*Anagallis arvensis***.** Atlas des plantes
de France, A. Masclef (1891)

Effects of different energy cropping systems on plant diversity in Central German agricultural landscapes

Dissertation

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Charlotte Seifert

aus

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Referent/In: Prof. Dr. Christoph Leuschner

Korreferent/In: Prof. Dr. Erwin Bergmeier

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Wheat field in Central Germany. Picture by Charlotte Seifert, August 2012.

Ein Weizenfeld

Weil es die Ähre verschmäht, sich mit der Farbe zu zieren,
Hat die Natur ihr den Mohn dicht an die Seite gestellt;
Jener hat sie die Kraft vertraut, den Menschen zu nähren,
Diesem verlieh sie den Reiz, welcher sein Auge erfreut.
Jene frage drum nicht: Wo sprießen dir nützliche Körner?
Oder diesen: Wo trägst du den erquicklichen Schmuck?
Wenn die eine uns fehlte, so könnten wir freilich nicht leben,
Aber wir möchten es nicht, wäre der and're nicht da!

Christian Friedrich Hebbel (1813-1863)
German poet and dramatist

Impressions from an extensively managed field margin near Göttingen in Lower Saxony in June 2012. Pictures by Caroline Focke.

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Chapter 3

Appendix 3.A Frequency of occurrence of species according to crop type and in-field position

Abbreviations and acronyms

Abb.	Abbreviation
AD	*Anno domini*
a.m.	*Ante meridiem*, before noon
a. s. l.	Above sea level
BEST	Research framework 'Bioenergie Regionen Stärken' ('Strengthening Bioenergy Regions')
BfN	Bundesamt für Naturschutz (German Federal Agency for Nature Conservation)
BKG	Bundesamt für Kartographie und Geodäsie (German Federal Agency for Cartography and Geodesy)
BLE	Bundesanstalt für Landwirtschaft und Ernährung (German Federal Office for Agriculture and Food)
BMBF	German Federal Ministry for Education and Research
BMEL	German Federal Ministry for Food and Agriculture (since 2014)
BMELV	German Federal Ministry for Food, Agriculture and Consumer Protection (before 2014)
BMU	German Federal Ministry for the Environment, Natur Conservation and Nuclear Safety
BS	Base saturation (%)
BtL	Biomass to liquid fuels (conversion technology for biomass)
BÜK50	Bodenübersichtskarte 1:50,000 (soil map, scale 1:50,000)
CCA	Canonical correspondence analysis
CEC	(Potential) cation exchange capacity
Chpt.	Chapter
C_{inorg}	Inorganic soil carbon
CO_2e	CO_2-equivalents (measure for greenhouse gas emissions)
C_{org}	Organic soil carbon
cp.	Compare
C_{tot}	Total soil carbon
DCA	Detrended correspondence analysis
DMK	Deutsches Maiskomittee e. V. (German Maize Committee, agricultural association)
DW	Dry weight
DWD	Deutscher Wetterdienst (Meteorological Service of Germany)
E	East
ECEC	Effective cation exchange capacity
EEA	European Environmental Agency
ed/eds	Editor(s)
e.g.	*Exempli gratia*, for example
et al.	*Et alii/et aliae*, and others
EU	European Union
EU-27	The 27 European Member States from January 2006 to June 2013
FI	Field interior
Fig.	Figure
FM	Field margin
FNR	Fachagentur Nachwachsende Rohstoffe e. V. (coordinating institution for research and development in the field of renewable resources in Germany)
GHG	Greenhouse gas
GK50	Geologische Karte 1:50,000 (geological map, scale 1:50,000)

GÜK500	Geologische Übersichtskarte 1:500,000 (geological map, scale 1:500,000)
HFA	Handbuch Forstliche Analytik (book on analysis methods in the forestry sector)
HI	Herbicide use intensity index
HRAC	Herbicide Resistance Action Committee (association of major herbicide producers)
HVO	Hydrogenated vegetable oils
ICP	Inductively coupled plasma analyses
i.e.	*Id est*
ILUC	Indirect land use change
I.M.A.	Information.medien.agrar e. V. (association for information, media and agriculture)
Inhib.	Inhibitor
LBEG	Landesamt für Bergbau, Energie und Geologie Niedersachsen (State Authority for Mining, Energy and Geology of Lower Saxony)
LPG	Landwirtschaftliche Produktionsgenossenschaft (agricultural production cooperatives in the former German Democratic Republic)
LS	Lower Saxon hills (study area)
LWA	Landwirtschaftsamt (German county office for agriculture in Thuringia)
LWK	Landwirtschaftskammer (German county office for agriculture in Lower Saxony)
N	North
NABU	Naturschutzbund Deutschland e. V. (nature conservation association)
n.d.	No data
No.	Number
NRW	North Rhine-Westphalia
ns	Not significant
N$_{tot}$	Total soil nitrogen
p./pp.	Page(s)
PAR	Photosynthetically active radiation
PAR$_{inc}$	Incident PAR flux density (μmol m^{-2} s^{-1})
PAR$_{trans}$	Transmitted PAR (μmol m^{-2} s^{-1})
PC	Principal component
PCA	Principal component analysis
PCNM	Principal coordinates of neighbour matrices
p.m.	*Post meridiem*, after noon
Prop.	Proportion
PS II	Photosystem II
pyr	Year preceding the field surveys
R²	Coefficient of determination (regression), redundancy statistic (RDA, CCA)
R²$_{adjusted}$	Adjusted R² (see chapter 1.3.4.1)
RDA	Redundancy analysis
SE	Standard error of the mean
SOM	Soil organic matter
SRC	Short rotation coppice
SRU	Sachverständigenrat für Umweltfragen (German Advisory Council on the Environment)
syr	Year of field surveys
TB	Thuringian Basin (study area)
TLL	Thüringer Landesanstalt für Landwirtschaft (Thuringian State Office for Agriculture)

TPAR	PAR transmissivity (= PAR_{trans}/PAR_{inc}, in %)
UBA	Umweltbundesamt (German Federal Environmental Agency)
UTM	Universal Transverse Mercator (projected coordinate system)
VCA	Variance component analysis
VG	Verwaltungsgemeinschaft (municipality)
WGS	World Geodetic System (geodetic reference system)

Chemical elements

Abb.	**Name**		
Al	Aluminium	**Mg**	Magnesium
Ba	Barium	**Mn**	Manganese
C	Carbon	**N**	Nitrogen
Ca	Calcium	**Na**	Sodium
Cl	Chlorine	**O**	Oxygen
Fe	Iron	**P**	Phosphorus
H	Hydrogen	**S**	Sulphur
K	Potassium		

Units

Abb.	**Name**	**Description**	**Expression in terms of SI base units**
°	Degree	Angle, geographical coordinate	$1° = 0.0175$ rad $= 0.0175$ m m^{-1}
'	Minute	Angle, geographical coordinate	$1' = 1/60° = 0.0003$ rad (or m m^{-1})
°C	Degree Celsius	Temperature	0 °C $= 273.15$ K
g	Gram	Mass	–
h	Hour	Time	1 h $= 3600$ s
ha	Hectare	Area	1 ha $= 10,000$ m²
J	Joule	Energy, work, amount of heat	1 J $= 1$ m² kg s^{-2}
L	Liter	Volume	1 L $= 10$ cm³
m	Meter	Length	–
min	Minute	Time	1 min $= 60$ s
mol	Mole	Amount of substance	–
t	Ton	Weight	1 t $= 1000$ kg
yr, yrs	Year(s)	Time	1 yr $= 31,536,000$ s

Unit prefixes

Abb.	Name	Value	Abb.	Name	Value
n	nano	10^{-9}	**k**	kilo	10^{3}
µ	micro	10^{-6}	**M**	mega	10^{6}
m	milli	10^{-3}	**G**	giga	10^{9}
c	centi	10^{-2}	**T**	tera	10^{12}
d	deci	10^{-1}	**P**	peta	10^{15}
			E	exa	10^{18}

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Summary

The rich biodiversity of the Central European farmland is to a large extent directly dependent on human management decisions. Diverse and dynamic biocoenoses have developed in mutual interactions with farming practices, and have been an integral part of agricultural land ever since the Neolithic. In the middle of the 20th century, when synthetic biocides, mineral fertilisers and more powerful machinery became widely available, an unprecedented, steady and rapid erosion of farmland biodiversity commenced. Despite political declarations of intentions to slow biodiversity loss, pressure on farmland biodiversity has been further increasing in recent years. In the face of climate change and to reduce the dependency on limited fossil fuels, the subsidy-driven bioenergy boom is increasing the demand for arable land to cultivate the required feedstocks.

Only few scientific publications have yet addressed the consequences of the rapid expansion of energy cropping on farmland plant diversity. This thesis aimed to contribute to filling this gap, by recording the habitat conditions (field management, light regime, soil chemical properties) and the plant diversity in different energy cropping systems (maize for anaerobic digestion, oilseed rape for biodiesel and poplar/willow short rotation coppices (SRC) for wood chip combustion). Subsequently, the energy cropping systems were compared to conventionally managed food/fodder crops (winter-sown wheat and barley) and to winter cereal fields managed extensively according to an agri-environmental scheme (AES).

The results show clearly that the underlying root causes of the strong decline in arable plant diversity extent far beyond energy cropping. We found the arable plant assemblages in the field interior to be extremely impoverished in energy maize and oilseed rape fields as well as in conventionally managed winter-sown wheat or barley (on average 3–6 species 100 m²). The mean nitrogen fertilisation rate (195 kg N ha^{-1} yr^{-1}) and herbicide use intensity (HI = 1.8) were shown to be similarly high for maize and winter wheat. Excessive phosphorus fertilisation (on average 96 kg P_2O_5 ha^{-1} yr^{-1}) was, however, discerned as a threat to plant diversity which mainly applies to maize production. Maize was also found to be more shading (11% PAR transmissivity vs. 19% in winter cereals) and consequently less species rich at the field margins than winter cereals (11 and 15 species 100 m^{-2} respectively). Across study regions, the arable plant communities of conventionally managed fields were typically dominated by the same set of only 5–10 common weedy species. We found (summer-sown) maize stands to offer habitats to a slightly different set of arable plant species than fields cultivated with winter-sown crops. A balanced mixture of maize, oilseed rape and winter cereals at the landscape scale consequently offers habitats to a wider range of arable plant species than any of these crops alone. Habitat conditions on extensively managed fields cultivated according to an AES were, however, found to be much more heterogeneous and suitable for a far greater range of species, including rare and threatened taxa (on average 21 species in the interior and 33 species 100 m^{-2} at the field margins).

Our models also demonstrate that moderate reductions in herbicide use intensity or fertilisation while staying within the range of currently practiced conventional farming techniques can hardly be expected to reliably halt the decline in arable plant diversity. Novel, regionally adapted approaches and extensively managed arable habitats are urgently needed.

The plant diversity of SRC was found to decline strongly with plantation age. Young, low-input SRCs showed a community composition similar to early successional fallow land. Contrarily, 5–8 year old densely planted SRCs were found to be dominated by a set of few generalist, nitrophilous species. They were more shading (1–4% PAR transmissivity) and only slightly more species rich (8–19 species 75 m^{-2}) than annual energy crops, despite not being fertilised or treated with herbicides for several years. The planting of 5–20 m wide SRC strips dividing larger fields with annual crops, can nevertheless be recommended in structurally impoverished, intensively managed agricultural landscapes, particularly if measures to increase the variability of habitat conditions in the coppices are applied.

To address the root causes of plant diversity loss on farmland it seems, however, additionally paramount to create a permanent network of extensively managed field sanctuaries to maintain viable source populations which can potentially disperse to the wider agricultural landscapes in the future. To effectively inform policy makers, future research on energy cropping and farmland plant diversity should broaden the focus from the field to the landscape scale and address remaining open questions with regard to the interactions of arable plant diversity with other taxonomic groups.

Zusammenfassung

Der große Artenreichtum der mitteleuropäischen Kulturlandschaften ist zu weiten Teilen unmittelbar von bestimmten extensiven Formen der Landbewirtschaftung abhängig. Vielfältige, artenreiche Biozönosen waren und sind ein wichtiger Bestandteil landwirtschaftlicher Nutzflächen. Sie haben sich seit dem Neolithikum im Wechselspiel mit den landwirtschaftlichen Wirtschaftsweisen kontinuierlich weiterentwickelt. Erst als synthetische Pflanzenschutzmittel, Mineraldünger und leistungsstarke Landmaschinen in der zweiten Hälfte des 20. Jahrhunderts weite Verbreitung erfuhren, setzte ein bis dato beispielloser, stetiger und schneller Rückgang der Artenvielfalt von Äckern, Wiesen und Weiden ein. Trotz politischer Absichtserklärungen den raschen Artenschwund zu stoppen, hat sich der Druck auf die verbleibenden Populationen auch in den letzten Jahren weiter erhöht. Der Klimawandel und der Wunsch die Abhängigkeit von endlichen fossilen Energieträgern zu reduzieren, haben dazu geführt, dass die Nachfrage nach Ackerland zum Anbau von Energiepflanzen, bedingt durch einschlägige Subventionen, stark gestiegen ist.

Nur wenige wissenschaftliche Publikationen haben sich bisher mit den Folgen der schnellen Ausweitung des Energiepflanzenanbaus Ackerflora beschäftigt. Diese Dissertation soll dazu beitragen, diese Lücke durch Untersuchungen der Habitatbedingungen (Bewirtschaftungsmethoden, Lichtklima im Bestand, Bodenchemie) und der Phytodiversität in verschiedenen Energiepflanzenbeständen (Silomais zur Biogaserzeugung, Winterraps zur Biodieselherstellung, Kurzumtriebsplantagen (KUP) aus Pappeln oder Weiden zur Hackschnitzelverbrennung) zu schließen. Die Energiepflanzenbestände wurden zudem mit konventionellem Wintergetreide zur Nahrungs- und Futtermittelproduktion (Winterweizen und -gerste) und mit extensiven Weizenbeständen aus Agrarumweltmaßnahmen (AUM) verglichen.

Die Untersuchungsergebnisse weisen eindeutig darauf hin, dass die eigentlichen Ursachen des starken Phytodiversitätsverlustes auf Äckern weiter reichen und grundlegender sind als der Anbau von Energiepflanzen. Es wurde gezeigt, dass die Phytozönosen im Inneren von Bioenergiemais- und Rapsäckern sowie auch in Wintergetreidebeständen extrem verarmt sind (im Mittel 3–6 Arten 100 m^{-2}). Die durchschnittliche Stickstoffdüngung (195 kg N ha^{-1} a^{-1}) und der Herbizidbehandlungsindex (HI = 1.8) waren sowohl in Mais- als auch in Winterweizenbeständen ähnlich hoch. Eine Bedrohung für die Pflanzenartenvielfalt durch übermäßige Phosphatdüngung (im Mittel 96 kg P_2O_5 ha^{-1} a^{-1}) wurde jedoch vor allem beim Maisanbau festgestellt. Zudem war die Beschattung in Maisbeständen erhöht (11 % PAR Transmissivität im Vergleich zu 19 % in Wintergetreide), was mit einem Rückgang der mittleren Artenzahlen am Feldrand einherging (11 Arten 100 m^{-2} im Mais, 15 Arten 100 m^{-2} in Wintergetreide). Die Pflanzengesellschaften konventioneller Äcker wurden regionsübergreifend von nur 5–10 häufigen Problemunkrautarten dominiert. Maisbestände wiesen als Sommerkulturen eine von Winterkulturen leicht abweichende Artenzusammensetzung auf. Eine ausgeglichene Mischung von

konventionellem Mais, Winterraps und Wintergetreide in der Agrarlandschaft bietet mehr Arten einen geeigneten Lebensraum als jede dieser Kulturen für sich genommen. Insgesamt waren die Habitatbedingungen auf den im Rahmen von AUM extensiv bewirtschafteten Äckern jedoch wesentlich heterogener und boten einer weitaus größeren Zahl von Pflanzenarten (darunter auch seltene und bedrohte Taxa) geeignete Lebensbedingungen. Im Mittel wurden 21 Arten 100 m^{-2} im Inneren und 33 Arten am Feldrand gezählt. Auch die Regressionsmodelle weisen darauf hin, dass moderate Beschränkungen der Herbizidbehandlungsintensität oder der Düngung, die sich im Rahmen der zur Zeit üblichen konventionellen Bewirtschaftspraktiken bewegen, kaum dazu beitragen den Rückgang der Ackerflora zu stoppen. Neue, an die jeweiligen regionalen Gegebenheiten angepasste Konzepte und extensive Ackerhabitate sind hierzu unerlässlich.

Die Vegetationsaufnahmen zeigen zudem, dass die Phytodiversität von KUP mit dem Alter der Plantagen stark zurückgeht. Junge KUP, ohne Düngung und mit nur geringem Herbizideinsatz, wiesen eine mit frühen Sukzessionsstadien von Ackerbrachen vergleichbare Artenzusammensetzung auf. Im Gegensatz hierzu wurden die 5–8 jährigen, dichtgepflanzten Plantagen von wenigen stickstoffliebenden Habitatgeneralisten dominiert. Obwohl sie seit mehreren Jahren weder gedüngt noch mit Herbiziden behandelt wurden, waren diese Bestände zudem dunkler (1–4 % PAR Transmissivität) und nur wenig artenreicher (8–19 Arten 75 m^{-2}) als einjährige Energiepflanzenkulturen. Die Pflanzung von 5–20 m breiten Energieholzstreifen zur Zerteilung großer Schläge in strukturarmen, landwirtschaftlichen Intensivregionen kann jedoch aus naturschutzfachlicher Sicht empfohlen werden, insbesondere wenn Maßnahmen nur Erhöhung der Habitatvielfalt in den Plantagen ergriffen werden.

Um die grundlegenden Ursachen des Pflanzenartenschwundes in Agrarlandschaften zu beheben, erscheint es jedoch darüber hinaus dringend notwendig, ein langfristig angelegtes Netzwerk extensiver Feldflorareservate zu begründen, um dauerhaft überlebensfähige Ackerwildkrautpopulationen zu erhalten und eine Ausbreitung dieser Arten in die weitere Agrarlandschaft in Zukunft wieder zu ermöglichen. Um politische Entscheidungsfindungen zu unterstützen, erscheint es notwendig im Rahmen von zukünftigen Forschungsprojekten den Blickwinkel vom Feld auf die Landschaftsebene zu erweitern und verbleibende offene Fragen über die Wirkungen der Ackerflora auf andere taxonomischen Gruppen zu klären.

Chapter 1

Introduction

1.1 Research context

1.1.1 Plant diversity in agricultural landscapes

1.1.1.1 Arable plant assemblages as a mirror of agricultural land use practices

The European landscape of today is the result of the long-term regional climate history and millennia of human interference (Lang et al., 2003; Behre, 2008). Without human influence the largest part of Europe, in particular of Central Europe, would today be covered by forests (Bohn et al., 2000/2003; Ellenberg & Leuschner, 2010). Since human settlements became common in the Neolithic, forests were increasingly used as wood pastures, cleared for crop production and logged to meet the growing demand for fire wood and construction material (Lang et al., 2003). The intensity of this process showed strong spatial and temporal variations and fluctuations (Lang et al., 2003; Behre, 2008). It resulted in dynamic mosaic landscapes with a mixture of non-forested and forested habitats and led to the development of the species-rich biocoenoses of open habitats which we know today.

The farmland flora, i.e. the flora of (fallow) arable land, pastures and meadows, evolved alongside the development of new farming techniques since the Neolithic (Willerding, 1986; Pott, 1992; Hofmeister & Garve, 2006; Ellenberg & Leuschner, 2010). During the Neolithic, crop production was commonly interrupted by long-term fallow periods. Consequently, the respective plant communities were found to be dominated by perennial species (Willerding, 1986; Pott, 1992). The differentiation between arable plant communities, dominated by annual plant species and grassland communities dominated by perennials became increasingly clear since medieval times, when fallow periods became shorter and less common (Willerding, 1986). A more detailed account of the main agricultural developments in the study areas and the impacts on arable plant diversity is given in section 1.2.2.

Genotypes allowing for a high degree of phenotypic plasticity are typically most successful in arable habitats (Imam & Allard, 1965; Hermanutz & Weaver, 1996). Other favourable properties include the production of large quantities of seeds, the ability to easily disperse over large distances, a resistant

seed coat, heterocarpy, the rapid completion of the life cycle, the long-term maintenance of germinability and/or the presence of underground storage organs which survive regular anthropogenic soil disturbances (Hofmeister & Garve, 2006).

Many of the farmland species (the so called archaeophytes) were introduced from the steppic habitats of the Middle East or the Mediterranean to the Central European agricultural habitats (Fig. 1.1; Ellenberg & Leuschner, 2010). Another group of species originates from naturally open habitats such as open shores and natural forest clearings and is considered native to Central Europe (Schneider et al., 1994). Some arable plant species such as common corncockle (*Agrostemma githago*), whiskered brome (*Bromus grossus*) and flax dodder (*Cuscuta epilinum*) are today completely dependent on the arable habitats in which they have evolved. There are no or hardly any known populations of these species in natural habitats which could provide the necessary conditions to serve as sanctuaries for their conservation. Historically, seeds of *A. githago* were harvested and re-sown together with the grain. Since seed cleaning techniques reliably discard corncockle seeds from the harvested grain, the species suffered strong declines and is nowadays threatened with extinction (Ludwig & Schnittler, 1996). *B. grossus* is also critically endangered (Ludwig & Schnittler, 1996) and listed in Annex II of the EU habitats directive, due to the loss of extensively managed spelt fields. As a consequence of the cessation of flax cultivation, *C. epilinum* is nowadays extinct in Germany (Ludwig & Schnittler, 1996). As shown above, the arable flora often directly reflects field management practices and can therefore be regarded as a mirror of the human land use history and as an important and integral part of the European nature and culture.
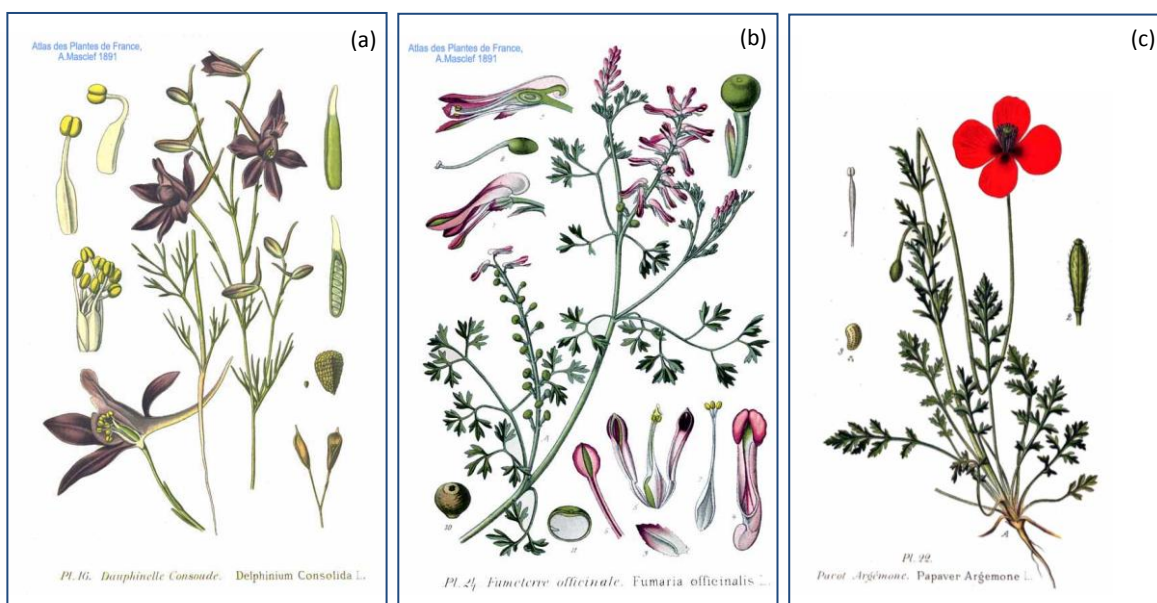
Fig. 1.1 Examples of arable archaeophytes (a) field larkspur *Consolida regalis*, (b) common fumitory *Fumaria officinalis* and (c) pale poppy *Papaver argemone*. All drawings are from the *Atlas des Plantes de France* by Amédée Masclef (1891).

1.1.1.2   The role of arable plants in agroecosystems

Besides their role as crop competitors, arable plant species fulfil a diverse set of ecological functions in agro-ecosystems. The concept of arable plants as being merely detrimental is therefore gradually being replaced by a more integrated view which values the ecological benefits arising from diverse arable plant assemblages.

Being the basis of the farmland food web, arable plants provide important resources for herbivores and decomposers (Parish et al., 2009). Concomitantly, this affects the predators of higher trophic levels. Today far less than 1% of the total summer biomass of high-input arable fields consists of non-crop arable plants (Parish et al., 2009). Research shows that the rapid and ongoing decline in farmland bird species is partly due to the lack of feeding resources (i.e. seeds and arthropods) in impoverished, high-input cropping systems (Marshall et al., 2003; Taylor et al., 2006; Evans et al., 2011). It has also been shown that more diverse arable plant communities are less suitable for mass reproductions of herbivorous crop pests (Altieri, 1999). This has been linked to the increased presence of antagonistic pest predators which find better habitat conditions in more diverse fields (Altieri, 1999; Jordan & Vatovec, 2004; Evans et al., 2011). Diverse arable plant communities, furthermore, contribute to the maintenance of favourable levels of plant-pollinator interactions in agricultural landscapes (Gabriel & Tscharntke, 2007).

Jordan & Vatovec (2004) highlight that the arable flora can contribute to reducing soil erosion, slowing nutrient loss, increasing soil organic matter, improving nitrogen levels and conserving soil water. Additionally, positive effects on decomposer communities have been observed as the colonization of bare soil by arable plants typically leads to an increase in soil microbial biomass and soil arthropods (Wardle, 1992, 1995).

These examples show that arable non-crop plants, in addition to serving as a source of joy and beauty to passers-by and as allegory and inspiration to poets and painters, are a vital key element of agricultural ecosystems. Maintaining tolerable levels of arable plants in agricultural landscapes would consequently not only contribute to the protection of a highly specialised and highly threatened group of species, but it would positively affect many other taxa and ecosystem processes.

1.1.1.3   Current conservation status

The current conservation status of arable plants is distressing particularly in northern and western Europe, due the negative impacts of the intensification of land use (i.e. increased fertilisation and herbicide use, narrow crop rotations, very effective seed cleaning techniques and increasingly homogeneous agricultural landscapes; Sutcliffe & Kay, 2000; Fried et al., 2009; Storkey et al., 2012). The species richness and abundance of arable non-crop species has been severely reduced since the

1950s in many western European countries (Sutcliffe & Kay, 2000; Fried et al., 2009; Kohlbrecher et al., 2012; Meyer et al., 2013). A study from Central Germany showed reductions of the regional species pool of 23%, translating into the loss of 68 out of 301 recorded arable plant species (Meyer et al., 2013). In Germany, there are approximately 300 specialised arable plant species of which > 55% are categorised as threatened in at least one federal state (Hofmeister & Garve, 2006). 71 taxa (> 20%) are even threatened at the national level (Fig. 1.2; Ludwig & Schnittler, 1996; Hofmeister & Garve, 2006). Field margins are often the last refuges for the remaining arable plants (Marshall & Moonen, 2002; Fried et al., 2009). In the field interior, the arable flora is commonly reduced to a set of 5−10 common, weedy plant species which can resist herbicide applications and are able to compete with the crop on highly fertilised soils (Meyer et al., 2013). As the field interior of conventionally managed fields is no longer suitable as habitat for most arable plant species, and increases in field size dwarfed the amount of land allotted to the field margins, the available habitat for arable plant species has been reduced by 95% in Northern Germany since the 1950s (Hötker & Leuschner, 2014).
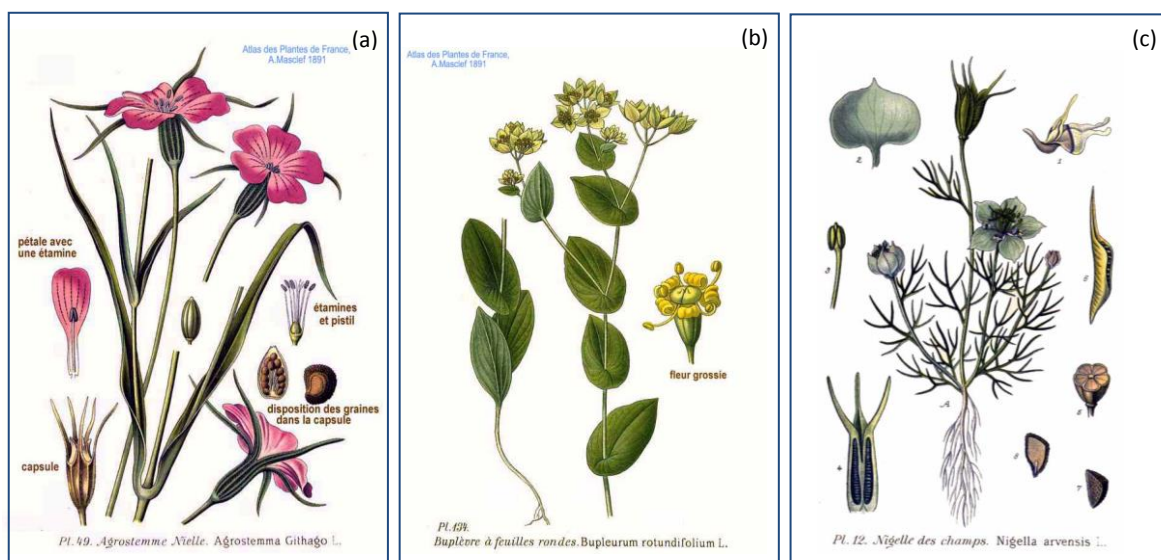
Fig. 1.2 Examples of nationally endangered arable plant species (a) common corncockle *Agrostemma githago* (critically endangered, category 1) (b) hare's ear *Bupleurum rotundifolium* (critically endangered, category 1) and (c) black bread weed *Nigella arvensis* (endangered, category 2) (Ludwig & Schnittler, 1996). All drawings are from the *Atlas des Plantes de France* by Amédée Masclef (1891).

As shown above, arable plant communities, which are the result of several thousand years of evolution alongside humans, are severely endangered. Concepts for halting the erosion of the biodiversity of agricultural ecosystems are currently not receiving the necessary political and financial support to be effective (Hötker & Leuschner, 2014). Changes in agricultural practices which potentially further increase the pressure on arable plant communities can make the recovery of a diverse and dynamic arable flora even more difficult in the future. It is therefore of particular importance to fully assess the impacts of any large scale changes in agricultural land use practices on arable plant diversity.

1.1.2 Energy cropping

1.1.2.1   Contribution to the total energy supply

Global importance of bioenergy

Currently, approximately 10% (= 52 EJyr[-1]) of the world's primary energy supply is covered by biomass (Fig. 1.3; Vakkilainen et al., 2013). Biomass can be used to produce thermal energy, electricity and transport fuels. With a share of approximately 90% of the global bioenergy production, woody biomass is the most important bioenergy feedstock (Nemestóthy, 2012). Although the global biomass trade is growing rapidly, it only made up about 5 % of the total use of biomass for energy in 2011 (excluding intra EU trade; Vakkilainen et al., 2013). The direct trade represented 0.5 EJyr[-1] in 2011 (including wood pellets, biodiesel, palm oil, ethanol, fuel wood and charcoal) and the indirect trade, i.e. energy production from residues of wood which had primarily been imported to be used as raw material, represented 0.6 EJyr[-1] (Vakkilainen et al., 2013). The domestic and the intra EU biomass production is therefore of great importance with regard to the environmental impacts of the sector in Europe.
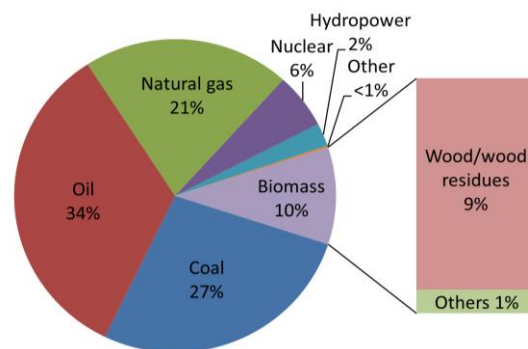
Fig. 1.3 Contribution of bioenergy to the global primary energy consumption of approximately 510 EJyr[-1]. Based on data from Vakkilainen et al. (2013) and Nemestóthy (2012).

Bioenergy in the European Union

In the European Union (EU-27) 4.3 EJyr[-1], equalling approximately 6% of the total primary energy consumption, were produced from biomass in 2010 (EEA, 2012, 2013b). The demand for bioenergy is estimated to more than double to 10.0 EJyr[-1] by 2020 to comply with the targets of the national renewable energy action plans of the EU member states (Bentsen & Felby, 2012). Currently, approximately 76% of the bioenergy is produced from woody material (wood and wood residues), whereas 24% is based on energy cropping in an agricultural context and the processing of animal manures (Fig. 1.4; EEA, 2013b). In 2008, most energy crops grown on EU farmland were annual food and fodder crops such as oilseed rape (59%), sun flower (20%), cereals (11%) and maize (7%) (EEA, 2013a). Germany and France were the main producers of energy crops with 1.53 and 1.35 million ha

respectively, followed by Poland and Romania with 0.76 and 0.57 million ha (Panoutsou et al., 2011). Since 2008 the sector has developed rapidly, but more recent figures at a European scale are surprisingly not available (Don et al., 2012). The recent literature shows, however, that the general picture has changed only little. Annual energy crops are still predominant and perennial energy crops, such as short rotation coppices, are not yet grown at large scales across the EU (Don et al., 2012).
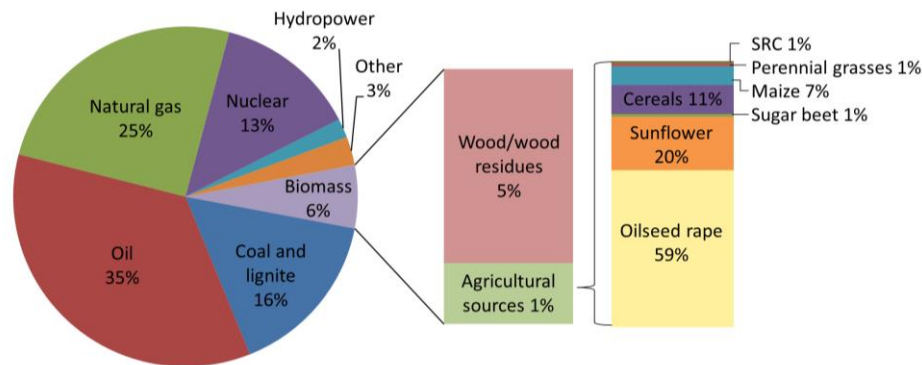
Fig. 1.4 Contribution of bioenergy to the primary energy consumption of the EU (EU-27) in 2010 (in total approximately 72 EJyr^{-1}) and contribution of different crops to the renewable energy mix (based on 2006–2008 data). No data on the contribution of animal manures to energy production from agricultural sources in the EU were available. Animal manures are therefore no represented in this figure, although they are subsumed under "agricultural sources". SRC = Short rotation coppice. Adapted from EEA (2013a & 2013b).

The case of Germany – 'Energiewende'

Germany, currently the largest producer of energy crops in the EU (Don et al., 2012), is pursuing an ambitious agenda for the transition from fossil to renewable energies, the so called 'Energiewende' (BMU & BMELV, 2010). In 2012, 8.2% (equalling 734 PJ) of the final energy consumption was covered by biomass (Fig. 1.5; FNR, 2013a). Biomass for heating (mostly solid fuels) represented approximately 59%, biomass for power generation 24% and for transport fuels 17% of this fraction (FNR, 2014b). This means that 9.2% of the heating, 7.5% of the power generation, and 5.7% of the transport fuels in Germany were produced from biomass (FNR, 2013e, 2014a, 2014d). Projections for 2020 expect an increase of these fractions to 9.7% for heat, 8.0% for power and 12.0% for transport fuels (BMU & BMELV, 2010). With the exception of transport fuels these targets are likely to be reached well before 2020.

Solid fuels (mostly wood and wood residues) are the most important biomass resource and the source of 81% of the thermal energy and 31% of the power generation from biomass in Germany (Fig. 1.5;

(FNR, 2013k, 2014d). The contribution of solid biofuels from agricultural sources, such as short rotation coppices or crop residues, to the national energy supply remains low (see section 1.1.2.2.).
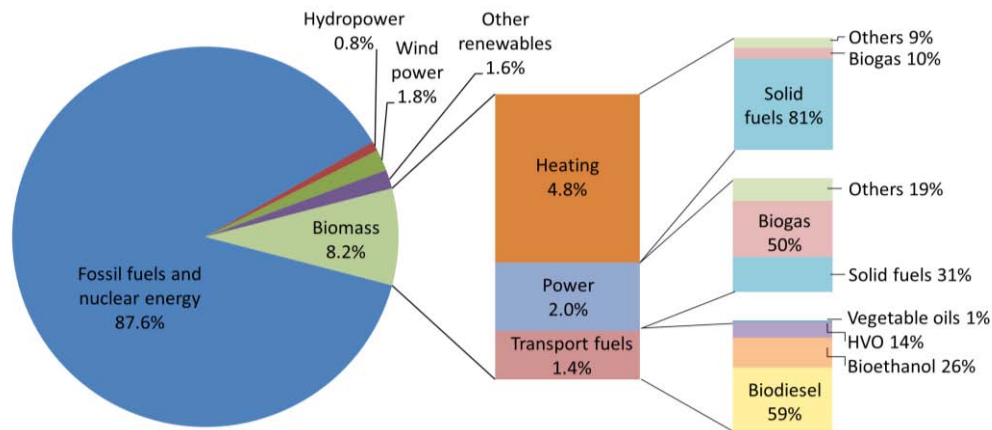
Fig. 1.5 Contribution of bioenergy to the final energy consumption in Germany in 2012 (approximately 9.0 EJyr^{-1}) according to the type of energy and the type of fuel. HVO = Hydrogenated vegetable oils. Adapted from FNR (2012, 2013a, 2013k, 2013l, 2014b, 2014d).

Unlike other EU countries, Germany strongly supports the establishment of farm-based plants for anaerobic digestion (also called 'biogas plants') through national subsidies (BMU & BMELV, 2010) and consequently has the largest biogas sector (Don et al., 2012). Biogas plants produce 50% of the power and approximately 10% of the heat generated from biomass in Germany (Fig. 1.5, (FNR, 2013k, 2013l, 2014d). These fractions sum up to 1.3% of the final energy consumption or approximately 3.8% of the total power generation and 1% of the total heat production (FNR, 2013a, 2013k, 2014a). The contribution of purified biogas (bio methane) to the consumption of transport fuels in Germany is with approximately 0.06% only marginal (FNR, 2012, 2013e).

Germany is also the largest producer of biodiesel and the second largest producer of bioethanol in the EU (Don et al., 2012). Biodiesel accounted for 3.4% of the national transport fuel consumption in 2012 and bioethanol covered 1.5% (FNR, 2013e).

1.1.2.2   *Conversion technologies and biomass feedstocks*

Solid biofuels

Solid biofuels can be used to produce heat, power and even transport fuels either through combustion (directly or by thermo-chemical conversion to charcoal) or via gasification (Kappas, 2013; FNR, 2014c). Detailed descriptions of the different combustion and gasification technologies are

available from Luque et al. (2011) and a good summary on the subject can be found in FNR (2014c). Wood and wood residues from forestry have by far the greatest share of all solid biofuels in Germany and worldwide (FNR, 2014c). As forest resources are limited (Verkerk et al., 2011), the importance of solid biofuels produced on farmland is starting to increase. All crops and crop residues can theoretically be used as solid biofuels after the appropriate pre-processing (i.e. sizing, drying and/or densification of the material). Examples include short rotation coppices (SRC; i.e. fast growing trees such as poplar or willow grown on farmland), cereals (the whole plant, the straw or the grain), miscanthus (*Miscanthus × giganteus*) and biomass from permanent grasslands and leys. Solid biofuels were grown on only 11,000 ha farmland in 2013 in Germany, i.e. approximately 0.07% of the agriculturally used area (Fig. 1.6; FNR, 2013c). Estimates suggest that short rotation coppices currently cover between 6,000 –10,000 ha (Hagemann et al., 2013). Miscanthus was grown on approximately 3,000 ha in 2011 (Pude, 2012). Both crops are estimated to yield about $50 – 300$ GJha^{-1}yr^{-1} depending on the conversion technology, the site conditions and the field management (SRU, 2007; FNR, 2014c). Due to these high energy yield potentials, which can compete with anaerobic digestion of maize silage ($50 – 200$ GJha^{-1}yr^{-1}; SRU, 2007; Grassini & Cassman, 2012), SRC and miscanthus are potentially economically and ecologically interesting options for the bioenergy sector. This is one reason why the area cultivated with SRC and miscanthus is expected to expand substantially during the next decade. Aust (2012) found a realizable potential of approximately 0.7 million ha (or 145 PJyr^{-1}) for the cultivation of SRC in Germany. The FNR (2014c) estimated a potential for energy production from solid biofuels grown on arable land of 180 PJ for the year 2007 and an increase up to > 700 PJyr^{-1} by 2020. Surprisingly, no data on the currently realised energy production from solid biofuels from agricultural systems is available, but with regard to the small area cultivated with SRC and miscanthus it is bound to be much less than the estimated potentials (in the range of $1 – 3$ PJyr^{-1}) (SRU, 2007; FNR, 2014c).

Biomass for anaerobic digestion

Biogas is produced in a bio-chemical conversion process through anaerobic digestion (Kappas, 2013). Biogas plants can use a variety of feedstocks, as long as these have a high content of crude proteins and crude fats, and a low content of dry matter and lignocellulosic compounds (Amon et al., 2007a; Amon et al., 2007b). Methane (CH_4) is the main combustible compound of biogas making up between 50–75% of the gas mixture (FNR, 2013b). A comprehensive description of the technical installations, the bio-chemical process and the suitability of different substrates for anaerobic digestion can be found in FNR (2013b) and Luque et al. (2011). German biogas plants are currently run on animal manures (41%) and silage from energy crops such as maize (40%), grass (6%) and cereals (4%) (FNR, 2013g, 2013h). Maize is the preferred energy crop as it produces the highest biomass and methane yields per hectare and can be cultivated with readily available machinery (Amon et al.,

2007a; Boehmel et al., 2008). Maize for biogas production is currently grown on 0.8 million ha (about 7% of the arable land) and represents 32% of the total maize production in Germany (FNR, 2013f; Destatis, 2014). The area cultivated with energy maize has thus more than tripled since 2007 (FNR, 2013d) and is expected to keep expanding rapidly during the next decade (BMU & BMELV, 2010). The area cultivated with alternative biogas feedstocks, e.g. silage from cereals, sugar beet, sorghum, sunflower, cup plant, clover leys and grassland biomass as well as wild flower mixtures is with 0.35 million ha much smaller (FNR, 2013c, 2013f). No current data on the exact area of each of these alternative feedstocks are available for Germany.
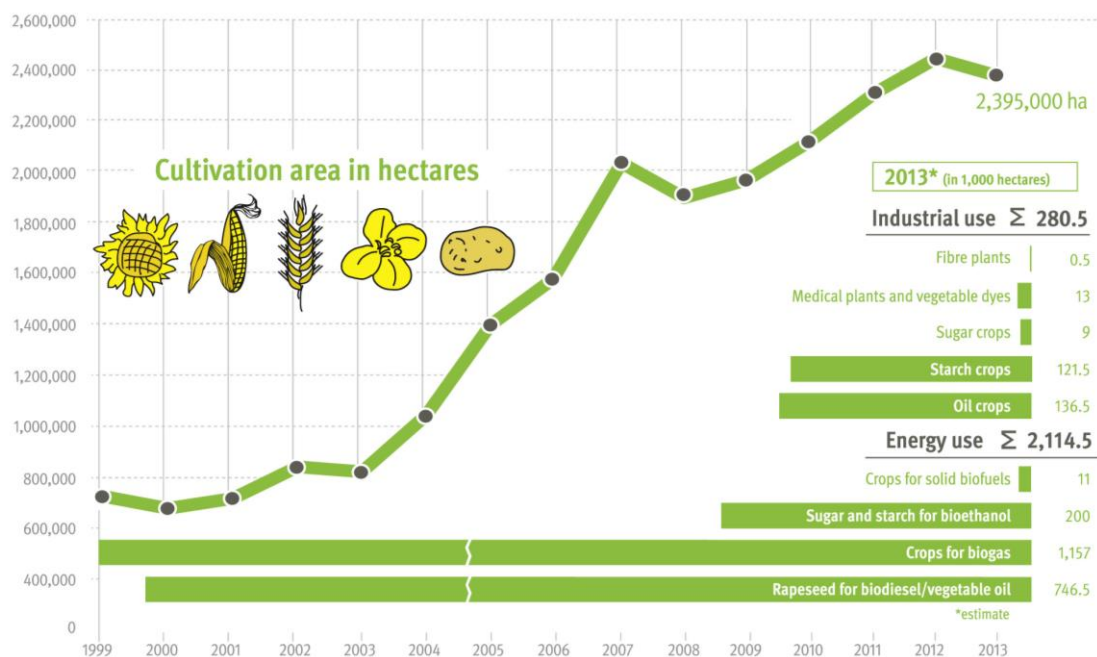
Fig. 1.6 Area used for the cultivation of renewable resources in Germany (1999–2013). Adapted from FNR (2013c).

