



DISSERTATION

Fragmentation of landscapes: modelling ecosystem services of transition zones

by Martin Schmidt

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A map is not the territory it represents,

but, if correct, it has a similar structure to the territory, which accounts for its usefulness.

Alfred Korzybski

Abstract

For millennia, humans have affected landscapes all over the world. Due to horizontal expansion, agriculture plays a major role in the process of fragmentation. This process is caused by a substitution of natural habitats by agricultural land leading to agricultural landscapes. These landscapes are characterized by an alternation of agriculture and other land use like forests. In addition, there are landscape elements of natural origin like small water bodies. Areas of different land use are beside each other like patches, or fragments. They are physically distinguishable which makes them look like a patchwork from an aerial perspective. These fragments are each an own ecosystem with conditions and properties that differ from their adjacent fragments. As open systems, they are in exchange of information, matter and energy across their boundaries. These boundary areas are called transition zones. Here, the habitat properties and environmental conditions are altered compared to the interior of the fragments. This changes the abundance and the composition of species in the transition zones, which in turn has a feedback effect on the environmental conditions.

The literature mainly offers information and insights on species abundance and composition in forested transition zones. Abiotic effects, the gradual changes in energy and matter, received less attention. In addition, little is known about non-forested transition zones. For example, the effects on agricultural yield in transition zones of an altered microclimate, matter dynamics or different light regimes are hardly researched or understood. The processes in transition zones are closely connected with altered provisioning and regulating ecosystem services. To disentangle the mechanisms and to upscale the effects, models can be used.

My thesis provides insights into these topics: literature was reviewed and a conceptual framework for the quantitative description of gradients of matter and energy in transition zones was introduced. The results of measurements of environmental gradients like microclimate, aboveground biomass and soil carbon and nitrogen content are presented that span from within the forest into arable land. Both the measurements and the literature review could not validate a transition zone of 100 m for abiotic effects. Although this value is often reported and used in the literature, it is likely to be smaller.

Further, the measurements suggest that on the one hand trees in transition zones are smaller compared to those in the interior of the fragments, while on the other hand less biomass was measured in the arable lands' transition zone. These results support the hypothesis that less carbon is stored in the aboveground biomass in transition zones. The soil at the edge (zero line) between adjacent forest and arable land contains more nitrogen and carbon content compared to the interior of the fragments. One-year measurements in the transition zone also provided evidence that microclimate is different compared to the fragments' interior.

To predict the possible yield decreases that transition zones might cause, a modelling approach was developed. Using a small virtual landscape, I modelled the effect of a forest fragment shading the adjacent arable land and the effects of this on yield using the MONICA crop growth model. In the transition zone yield was less compared to the interior due to shading. The results of the simulations were upscaled to the landscape level and exemplarily calculated for the arable land of a whole region in Brandenburg, Germany.

The major findings of my thesis are: (1) Transition zones are likely to be much smaller than assumed in the scientific literature; (2) transition zones aren't solely a phenomenon of forested ecosystems, but significantly extend into arable land as well; (3) empirical and modelling results show that transition zones encompass biotic and abiotic changes that are likely to be important to a variety of agricultural landscape ecosystem services.

Zusammenfassung

Seit Jahrtausenden werden Landschaften weltweit maßgeblich durch den Menschen gestaltet. Insbesondere die Landwirtschaft hat durch Fragmentierung, der teilweisen Umwandlung natürlicher Lebensräume in landwirtschaftlich genutzte Flächen, großen Einfluss, so dass Agrarlandschaften entstanden. Diese zeichnen sich durch einen Wechsel von agrarischer und anderer Nutzung, wie beispielsweise Forst, aus. Hinzu kommen Flächen, die auf eine natürliche Entstehung zurückzuführen sind, wie etwa Kleingewässer. Kleinere und größere Flächen der unterschiedlichen Nutzung liegen als Flecken bzw. Fragmente nebeneinander. Durch die physische Differenzierbarkeit der Flächennutzung aus der Vogelperspektive werden Agrarlandschaften oft auch als Flickwerk ("Patchwork") bezeichnet. Diese Fragmente sind Ökosysteme, die sich in ihren Eigenschaften voneinander unterscheiden.

Die Fragmente als Ökosysteme sind offene und komplexe Systeme und stehen im Austausch mit angrenzenden Fragmenten. Die Bereiche, in denen der Austausch von Stoffen, Energie und Informationen stattfindet, sind deren Übergangszonen. Durch den Austausch verändern sich die vorherrschenden Eigenschaften der jeweils angrenzenden Fragmente in den Übergangszonen. Stoffflüsse beeinflussen dabei die in den Übergangszonen lebenden Organismen und können die Artenzusammensetzung und Population verändern. Gleichwohl hat dies Rückkopplungseffekte auf die Flüsse von Stoffen, Informationen und Energie selbst.