Biomass for liquid transport fuels

Technically, there is a range of liquid biofuels for transport available, including among others biodiesel, vegetable oils, bioethanol, hydrogenated vegetable oils (HVO), biobutanol, and biomass to liquid fuels (BtL) (Schmitz et al., 2009). Only the first four of these options are currently available on the German market with biodiesel accounting for approximately 59%, bioethanol for 26%, HVO for 14% and vegetable oil for 1% of the German biofuel consumption (Fig. 1.5; FNR, 2012). The substrate requirements differ between the different kinds of transport biofuels. For a detailed description of the conversion processes refer to Luque et al. (2011).

Oilseed rape is, with a share of 85%, the most important resource for biodiesel production in Germany, followed by waste fats (10%), palm oil (2%) and soy (3%) (FNR, 2013j). Oilseed rape for

biodiesel and (hydrogenated) vegetable oil production was grown on 0.79 million ha equalling approximately 7% of the arable land in Germany in 2012 (FNR, 2013c; Destatis, 2014). This means that about 60% of the total rapeseed acreage in Germany was used for biofuel production (BLE, 2013). The total area grown with oilseed rape increased between 2000 and 2012 from 1.08 million ha to 1.30 million ha (BLE, 2013) due to high commodity prices. This overall increase was steady but with minor fluctuations until 2007 (BLE, 2013). Since 2007 no further increase of the rapeseed acreage was observed (BLE, 2013). As opposed to maize, oilseed rape reacts to continuous cropping in monoculture with strong yield declines of up to 25% (Hilton et al., 2013). To avoid yield declines oilseed rape can only be grown every third to fourth year at the same site, which generally restrains the expansion (Hilton et al., 2013). Alternative oil crops such as sunflowers or flax are not used for biodiesel production to any significant extent in Germany.

Plants for bioethanol production, such as sugar beet or cereals, were grown on 0.2 million ha in 2012 and 2013 (FNR, 2013c).

1.1.2.3 Environmental impacts of energy cropping

As shown in the previous section the production of biomass as a renewable energy source affects large proportions of land. It can lead to substantial changes in the kind and intensity of land use and land management. The environmental impacts of energy cropping, i.e. the impacts on biodiversity, greenhouse gas emissions, nutrient and water cycles, differ between energy cropping systems. When assessing the impacts of energy cropping the baseline, i.e. the former land use against which the energy crop is compared, is of fundamental importance (St. Clair et al., 2008; Immerzeel et al., 2014). Impacts can furthermore either be direct, i.e. caused at the site by the cultivation of the specific energy crop in comparison to the previous land use or indirect, i.e. caused by changes in land use elsewhere which were induced by the displacement of the kind of land use which is being replaced by energy cropping (Fargione et al., 2008; Lapola et al., 2010; Immerzeel et al., 2014). This work mainly deals with the direct impacts of energy cropping. As the indirect impacts are nevertheless of great importance to fully understand the consequences of energy cropping, they are also shortly discussed afterwards.

Direct impacts on biodiversity

A large body of research shows that the biodiversity of agricultural landscapes generally benefits from a heterogeneous landscape structure creating a mosaic of different habitats (Benton et al., 2003; Tscharntke et al., 2005). Another essential requirement is the availability of sufficient land with conditions suitable for reproduction and survival (Benton et al., 2003; Tscharntke et al., 2005).

Many farmland species are dependent on the presence of low-input grasslands, croplands and fallow lands (Wesche et al., 2012; Meyer et al., 2013).

The increasing demand for land to produce feedstocks for bioenergy production is affecting farmers' land use decisions, which in turn can affect the biodiversity of agricultural landscapes (Dauber et al., 2010; Immerzeel et al., 2014). The effects of bioenergy feedstocks on biodiversity differ widely depending on the type and intensity of land management required for its production and the type of land use which is being replaced (Immerzeel et al., 2014). In comparison with annual energy crops and/or other conventionally managed, high-input cropping systems, perennial 'second-generation' energy crops, such as SRC or miscanthus, are typically expected to have positive effects on biodiversity (Dauber et al., 2010; Immerzeel et al., 2014). When compared to grasslands, fallow land, low-input cropland or forests the effects are less clear and the impacts on biodiversity will often be neutral or negative (Immerzeel et al., 2014; Meyer et al., 2014).

As the pressure on agricultural biodiversity is already extremely high (cp. section 1.1.1.3), the conversion of species rich agricultural habitats such as fallows or grasslands to energy croplands is of particular concern (Glemnitz et al., 2008). The strong decline in fallow land (set-aside) since the year 2000 is at least partially due to the increased demand for land for bioenergy production (NABU, 2007; Glemnitz et al., 2008). Since 1992 and until 2007, farmers were required to take a certain amount (initially 15%, later 10%, then 5% and from 2007 onwards 0%) of their acreage out of production, to reduce grain surpluses and stabilise global food market prices. Biodiversity, and arable plant diversity in particular, typically greatly benefited from this set-aside requirement (Tscharntke et al., 2011). The production of renewable resources, such as bioenergy feedstocks was, however, allowed on the set-aside land, reducing its value for biodiversity conservation. This option was increasingly used since governmental subsidies additionally supported the production of renewable resources and the establishment of farm-based plants for anaerobic digestion throughout Germany (NABU, 2007). Demand for land for bioenergy production (in conjunction with increasing global demands for agricultural products), finally led to the complete abolishment of the European set-aside requirement in 2009 with severe negative impacts for the agricultural biodiversity. In 2013 only 200,000 ha of fallow arable land (i.e. less than 1.7% of the arable land) were left in Germany, representing a reduction by approximately 70% since 2007 (Destatis, 2007; NABU, 2007; Destatis, 2014). Fallow land is today no longer widely available as a refuge for agricultural biodiversity.

Another important issue is the loss of permanent grassland observed throughout Germany in recent years (BfN, 2014). Grassland is increasingly being abandoned or converted to cropland as its economic profitability is decreasing, due to changes in the structure of livestock production, strong subsidies for energy feedstocks and high commodity prices for cereals and other crops (Nitsch et al., 2012; BfN, 2014; Laggner et al., 2014). Since 1990, the grassland area in Germany decreased by >

0.88 million ha to about 4.6 million ha in 2013 (BfN, 2014). Although the direct and indirect impacts of energy cropping on the conversion of species rich grasslands are hard to quantify, it has been shown that the probability of grassland conversion increases in the vicinity of biogas plants (Laggner et al., 2014). The habitat quality of the remaining grassland also decreased drastically as intensification progressed (BfN, 2014). As biomass from highly fertilised grasslands with a high cutting frequency is better suitable as substrate for anaerobic digestion than biomass from extensively managed grasslands, the use of grassland biomass as a biogas feedstock can additionally incentivise grassland intensification (Ortloff, 2012). Biomass from extensively managed grasslands has a higher content of lignocellulosic compounds and is therefore better suitable for combustion instead of anaerobic digestion (Ortloff, 2012).

On the other hand, it has also been shown, that the majority of the bioenergy feedstocks are grown on former arable land (Laggner et al., 2014) leading to shifts in crop rotations and changes in crop frequencies at the landscape scale (Steinmann & Dobers, 2013). The impacts of the expansion of energy cropping on arable land are therefore in the focus of the research presented in this thesis. Two issues are of particular importance with regard to the impacts of energy cropping on the biodiversity of arable land: (a) changes in field management on conventionally managed arable land and (b) the intensification of land use, i.e. the loss of extensively managed, low-input cropland. The consequences of both issues have been addressed in the present work.

The direct impacts of 'first generation', annual energy crops on biodiversity have only rarely been addressed by research (Immerzeel et al., 2014; Meyer et al., 2014), although they dominate the energy cropping sector in Europe (EEA, 2013a). The available results indicate that effects differ strongly between different targeted taxa. Skylarks (*Alauda arvensis*) have been shown to be particularly negatively affected by maize cultivation (Nagy et al., 2009; Gevers et al., 2011; Everaars et al., 2014; Sauerbrei et al., 2014), whereas a carabid beetle and a linyphiid spider were found to benefit (Gevers et al., 2011). The status of the arable flora in annual energy cropping systems has yet received little attention. Available results show negative effects of maize cropping on arable plant diversity (Glemnitz et al., 2008; Waldhardt et al., 2011; Meyer et al., 2014).

Direct impacts on the abiotic environment and soil organic matter cycles

Apart from the impacts on biodiversity, energy cropping also affects land based greenhouse gas (GHG) emissions, nutrient and water cycles. Estimates of the contribution of bioenergy to the reduction of GHG emissions to mitigate climate change differ widely depending on the feedstocks, the conversion technologies and the parameter setting of the respective model (Cherubini & Strømman, 2011). Although the majority of the scientific assessments found net reductions of GHG emissions

when bioenergy replaces fossil fuels, significant uncertainties remain, particularly with regard to the impact of land use change and land management change (Cherubini & Strømman, 2011). According to the FNR (2013i), the use of bioenergy in Germany led to a reduction of GHG emissions of 70.8 million t CO_2-equivalents (CO_2e) in 2012. This is a considerable amount equalling 7.5% of the total German GHG emissions of 940 million t CO_2e in 2012 (UBA, 2014). No information is available to which extent land use change (i.e. GHG emissions from the conversion of permanent grassland to arable land) and changes in the intensity of land management (resumption of high-input production on former fallow land) were incorporated in this evaluation.

Despite these uncertainties, general patterns with regard to the GHG balance of particular bioenergy feedstocks and conversion technologies have been observed. Firstly, the net GHG reductions from transport biofuels are generally lower than the reductions due to heat and power generation from biomass (Rowe et al., 2009; Cherubini & Strømman, 2011). In Germany, transport biofuels were responsible for only 6.6% of the GHG reduction from bioenergy (4.7 million t CO_2e), whereas heat accounted for 53.6% and power generation for 39.7% (FNR, 2013i). Secondly, 'second generation' energy crops, i.e. perennial crops such as SRC or miscanthus, which are not used for food and/or fodder production, have been found to contribute more strongly to GHG reductions than 'first generation' annual energy crops such as maize or oilseed rape (Rowe et al., 2009; Cherubini & Strømman, 2011; Havlík et al., 2011). Thirdly, energy cropping on former arable land will have a more positive effect on the GHG balance than energy cropping which involves the conversion of grassland or forest (St. Clair et al., 2008; Rowe et al., 2009). Pre-harvest GHG emissions from oilseed rape production have been found to greatly exceed emissions from intensively managed grasslands and to be similar to emissions from cereal cropping (St. Clair et al., 2008). GHG emissions from low-input SRC and miscanthus can on the other hand be equal or even lower than emissions from intensively managed grasslands (St. Clair et al., 2008).

Energy cropping also affects soil carbon cycles. The soil organic matter (SOM) balance largely depends on the kind of energy crop, the local soil and climate conditions, the soil cover and the frequency and depth of tillage. Maize cultivation generally contributes to the reduction of the SOM content (Willms, 2013). Appropriate management, including diverse crop rotations, the planting of catch crops and the recirculation of crop residues and digestate from anaerobic digestion, is necessary to avoid SOM losses (Willms, 2013). As compared to annual crops, perennial energy crops, such as SRC or miscanthus, have the potential to sequester additional carbon in the soil biomass (Don et al., 2012). A recent review (Don et al., 2012) revealed, however, no positive effect on the C-balance if perennial energy crops were established on former grassland.

There is a large body of literature dealing with the impacts of energy cropping on soil nitrogen cycles and N-leaching. Annual energy crops generally need higher and more regular fertiliser inputs than for

instance SRC or other perennial energy crops. In comparison with annual cropping systems, second generation energy crops are therefore expected to reduce N-surpluses and leaching to the groundwater (Aronsson et al., 2000; Pugesgaard et al., 2014). Impacts of annual energy crops on N-leaching are, however, expected to differ widely depending on the actual field management practices and the local site conditions (Gebel et al., 2013). An assessment of different annual energy crops from Saxony nevertheless shows potential reductions in N-leaching to the groundwater with increased cultivation of maize and oilseed rape and reductions in winter cereal cultivation if the conversion of grassland to arable land is avoided (Gebel et al., 2013).

Energy crops also differ in their water-use efficiency and exhibit widely differing rates of evapotranspiration. Concerns have been raised that the cultivation of water-demanding energy crops, such as SRC, can reduce groundwater recharge and affect the hydrology of local catchments if grown in large quantities (Dimitriou et al., 2009). Scientific results indicate that the effects of energy crops on local water cycles are highly site-specific and differ according to the crop and the variety, the local precipitation and soil conditions (Dimitriou et al., 2009). Proposals have been made to label energy crop reproductive material according to its water use strategy in order to enable farmers to choose locally appropriate varieties (Schmidt-Walter et al., 2014).

With the exception of water consumption, second generation, perennial energy crops are expected to improve the environmental performance of energy cropping as compared to annual energy crops. Positive effects are, however, much less likely, if the establishment of energy crops directly or indirectly involves the conversion of permanent grasslands or other natural or semi-natural habitats.

Indirect impacts

Indirect land use change (ILUC) occurs when land with a high carbon storage capacity (generally implying land with biodiversity-rich natural or semi-natural land cover) is converted to farmland to meet the demand for commodities displaced by the production of biofuel feedstocks (Plevin et al., 2010). Examples include the encroachment of pastures into the Amazonian forest caused by the replacement of livestock farming by soy and biofuel production at its original site (Arima et al., 2011), and the conversion of Indonesian lowland tropical rainforest and the ploughing up of grasslands in the US to substitute farmland for food/fodder production lost to oil palm or corn cultivation for biofuel production at other locations (Fargione et al., 2008; Searchinger et al., 2008). Indirect land use change not only affects the sustainability of energy cropping in the tropics and the prairies of North America, but can also be observed in the cultural landscapes of Europe. In Germany, grassland which is converted to cropland is in most cases not directly used for energy cropping, but may be used to produce other crops which are competing with the energy crops for land (Laggner et al., 2014). The

effects of ILUC on GHG emissions and biodiversity are hard to quantify scientifically, but have been shown to potentially exceed the direct effects of energy cropping (Lapola et al., 2010; Plevin et al., 2010; Dumortier et al., 2011; Immerzeel et al., 2014).

1.2 The study areas

1.2.1 Location, geology and climate

The data collection for this work was carried out in two study regions in Central Germany (Fig. 1.7 & 1.8). For the purpose of the following chapters they will be referred to as the 'Lower Saxon hills' (LS, 1,118 km²) and the 'Thuringian Basin' (TB, 1058 km²). As this work was part of the larger research framework 'BEST – Bioenergie Regionen Stärken' ('Making bioenergy regions stronger') research from a wide variety of scientific disciplines took place in these regions in 2011 and 2012. The first study region, the Lower Saxon hills, is defined by the administrative borders of the county of Göttingen in southern Lower Saxony. The second study region, the Thuringian Basin, extends from the city of Mühlhausen in the west to Kölleda in the east and from Erfurt in the south to Hornsömmern in the north. It ranges across parts of four Thuringian counties, i.e. the county of Unstrut-Hainich, the county of Gotha, the county of Sömmerda and the city of Erfurt.
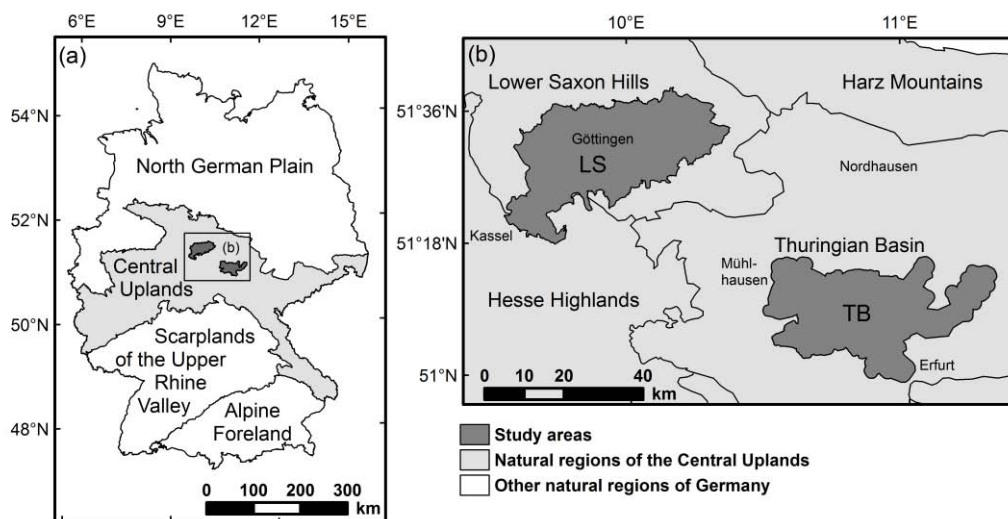
Fig. 1.7 Location of the two study regions (a) in Germany and (b) within the German Central Uplands (LS – Lower Saxon hills, TB – Thuringian Basin). Borders of LS are congruent with the county of Göttingen in Lower Saxony. TB represents the parts of the Thuringian Basin which were part of the larger research framework BEST in which this work was conducted. The map is based on Universal Transverse Mercator (UTM) projection; zone 32 N (WGS 1984). Biogeographical boundaries were adopted from Meynen & Schmithüsen (1953-1962).
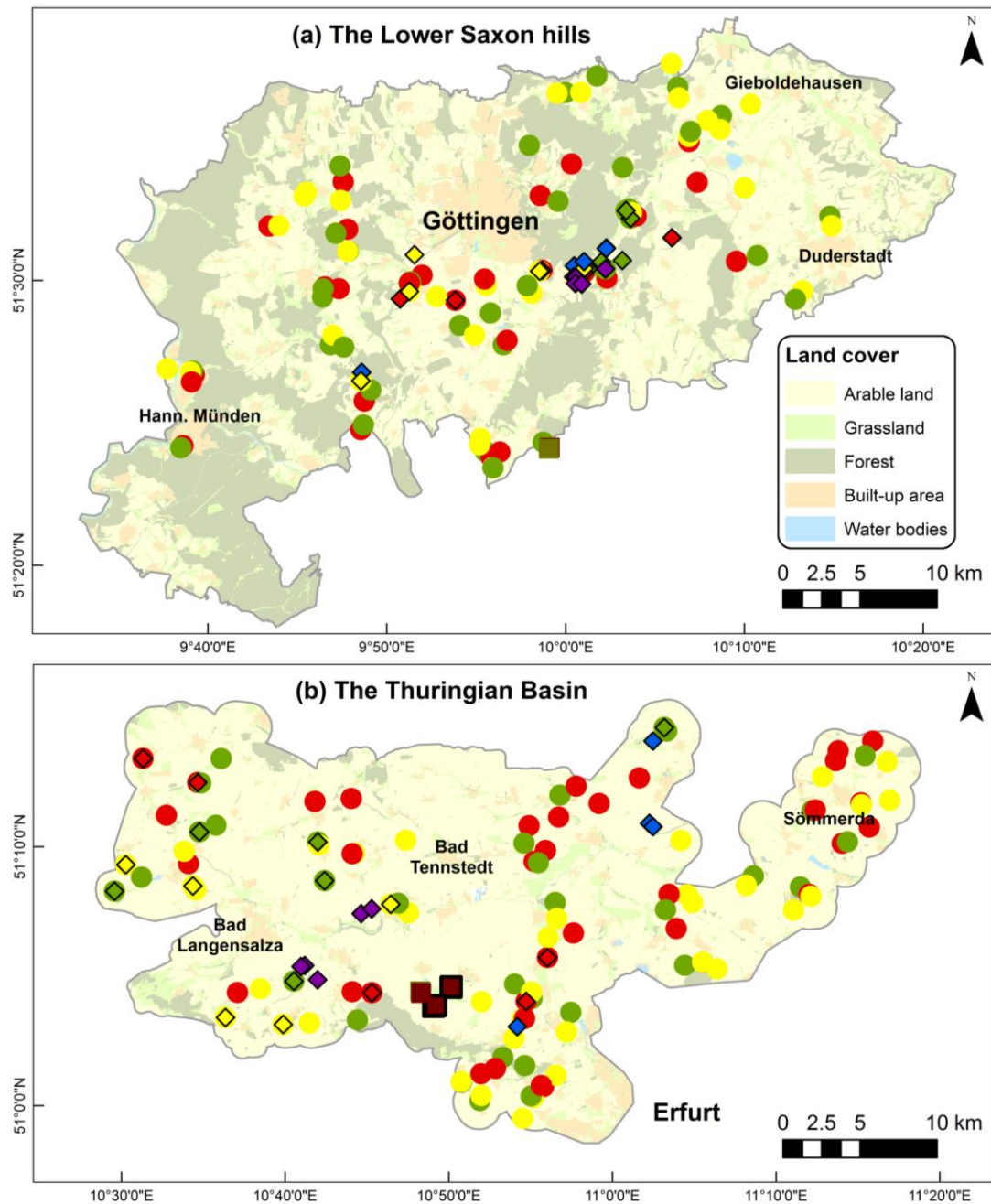
Fig. 1.8 Location of the sampling sites within the two study regions (a) the Lower Saxon hills and (b) the Thuringian Basin. The maps are based on Universal Transverse Mercator (UTM) projection; zone 32 N (WGS 1984).

Both study regions are characterised by a hilly countryside with plains highly suitable for agricultural production due to glacial loess deposits. The Lower Saxon hills are dominated by a mosaic of geological sediments from the Germanic Trias, including Buntsandstein, Muschelkalk and to a lesser extent also Keuper (Fig. 1.9). Loess deposits are mainly found in the Leine valley west of Göttingen and in the central part of the northern Eichsfeld in the east of the county. Contrarily, the Thuringian Basin is characterised by a mosaic of loess and loam deposits in the valleys and Keuper sediments on the ridges and plateaus. The main Muschelkalk belt is the 'Fahner Höhe' ridge at the southern edge of the study region. Buntsandstein sediments are lacking in the Thuringian Basin.

The altitudinal range of both study areas is similar (Lower Saxon hills 104–579 m a.s.l.; Thuringian Basin 134–413 m a.s.l.), but the climatic conditions differ considerably. Due to the east-west gradient of continentality in Central Europe, the climate in the Thuringian Basin is drier (mean annual precipitation 500–550 mm) and slightly warmer (mean annual temperature 9.2 °C) than in the Lower Saxon hills (650 mm and 8.7 °C respectively; DWD, 2013).

1.2.2   Land use history

Archaeological records confirm the presence of human settlements and agricultural practices as early as 7000 years before present in the two study regions (Beier & Einicke, 1994; Jantz, 2008).

The Neolithic people already cultivated emmer and einkorn wheat (*Triticum dicoccon* and *T. monococcum* respectively), pea (*Pisum sativum*), lentil (*Lens culinaris*) and less commonly opium poppy (*Papaver somniferum*) and barley (*Hordeum vulgare*) (Behre, 2008).

These crops were cultivated on cleared forests plots. When the soil was depleted, the cultivation shifted to another site and the field was left fallow to regenerate for several years and/or used for grazing purposes (Willerding, 1986). This kind of management, also called 'Feldgraswirtschaft' (i.e. the mixture of cropland and grassland periods), was practiced widely until the medieval ages throughout Central Europe (Pott, 1992). It led to the development of weed assemblages which rather resembled today's grassland plant communities due to the high prevalence of perennial species (Pott, 1992; Hofmeister & Garve, 2006). Typical cropland species such as *Bromus secalinus*, *Lapsana communis* or *Fallopia convolvulus* have nevertheless been detected regularly in Neolithic fossil records (Hilbig et al., 2013). Lists of arable plant species found in Neolithic fossil records from Central Europe have been published by Willerding (1986) and Jacomet et al. (1991).

An important change in field management occurred during the early to high medieval when the more efficient three-field rotation system ('Dreifelderwirtschaft') replaced the 'Feldgraswirtschaft' in
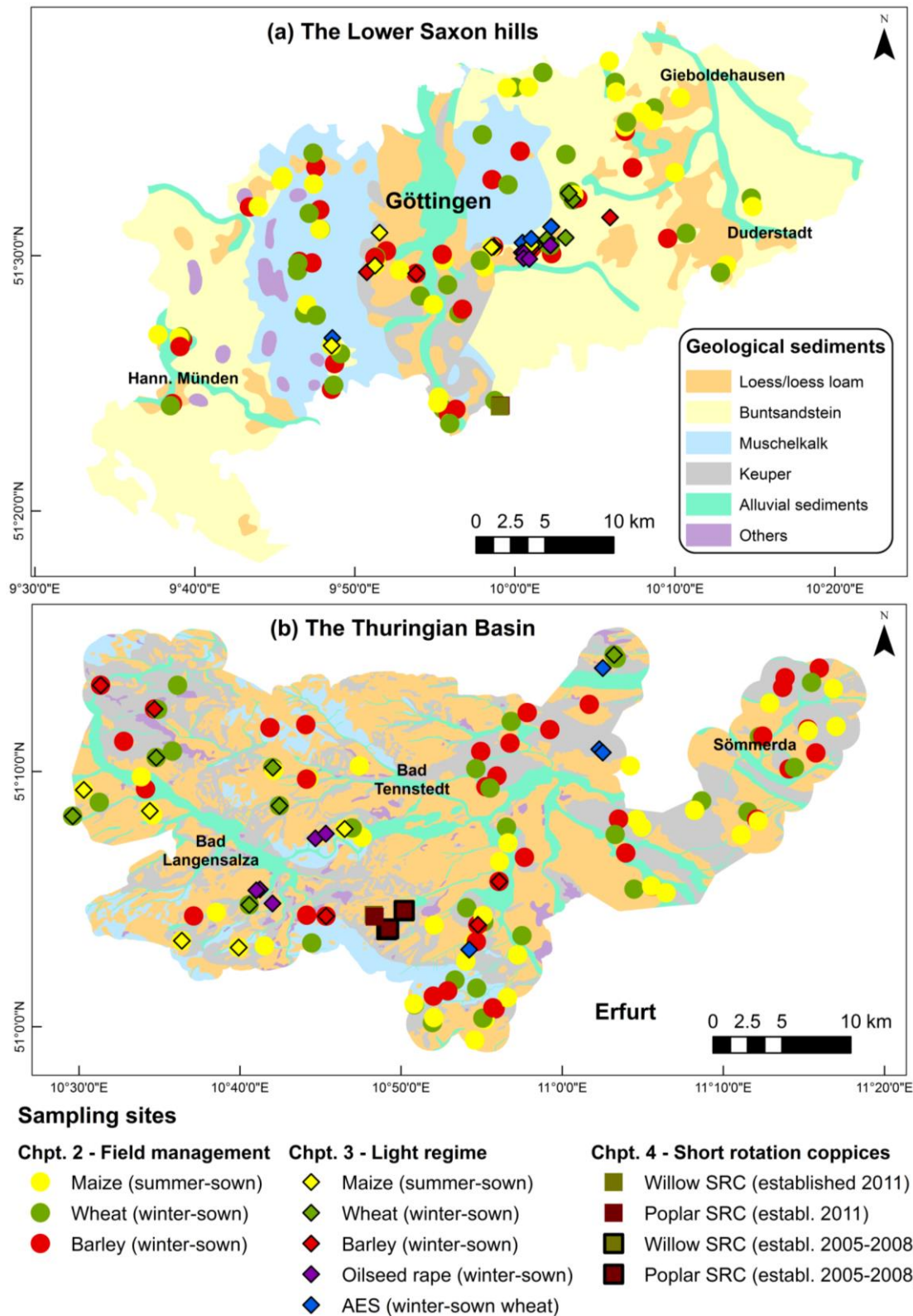
Fig. 1.9 Geology of the study regions (a) the Lower Saxon hills and (b) the Thuringian Basin. Map (a) is based on the geological map for Lower Saxony (GÜK500, 1:500,000; LBEG, 2014) and (b) is based on the soil geological map of Thuringia (1:100,000; Rau et al., 2000). Both maps are presented in Universal Transverse Mercator (UTM) projection; zone 32 N (WGS 1984).

Central Europe (Pott, 1992; Karg, 1995). The crop rotation in the three-field system consisted of winter sown cereals, summer-sown crops (mostly cereals or legumes) and a fallow year during which fields were used as pastures (Pott, 1992). These three types of land use rotated across the holding. Due to the shortening of the fallow period, perennial species became less important in the arable plant assemblages and the proportion of annual species increased (Willerding, 1986; Pott, 1992; Karg, 1995). Historical sources confirm that the three-field system was the prevalent form of field management in the Thuringian Basin and in the Lower Saxon hills since medieval times and was practiced until the middle of the 19th century in many parts of the study areas (Scheidemann, 1959; Rockstuhl, 2012; VG Schlotheim, n.d.). With the course of time, improvements of the three-field system such as land ameliorations, the introduction of new crops (i.e. potatoes and other root crops), improved breeds, and the introduction of mineral fertilisers in the 19th century led to changes in habitat conditions for the arable plant assemblages (Hofmeister & Garve, 2006). As fallow periods became less common and were largely replaced by root crop cultivation annual disturbance was prevalent on many fields. The importance of perennials declined further and species rich plant communities dominated by annual plant species developed (Pott, 1992). These species rich arable plant assemblages were widespread until the middle of the 20th century.

A major transition in agricultural practices in the study areas occurred after the Second World War. With the wider introduction of synthetic herbicides, the increased availability of mineral fertilisers, improvements in seed cleaning techniques and technical advances of the field machinery the diversity of the arable plant assemblages started to impoverish rapidly (Hofmeister & Garve, 2006). With the new machinery, land amelioration progressed, leading to increased field sizes and the loss of landscape structural elements such as field margins, wetlands, hedges and copses (Pott, 1992).

This process of intensification differed between the study regions, due to differences in agricultural policies on both sides of the inner German border after 1949. As part of the agrarian reform in the German Democratic Republic many land owners in the Thuringian Basin were expropriated. The land was collectivised and farmed by agricultural production cooperatives (in German: Landwirtschaftliche Produktionsgenossenschaft or LPG). Small fields were joined and landscape features such as hedges, copses, rocks and ponds were removed to allow for more efficient agricultural production with larger machinery (Humm, 1999). A highly productive agricultural landscape with very large fields and few structural landscape elements developed. After the German reunification, the state-owned agricultural land was returned into private hands or reallocated to the regional administrative bodies (federal states or municipalities) to whom it formerly belonged (Dells, 2008). Many federal states, municipalities and private land owners subsequently rented their land to large, privately owned, agricultural cooperatives which are today responsible for farming the land. Consequently, the re-privatization of land did not result in overall decreases in field size. The average size of an administrative field unit is today 18.8 ha in the Thuringian Basin and fields of 20–100 ha dominate the landscape.

In the Lower Saxon hills no such collectivization took place and the average field size is therefore smaller until today (6.6 ha). Only very few fields exceed 20 ha. Landscape features such as hedges and tree lines are nevertheless not very common, particularly in the areas with fertile loess soils. As a large number of family farm businesses were and are still economically directly dependent on the agricultural production in Western Germany, land use intensity is high and fertile land is only rarely spared from production. Due to these developments, many characteristic and formerly common plant species of arable habitats, such as the common corncockle (*Agrostemma githago*, Fig. 1.2) or the summer pheasant's eye (*Adonis aestivalis*), suffered strong declines in Central Germany (Meyer et al., 2013) and are nowadays threatened in many regions of Central Europe.

1.2.3   Energy cropping in the study areas

With regard to the importance of energy cropping, the two study areas can be regarded as representative of agricultural areas in western and eastern Germany with fertile soils and a prevalence of arable farming (I.M.A, 2005). Livestock farming is also present in both regions, but is of minor importance as compared to other regions of Germany (I.M.A, 2005). The production of silage maize both for fodder and as biogas feedstock is therefore traditionally less important in both study regions than in regions with a focus on livestock farming. In the Lower Saxon hills, approximately 6% of the arable land was cultivated with silage maize in 2010 (LWK Niedersachsen, 2010). Assuming that 32% of the silage maize is used for energy production (i.e. the national average) this results in 2% of the arable land being cultivated with energy maize. The agricultural literature and interviews with the farmers showed that no systematic differences in the management of energy maize fields and fodder maize fields are to be expected. The influence of energy maize production on arable plant diversity is therefore assumed to originate exclusively from the expansion of the area cultivated with maize. The largest biogas plant in the Lower Saxon hills was established in 2011 and produces about 190 TJ of energy (power and heat) annually (Biogas Göttingen, 2013). It mainly uses maize silage as substrate (Biogas Göttingen, 2013). According to the operators, the necessary feedstocks are produced on approximately 600 ha of arable (Biogas Göttingen, 2013). This results in about 4.8% of the arable land within a circle of approximately 8 km around the plant being used for energy production (assuming that 50% of the land within this circle is cropland) (Biogas Göttingen, 2013). This impact on agricultural production is considerable, but the effects are far from producing "maize deserts" or resulting in large-scale maize monocultures in the region. For comparison, winter-sown wheat is produced each year on about 45% of the arable land in the Lower Saxon hills (Fig. 1.10; LWK Niedersachsen, 2010).

The situation is similar in the Thuringian Basin. In the county of Sömmerda silage maize was grown on 8% of the arable land in 2010 (LWA Sömmerda, 2011). Approximately 50% of the silage maize,

representing 4% of the arable land, was used for anaerobic digestion (LWA Sömmerda, 2011). Sömmerda is the county with the highest density of plants for anaerobic digestion in the Thuringian Basin (TLL, 2014). In the county of Unstrut-Hainich maize (including silage maize and grain maize) was produced on only 5% of the arable land in 2010 (LWA Leinefelde-Worbis, 2011). The prevalence of energy maize production is therefore also generally rather low in the second study area.

A different picture is found for oilseed rape which was grown on 19% of the arable land in the Lower Saxon hills, on 16% of the arable land in the county of Sömmerda and on 20% of the arable land in the county of Unstrut-Hainich in 2010 (LWK Niedersachsen, 2010; LWA Leinefelde-Worbis, 2011; LWA Sömmerda, 2011). Assuming a biodiesel end use for 60% of the production (the national average) 11.4%, 9.6% and 12% of the arable land was used for biodiesel feedstocks respectively. These values greatly exceed the national average of 7% (FNR, 2013c; Destatis, 2014). Farmers are often not aware at the beginning of the season whether their rapeseed produce is going to be used for biodiesel production or for other purposes by the purchaser. The management of oilseed rape fields is therefore also not influenced by the end use.
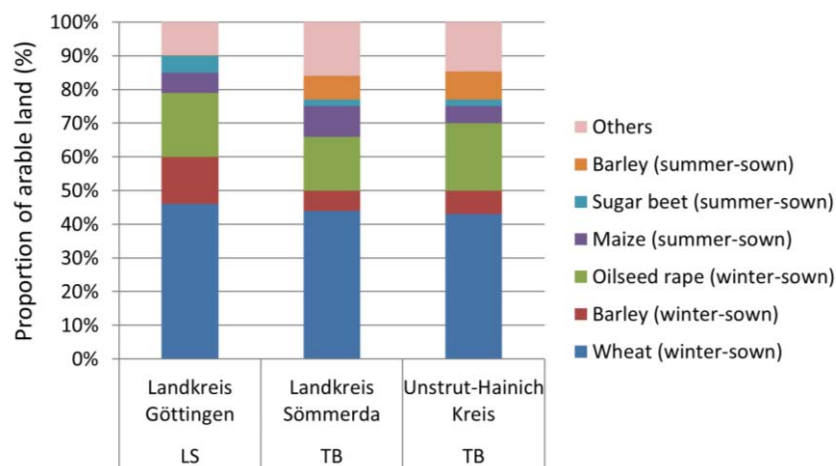
Fig. 1.10 Proportion of arable land used for the production of different crops in the two study regions in 2010. Data from the county of Göttingen (Landkreis Göttingen, LWK Niedersachsen, 2010) represent the situation in the Lower Saxon hills (LS), whereas data from the county of Sömmerda (Landkreis Sömmerda, LWA Sömmerda, 2011) and the county of Unstrut-Hainich (Unstrut-Hainich Kreis, LWA Leinefelde-Worbis, 2011) represent the Thuringian Basin (TB). Maize includes silage maize for fodder and biogas production as well as grain maize. Statistics on the specific share of biogas maize were not consistently available at the county level.

Solid biofuels, such as short rotation coppices, are only produced on few fields in both study regions by farmers who are curious to test new land use options. The current impact on agricultural production at landscape scale is therefore still negligible.

As shown above, the two study regions are not among the current hotspots of bioenergy production in Germany. They are nevertheless suitable for studying the impacts of biomass production for energy use, as they can beregarded as representative of the wider German agricultural landscape in regions shaped by arable farming, rather than livestock production. The Lower Saxon hills represent the current agricultural situation in the western German regions, whereas the Thuringian Basin is indicative of the situation in the eastern German federal states of Central Germany.

1.3   General description of the applied methods

1.3.1   Vegetation sampling

The composition of the plant assemblages of energy cropping systems (maize for anaerobic digestion, oilseed rape for biodiesel, short rotation coppices for combustion) and control plots (conventional wheat and barley, fields managed according to agri-environmental schemes) was recorded using standard phytosociological methods (Dierschke, 1994; Wilmanns, 1998). Within a fixed sampling plot the cover (≈abundance) of all plant species present was recorded. To allow for comparisons with previous studies we used a standard plot size of 100 m^2 (2 m × 50 m) in arable fields (compare van Elsen, 1994) and a plot size of 25 m² (1 m × 25 m) in the short rotation coppices. The Braun-Blanquet cover abundance scale with modifications according to Wilmanns (1998) was applied consistently for all vegetation assessments (Table 1.1). It allows characterizing plant assemblages based on the cover and/or abundance of each species present.

Additionally, the total plant cover, the height, cover and development stage of the crop, the mean height and total cover of weeds, the height and cover of the tree/shrub layer, the cover of the moss layer and the total flower cover at the time of the survey were estimated visually. An exemplary data sheet as used for the vegetation survey can be found in appendix 1.A. The abiotic environmental variables recorded for each plot included the geographic coordinates, slope and aspect, elevation a. s. l. and the soil texture.

Table 1.1 Braun-Blanquet cover abundance scale according to Wilmanns (1998).

Symbol	No. of individuals	Cover
r	1	< 5 %
+	2–5	< 5 %
1	6–100	< 5 %
2m	> 100	< 5 %
2a	-	≥ 5 %, but < 15 %
2b	-	≥ 15 %, but < 25 %
3	-	≥ 25 %, but < 50 %
4	-	≥ 50 %, but < 75 %
5	-	≥ 75 %, but < 100 %

1.3.2   Soil sampling and analyses

1.3.2.1   pH

To measure the pH we used a standard procedure as described in HFA (2005). 10 g of fresh soil were mixed with 25 mL of demineralised water. After a minimum of 10–12 hours of immersion, the pH in H_2O was measured with a pH-meter. After adding 1.86 g of KCl and waiting for its complete dissolution, the pH was measured again with the same pH meter to obtain pH_{KCl}.

1.3.2.2   Effective cation-exchange capacity (ECEC) and base saturation

The effective cation-exchange capacity (ECEC, in $mmol_c$ kg^{-1}) is a measure for the sum of the charges of all exchangeable cations of a soil. It describes the amount of cations which are available for plant growth at intermediate time scales. Contrary to the potential cation-exchange capacity (CEC), ECEC is independent of the soil pH and can therefore be used to compare cation availability across a wide range of different soils. For the purpose of this work ECEC was determined by inductively coupled plasma analysis (ICP) of a $BaCl_2$ soil percolate (HFA, 2005). 2.5 g of fresh soil were percolated with 100 mL of 0.1 mol L^{-1} $BaCl_2$-solution during 3 h. To avoid biotic contamination, the percolate was subsequently mixed with 5 mL of 65% HNO_3 and kept in the freezer. We used ICP analysis to determine the specific contents of the different cations in the soil samples. The ICP analysis works based on atomic emission spectroscopy, i.e. it measures the specific light emissions of ions which, at extremely high temperatures, return from an excited electronic state to the ground electronic state (Boumans, 1987).

Based on the ICP results the base saturation (BS, in %) was calculated as the sum of the measured amounts of all basic cations divided by the sum of all measured cations (eq. 1.1).

$$BS = \frac{[Ca^{2+} + Mg^{2+} + K^+]}{[Ca^{2+} + Mg^{2+} + K^+ + Al^{3+} + Mn^{2+} + Fe^{2+} + H^+]} \quad \text{(eq. 1.1)}$$

1.3.2.3   C/N-ratio and soil organic matter content

The C/N-ratio of the soil was determined with a C/N-elemental analyser (vario EL III, Elementar, Hanau, Germany). The total amount of carbon (C_{tot}) and nitrogen (N_{tot}) were determined from samples of 10 mg of dried soil prepared for ignition in tin capsules. The C/N-elemental analyser measures C and N based on catalytic tube combustion in an oxygenated CO_2-atmosphere at high temperatures (Elementar Analysesysteme GmbH, 2005). The freed gases are separated from each other and quantified with a thermal conductivity detector.

The amount of organic carbon (C_{org}) in the soil samples was obtained based on the loss-on-ignition method. Samples were heated to 450 °C for at least 5 h to burn all organic carbon. Subsequently, the amount of the remaining inorganic carbon (C_{inorg}) was measured with the C/N-elemental analyser based on 20 mg of the raw ash. C_{inorg} was subtracted from the total carbon (C_{tot}) to obtain C_{org}. An estimate of the soil organic matter content (in %) was obtained by multiplying C_{org} with the factor 1.72 (Baize, 1993).

1.3.2.4   *Plant-available phosphorus*

In the soil, phosphorus (P) is part of a range of organic and inorganic compounds. The vast majority of the P is not immediately available to be absorbed by plants as it is bound in minerals or is part of the soil organic matter. For ecological studies the plant-available P, i.e. the amount of P in the soil which can easily be absorbed by plants, is of importance. To measure the amount of plant-available P, anion-exchange resin bags can be used to simulate the P-exchange at the roots (Sibbesen, 1977). This method captures both the soluble P and non-stable inorganic P-compounds of the soil solution (Sibbesen, 1977). The fraction of non-stable inorganic compounds which is absorbed by the plants depends on the concentration of these compounds in the soil solution. This process can be reproduced with the resin bag method. To enable the P to bind to the resin, 1 g of fresh soil and a resin bag were immersed in 30 mL demineralised water and the mixture was agitated for 16 hours. In a second step, the P was extracted again from the resin bag by washing the bag twice for 30 min, first with 25 mL of 10% NaCl-solution and afterwards with 25 mL of 2% NaOH-solution. Both solutions with their respective P-fractions were joined. The P-content of the complete solution was measured colourimetrically with the molybdate/ascorbic acid method as modified by Murphy & Riley (1962).

1.3.3   Measuring PAR transmissivity

To determine the shading effect of high-yielding energy crops, the proportion of photosynthetically active radiation reaching the ground in energy crop stands and control crops (PAR transmissivity, in %) was determined. Quantum sensors (LI-190 Quantum Sensor, LI-COR, Lincoln, Nebraska, USA; Biggs et al., 1971) were used to measure the PAR flux density (in µmol photons $m^{-2}s^{-1}$) above and below the crop canopy. To measure how much PAR was available for weed growth below the crop canopy, one PAR sensor was positioned at ground level ~15 cm above the soil surface in order to record the light climate experienced by the weeds (transmitted PAR; PAR_{trans}). A second PAR sensor was placed ~50 cm above the crop canopy for synchronously measuring incident PAR flux density (PAR_{inc}). Incident and transmitted PAR were recorded synchronously with a LI-1400 data logger (LI-COR, Lincoln, Nebraska, USA). PAR transmissivity (TPAR, in %) was calculated as

$$TPAR = \frac{PAR_{trans}}{PAR_{inc}} * 100 \qquad\qquad \text{(eq. 1.2)}$$

To avoid biased readings, sensors were placed with great care in the stand in order to avoid damaging the crop canopy. The mobile sensor was mounted at the end of a 1.5 m-long stick which allowed conducting the measurements at a distance from the operator where he influenced radiation flux density only minimally. Both sensors rotated freely so that they were always in horizontal position. In each survey plot, ten measuring points were randomly selected and five consecutive light measurements taken at time steps of ~5 s at each point. Mean TPAR per plot was obtained by averaging over the five consecutive measurements per point and subsequently calculating the mean across the ten points per plot.