In der Forschung ist bereits viel über die Auswirkungen auf Organismen in den Übergangszonen bekannt, insbesondere für bewaldete Gebiete. Weniger beforscht sind abiotische Effekte, insbesondere die graduellen Veränderungen von Stoffen und Energie in der Übergangszone. Diese sind jedoch eng verwoben in die Prozesse, die zu regulierenden und bereitstellenden Ökosystemleistungen wie beispielsweise landwirtschaftlichen Erträgen oder Kohlenstoffspeicherung beitragen. Darüber hinaus gibt es wenig Forschung zu den Übergangszonen von nicht-bewaldeten Übergangszonen, wie etwa Äckern.

In der vorliegenden Arbeit präsentiere ich die Ergebnisse einer Literaturrecherche und einen Ansatz zur quantitativen Beschreibung von Stoff- und Energieflüssen in Übergangszonen. Darüber hinaus analysiere ich Messungen eben jener abiotischen Effekte in Übergangszonen. Sowohl die Messungen als auch die Auswertung der Literatur ergab, dass viele Autoren die Übergangszone in Bezug auf Umweltgradienten und deren Einfluss auf Ökosystemleistungen mit 100 m überschätzen. Sie ist oft kleiner.

Die Messungen ergaben außerdem, dass Bäume in der Übergangszone kleiner sind und dadurch vermutlich weniger Kohlenstoff speichern als vergleichbare Bäume im Inneren dieser Fragmente. An Wald angrenzende Ackerkulturen zeigen ebenfalls einen geringeren Aufwuchs an Biomasse. Im Boden genau an der Grenze zwischen Wald- und Ackerfragmenten waren sowohl Stickstoff als

auch Kohlenstoff erhöht. Einjährige Messungen in Brandenburg ergaben, dass das Mikroklima in der Übergangszone im Vergleich zum Inneren der Fragmente ebenfalls verändert war.

Um genauer zu verstehen, was die Ertragsminderung in der ackerbaulichen Übergangszone induziert, wurde ein Modellierungsansatz entwickelt. Die Beschattung durch einen virtuellen Wald wurde im agrarischen Simulationsmodell MONICA als Variable benutzt, um eine potentielle Ertragsminderung zu simulieren. Ein Minderertrag in der Übergangszone konnte auf diese Weise nachgewiesen und mit einer verminderten Solarstrahlung in Verbindung gebracht werden. Die simulierten Ergebnisse wurden anschließend für die Beispielregion Brandenburg für die gesamte landwirtschaftlich genutzte Fläche skaliert.

Insbesondere in drei Punkten trägt diese Arbeit zum wissenschaftlichen Fortschritt bei: 1) Übergangszonen sind sehr wahrscheinlich kleiner als bislang in der wissenschaftlichen Literatur angenommen; 2) Übergangszonen sollten ganzheitlich, über die Grenze zwischen Wald und Feld hinweg betrachtet werden; 3) Messungen und Modellierung zeigen einen Zusammenhang zwischen Mikroklima, Stoffdynamik und Ökosystemleistungen in Übergangszonen von Agrarlandschaften.

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1. Introduction

1.1. Motivation

1.1.1. Fragmentation

Landscapes all over the world are massively influenced by human activities. What natural or semi-natural habitats remain are becoming more and more fragmented, leading to vulnerable ecosystems and endangered native species. A main reason for this is that the area of contiguous intact forest has been decreasing for millennia due to deforestation and the expansion of agricultural land (Williams, 2006). The intrusion of agriculture in landscapes often alters their original properties and functions.

Agricultural landscapes comprise a variety of land uses, but are dominated by agriculture. Agriculture is characterized by land use practices like crop rotation, soil management or mowing and grazing. Most often, these land use practices alter the natural conditions in a way, that areas with different land uses are easily distinguishable because they are physically separated (Ries et al., 2004a). Especially intensive land use practices and landownership make a distinction of arable land and e.g. forests possible from an aerial perspective. These distinct areas are often called patches or fragments (Wu and Loucks, 1995). In the development of agricultural landscapes arable land often replace natural habitat fragments. The area of the latter gets smaller in total and the fragments are often farer apart from each other.

Fragments in agricultural landscapes are complex ecological systems. They differ from each other in the composition of species and non-living components. Constantly, the process of fragmentation of agricultural landscapes largely changes these ecosystems' properties and functioning by mixing zones of different habitat quality and ecological features at the boundaries of fragments. These zones are called transition zones. They are areas of active and passive exchange of matter, information and energy and have different properties than the native forest, plain pasture or arable lands present in the landscape (Schmidt et al., 2017).

1.1.2. Transition zones

Transition zones¹ and the influence on environmental conditions in transition zones have been intensely studied over the last decades (e.g. Chen et al., 1993; Murcia, 1995; Ries et al., 2004). In

¹ In order to be consistent in this thesis, definitions that are introduced in Section 2 are already used in the introduction. See Table 2.1 for all definitions.

many articles the term 'edge' is used although a transition zone is meant. The term 'edge' is rather vague as it is often a proxy for a point and a length in the literature. Transitional gradients can be described with a depth, distance or length, an index for the spatial extent of a single gradient in a transition zone measured from the zero point between two fragments (e.g. Chen et al., 1995; Gehlhausen et al., 2000; Harper et al., 2005). The zero point can be "the point of edge creation or point of edge maintenance" (Murcia, 1995). This concept can be extended to a zero line from which the length of significant transitional gradients has positive or negative values respectively diametrically opposed directions perpendicular to the zero line. The difference of an environmental condition expressed in a metric value compared for adjacent patches is called magnitude of transitional gradients (Harper et al., 2005).

The circumstance of various definitions in the scientific literature has led to conceptual frameworks to identify and define edges as ecological boundaries in two or three dimensions (Cadenasso et al., 2003a; Strayer et al., 2003). Although Cadenasso et al. (2003a) already define ecological boundaries as "not limited by system or scale", Yarrow and Marín (2007) subordinate ecological boundaries and 'edge' to the more generalized term 'transition zones'. Yet, a consistent and comprehensive framework to describe and specifically quantify ecological processes in transition zones, their magnitude and extent is not presented in the literature, to my knowledge. This lack of knowledge led to the first research question.

First Research Question

Can a framework be developed to quantitatively describe the ecological processes and their changes across transition zones?

The necessity for clear definitions and a quantitative framework follows from the literature. Some authors use a rather random length of transitional gradients from the zero line (edge) into the fragment to calculate global shares of transition in fragments. For example, 74% of the total forest area in England (Riutta et al., 2014), 74% of semi-deciduous savanna forest (Hennenberg, 2005), almost 50% of all Brazilian Atlantic rain forests (Ribeiro et al., 2009) and 44% of continental United States forest (Riitters et al., 2002) are within a transition zone of 90 to 100 m. Globally, Haddad et al. (2015) calculated that 20% of forested land was located in a 100 m transition zone within forests. The length of 100 m is most often not measured in these papers. It is only a proxy for the calculation of the area and has to be compared with measurements for the relevant environmental gradients and ecosystem services in other papers. This is only possible with a common quantitative framework that makes results comparable across continents and between scientific disciplines.

Beyond the spatial extent of transition zones it is important to have a comprehensive knowledge on the magnitude of environmental gradients and the degree of alterations of ecological processes. Information on the magnitude allows a categorisation whether the effects in the transition zones are positive or negative, and if so, for whom or what. Further, a comparison of processes and variables allows finding correlations and causalities. This would help to disentangle the processes, understand the importance of transition zones and calculate the effects, e.g. by multiplying the amount of a yield reduction due to shading in a transition zone by the respective area.

1.1.3. Abiotic effects

Fragmentation affects microclimate, which leads to changes in the abundance and diversity of plant communities. This, in turn, can further alter the microclimate (Chen et al., 1992; Laurance et al., 2011; Saunders et al., 1991). Different taxa and cycles of matter production and decomposition respond positively or negatively to the changes in microclimate caused by transitions (Godefroid et al., 2006; Heithecker and Halpern, 2007; Magnago et al., 2015). With respect to altered decomposition rates and primary production (Chen et al., 1992) within these transition zones, Ewers and Banks-Leite (2013) hypothesize that as global climate changes, microclimatic transitional gradients will expand and get steeper and the transition zones of forests will gain more and more importance. Merging the results of different experiments could lead to a more comprehensive understanding of the ecological processes and ecosystem functions in transition zones. The current lack of knowledge consolidation around the magnitude of transitional gradients and the spatial extent of transition zones led to the second research question².

Second Research Question

What is currently known about how biogeochemical and biogeophysical processes and derived ecosystem services change across transition zones?