All PAR readings were obtained around noon time (10 a.m. to 3 p.m.) on days with homogeneous sky conditions without rapid fluctuation of incident radiation. Although sky conditions and sun angle necessarily varied between different measurement days, we are confident that our data are not biased by these variations, as Hipps (1983) showed that in late-season crop stands with high leaf area indices, measurements of PAR interception and transmission are not influenced by sun angle and cloud cover.

1.3.4 Data analysis

1.3.4.1 General comments on the statistical approaches used

In this section a short general overview of the main statistical approaches applied will be given. More specific descriptions of the data analyses can be found in the 'methods' sections of the respective chapters. The main analyses of this thesis applied a correlative approach, i.e. it was determined how well variations in arable plant diversity could be represented by (multiple) linear regression models with a set of predictors. For multivariate response variables we applied redundancy analyses (RDA) or canonical correspondence analyses (CCA). Both techniques are also correlative approaches. In CCA, the components of the response matrix *A* (species matrix) and the predictor matrix *B* (i.e. management, environmental and spatial factors) are extracted in a way which maximises their correlation in a single step (van den Wollenberg, A. L., 1977). In RDA two steps are executed. First, the components of the response matrix *A* are extracted maximising their correlation with the predictor matrix *B*. The components of the resulting new matrix *C* are then extracted by maximising their correlation with the response matrix *A* (van den Wollenberg, A. L., 1977). The redundancy statistic R^2 (more precisely termed 'regression coefficient of determination' for univariate regression analyses) gives the percentage of the variation of the response matrix *A* explained by the predictor matrix *B* (Peres-Neto et al., 2006). For RDA the redundancy statistic can be directly determined (van den Wollenberg, A. L., 1977). For CCA the proportion of the total inertia attributable to the predictor

matrix is commonly used in ecological research as redundancy statistic (Peres-Neto et al., 2006; Legendre, 2008). R^2 varies with the degrees of freedom of the respective model and with the sample size (Peres-Neto et al., 2006). If several models with different numbers of predictors and/or differing in sample size are to be compared R^2 needs to be adjusted to account for these variations. $R^2_{adjusted}$, defined as

$$R^2{}_{adjusted} = \frac{n-1}{n-p-1}(1-R^2)$$ (eq. 1.3),

where n is the sample size and p is the number of predictors, is commonly applied for this purpose in ecological assessments (Peres-Neto et al., 2006; Legendre, 2008).

Univariate and multivariate correlative statistics are useful tools which can be used to explore and also to refute hypotheses. Correlative statistics can also support hypotheses, but they can never be interpreted as prove of a causal relationship between the predictor and its response variable. The results presented in the following chapters should be interpreted with this limitation in mind.

1.3.4.2   Variation partitioning

In ecological systems it is common to be confronted with a wide range of predictor variables which are interrelated with each other. These predictor variables can often be clustered and organised in subject-specific groups or 'sets of predictors'. To rank the different sets of variables according to their importance and to explore the relationships between different sets, 'variation partitioning' is commonly applied in community ecology (Legendre, 2008). Variation partitioning can be used to discern the part of the variation which is exclusively attributable to a certain set of predictors ([a], [b] or [c] in Fig. 1.11), while controlling for the effects of other sets and to quantify the variation which cannot be attributed to a single set of predictors.

The different steps of the variation partitioning according to Legendre (2008) will be illustrated here by using an example set of predictors represented by the red circle in Fig. 1.11. Firstly, the 'gross effect' ([A] = [a]+[ab]+[ac]+[abc]) of each set of predictors is determined by regressing it against the response matrix without incorporating any conditional factors. The fractions [a], [b] and [c] can be obtained directly from partial regression, RDA or CCA by partialling out the effects of all other predictors in the model. The significance of these so called 'net effects' can therefore also be determined directly. To obtain [ab], [ac], [bc] and [abc] further intermediate steps are necessary. The fraction [ab] can be determined by subtracting [a] and [b] from the variation explained by [AB] after partialling out [C]. Likewise, [ac] can be determined by subtracting [a] and [c] from the variation explained by [AC] after partialling out [B]. [abc] is then determined by subtracting [a], [ab] and [ac] from [A]. The significance of the shared fractions can therefore not be determined (Legendre, 2008).

As some degree of correlation among the predictors is common in ecological research, shared fractions occassionally turn out to be negative (Legendre, 2008).
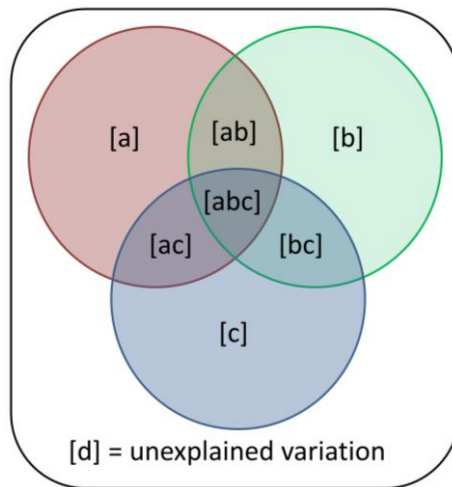
Fig. 1.11 Venn diagram representing the partitioning of variation between three sets of predictors. The rectangle represents 100% of the variation in the response matrix. The fractions [a], [b] and [c] represent the 'net effect' of the respective set of predictors, whereas the sum of all fractions encompassed by a circle (e.g. [A] = [a]+[ab]+[ac]+[abc]) represents the 'gross effect'. Adapted from Legendre (2008).

1.4 General objectives and outline of the chapters

As shown above, important changes in agricultural production patterns are currently taking place in Germany and at the European scale in order to meet the growing demand for bioenergy. Crop frequencies and crop rotations on arable land are changing rapidly as more and more energy crops, such as maize and oilseed rape, are being cultivated (Steinmann & Dobers, 2013). Changes in cropping patterns are associated with changes in field management practices concerning the timing, quality and quantity of cultivation, fertilisation and herbicide use. Consequently, they can lead to changes in other abiotic factors associated with the field management (e.g. the light climate in the crop stands). In the context of the strong and ongoing decline of arable plant diversity (Sutcliffe & Kay, 2000; Fried et al., 2009; Storkey et al., 2012; Meyer et al., 2013), it seems urgently necessary to investigate the impacts of these large scale changes in crop frequencies, field management practices and abiotic conditions on arable plant diversity. As arable plants are at the basis of the farmland food web, impacts can be expected to additionally affect organisms of higher trophic levels which depend on arable plants for food, reproduction and shelter. Only a sound understanding of the key factors affecting arable plant diversity will make it possible to develop effective strategies and tools to reconcile agricultural production and biodiversity conservation. This work aims to contribute to fill this research gap.

In **chapter 2**, an analysis of the effects of changes in field management practices on arable plant diversity is presented. Records of the arable flora in conventionally managed fields of (energy) maize, winter-sown wheat and winter-sown barley were related to a comprehensive set of field management factors (including fertilisation, herbicide use, conversion tillage, crop rotation and land use history). The relative importance of management vs. environmental factors for arable plant diversity was quantified, and the management factors with the greatest impact were determined. To address the hypothesis that the expansion of maize production negatively affects arable plant diversity if it replaces traditional food and fodder crops, we investigated the importance of the choice of crop for weed cover, species richness and community composition on arable land and discerned interactions with differences in field management between crops.

In **chapter 3** the effects of changes in crop stand density and light regime in different conventionally managed crop stands (maize, oilseed rape, winter-sown wheat and winter-sown barley) on arable plant diversity were investigated. We tested the hypothesis that less light reaches the ground in tall-growing maize fields than in fields of oilseed rape or cereals leading to less favourable habitat conditions for the arable flora. Additionally, we compared the light regime between conventionally managed fields and fields managed according to an agri-environmental scheme for the protection of arable plant species. More specifically, the following questions were addressed: (1) How does the light regime at ground level vary between cropping systems? (2) Which spatial scale of observation contributes most to the variation in light conditions? (3) How much of the variation in PAR transmissivity between arable fields can be explained by crop performance, field management and/or regional effects?

Whereas chapters 2 and 3 are dealing with the impacts of annual, widely cultivated energy crops on arable plant diversity, **chapter 4** focusses on an alternative, 'second-generation' bioenergy feedstock, which is still in the 'investigation phase'. Due to potentially high yields and low requirements of fertilisation and herbicide use, short rotation coppices (SRCs) are widely discussed in the scientific community as a promising feedstock to reduce the negative ecological effects of energy cropping. Chapter 4 investigates the impacts of SRC (willow and poplar) on the plant diversity of agricultural landscapes and discusses ways to increase the value of SRC as habitat for vascular plants. The following research questions were addressed: (1) How do habitat conditions and the associated plant communities in SRCs change with the time since planting? (2) How do habitat conditions in SRCs during the different growth stages compare to habitat conditions in other types of land use? Based on these findings the implications for plant species conservation were discussed and management recommendations to maximise positive effects of SRCs on plant diversity were derived.

In **chapter 5**, the main findings of the previous chapters are summarised and discussed in the wider ecological and political context.

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Chapter 2

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Seifert, C., Leuschner, C., Culmsee, H. (2015)

Arable plant diversity on conventional cropland – The role of crop species, management and environment

Abstract

Arable plant assemblages are affected by a wide range of management and environmental factors. It is yet unclear which factors are the most important predictors of arable plant diversity and composition in contemporary agricultural landscapes. The dramatic decline in agrobiodiversity calls for strategies to reconcile production and biodiversity goals on the basis of sound knowledge about the determinants of farmland biodiversity. Based on surveys of the arable flora on 240 conventional maize and winter cereal (wheat and barley) fields in two regions of Germany, we assessed the importance of a large number of management, environmental and spatial predictors for arable plant abundance, species richness and community composition. All conventionally managed fields had very low plant diversity (3-6 species 100 m²). A complete model with all examined factors explained 40 and 32% of the variation in species richness and 42 and 36% of the variation in community composition for the field interior and the field margin, respectively. The choice of crop and other management factors (sowing date, fertilisation, herbicide use and crop rotation) were identified as important predictors of arable plant diversity in contemporary fields. Our results show that unless agrochemical input is markedly reduced, the potential for increasing the floristic diversity of high-input cropland is small.

2.1 Introduction

Arable non-crop plants play a vital and diverse role in farmland ecosystems, extending beyond being merely detrimental crop competitors. They are the basis of the cropland food web and provide forage, shelter and habitats for reproduction for a wide range of insects, birds and mammals (Marshall et al., 2003; Parish et al., 2009). It has also been shown that arable plants support beneficial arthropods that aid in suppressing pest populations (Altieri, 1999; Evans et al., 2011) and help to maintain favourable levels of plant-pollinator interactions in agricultural landscapes (Gabriel & Tscharntke, 2007). In many Northern and Western European countries, where arable plants are confronted with a continuous process of intensification of field management since the middle of the 20th century, arable plant diversity has been decreasing rapidly (Sutcliffe & Kay, 2000; Fried et al., 2009; Meyer et al., 2013). Many studies have shown that farming without herbicides and with reduced amounts of fertiliser input (i.e. organic farming or traditional low-input farming practices) benefits arable plant diversity and supports associated ecosystem services, such as pollinator and pest predator communities (Hole et al., 2005; Kleijn et al., 2009). Yet in many cases, yields and revenues from conventionally managed, high-input fields are higher (Seufert et al., 2012), and calls to seek opportunities for enhancing arable plant diversity and abundance without compromising crop yields remain common. While reconciling these two goals might seem contradictory in the light of standard models of yield loss and weed density (Cousens, 1985), a better understanding of the abiotic and biotic controls of arable plant communities can potentially highlight new ways for an integrated weed management. The Resource Pool Diversity Hypothesis of Smith et al. (2010), for instance, postulates that diverse soil resource pools can decrease the competition between weeds and crops and allow for higher weed abundance without reducing crop yields. In this study, we aimed at disentangling the complex interaction between environmental (soil, climate) and management-related factors (crop type, phenology, cultivation practices, fertilisation, herbicide use) in their effect on the diversity and composition of arable plant assemblages in the interior and at the margin of conventionally managed maize and winter cereal (wheat and barley) fields. A better understanding of the key factors determining biodiversity on agricultural land can potentially be used to optimise strategies for enhancing arable plant diversity without compromising production goals.

The term "plant diversity" is used here to summarize different characteristics of plant assemblages and is not restricted to simply assessing the number of species. The influence of different management and environmental factors can be expected to vary between diversity metrics (Fried et al., 2008). To address different aspects of arable plant diversity, we assessed the effects of field management and environmental factors on weed cover, species richness and community composition. Whereas the latter two have been assessed by previous studies (Andreasen and Scovgaard, 2009; Fried et al., 2008; Lososová et al., 2004; Pinke et al., 2012 and references therein), weed cover has only rarely been investigated (Pyšek et al., 2005) despite being a crucial factor for farmland management decisions. A

sufficiently dense weed cover composed of a diverse set of species with different traits implies more resources for a wider range of consumers and decomposers than mono-specific crop stands (which might nevertheless provide the same or even higher total biomass) (Marshall et al., 2003). At the same time, a higher weed cover negatively affects crop growth and yields and is therefore unfavourable from the crop production perspective (Cousens, 1985; Smith et al., 2009; Tang et al. 2014). Ideally, weed management at a regional scale would aim at maintaining high species richness and the complete set of regionally adapted arable plant assemblages while reducing weed cover to a level which balances the need to provide enough resources for maintaining functional pollinator and pest-predator communities with the objective to minimise yield loss (Marshall et al., 2003; Smith et al., 2009; Tang et al. 2014).

We expect management practices such as herbicide treatments, fertilisation, tillage, crop rotation and land use history to strongly influence all three response variables. Several studies investigated the impact of sowing date, tillage and N-fertilisation (Fried et al., 2008; Kleijn et al., 2009; Lososová et al., 2004; Pinke et al., 2012), but none of these studies examined the complete range of fertiliser applications, herbicide use intensity and aspects of crop rotation in their effect on the biological diversity of agricultural ecosystems.

Herbicide treatments aim at reducing weed cover and can also influence species richness and community composition by eradicating certain species completely and consequently fostering others (Délye, 2005; Owen, 2008). Both the overall intensity of herbicide applications and the types of active ingredients can be expected to determine these effects (Menne and Köcher, 2012). Herbicides can be classified according to their sites of action (SoA) as proposed by the Herbicide Resistance Action Committee (HRAC, Menne and Köcher, 2012). We expect herbicides targeted at a certain SoA to differ in their effectiveness in eradicating certain species and to most strongly affect a similar set of species (Délye, 2005, Owen, 2008, Nagai and Taya, 2015). This leads to the hypothesis that certain HRAC groups of herbicides have stronger effects on plant diversity than others and that a dose-dependent response between the three investigated diversity metrics and the herbicides of certain HRAC group exists.

N fertilisation is known to increase the competitive ability of the crop and to favour certain weed species, whereas it negatively affects species preferring less nutrient-rich soils (Kleijn and van der Voort, 1997; Pyšek and Lepš, 1991; Tang et al. 2014). P, K and S fertilisation without N input has been shown to more strongly favour weed than crop biomass (Grant et al., 2007; Tang et al. 2014). Consequently, we formulate the hypotheses that crop cover increases with N fertilisation, whereas weed cover and species richness decrease and the community composition shifts. The effects of P, K and S fertilisation are expected to vary depending on N fertilisation.

Mouldboard ploughing reduces weed abundance and has been found to reduce weed cover, but to have only little effect on species richness (Ball, 1992; Teasdale et al., 1991). Depending on the timing of ploughing it can, however, be expected to influence community composition (Ball, 1992). This leads to the hypothesis that weed cover and community composition are mostly affected by deep ploughing, whereas species richness is not.

We also hypothesise that land use history and crop rotation will have strong impacts on community composition through their longer term influence on the seedbank (Ball, 1992). With regard to weed cover and species richness we expect these effects to be overridden by the more immediate effects of chemical and mechanical weed control.

Previous studies on the effects of field management on arable plant assemblages in conventional fields have found the current and, to a lesser extent, also the preceding crop type to be major drivers of arable plant species richness and community composition (Andreasen and Scovgaard, 2009; Fried et al., 2008; Lososová et al., 2004; Pinke et al., 2012). We expect the factor crop type to represent information about crop-specific management practices relating to the timing, type and quantity of cultivation, fertilisation and herbicide use.

In accordance with results by Fried et al. (2008) and Lososová et al. (2004), we expect community composition to be affected by macro-environmental factors, such as soil and climate conditions and by micro-environmental parameters such as slope, aspect and crop cover regulating, for instance, radiation input and water availability. Management practices will also affect community composition, but the effects can be regarded as secondary, i.e. acting on the community within the limits of their suitable environmental envelope. Weed cover and species richness are expected to be more strongly determined by the immediate effects of management practices and to a lesser extent by environmental factors than community composition.

Available studies also indicate that the relative importance of environmental factors with regard to field management can be expected to be higher if gradients are larger, i.e. at cross-regional or nationwide scales (Cimalová and Lososová, 2009; Fried et al., 2008; Pinke et al., 2012). The present study encompasses relatively small gradients in climate and soil variables as compared to previous assessments (Andreasen and Scovgaard, 2009; Fried et al., 2008; Lososová et al., 2004; Pinke et al., 2012). We therefore expect the relative importance of management vs. environmental factors to be high as compared to these assessments. As field margins are in contact with adjacent habitats (Marshall and Arnolds, 1995) and commonly receive less fertiliser and herbicide input (Wilson and Aebischer, 1995), we also expect the importance of field management factors to decreases towards the field edge (Pinke et al., 2012).

These hypotheses can be summed up under the following main research questions:

1) What is the relative importance of management factors vs. environmental factors for arable plant diversity in conventional cropping systems at a regional scale?

2) Which management factors have the largest influence on arable plant diversity and how are they interacting with other management, environmental and spatial factors?

3) How important is the choice of crop for arable plant diversity in modern cropping systems and how does this factor interact with other management factors? What are the main differences in field management between the three studied crops?

2.2 Methods

2.2.1 Study areas

Two regions in Central Germany, the Lower Saxon hills (LS, 1118 km²) and the Thuringian Basin (TB, 1058 km²), were chosen as study regions (Fig. 2.1). The climate in the Thuringian Basin is more continental, i.e. drier and slightly warmer than in the Lower Saxon hills (mean annual precipitation: LS 650 mm, TB 500–550 mm; mean annual temperature: LS 8.7 °C, TB 9.2 °C). Both regions share a long history of arable farming (since the Neolithic) due to the presence of highly fertile loess soils.
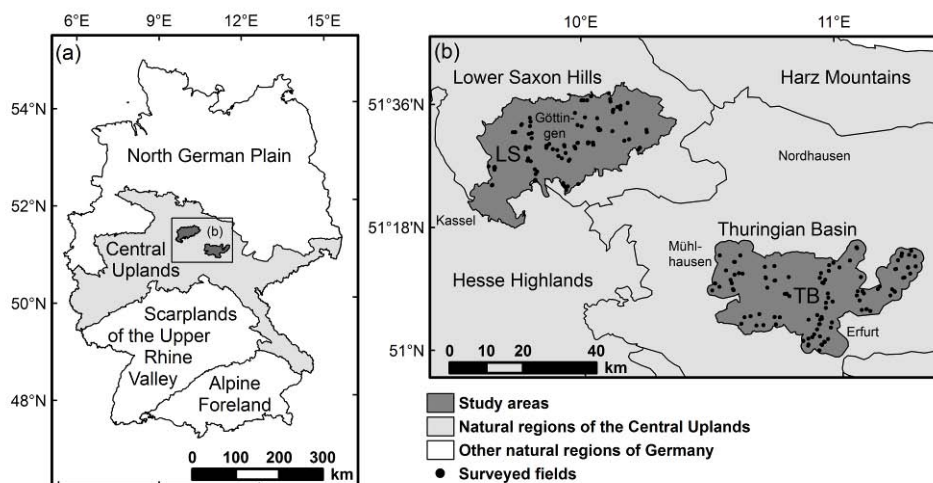
Fig. 2.1 Location (a) of the two study regions (LS = Lower Saxon hills, TB = Thuringian Basin) in Germany and (b) of the surveyed fields within the study regions (95 fields in LS, 111 fields in TB). The map is based on Universal Transverse Mercator (UTM) projection, zone 32 N (WGS 1984). Biogeographical boundaries were adopted from Meynen & Schmithüsen (1953-1962).

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Arable land represents 43% of the total area in the Lower Saxon hills and 78% of the area in the Thuringian Basin. Today's crop rotations in both study regions are dominated by the winter-sown crops wheat, barley and oilseed rape (approximately 2/3 of the arable land (LWA Leinefelde-Worbis, 2011; LWA Sömmerda, 2011; LWK Niedersachsen, 2010). Silage maize for fodder or energy use was cultivated on 6–7% of the arable land (LWA Leinefelde-Worbis, 2011; LWA Sömmerda, 2011; LWK Niedersachsen, 2010).

2.2.2 Sampling design

Twenty conventionally managed fields were surveyed per crop (maize, winter-sown wheat, winter-sown barley), sampling year (2011 or 2012) and study region, yielding a total of 240 surveyed fields. Farmers were selected with the aim to achieve full geographical coverage of the study regions, but were otherwise contacted at random without regard to farm size. Fields were randomly selected from the pool of fields offered by the participating farmers, with the condition of a minimum distance of 2 km between surveyed fields of the same crop within a sampling year. This high threshold ensures that direct dispersal events will only occur extremely rarely as most dispersal takes place within distances <200 m (Bergelson et al., 1993). On each field, two plots of 2 m × 50 m size were surveyed. One plot was located parallel to the field margin starting with the first sown row of crops. The second one was placed parallel to the first plot and at least 10 m away from the field. Vegetation surveys and soil sampling took place between 26 May and 8 August 2011 and 2012. Sampling started within the barley fields which are the first to be harvested, then gradually shifted towards wheat fields and finished with the maize fields which are harvested last.

2.2.3 Sampling of arable weed assemblages

Within each 100 m² sampling plot, all arable plant species and their respective covers were estimated using the Braun-Blanquet cover abundance scale consisting of nine classes (r, +, 1, 2m, 2a, 2b, 3, 4, 5) according to Wilmanns (1998). Additionally, total crop cover and total weed cover were estimated visually. A complete species list is given in appendix 2.A.

2.2.4 Soil sampling and analyses

Soil samples were taken at four random locations within each vegetation sampling plot from the uppermost 10 cm of the soil column after removing the litter using a soil corer of 5.1 cm diameter. The four samples per plot were thoroughly mixed and stored at 4 °C. The pH was measured according to HFA (2005). The base saturation ($[Ca^{2+}, Mg^{2+}, K^+]/[Ca^{2+}, Mg^{2+}, K^+, Al^{3+}, Mn^{2+}, Fe^{2+}, H^+]$, in %) was calculated from an analysis of the effective cation-exchange capacity via percolation with $BaCl_2$

solution (0.1 mol L^{-1}) and inductively coupled plasma (ICP) analysis of element concentrations in the percolate based on methods described in HFA (2005). The concentration of plant-available phosphorus was estimated with anion-exchange resin bags (Sibbesen, 1977). The amount of extracted P was measured colourimetrically with the molybdate/ascorbic acid method. The total amount of carbon (C_{tot}) and nitrogen (N_{tot}) and the C/N-ratio of the samples were determined with a C/N-elemental analyser (vario EL III, Elementar, Hanau, Germany). To detect the concentration of organic carbon (C_{org}), samples were heated to 450 °C to burn all organic carbon. Subsequently, the amount of the remaining inorganic carbon was measured and subtracted from the total carbon (C_{tot}) concentration. An estimate of the soil organic matter content (in %) was obtained by multiplying C_{org} with the factor 1.72.

2.2.5 Derivation of environmental, management and spatial variables

2.2.5.1   *Management variables*

A concise summary of all investigated management variables is given in Table 2.1. For each surveyed field, the participating farmers answered a standardised questionnaire (appendix 2.B). They were asked to provide information on land use history, on the crops and catch crops planted during the ten years prior to the survey, on the amount, timing and kind of fertilisation, herbicide applications and tillage operations during the year of survey (syr) and during the preceding year (pyr). Due to incomplete or inconsistently filled questionnaires, 34 fields had to be excluded from further analyses, resulting in a total of 206 sampling units (71 maize fields, 71 wheat fields and 64 barley fields).

Herbicide applications were found to involve a wide range of products and active ingredients. To reduce the complexity of this information, we derived the herbicide use intensity index (HI, eq. 2.1) as proposed by Roßberg et al. (2002). For each field, HI was obtained by summing up the products of the proportion of the field area sprayed (p_{area}) and the actually applied fraction of the maximum admitted application rate (p_{rate}) for each herbicide. The factor p_{area} was usually set to 1, as the treatments generally affected the entire sampling plots. In few cases, where only part of the field was sprayed, the factor p_{area} was used to minimise potential bias.

$$HI = \sum(p_{area} * p_{rate}) \qquad (eq. 2.1)$$

All herbicides were also classified by the sites of action of their active ingredients according to the classification system of the Herbicide Resistance Action Committee (HRAC) (Menne and Köcher, 2012; appendix 2.C). The amount of applied herbicide within each HRAC group (in kmol ha^{-1}) was determined for each field and summed up for the study year and the previous year.

Table 2.1 Description of predictors used for the analyses in the nine sets of variables. If correlations with R > 0.5 were found for a predictor, the correlated variables are given. Syr = year of survey, pyr = preceding year. Details on the categories of soil type, geological substrate and the HRAC classification are given in appendix 2.C.

Group	Set of variables	Variable	Description	Correlations with R > 0.5
Environmental factors	Macro environment	Region	Study region (LS or TB) [a]	Precip., sunsh. h.
		Elevation	Elevation [m a.s.l.]	Temp., precip.
		Precipitation	Mean precipitation during veget. period [mm]	Region, elev., temp., precip.
		Temperature	Mean temperature during veg. period [°C]	Elev., precip.
		Sunshine hours	Mean # of sunshine hours during veget. period [h]	Region, precip.
	Micro environment	Slope	Slope [°]	-
		Aspect	0 = (NW, NNW, N, NNE, NE) 0.5 = (ENE, E, ESE, WNW, W, WSW) 1 = (SW, SSW, S, SSW, SE)	-
		Crop cover	Total crop cover per plot [%]	-
		Tree/shrub cover	Total tree/shrub cover [%], only at field edge	-
	Soil	Geological substrate	16 categories	-
		Soil type	16 categories	-
		pH	pH (in H_2O)	-
		log(BS)	log(1-base saturation)	-
		C/N ratio	C/N ratio [g g^{-1}]	-
		SOM	Soil organic matter content [%]	-
		P	Plant-available P [μmol P g$_{(dry\ weight)}^{-1}$]	-
Management factors	Crop	Crop$_{syr/pyr}$	Current (syr) or preceding crop (pyr)	Sowing date
		Catch crop$_{syr/pyr}$	Binary (y/n): Catch crop prior to main crop?	-
		Sowing date	Sowing date given as # of weeks after August 1	Crop$_{syr}$
	Fertilisation	N, P, K, S, Mg (each for syr & pyr)	Amount of N, P, K, S, Mg fertilisation [kg*ha^{-1}]	K & P; K & Mg; Mg & log(Ca); in the resp. year
		log(Ca)$_{syr}$, log(Ca)$_{pyr}$	logarithm of Ca fertiliser amount [kg ha^{-1}]; (log(Ca+0.1))	Mg$_{syr}$, Mg$_{pyr}$
	Herbicides	HI$_{syr/pyr}$	Herbicide use intensity index (see Methods section)	# of mechanisms of action$_{syr/pyr}$
		# of mech. of action$_{syr/pyr}$	Number of mechanisms of action of herbicides used (see Methods section)	HI$_{syr/pyr}$
		A, B, C$_1$, C$_2$, C$_3$, E, F$_1$, F$_2$, F$_3$, F$_4$, G, K$_1$, K$_3$, N, O [b]	Amount of herbicide from the respective HRAC group [1000 mol ha^{-1}], sum of syr & pyr	C$_1$ with prop. of maize & winter-sown crops
	Conversion tillage	Ploughed last 2 yrs y/n	Binary (y/n): Deep ploughed during 2 years prior to survey?	-
		Ploughed last 5 yrs y/n	Binary (y/n): Deep ploughed during 5 years prior to survey?	-
	Crop rotation (10 years prior to survey)	# of crops	Number of different crops in crop rotation	-
		Prop. of maize/wheat/winter crops/legumes/fallow & grassland/biennial & perennial crops	Proportion of years with maize/winter-sown wheat/winter-sown crops/legumes/fallow & grassland/biennial & perennial crops [%]	Maize & wheat & winter crops & C$_1$; fallow + grassland & biennials + perennials
	Land use history	Years since conversion to cropland	4 categories: ≤ 10; > 10 & ≤ 30; > 30 & ≤ 50; > 50 years	-
		Previous land use	6 categories: permanent grassland, fallow land, vegetables, fruit trees, forest, unknown	-

a) See Fig. 2.1;
b) Categories of the HRAC herbicide classification system (Menne and Köcher, 2012, cp. appendix 2.C)

2.2.5.2   Environmental variables

The mean air temperature, the mean precipitation and the mean number of sunshine hours during the vegetation period (March – September) were obtained for each plot from spatially explicit data (resolution 1 × 1 km) based on long-term means for the period 1971-2000 (DWD, 2013). Elevation above sea level (a. s. l.) was derived from the digital elevation model for Germany (resolution: extent 25 × 25 m, height 0.1 m) (BKG, 2005). Variables characterising the local environment on the plots (slope, aspect, crop cover and tree/shrub cover; termed 'micro environment' hereafter) were determined directly in the field. In addition to the measured soil chemistry variables, soil and geological maps of the study areas (LS – BÜK50, GK50; Boess et al., 2004; LBEG, 2010; TB – 'Thuringian soil geological map'; Rau et al., 2000) were used to identify the dominant soil type and the geological substrate for each plot (appendix 2.C).

2.2.5.3   Spatial factors

The spatial variables were derived from the geographical coordinates (coordinate system UTM zone 32 N, WGS 1984) using principal coordinates of neighbour matrices (PCNM). These spatial variables are calculated as principal components of a truncated matrix of Euclidian distances between sampling sites (Borcard and Legendre, 2002). PCNM variables are able to fully represent all spatial relations within a given spatially explicit data set and can be used to correct for spatial autocorrelation (Borcard and Legendre, 2002). We used the *pcnm* function of the R-package vegan (R version 3.0.1) and applied a truncation threshold of 40 km. The threshold of 40 km was obtained as the longest distance in the minimum spanning tree of the matrix of Euclidian distances (rounded; Borcard and Legendre, 2002). After pre-selecting all spatial variables with $p \leq 0.2$ for each response variable, backward selection was applied to identify PCNM variables with $p \leq 0.05$ (R-package vegan, functions rda, ordistep) to be included in further analyses (Borcard & Legendre, 2002).

2.2.6 Statistical analyses

2.2.6.1   Model selection and variation partitioning

All statistical analyses were performed using R (versions 2.11.1 and 3.0.1; R Development Core Team, 2010). The crop itself, any woody species (including *Rubus fruticosus* agg.) and representatives of volunteer grain (*Brassica napus*, *Hordeum vulgare*, *Triticum aestivum*, *Secale cereale* and ×*Triticosecale*) were not included in the analyses of species richness and community composition. The environmental and management variables were allocated to one of nine subject-specific sets of variables (environmental variables: macro environment, micro environment, soil; management variables: crop, fertilisation, herbicides, conversion tillage using a mouldboard plough, crop rotation and land use history, Table 2.1). All variables were tested for collinearity (within and between sets). The variable "sowing date" was very strongly correlated to the crop type and was therefore excluded

from the analyses. Further correlations were found to occur only within certain sets of variables. If a large number of variables of a set was highly correlated (as occurred for macro environment, fertilisation and crop rotation), the axes of a principal components analysis (PCA) were used as variables instead. The first two axes (PC1 – increasing precipitation and decreasing temperature, PC2 – decreasing elevation, together explaining 96% of the total variation; appendix 2.D) were used for characterising the macro environment, and the first four axes for fertilisation (PC1 – decreasing application of organic fertilisers including N, P, K, Ca, Mg, PC2 – gradient of S fertilisation, PC3 – increasing Ca and Mg fertilisation and decreasing P input, PC4 – gradient of N fertilisation; together explaining 88%) and crop rotation (PC1 – intensity gradient: decreasing proportion of winter crops and increasing proportion of perennial crops and fallow, PC2 – decreasing proportion of maize, PC3 – decreasing number of crops, PC4 – decreasing proportion of legumes, explaining 92% of the variation; appendix 2.D).

Full models were derived for each set of variables based on theoretical reflections about potential mechanisms acting on arable plant diversity (appendix 2.D). Subsequently, a backward selection process (R-package vegan, function *ordistep*) was used and only variables which contributed significantly ($p \leq 0.05$) were retained in the final models (Table 2.2). (Partial) multiple linear regression models were calculated for the response variables total cover of arable non-crop plants ('weed cover') and species richness per plot (R-package vegan, function *rda*, Oksanen et al., 2013). All models were checked visually for normality of residuals and homogeneity of variances. The total weed cover was log-transformed before the analyses to avoid problems with heteroscedasticity. For quantifying the effects on community composition, we applied (partial) redundancy analyses (R-package vegan, function *rda*). Beforehand, a Hellinger transformation was applied to the matrix of species abundances per plot to avoid problems with long gradients resulting in many zeros in the data set (Legendre and Gallagher, 2001). In all models, the effects of sampling year and spatial autocorrelation (all significant PCNM variables) were partialled out to account for effects inherent to the sampling design.

The final models (Table 2.2) were used to obtain the 'gross' effect of each set of variables. 'Net' effects were calculated by modelling the explanatory power of each set of variables while partialling out the effects of all other significant variables from the reduced models (Legendre, 2008). We always reported adjusted R^2 (Peres-Neto et al., 2006). Variation that was explained by the model containing the significant variables from all sets but that could not be attributed to a single set of variables was called 'shared variation' (Legendre, 2008). By summing up the gross effects of the pertinent sets of variables, the maximum amount of variation explained by environmental, management or spatial factors was determined. A certain fraction of the total shared variation could be attributed to each of these three groups of factors by subtracting the maximum amount explained by the other two groups

of factors minus the net amount explained by each pertinent set of variables from the total amount of shared variation.

Table 2.2 Final models for each of the six response variables sorted by the nine sets of predictors. Variables from Table 2.1 were only retained in the model if they contributed significantly ($p \leq 0.05$). In all models, the year of survey and the significant PCNM variables were included as co-variables to correct for differences between sampling year and spatial autocorrelation.

	Final models after backward selection	
Weed cover	*Field interior*	*Field margin*
Crop	$crop_{syr}$	-
Fertilisation	$PC1_{syr}*PC1_{pyr} + PC2_{syr} + PC2_{pyr}$	$PC2_{syr} + PC2_{pyr}$
Herbicides	C3 + F2 + K1	-
Conversion tillage	-	-
Crop rotation	PC1	-
Land use history	time since conversion to cropland	-
Macro environment	-	-
Micro environment	-	-
Soil	soil type + SOM	pH
Spatial factors	6 significant PCNMs	15 significant PCNMs
Species richness		
Crop	$crop_{syr}$	$crop_{syr}$
Fertilisation	$PC1_{syr} + PC2_{syr} + PC2_{pyr} + PC3_{syr}*PC3_{pyr}$	$PC2_{pyr}$
Herbicides	C1 + K3	-
Conversion tillage	-	-
Crop rotation	-	-
Land use history	-	-
Macro environment	-	-
Micro environment	-	crop cover
Soil	-	geological substrate + pH + SOM
Spatial factors	10 significant PCNMs	4 significant PCNMs
Community composition		
Crop	$crop_{syr}$	$crop_{syr}$
Fertilisation	$PC1_{syr}+PC2_{syr}+PC3_{syr}$	$PC1_{syr} + PC2_{syr}$
Herbicides	C1 + G	B + C1
Conversion tillage	ploughing y/n (last 5 yrs)	-
Crop rotation	PC1 + PC2 + PC3	PC1 + PC2 + PC3
Land use history	time since conversion to cropland	-
Macro environment	PC2	PC2
Micro environment	-	slope*aspect + crop cover
Soil	soil type*geological substrate + P	soil type + geological substrate + pH + C/N ratio + SOM + P
Spatial factors	5 significant PCNMs	8 significant PCNMs

2.2.6.2 *Comparisons between crops*

The non-parametric Behrens-Fisher-Test for multiple comparisons (R-package npmc, function *npmc,* Helms and Munzel, 2009) was used to test for differences in arable plant cover, species richness and the explanatory variables between the three different crops.

2.2.6.3 *Analysing the influence of crop type*

For analysing the influence of the current crop on arable plant diversity in more detail, we conducted a separate analysis including only those sets of management variables which contained

variables showing significant differences between crops (fertilisation, herbicide use and crop rotation). Additionally, we added the variable sowing date, which had not been included in the previous analysis, to explore whether it was possible to discern which factor (current crop or sowing date) was more important. When interpreting the results the strong correlation between both variables was taken into account.

By combining the reduced models for each of these sets of variables, we obtained the maximum amount of variation explained by all potentially crop-related management parameters for all response variables. The gross and the net effects for each set of variables were calculated. We differentiated between the net effect of each set when partialling out all other variables (net$_{all}$) and the net effect when partialling out the effect of the crop (net$_{crop}$). We also calculated the net effect of the current crop while controlling for each of the other sets of variables one by one (crop$_{var}$).

2.3 Results

2.3.1 General importance of management factors for arable plant diversity

Arable plant diversity (weed cover, species richness and community composition) was found to be explained to a large extent by factors from all three factor groups (management, environmental and spatial factors). All factors together explained 32–43% of the total variation in weed cover, species richness and community composition (gross effects, R^2 = 0.32–0.43, $R^2_{adjusted}$ = 0.13–0.34) (Table 2.3). Field management factors were found to be important explanatory variables for the arable plant cover, the species richness per plot and the community composition (Fig. 2.2), accounting for 6–48% of the total explained variation. At the field margins, the explanatory power of the management factors was by 49–86% smaller than in the field interior for the three response variables. Field management was more important than environmental factors when explaining variation in species richness (field interior) and in weed cover (field interior and margin), but it was less important for species richness at the field margin and for community composition in the interior and at the margin. Fertilisation was the only factor which showed a significant net effect on weed cover both at the field margin and in the field interior after correcting for spatial autocorrelation (Table 2.3).

The explanatory factors from different sets of variables interacted strongly and the net effects were usually much smaller than the gross effects. The proportion of explained variation, which could not be attributed to a certain group of variables (management, environment or spatial factors), varied between 2 and 25% of the total explained variation, while the share of variation not attributable to a certain set of variables within these groups varied between 0–30%.

Table 2.3 Gross and net effects ($R^2_{adjusted}$) of different sets of variables on weed cover, species richness and community composition in 206 fields in Central Germany. Note that only the significant variables from the final models (Table 2.2) have entered the analyses presented here. The number and kind of variables included in each set may differ between response variables. Please refer to table 2.2 for interpretation. For further details on the calculation see Methods section; ns $p > 0.05$; * $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$.

	Weed cover				Species richness				Community composition			
	Field interior		Field margin		Field interior		Field margin		Field interior		Field margin	
	gross	*net*	gross	*net*	gross	*net*	gross	*net*	gross	*net*	gross	*net*
Total	30.8**	-	34.3**	-	32.9**	-	23.7**	-	14.2**	-	12.7**	-
Crop	8.8**	*ns*	ns	*ns*	9.5**	*2.0*	3.9**	*2.8*	0.6**	*ns*	0.9**	*0.3**
Fertilisation	12.9**	*5.5**	2.5**	*2.1*	9.2**	*ns*	4.4**	*3.7*	0.7**	*ns*	0.5**	*ns*
Herbicides	5.9**	*ns*	ns	*ns*	4.9**	*2.2*	ns	*ns*	0.8**	*ns*	0.7**	*0.3*
Conversion tillage	ns	*ns*	ns	*ns*	ns	*ns*	ns	*ns*	0.7**	*0.2*	ns	*ns*
Crop rotation	3.3**	*ns*	ns	*ns*	ns	*ns*	ns	*ns*	1.4**	*ns*	1.1**	*0.5*
Land use history	6.9**	*ns*	ns	*ns*	ns	*ns*	ns	*ns*	2.3*	*1.4*	ns	*ns*
Total management	19.9**	-	2.5**	-	13.3**	-	8.9**	-	3.1**	-	1.9**	-
Macro environment	ns	*ns*	ns	*ns*	ns	*ns*	ns	*ns*	0.2*	*ns*	0.3**	*ns*
Micro environment	ns	*ns*	ns	*ns*	ns	*ns*	2.3**	*ns*	ns	*ns*	1.1**	*ns*
Soil	5.9*	*ns*	1.2*	*ns*	ns	*ns*	8.5**	*6.4*	8.5**	*7.0*	6.9**	*5.6**
Total environment	5.9*	-	1.2*	-	ns	-	10.6**	-	8.5**	-	7.2**	-
Spatial factors	10.1**	*11.1**	31.2**	*30.8***	20.1**	*19.1**	7.3*	*3.5*	2.4**	*1.7*	3.9**	*2.8**
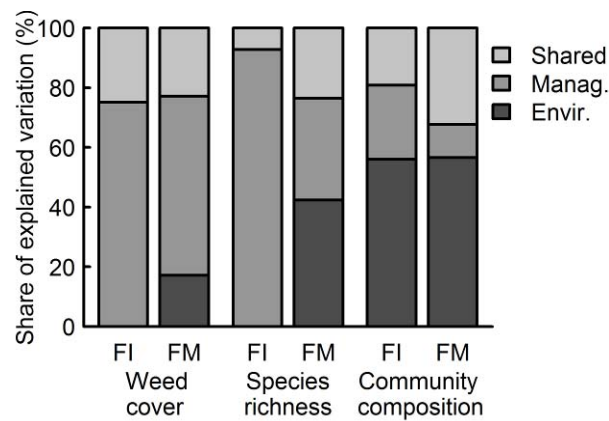
Fig. 2.2 Proportion of explained variation in arable plant cover, species richness and community composition attributable to field management and environmental factors in the interior (FI) and at the margin (FM) of 206 conventionally managed fields in Central Germany.

2.3.2 Effects of specific management factors on arable plant diversity and interactions with environmental and spatial factors

Within the group of management factors, current crop was identified as a significant explanatory factor for all response variables except weed cover at the field margin (Table 2.2). Neither the preceding crop nor the planting of catch crops were found to be significant factors for any of the response variables (Table 2.2).

In contrast, fertilisation during the year of survey and/or the preceding year was retained in the final models for all response variables (Table 2.2). We found both the amount of N, P, K, Ca and Mg (PC1) and S fertilisation (PC2) to be important explanatory factors (Table 2.2). Interestingly, the effect of S fertilisation persisted independently of the effects of other predictors and was found to be more important for weed cover and species richness than the other fertilisers (Table 2.2 & 2.3, Fig. 2.3, 2.4a). S fertilisation in the year preceding the vegetation surveys was found to be positively correlated to weed cover and species richness (Fig. 2.3 & 2.4a). Even the relationship between weed cover and crop cover was found to be mediated by S fertilisation. Surprisingly, crop cover was just about not significantly correlated to weed cover accounting only for the effects of crop type, spatial autocorrelation and sampling year ($y = -1.64$ x; $R^2 = 0.02$; $p = 0.053$). Only after additionally partialling out the effect of S fertilisation in the preceding year the relationship became significant as expected ($y = -1.83$ x; $R^2 = 0.02$; $p = 0.033$).
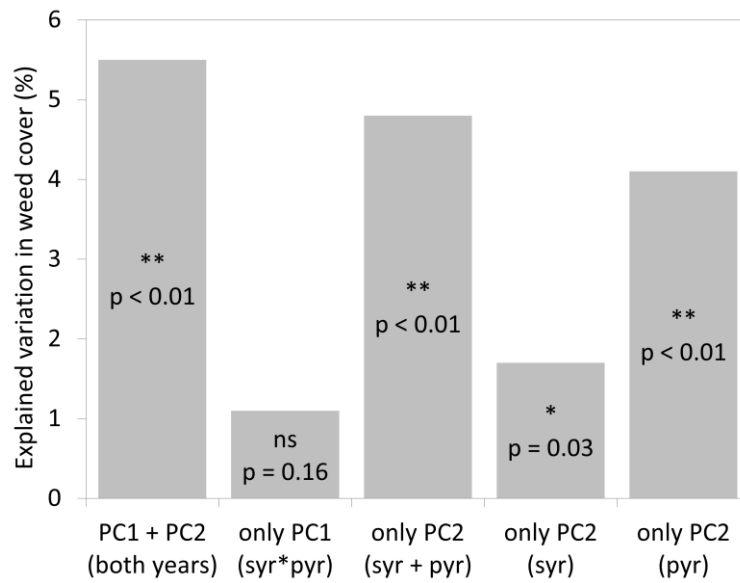
Fig. 2.3 Proportion of variance in weed cover in the field interior explained by fertilisation in the sampling year (syr) and in the preceding year (pyr). Fertilisation effects are represented as principal components (appendix 2.D). The bars show net effects after partialling out the effect of crop, sampling year and spatial autocorrelation. Only combinations of factors which were retained in the final models (Table 2.2) are presented here.
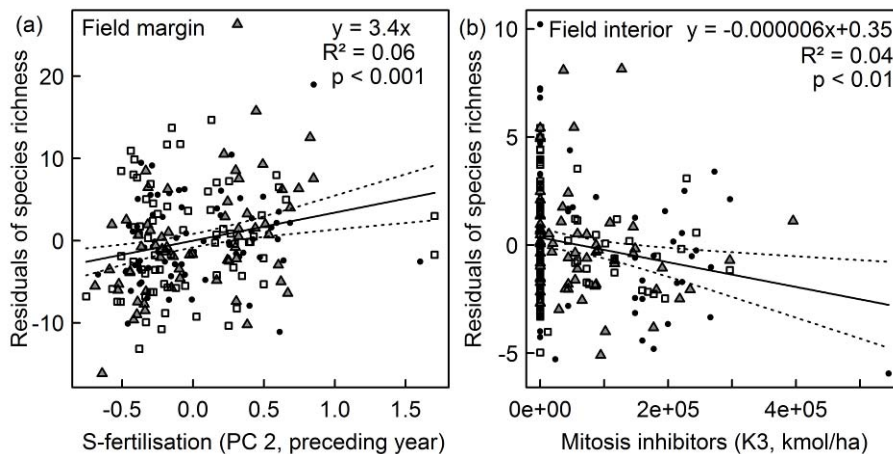
Fig. 2.4 Relationship between (a) species richness at the field margin and S-fertilisation during the preceding year and (b) species richness in the field interior and the amount of herbicides from class K_3 (mitosis inhibitors) after partialling out the effects of crop, sampling year and spatial autocorrelation. The solid line represents the linear regression line while the dotted lines are the 95% confidence intervals. Black circles = maize, white squares = winter-sown wheat, grey triangles = winter-sown barley.

Certain groups of herbicides (B, C_1, C_3, F_2, G, K_1 and K_3, cp. Table 2.2 & 2.3, Fig. 2.4b) also explained part of the variation for all response variables. The effect on weed cover and species richness was, however, restricted to the field interior. A significant net effect of the use of inhibitors of branched chain amino acid synthesis (B), photosynthesis inhibitors at PS II (C_1) and mitosis inhibitors (K_3) was found for species richness in the field interior and community composition at the field margin. Interestingly, the herbicide use intensity index (HI) did not explain any significant fraction neither in the field interior nor at the field margin.

The importance of conversion tillage, crop rotation and land use history was less consistent across the response variables (Table 2.2 & 2.3). Crop rotation was found to influence weed cover and community composition, whereas it could not explain species richness. Community composition was related to the first three principal components of crop rotation, indicating that all included variables were potentially important. A net effect of crop rotation on community composition was detected at the field margin (0.5%).

Among the environmental factors, the soil chemical variables were found to be potentially most important. Significant net effects of soil parameters were detected for species richness at the field margin (6.4%) and for community composition both at the field margin (5.6%) and in the field interior (7.0%).

2.3.3 Importance of the current crop and interactions with other management factors

Margins of maize fields were with on average 11 weed species 100 m^{-2} less species-rich than margins of winter wheat fields with 15 species 100 m^{-2} (Fig. 2.5). For the field interior, where species richness was generally extremely low, the opposite picture was found. Winter-sown cereals were with 3 species 100 m^{-2} even species-poorer than maize fields with 6 species 100 m^{-2}.

The current crop potentially explained a substantial amount of the variation in weed cover, species richness and community composition (Table 2.4). The crop effects were, however, restricted to the field interior and were interrelated with sowing date, fertilisation, herbicide use and crop rotation. Maize fields received about 2–5 times more organic fertilisers (on average 23.6 Mg ha^{-1}yr^{-1}, including animal manures and digested residues from biogas plants) than winter cereal fields. As a consequence, the added doses of P, K, Ca and Mg were 2–6 times higher in maize fields (Table 2.5). N input was not higher on maize fields as compared to winter cereal fields, as higher N-inputs from organic fertilisers were balanced with lower mineral N fertilisation. Herbicide use intensity was found to be higher in maize and winter-sown wheat than in winter-sown barley (Table 2.5). The types of herbicides applied also differed substantially between crops.
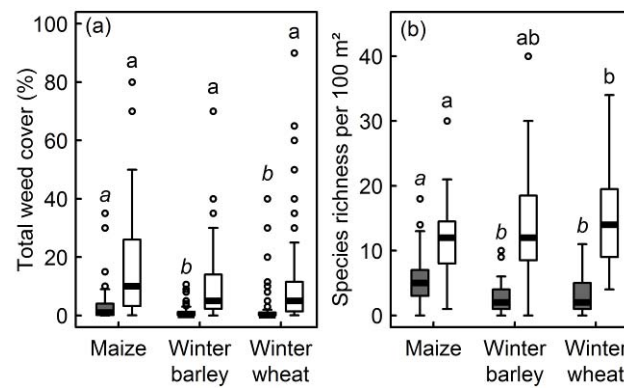
Fig. 2.5 Differences in (a) total weed cover (in %) and (b) species richness per sampling plot between fields cultivated with maize (n=71), winter barley (n=64) and winter wheat (n=71). Plots from the field interior (grey bars, italic letters) and from the field margin (white bars, regular letters) were analysed separately. Different letters indicate significant differences between crops at $p \leq 0.05$.

No net effect of the crop was detectable after controlling for the effects of the other management factors. The sowing date and the crop were so tightly correlated, that neither factor explained any significant part of the variation when the other was controlled for, although taken individually both showed high explanatory power. The complementarity of the effects of the crop with fertilisation, herbicides and crop rotation was less strong. Although controlling for crop usually reduced the amount of variation explained by these factors, the remaining fractions were in many cases still significant (for details see Table 2.4). Likewise, the crop usually still explained significant parts of the variation when corrected for the effects of fertilisation, herbicides or crop rotation.

2.4 Discussion

2.4.1 General importance of management factors for arable plant diversity

The total amount of variation explained by the complete reduced models was high in our study as compared to previous studies with similar goals (Fried et al., 2008; Pinke et al., 2012). For community composition, we found the complete set of variables to explain 42% (R^2) in the field interior (20 variables, $R^2_{adj} = 14\%$) and 36% at the field margin (26 variables, $R^2_{adj} = 13\%$) including management, environmental and spatial predictors. Pinke et al. (2012) were able to explain 25% (R^2) of the variation in community composition of 243 arable fields in Hungary with 24 variables describing the field management, environmental factors and the site context. Fried et al. (2008) explained 35% (proportion of total inertia from CCA) of the variation in community composition from 694 fields throughout France with 14 management and environmental variables. The amount of explained variation was similarly high for weed cover (field interior: $R^2 = 43\%$ / margin: $R^2 = 40\%$) and species richness (40% / 32%). We are not aware of other similarly comprehensive investigations

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for these two response variables in conventionally managed farmland. The complex interactions of different predictors in agroecosystems and the noise which is necessarily generated in the data collection process define the limits of current statistical approaches for this type of assessments.

Our results confirm the general importance of field management factors for arable plant diversity, reaffirming that human influence strongly shapes arable plant assemblages. We found weed cover, species richness and community composition to respond to different sets of predictors, confirming previous results by Fried et al. (2008). Management factors were able to explain large proportions of the variations in weed cover and species richness in the field interior, whereas environmental factors did not contribute significantly. In contrast and in accordance with Pinke et al. (2012), our data show management factors to be less important than environmental factors for explaining community composition. This confirms our hypothesis that environmental factors acting as barriers to the dispersal and establishment of new species remain important for shaping community composition even in landscapes highly influenced by human management.

Table 2.4 Gross and net effects ($R^2_{adjusted}$) of management factors found to be interacting with the cultivated crop. *Net$_{all}$* gives the net effect after controlling for all other management factors, *net$_{crop}$* gives the effect after controlling for the effects of crop, and *crop$_{var}$* gives the net effect of crop after controlling for the respective variable. Note that only the significant variables from the final models (Table 2.2) have entered the analyses presented here. The number and kind of variables included in each set may differ between response variables. Please refer to table 2.2 for interpretation. For further details on the calculation see Methods section; ns $p > 0.05$; * $p \leq 0.05$; ** $p \leq 0.01$; *** $p \leq 0.001$.

Weed cover	Field interior				Field margin			
	gross	net$_{all}$	net$_{crop}$	crop$_{var}$	gross	net$_{all}$	net$_{crop}$	crop$_{var}$
Crop	8.8**	0			0	-		
Sowing date	9.7**	0	0	0	0	-	-	-
Fertilisation	12.9**	8.0**	5.1**	0	2.5**	2.5**	1.8*	-
Herbicides	5.9**	0	0	3.5*	0	-	-	-
Crop rotation	3.3**	0	0	6.1**	0	-	-	-
Total explained var.	17.4**				2.5**			
Species richness								
Crop	9.5**	0			3.9**	0		
Sowing date	10.0**	0	0	0	3.1**	0	0	0
Fertilisation	9.2**	0	0	1.8*	4.4**	4.5**	5.0**	4.5**
Herbicides	4.9**	2.1*	2.6**	7.2**	0	-	-	-
Crop rotation	0	-	-	-	0	-	-	-
Total explained var.	13.1**				10.8***			
Community composition								
Crop	0.6**	0			0.9**	0		
Sowing date	0.7**	0	0	0	0.8**	0	0	0.1*
Fertilisation	0.7**	0	0	0.2*	0.5**	0	0	0.4***
Herbicides	0.8**	0	0.2*	0.2*	0.7**	0.2*	0.3**	0.5***
Crop rotation	1.4**	1.2***	1.2**	0.4***	1.1**	0.6*	0.7***	0.6***
Total explained var.	2.4***				1.9***			

Table 2.5 Differences in macro- and micro-environmental factors, soil chemistry and field management practices (fertilisation, herbicide use and crop rotation) between stands of conventionally managed maize, winter-sown wheat and winter-sown barley. Means and standard deviation (±SE) are presented. Different letters indicate significant differences between groups (separate analyses for the field interior and the field margin, non-parametric Behrens-Fisher-test for multiple comparisons, $P \leq 0.05$). No differences in fertiliser input or herbicide applications between the three crops were found for the year preceding the study. Results are therefore not shown here. HRAC = Herbicide classification system from the Herbicide Resistance Action Committee. Syr = year of survey, pyr = preceding year.

	Field interior			Field margin (only if different from interior)		
	Maize	**Wheat**	**Barley**	**Maize**	**Wheat**	**Barley**
Sample size	71	71	64	71	71	64
Sowing date (days)	22 Apr±4	7 Oct±10	22 Sep±5	22 Apr±4	7 Oct±10	22 Sep±5
Macro environment						
Elevation (m a. s. l.)	215 ± 5^a	234 ± 7^a	222 ± 6^a			
Precipitation (Mar.–Sept.)	386 ± 4^a	394 ± 5^a	383 ± 5^a			
Temperature (Mar.–Sept.)	12.6 ± 0.1^a	12.4 ± 0.1^a	12.6 ± 0.1^a			
Sunshine hours (Mar.–Sept.)	1160 ± 2^a	1156 ± 2^a	1160 ± 2^a			
Micro environment (field interior)						
Slope (°)	1.1 ± 0.1^a	$\mathbf{1.8\pm0.2^b}$	$\mathbf{1.8\pm0.2^b}$	1.2 ± 0.2^a	1.7 ± 0.2^a	1.8 ± 0.2^a
Crop cover (%)	$\mathbf{66.2\pm2.4^a}$	57.7 ± 1.5^b	66.0 ± 1.6^a	$\mathbf{49.0\pm2.0^a}$	39.7 ± 1.4^b	$\mathbf{46.1\pm1.7^a}$
Tree/shrub cover (%)	0	0	0	0.9 ± 0.3^a	2.5 ± 1.2^a	1.1 ± 0.3^a
Soil chemistry (field interior)						
pH	7.1 ± 0.1^a	7.0 ± 0.1^a	7.1 ± 0.1^a	7.2 ± 0.1^a	7.2 ± 0.1^a	7.2 ± 0.1^a
Base saturation (%)	99.9 ± 0.1^a	99.5 ± 0.2^a	99.7 ± 0.1^a	99.8 ± 0.1^a	99.6 ± 0.2^a	99.8 ± 0.1^a
C/N-ratio	9.2 ± 0.1^a	$\mathbf{9.4\pm0.2^{ab}}$	$\mathbf{9.6\pm0.1^b}$	9.3 ± 0.1^a	9.7 ± 0.2^a	9.7 ± 0.2^a
SOM (%)	3.6 ± 0.1^a	3.5 ± 0.1^a	3.8 ± 0.1^a	3.6 ± 0.1^a	3.6 ± 0.1^a	3.8 ± 0.1^a
Plant-available P (µmol P * $g_{(dry\ weight)}^{-1}$)	2.8 ± 0.2^a	2.3 ± 0.1^a	2.0 ± 0.1^a	$\mathbf{2.7\pm0.2^a}$	$\mathbf{2.2\pm0.1^{ab}}$	1.9 ± 0.2^b
Fertilisation (year of survey)						
N-fertilisation (kg/ha)	$\mathbf{183\pm6^{ab}}$	$\mathbf{206\pm5^a}$	170 ± 5^b			
P-fertilisation (kg/ha)	$\mathbf{96\pm5^a}$	30 ± 4^b	22 ± 3^b			
K-fertilisation (kg/ha)	$\mathbf{117\pm8^a}$	57 ± 8^b	26 ± 4^c			
S-fertilisation (kg/ha)	12 ± 1^a	$\mathbf{26\pm2^b}$	$\mathbf{25\pm2^b}$			
Ca-fertilisation (kg/ha)	$\mathbf{221\pm60^a}$	38 ± 14^b	36 ± 13^b			
Mg-fertilisation (kg/ha)	$\mathbf{36\pm5^a}$	18 ± 2^b	12 ± 1^c			
Organic fertilisers (1000 kg/ha)	$\mathbf{23.6\pm1.9^a}$	10.0 ± 1.3^b	4.2 ± 0.8^c			

Table 2.5 (continued)

	Field interior			Field margin (only if different from interior)		
	Maize	**Wheat**	**Barley**	**Maize**	**Wheat**	**Barley**
Herbicide use						
Herbicide use intensity index (syr)	**1.7±0.1[a]**	**1.9±0.1[a]**	1.4±0.1[b]			
# of mechanisms of action (syr)	**3.5±0.1[a]**	3.0±0.1[B]	**3.1±0.1[ab]**			
A (ACCase inhibitors, kmol/ha)	3.2±0.8[a]	**10.7±1.8[b]**	**12.5±1.9[b]**			
B (ALS/AHAS inhib., kmol/ha)	**23.3±1.2[a]**	16.1±1.2[b]	10.4±1.4[c]			
C_1 (PS-II inhib., kmol/ha)	**107.8±4.0[a]**	34.1±4.4[b]	8.0±2.3[c]			
C_2 (PS-II inhib., kmol/ha)	12.9±4.4[a]	**37.5±9.7[ab]**	**57.0±9.5[b]**			
C_3 (PS-II inhib., kmol/ha)	**34.8±3.6[a]**	10.3±2.5[b]	10.1±2.7[b]			
E (Protoporphyrinogen oxidase inhib., kmol/ha)	0.9±0.2[a]	2.0±0.4[a]	7.6±2.6[a]			
F_1 (Carotenoid biosynthesis inhib., kmol/ha)	7.7±1.9[a]	14.4±2.2[b]	**31.3±3.1[c]**			
F_2 (Carotenoid biosynthesis inhib., kmol/ha)	**12.9±1.5[a]**	5.3±1.1[b]	0.5±0.3[c]			
F_3 (Carotenoid biosynthesis inhib., kmol/ha)	1.6±0.5[a]	**8.9±7.1[b]**	**0.2±0.1[b]**			
F_4 (Carotenoid biosynthesis inhib., kmol/ha)	0.0±0.0[a]	0.3±0.2[a]	0.8±0.4[a]			
G (EPSP synthase inhib. =glyphosate, kmol/ha)	106.7±13.1[a]	89.5±11.2[a]	74.2±10.0[a]			
K_1 (Inhib. of microtubule assembly, kmol/ha)	11.9±5.4[a]	25.5±8.6[a]	35.2±8.8[A]			
K_3 (Mitosis inhib. kmol/ha)	**75.8±10.6[ab]**	43.9±7.0[a]	**64.4±7.6[b]**			
N (Inhib. of lipid synthesis, kmol/ha)	9.9±5.6[a]	41.9±12.7[a]	15.7±8.5[a]			
O (Synthetic auxins, kmol/ha)	26.0±4.4[a]	**71.5±7.6[b]**	**54.9±6.7[b]**			
Crop rotation (last 10 yrs. before study)						
# of crops	4.1±0.1[a]	4.0±0.1[a]	4.3±0.1[a]			
Prop. of maize (%)	**27±2[a]**	10±1[b]	9±1[b]			
Prop. of winter wheat (%)	37±1[a]	**51±1[b]**	37±1[a]			
Prop. of winter-sown crop (%)	58±2[a]	**76±2[b]**	**77±2[b]**			
Prop. of legumes (%)	2±1[a]	3±1[a]	2±1[a]			
Prop. of fallow period/grassland (%)	3±1[a]	0±0[a]	1±1[a]			
Prop. of biennial/perennial crops (%)	4±1[a]	0±0[a]	3±1[a]			

With on average 3–6 species 100 m^{-2}, the interior of conventionally managed maize and winter cereal fields was found to be extremely species-poor. This is in accordance with Meyer et al. (2013) who found a median of six species per plot on loamy soils in Germany. More than half of the recorded arable plants in the field interior belonged to one of only nine very common weedy species (*Fallopia convolvulus*, *Galium aparine*, *Polygonum aviculare*, *Chenopodium album*, *Viola arvensis*, *Convolvolus arvensis*, *Alopecurus myosuroides*, *Cirsium arvense* and *Bromus sterilis*, in decreasing order of importance). On extensively managed, low-input arable land, much higher species richness and abundances have been recorded in previous studies (e.g. 21–35 species 100 m^{-2}, Albrecht et al., 2009; Seifert et al., 2014; cp. also Kleijn and van der Voort, 1997; Walker et al. 2007). This extremely low diversity combined with very low arable plant cover (67% of all fields had a cover value $\leq 1\%$) indicates that modern high-input cropland must represent an extremely impoverished habitat for insects, birds and mammals. Most species, including many pest predators, depend upon more heterogeneous habitats in terms of floristic diversity.

As expected, the explanatory power of field management factors was higher in the field interior than at the field margins. Field margins tend to receive less fertiliser and herbicides (Wilson and Aebischer, 1995) and crop yields are usually lower (Sparkes et al., 1998). Being ecotones, field margins are also influenced by a wider range of factors than the interior of arable fields, including inputs from the adjacent habitats (Marshall and Arnolds, 1995). These effects lead to strongly increased cover values and species numbers at the field margin (Fig. 2.5). Correspondingly, we found the influence of environmental factors on arable plant diversity to be higher at the field margin than in the field interior, possibly due to the release of the arable plant assemblages from chemical stressors and competition with the crop (Pinke et al., 2012).

In this study, we tested for the effects of fertiliser and herbicide applications on arable plant diversity at the field margins, assuming a linear relationship with the application rates in the field interior. This was a necessary simplification. More exactly, the difference in management intensity between the margin and the interior is expected to vary depending on the site conditions, the machinery used, and the weather conditions on the day of application and ultimately even on the farmer's mood and preferences (Wilson and Aebischer, 1995). As a result, the reduced explanatory power at the field margin might partly be due to less accurate data on fertiliser and herbicide input.

2.4.2 Effects of particular management factors on arable plant diversity and interactions with environmental and spatial factors

The current crop, fertilisation and to a lesser extent also herbicide use as well as soil characteristics were found to be the most important predictors of arable plant diversity in Central European high-input croplands, apart from spatial factors (Fried et al, 2008; Lososová et al., 2004;

Pinke et al., 2012). Due to many inter-relations between field management factors, it seems impossible to single out any particular factor as most important for reducing or enhancing arable plant diversity.

Fertilisation was found to be a promising target as it was shown to be related to all three response variables. Contrary to results from previous studies (Pyšek and Lepš, 1991; Storkey et al., 2012), N fertilisation was not singled out as the most important explanatory factor. The data reveal that rates of N, P, K and Mg fertilisation were too highly correlated to allow for their statistical separation via the approach we applied and were therefore all subsumed in one principal component. Although we found significant gross effects of PC1 in the field interior, no net effects of this group of fertilisers persisted. This was due to the fact that the highest inputs of these nutrients resulted from organic fertilisers which contain all four elements and are applied at highest rates in maize fields and much more rarely in barley fields (Table 2.5). We could therefore not separate the effects of fertilisation from the overall crop effect. The effects of herbicide applications might also have overridden the fertilisation effect.

S fertilisation represented a second gradient (PC2) and was statistically independent from the other types of fertilisers. S fertilisers were applied mainly in the form of ammonium sulphate or ammonium nitrate sulphate. The fact that sulphur fertilisation during the year preceding the survey was positively correlated to species richness (Fig. 2.4a) and weed cover (Fig. 2.3) is interesting. As proposed by Smith et al. (2009), sulphur fertilisation potentially increases resource diversity in the soil and consequently reduces the competition between weeds and crops. Fields with S fertilisation might therefore support a higher weed cover without decreases in crop yields (Grant et al., 2007; Smith et al., 2009). The Resource Pool Diversity Hypothesis (Smith et al., 2009) was also supported by the fact that the negative relationship between crop cover and weed cover became significant only after partialling out the confounding effect of S fertilisation. The diversification of soil resource pools might therefore indeed represent an opportunity for enhancing arable plant diversity without compromising crop yields. These conclusions must remain highly tentative and need further experimental verification. In the light of extremely low weed cover values in the field interior, it seems, however, unlikely that the magnitude of such effects is large enough to positively influence species of higher trophic levels or to halt the on-going decline in arable plant diversity (Meyer et al., 2013; Storkey et al., 2012).

Contrary to our hypothesis the herbicide use intensity index (HI) was not found to explain any of the plant diversity metrics. Several aspects might contribute to this unexpected lack of fit: (1) The maximum admitted application rate, which is an important parameter for calculating HI, is determined by governmental bodies based on maximum residue levels in the final products (EC, 1991-2013). It does not necessarily imply any comparability between herbicides in terms of effectiveness for eradicating weeds. The HI might thus not be a suitable proxy for ecological assessments of herbicide pressure on weed assemblages. (2) We would expect a certain threshold value above which any further

increase in herbicide dosage will not lead to further decrease in weed diversity. This threshold might have been exceeded on the fields with extremely low weed cover, i.e. potentially on 2/3 of all assessed fields. Our study highlights the importance of the development of a more appropriate herbicide intensity index for assessing herbicide impacts in agro-ecosystems. The strong inter-relationships we found between different sets of explanatory factors show that the development of such an index and the determination of any potential threshold value can only be realised based on experimental assessments and controlling for confounding factors. Such an index would be a useful tool for determining the external economic impacts, i.e. the "cost of biodiversity loss" (TEEB, 2010), of herbicide applications on a field by field basis.

We found the approach to use the number of molecules of the active ingredients of herbicides to be more promising. Structuring the herbicide dosage data by HRAC groups for investigating differences in the impacts of particular modes of action yielded interesting new insights. We found the amount of mitosis inhibitors (K_3) to exhibit significant net negative effects on species richness in the field interior after partialling out the effects of crop, sampling year and spatial autocorrelation (Fig. 2.4b). Common mitosis inhibitors applied on the surveyed fields were flufenacet and pethoxamid. We did not find similarly large effects for the other groups of herbicides. These differential effects between HRAC groups might be due to several reasons. (1) The degree of selectivity of target weeds and crops might vary between HRAC classes, e.g. the different sulfonylureas from group B are commonly targeted at a certain crop and certain weed species (Drobny et al, 2012), whereas the mitosis inhibitors from group K_3 affect a broader set of species and crops (Böger, 2003; Busi, 2014). K_3 herbicides are therefore more likely to exhibit a dose-dependent "group effect" as postulated in our initial hypothesis. (2) The effect of herbicide groups which target mainly one of the three assessed crops, such as C_1 herbicides which are only applied in maize (Table 2.5), can statistically not be separated from the overall effect of the crop. As K_3 herbicides are applied across all three crops, they have higher potential to show significant net effects.

Among management factors crop rotation, conversion tillage with a mouldboard plough and land use history were found to be particularly important for explaining community composition. Our initial hypotheses were thus largely confirmed. This leads to the conclusion that dynamic mosaic landscapes including a wide range of different annual and perennial crops, fields with and without conversion tillage and a range of land uses other than crop production will in the long term secure highest levels of arable plant diversity (see also Gabriel et al. 2006). Contrary to our initial hypothesis and to data from literature (Ball, 1992; Teasdale et al., 1991), mouldboard ploughing was not found to decrease weed cover. The effect might have been overridden by the stronger effects of herbicides and fertilisation.

The fact that soil parameters were found to play a prominent role in explaining community composition highlights the importance to maintain landscapes with diverse soil conditions. In the light of increasing homogenisation of habitat conditions on arable land through eutrophication and increasingly larger agricultural units (Smart et al., 2006), it is vital to conserve mesophilous and nutrient-poor habitats for maintaining biodiversity (Storkey et al., 2012).

2.4.3 Importance of the current crop and interactions with other management factors

Our results confirm that the current crop is a sound proxy of management techniques. In particular, it is the only easily derived predictor which subsumes aspects of crop phenology and timing of management practices. More detailed data on fertilisation, herbicide applications and crop rotation will, however, add valuable additional information and should wherever possible be incorporated.

The analysis of the interrelationship between crop type and management factors confirmed that the management of the winter-sown cereals wheat and barley diverged in many aspects from the summer-sown crop maize. Overall N fertilisation was not found to differ between crops, but maize fields received the highest doses of organic fertiliser leading to extremely high P inputs. Doses of >100 kg P_2O_5 ha^{-1} yr^{-1}, as recorded for almost 50% of the maize fields, are likely to exceed the nutrient removal with maize harvest (LWK NRW, 2012) and can lead to substantial P leaching to the groundwater, particularly in maize-dominated crop rotations. Herbicide applications in maize fields differ compared to winter cereal fields in timing of applications and types of the herbicides. Systematic differences in overall quantity of herbicide use were not identified and the herbicide use intensity index was equally high in maize fields and in winter wheat fields. The HI values in the study region were found to be comparable to values found by comprehensive assessments across Germany (Roßberg, 2011). Differences in timing of herbicide treatments are difficult to capture and were represented in this study by the variable "current crop" and "sowing date". Both variables were found to be highly significant. Due to the strong correlation between both, it was, however, not possible to distinguish between the effects of timing, type and size of the differences in management practices represented by these two variables. New approaches for more adequately incorporating temporal aspects of field management in scientific impact assessments can potentially help to increase the explanatory power of the respective models.

2.5 Conclusions

The maintenance of tolerable levels of arable plants in conventional crop systems is a key requirement to allow for the protection of a basic level of biodiversity in agricultural landscapes, including pollinators and pest-predators. We found some indication that further research with regard to

the Resource Pool Diversity Hypothesis (Smith et al., 2009) and more particularly concerning S fertilisation might yield opportunities for increasing arable plant diversity without compromising yields. The expected magnitude of the benefits of this approach is, however, unlikely to be large enough to halt the on-going decline in farmland biodiversity. The results we present in this study show clearly that the necessary increase in arable plant diversity to restore functional farmland food webs cannot be achieved by simply altering certain management factors, such as fertilisation or herbicide use, within the range of current conventional farming practices. Well researched approaches such as sparing field margins from fertilisation and herbicide treatments (Kleijn and van der Voort, 1997; Walker et al. 2007) should be favoured as viable options as they balance biodiversity benefits and production goals. In line with Garnett et al. (2013) it seems necessary to rethink the current farming approaches more radically; inventing new technologies and adapting farming practices to specific regional requirements should be options. In the meanwhile, it is urgently needed to conserve appropriate habitats as sanctuaries for the arable flora in regions such as Germany, where this group of species is suffering from rapid decline (Meyer et al., 2013; Storkey et al., 2012). In Germany, this type of high-nature-value farmland has dropped to extremely low levels due to financial unattractiveness of the respective agri-environmental schemes in the face of strong support for energy crop production and high commodity prices for agricultural products. The fact that extensively managed arable land will also not be eligible as 'ecological focus area' in Germany under the new EU common agricultural policy highlights the necessity to improve the financial support of the pertinent schemes in the framework of the EU rural development programmes.

2.6 Acknowledgements

2.7   References

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Chapter 3

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Seifert, C., Leuschner, C., Meyer, S. & Culmsee, H. (2014)

Inter-relationships between crop type, management intensity and light transmissivity in annual crop systems and their effect on farmland plant diversity

Abstract

The recent boost of energy cropping in Central and Western Europe has greatly increased the demand for farmland leading to rapid land use change in many cultural landscapes. First-generation energy crops are now cultivated at more than 15% of Germany's arable land, but the consequences of this change in crop frequencies for agro-biodiversity are largely unknown. Concerns have been raised that this development might accelerate biodiversity loss due to high crop cover and reduced light availability in energy crop stands which could further deteriorate the growing conditions for declining arable plant species. We analysed the transmissivity for photosynthetically active radiation (TPAR) in conventionally managed maize and oilseed rape fields (energy crops) and winter cereal fields (food/fodder crops) in Central Germany and contrasted it with TPAR measured in wheat fields managed according to agri-environmental schemes (AES). Secondly, we analysed the relation between light intensity and arable plant diversity metrics with respect to effects of field management and geographical differences. Light availability was lowest in maize stands (6% TPAR) followed by winter cereals and oilseed rape (10–13 %). Field margins were brighter than field centres (17% vs. 10%). Highest light transmissivity was measured on AES fields (57%), which was associated with elevated plant diversity. Light availability explained a significant fraction of the variation in species richness also on conventionally managed field margins (R^2 = 8%). Effects of light availability on community composition were found only when the least intensive systems (margins of conventional and AES wheat fields) were analysed. The main detrimental effect of the expansion of energy cropping on farmland habitat diversity is the loss of extensively managed farmland where light availability is higher. Reduced fertilizer use on conventional field margins will increase light availability and thus improve habitat conditions for arable plant species.

3.1 Introduction

During the last decade, new political frameworks for energy supply and agricultural production have led to large-scale changes in the relative frequency of major crop types grown in Western and Central Europe which fuelled the conversion of extensively managed fields, fallow land, pastures and meadows to intensively managed crop systems (Nitsch et al., 2012; Steinmann and Dobers, 2013). One major driver behind these changes are subsidies paid for the cultivation of energy crops (Charles et al., 2013; Khanna and Chen, 2013) which have been advocated by scientists and politicians as a tool to reduce CO_2 emissions for mitigating climate change (BMU and BMELV, 2010; IPCC, 2011).

Germany has declared particularly ambitious goals for replacing fossil fuels by renewable energies and the government pushes the transition by strong financial support (Britz and Delzeit, 2013). As a consequence, the area cultivated with energy crops in Germany tripled from less than 0.7 million ha in 2000 to ~2.1 million ha in 2012 equalling about 18% of the arable land (FNR, 2013a). This rapid expansion is currently mostly related to annual ('first-generation') energy crops (>90% of all energy crops grown on farmland) as these can be cultivated and handled with well-established, widely available machinery (FNR, 2013a, b). The two major energy crops in Germany are maize (*Zea mays* L.) and oilseed rape (*Brassica napus*, L.) (FNR, 2013a, b). In 2012, maize for biogas production (representing 32% of the total maize acreage) was grown on approximately 7% of the arable land, and oilseed rape for biofuel and vegetable oil (representing 70% of the total rapeseed acreage) was grown on 8% of the arable land (Destatis, 2013; FNR, 2013a, b). Political strategies project that the proportion of land used for energy crop cultivation might increase up to 34% by 2020 (BMU and BMELV, 2010). This fundamental change in crop frequencies and land use intensity will also alter the environmental conditions determining the biodiversity in bioenergy landscapes.

Light is arguably the most important resource affecting the growth and survival of the arable flora. It determines photosynthetic carbon gain and regulates developmental processes such as seed dormancy, germination, and photomorphogenesis (Ballaré and Casal, 2000; Holt 1995). Over a broad range of light intensities, productivity increases linearly with the amount of absorbed photosynthetically radiation (PAR, 380-760 nm). While the low-input cropping systems of the past were characterised by relatively high light transmission through the crop, low light availability is likely an important limiting factor for arable plant growth and survival in the dark understory of modern crop stands (Kruk et al., 2006; Olsen et al., 2005). Reduced radiation transmission through the crop can alter the competitive ability of arable plants and crops (Bornkamm, 1961; Liu et al., 2009) and may affect other environmental factors, which vary with stand density and related radiation transmission, such as air humidity and air temperature in the stand interior.

Agricultural ecosystems are unique in the way that humans shape habitat conditions more strongly than in any other ecosystem (Ellenberg and Leuschner, 2010). More than climate and geological

conditions, the choice of crop and the type of field management determine the light regime in crop stands (Ellenberg and Leuschner, 2010; van Elsen, 1994). The importance of light availability for the structure and composition of arable plant assemblages has already been emphasized by Rademacher (1939, 1950), but has led to only few empirical investigations thereafter (e.g. Kruk et al., 2006; Liu et al., 2009; Olsen et al., 2005). Certainly, large-scale agricultural intensification and changes in crop rotations in many regions of Central and Western Europe in the last decades must have had profound effects on the light regime in and below crop stands at the field and landscape scale. In particular, concerns have been widely raised that light availability in maize stands is reduced as compared to other commonly grown crops, affecting farmland plant diversity. The sustainability of high-input maize production to fuel biogas plants has therefore been questioned. A diverse arable flora may not only help to halt erosion and reduce nitrate leaching, it is also an important food resource and/or breeding habitat for the pollinating and pest-controlling insects and birds of the agricultural landscape (Jordan and Vatovec, 2004; Marshall et al., 2003; Parish et al., 2009). The role of altered light conditions in explaining the large losses in farmland biodiversity during the past 50 years (e.g. Fried et al., 2009; Meyer et al., 2013; Storkey et al., 2012; Sutcliffe and Kay, 2000) has not yet been quantified and its relative importance with respect to other chemical and mechanic stressors in modern cropping systems remains unknown.

Previous PAR measurements in modern crop stands were primarily done for the purpose of parameterizing of crop growth and yield models (Daughry et al., 1992; Gallo et al., 1995; Hipps, 1983), but did not deal with weed growth conditions. Our study for the first time links light measurments in modern high-yield crop stands to the weed vegetation using a systematic, replicated sampling design. We did this in regular cropland used for food, fodder or energy production and did not study experimental stands. Apart from a few exemplary measurements (van Elsen, 1994), such an approach has not been adopted before. This will allow drawing conclusions on the influence of changes in crop rotations on the light regime in crop stands and on weed growth conditions on the landscape level.

We addressed the following questions: (1) How does the light regime at ground level, expressed as PAR transmissivity (TPAR), vary between cropping systems? (2) Which spatial scale of observation contributes most to the variation in light conditions? (3) How much of the variation in TPAR in arable fields can be explained by crop performance (crop cover), management (choice of crop, conventional vs. AES cropping) and/or regional effects? (4) How does TPAR relate to the structure and composition of arable plant assemblages (weed cover, species richness and community composition) in different cropping systems in comparison to othe explanatory factors such as field management and study region?

3.2 Methods

3.2.1 Study area

The fields investigated were selected in two regions in the central uplands of Germany, which are characterized by a hilly countryside with plains highly suitable for productive arable farming (Fig. 3.1). The first study region, the Lower Saxon hills (LS), is situated in the county of Göttingen (1118 km²; elevation 104–579 m a.s.l.). Mean annual precipitation is 650 mm and mean annual temperature 8.7°C (DWD, 2013). The second study regions is located in the agricultural plains of the Thuringian Basin (TB; 1058 km²; 134–413 m a.s.l.). The climate in the Thuringian Basin is drier (mean annual precipitation: 500–550 mm) and slightly warmer (mean annual temperature: 9.2°C) than in the Lower Saxon hills (DWD, 2013). Due to differences in agricultural policies on both sides of the inner German border before 1990, the agricultural landscapes in the two study regions differ structurally. The proportion of farmland is higher in the Thuringian Basin than in the Lower Saxon hills (78% vs. 44%, respectively) and fields are on average larger [average size of an administrative field unit 18.8 ha (TB) vs. 6.6 ha (LS)].
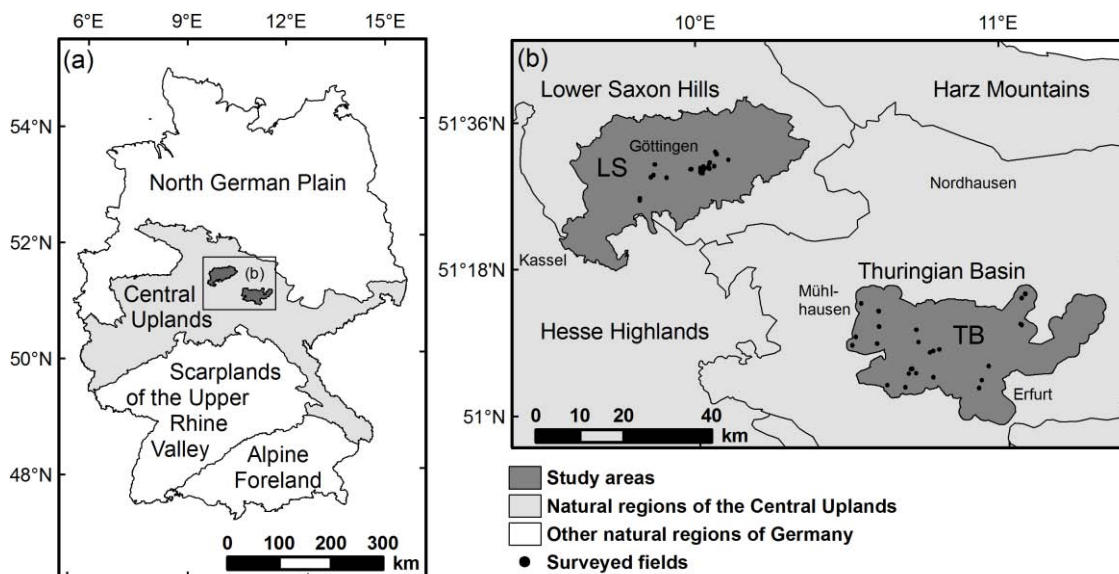
Fig. 3.1 Study areas in Lower Saxony and Thuringia, Germany. (a) Location in Germany; (b) delimitation of the study areas within the Lower Saxon hills (LS) and the Thuringian Basin (TB) with the distribution of the 25 surveyed fields in each study area (dots). The map is based on Universal Transverse Mercator (UTM) projection, zone 32 N (WGS 1984). Biogeographical boundaries are adopted from Meynen and Schmithüsen (1953-1962).

3.2.2 Sampling design

A total of 50 arable fields, equally distributed across both study regions, were selected for vegetation surveys, which were conducted in the summers of 2011 and 2012. In each region, five fields of four different crops (maize, oilseed rape, winter barley and winter wheat) under conventional management (i.e. application of fertilizers and herbicides according to common agricultural practice) were selected. We included only fields with a closed crop stand without any signs of crop growth failure. We did not distinguish between fields on which maize or oilseed rape was specifically grown for the purpose of energy production and fields where these crops were cultivated for food/fodder production. Interviews with farmers and energy plant operators, consultation of the agricultural literature (Lütke-Entrup and Schäfer, 2011) and earlier analyses of available field management data for 28 maize fields (Seifert et al., unpublished data) revealed not differences in fertilizer application or weed management between these two types of product usage. As far as oilseed rape is concerned, farmers are often not even aware of what their product is finally going to be used for by the purchaser. Impacts of energy cropping on biodiversity were thus assumed to arise exclusively from changes in the relative frequency of the crops in time and space and from the conversion of extensively managed fields, fallows, pastures and meadows to intensively managed arable land. Additionally, five winter wheat fields, which were managed according to an agri-environmental scheme (AES) for the protection of rare arable plant species, were studied in each region. On AES fields, no herbicides or fertilizers had been applied in the year of study, but sowing and tillage was done with the same intensity and timing as on adjacent conventionally managed fields.

On each field, two plots of 2 m × 50 m (i.e. 100 m²) were established for the surveys, the first one along the field margin (FM, starting with the first crop row) and the second one in the field interior (FI, with at least 10 m distance to the field edge parallel to the first plot). 100 m² is a common relevé size used in Central European arable field surveys (Meyer et al., 2013). Most AES fields were only 12-20 m wide; thus, the plots in the field interior were placed with 6–9 m distance to the field edge. Field surveys, which included vegetation sampling and PAR measurements, were conducted in the period when the crops were in their respective milk stages (around 3–6 weeks before harvesting) and crop cover was estimated to be at its maximum (i.e. winter barley: 8–16 June; oilseed rape: (24–29 June; winter wheat 14 June–8 July; maize: 30 July–8 August).

3.2.3 Sampling of arable weed assemblages

The cover and composition of all vascular plant species was recorded in the 100 m² plots using the Braun-Blanquet cover abundance scale consisting of nine classes according to Wilmanns (1998). Additionally, total crop cover and total weed cover were estimated visually. Species nomenclature follows Buttler et al. (2013). A comprehensive species list is given in appendix 3.A.

3.2.4 PAR measurement and calculation of TPAR

Quantum sensors (LI-190 Quantum Sensor, LI-COR, Lincoln, Nebraska, USA; Biggs et al., 1971) were used to measure the flux density of photosynthetically active radiation (PAR, expressed as μmol photons $m^{-2}s^{-1}$) above and below the crop canopy around noon time (10 a.m.–3 p.m.) on days with homogeneous sky conditions without rapid fluctuation of incident radiation. Although the sky conditions and sun angle necessarily varied between different measurement days, we are confident that our data are not biased by these variations, as Hipps (1983) showed that in late-season crop stands with high leaf area indices, measurements of PAR interception and transmission are not influenced by sun angle and cloud cover.

To measure how much TPAR was available for weed growth below the crop canopy, one PAR sensor was positioned at ground level ~15 cm above the soil surface in order to record the light climate experienced by the weeds (transmitted PAR; PAR_{trans}). A second PAR sensor was placed ~50 cm above the crop canopy for synchronously measuring incident PAR flux density (PAR_{inc}). To avoid biased readings, sensors were placed with great care in the stand in order to avoid damaging the crop canopy. The mobile sensor was mounted at the end of a 1.5 m-long stick which allowed conducting the measurements at a distance from the operator where he influenced radiation flux density only minimally. Both sensors rotated freely so that they were always in horizontal position. In each survey plot, ten measuring points were randomly selected and five consecutive light measurements taken at time steps of ~5 s at each point. Incident and transmitted PAR were recorded synchronously with a LI-1400 data logger (LI-COR, Lincoln, NE, USA). PAR transmissivity (TPAR, in %) was calculated as

$$TPAR = (PAR_{trans} / PAR_{inc}) \times 100 \qquad (eq. 3.1)$$

Mean TPAR per plot was obtained by averaging over the five consecutive measurements per point and subsequently calculating the mean across the ten points per plot. Both sensors were calibrated against each other before the start of the measurements.

3.2.5 Statistical analyses

3.2.5.1 *Analysis of transmissivity data*

Differences in TPAR (and also in the vegetation parameters crop cover, weed cover, species richness and light demand) between the different cropping systems were analysed using the non-parametric Behrens-Fisher test for multiple comparisons (R-package npmc; Helms and Munzel, 2009). We used Wilcoxon's signed rank test for paired samples to test for differences between the FI and the FM for the different cropping systems. All statistical tests in this study were performed with R software, version 2.11.1 (R Development Core Team, 2010).

Under the assumption that the spatial scale of observation in this study – within-field, between fields, between cropping systems and between regions – increases with each step in this sequence, we used variance component analysis (VCA) with nested random effects to analyse at which scale most of the variation in TPAR originates (R-package lme4; Bates, 2013; Crawley, 2007; Searle et al., 2006). Measurement points were nested within fields, fields within cropping systems (only for analyses across different crop types) and cropping systems or fields within study regions. Due to the particular error structure of percentage data, we expected binomial errors and used a logit link function.

To examine how much of the variation in TPAR in the arable fields was explained by crop cover (in %), field management (crop; conventional management vs. AES) and region, we conducted linear regression analyses (R-package: vegan; Oksanen et al., 2011). For quantifying the intersections between factors (i.e. the fractions of variation statistically not attributable to any single factor; not referring to interactions), we used variance partitioning (Legendre, 2008), when several factors were significant.

3.2.5.2   *Analysing the effects of TPAR on the diversity of weed assemblages*

The arable plant assemblages were described by their total weed cover, species richness and community composition. The crop itself, any woody species (including *Rubus fruticosus* agg.), representatives of volunteer grain (*Brassica napus*, *Hordeum vulgare*, *Triticum aestivum*, *Secale cereale* and ×*Triticosecale*) and specimens belonging to one of ten taxa represented with several species in the data set and which were not identifiable to species level, were excluded from the analyses concerning the arable plant assemblages (cp. appendix 3.A). Two plots from the FI (conventional wheat and barley) had to be excluded from certain analyses (CCA) due to lacking weed species in the plot.

Variance partitioning (Legendre, 2008) was used to quantify the explanatory power of TPAR for each of the four response variables (weed cover, species richness and community composition) in relation to field management factors and regional effects. Based on linear regressions (for weed cover and species richness) and canonical correspondence analyses (CCA, for community composition; R-package vegan), we obtained (1) the fractions of variance in the data set explained, if only one of the three explanatory factors [TPAR, management (crop; conventional vs. AES), region (LS vs. TB)] was considered, or if several factors were significant, and (2) the fractions of variance shared by any combination of two or all three factors.

3.3 Results

3.3.1 TPAR variability within and between different cropping systems

The PAR transmissivity of conventionally managed fields did not differ between the two study regions when analysed separately for the four crops and for the field interior and the field margin (appendix 3.B); the data were thus pooled in subsequent analyses. Across all 40 conventional fields, TPAR showed considerable variation (range in the field interior: 1.9–18.9%, mean: 9.9% ± 0.7 SE; field margin: 4.3–40.3%, mean: 16.9% ± 1.3; Fig. 3.2a, appendix 3.C). The coefficients of variation in the four crops ranged between 25–47% in the field interior and 35–61% at the field margin. TPAR was lower in maize fields than in the other three crops both in the interior (means: 5.9 vs. 10–13%; difference significant to oilseed rape) and at the margin (means: 10.9 vs. 18–19%; difference significant to oilseed rape and barley). TPAR was 50–80% higher at the margin than in the field interior.

Compared to conventionally managed wheat fields, TPAR in AES wheat fields was 5.5 fold (field interior) and 3.8 fold (field margin) higher and more variable between the fields and regions (range in the field interior: 35.1–83.3%, mean = 56.9% ± 4.9; range at the field margin: 39.6–94.1%, mean: 69.1% ± 6.1; Fig. 3.3a). Conventional and AES wheat fields contrasted also with respect to the frequency distribution of TPAR values (Fig. 3.4). Data showed positively skewed distributions with clumping of values below 20% TPAR in the conventional fields, but Gaussian or negatively skewed distributions with a modus >50% TPAR in the AES fields.
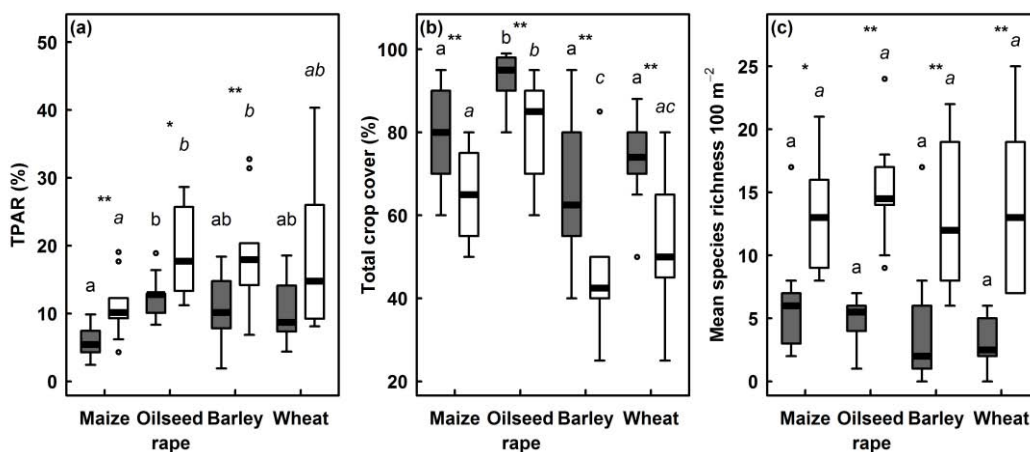
Fig. 3.2 Differences in (a) PAR transmissivity (TPAR), (b) total crop cover, and (c) mean number of weed species 100 m^{-2} of the weed assemblages (cover weighted) in the field interior (FI, grey boxplots) and at the field margin (FM, white boxplots) of four different crops in conventionally managed arable fields in the central uplands of Germany. Different letters indicate significant differences between crops (Behrens-Fisher test, $p \leq 0.05$, n=10 fields per group) for the FI (regular letters) and the FM (italic letters). Asteriscs indicate significant differences between plots in the FI and at the FM for a crop (Wilcoxon signed rank test; * $p \leq 0.05$; ** $p \leq 0.01$; n=10 per group).

3.3.2 The spatial origin of TPAR variation

Hierarchical variance component analysis (VCA) confirmed that the variation in light conditions in conventionally managed arable fields originated at the local rather than the regional scale (Table 3.1). Total TPAR variance was similarly large in the interior and at the margin of conventional fields, but the proportion of variance assignable to the different spatial scales differed. At field margins, variability was highest between different fields of a crop (48%) and differences between crops contributed only a small fraction of variance (9%). In the field interior, the choice of crop was more important (21%) and between-field variability was reduced (36%). Within-field variability contributed similarly at field margins (42%) and in the field interior (43%).

Separate analyses for maize, oilseed rape, barley and wheat revealed differences in the origin of variation between the crops. The highest TPAR variation (within-field to region) was registered in conventional wheat fields, the lowest in oilseed rape in the field interior and in barley at the field margin. In barley fields, between-field variation was of particular importance (field interior: 92%, field margin: 73%). For the other three crops, the contribution of between-field variation was low in the field interior (18–29%), but increased towards the field margin (32–54%). The region explained only a very small fraction of TPAR variation in the conventionally managed crops and a significant effect was only detected for conventional wheat (field interior) and oilseed rape (field margin) but not in the other crops.
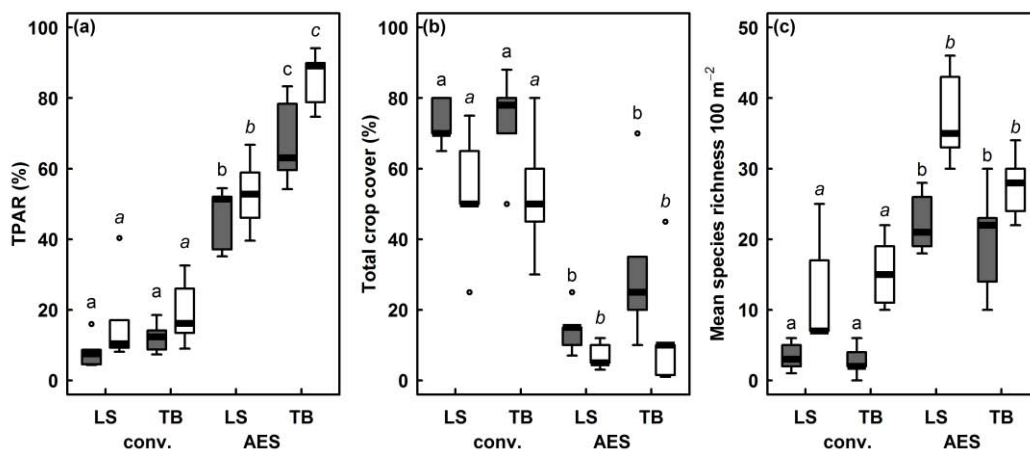
Fig. 3.3 Differences between conventionally managed wheat fields (conv., n=10) and wheat fields managed according to an agri-environmental scheme (AES, n=10) in the Lower Saxon hills (LS) and in the Thuringian Basin (TB), distinguishing between plots in the field interior (FI, grey boxplots) and at the field margin (FM, white boxplots) in (a) TPAR (in %), (b) total crop cover (in %), and (c) mean species richness per plot. Different letters indicate significant differences between crops (Behrens-Fisher test, $p \leq 0.05$, n=5 per group) for the FI (regular letters) and the FM (italic letters).

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Table 3.1 Variance of TPAR attributable to within-field, between-field, between-crop and between-region variation according to hierarchical variance component analysis (VCA with binomial errors and logit links); the nesting structure consisted of measurement points within fields with different kinds of crops in the two study regions added tot the model as random variables. Proportions (in %) of the total variance are given in brackets. All fields, all five studied cropping systems; Conv., conventionally managed; AES, managed under an agri-environmental scheme.

	n	Field interior (FI)					Field margin (FM)				
		Within field	Between fields	Between crops	Between regions	Total	Within field	Between fields	Between crops	Between regions	Total
All fields	50	0.29 (14.8)	0.24 (12.6)	1.41 (72.7)	0.00 (0.0)	1.94	0.27 (12.8)	0.32 (15.5)	1.51 (71.7)	0.00 (0.0)	2.10
Conv. crops	40	0.27 (43.4)	0.22 (35.6)	0.13 (21.1)	0.00 (0.0)	0.62	0.28 (42.4)	0.32 (48.2)	0.06 (9.4)	0.00 (0.0)	0.66
Wheat (conv., AES)	20	0.42 (14.8)	0.24 (8.4)	2.20 (76.8)	0.00 (0.0)	2.86	0.34 (10.6)	0.47 (14.8)	2.37 (74.7)	0.00 (0.0)	3.18
Maize	10	0.35 (70.7)	0.15 (29.3)		0.00 (0.0)	0.50	0.40 (67.8)	0.19 (32.2)		0.00 (0.0)	0.59
Oilseed rape	10	0.19 (79.3)	0.05 (20.7)		0.00 (0.0)	0.24	0.20 (51.8)	0.17 (42.8)		0.02 (5.4)	0.39
Barley	10	0.04 (8.2)	0.43 (91.8)		0.00 (0.0)	0.47	0.10 (27.3)	0.27 (72.7)		0.00 (0.0)	0.37
Wheat (conv.)	10	0.50 (64.8)	0.14 (17.6)		0.13 (17.6)	0.77	0.43 (46.5)	0.50 (53.5)		0.00 (0.0)	0.93
Wheat (AES)	10	0.36 (38.9)	0.34 (37.2)		0.21 (23.9)	0.91	0.23 (16.2)	0.36 (25.7)		0.81 (58.1)	1.40

Adding AES fields to the sample more than tripled the total variance of TPAR. Between-region variation was an important source of TPAR variation in AES wheat fields contributing with 58% (field margin) and 24% (field interior) to total variation (Table 3.1). Thus, AES fields are an important source of landscape-scale heterogeneity in the cropland light regimes.

3.3.3 Drivers of TPAR in arable fields

TPAR and management factors were mutually related. The regression analyses showed that field management factors (crop type and management intensity) best explained the variation in TPAR (Table 3.2). The crop type explained 23% of the TPAR variation in the interior of conventional fields and management intensity (conventional vs. AES) explained 81% in the interior and 73% at the margins when wheat fields were analysed separately. Crop cover co-varied tightly with management intensity, but was found to explain less variation (field interior: 59%, field margin: 51%). Management intensity still explained about 20% of TPAR variation both in the field interior and at the margins after controlling for the effect of crop cover. In conventionally managed fields, estimated crop cover varied between 40–99% in the field interior (mean: 78% ± 5 SE) and between 25–95% at the margin (mean: 61% ± 6) (Fig. 3.2b; difference significant at $p \leq 0.001$; appendix 3.C). On AES wheat fields, total crop cover was much lower (FI, mean: 23% ± 6; FM, mean: 10% ± 4; Fig. 3.3b) than on conventional wheat fields in both study regions (Fig. 3.3b; difference significant at $p \leq 0.01$; appendix 3.C). Differences between sampling regions did not explain the variation in TPAR.
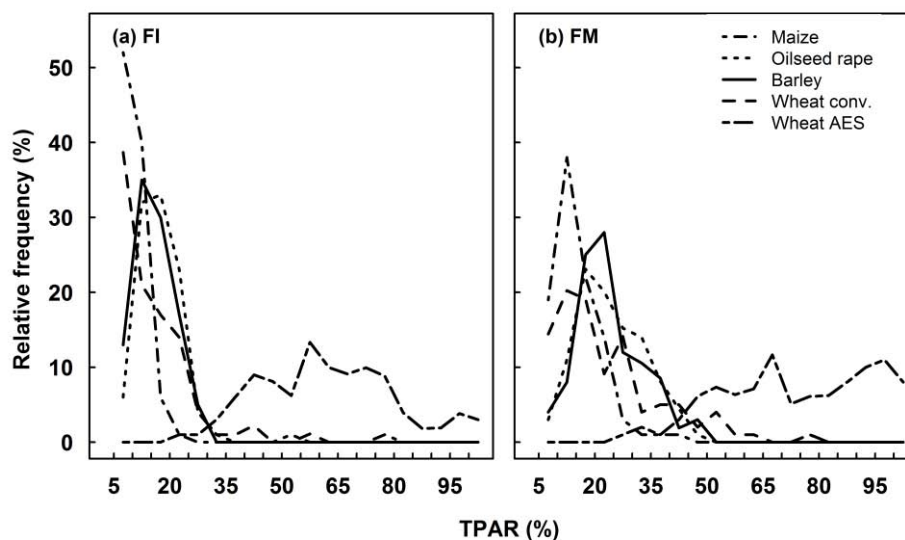
Fig. 3.4 Relative frequency of classes of PAR transmissivity (TPAR) to the soil surface (interval width: 5%) in four different conventionally managed crops and winter wheat managed according to an agri-environmental scheme (AES) for (a) the field interior and (b) the field margin in the central uplands of Germany (n=10 fields per category).

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Table 3.2 Fractions of the variation in TPAR explained by crop cover, management factors (crop, management intensity: conventional management (conv.) vs. agri-environmental schemes (AES)), and region (Lower Saxon hills vs. Thuringian Basin), in the field interior (FI) and at the field margin (FM) of conventionally managed fields with four different crops and of wheat fields under conventional and AES management, assessed in the central uplands of Germany. Fractions are the adjusted R^2 from redundancy analyses (RDA). Intersections were obtained from variation partitioning (Legendre, 2008).

		TPAR	
		FI	FM
Conv. crops	$n =$	40	40
Crop cover + Management + Region		**35.2**[**]	**17.4**[*]
Crop cover (total)		ns	ns
Management (total)		**23.4**[**]	ns
Region (total)		ns	ns
Crop cover∩Management		-	-
Crop cover∩Region		-	-
Management∩Region		-	-
Crop cover∩Management∩Region		-	-
Wheat fields (conv., AES)	$n =$	20	20
Crop cover + Management + Region		**87.0**[**]	**80.4**[**]
Crop cover (total)		**59.1**[**]	**50.5**[**]
Management (total)		**81.3**[**]	**72.9**[**]
Region (total)		ns	ns
Crop cover∩Management		60.2	53.0
Crop cover∩Region		-	-
Management∩Region		-	-
Crop cover∩Management∩Region		-	-

∩ indicates that the variation explained is shared between the respective components. These intersections cannot be tested for significance (Legendre, 2008). Intersections were only assessed when all components explained a significant fraction of the variation in the single factor models. Models were tested with ANOVA (ns, $p > 0.05$; *, $p \leq 0.05$; **, $p \leq 0.01$).

3.3.4 Influence of TPAR on weed assemblages

3.3.4.1   *Influence of TPAR on weed cover*

TPAR was not found to explain a significant part of the variation in weed cover between conventional fields, neither in the field interior nor at the margin. The total weed cover in conventionally managed fields was generally very low with on average 3% (± 2 SE) cover in the field interior and 16% (± 3) at the margin (appendix 3.C). Differences between crops or regions also did not capture a significant part of these subtle weed cover variations.

In contrast, regression analyses showed that weed cover in the field interior was largely explained by the factors TPAR (34%) and field management (38%), both being highly correlated to each other, when conventional and AES wheat fields were assessed. On the other hand, none of the investigated factors explained the difference in weed cover at the field margin in conventional vs. AES wheat fields. Weed cover was with on average 38% ($\pm$ 8) by factor 9.5 higher than in conventionally managed wheat fields (mean: 4% $\pm$ 4). At the margin, weed cover differed much less between AES fields (mean: 28% $\pm$ 8) and conventional wheat fields (mean: 17% $\pm$ 4). Variation in weed cover mainly seems to be driven by factors other than the ones included in this study.

3.3.4.2   Influence of TPAR on species richness

Contrary to weed cover, species richness per plot was related to TPAR when management intensity was reduced (i.e. at the margin of conventional fields and when AES fields were included in the analysis, Table 3.3). On conventional field margins, species richness increased with TPAR (correlation coefficient $R = 0.32$, $p = 0.05$, n = 40) and TPAR explained 8% of the variation in species richness. The increased variability in light conditions and the reduction of management intensity at the field margins and on AES fields was also found to go along with increased variability in species richness (Fig. 3.2c and 3.3c). When contrasting conventional and AES wheat fields, both TPAR and management intensity were highly significant explanatory factors for species richness. Both factors co-varied tightly and together explained 83% of the variation in species richness in the interior (with 73% being shared between the factors, not being exclusively attributable to either one). At the field margins, both factors together explained 50% of the variation, with 35% being shared between the factors. In contrast to the results from conventionally managed fields, species richness decreased with increasing TPAR on AES field margins where other factors than light seemed to limit plant growth (correlation coefficient $R = -0.65$, $p = 0.04$, n = 10). In the interior of conventional fields, differences in species richness between fields were very small and neither TPAR nor crop or region explained this small variation.

Species richness per plot did not differ between the four conventional crops neither for the field margins nor for the field interior (Fig. 3.2c). Weed assemblages of AES wheat fields were, however, much more species-rich than conventionally managed wheat fields ($p \leq 0.05$; Fig. 3.3c). Conventional fields (all crops averaged) had on average five species per plot in the interior and 14 species at the field margins, while 21 and 33 species, respectively, were recorded in the AES fields. Apart from few exceptions, we found the most species-rich conventionally managed field margins to still be less species-rich than the poorest AES field margins. There was, however, considerable overlap in species richness between conventionally managed margins and the centres of AES fields.

We recorded a total of 157 plant species in the 100 plots, of which 155 taxa were found at the field margin and 90 in the field interior (appendix 3.A). 121 species occurred in the ten investigated AES fields and 101 species in the 40 conventionally managed fields. With 24 species restricted to the margin of conventional fields (maize: 8 species; oilseed rape: 2; winter barley: 7; winter wheat: 5) and 31 species restricted to the margin of AES fields, field margins were very important for the regional species pool. Three species were restricted to the interior of AES fields, but no species occurred solely in the interior of conventionally managed fields.

Table 3.3 Fractions of variance (adjusted R^2, in %) explained by TPAR, management factors (crop; management intensity: conventional management (conv.) vs. agri-environmental schemes (AES), and region (Lower Saxony uplands vs. Thuringian Basin)) for total weed cover, species richness and the community composition of weed assemblages on arable fields in the central uplands of Germany. Fractions are based on redundancy analyses (RDA, for weed cover and species richness) or canonical correspondence analyses (CCA, for community composition) and intersections were obtained from variation partitioning.

	Weed cover		Species richness		Community composition	
	FI	FM	FI	FM	FI	FM
Conv. crops $n =$	40	40	40	40	38	40
TPAR + Management + Region	ns	ns	ns	ns	**10.4**[*]	**13.2**[**]
TPAR (total)	ns	ns	ns	7.7[*]	ns	ns
Management (total)	ns	ns	ns	ns	9.4[*]	10.1[**]
Region (total)	ns	ns	ns	ns	ns	2.3[**]
TPAR∩Management	-	-	-	-	-	-
TPAR∩Region	-	-	-	-	-	-
Management∩Region	-	-	-	-	-	0.1
TPAR∩Management∩Region	-	-	-	-	-	-
Wheat fields (conv., AES) $n =$	20	20	20	20	19	20
TPAR + Management + Region	**31.8**[*]	ns	**83.3**[**]	**62.2**[**]	**12.4**[**]	**18.3**[**]
TPAR (total)	**33.6**[**]	ns	**73.8**[**]	**35.2**[*]	ns	**8.0**[**]
Management (total)	**37.9**[**]	ns	**79.9**[**]	**63.4**[**]	**4.0**[*]	**6.8**[**]
Region (total)	ns	ns	ns	ns	**6.9**[**]	**7.4**[**]
TPAR∩Management	36.8	-	73.3	34.9	-	0.7
TPAR∩Region	-	-	-	-	-	1.0
Management∩Region	-	-	-	-	0	0
TPAR∩Management∩Region	-	-	-	-	-	0

∩ indicates that the variation explained is shared between the respective components (i. e. it cannot be attributed exclusively to one of the components). These intersections cannot be tested for significance (Legendre, 2008). Intersections were only assessed when all components explained a significant fraction of the variation in the single factor models. Models were tested with ANOVA (ns, $p > 0.05$; *, $p \leq 0.05$; **, $p \leq 0.01$). FI, field interior; FM, field margin.

3.3.4.3   Influence of TPAR on community composition

A relationship between TPAR and community composition was only found when the most extensively managed cropping systems (margins of conventional and AES wheat fields) were

assessed. Here, TPAR explained the highest fraction of the variation (8%). The contribution of TPAR was largely independent of the fractions explained by management intensity (7%, 1% shared with TPAR) and region (7%, 1% shared with TPAR). *Adonis aestivalis*, *Euphorbia exigua* and *Falcaria vulgaris* are examples of species which preferred wheat fields with high TPAR, whereas *Fumaria officinalis*, *Lamium purpureum* and *Myosotis arvensis* were less light-demanding and occurred more often when TPAR was low.

In the conventional fields, not TPAR but the choice of crop explained the largest fraction of variation in community composition in the field interior (9%) as well as at the field margin (10%, Table 3.3). We found *Chenopodium hybridum*, *Echinochloa crus-galli*, *Persicaria lapathifolia* and *Solanum nigrum* to prefer maize fields over oilseed rape and winter cereals (cp. appendix 3.A). *Descurainia sophia*, *Matricaria recutita*, *Myosotis arvensis*, *Papaver rhoeas* and *Thlaspi arvense* are examples of species which were commonly found in oilseed rape, barley and wheat fields, but were absent in maize fields. We could not identify any species which showed a clear preference for any of the three winter-sown crops (oilseed rape, barley and wheat), apart from *Capsella bursa-pastoris*, which was most commonly found in oilseed rape.

In addition, weed assemblages at the margin of conventional fields differed between the two study regions which explained 2% of the variation in community composition. The influence of region on the composition of the weed assemblages was even larger in AES fields (7 and 6% of variance explained in the field interior and at the margin, respectively) than in conventional cropping systems.

There was a large number of species regularly occurring on AES fields (frequency > 40%) that were completely absent in conventional fields (e. g. *Geranium columbinum*, *Euphorbia exigua*, *Arenaria serpyllifolia*, *Valerianella dentata*, *Medicago lupulina*). Rare species, such as *Adonis flammea*, *Bupleurum rotundifolium* and *Euphorbia falcata* (all listed as critically endangered in Ludwig and Schnittler, 1996) were also restricted to AES fields.

3.4 Discussion

3.4.1 Light transmission patterns in different cropping systems

With average PAR transmissivities in the range of 10–13% in the field interior at the time of peak crop biomass, conventionally managed wheat, barley and oilseed rape fields are equally or less shading than temperate grasslands (Lane et al., 2000; Tappeiner and Cernusca, 1989) and less shading than closed broad-leaved forests (Ellenberg and Leuschner, 2010). Nevertheless, these modern crop stands are nowadays probably too dark for many typical light-demanding arable weed species. Experimental evidence shows that many arable species grow best at high illumination (Bornkamm, 1961; Fitter and Ashmore, 1974; Holt, 1995). Tall-growing maize, the crop with the largest increase in

cultivation area in Germany during the recent boost of energy farming (Steinmann and Dobers, 2013), reduces transmissivity to lower levels than the other three conventional crops, and the range of transmissivities measured was smaller in maize than in all other crops. With a mean transmissivity of 6%, maize fields are as dark as fertile grasslands and mature spruce forests (Ellenberg and Leuschner, 2010).

The most important TPAR gradient existing in conventional high-input cropland is the difference between field margin and interior. In general, TPAR was not only higher (average across all four conventional crops: 17%) at the field margin than in the interior, but also more variable ranging 4 to 40%). Field margins therefore have a higher potential to increase habitat diversity in terms of light availability than the field interior. Consequently, the TPAR reduction in maize stands is ecologically most relevant at the field margin, where the highest diversity of arable plants is usually found (Fried et al., 2009; van Elsen, 1994). We assume that the differences between field edge and field interior are attributable to a set of interacting factors which influence both light conditions and the arable flora. Due to technical and legal constraints, margins tend to receive less crop seeds, fertilizer and herbicides (Wilson and Aebischer, 1995) and crop yields are typically lower (Sparkes et al., 1998). In accordance with these findings, we found crop cover to be on average 17% higher in the interior than at the margin.

One important result of this study is that the variability in PAR transmissivity in conventionally managed fields is generated almost exclusively at the local scale, whereas regional differences in TPAR were negligible (except for the interior of wheat fields). Similarly, between-field variation within a region was of minor importance in the field interior, at least for maize, oilseed rape and wheat. We assume that between-field differences in TPAR, if they exist, mainly reflect different management practices and gradients in soil and climate conditions. Consequently, our data evidence a general homogenization of site conditions and management practices that has taken place in Central and Western Europe's farmland during the last decades. Formerly existing differences in weed management, fertilization and soil conditions across farms and regions (Meyer et al., 2013) have become less important for crop growth and TPAR in conventional cropping systems. The homogenization of habitat conditions at the local and landscape scale is a major driver of impoverishment in the regional species pools (Kohlbrecher et al., 2012; Meyer et al., 2013; Smart et al., 2006). The three-fold increase in total TPAR variance after adding the ten AES fields to the sample shows that extensively managed arable fields greatly increase the heterogeneity of light conditions inside crop stands creating ecological niches for weeds with higher light demand.

Even though field margins of conventionally managed fields also offer considerable variation in light condition, there was little overlap with light conditions measured on AES fields. The most illuminated conventional field margins were only just as bright as the darkest AES fields. A considerable part of

the differences in TPAR between conventional and AES field margins can be explained by differences in crop cover (50%). These differences in crop cover are thought to be driven by several interacting factors related to field management and environmental conditions. This supports the results of Kleijn and van der Voort (1997) who showed that fertilisation of crops synchronously increased biomass production and lowered light penetration, thereby reducing plant growth and species richness at the level of single species and at the level of weed communities. The absence of fertilization on AES fields reduces the productivity of the sites, but this effect interacts with site-specific soil conditions. The AES fields of our study were typically located on less productive soils within the landscape (mostly upslope of adjacent conventionally managed fields) as these are the only locations where rare arable species persist until today. Site conditions were, however, diverse. The AES fields contained both the fields with lowest and those with highest weed cover (range: 9–100%) of this study. The missing application of herbicides releases arable non-crop plants from chemical stress and increases the competition between crops and weeds on AES fields as compared to conventional field margins. In combination with the only moderate to low productivity of the sites, this resulted in either a sparse or a dense and species-rich weed cover with co-existence of different growth forms. All AES fields were found to offer diverse light conditions in the upper range of the spectrum (35–94%). Fertilizer input and herbicide application on conventional field margins, typically in combination with more fertile soils, increase the competitive strength of the crop, lead to a denser crop canopy and reduce light transmission.

3.4.2 The influence of TPAR on the diversity of weed assemblages

The main determinants of weed species richness in the 50 investigated conventional and AES fields were TPAR and management intensity. Both factors are highly interacting, because high PAR transmissivity was only found on AES fields. AES management also increased weed cover, but the difference to conventional fields was significant only in the field interior and not at the field margin. We explain this surprising result by the fact that some margins of AES fields are located on particularly nutrient-poor and often relatively dry soils supporting only sparse vegetation cover. Correspondingly, TPAR did not explain weed cover at the field margin suggesting stronger growth limitation by nutrient and water shortage than by low light on the less productive AES fields (Kleijn and van der Voort, 1997). Belowground constraints of productivity are thought to be the main drivers of weed species richness in extensively managed herbicide-free arable fields, because they reduce the competitive ability of the crop (Jørnsgård et al., 1996) and of certain competitive N-demanding weeds.

Both weed cover (median <1%) and species richness (on average <5 species per 100 m²) were extremely low in the interior of conventional fields, which compares well with surveys by Meyer et al. (2013) in Central and Northern Germany. The species richness and cover of these impoverished weed assemblages seem to be primarily under the control of chemical and mechanical weed management

(Marshall et al., 2001), since we found no effect of TPAR, region or crop type in the interior of conventional fields.

The influence of light availability on species richness was larger at the field margins, where it was the only influential factor. Species richness was found to be significantly correlated to light availability, but the relatively low correlation coefficient of 0.3 indicates that factors other than light influence species richness on field margins as well. Our results show that these other factors do not co-vary with the kind of crop cultivated on the respective field margin. Regional (i.e. macroclimatic) effects can also be excluded as drivers of species richness on conventional field margins, as we found no significant effect in the regression analyses. It appears that reduced herbicide loads, i.e. the partial release from chemical stress, are mostly responsible for the stronger influence of TPAR on species richness at the margins.

The high variability of light conditions at conventional field margins was found to go along with a high variability in species richness. Species numbers found on the most species-rich conventional field margins were comparable to species numbers in the interior of relatively species-poor AES fields. Field margins are among the last remaining strongholds of arable plant diversity in modern farmland because management intensity is lower and light transmission higher than in the field interior (Fried et al., 2009; Marshall and Moonen, 2002). However, intensification has greatly reduced the diversity of weed communities at the margins as well (Leuschner et al., 2013; Meyer et al., 2013; Storkey et al., 2012), and the ongoing increase in mean field size has dwarfed the fraction of a field being allotted to the field margin. Our results indicate that increasing the light availability on conventional field margins through reduced fertilization and thus lowered crop cover represents a promising measure of improving the habitat quality for arable plants in agricultural landscapes.

None of the four tested crops can be recommended as particularly suitable for enhancing light availability and species richness at conventional field margins. Compared to the other three investigated crops, maize fields were the darkest, but this gradient was not reflected in a significant difference in species richness between maize field margins (on average 13 species per plot) and margins of the other three crops (14–15 species per plot). Contrary to species richness, the community composition of conventional field margins was not influenced by TPAR, but showed a significant influence of crop type with different crops being preferred by different sets of weed species. We found a set of species, such as *Echinochloa crus-galli* and *Solanum nigrum*, which preferred maize fields over the other three crops, and a set of species apparently preferring oilseed rape, barley and wheat over maize (e.g. *Matricaria recutita*, *Papaver rhoeas*). Differences in the life cycle of these weed species suggest that their occurrence is promoted or inhibited by the specific phenology of the crop and the timing of herbicide treatments and ploughing, while differences in the light regime at maximum crop cover are of secondary importance. Oilseed rape, winter barley and winter wheat are

sown in autumn and maize as a thermophilous species in late spring. Weed species preferentially associated with maize germinate late in spring, when the effect of herbicides applied to the young maize plants has vanished. Several of them are C4-grasses with elevated temperature optimum, which are unable to withstand the Central European winter as juvenile plants. In contrast, the weeds found preferentially with winter-sown wheat, barley and oilseed rape are all facultative winter annuals which obligatorily germinate in autumn or early in spring (Krumbiegel, 2002; Trefflich et al., 2002). Thus, one way to promote the phytodiversity of arable land is the combination of crops with different phenologies.

3.5 Conclusions

The key constraints for diverse weed communities in modern cropping systems are the very effective weed management and the high fertilizer inputs, amplified through dense crop cover and related low light transmissivity to the ground. Our results demonstrate that conventionally managed energy crops (maize, oilseed rape) and conventionally managed winter cereals (wheat, barley) do not systematically differ in terms of weed cover and species richness, even though maize fields were darker than cereal fields in our study regions. Increasing first-generation energy crop cultivation will therefore not necessarily reduce species richness in agricultural landscapes, as long as it does not replace species-rich extensively managed arable fields, fallow land, pastures or meadows. A balanced mix of winter-sown crops (e.g. winter cereals, oilseed rape) and summer-sown crops (e.g. maize) should have a positive influence on the size of the regional arable species pool by creating habitats for species with different phenologies and radiation requirements. However, increasing the diversity of intensively managed crops in the farmland does not solve the agrobiodiversity crisis and it does not generate habitats for the large number of endangered weed species.

Our results do not support claims about a positive synergy between bioenergy production and biodiversity conservation, and concerns about negative biodiversity implications of energy cropping with first-generation energy crops could not be refuted. In modern farmland, an inevitable trade-off exists between maximizing biomass yields and sustaining biodiversity (Gabriel et al., 2013), no matter what kind of crop (food, fodder or bioenergy) is produced. Our results show convincingly that a diverse arable flora is dependent on sites with low to medium nutrient availability, reduced crop cover and corresponding higher light availability and the absence of herbicide application. In order to comply with international and national biodiversity targets, an expansion of these kinds of arable habitats remains crucial. EU and German energy policy has created economic circumstances which strongly favour the intensification of land use practices. Attempts to protect species-rich farmland habitats as part of a diverse cultural landscape suffer from disproportionate disadvantages. The EU set-aside scheme which had a strong positive impact on farmland biodiversity has been abolished in 2007

to meet the demand for land for energy cropping. The amount of grassland in Germany declined by 5% (250,000 ha) between 2003 and 2012 (Behm, 2012), largely due to conversion to arable land (Nitsch et al., 2012). The bioenergy boost's most deleterious effect on biodiversity is related to its high demand for land which threatens the last strongholds of agrobiodiversity. In the meantime, prices for arable land keep soaring (Emman et al., 2011), and agri-environmental schemes, which compensate farmers for managing arable land extensively, suffer from financial unattractiveness. Legal measures, such as a reform of the EU Common Agricultural Policy (CAP) in the direction of a stronger support of farmers who maintain biodiversity (Oppermann et al., 2012), and the establishment of networks of agrobiodiversity sanctuaries with biodiversity-friendly extensive cropping, are urgently needed.

3.6 Acknowledgements

3.7 References

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Chapter 4

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Seifert, C., Leuschner, C., Culmsee, H. (2015)

Short rotation coppices as habitat for vascular plants

4.1 Introduction

The idea to produce wood chips from fast-growing trees cultivated in the form of short rotation coppice (SRC) on arable land arose from the need to find sustainable solutions to meet the increasing demand for wood products and energy while at the same time reducing green-house gas emissions and halting the overexploitation of forests (Caslin et al., 2010). The strong increase in the global demand for wood chips during the last decade has been driven by current policy frameworks targeting the reduction of greenhouse gas emissions and the diversification of the energy supply. Between 2000 and 2010, the global net solid biofuel trade increased sixfold, from 3.5 to 18 million t a^{-1} (Lamers et al., 2012). Europe is the most important player on the global wood chip market, covering about two thirds of the global net solid biofuel trade in 2010 (Lamers et al., 2012). Sustainable wood production not only implies maintaining soil fertility and avoiding N_2O emissions but also preserving biodiversity. The impact of SRC on biodiversity has been an important research focus in recent years (Rowe et al., 2009; Dauber et al., 2010). The effects on biodiversity will differ with the regional setting in which SRC is cultivated. In regions such as western and central Europe, where the most dramatic decline in species richness in the last decades has occurred on farmland rather than in forests (Wesche et al., 2012; Meyer et al., 2013), the type of farming system substituted by SRC will influence the impact on biodiversity most strongly, the effect depending on whether the previous land use was more or less beneficial for biodiversity than SRC (Rowe et al., 2009). Once established, plant community composition in SRC is affected by the soil and climate conditions, the land use history and the surrounding landscape (Baum et al., 2012b, Simmering et al., 2013).

Within the framework of the BEST research project, the effects of SRC on plant diversity were investigated in two study regions in central Germany. The aim of the study was to characterise SRC as

habitat for vascular plants and to evaluate its potential contribution to the conservation of plant species diversity in European cultural landscapes. Data from vegetation surveys, analyses of soil parameters and measurements of the light regime in the SRC stands studied were compared with literature data from alternative land use types, such as arable land, grassland and forest. More specifically, the following research questions were addressed:

(1)   How do habitat conditions and the associated plant communities in SRC change with time since planting?

(2)   How do habitat conditions prevailing in SRC during the different growth stages of the first rotation compare with habitat conditions associated with other types of land use?

(3)   What are the implications of these findings for the conservation of vascular plant species diversity?

(4)   How can SRC be managed to maximise positive effects on vascular plant diversity?

4.2 Methods

4.2.1 Study sites

The study was carried out in 10 SRC plantations of different ages located in two study regions in central Germany (Niedersächsisches Hügelland and Thüringer Becken) and planted with either poplar (variety Max 1; *Populus maximowiczii* × *P. nigra*) or willow (variety Tordis; (*Salix schwerinii* x *S. viminalis*) × *S. viminalis*) (Table 4.1). Six plantations were newly established on arable land in early spring 2011 at three different sites. Four older plantations established in 2005, 2007 and 2008 were also surveyed. As no older plantations existed in the Niedersächsisches Hügelland, only newly established plantations were surveyed in this region. The landscape context, climate and soil conditions were similar for all sites within each region but differed considerably between the two study regions. The climate of the Niedersächsisches Hügelland was more oceanic than that of the Thüringer Becken (mean annual precipitation of 650 and 500 – 550 mm, respectively; mean annual temperature of 8.7 and 9.2°C, respectively), and the agricultural landscape was characterised by smaller fields and a higher diversity of landscape elements such as field margins and hedges. The soils in the hill landscape of the Niedersächsisches Hügelland were also less loamy and had a lower pH than in the Thüringer Becken.

All of the SRCs in the Thüringer Becken (see Table 4.1) were established on former arable land, which had been used to produce crops using conventional farming techniques for at least 100 years. Only at the Dachwig site had the land been left fallow for 2 years before the establishment of the SRC.

The site at Reiffenhausen was historically arable land, but was not cropped during the 1990s to comply with the European set-aside requirements. On the land now hosting the poplar plantation at that site and on the western half of the willow plantation, crop production was resumed in 2009. The eastern part of the willow plantation remained fallow until the SRC was established in 2011.

Table 4.1 Characteristics of the SRC plantations studied.

ID	Site	Tree genus	Year of planting	Nearest village	Study region	Area (ha)	Row spacing (m)[a]
B1	BERTA I (new)	Poplar	2011	Großfahner		0.3	0.75/1.50
		Willow	2011		Thuringian Basin	0.3	0.75/1.50
B2	BERTA II	Poplar	2011	Döllstädt		0.5	0.75/1.50
		Willow	2011			0.5	0.75/1.50
R	Reiffen-hausen	Poplar	2011	Reiffen-hausen	Lower Saxon Hills	0.5	0.75/1.50
		Willow	2011			0.6	0.75/1.50
B1o	BERTA I (old)	Poplar	2007	Großfahner		0.3	0.80/0.80
		Willow	2005		Thuringian Basin	0.4	1.50/1.50
D	Dachwig	Poplar	2008	Dachwig		0.3	1.00/1.00
		Willow	2008			0.3	0.75/2.50

[a] Most of the plantations were planted in narrow twin rows with larger spaces between the twin rows. Row spacing distance between rows in a twin row/distance between the twin rows.

The landscape surrounding all of the SRCs in the Thüringer Becken was dominated by large-scale arable land use for more than a century. The nearest forest was approximately 2 km away from the sites B1, B1o and B2 and 3.5 km from site D (see Table 4.1 for abbreviations). The poplar and willow plantations at Reiffenhausen were adjacent to arable land set aside during the 1990s. To the north and north-east, they bordered the former inner German border, which was characterised by a mosaic of open grassland with solitary trees, hedges and woods. The closest forest was located across the old border in Thüringen, less than 0.5 km away.

No fertiliser was applied in the SRC studied. Weed management consisted of cutting weeds with a mower between the twin rows at irregular intervals. Herbicide application (Thuringian sites: fluazifop-p; Reiffenhausen: diflufenican, flurtamone, flufenacet) took place in April/May 2011. Due to the late application, however, many weeds were already too mature for the herbicides to have an optimal effect. The recommended application periods to maximise weed suppression would have been either just before or directly after germination or, in the case of fluazifop-p, from the time of emergence of the second to the fourth pair of leaves of the targeted weeds.

4.2.2 Vegetation sampling and analysis

In each SRC, three plots of 1 m × 25 m were randomly placed inside the stand, parallel to the planted rows, with a minimum distance of 10 m between plots. Three more plots of the same size were

surveyed at the edge of each SRC starting with the first row of planted trees. The cover of all vascular plant species (with the exception of the planted trees) was recorded according to the Braun-Blanquet cover abundance scale in each plot (Wilmanns, 1998). The surveys within all of the SRCs took place (i) in June/July 2011, (ii) in April 2012 and (iii) in June/July 2012. The surveys at the edges took place during the latter two periods. The Dachwig SRC was only surveyed in July 2012, and only the interior of the stands was considered.

The vegetation data were analysed with respect to habitat preference, naturalisation status and red list status of the species. The records from all three plots per SRC were pooled. The species numbers mentioned in this paper always refer, therefore, to a plot size of 75 m². The habitat preference of a species was assigned based on the main occurrence of the species in a vegetation formation (*Hauptvorkommen*) according to FloraWeb (BfN, 2014). Species not clearly assignable to one habitat type were classified as habitat generalists. Data about the naturalisation status were obtained from the BiolFlor database (BfN, 2002). Native species were distinguished from archaeophytes (naturalised prior to 1492 AD) and neophytes (naturalised after 1492 AD). The red list status was obtained from Ludwig and Schnittler (1996).

4.2.3 Soil sampling and analyses

Soil samples were taken at four random locations within each vegetation sampling plot (with the exception of the Dachwig SRC) from the uppermost 10 cm of the soil column with a soil core of 5.1 cm diameter in summer 2011. The four samples per plot were thoroughly mixed and stored at 4 °C. The pH of the fresh soil was measured with a pH meter in H_2O in accordance with HFA (2005). The base saturation ($[Ca^{2+}, Mg^{2+}, K^+]/[Ca^{2+}, Mg^{2+}, K^+, Al^{3+}, Mn^{2+}, Fe^{2+}, H^+]$, in %) was calculated from an analysis of the effective cation-exchange capacity via percolation with $BaCl_2$ solution (0.2 N) based on methods described in HFA (2005). The plant-available phosphorus was extracted with anion-exchange resin bags (Sibbesen, 1977). The total amount of carbon (C_{tot}) and nitrogen (N_{tot}) and the C/N-ratio for each sample were determined with a C/N-elemental analyser (Vario EL III, Elementar, Hanau, Germany). To determine the amount of organic carbon (C_{org}), samples were heated to 450°C. In the process, all of the organic carbon was burned away, and the quantity of the remaining inorganic carbon was measured and subtracted from the total carbon (C_{tot}). An estimate of the soil organic matter content (in %) was obtained by multiplying C_{org} with the factor 1.72.

4.2.4 Measurements of PAR-transmission

Quantum sensors (LI-190 Quantum Sensor, LI-COR, Lincoln, Nebraska, United States) were used to measure the flux density of photosynthetically active radiation (PAR, expressed as μmol photons m^{-2} s^{-1}) above and below the tree canopy around noon (10 a.m. to 3 p.m.) on days with

uniformly overcast sky. One PAR sensor was positioned at ground level, approximately 15 cm above the soil surface, in order to record the light climate experienced by herbaceous plants in SRC (transmitted PAR; PAR_{trans}). A second PAR sensor was placed approximately 1 m above the tree canopy (in the case of young stands) or in the open about 30–50 m south of the SRC (older stands) to measure incident PAR flux density (PAR_{inc}). In each SRC, 10 measuring points were randomly selected and 5 consecutive light measurements taken at time intervals of approximately 5 s at each point. With the help of a 100 m cable, PAR_{trans} and PAR_{inc} were recorded synchronously with a LI-COR LI-1400 data logger. PAR transmissivity (TPAR, in %) was calculated as

$$TPAR = (PAR_{trans} / PAR_{inc}) \times 100 \qquad \text{(eq. 4.1)}$$

The mean TPAR per SRC was obtained by averaging over the five consecutive measurements per point and subsequently calculating the mean across the 10 points per SRC.

4.3 Results and Discussion

4.3.1 Habitat conditions and plant diversity in newly established short rotation coppices

Habitat conditions in the first and second year SRC were similar to habitat conditions on cropland left fallow. In July of the year of planting of the cuttings, the total tree cover in the SRC stands was about 1% and the mean PAR transmittance was 89% (Fig. 4.1), creating optimal conditions for pioneer vegetation. On fertile soils, canopy closure may be achieved in SRC in the second year after planting, if weather conditions are favourable (Tubby and Armstrong, 2002). In this study, this was far from being the case, even on the loess sites. The tree cover was still relatively low in the second summer (<18%), but PAR transmittance decreased to 66%. As was mentioned in section 4.2.1, weed management in the newly established SRCs did not follow standard agricultural techniques (cf. Tubby and Armstrong, 2002; Caslin et al., 2010). The sites were not prepared with glyphosate in either autumn or spring before planting, and the application of herbicides was too late to substantially affect many weeds. Strong weed growth in combination with an exceptionally dry spring season in 2011 retarded the growth of both poplars and willows. If SRC is to be cultivated at a larger scale, farmers will in all likelihood need to follow SRC establishment guidelines to keep young crops weed free (Tubby and Armstrong, 2002; Caslin et al., 2010). This will inevitably decrease considerably the value of SRC as habitat for vascular plant species during the first years after planting (Gustafsson, 1987; Cunningham et al., 2004).
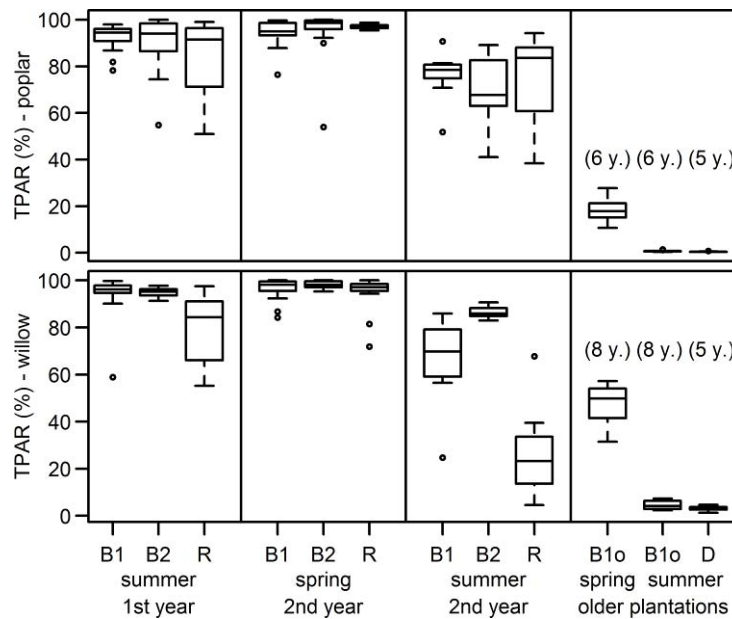
Fig. 4.1 PAR transmittance (TPAR, in %) in poplar and willow SRC in central Germany. See Table 4.1 for site descriptions (abbreviations).

The narrow C/N-ratio and the low soil organic matter content (Table 4.2) in the SRCs also resembled conditions on arable land (C. Seifert, C. Leuschner & H. Culmsee, unpublished data). The C/N-ratio was lower than values measured in older SRC in northern Germany and Sweden (Baum et al., 2012b). The availability of phosphorus for plants was within the lower range of values usually found on arable fields in the study regions (C. Seifert, C. Leuschner & H. Culmsee, unpublished data), as no P fertilisation took place in the plantations.

Table 4.2 Soil chemical properties of SRC in central Germany.

Site	Tree species	pH	Base saturation (%)	Plant-available P (μmol/gDW)	C/N-ratio (g/g)	Soil organic matter (%)
BERTA I	Poplar	7.5±0.1	99.95±0.04	0.7±0.1	8.64±0.38	2.95±0.16
	Willow	7.6±0.1	99.97±0.03	0.7±0.1	8.44±0.34	2.91±0.11
BERTA II	Poplar	7.4±0.2	99.93±0.03	0.7±0.1	8.83±0.03	2.85±0.07
	Willow	7.5±0.2	99.99±0.00	0.6±0.1	8.93±0.32	3.00±0.17
Reiffenhausen	Poplar	5.3±0.1	99.78±0.05	1.8±0.2	9.90±0.16	2.01±0.04
	Willow	5.3±0.2	98.97±0.46	2.0±0.2	9.25±0.24	2.78±0.09
BERTA I (old stands)	Poplar	7.5±0.1	99.80±0.12	0.8±0.1	7.40±0.28	2.82±0.06
	Willow	7.4±0.1	99.98±0.02	1.2±0.1	9.06±0.10	3.90±0.07

Values are the mean (±SE) of three samples per SRC. DW: dry weight. The first three sites refer to 1-year-old SRC.

Plant species richness and community composition were found to differ strongly between SRCs of different age classes (Fig. 4.2). The plant communities and the succession patterns found in the newly

established SRC were typical of first- and second-year fallows (Steffan-Dewenter and Tscharntke, 1997; Tscharntke et al., 2011). On average, 25 species were recorded per 75 m² in the poplar and 28 species in the willow plantations (appendix 4.A). More than 80% were annuals associated with arable land, probably mostly emerging from the seed bank. The remaining 20% were species associated with nitrophilous herbaceous vegetation or habitat generalists. The average species richness increased to 38 species in the poplar and 33 species in the willow plantations in the second year. This result is in accordance with earlier studies reviewed by Baum et al. (2009), who found species numbers, and the number of annual species, to increase during the first 2 years and to decrease thereafter with increasing closure of the tree canopy. The additional species found during the second year were mainly perennial species associated with nitrophilous herbaceous vegetation, grassland or forest (saplings of woody species, no forest herbs), or perennial habitat generalists (Fig. 4.2, cf. Fry and Slater, 2009). The number of indigenous species also increased remarkably from the first to the second year (Fig. 4.3). By contrast, the number of archaeophytes remained stable or decreased slightly. No species red listed for Germany were found in the SRC. Several other studies of SRC in the Netherlands, Germany and Sweden also recorded no or only few rare or red listed species (Vonk, 2008; Baum et al., 2012b). In clear contrast, to this, however, Delarze and Ciardo (2002) found 18 red list species in a newly established SRC in Switzerland.
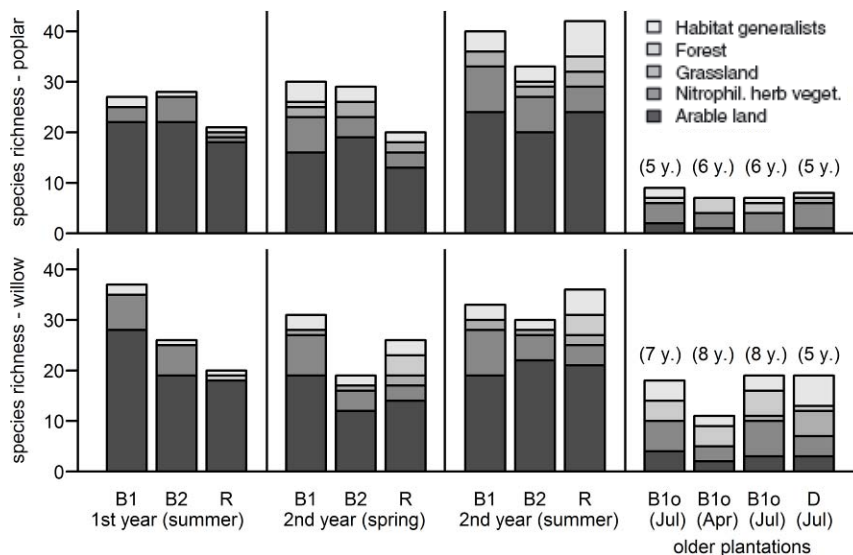
Fig. 4.2 Number of vascular plant species found in SRC in central Germany by habitat preference of the species. For site descriptions (abbreviations), see Table 4.1. The area sampled per SRC was 75 m².

These general trends were consistent across all three sites with newly established SRC, for both poplar and willow stands, although the soil analyses indicated differences in site conditions between the plantations in the Thüringer Becken and those in the Niedersächsisches Hügelland (Table 4.2). The

Reiffenhausen SRC was situated on middle Buntsandstein sediments, whereas the SRC in Thüringen were planted at sites on Pleistocene loess deposits. The soil of the Reiffenhausen SRC was more acidic (pH = 5.3) than in the Thüringer Becken (pH ~7.5). The base saturation was very high in all soils under SRC (> 99%) and was in the upper range of values measured by Baum et al. (2012b) in SRC on acidic arable soils in northern Germany and Sweden.
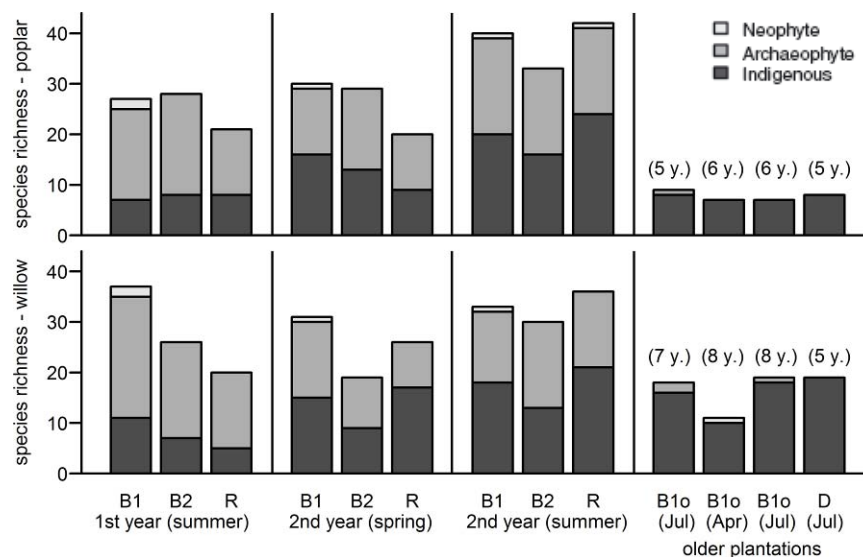
Fig. 4.3 Number of vascular plant species found in SRC in central Germany by naturalisation status. For site descriptions (abbreviations), see Table 4.1. The area sampled area per SRC was 75 m².

4.3.2 Habitat conditions and plant diversity in older short rotation coppice

Species richness in the older SRC with a closed canopy was much lower than in the first- and second-year plantations. In summer, 79 species per 75 m² were found in the old poplar plantations (PAR transmittance 0.5–0.7 %) and 18–19 species in the old willow SRC (PAR transmittance 3.2–4.6 %; Figures 4.1 and 4.2). The share of annual species associated with arable land decreased to <20%. The species numbers in the older poplar plantations were within the upper part of the range of values recorded on intensively managed arable land (Meyer et al., 2013; Seifert et al., 2014) and light availability on the ground was lower than in mature spruce forests (Ellenberg and Leuschner, 2010). The total herb cover value in the old poplar plantations did not exceed 5%, apart from one exception where stinging nettle (*Urtica dioica*) covered almost 20% of the sampling plot. The recorded species numbers in the old poplar plantations were found to be considerably lower than numbers recorded by Glaser and Schmidt (2010) who found on average 17 (clone Beaupré, PAR transmittance of 25–35%, dominance of few highly competitive species) and 27 species 100 m² (clone Max 1, PAR transmittance of 5–7%) in two poplar plantations of unknown age, but not recently established, in

Sachsen. These differences can be explained by the far lower light transmittance in the poplar SRC surveyed as part of this study. The old willow SRC stands had a wider row spacing (Table 4.1) and correspondingly higher PAR transmittance, species richness and understory vegetation cover. Nevertheless, the light availability in the willow SRC was still in the lower range of values measured in mature forests (Ellenberg and Leuschner, 2010) and in maize fields (Seifert et al., 2014). The most dominant species in the willow SRC with cover values of up to 75% was couch grass (*Elymus repens*), a noxious weed species feared by farmers. Other important species were stinging nettle (*Urtica dioica*), hedge bindweed (*Calystegia sepium*), creeping thistle (*Cirsium arvense*) and cleavers (*Galium aparine*), all common generalist weedy species of nitrophilous herbaceous vegetation and arable land. Vegetation records by other authors (Gustafsson, 1987; Britt et al., 2007; Glaser and Schmidt, 2010; Baum et al., 2012b) confirmed the finding that mature SRC is commonly dominated by a small set of generalist perennial herbs with no conservation status. In the older plantations, the importance of archaeophytes of arable habitats decreased drastically as compared to first- or second-year stands (Fig. 4.3). The transition pattern over time, from annual species associated with arable land to perennials associated with various habitat types, including forest species, concurred with findings by other authors and is typical for SRC (Delarze and Ciardo, 2002; Britt et al., 2007; Baum et al., 2012a). Species richness in spring, before the leaf flush, was found to be as low (poplar) or about 40% lower (willow) than in summer (Fig. 4.2). Contrary to findings by Gustafsson (1987) from older SRC in Sweden, no typical 'spring' species (e.g. forest geophytes or 'early bird' winter annuals) were found, and all species present in spring persisted also during summer. These differences may, however, have been the result of different land use histories and landscape settings between the sites in the two studies.

Habitat conditions in the older SRC were not directly comparable to any other land use type. PAR transmittance in these plantations was drastically reduced and similar to very dense stands of young trees during initial phases of forest regeneration (Dölle and Schmidt, 2009). The C/N-ratio and the soil organic matter content in the upper mineral soil layer (Table 4.2) were similar to the values measured on arable fields (compare chapter 2) and narrower (C/N) and lower (organic matter) than characteristic values from forest soils (Ellenberg and Leuschner, 2010; Yang and Luo, 2011). Due to the dense canopy, only few of the species associated with open habitats were able to persist. Herbaceous forest species were not found either (with the exception of wind dispersed milletgrass, *Milium effusum*), as the respective propagules were not present at the sites due to the long standing agricultural use. Stand age and distance to the next forest are the most important factors determining the potential for forest herbs to colonise new SRC (Verheyen et al., 2003). Colonisation from nearby forests (2.0–3.5 km away) is likely to take much longer than 5–8 years. The timeframe of 20–30 years, which is often cited as the maximum age for SRC (Caslin et al., 2010), might not be enough to allow for the establishment of many herbaceous forest species of high conservation value particularly in agricultural landscapes with little forest cover (Gustafsson, 1987; Hardcastle, 2006). Therefore, and due to the dramatic pace

of biodiversity loss in agricultural habitats, it seems appropriate to focus on optimising habitat conditions in SRC for species associated with cultural landscapes and farmland habitats, rather than for forest species.

4.3.3 Implications for plant species conservation

The data presented in this paper, in conjunction with results from previous studies, indicated that the suitability of SRC as habitat for vascular plant species in agricultural landscapes varies widely with the age of the plantation (cf. Baum et al., 2012b) and with the intensity of SRC management. When assessing whether establishing a SRC will benefit plant diversity, it is essential to consider the type of land use being replaced and the kind of SRC replacing it. Fig. 4.4 represents an attempt to place SRC in the context of other common agricultural land use types with regard to management intensity and its potential contribution to maintaining plant diversity. Low-input, structurally diverse SRC assumes an intermediate position between multispecies hedges and copses and high-input SRC in terms of its ability to support plant diversity. It has the highest potential to be both economically and ecologically viable. Replacing the few remaining areas of agricultural landscape with land use types that support very high plant diversity, such as extensively managed grasslands, croplands and fallows with SRC would decrease plant diversity, whereas planting low-input SRC on intensively managed arable land has the potential to increase the plant diversity of agricultural landscapes. It must be stressed that the management intensity of arable land, grassland and woody landscape elements is
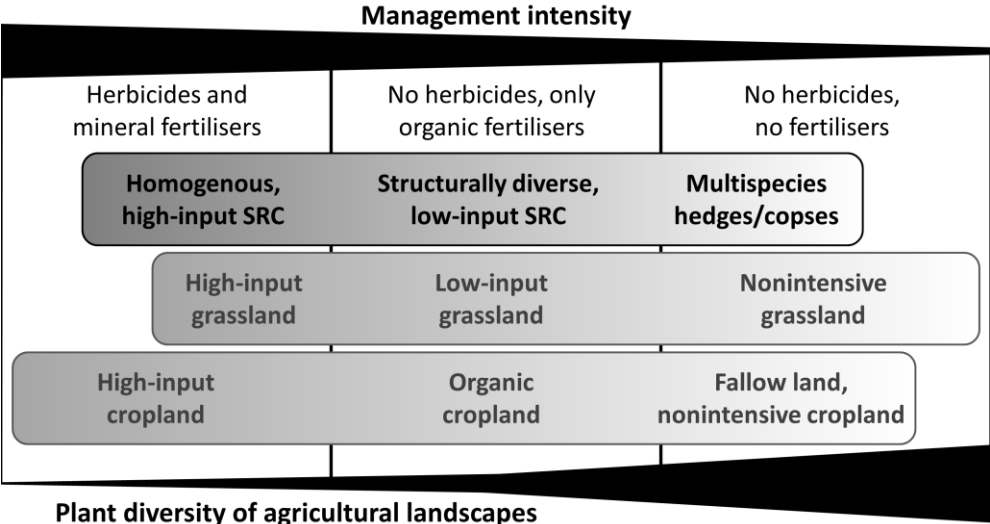
Fig. 4.4 Value of SRC for plant diversity in agricultural landscapes relative to other land use options and management intensity (based on Hoogeveen et al., 2005; Gabriel et al., 2013). Maximum diversity is expected to be achieved where a regionally adapted mosaic of woody landscape elements, grassland and arable land exists.

a continuum and that effects of reductions in management intensity can vary widely under different environmental conditions. Therefore, Fig.4.4 is necessarily a simplification. Additionally, the target species of conservation efforts, or group of species, must be taken into consideration. Rare arable archaeophytes will not find suitable conditions after canopy closure; these are better served by nonintensive cropping or regularly renewed fallows. In landscapes with few structural elements, where large fields and high-input cropping systems prevail due to the presence of fertile soils, SRC can contribute to maintaining a basic level of plant diversity and serve as refuges and feeding resources for species from higher trophic levels, such as insects and birds. This complexity implies that management decisions aimed at enhancing habitat diversity must always be site specific.

4.3.4 Management recommendations for the enhancement of plant diversity

The management of SRC can differ widely from homogenous, high-input plantations to structurally diverse, low-input coppices (Gustafsson, 1987; Cunningham et al., 2004; Baum et al., 2009; Fig. 4.4). In general, from the perspective of plant species conservation, the characterisation 'high input' may still apply at considerably lower fertilisation and herbicide application rates than commonly occur on conventional cropland (cf. Storkey et al., 2012; Gabriel et al., 2013). The SRCs surveyed in this study were situated midway along the continuum between these two extremes, being 'low input' but structurally homogeneous. Factors influencing the value of SRC for plant diversity are (i) the amounts and timing of fertiliser input and herbicide application, (ii) the diversity of tree species and varieties planted, (iii) the temporal and spatial heterogeneity of habitat conditions in SRC (Delarze and Ciardo, 2002; NABU, 2008; Weih, 2008; Glaser and Schmidt, 2010; Baum et al., 2012b) and (iv) the structure and complexity of the surrounding landscape (Baum et al., 2012b). The data presented in this chapter indicated that first- and second-year SRC can be a valuable habitat for farmland plant species, where little or no herbicides and no fertiliser are applied. Cunningham et al. (2004) showed that commercial SRC with higher herbicide and fertiliser inputs has lower vegetation cover and species richness than SRC planted for non-commercial research purposes.

SRC stands with a higher degree of tree variability might habitat con provide more diverse habitat conditions, also for the accompanying flora as tree species, for instance, differ in terms of phenology and leaf area index, support different phytophagous species and produce material with different degrees of decomposability. Planting more than one genotype per SRC is the most basic step towards increasing crop biodiversity and is strongly recommended for phytosanitary reasons (Caslin et al., 2010; Georgi et al., 2013; Georgi and Müller, 2013). Using several species of fast-growing trees and interspersing indigenous trees and shrubs in the SRC can further increase the biodiversity value (NABU, 2008; Unseld et al., 2011).

In order to maximise the variability of habitat conditions in SRC, four management decisions are of particular importance. Firstly, tree spacing and canopy density influence PAR transmittance and herb diversity (Delarze and Ciardo, 2002). The differences in plant diversity between the older poplar and the willow SRCs presented here were at least partially due to differences in row spacing and light availability in the understory. Irregular spacing and the occasional introduction of canopy openings can further increase habitat diversity. The second important factor is the length of the harvesting cycle (Gustafsson, 1987; Baum et al., 2012b). Short harvesting cycles create open habitat conditions more frequently, increasing plant diversity. Several authors recommended rotation times of three years for willow SRC (Gustafsson, 1987; Caslin et al., 2010). Thirdly, harvesting parts of the plantation in different years increases the diversity of growth stages within the SRC and correspondingly broadens the range of site conditions found in the plantation (Baum et al., 2009). Surveys from traditional German coppice woodlands show that this mosaic of of different growth stages is important to maintain higher degrees of phytodiversity (Conrady, 2007). To our knowledge, no records of plant species diversity directly after harvesting have been published yet. Baum et al. (2012a) showed, however, that species composition varied with the total age of the plantation rather than with the time since the last harvest. Generally, competitive perennial species such as couch grass are likely to have spread by the time of the first harvest (Gustafsson, 1987; Baum et al., 2012a; Baum et al., 2012b) and threaten to suppress less competitive annuals or archaeophytes even after harvest. The
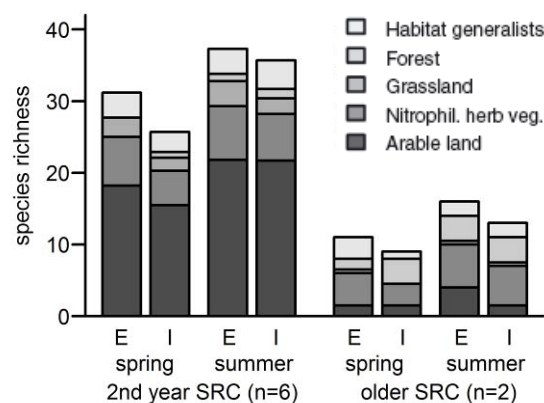
Fig. 4.5 Number of vascular plant species found at the edges (E) and in the interiors (I) of SRC in central Germany by habitat preference of the species. Species numbers per 75 m² were averaged across all of the studied SRC of an age class, with stands of willow and poplar pooled.

fourth point, which is of particular importance, is the size of the plantation and its edge length (Unseld et al., 2011). The edges of SRC were by 4–19% richer in species than the corresponding stand interior (Fig. 4.5). The occurrence of plant species of arable land, grasslands and nitrophilous herb vegetation showed a tendency to increase towards the edge, while forest species decreased. This is in accordance

with the findings by Cunningham et al. (2004), who found species richness to decline from the edge towards the centre of the SRC during the first 2 years after establishment. Smaller SRC with longer edges also contributed more strongly to biodiversity conservation than larger, homogenous plantations in previous studies (Cunningham et al., 2004; NABU, 2008; Weih, 2008; Baum et al., 2009). Agroforestry systems in which SRC strips alternate with grassland or cropland are an interesting option for maximising edge length and they can even increase the revenue per hectare (Unseld et al., 2011; Bärwolff et al., 2011; Bärwolff et al., 2012; Hartmann and Lamersdorf, 2015).

4.4 Conclusions

SRC can be either beneficial or detrimental to plant species diversity on the landscape scale, depending on the targeted species, the management intensity of the SRC, the land use type being replaced and the surrounding landscape context (Fig. 4.4). The habitat conditions for vascular plants prevailing in SRC vary extremely quickly, particularly in the early part of each harvesting cycle, making it difficult for habitat specialists to persist (Delarze and Ciardo, 2002). Consequently, older SRC was dominated by a limited set of generalist species, and all of the younger and older SRC stands investigated completely lacked red list plant species. SRC is unlikely to attain the status of habitat of high nature conservation value. The conversion of ecologically valuable habitats, such as permanent grasslands and other man-made European Natura habitat types, to SRC should, therefore, be avoided (BfN, 2010). For economic reasons, it is likely that SRC and other second-generation energy crops, such as *Miscanthus* (Harvolk et al., 2014), will preferentially be planted on marginal sites and not on very fertile soils where high yields can be obtained with annual crops. This has become even more valid since the onset of the bioenergy boom, with high pressure even on marginal arable lands, where the last strongholds of agro-biodiversity remain. Depending on the previous use of these lands, the conversion of marginal land to SRC will often conflict with biodiversity conservation. However, narrow SRC strips dividing large, uniform arable fields can contribute to the conservation of plant species diversity in landscapes dominated by high-input agriculture. Such landscape-compatible stands (BfN, 2012) can enhance landscape structure and habitat connectivity. By contrast, large-scale, homogenous SRC cannot be expected to enhance plant species diversity, even in intensively managed landscapes. Potential governmental subsidies for SRC (i.e. in the form of agri-environment measures or the recognition as ecological focus areas) should only support structurally diverse, low-input SRC. Additionally, schemes should allow for site-specific decision making in order to avoid the conversion of valuable, species rich farmland habitats to SRC.

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Chapter 5

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Synthesis

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5.1 Influences of energy cropping on the plant diversity of agricultural landscapes

5.1.1 Annual energy crops (maize, oilseed rape)

5.1.1.1 Research context and approaches

The assessments of the biodiversity impacts of annual energy crops presented in this thesis mainly focussed on maize cultivation for biogas. This focus was chosen to reflect current research priorities. The rapid expansion of maize cultivation has caused major concerns among scientists, conservationists and also among the wider public about negative ecological impacts of energy cropping. The concerns are fuelled by the fact that in regions with a strong focus on livestock farming the agricultural landscape is today characterized by maize production. In north-west Germany, the share of maize of the total arable land typically exceeds 50% and can reach up to 76% (DMK, 2010a). This equals shares of 30–48% of the total agriculturally used area (including grasslands) in these regions (DMK, 2010b). Tall-growing maize is characterizing the landscapes and these rapid changes in crop frequencies are widely perceived by the public. Oilseed rape, the second important energy crop in Germany, is not reaching such high densities in any German region. A theoretical maximum of 25–33% of the arable land can be cultivated with oilseed rape each year as this crop cannot be grown repeatedly without interruptions on the same field. Additionally, it is not tall-growing and its beautiful flowering aspect in late spring enhances recreational landscape quality for people. Concerns about negative ecological impacts of oilseed rape production are therefore much less commonly voiced, although for instance Diekötter et al. (2010) found oilseed rape production to distort plant-pollinator interactions in the landscape surrounding the fields. We therefore also, additionally, included oilseed rape in the assessments in chapter 3 to account for its importance in the study areas. All assessments were carried out in two study regions in Central Germany, the Lower Saxon hills and the Thuringian Basin, both characterized by fertile arable soils. These regions can be regarded as representative of landscapes with a high prevalence of arable farming. All surveys were done in fields managed conventionally or under an agri-environmental scheme as part of the usual farming activities in the study areas. Vegetation surveys, soil sampling and measurements of PAR transmissivity took place in mature crop stands approximately 1–6 weeks before harvesting of the specific crop.

5.1.1.2   Key findings

Arable plant diversity and growing conditions in the field interior and at the field margins

We found the field interior of all crops, energy crops as well as food and/or fodder crops, under conventional, high-input management to be extremely species poor. We recorded on average 6 species 100 m^{-2} in maize (n = 71), 3 species 100 m^{-2} in winter-sown wheat and barley (n = 71 and 64 respectively) and 5 species 100 m^{-2} in oilseed rape (n = 10). Other studies from Central Germany also found the field interior to be extremely impoverished with regard to arable plant diversity. Meyer et al. (2013) found on average 6 species 100 m^{-2} in conventionally managed fields of different crops on loamy soils. These strikingly low numbers, in conjunction with the low cover values recorded (median across all crops < 0.5% total weed cover), indicate that the interior of arable fields must represent an extremely impoverished habitat for insects, birds and mammals. In accordance with Parish et al. (2009), this illustrates that today far less than 1% of the total summer biomass of high-input arable fields consists of non-crop arable plants. Only those animals able to survive by solely feeding on and reproducing in the crop itself, i.e. crop pests, can persist in such homogeneous environments. Most animals, including many pest predators, depend upon more heterogeneous habitats in terms of floristic diversity.

A particularly concerning result of this study was that we found the margins of maize fields to be even less species rich (on average 11 species 100 m^{-2}, n = 71) than the margins of fields with winter-sown wheat (15 species 100 m^{-2}, n = 71). Winter-sown barley exhibited intermediate species richness with 13 species 100 m^{-2} (n = 64), whereas oilseed rape harboured on average 15 species 100 m^{-2} (n = 10). These results are largely in accordance with Waldhardt et al. (2011) who found on average 18 species 100 m^{-2} on maize field margins, but 27 species in winter-sown wheat, 30 species in winter-sown barley and 28 species in oilseed rape in Southern Hesse. Differences in overall species numbers between the two studies are thought to originate from differences in the regional species pools between Southern Hesse, the Lower Saxon hills and the Thuringian Basin.

Field margins of maize fields were also found to be more strongly shaded than margins of other crop fields. PAR transmissivity to the ground reached on average only 11% in maize field margins, but 18% in field margins of winter-sown wheat and 19% in winter-sown barley and oilseed rape. The shading effect of the crop was found to explain a significant part ($R^2_{adjusted} = 7.7\%$) of the plant species richness at the field margins. As opposed to the field interior, field margins were found to be suitable habitats for a wide variety of arable plant species. Although they represent only approximately 4% of the total cropland (Hötker & Leuschner, 2014), field margins are the last refuges for a wide range of arable plants (Marshall & Moonen, 2002; Fried et al., 2009). The effects of the expansion of maize cropping on the arable plant diversity of field margins are therefore particularly important at the landscape scale with regard to the maintenance of arable plant diversity.

Growing conditions for arable plants in the field interior are worse than at the field margins due to a range of different factors concerning the field management and related abiotic factors. Contrary to the field margins, the field interior receives the full fertiliser and herbicide doses and is not influenced as strongly by adjacent, potentially more species rich, habitat types (Wilson & Aebischer, 1995). The crop cover in the field interior and concomitantly also the yields are therefore typically higher than at the field margins (Sparkes et al., 1998). Consequently, the light transmissivity is lower in the field interior, with maize having the strongest shading effect. On average 6% of the incoming PAR reached the ground in maize fields, 13% in oilseed rape, 11% in barley and 10% in wheat fields. Differences between the four assessed crops explained 23.4% ($R^2_{adjusted}$) of the variation in PAR transmissivity at the field level. Maize fields were found to be as dark as fertile grasslands and mature spruce forests, whereas the stands of the other three crops were less shading than temperate grasslands or closed broad-leaved forests (Tappeiner & Cernusca, 1989; Lane et al., 2000; Ellenberg & Leuschner, 2010). All conventionally managed crop stands were far more shading than wheat stands managed extensively (i.e. without fertilisation and herbicides applications) according to an agri-environmental scheme with on average 57% PAR transmissivity. Conventionally managed, high-input crop stands are nowadays probably too dark for many typical light-demanding arable plant species. Considering that the "field interior" is representative of about 96 % of the total arable land (Hötker & Leuschner, 2014), and that > 90 % of the arable land in Germany are managed conventionally with annual crops or vegetables (excluding fallow arable land, perennial (clover) leys and organic annual crops and vegetables) (Destatis, 2013, 2014), the potentially available habitat for many arable species has been reduced severely during the past 60 years.

The influence of crop choice and field management

Strong and diverse inter-relationships between different factors affecting arable plant diversity make it extremely difficult to quantify the importance of single predictors. The light transmissivity, for instance, is influenced by the crop stand density which in turn is affected by the choice of crop, the amounts of fertilisation, the herbicide applications and the soil and climatic conditions at the site. All these factors also affect arable plant growth either directly or indirectly by influencing other predictors. With the help of variation partitioning, we aimed to gain insights into this complex network of interdependencies. A complete model with all examined field management, environmental and spatial factors explained 40 and 32 % of the variation in species richness and 42 and 36 % of the variation in community composition for the field interior and the field margin, respectively. These values are high as compared to previous studies by Pinke et al. (2012). The fact that even with a very comprehensive set of predictors only less than 50% of the variation in arable plant diversity can be explained, illustrates the difficulty of devising simple, straightforward management recommendations for enhancing arable plant diversity within conventional crop systems. The complex interactions of

different predictors in agroecosystems and the noise which is necessarily generated in the data collection process define the limits of current statistical approaches for this type of assessments.

Field management factors were generally found to be more important than environmental factors when explaining variation in species richness (field interior) and in weed cover (field interior and margin). On the other hand, they were less important than environmental factors for species richness at the field margins and for community composition in the interior and at the margin. These results confirm the hypothesis that field management affects species richness and abundance more strongly than community composition. The latter is thought to be driven more strongly by the environmental conditions which define the local species pools and to respond to changes in field management at much larger time-scales. It also confirms that field management factors exert a stronger influence in the field interior as opposed to the field margin. At the field margins, the explanatory power of the management factors was by 49–86% smaller than in the field interior for the three response variables. Overall, 6–48% of the total explained variation in arable plant diversity was attributable to field management factors, whereas 0–49% was due to varying environmental conditions. The remainder was either attributable to spatial factors or could not be attributed to any of these three groups.

The current crop, fertilisation and to a lesser extent also herbicide use as well as soil characteristics were found to be the most important predictors of arable plant diversity in Central European high-input croplands, apart from spatial factors. Weed cover, species richness and community composition responded to different sets of factors. The current crop was found to potentially explain a substantial amount of the variation in all three response variables. The effect of crop type is thought to originate from crop-specific management practices relating to the timing, quality and quantity of cultivation, fertilisation and herbicide use. Supporting this hypothesis, the crop effects were found to be interrelated with sowing date, fertilisation, herbicide use and crop rotation. Changes in cropping patterns at the landscape scale, such as occurring during the current bioenergy boom, will therefore affect arable plant diversity. The impacts can either be positive or negative. A balanced mixture of maize and winter cereals offers habitats to a wider range of arable plant species than any of these crops alone. On the other hand, the results presented in this thesis also clearly show that the effects of increasing crop diversity on conventionally managed arable land are only of secondary importance as compared to the effects of substantially decreasing land use intensity. Extensively managed fields without fertiliser or herbicides use on nutrient-poor soils showed a much higher plant diversity (on average 21 species 100 m^{-2} in the field interior and 33 species 100 m^{-2} at the field margins, including the presence of rare and endangered species) than conventionally managed fields. The type of field management (conventional vs. extensive) alone explained 80% (gross effect, $R^2_{adjusted}$, n = 20) of the variations in species richness among fields of winter-sown wheat, whereas the current crop was found to explain only 10% (gross effect, $R^2_{adjusted}$, n = 206, no significant gross effect of crop type found at n = 40) of the variations in species richness among conventionally managed fields.

Considerable differences in field management were detected between maize, winter-sown wheat and winter-sown barley. Maize fields received about 2–5 times more organic fertilisers (on average 23.6 Mg/ha*year, including animal manures and digested residues from biogas plants) than winter cereal fields. As a consequence, the added doses of P, K, Ca and Mg were 2–6 times higher in maize fields. N-input was not higher on maize fields as compared to winter cereal fields, as higher N-inputs from organic fertilisers were balanced with lower mineral N-fertilisation. As maize growth benefits from high P-doses and the reduction potential with regard to mineral fertilisation is limited, P-fertilisation in maize fields was found to be much higher than in cereal fields. These values are highly alarming. Doses of >100 kg P_2O_5 ha^{-1} yr^{-1}, as recorded for almost 50% of the maize fields, are highly likely to exceed the nutrient removal with maize harvest and can lead to substantial P-leaching to the groundwater with negative effects on a wide range of ecosystems (Sibbesen & Runge-Metzger, 1995; Sibbesen & Sharpley, 1997; LWK NRW, 2012).

The herbicide use intensity index (HI) in the study regions was found to be comparable to values found by more comprehensive assessments across Germany (Roßberg, 2011). In accordance with these assessments, herbicide use intensity was found to be higher in maize and winter-sown wheat than in winter-sown barley. The implications of the use of different active ingredients are hard to discern in detail, but our analyses highlight the importance of such assessments. Certain groups of photosynthesis inhibitors at PS II (C_1, C_3), in our study area mainly represented by terbuthylazine and bromoxynil, are commonly applied in maize cultures, but are not suitable for winter cereals. Group C_2 (i.e. isoproturon) on the other hand is suitable for winter cereals, but not for maize. We found the amount of terbuthylazine (C_1) to potentially affect community composition. In landscapes where maize has become the dominant crop, as for example in certain parts of north-west Germany, such shifts in the herbicides' mechanisms of action can lead to durable shifts in community composition if applied over longer time periods.

Fertilisation was found to be a promising leverage for enhancing arable plant diversity as it was shown to be related to weed cover, species richness and community composition. The quality and quantity of herbicide applications was also shown to be potentially important. Yet, no net effects of changes in nitrogen and phosphorus fertilisation rates were found after partialling out the effects of other management factors. Likewise it could not be discerned why certain groups of herbicides, i.e. mitosis inhibitors (HRAC group K_3), showed dose-dependent effects on species richness in the field interior, whereas others did not (compare chapter 2). The fact that we did not find clearer dose-dependent effects of fertilisers and herbicides on arable plant diversity maybe due to overall levels of land use intensity being comparatively high on all conventionally managed fields. It has been shown that biodiversity does not respond linearly to changes in land use intensity (Kleijn et al., 2009). There may be thresholds above which a reduction in fertilisation or herbicide use will not affect biodiversity to any measurable extent. Fertiliser and herbicide applications on the surveyed conventionally managed

fields are likely to have been exceeding such thresholds for many years. Kleijn et al. (2009) found that plant species richness on arable land increased markedly with reductions in fertiliser input only below doses of 100 kg N ha^{-1} yr^{-1}. Very little change was observed at doses >150 kg N ha^{-1} yr^{-1}, as were applied on 93% of the surveyed fields in the present study during the year of survey or the preceding year. Herbicide applications might exacerbate this effect, as farmers might follow a 'zero-tolerance' strategy with regard to weed infestation independent of the fertilisation rate. As it is difficult to determine optimal application rates for weed management, unnecessary 'insurance' applications of herbicides are also thought to be a common phenomenon complicating the assessment. Our results underline that it will hardly be possible to slow the rapid erosion of biodiversity in agricultural landscapes by solely relying on approaches which reduce land use intensity, but stay within the range of currently practised conventional management techniques.

5.1.2 Short rotation coppices

5.1.2.1   *Research context and approaches*

The second part of the present thesis focused on alternative, 'second generation' energy feedstocks. Short rotation coppices (SRCs), i.e. fast growing trees grown on farmland and repeatedly harvested in cycles of 3–7 (–20) years, are a particularly promising energy feedstock for Central Europe. The expected energy yields per hectare and year can potentially be equally high or even higher than energy yields from the anaerobic digestion of maize (SRU, 2007). As opposed to most other second generation energy crops, native tree species, such as certain native willows and poplars are suitable for cultivation as SRCs. As perennial crops, overall fertiliser and herbicide applications in SRCs are generally expected to be lower than in annual energy crops. SRCs are therefore commonly presented as an ecologically friendly alternative to first generation, annual energy crops. With regard to the effects on biodiversity and more specifically on farmland plant diversity, the impacts of a potential large-scale establishment of SRC are less clear and are therefore in the focus of chapter 4 of this thesis. All assessments presented here are based on vegetation, soil and light transmissivity surveys in 10 SRCs (5 × willow, 5 × poplar) of different age classes in the Lower Saxon hills and the Thuringian Basin established on former arable land.

5.1.2.2   *Key findings*

The results clearly confirm the hypothesis that the habitat conditions and concomitantly also the flora of SCRs changes with the age of the plantation. Habitat conditions in the recently (in 2011) established SRCs on arable land were shown to be similar to habitat conditions on fallow arable land, as long a no or only little herbicides and fertilisers are applied on the coppice. The low tree cover

(1%), the high PAR transmissivity (on average 89%), the low soil organic matter content (2.8%) and the narrow C/N-ratio (9.0) in the upper soil layer in first year SRCs were found to be comparable to arable land which was recently left fallow. Under these conditions relatively species rich plant communities with a high proportion of arable plants (natives as well as archaeophytes) developed. Species richness increased from the first to the second year after establishment from on average 27 species 75 m^{-2} to 36 species 75 m^{-2}. Tree cover increased as well, but remained mostly clearly below 20%. PAR transmissivity decreased to 66%. Community composition on the first and second year SRCs resembled plant assemblages typical for early successional fallow land (Steffan-Dewenter & Tscharntke, 1997; Tscharntke et al., 2011). No rare or red-listed species were detected on the coppices.

Species richness in the 5–8 year old willow and poplar plantations was drastically reduced as opposed to the establishment phase. On average 8 species 75 m^{-2} were recorded in the poplar plantations and 19 species 75 m^{-2} in the willow coppices. Tree row spacing was wider and light transmissivity higher in the willow coppices (row spacing > 1.5 m; PAR transmissivity approximately 4%) as opposed to the poplar stands (≤ 1m; <1%). Mature poplar plantations were found to be darker than maize fields and mature spruce forests and nearly as species poor as arable land (compare chapter 2; Ellenberg & Leuschner, 2010). The community composition also changed drastically. The importance of arable plant species and archaeophytes decreased strongly. The remaining plant species were in their majority associated with nitrophilous herbaceous vegetation or were habitat generalists. Couch grass (*Elymus repens*) was the dominant species in the willow coppices. Other important species were stinging nettle (*Urtica dioica*), hedge bindweed (*Calystegia sepium*), creeping thistle (*Cirsium arvense*) and cleavers (*Galium aparine*), all common generalist weedy species of nitrophilous herbaceous vegetation and arable land. Only one forest species, the wind dispersed milletgrass (*Milium effusum*), was recorded at one site. Colonisation by most forest species is likely to take much longer than 5–8 years, despite the nearest forests being only 2.0–3.5 km away. Even the time frame of 20–30 years, which is often cited as the maximum age for SRCs (Caslin et al., 2010), might not be enough to allow for the establishment of many herbaceous forest species of high conservation value particularly in agricultural landscapes with little forest cover (Gustafsson, 1987; Hardcastle, 2006). Therefore, and due to the dramatic pace of biodiversity loss in agricultural habitats, it seems appropriate to focus on optimising habitat conditions in SRCs for species associated with cultural landscapes and farmland habitats, rather than for forest species.

Based on these findings and a wider literature review management recommendations for SRCs were given in order to enhance their value as habitats for vascular plants. These recommendations include: (1) avoiding the use of herbicides and fertilisation; (2) cultivating several tree species with a number of different intra-specific genotypes and interspersing other native trees and shrubs; (3) increasing the variability in habitat conditions in time and space in the coppices by choosing a wide tree spacing, using a short harvesting cycle, partially harvesting the coppice at irregular time-steps, decreasing

plantation size to ≤ 1 ha and maximising plantation edge length (Delarze & Ciardo, 2002; NABU, 2008; Weih, 2008; Baum et al., 2012).

5.1.3 Short rotation coppices – a biodiversity friendly alternative to annual energy crops?

The question set out in the headline will be answered here with regard to the impacts of both energy cropping systems on the plant diversity of agricultural landscapes. Other taxonomic groups and the wider ecological contexts such as the impacts on soil organic matter accumulation, nutrient leaching, soil erosion and greenhouse gas emission will not be discussed as they are partially still a matter of scientific debate and were not addressed in this thesis.

As outlined in chapters 2 and 3, maize and oilseed rape cultivation as currently practiced on conventional farms are similarly detrimental for farmland plant diversity as the cultivation of winter-sown wheat or barley. The field interior of all four crops was found to be extremely species poor. At the margins, maize fields even showed the lowest species richness of all tested crops, likely due to lower light availability.

The results presented in chapter 4 show on the other hand that very dense, structurally homogeneous SRCs can be nearly as species poor as conventionally managed maize or oilseed rape fields. These SRCs were also found to be much more shading than maize fields or even old-growth forests, making them completely unsuitable as habitats for most farmland species. There is a range of easily implemented measures to decrease canopy density, increase the structural diversity and improve the quality of SRCs as habitats for farmland plant species. Adopting these measures will, however, in many cases inevitably result in yield reductions. Consequently, it seems optimistic to expect SRCs to contribute strongly to enhancing the plant diversity of agricultural landscapes without imposing additional requirements. Even less densely planted SRCs, such as the willow coppices surveyed for this study, were shown to mostly provide habitats to a range of common, generalist and mostly problematic weedy species. Their value for plant species conservation in agricultural landscapes is therefore also low, even if species richness is higher than in many maize, oilseed rape or winter cereal fields.

In intensively managed agricultural landscapes it can nevertheless be beneficial for biodiversity to intersperse SRC among the annual energy crops. The concept of planting a number of narrow parallel SRC strips (e.g. 5–20 m width) in regular distances to divide large arable fields planted with annual crops seems particularly well suited for maximising the biodiversity benefits from SRC cultivation (Unseld et al., 2011). These 'agro-forestry systems' can help to increase the habitat diversity of homogeneous, structurally impoverished agricultural landscapes. SRCs are, however, not a suitable

substitute for the loss of fallow lands or grasslands. They can also not compensate for the current decline in conservation headlands and extensively managed arable land.

The establishment phase of SRC was shown to be the period most interesting from the farmland plant conservation perspective. If the use of herbicides and fertilisers is avoided during the establishment phase, species rich plant assemblages similar to early successional fallow land were shown to develop. Poplars and willows planted as stockings are unfortunately hardly able to compete with this spontaneous arable vegetation resulting in growth delays and initial yield declines if left unmanaged. Weed management can either be done semi-manually with small machines, or with herbicides. Under the scenario that SRCs are cultivated at larger scales, it seems likely that herbicides will be applied routinely during the establishment phase to aid tree establishment and reduce the need for labour intensive mechanical weeding. This would, however, considerably decrease the value of SRC as habitats for farmland plant species. Additional applications of fertiliser to maximise yields would further deteriorate the habitat quality.

Our results show that SRCs cannot be expected to be a suitable tool for halting the rapid erosion of farmland plant diversity. It seems rather necessary to direct their establishment towards intensively managed agricultural landscapes in order to achieve benefits for farmland plant diversity. Appropriate planning procedures also seem paramount to avoid negative impacts which are to be expected if fallow land, low-input grassland or extensively managed arable land is converted to SRC. These sites, making up a considerable proportion of what is commonly referred to as 'marginal land', are the last existing refuges for many plant species in contemporary agricultural landscapes. Their loss would likely counterbalance any positive biodiversity effects of SRC.

Finally, the question arises whether the expansion of SRC cultivation can actually be expected to lead to declines in energy maize or oilseed rape production. Wood chips from SRC are used for energy generation via combustion in combined heat and power plants. They are not suitable as feedstock for anaerobic digestion (biogas production) due to the high content of lignocellulosic compounds and can therefore not be regarded as a direct substitute for energy maize. As wood chips do not contain extractable amounts of vegetable oil, they can also not replace oilseed rape for biodiesel production. At short and intermediate timescales SRC can therefore not be expected to decrease annual energy crop production as biogas plants and biodiesel refineries will keep requiring appropriate feedstocks independent of the developments on the market for solid biofuels. SRCs might, however, contribute to slowing the expansion of annual energy cropping at intermediate timescales. In the long run anaerobic digestion and biodiesel production might even be replaced by more ecologically friendly conversion technologies, such as cogeneration units or biomass to liquid fuels (BtL). As many biogas plants have been established only recently, and are expected to have long operational life spans, there is nevertheless a considerable danger that the widespread cultivation of SRC would additionally increase

the demand for land for bioenergy feedstocks at least during the next two decades. The negative consequences of increasing land prices due to high demand are already noticeable in contemporary agricultural landscapes. Grasslands are being converted to cropland (BfN, 2014), fallow land is increasingly reconverted to production (Destatis, 2014) and the financial attractiveness of agri-environmental schemes dwindles (Meyer et al., 2010). In conclusion, it seems necessary to carefully consider all these issues when portraying SRC as a biodiversity friendly alternative to annual energy crop production.

5.1.4 General conclusions

In conclusion, the results of this study show clearly that the underlying root causes of the strong decline in arable plant diversity during the last decades extent far beyond energy cropping. Energy cropping was shown to potentially aggravate the situation further but can rather be seen as the 'tip of the iceberg'. The impacts of energy cropping also differ widely depending on the structure of the agricultural sector and the soil and climate conditions in different regions. Extremely high proportions of maize on arable land are found for instance in north-west Germany (up to 76%; DMK, 2010a), whereas the share of maize of the arable land did not exceed 10% in both study areas.

Negative ecological impacts of energy cropping, particularly of energy maize cultivation, are commonly addressed in the media and are therefore widely discussed. This study highlights that these discussions commonly overlook that the conventional production of winter-sown wheat can be just as intensive as the production of energy maize. Nitrogen fertilisation and herbicide use intensity were shown to be similarly high for maize and winter wheat. Excessive phosphorus fertilisation was, however, discerned as a threat to plant diversity which mainly applies to maize production, due to far higher inputs of organic fertilisers. Maize was also found to be more shading and consequently less species rich at the field margins than winter cereals. On the other hand, we found (summer-sown) maize stands to offer habitats to a different set of arable plant species than fields cultivated with winter-sown crops, such as oilseed rape, wheat or barley. In agricultural landscapes otherwise dominated by winter-sown crops, maize fields can therefore also be expected to positively influence arable plant diversity at the landscape scale.

We concluded that maize and oilseed rape are most detrimental to arable plant diversity if their proportion in the landscape and in the crop rotations becomes too high, or if species-rich, low-input farmland habitats are lost due to energy crop production. These facts indicate that the impacts of energy cropping differ widely between regions. Regionally specific subsidies or the inclusion of energy cropping in the existing landscape planning framework could help to prevent the most detrimental effects on arable plant diversity.

A similar conclusion as for annual energy crops can also be drawn for SRC. Compared to maize or oilseed rape production, SRC can, however, increase plant diversity if grown in highly intensified production landscapes and managed in a biodiversity-friendly way. The set of species which is likely to survive in SRC after canopy closure differs strongly from the arable plant assemblages found in annual cropping systems. It therefore seems necessary to identify priority groups of species to be targeted by conservation efforts in agricultural landscapes.

5.2   Open questions, uncertainties and future research perspectives

This thesis contributed to filling the 'hard data' gap which existed with regards to the consequences of energy crop production on the plant diversity of agricultural landscapes. At the outset of the work in November 2010, only very few publications with vegetation surveys in conventionally managed maize fields, oilseed rape fields and SRC were available (e.g. Mehrtens, 2005; Glemnitz et al., 2008; Baum et al., 2009). Based on field surveys and supported by the questionnaires on field management completed by the participating farmers, it was possible to discern potential relationships between plant diversity metrics and field management, climate and soil conditions and the light regime in energy cropping systems and control plots (conventionally managed winter-sown wheat and barley and wheat fields managed according to an agri-environmental scheme). This thesis can be used as a base for further explorations of the subject.

Many new questions arose on the details of the links between the assessed factors. As for all regression analyses, the correlations found do not necessarily represent genuine causal relationships. The results of the respective regression analyses supported our hypotheses in many cases, in others they led to new open questions. It remains for instance unclear, why the species richness of conventionally managed field margins increased if the sulphur fertilisation was high in the year preceding the survey. This effect might have arisen by chance, but a genuine link cannot be excluded based on current scientific knowledge. All open questions which arose with regard to the validity of the causal relationships assumed can only be answered in an experimental setting where confounding factors can be controlled for.

There are three further sets of research questions which arose during the process of conducting this project. Firstly, it seems important to assess the impacts of energy cropping on biodiversity at the landscape scale. Open questions which remain are for example: (1) Do high proportions of energy maize reduce arable plant diversity at the landscape scale? (2) Is β-diversity affected by energy cropping? (2) Which proportion of maize, oilseed rape or SRC cultivation is beneficial or tolerable with regard to the maintenance of plant diversity in different regional settings? (3) How should the different energy crops be distributed in the landscape to minimise the threats and maximise the

benefits for farmland plant diversity? Some of these questions might be answered based on available data gathered during the course of this project, but which could not be analysed until know.

The second important aspect concerns the link between arable plant diversity and other taxonomic groups. Some interesting questions which have not yet been fully addressed in the scientific literature are (1) how is arable plant diversity linked to soil nutrient cycling and to the diversity of the soil macro- and microorganisms? (2) What is the contribution of arable plants to above-ground ecosystem services such as pollination and pest control and how does it relate to the diversity of the respective taxonomic groups? (3) What is the role of the arable flora for maintaining bird diversity in agricultural landscapes? (4) And how are these ecosystem services of arable plants affected by energy cropping?

A third interesting aspect would be to explore possibilities of ranking different management approaches and conservation measures for agricultural landscapes according to their potential for maintaining or enhancing plant diversity in different regional settings. This information would be very useful for policy makers and could help to guide the debate about appropriate measures for biodiversity conservation in agricultural landscapes. Potential questions are for instance: (1) Is it possible to derive objective guidelines to rank or categorize conservation measures according to their function and impact (e.g. in 'fundamental' and 'conditional' measures)? (2) Will such guidelines lead to different rankings in different regional settings? (3) How do segregative and integrative conservation approaches need to be linked in a certain regional setting in order to achieve optimal results in terms of reconciling biodiversity conservation, food and energy production?

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Appendix

Appendix 1.A Data sheet as used for the vegetation survey.

No.	Date:	Location:		Exposition:	Slope:	Elevation a. s. l.:
Name:	Quadrat size:					
Edge/Inside:	Orientation of long side:	Along tractor lane	Across tractor lanes			
		Edge centre coordinate:		Coord. No.:		Coord. Syst.:
Total plant cover (%):		Cover moss layer (%):		Crop:		
Total crop cover (%):		Max. height of crop:		Mean height of crop (MCH):		
Dev. stage of crop (acc. to BBCH):		Total cover bare soil (%):		Total flower cover (%):		
Total tree/shrub cover (%):		Max. height of tree/shrub layer:		Mean height of tree/shrub layer:		
Total weed cover (%):		Max. height of weeds:		Mean height of weeds:		
Total cover weed layer 1 (WL1, <50 cm, %):		Max. height of WL1 weeds:		Mean height of WL1 weeds:		
Total cover weed layer 2 (WL2, >50 cm, %):		Max. height of WL2 weeds:		Mean height of WL2 weeds:		
Tot. cov. weeds above MCH (WLA):		Max. height of WLA weeds:		Mean height of WLA weeds:		
Tot. cov. Weeds below MCH (WLB):		Max. height of WLB weeds = Max. height of crop		Mean height of WLB weeds:		
Soil texture:	Soil description:					Soil sample no:

Notes:

Species	Cover acc. to BB					Dev. Stage (BBCH)
	Tot.	WL1	WL2	WLA	WLB	
1						
2						
3						
4						
5						
6						
7						

Appendix 1.A

Species	Cover acc. to BB					Dev. Stage (BBCH)
	Tot.	WL1	WL2	WLA	WLB	
8						
9						
10						
11						
12						
13						
14						
15						
16						
17						
18						
19						
20						
21						
22						
23						
24						
25						
26						
27						
28						
29						
30						

Notes:

Appendix 2.A Frequency of occurrence of species recorded on 206 conventionally managed maize, winter wheat and winter barley fields in the Lower Saxon hills and the Thuringian Basin.

Species name	Field interior				Field margin				Total
	Maize	Wheat	Barley	Sum	Maize	Wheat	Barley	Sum	
No. of plots	71	71	64	*206*	71	71	64	*206*	**412**
Achillea millefolium	0	0	0	*0*	0	4	5	*9*	**9**
Adonis aestivalis	0	0	1	*1*	1	2	5	*8*	**9**
Aegopodium podagraria	0	0	0	*0*	0	0	1	*1*	**1**
Aethusa cynapium	3	2	2	*7*	3	11	3	*17*	**24**
Agrimonia eupatoria	0	0	0	*0*	0	0	1	*1*	**1**
Agrostis capillaris	0	0	0	*0*	0	2	0	*2*	**2**
Agrostis stolonifera	0	0	0	*0*	2	6	4	*12*	**12**
Alliaria petiolata	0	0	1	*1*	0	1	2	*3*	**4**
Allium vineale	0	0	0	*0*	0	1	0	*1*	**1**
Alopecurus myosuroides	8	9	12	*29*	13	19	22	*54*	**83**
Alopecurus pratensis	0	0	0	*0*	0	1	1	*2*	**2**
Alyssum alyssoides	0	0	0	*0*	0	1	0	*1*	**1**
Amaranthus retroflexus	4	0	0	*4*	6	1	0	*7*	**11**
Anagallis arvensis	2	1	0	*3*	2	2	0	*4*	**7**
Anagallis foemina	0	0	0	*0*	0	1	1	*2*	**2**
Anchusa arvensis	0	0	0	*0*	0	0	1	*1*	**1**
Anthemis cotula	0	0	0	*0*	0	2	0	*2*	**2**
Anthriscus caucalis	0	0	0	*0*	0	0	1	*1*	**1**
Anthriscus sylvestris	0	0	0	*0*	0	1	2	*3*	**3**
Apera spica-venti	1	1	1	*3*	1	7	2	*10*	**13**
Aphanes arvensis	0	0	1	*1*	0	5	4	*9*	**10**
Arabidopsis thaliana	0	0	0	*0*	0	0	1	*1*	**1**
Arctium species	2	0	1	*3*	10	12	2	*24*	**27**
Arenaria serpyllifolia	0	0	0	*0*	0	1	0	*1*	**1**
Arrhenatherum elatius	0	0	0	*0*	2	11	6	*19*	**19**
Artemisia vulgaris	0	0	0	*0*	0	2	0	*2*	**2**
Asperugo procumbens	0	0	0	*0*	0	0	1	*1*	**1**
Atriplex patula	1	2	2	*5*	4	8	3	*15*	**20**
Avena fatua	8	3	1	*12*	10	6	3	*19*	**31**
Ballota nigra	0	0	0	*0*	1	1	1	*3*	**3**
Barbarea vulgaris	1	0	0	*1*	0	1	0	*1*	**2**
Bidens species	1	0	0	*1*	0	0	1	*1*	**2**
Bromus arvensis	0	0	0	*0*	1	0	0	*1*	**1**
Bromus commutatus ssp. *decipiens*	0	1	3	*4*	0	7	6	*13*	**17**
Bromus hordeaceus	0	0	0	*0*	0	4	5	*9*	**9**
Bromus secalinus	0	3	2	*5*	0	8	5	*13*	**18**
Bromus sterilis	4	11	11	*26*	12	41	41	*94*	**120**
Bryonia dioica	1	0	0	*1*	0	0	0	*0*	**1**
Buglossoides arvensis	0	0	0	*0*	1	0	1	*2*	**2**
Bunias orientalis	0	0	0	*0*	0	2	1	*3*	**3**
Bupleurum rotundifolium	0	1	0	*1*	0	1	1	*2*	**3**

Species name	Field interior				Field margin				Total
	Maize	Wheat	Barley	Sum	Maize	Wheat	Barley	Sum	
Calystegia sepium	2	0	0	*2*	3	0	1	*4*	**6**
Camelina microcarpa	0	0	0	*0*	0	2	0	*2*	**2**
Campanula rapunculoides	0	0	0	*0*	0	1	0	*1*	**1**
Capsella bursa-pastoris	4	2	0	*6*	10	7	6	*23*	**29**
Carduus acanthoides	0	1	0	*1*	1	9	6	*16*	**17**
Carduus crispus	0	0	0	*0*	1	0	0	*1*	**1**
Carex hirta	0	0	0	*0*	1	0	0	*1*	**1**
Caucalis platycarpos	0	0	0	*0*	0	2	4	*6*	**6**
Centaurea cyanus	0	1	0	*1*	0	4	0	*4*	**5**
Centaurea jacea	0	0	0	*0*	0	0	1	*1*	**1**
Centaurea scabiosa	0	0	0	*0*	0	1	2	*3*	**3**
Chaerophyllum aureum	0	0	0	*0*	1	0	0	*1*	**1**
Chaerophyllum temulum	0	0	0	*0*	0	1	0	*1*	**1**
Chenopodium album	24	18	11	*53*	32	37	22	*91*	**144**
Chenopodium hybridum	9	0	1	*10*	11	1	0	*12*	**22**
Cichorium intybus	0	0	0	*0*	0	0	1	*1*	**1**
Cirsium arvense	15	6	8	*29*	39	35	33	*107*	**136**
Cirsium vulgare	0	0	0	*0*	3	7	1	*11*	**11**
Consolida ajacis	0	0	0	*0*	0	1	0	*1*	**1**
Consolida regalis	0	2	2	*4*	1	9	6	*16*	**20**
Convolvulus arvensis	15	8	10	*33*	35	45	37	*117*	**150**
Cruciata laevipes	0	1	0	*1*	0	1	0	*1*	**2**
Dactylis glomerata	0	0	0	*0*	14	10	11	*35*	**35**
Daucus carota	0	0	0	*0*	1	5	1	*7*	**7**
Deschampsia cespitosa	0	0	0	*0*	0	1	0	*1*	**1**
Descurainia sophia	0	1	1	*2*	1	16	11	*28*	**30**
Echinochloa crus-galli	15	0	0	*15*	14	1	0	*15*	**30**
Echinops sphaerocephalus	0	0	0	*0*	0	1	1	*2*	**2**
Elymus repens	12	3	4	*19*	36	51	34	*121*	**140**
Epilobium angustifolium	0	0	0	*0*	0	1	0	*1*	**1**
Epilobium ciliatum	0	0	0	*0*	0	1	3	*4*	**4**
Equisetum arvense	7	3	0	*10*	18	19	11	*48*	**58**
Erodium cicutarium	1	0	0	*1*	2	3	2	*7*	**8**
Eryngium campestre	0	0	0	*0*	0	3	0	*3*	**3**
Erysimum cheiranthoides	1	0	0	*1*	1	0	0	*1*	**2**
Euphorbia cyparissias	0	0	0	*0*	0	3	1	*4*	**4**
Euphorbia exigua	1	1	2	*4*	2	6	1	*9*	**13**
Euphorbia helioscopia	10	3	1	*14*	19	18	10	*47*	**61**
Euphorbia platyphyllos	1	0	2	*3*	2	6	3	*11*	**14**
Falcaria vulgaris	1	0	0	*1*	5	4	8	*17*	**18**
Fallopia convolvulus	37	31	19	*87*	51	49	29	*129*	**216**
Festuca ovina ag.	0	0	0	*0*	0	1	2	*3*	**3**
Festuca pratensis	1	0	0	*1*	2	4	3	*9*	**10**
Festuca rubra ag.	0	0	0	*0*	1	5	1	*7*	**7**
Fragaria viridis	0	0	0	*0*	0	0	1	*1*	**1**

Species name	Field interior				Field margin				Total
	Maize	Wheat	Barley	Sum	Maize	Wheat	Barley	Sum	
Fumaria officinalis	6	4	3	*13*	6	11	9	*26*	**39**
Fumaria vaillantii	1	2	2	*5*	4	6	7	*17*	**22**
Galeopsis tetrahit	0	0	0	*0*	0	4	2	*6*	**6**
Galinsoga quadriradiata	1	0	0	*1*	1	0	0	*1*	**2**
Galium album	0	0	0	*0*	0	2	1	*3*	**3**
Galium aparine	23	26	13	*62*	46	55	40	*141*	**203**
Geranium columbinum	0	0	0	*0*	1	0	4	*5*	**5**
Geranium dissectum	8	2	2	*12*	28	15	8	*51*	**63**
Geranium molle	1	0	0	*1*	1	0	0	*1*	**2**
Geranium pusillum	12	2	1	*15*	25	19	19	*63*	**78**
Geum urbanum	0	0	0	*0*	1	3	1	*5*	**5**
Glechoma hederacea	0	0	0	*0*	1	2	3	*6*	**6**
Gnaphalium uliginosum	0	0	0	*0*	0	0	1	*1*	**1**
Helianthus annuus	1	0	0	*1*	0	0	0	*0*	**1**
Heracleum sphondylium	0	0	1	*1*	1	2	3	*6*	**7**
Hieracium aurantiacum	0	0	0	*0*	0	1	0	*1*	**1**
Holcus lanatus	0	0	0	*0*	1	1	0	*2*	**2**
Holcus mollis	0	0	0	*0*	0	1	3	*4*	**4**
Hordeum murinum	0	0	0	*0*	0	0	1	*1*	**1**
Hypericum perforatum	0	0	0	*0*	0	1	3	*4*	**4**
Impatiens glandulifera	0	0	0	*0*	0	0	1	*1*	**1**
Juncus bufonius	0	1	0	*1*	0	0	0	*0*	**1**
Kickxia elatine	1	0	0	*1*	0	0	0	*0*	**1**
Lactuca serriola	0	1	0	*1*	0	7	2	*9*	**10**
Lamium album	0	0	0	*0*	1	3	0	*4*	**4**
Lamium amplexicaule	0	2	1	*3*	0	4	4	*8*	**11**
Lamium purpureum	5	7	6	*18*	11	23	15	*49*	**67**
Lapsana communis	0	2	1	*3*	2	7	5	*14*	**17**
Lathyrus tuberosus	1	0	0	*1*	5	6	6	*17*	**18**
Lepidium draba	1	0	0	*1*	2	3	1	*6*	**7**
Lolium multiflorum	0	0	1	*1*	2	0	0	*2*	**3**
Lolium perenne	3	0	1	*4*	8	8	7	*23*	**27**
Lotus corniculatus	0	0	0	*0*	0	0	1	*1*	**1**
Malva neglecta	0	0	0	*0*	2	0	0	*2*	**2**
Matricaria discoidea	0	0	0	*0*	0	1	2	*3*	**3**
Matricaria recutita	6	1	3	*10*	8	5	12	*25*	**35**
Medicago lupulina	0	0	0	*0*	1	2	1	*4*	**4**
Medicago sativa	0	0	0	*0*	2	0	0	*2*	**2**
Melilotus officinalis	0	0	0	*0*	0	1	0	*1*	**1**
Mentha arvensis	1	0	0	*1*	1	0	0	*1*	**2**
Mercurialis annua	2	0	0	*2*	3	0	0	*3*	**5**
Myosotis arvensis	4	6	1	*11*	4	17	8	*29*	**40**
Nonea pulla	0	0	0	*0*	0	1	0	*1*	**1**
Ononis spinosa	0	0	0	*0*	0	0	1	*1*	**1**
Onopordum acanthium	0	0	0	*0*	0	1	0	*1*	**1**

Species name	Field interior				Field margin				Total
	Maize	Wheat	Barley	Sum	Maize	Wheat	Barley	Sum	
Panicum sp.	0	0	0	*0*	1	0	0	*1*	**1**
Papaver argemone	0	0	0	*0*	0	1	0	*1*	**1**
Papaver dubium	0	0	0	*0*	0	2	0	*2*	**2**
Papaver rhoeas	0	3	4	*7*	2	27	28	*57*	**64**
Pastinaca sativa	0	0	0	*0*	0	0	1	*1*	**1**
Persicaria amphibia	0	1	1	*2*	1	1	0	*2*	**4**
Persicaria lapathifolia	9	2	0	*11*	9	4	3	*16*	**27**
Persicaria maculosa	1	1	0	*2*	3	0	1	*4*	**6**
Phleum pratense	0	0	0	*0*	1	2	0	*3*	**3**
Phragmites australis	0	0	0	*0*	3	1	2	*6*	**6**
Plantago lanceolata	0	0	0	*0*	1	1	2	*4*	**4**
Plantago major	0	0	0	*0*	0	1	2	*3*	**3**
Plantago media	0	0	1	*1*	0	0	1	*1*	**2**
Poa angustifolia	0	0	0	*0*	0	1	0	*1*	**1**
Poa annua	1	2	1	*4*	2	13	7	*22*	**26**
Poa pratensis	0	0	1	*1*	0	4	6	*10*	**11**
Poa trivialis	1	0	1	*2*	3	10	13	*26*	**28**
Polygonatum multiflorum	0	0	0	*0*	0	1	0	*1*	**1**
Polygonum aviculare	29	16	14	*59*	41	40	32	*113*	**172**
Potentilla anserina	0	0	0	*0*	3	0	1	*4*	**4**
Potentilla reptans	0	0	0	*0*	3	5	2	*10*	**10**
Ranunculus bulbosus	0	0	0	*0*	0	1	0	*1*	**1**
Ranunculus repens	0	0	0	*0*	0	0	3	*3*	**3**
Raphanus raphanistrum	1	0	1	*2*	0	1	1	*2*	**4**
Reseda lutea	0	0	0	*0*	0	1	2	*3*	**3**
Rumex acetosella	0	0	0	*0*	0	1	0	*1*	**1**
Rumex crispus	4	0	0	*4*	6	2	3	*11*	**15**
Rumex obtusifolius	0	0	0	*0*	1	0	0	*1*	**1**
Sanguisorba minor	0	0	0	*0*	0	0	1	*1*	**1**
Scleranthus annuus	0	0	0	*0*	0	1	0	*1*	**1**
Securigera varia	0	0	0	*0*	0	0	1	*1*	**1**
Senecio vulgaris	0	0	0	*0*	1	0	0	*1*	**1**
Setaria pumila	0	0	0	*0*	2	0	0	*2*	**2**
Setaria viridis	1	0	0	*1*	1	0	0	*1*	**2**
Sherardia arvensis	0	1	0	*1*	0	1	0	*1*	**2**
Silene latifolia ssp. *alba*	0	1	0	*1*	2	3	1	*6*	**7**
Silene noctiflora	3	1	2	*6*	5	1	0	*6*	**12**
Silene vulgaris	0	0	0	*0*	0	0	1	*1*	**1**
Sinapis alba	2	1	0	*3*	4	2	0	*6*	**9**
Sinapis arvensis	1	0	0	*1*	2	2	6	*10*	**11**
Sisymbrium officinale	4	1	1	*6*	3	5	6	*14*	**20**
Solanum nigrum	9	0	0	*9*	13	0	1	*14*	**23**
Solidago gigantea	0	0	0	*0*	0	0	1	*1*	**1**
Sonchus arvensis	1	0	0	*1*	1	1	1	*3*	**4**
Sonchus asper	1	1	0	*2*	7	2	1	*10*	**12**

Species name	Field interior				Field margin				Total
	Maize	Wheat	Barley	Sum	Maize	Wheat	Barley	Sum	
Sonchus oleraceus	0	0	1	*1*	1	0	1	*2*	**3**
Stachys palustris	0	0	0	*0*	1	0	0	*1*	**1**
Stachys sylvatica	0	0	0	*0*	0	0	1	*1*	**1**
Stellaria media	6	0	0	*6*	9	2	4	*15*	**21**
Symphytum officinale	1	0	0	*1*	1	0	0	*1*	**2**
Tanacetum vulgare	0	0	0	*0*	0	1	1	*2*	**2**
Taraxacum sp.	9	6	0	*15*	6	9	3	*18*	**33**
Thlaspi arvense	6	2	3	*11*	16	14	19	*49*	**60**
Tragopogon pratensis	0	0	0	*0*	1	0	0	*1*	**1**
Trifolium pratense	1	0	0	*1*	1	0	1	*2*	**3**
Trifolium repens	1	0	0	*1*	0	0	1	*1*	**2**
Tripleurospermum perforatum	5	7	1	*13*	11	19	15	*45*	**58**
Trisetum flavescens	0	0	0	*0*	0	0	1	*1*	**1**
Tussilago farfara	0	0	0	*0*	1	1	0	*2*	**2**
Urtica dioica	1	0	0	*1*	9	6	6	*21*	**22**
Urtica urens	0	0	0	*0*	1	0	0	*1*	**1**
Valerianella locusta	0	0	0	*0*	0	3	0	*3*	**3**
Veronica arvensis	0	1	0	*1*	0	5	2	*7*	**8**
Veronica chamaedrys	0	0	0	*0*	0	1	0	*1*	**1**
Veronica filiformis	0	0	0	*0*	0	1	0	*1*	**1**
Veronica hederifolia	0	7	3	*10*	0	16	15	*31*	**41**
Veronica persica	9	2	0	*11*	25	22	8	*55*	**66**
Veronica polita	5	2	3	*10*	15	7	7	*29*	**39**
Vicia cracca	0	0	0	*0*	2	3	0	*5*	**5**
Vicia hirsuta	0	0	0	*0*	4	2	4	*10*	**10**
Vicia sativa	0	0	0	*0*	1	0	0	*1*	**1**
Vicia tenuifolia	0	0	0	*0*	0	0	1	*1*	**1**
Vicia tetrasperma	0	0	0	*0*	4	1	3	*8*	**8**
Viola arvensis	20	21	12	*53*	21	36	32	*89*	**142**
Viola hirta	0	0	0	*0*	1	0	0	*1*	**1**
Vulpia myuros	0	1	0	*1*	0	1	1	*2*	**3**
Anagallis sp.	0	0	0	*0*	0	1	0	*1*	**1**
Apiaceae sp.	0	0	1	*1*	0	0	1	*1*	**2**
Asteraceae sp.	0	0	2	*2*	0	4	7	*11*	**13**
Brassicaceae sp.	0	0	0	*0*	0	2	1	*3*	**3**
Bromus sp.	1	0	0	*1*	0	0	0	*0*	**1**
Caryophyllaceae sp.	0	0	0	*0*	0	1	0	*1*	**1**
Crataegus sp.	0	0	0	*0*	0	3	1	*4*	**4**
Epilobium sp.	0	0	0	*0*	1	1	0	*2*	**2**
Fumaria sp.	0	2	0	*2*	1	3	2	*6*	**8**
Geranium sp.	0	0	0	*0*	0	2	0	*2*	**2**
Hieracium sp.	0	0	0	*0*	1	0	0	*1*	**1**
Medicago sp.	0	0	0	*0*	0	0	1	*1*	**1**
Papaver sp.	0	0	0	*0*	1	1	0	*2*	**2**

Species name	Field interior				Field margin				Total
	Maize	Wheat	Barley	Sum	Maize	Wheat	Barley	Sum	
Persicaria sp.	0	0	0	*0*	0	0	1	*1*	**1**
Poaceae sp.	1	0	0	*1*	3	0	0	*3*	**4**
Ranunculus sp.	0	0	2	*2*	0	0	2	*2*	**4**
Rumex sp.	1	0	0	*1*	4	0	1	*5*	**6**
Sonchus sp.	0	0	0	*0*	1	1	2	*4*	**4**
Trifolium sp.	0	0	0	*0*	1	0	2	*3*	**3**
Valerianella sp.	0	0	0	*0*	0	1	0	*1*	**1**
Veronica sp.	0	0	0	*0*	0	1	0	*1*	**1**
Vicia sp.	1	0	0	*1*	2	0	0	*2*	**3**
Trees/shrubs									
Acer negundo	3	0	0	*3*	4	0	0	*4*	**7**
Acer sp.	4	2	2	*8*	5	3	5	*13*	**21**
Aesculus hippocastanum	0	0	0	*0*	1	0	0	*1*	**1**
Carpinus betulus	0	0	1	*1*	0	0	2	*2*	**3**
Cornus sp.	0	0	0	*0*	1	0	0	*1*	**1**
Fraxinus excelsior	2	5	2	*9*	2	5	5	*12*	**21**
Prunus avium	0	0	1	*1*	0	2	3	*5*	**6**
Prunus domestica	0	0	0	*0*	1	4	1	*6*	**6**
Prunus padus	0	0	0	*0*	0	0	1	*1*	**1**
Prunus sp.	0	2	0	*2*	0	0	1	*1*	**3**
Prunus spinosa	0	0	0	*0*	0	3	1	*4*	**4**
Quercus sp.	0	0	0	*0*	2	0	0	*2*	**2**
Robinia pseudoacacia	1	0	0	*1*	1	0	0	*1*	**2**
Rosa sp.	1	0	0	*1*	0	1	5	*6*	**7**
Rubus fruticosus ag.	1	0	0	*1*	5	8	7	*20*	**21**
Tilia sp.	0	0	1	*1*	0	0	1	*1*	**2**
Volunteer crops									
Avena sativa	0	0	0	*0*	1	0	0	*1*	**1**
Brassica napus	21	7	8	*36*	18	5	13	*36*	**72**
Hordeum vulgare	0	6	0	*6*	0	6	0	*6*	**12**
Pisum sativum	0	0	0	*0*	1	0	0	*1*	**1**
Secale cereale	0	2	2	*4*	0	3	4	*7*	**11**
×Triticosecale	0	2	1	*3*	0	3	2	*5*	**8**
Triticum aestivum	12	1	24	*37*	7	0	22	*29*	**66**
Triticum durum	0	0	1	*1*	0	0	0	*0*	**1**

Appendix 2.B Questionnaire about the field management answered by the 60 participating farmers
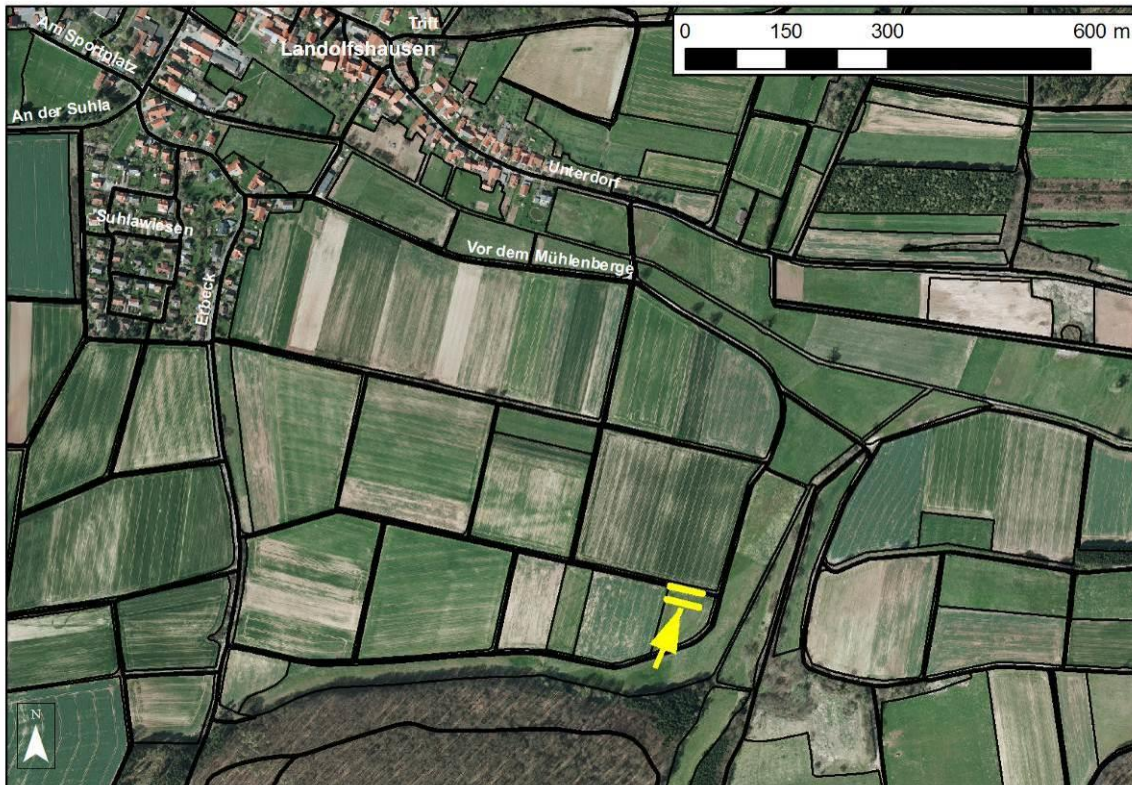
Landnutzungsgeschichte:

1) Seit wann ist der Schlag Ackerland? (bitte Zutreffendes markieren)

___ seit weniger als 10 Jahren   ___ seit 30-50 Jahren

___ seit 10-15 Jahren   ___ seit 50-100 Jahren

___ seit 15-20 Jahren   ___ seit mehr als 100 Jahren

___ seit 20-30 Jahren   ___ nicht bekannt, aber mindestens seit _____ Jahren

2) Vorherige Nutzung (bitte Zutreffendes markieren):

___ Dauergrünland   ___ anders: _____

___ Wald   ___ nicht bekannt

Bemerkungen:

335 Schlagname: _____   Frucht 2012: Winterweizen (Sorte:_____)

Fruchtfolge 2002 - 2012:

| Jahr | Hauptfrucht | Nutzung zur Bioenergie-produktion? | Anschließend Zwischenfrucht? | | | |
			Ja	Nein	Wenn ja, bitte Frucht angeben.	Nutzung zur Bioenergie-produktion?
2002						
2003						
2004						
2005						
2006						
2007						
2008						
2009						
2010						
2011						
2012						

Bemerkungen:

Appendix 2.B

335 Schlagname: _____   Frucht 2012: Winterweizen (Sorte:_____)

Pflanzenschutz

Anstatt des Eintragens in die unten bereitgestellten Felder können Sie auch einen Ausdruck über den Pflanzenschutz während der letzten beiden Saisons (vom 01. August **2010** – 31. Juli **2012**) beifügen.
Bitte darauf achten, dass **Datum, Produktbezeichnung** und **Mengen**angaben (**mit Einheit!**) vorhanden sind!

Vorletzte Saison: 01. August **2010** – 31. Juli **2011**

DATUM	PRODUKTBEZEICHNUNG	MENGE (in Liter oder Kg pro Hektar)

Letzte Saison: 01. August **2011** – 31. Juli **2012**

DATUM	PRODUKTBEZEICHNUNG	MENGE (in Liter oder Kg pro Hektar)

Bemerkungen:

335 Schlagname: _____   Frucht 2012: Winterweizen (Sorte:_____)

Organische und mineralische Düngemittel

Bitte Düngemittel eintragen oder Ausdruck beifügen. Bitte darauf achten, dass **Datum, Bezeichnung** und **Mengen**angaben (**mit Einheit!**) vorhanden sind!
Bitte **organische und mineralische** Düngemittel angeben.

Vorletzte Saison: 01. August 2010 – 31. Juli 2011

DATUM	BEZEICHNUNG	MENGE (in Liter oder Kg pro Hektar)

Letzte Saison: 01. August 2011 – 31. Juli 2012

DATUM	BEZEICHNUNG	MENGE (in Liter oder Kg pro Hektar)

Bemerkungen:

Appendix 2.B

335 Schlagname: _____   Frucht 2012: Winterweizen (Sorte:_____)

Mechanische Bearbeitung (pflügen, grubbern, drillen etc.)

Zutreffendes bitte ankreuzen:

___ Vollständig pfluglose Bearbeitung seit _____

___ Regelmäßig pflügende Bearbeitung seit _____

___ Bearbeitung wechselnd je nach Feldfrucht

___ Bearbeitung anders: _____

Für die letzten beiden Saisons bitte Art der Bearbeitung und Datum angeben oder Ausdruck beifügen. Besonders wichtig sind das **Pflügen** und die **Aussaattermine**.

Vorletzte Saison: 01.August 2010 bis 31. Juli 2011

DATUM	Art der Bearbeitung

Letzte Saison: 01.August 2011 bis 31. Juli 2012

DATUM	Art der Bearbeitung

Bemerkungen:

Appendix 2.C Additional information for the variables soil type (a), geological substrate (b) and the HRAC herbicide categories (c)

(a) Number of plots for each soil type separated by crop and in-field position.

soil type	abb.	field interior				field margin				total
		maize	wheat	barley	sum	maize	wheat	barley	sum	
Total no. of plots		71	71	64	*206*	71	71	64	*206*	**412**
Vega (alluvial)	AB	7	1	5	*13*	7	2	5	*14*	**27**
Tschernitza (alluvial)	AT	3	2	2	*7*	2	3	2	*7*	**14**
Brown Earth	BB	5	6	4	*15*	5	6	4	*15*	**30**
Brown Earth – Pelosols	BB-DD	0	1	0	*1*	0	1	0	*1*	**2**
Brown-Earth – Regosols	BB-RQ	1	0	2	*3*	1	0	2	*3*	**6**
Brown Earth – Pararendzinas	BB-RZ	0	1	0	*1*	0	1	0	*1*	**2**
Albeluvisols	BF	0	0	1	*1*	0	0	1	*1*	**2**
Luvisol - Chernozems	BL-TT	4	5	1	*10*	4	5	0	*9*	**19**
Pelosols	DD	9	9	12	*30*	7	9	13	*29*	**59**
Rendzinas	RR	1	3	1	*5*	1	3	2	*6*	**11**
Pararendzinas	RZ	7	13	9	*29*	8	13	9	*30*	**59**
Stagnosol – Brown Earths	SS-BB	5	1	2	*8*	5	1	2	*8*	**16**
Stagnosol – Luvisols	SS-BL	15	12	9	*36*	15	12	9	*36*	**72**
Chernozems	TT	12	15	14	*41*	14	14	13	*41*	**82**
Chernozem – Luvisols	TT-BL	0	1	0	*1*	0	1	0	*1*	**2**
Colluviums	YK	2	0	2	*4*	2	0	2	*4*	**8**

(b) Number of plots for each geological substrate separated by crop and in-field position.

sediment	field interior				field margin				total
	maize	wheat	barley	sum	maize	wheat	barley	sum	
Total no. of plots	71	71	64	206	71	71	64	206	412
Loess	25	21	16	62	24	21	15	60	122
Loess over Bunter Sandstone	2	3	1	6	3	3	1	7	13
Loess over Muschelkalk	1	3	2	6	1	3	1	5	11
Upper Bunter Sandstone	4	4	2	10	4	4	2	10	20
Middle Bunter Sandstone	2	4	3	9	2	4	3	9	18
Lower Bunter Sandstone	1	2	0	3	1	2	0	3	6
Upper Muschelkalk	2	5	3	10	2	5	4	11	21
Middle Muschelkalk	2	4	1	7	2	4	1	7	14
Lower Muschelkalk	1	0	1	2	1	0	1	2	4
Middle Keuper	9	12	16	37	9	11	17	37	74
Lower Keuper	8	6	6	20	9	6	6	21	41
Pleistocene cover sediments over Bunter Sandstone	0	1	1	2	0	1	1	2	4
Pleistocene fluvial terraces	3	4	5	12	2	4	5	11	23
Glacial till	0	0	1	1	0	0	1	1	2
Alluvial sediments (Holocene)	11	2	6	19	11	3	6	20	39

(c)   HRAC herbicide classification system according to Menne and Köcher (2012). Only groups with substances applied on the study fields are shown.

HRAC group	Mechanism of action	Chemical family	Number of fields
A	ACCase inhibitors	Aryloxyphenoxy-propionates, cyclohexanediones, phenylpyrazolines	61
B	ALS/AHAS inhibitors	Sulfonylureas, imidazolinones, triazolopyrimidines, pyrimidinyl(thio)benzoates, sulfonylaminocarbonyl-triazolinones	197
C_1	PS-II inhibitors	Triazines, triazinones, triazolinones, uracils, pyridazinones, phenylcarbamates	105
C_2	PS-II inhibitors	Ureas, amides	41
C_3	PS-II inhibitors	Nitriles, benzothiadiazinones, pheny-pyridazines	63
E	Protoporphyrinogen oxidase inhibitors	Dephenyl ethers, phenylpyrazoles, N-phenylphthalimides, thiadiazoles, oxadiazoles, oxazolidinediones, pyrimidindiones, others	44
F_1	Carotenoid biosynthesis inhibitors (inhibition of the phytoene desaturase)	Pyridazinones, pyridinecarboxamides, others	84
F_2	Carotenoid biosynthesis inhibitors (inhibition of 4-hydroxyphenyl-pyruvate-dioxygenase)	Triketones, isoxazoles, pyrazoles, others	45
F_3	Carotenoid biosynthesis inhibitors (unknown target)	Triazoles, diphenylethers, ureas	10
F_4	Carotenoid biosynthesis inhibitors (inhibition of 1-deoxy-D-xylulose 5-phosphate synthase)	Isoxazolidinones	3
G	EPSP synthase inhibitors	Glycines (includes glyphosate)	91
K_1	Inhibitors of microtubule assembly	Dinitroanilines, phosphoroamidates, pyridines, benzamides, benzoic acids	18
K_3	Mitosis inhibitors (inhibition of VLCFAs)	Chloroacetamides, acetamides, oxyacetamides, tetrazolinones, others	96
N	Inhibitors of lipid synthesis (not ACCase inhibition)	Thiocarbamates, phosphorodithioates, benzofurans, chloro-carbonic-acids	11
O	Synthetic auxins	Phenoxy-carboxylic-acids, benzoic acids, pyridine carboxylic acids, quinolone carboxylic acids	115

Appendix 2.D

(a) Loadings of the principal components (PCs) from PCA of different groups of environmental variables. PCs were entered in the statistical analyses to avoid problems with collinearity for sets of factors with highly correlated variables (macro environment, fertilisation, crop rotation).

Principle components	PC1	PC2	PC3	PC4
Macro environment				
Total explained variation (%)	*72.3*	*23.6*		
Region	**0.85**	0.49	-	-
Elevation a. s. l.	0.63	**-0.77**	-	-
Precipitation	**0.98**	0.06	-	-
Temperature	**-0.90**	0.40	-	-
Sunshine hours	**-0.86**	-0.43	-	-
Fertilisation (year of survey)				
Total explained variation (%)	*56.7*	*16.9*	*8.3*	*6.8*
N	-0.67	-0.49	-0.26	0.46
P	**-0.74**	0.24	-0.48	-0.36
K	**-0.89**	-0.01	-0.14	-0.06
S	-0.15	**-0.92**	0.12	-0.34
logCa	**-0.87**	0.12	0.32	-0.05
Mg	**-0.78**	0.15	0.39	-0.03
Organic fertilisers	**-0.89**	0.09	0.05	0.14
Fertilisation (preceding year)				
Total explained variation (%)	*55.6*	*15.8*	*9.1*	*7.9*
N	-0.66	0.42	-0.01	-0.62
P	**-0.77**	-0.16	-0.30	0.01
K	**-0.86**	-0.08	-0.29	0.07
S	-0.12	**0.93**	-0.06	0.32
logCa	**-0.86**	-0.09	0.29	0.21
Mg	**-0.76**	-0.02	0.58	0.01
Organic fertilisers	**-0.89**	-0.13	-0.21	0.14
Crop rotation				
Total explained variation (%)	*39.6*	*22.2*	*19.5*	*10.7*
# of crops	0.17	-0.16	**-0.83**	0.46
Prop. of maize	0.50	**-0.73**	0.30	-0.24
Prop. of wheat	**-0.79**	0.30	0.18	-0.19
Prop. of winter crops	**-0.85**	0.36	0.00	0.12
Prop. of legumes	0.28	0.33	-0.66	-0.60
Prop. of fallow/grassland	**0.70**	0.54	0.34	0.26
Prop. of biennials/perennials	**0.76**	0.61	0.11	-0.04

(b) Complete theoretical models before backward selection for each of the six response variables sorted by the nine sets of predictors.

Set of variables	Full models
Crop	$crop_{syr}*crop_{pyr} + crop_{syr}*catch\ crop_{syr} + crop_{pyr}*catch\ crop_{pyr}$
Fertilisation	$PC1_{syr}*PC1_{pyr} + PC2_{syr}*PC2_{pyr} + PC3_{syr}*PC3_{pyr} + PC4_{syr}*PC4_{pyr}$
Herbicides	intensity index$_{syr}$ * intensity index$_{pyr}$ + A + B + C1 + C2 + C3 + E + F1 + F2 + F3 + F4 + G + K1 + K3 + N + O
Conversion tillage	ploughing y/n (last 2 yrs) + ploughing y/n (last 5 yrs)
Crop rotation	PC1 + PC2 + PC3 + PC4
Land use history	time since conversion to cropland*previous land use
Macro environment	PC 1*PC 2
Micro environment	slope*aspect + crop cover (+ tree/shrub cover)
Soil	soil type*geological substrate + pH + C/N ratio + SOM + P
Spatial factors	all positive PCNM variables (103-107, depending on response variable)

Appendix 3.A Species list from vegetation surveys on 50 arable fields in the Lower Saxon hills and the Thuringian Basin in the Central Highlands of Germany, giving the number of plots for each cropping system and in-field position of the plot (FI, field interior; FM, field margin), in which the 157 weed species have been recorded. Conv., conventional management; AES, agri-environmental scheme; red list status in Germany (1, critically endangered; 2, endangered; 3, vulnerable; V, near threatened; D, data deficient; *, not threatened) based on Ludwig and Schnittler (1996).

| | FI | | | | | FM | | | | | | Total | Red list status |
	All conv. crops	Maize	Oilseed rape	Barley	Wheat conv.	Wheat AES	All conv. crops	Maize	Oilseed rape	Barley	Wheat conv.	Wheat AES		
Total number of plots (*n*)	40	10	10	10	10	10	40	10	10	10	10	10	100	
Achillea millefolium	0	0	0	0	0	1	2	0	0	1	1	0	3	*
Adonis aestivalis	0	0	0	0	0	4	1	0	0	1	0	5	10	3
Adonis flammea	0	0	0	0	0	1	0	0	0	0	0	0	1	1
Aegopodium podagraria	0	0	0	0	0	0	1	0	1	0	0	0	1	*
Aethusa cynapium	1	0	0	1	0	1	0	0	0	0	0	2	4	*
Agrimonia eupatoria	0	0	0	0	0	2	0	0	0	0	0	2	4	*
Agrostis stolonifera	0	0	0	0	0	0	1	0	0	1	0	1	2	*
Allium oleraceum	0	0	0	0	0	1	0	0	0	0	0	2	3	*
Allium vineale	0	0	0	0	0	2	0	0	0	0	0	4	6	*
Alopecurus myosuroides	7	1	1	2	3	4	14	3	4	3	4	4	29	*
Alopecurus pratensis	0	0	0	0	0	0	1	0	0	1	0	0	1	*
Alyssum alyssoides	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Amaranthus retroflexus	0	0	0	0	0	0	1	1	0	0	0	0	1	*
Anagallis arvensis	0	0	0	0	0	4	1	1	0	0	0	7	12	*
Anagallis foemina	0	0	0	0	0	2	0	0	0	0	0	3	5	*
Anthemis cotula	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Anthriscus sylvestris	0	0	0	0	0	0	1	0	1	0	0	0	1	*
Apera spica-venti	0	0	0	0	0	3	2	0	1	0	1	3	8	*
Aphanes arvensis	0	0	0	0	0	4	2	0	2	0	0	4	10	*
Arctium lappa	0	0	0	0	0	0	3	3	0	0	0	0	3	*
Arenaria serpyllifolia	0	0	0	0	0	5	0	0	0	0	0	4	9	*
Arrhenatherum elatius	0	0	0	0	0	0	8	0	4	1	3	3	11	*
Artemisia vulgaris	0	0	0	0	0	0	1	0	0	0	1	0	1	*
Asperugo procumbens	0	0	0	0	0	0	1	0	0	1	0	0	1	3
Atriplex patula	2	1	0	1	0	1	1	0	1	0	0	1	5	*
Avena fatua	1	0	0	0	1	3	3	0	1	0	2	2	9	*
Barbarea vulgaris	2	1	0	1	0	0	2	0	0	1	1	0	4	*
Bromus hordeaceus	0	0	0	0	0	0	1	0	0	1	0	0	1	*
Bromus japonicus	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Bromus secalinus	1	0	0	0	1	1	2	0	0	0	2	1	5	*
Bromus sterilis	2	0	0	1	1	5	15	1	2	5	7	7	29	*
Bunias orientalis	0	0	0	0	0	1	0	0	0	0	0	1	2	*
Bupleurum rotundifolium	0	0	0	0	0	1	0	0	0	0	0	2	3	1
Camelina microcarpa	0	0	0	0	0	3	0	0	0	0	0	3	6	*
Camelina sativa	0	0	0	0	0	0	1	0	0	0	1	0	1	*
Campanula rapunculoides	0	0	0	0	0	0	0	0	0	0	0	2	2	*
Capsella bursa-pastoris	4	1	2	1	0	1	5	0	4	1	0	1	11	*
Carduus acanthoides	0	0	0	0	0	3	3	0	0	1	2	5	11	*
Carex hirta	0	0	0	0	0	0	1	1	0	0	0	0	1	*
Carum carvi	0	0	0	0	0	0	0	0	0	0	0	1	1	V
Caucalis platycarpos	0	0	0	0	0	1	0	0	0	0	0	3	4	3

	FI					FM						Total	Red list status
	All conv. crops	Maize	Oilseed rape	Barley	Wheat conv.	Wheat AES	All conv. crops	Maize	Oilseed rape	Barley	Wheat conv.	Wheat AES		
Total number of plots (*n*)	*40*	*10*	*10*	*10*	*10*	*10*	*40*	*10*	*10*	*10*	*10*	*10*	*100*	
Centaurea cyanus	1	0	0	0	1	2	1	0	0	0	1	2	6	*
Centaurea jacea	0	0	0	0	0	0	1	0	0	1	0	0	1	*
Centaurea scabiosa	0	0	0	0	0	0	0	0	0	0	0	4	4	*
Centaurea stoebe	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Cerastium holosteoides	0	0	0	0	0	0	0	0	0	0	0	2	2	*
Chenopodium album	8	3	1	2	2	2	18	4	5	5	4	1	29	*
Chenopodium hybridum	1	1	0	0	0	1	3	2	1	0	0	0	5	*
Cichorium intybus	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Cirsium arvense	4	1	1	2	0	7	22	7	5	7	3	10	43	*
Cirsium vulgare	0	0	0	0	0	0	3	0	1	1	1	0	3	*
Consolida regalis	0	0	0	0	0	7	2	0	0	2	0	7	16	3
Convolvulus arvensis	9	2	4	3	0	7	28	6	8	6	8	10	54	*
Cynoglossum officinale	0	0	0	0	0	1	0	0	0	0	0	0	1	*
Dactylis glomerata	0	0	0	0	0	0	9	5	2	2	0	5	14	*
Daucus carota	0	0	0	0	0	0	2	0	0	1	1	4	6	*
Descurainia sophia	3	0	3	0	0	3	8	0	3	2	3	1	15	*
Echinochloa crus-galli	3	3	0	0	0	0	2	2	0	0	0	0	5	*
Elymus repens	4	1	2	0	1	2	30	6	9	6	9	2	38	*
Epilobium ciliatum	1	0	1	0	0	0	2	0	1	1	0	0	3	*
Equisetum arvense	3	1	1	0	1	0	8	1	3	2	2	0	11	*
Erodium cicutarium	0	0	0	0	0	2	2	0	0	1	1	3	7	*
Eryngium campestre	0	0	0	0	0	0	0	0	0	0	0	5	5	V
Euphorbia cyparissias	0	0	0	0	0	0	1	0	0	0	1	4	5	*
Euphorbia esula	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Euphorbia exigua	0	0	0	0	0	4	0	0	0	0	0	6	10	*
Euphorbia falcata	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Euphorbia helioscopia	1	0	1	0	0	2	11	3	3	3	2	5	19	*
Euphorbia platyphyllos	0	0	0	0	0	1	1	0	0	0	1	3	5	*
Falcaria vulgaris	0	0	0	0	0	3	2	1	0	1	0	4	9	*
Fallopia convolvulus	21	7	3	5	6	7	28	6	7	8	7	6	62	*
Festuca brevipila	0	0	0	0	0	0	1	0	0	1	0	0	1	*
Festuca pratensis	0	0	0	0	0	0	3	0	2	1	0	1	4	*
Festuca rubra agg.	0	0	0	0	0	0	1	0	0	0	1	0	1	*
Fumaria officinalis	5	2	0	2	1	0	6	0	0	3	3	0	11	*
Fumaria vaillantii	1	0	0	1	0	1	4	2	0	1	1	0	6	*
Galeopsis tetrahit	0	0	0	0	0	0	1	0	0	0	1	0	1	*
Galium album	0	0	0	0	0	0	1	0	0	0	1	3	4	*
Galium aparine	16	3	4	3	6	6	35	7	10	8	10	4	61	*
Galium glaucum	0	0	0	0	0	0	0	0	0	0	0	1	1	V
Galium spurium	0	0	0	0	0	1	0	0	0	0	0	2	3	D
Galium tricornutum	0	0	0	0	0	2	0	0	0	0	0	2	4	3
Galium verum	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Geranium columbinum	0	0	0	0	0	5	0	0	0	0	0	6	11	*
Geranium dissectum	5	3	1	1	0	4	18	8	5	2	3	4	31	*
Geranium pusillum	4	2	2	0	0	6	13	4	3	2	4	3	26	*
Geum urbanum	0	0	0	0	0	0	1	0	1	0	0	1	2	*
Glechoma hederacea	0	0	0	0	0	0	3	1	1	1	0	1	4	*
Heracleum sphondylium	0	0	0	0	0	0	1	1	0	0	0	0	1	*
Hypericum perforatum	0	0	0	0	0	0	0	0	0	0	0	2	2	*
Knautia arvensis	0	0	0	0	0	1	0	0	0	0	0	3	4	*
Lactuca serriola	0	0	0	0	0	0	1	0	1	0	0	2	3	*
Lamium amplexicaule	1	0	0	0	1	0	2	0	0	0	2	2	5	*
Lamium purpureum	4	1	0	2	1	1	10	2	2	1	5	2	17	*

Appendix 3.A

	FI						FM							
	All conv. crops	Maize	Oilseed rape	Barley	Wheat conv.	Wheat AES	All conv. crops	Maize	Oilseed rape	Barley	Wheat conv.	Wheat AES	Total	Red list status
Total number of plots (*n*)	*40*	*10*	*10*	*10*	*10*	*10*	*40*	*10*	*10*	*10*	*10*	*10*	*100*	
Lappula squarrosa	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Lapsana communis	1	0	0	1	0	4	0	0	0	0	0	4	9	*
Lathyrus tuberosus	1	1	0	0	0	2	3	1	1	1	0	5	11	*
Lepidium campestre	0	0	0	0	0	1	0	0	0	0	0	3	4	*
Lepidium draba	0	0	0	0	0	1	1	0	0	1	0	2	4	*
Linum austriacum	0	0	0	0	0	0	0	0	0	0	0	3	3	*
Lolium perenne	0	0	0	0	0	0	1	1	0	0	0	0	1	*
Lotus corniculatus	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Matricaria recutita	2	0	0	2	0	1	7	1	2	4	0	0	10	*
Medicago lupulina	0	0	0	0	0	3	0	0	0	0	0	5	8	*
Microthlaspi perfoliatum	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Myosotis arvensis	6	1	3	1	1	3	11	1	6	1	3	3	23	*
Nonea erecta	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Ononis spinosa	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Onopordum acanthium	0	0	0	0	0	1	0	0	0	0	0	0	1	*
Papaver argemone	0	0	0	0	0	0	0	0	0	0	0	2	2	*
Papaver rhoeas	2	0	2	0	0	7	15	1	5	5	4	4	28	*
Persicaria lapathifolia	2	2	0	0	0	0	2	2	0	0	0	0	4	*
Plantago lanceolata	0	0	0	0	0	0	0	0	0	0	0	2	2	*
Poa annua	1	0	1	0	0	0	1	0	0	0	1	0	2	*
Poa pratensis	0	0	0	0	0	0	5	0	3	2	0	0	5	*
Poa trivialis	0	0	0	0	0	0	6	0	3	3	0	4	10	*
Polygonum aviculare	14	6	4	3	1	5	24	8	4	7	5	5	48	*
Potentilla anserina	0	0	0	0	0	0	1	1	0	0	0	0	1	*
Potentilla reptans	0	0	0	0	0	0	3	2	0	0	1	3	6	*
Ranunculus arvensis	0	0	0	0	0	1	0	0	0	0	0	1	2	3
Ranunculus repens	0	0	0	0	0	2	1	0	1	0	0	2	5	*
Reseda lutea	0	0	0	0	0	0	0	0	0	0	0	3	3	*
Rumex crispus	1	1	0	0	0	3	7	2	3	1	1	1	12	*
Rumex obtusifolius	0	0	0	0	0	1	1	1	0	0	0	0	2	*
Sanguisorba minor	0	0	0	0	0	0	0	0	0	0	0	3	3	*
Senecio vulgaris	0	0	0	0	0	0	1	1	0	0	0	0	1	*
Sherardia arvensis	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Silene latifolia ssp. alba	0	0	0	0	0	0	3	0	1	0	2	1	4	*
Silene noctiflora	1	1	0	0	0	0	1	1	0	0	0	1	3	V
Silene vulgaris	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Sinapis arvensis	0	0	0	0	0	1	1	0	1	0	0	2	4	*
Sisymbrium officinale	2	1	0	1	0	0	2	0	1	1	0	0	4	*
Solanum nigrum	3	3	0	0	0	0	3	3	0	0	0	0	6	*
Sonchus arvensis	0	0	0	0	0	0	2	1	0	1	0	0	2	*
Sonchus asper	0	0	0	0	0	1	3	0	3	0	0	1	5	*
Stachys palustris	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Stellaria media	1	0	0	1	0	3	6	3	1	1	1	1	11	*
Thlaspi arvense	4	0	2	2	0	1	10	1	3	5	1	2	17	*
Trifolium campestre	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Trifolium pratense	0	0	0	0	0	1	0	0	0	0	0	1	2	*
Trifolium repens	1	1	0	0	0	0	1	0	0	1	0	0	2	*
Tripleurospermum perforatum	4	1	2	0	1	7	15	3	5	3	4	5	31	*
Urtica dioica	1	0	1	0	0	0	10	3	2	1	4	0	11	*
Urtica urens	0	0	0	0	0	0	1	1	0	0	0	0	1	*
Valerianella dentata	0	0	0	0	0	4	0	0	0	0	0	5	9	V
Valerianella locusta	0	0	0	0	0	0	1	0	0	0	1	0	1	*
Veronica arvensis	0	0	0	0	0	3	0	0	0	0	0	4	7	*

	FI						FM						Total	Red list status
	All conv. crops	Maize	Oilseed rape	Barley	Wheat conv.	Wheat AES	All conv. crops	Maize	Oilseed rape	Barley	Wheat conv.	Wheat AES		
Total number of plots (*n*)	*40*	*10*	*10*	*10*	*10*	*10*	*40*	*10*	*10*	*10*	*10*	*10*	*100*	
Veronica hederifolia	2	0	1	1	0	0	6	0	0	3	3	1	9	*
Veronica persica	5	4	1	0	0	6	18	8	5	2	3	6	35	*
Veronica polita	1	1	0	0	0	3	4	3	0	0	1	5	13	*
Vicia angustifolia	0	0	0	0	0	1	0	0	0	0	0	6	7	*
Vicia cracca agg.	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Vicia hirsuta	0	0	0	0	0	2	0	0	0	0	0	2	4	*
Vicia sativa	0	0	0	0	0	4	1	1	0	0	0	4	9	*
Vicia villosa	0	0	0	0	0	0	0	0	0	0	0	1	1	*
Viola arvensis	14	4	7	1	2	7	18	3	6	4	5	8	47	*
Vulpia myuros	0	0	0	0	0	0	1	0	0	1	0	0	1	*

Cases excluded from the analyses as specimens were not identifiable to the required level:

Anagallis sp.	1	0	0	1	0	0	0	0	0	0	0	0	1	na
Arctium sp.	1	0	1	0	0	0	6	0	2	0	4	0	7	na
Fumaria sp.	0	0	0	0	0	0	1	0	0	1	0	0	1	na
Rumex sp.	1	0	0	1	0	2	1	1	0	0	0	0	4	na
Sonchus sp.	1	0	0	1	0	0	1	0	0	1	0	0	2	na
Taraxacum sp.	2	1	0	0	1	2	3	0	1	1	1	6	13	na
Trifolium sp.	0	0	0	0	0	0	0	0	0	0	0	1	1	na
Asteracea sp.	0	0	0	0	0	0	3	1	2	0	0	0	3	na
Caryophyllaceae sp.	0	0	0	0	0	0	1	0	0	1	0	0	1	na
Poaceae sp.	0	0	0	0	0	0	1	0	1	0	0	0	1	na

Appendix 3.B Mean TPAR (± SE) measured in the field interior (FI) and at the field margins (FM) of 50 arable fields with different cropping systems in two study regions in the Central Uplands of Germany (LS, Lower Saxon hills; TB, Thuringian Basin). Differences in TPAR between the study areas for each cropping system and in-field position of the plot were tested with the independent two-group Mann-Whitney U test (*, $p \leq 0.05$; **, $p \leq 0.01$).

Data set	n per location	TPAR (%) FI			FM		
		LS	TB	P	LS	TB	P
All conv. crops	20	9.1±1.1	10.6±0.9	0.29	15.8±2.2	18.0±1.5	0.13
Maize	5	5.7±1.3	6.1±1.1	0.84	8.8±1.1	13.1±2.4	0.10
Oilseed rape	5	12.2±1.8	12.8±1.1	0.55	16.3±3.2	22.3±2.5	0.31
Barley	5	10.2±2.9	11.4±1.7	1.00	21.1±5.0	17.0±1.3	0.70
Wheat conv.	5	8.2±2.1	12.2±2.0	0.22	17.0±6.0	19.4±4.3	0.70
Wheat AES	5	**46.0±4.1**	**67.7±5.6**	**0.02***[*]	**52.8±4.8**	**85.3±3.7**	**0.01**[**]

Appendix 3.C Means (± SE) of TPAR (in %), the crop cover, the weed cover and the total cover (= crop + weed cover, all in %) and the mean number of species in arable weed assemblages for four different conventionally managed crops (conv.) and winter wheat managed under an agri-environmental scheme (AES) from arable fields in the Central Uplands of Germany. As TPAR of AES fields was found to differ between the two study regions (LS: Lower Saxon hills, TB: Thuringian Basin), means are also given for each region. Capital letters indicate significant differences between cropping systems (grey shaded rows, nonparametric Behrens-Fisher test for multiple comparisons). Values from the field margin (FM) are significantly different from the field interior (FI) across cropping systems (grey shaded rows) for all six parameters (Wilcoxon signed-rank test; $n = 50$; *, $p \leq 0.05$; ***, $p \leq 0.001$).

Position	n per column	TPAR (%)			Crop cover (%)			Weed cover (%)			Species no.		
		FI	***	FM	FI	***	FM	FI	***	FM	FI	***	FM
All conv. crops	40	9.9±0.7		16.9±1.3	78.2±4.9		60.9±6.2	2.5±2.1		16.1±3.4	4.6±0.6		13.9±0.8
Maize	10	5.9±0.8[A]		10.9±1.4[A]	80.0±3.7[A]		64.6±3.4[A]	3.5±1.3[A]		20.6±4.2[A]	6.1±1.4[A]		13.1±1.4[A]
Oilseed rape	10	12.6±1.0[B]		19.3±2.2[B]	93.6±1.8[B]		81.0±3.8[B]	1.7±0.5[A]		13.7±4.4[A]	5.1±0.6[A]		15.0±1.3[A]
Barley	10	10.8±1.6[AB]		19.1±2.5[B]	66.0±5.3[A]		45.0±5.1[C]	0.5±0.3[A]		13.5±3.2[A]	4.1±1.6[A]		13.5±1.9[A]
Wheat conv.	10	10.2±1.5[AB]		18.2±3.5[AB]	73.1±3.4[A]		53.0±5.6[AC]	4.3±4.0[A]		16.7±3.8[A]	3.1±0.7[A]		14.0±2.1[A]
Wheat AES	10	56.9±4.9[C]		69.1±6.1[C]	23.2±5.9[C]		10.3±4.1[D]	37.6±8.5[B]		27.9±7.6[A]	21.1±2.0[B]		32.5±2.4[B]
Wheat AES (LS)	5	46.0±4.1		52.8±4.8	14.4±3.1		7.0±1.7	27.2±13.7		35.1±13.3	22.4±2.0		37.4±3.0
Wheat AES (TB)	5	67.7±5.6		85.3±3.7	32.0±10.3		13.5±8.1	48.0±9.2		20.6±7.5	19.8±3.5		27.6±2.1

Appendix 4.A List of species recorded on SRCs in the Thuringian Basin and in the Lower Saxon hills giving the naturalisation status, the habitat preference, the life form according to Raunkiær and the number of SRCs in which each species was recorded according to the position of the plot (*In* interior; *Ed* edge), the plantation age (in yrs.), the survey season (*Sp* spring; *Su* summer) and the tree species (*P* poplar; *W* willow). Naturarlisation status: *Arch.* archaeophyte, *neoph.* neophyte; habitat preferences: *G* grassland, *A* arable land, *N* nitrophilous vegetation, *F* forest, *Gen.* habitat generalists, life forms: *Hemicryp.* hemicryptophyte, *theroph.* therophyte, *chamaeph.* chamaephyte, *phaneroph.* phanerophyte). See chapter 4 for details on the assignment of the species to the different categories.

			Position	In	In	In	In	In	In	In	In	In	In	Ed	Ed	Ed	Ed	Ed	Ed	Ed	Ed
			Age (yrs.)	1	1	2	2	2	2	5-8	5-8	5-8	5-8	2	2	2	2	5-8	5-8	5-8	5-8
			Season	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su
			Tree species	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W
			Tot. # of SRCs	3	3	3	3	3	3	1	1	2	2	3	3	3	3	1	1	1	2
Species	**Status**	**Hab. pref.**	**Life form**																		
Achillea millefolium	Native	G	Hemicryp.	0	0	1	0	2	0	0	0	0	1	1	0	1	2	0	0	0	0
Aethusa cynapium	Arch.	A	Theroph.	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Agrostis stolonifera	Native	G	Hemicryp.	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0
Alopecurus myosuroides	Arch.	A	Theroph.	1	1	1	0	1	1	0	0	0	0	1	0	1	1	0	0	0	0
Amaranthus retroflexus	Neoph.	A	Theroph.	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Anagallis arvensis	Arch.	A	Theroph.	2	2	0	0	2	0	0	0	0	0	0	0	1	0	0	0	0	0
Anagallis foemina	Native	A	Theroph.	1	1	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0
Anthemis tinctoria	Arch.	N	Hemicryp.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Apera spica-venti	Native	A	Theroph.	1	0	0	0	1	1	0	0	0	0	1	0	1	1	0	0	0	0
Aphanes arvensis	Arch.	A	Theroph.	0	0	1	0	1	0	0	0	0	0	1	0	1	0	0	0	0	0
Arctium species	Native	N	Hemicryp.	1	2	2	2	2	2	1	0	0	0	2	2	2	1	1	1	1	1
Arenaria serpyllifolia	Native	G	Theroph.	0	0	2	2	2	2	0	0	0	0	2	2	2	2	0	0	0	0
Arrhenatherum elatius	Native	G	Hemicryp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Artemisia vulgaris	Native	N	Hemicryp.	1	1	1	1	2	1	0	0	0	1	1	1	2	1	0	1	0	1
Atriplex patula	Arch.	A	Theroph.	2	2	0	0	2	3	0	0	0	0	0	0	2	2	0	0	0	0

Position	In	In	In	In	In	In	In	In	In	In	Ed	Ed	Ed	Ed	Ed	Ed	Ed	Ed
Age (yrs.)	1	1	2	2	2	2	5-8	5-8	5-8	5-8	2	2	2	2	5-8	5-8	5-8	5-8
Season	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su
Tree species	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W
Tot. # of SRCs	3	3	3	3	3	3	1	1	2	2	3	3	3	3	1	1	1	2

Species	Status	Hab. pref.	Life form																		
Avena fatua	Arch.	N	Theroph.	1	2	0	0	1	1	0	0	0	0	0	0	1	2	0	0	0	0
Ballota nigra	Arch.	N	Chamaeph.	1	1	1	1	2	1	0	0	0	0	2	2	2	2	0	0	0	1
Bellis perennis	Arch.	G	Hemicryp.	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bromus hordeaceus	Native	G	Theroph.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Bromus secalinus	Arch.	A	Theroph.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Bromus sterilis	Arch.	A	Theroph.	1	0	2	3	3	3	0	0	0	0	3	3	3	2	0	0	0	0
Calamagrostis epigejos	Native	G	Geophyte	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Calystegia sepium	Native	N	Geophyte	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	1
Capsella bursa-pastoris	Native	A	Theroph.	0	1	3	3	2	2	0	0	0	0	3	3	3	3	0	0	0	0
Carduus acanthoides	Arch.	N	Hemicryp.	1	2	1	1	1	2	0	0	0	0	1	1	1	1	0	0	0	0
Centaurea cyanus	Arch.	A	Theroph.	1	0	1	0	1	1	0	0	0	0	1	0	1	1	0	0	0	0
Chenopodiaceae species	Arch.	A	Theroph.	0	0	1	1	0	0	0	0	0	0	3	1	0	0	0	0	0	0
Chenopodium album	Arch.	A	Theroph.	3	3	2	2	3	3	0	0	0	0	2	1	3	3	0	0	0	0
Chenopodium hybridum	Arch.	A	Theroph.	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cirsium arvense	Native	Gen.	Geophyte	2	3	3	3	3	3	0	0	1	2	3	3	3	3	1	1	1	2
Cirsium vulgare	Native	N	Hemicryp.	2	0	2	2	3	2	0	0	0	0	3	3	2	2	0	0	0	1
Convolvulus arvensis	Native	A	Geophyte	1	1	0	1	1	2	0	0	0	2	1	1	1	2	0	0	0	2
Crepis biennis	Arch.	G	Hemicryp.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Dactylis glomerata	Native	N	Hemicryp.	0	0	1	0	0	0	0	0	0	0	2	2	1	1	0	0	0	2
Daucus carota	Native	Gen.	Hemicryp.	0	0	1	1	1	1	0	0	0	0	2	2	0	0	0	1	0	0
Descurainia sophia	Arch.	A	Theroph.	1	1	2	2	2	2	0	0	0	0	2	2	2	2	0	0	0	0
Dipsacus fullonum	Arch.	N	Hemicryp.	0	1	0	1	0	1	0	0	0	0	1	1	1	1	0	0	0	0

			Position	In	In	In	In	In	In	In	In	In	In	Ed	Ed	Ed	Ed	Ed	Ed	Ed	Ed
			Age (yrs.)	1	1	2	2	2	2	5-8	5-8	5-8	5-8	2	2	2	2	5-8	5-8	5-8	5-8
			Season	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su
			Tree species	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W
			Tot. # of SRCs	3	3	3	3	3	3	1	1	2	2	3	3	3	3	1	1	1	2
Species	Status	Hab. pref.	Life form																		
Elymus repens	Native	N	Geophyte	2	2	3	3	2	2	0	1	2	2	3	3	3	2	1	1	1	2
Epilobium ciliatum	Neoph.	N	Hemicryp.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Epilobium lamyi	Native	N	Hemicryp.	0	0	0	1	1	1	0	0	0	1	0	1	0	1	0	1	0	1
Epilobium montanum	Native	N	Hemicryp.	0	0	0	0	1	1	0	0	0	0	0	0	1	1	0	0	0	0
Epilobium species	Native	N	Hemicryp.	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Equisetum arvense	Native	A	Geophyte	0	1	0	1	0	1	0	0	1	1	0	1	1	1	0	0	0	1
Erigeron acris	Native	Gen.	Theroph.	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0
Euphorbia exigua	Arch.	A	Theroph.	2	3	0	0	1	2	0	0	0	0	0	0	0	0	0	0	0	0
Euphorbia helioscopia	Arch.	A	Theroph.	2	3	0	1	0	2	0	0	0	0	1	1	1	0	0	0	0	0
Fallopia convolvulus	Arch.	A	Theroph.	2	3	0	0	2	1	0	0	0	1	0	0	1	3	0	0	0	1
Festuca pratensis	Native	G	Hemicryp.	1	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Fragaria vesca	Native	Gen.	Hemicryp.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Fumaria officinalis	Arch.	A	Theroph.	2	3	0	0	2	2	0	0	0	0	0	0	0	1	0	0	0	0
Fumaria species	Arch.	A	Theroph.	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0
Fumaria vaillantii	Arch.	A	Theroph.	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Galium album	Native	Gen.	Hemicryp.	0	0	1	0	1	0	0	0	0	0	1	1	1	1	0	1	0	2
Galium aparine	Native	A	Theroph.	3	2	3	3	3	3	1	1	0	2	3	2	3	3	1	1	1	2
Geranium dissectum	Arch.	A	Theroph.	0	0	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0
Geranium pusillum	Arch.	A	Theroph.	2	2	2	1	2	0	0	0	0	0	2	3	2	2	0	0	0	0
Geum urbanum	Native	N	Hemicryp.	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	1	0	1
Glechoma hederacea	Native	G	Geophyte	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	1	0	1
Helianthus annuus	Neoph.	Gen.	Theroph.	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Species	Status	Hab. pref.	Life form	In 1 Su P	In 1 Su W	In 2 Sp P	In 2 Sp W	In 2 Su P	In 2 Su W	In 5-8 Sp P	In 5-8 Sp W	In 5-8 Su P	In 5-8 Su W	Ed 2 Sp P	Ed 2 Sp W	Ed 2 Su P	Ed 2 Su W	Ed 5-8 Sp P	Ed 5-8 Sp W	Ed 5-8 Su P	Ed 5-8 Su W
Position				In	In	In	In	In	In	In	In	In	In	Ed	Ed	Ed	Ed	Ed	Ed	Ed	Ed
Age (yrs.)				1	1	2	2	2	2	5-8	5-8	5-8	5-8	2	2	2	2	5-8	5-8	5-8	5-8
Season				Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su
Tree species				P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W
Tot. # of SRCs				3	3	3	3	3	3	1	1	2	2	3	3	3	3	1	1	1	2
Helianthus tuberosus	Neoph.	Gen.	Geophyte	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Heracleum sphondylium	Native	N	Hemicryp.	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Hieracium lachenalii	Native	Gen.	Hemicryp.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Holcus lanatus	Native	G	Hemicryp.	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Holcus mollis	Native	Gen.	Geophyte	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Hypericum perforatum	Native	Gen.	Hemicryp.	0	0	1	0	1	0	0	0	0	1	0	0	1	0	0	0	0	1
Lactuca serriola	Native	A	Theroph.	0	0	2	3	2	3	0	0	0	0	1	2	2	3	0	0	0	1
Lamium amplexicaule	Arch.	A	Theroph.	1	1	2	1	1	0	0	0	0	0	1	2	0	0	0	0	0	0
Lamium purpureum	Arch.	A	Theroph.	0	1	0	1	0	1	0	0	0	0	0	1	1	0	0	1	0	1
Lathyrus tuberosus	Native	A	Hemicryp.	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1
Lepidium ruderale	Arch.	Gen.	Theroph.	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Lolium perenne	Native	G	Hemicryp.	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0
Malva neglecta	Arch.	A	Theroph.	0	0	1	1	0	1	0	0	0	0	2	1	1	1	0	0	0	0
Matricaria recutita	Arch.	A	Theroph.	1	1	0	0	1	1	0	0	0	0	1	0	1	1	0	0	0	0
Matricaria species	Arch.	A	Theroph.	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Medicago lupulina	Native	A	Theroph.	0	1	0	0	2	1	0	0	0	0	1	0	1	2	0	0	0	0
Milium effuSu	Native	F	Hemicryp.	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1
Mycelis muralis	Native	N	Hemicryp.	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Myosotis arvensis	Arch.	A	Theroph.	2	1	2	1	2	1	0	0	0	0	1	1	1	2	0	0	0	0
Onopordum acanthium	Arch.	N	Hemicryp.	0	2	1	1	0	1	0	0	0	0	1	2	1	2	0	0	0	0
Papaver dubium	Arch.	A	Theroph.	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Papaver rhoeas	Arch.	A	Theroph.	1	1	2	2	3	2	0	0	0	0	2	1	2	2	0	0	0	1

Species	Status	Hab. pref.	Life form	In 1 Su P	In 1 Su W	In 2 Sp P	In 2 Sp W	In 2 Su P	In 2 Su W	In 5-8 Sp P	In 5-8 Sp W	In 5-8 Su P	In 5-8 Su W	Ed 2 Sp P	Ed 2 Sp W	Ed 2 Su P	Ed 2 Su W	Ed 5-8 Sp P	Ed 5-8 Sp W	Ed 5-8 Su P	Ed 5-8 Su W
Tot. # of SRCs				3	3	3	3	3	3	1	1	2	2	3	3	3	3	1	1	1	2
Phragmites australis	Native	Gen.	Geophyte	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	1
Plantago major	Native	G	Hemicryp.	0	0	0	0	2	1	0	0	0	0	0	1	0	2	0	0	0	0
Poa annua	Native	A	Theroph.	1	0	3	2	1	0	0	0	0	0	3	3	1	1	0	0	0	0
Poa trivialis	Native	Gen.	Hemicryp.	0	0	0	0	1	1	0	1	0	2	0	0	1	2	0	0	0	1
Polygonum aviculare	Native	A	Theroph.	2	2	0	0	1	2	0	0	0	0	0	0	1	2	0	0	0	0
Polygonum lapathifolium	Native	A	Theroph.	1	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Polygonum persicaria	Native	A	Theroph.	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Potentilla reptans	Native	G	Hemicryp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Ranunculus repens	Native	Gen.	Hemicryp.	0	0	0	0	0	1	0	0	0	2	0	1	0	1	0	0	0	0
Reseda lutea	Arch.	N	Theroph.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Rumex crispus	Native	G	Hemicryp.	0	0	0	1	1	2	0	0	0	0	2	2	2	3	0	0	0	0
Rumex obtusifolius	Native	N	Hemicryp.	0	0	0	1	1	0	1	1	1	0	1	1	1	1	1	0	1	0
Senecio jacobaea	Native	Gen.	Hemicryp.	1	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Senecio vulgaris	Native	A	Theroph.	0	0	2	1	1	2	0	0	0	0	2	1	0	0	0	0	0	0
Silene dioica	Native	Gen.	Hemicryp.	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Silene latifolia	Native	N	Hemicryp.	0	0	0	1	1	1	0	0	0	0	0	2	0	2	0	0	0	0
Silene noctiflora	Arch.	A	Theroph.	1	0	1	0	2	0	0	0	0	0	1	0	1	1	0	0	0	0
Sinapis arvensis	Arch.	A	Theroph.	1	2	0	0	1	2	0	0	0	0	0	0	2	0	0	0	0	0
Sisymbrium officinale	Arch.	A	Theroph.	2	2	2	2	2	2	0	0	0	0	2	1	2	2	0	0	0	0
Solanum nigrum	Arch.	A	Theroph.	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Solidago virgaurea	Native	F	Hemicryp.	0	0	0	0	2	1	0	0	0	0	0	0	2	1	0	0	0	0
Sonchus asper	Native	A	Theroph.	1	2	0	0	2	2	0	0	0	0	0	0	1	1	0	0	0	1

			Position	In	In	In	In	In	In	In	In	In	In	Ed	Ed	Ed	Ed	Ed	Ed	Ed	Ed
			Age (yrs.)	1	1	2	2	2	2	5-8	5-8	5-8	5-8	2	2	2	2	5-8	5-8	5-8	5-8
			Season	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su
			Tree species	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W
			Tot. # of SRCs	3	3	3	3	3	3	1	1	2	2	3	3	3	3	1	1	1	2
Species	**Status**	**Hab. pref.**	**Life form**																		
Sonchus oleraceus	Native	A	Theroph.	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Sonchus species	Native	A	Theroph.	0	0	0	0	0	1	0	0	0	0	0	0	0	2	0	0	0	0
Stachys palustris	Native	G	Geophyte	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	1
Stellaria media	Arch.	A	Theroph.	1	0	2	1	1	0	0	0	0	0	2	2	1	1	0	0	0	0
Symphytum officinale	Native	Gen.	Hemicryp.	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1
Taraxacum species	Native	Gen.	Hemicryp.	0	0	3	3	3	3	0	1	0	0	3	3	3	3	1	1	0	0
Thlaspi arvense	Arch.	A	Theroph.	1	2	1	1	1	1	0	0	0	0	2	3	1	2	0	0	0	0
Torilis japonica	Native	N	Hemicryp.	0	0	0	0	1	1	0	0	1	2	0	0	2	1	0	0	0	2
Trifolium pratense	Native	G	Hemicryp.	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0
Trifolium repens	Native	G	Hemicryp.	0	0	0	1	0	0	0	0	0	0	1	1	1	1	0	0	0	0
Tripleurospermum perforatum	Arch.	A	Theroph.	3	3	3	3	3	3	0	0	0	0	3	3	3	3	0	0	0	0
Tussilago farfara	Native	N	Hemicryp.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Urtica dioica	Native	N	Chamaeph.	0	0	0	0	0	0	1	1	2	2	0	1	0	1	0	1	0	2
Veronica arvensis	Arch.	A	Theroph.	0	0	0	0	1	0	0	0	0	0	0	0	1	1	0	0	0	0
Veronica persica	Neoph.	A	Theroph.	1	1	1	1	1	1	0	1	0	0	2	2	2	2	0	0	0	0
Veronica polita	Arch.	A	Theroph.	1	1	1	1	1	0	0	0	0	0	2	1	1	1	0	0	0	0
Vicia cracca	Native	G	Hemicryp.	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1
Vicia tetrasperma	Native	A	Theroph.	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Viola arvensis	Arch.	A	Theroph.	2	3	3	3	2	3	0	0	0	0	3	3	3	3	0	0	0	0
Trees & shrubs																					
Acer campestre	Native	F	Phaneroph.	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0
Acer negundo	Neoph.	F	Phaneroph.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1

Species	Status	Hab. pref.	Life form	Position In	In	In	In	In	In	In	In	In	In	Ed	Ed	Ed	Ed	Ed	Ed	Ed	Ed
			Age (yrs.)	1	1	2	2	2	2	5-8	5-8	5-8	5-8	2	2	2	2	5-8	5-8	5-8	5-8
			Season	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su	Sp	Sp	Su	Su
			Tree species	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W	P	W
			Tot. # of SRCs	3	3	3	3	3	3	1	1	2	2	3	3	3	3	1	1	1	2
Acer pseudoplatanus	Native	F	Phaneroph.	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0
Carpinus betulus	Native	F	Phaneroph.	0	0	0	1	0	1	0	0	0	0	0	0	0	1	0	0	0	0
Cornus sanguinea	Native	F	Phaneroph.	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	1	0	1
Crataegus species	Native	F	Phaneroph.	0	0	0	1	1	1	0	1	0	0	0	0	0	0	0	0	0	0
Fraxinus excelsior	Native	F	Phaneroph.	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	1	0
Prunus avium	Native	F	Phaneroph.	0	0	1	0	0	0	0	1	0	1	0	0	0	0	0	1	1	1
Quercus species	Native	F	Phaneroph.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Ribes species	Native	F	Phaneroph.	0	0	0	1	0	0	0	1	0	1	0	0	0	0	0	0	0	0
Rosa species	Native	F	Phaneroph.	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	1	0	1
Rubus fruticosus ag.	Native	F	Phaneroph.	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Sambucus nigra	Native	N	Phaneroph.	0	0	1	0	1	0	0	0	1	0	1	1	0	0	0	0	0	0
Sorbus aucuparia	Native	N	Phaneroph.	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Viburnum species	Native	F	Phaneroph.	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Crop species																					
Brassica napus	Arch.	A	Theroph.	3	3	1	1	1	0	0	0	0	0	2	2	2	0	0	0	0	0
Hordeum vulgare	Arch.	A	Theroph.	1	1	1	1	1	0	0	0	0	0	1	1	1	0	0	0	0	0
Secale cereale	Arch.	A	Theroph.	1	1	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0
Triticum aestivum	Arch.	A	Theroph.	1	2	0	0	2	1	0	0	0	0	0	0	0	0	0	0	0	0

Further publications

Peer-reviewed journals

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Seifert, C. (2013) Auswirkung des Anbaus von Bioenergiepflanzen auf die Pflanzenartenvielfalt der Agrarlandschaft in Südniedersachsen und im Thüringer Becken. In: Feit, U. & Korn, H. (eds); Treffpunkt Biologische Vielfalt XII – Interdisziplinärer Forschungsaustausch im Rahmen des Übereinkommens über die biologische Vielfalt. *BfN-Skripten* **335**, 71–76.

Eigenständigkeitserklärung

Hiermit bestätige ich die vorliegende Dissertationsschrift eigenständig verfasst und keine anderen als die im Text angegebenen Quellen und Hilfsmittel verwendet zu haben.

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Ort, Datum, Unterschrift