Most of the research concerning transition zones and their effects has focused on diversity and abundance of animals and plants (Ries et al., 2004). Compared to biotic effects, abiotic effects have rarely been investigated, and if so, the studies on abiotic effects are hardly comparable (Murcia, 1995). Important abiotic factors include air and soil temperature as well as moisture, vapour pressure, solar radiation and wind (Chen et al., 1995; Gehlhausen et al., 2000; Matlack, 1993). These microclimatic factors are key drivers of the primary and secondary responses to transition zones,

² The ecosystem services approach was used for a better understanding and to make variables from different disciplines comparable.

including matter cycling (Chen et al., 1995) or the diversity and abundance of species (Ries et al., 2004). Microclimate effects are limited spatially, defined as those less than 2 km in horizontal extension (in case of fragments perpendicular to the zero line) as well as temporally, describing processes which can be very short in terms of 'climate', e.g. one growing season. In particular, the last point has to be taken under detailed consideration because of the dependence of the microclimate on the wider regional climate (Didham and Lawton, 1999). Little is known about seasonal changes of microclimate in transition zones. Owing to the fact that most studies have taken place in America, Australia and the Tropics, investigations of abiotic factors in transition zones should be made in the European temperate climate zone to have a better understanding about climate influence on microclimate in these regions. Moreover, this would be an important contribution to a global understanding of transitional gradients.

Apart from microclimatic factors, transitional gradients of matter dynamics are rarely investigated. This is disadvantageous, as matter dynamics are strongly driven by microclimate and vice versa, and matter dynamics can have important influences on microclimate in transition zones. Existing literature describes higher decomposition rates in forested transition zones (3 to 4 times within 50 m) due to higher soil moisture, fungi abundance and temperature (Chen et al., 1993; Didham, 1998; Saunders et al., 1991). These effects, in addition to microclimatic stressors (Ziter et al., 2014) cause lower carbon stocks in transition zones compared to the forest interior (de Paula et al., 2011). For nitrogen, a higher qualitative availability and a narrower C:N-ratio in comparison to grassland (Johnson and Wedin, 1997) have been found. Higher nitrate rates in transition zones compared to forest interiors and higher dissolved inorganic nitrogen levels (Weathers et al., 2001) are also mentioned in the literature.

Most of the research on the biologic, physical, and microclimate dynamics of transition zones has been performed in forested transition zones. Very little is known about non-forested transition zones. Moreover, only a few studies exist that measure microclimate and matter dynamics along transects that span gradients from agricultural land into forest. Also the temporal resolution and timespan of measurements is often limited to a few weeks or just a couple points in time. The third research question is meant to add knowledge in the light of these gaps.

Third Research Question

What are length and magnitude of environmental gradients in transition zones from forest to agricultural land in the temperate zone?

1.1.4. Modelling

The first three research questions focus on improving our insight into the spatial extent of and alterations caused by transition zones that affect ecosystem services. To develop a deeper understanding of transition zones, the complexity of changes in ecological processes across transition zones and the mechanisms behind these changes have to be disentangled, especially at landscape scales. Yet, there are currently limited approaches for modelling spatial interactions within agricultural landscapes that take into account the influence of transitional gradients on ecosystem services. To simulate these effects in transition zones, models have to be spatially explicit, which means they have to take into account environmental gradients in every transition zone in the landscape. Moreover, the magnitude and dynamics of these gradients have to be considered. Ideally, the different fragments simulated will be able to virtually exchange fluxes of matter, energy and information (e.g. genetic information). Comparing the conditions in transition zones to the interiors of fragments may help reveal relevant differences that at a landscape-level should alter ecosystem service or plant growth models, especially in highly fragmented agricultural landscapes where the length of ecosystem boundaries per unit area is high.

With respect to the length of significant transitional gradients the models could be quite simple: most abiotic factors seem to be a function of distance, e.g. decrease with increasing distance (Didham and Lawton, 1999; Murcia, 1995). Incorporating transition zones into models that predict matter dynamics and microclimatic effects could have important implications for global C storage calculations or models that predict matter turnover rates, e.g. in forested transition zones. Another important scientific discipline with respect to transition zones in agricultural landscapes is crop modelling. With respect to agro-ecosystems, the impact of shading on yield and soil moisture are the most important abiotic effects of edges and transition zones. As yield is one of the most important provisioning ecosystem services the fourth research question was formulated according to that.

Fourth Research Question

What is the impact of shading on yield in transition zones of agricultural land and how can we model them?

1.2. Structure of the thesis and contributions

To answer the above listed research questions this thesis comprises three research articles and one data publication (Table 1.1). The literature review (Section 2) is open access and published in a peer-reviewed journal, two research articles are published in open access peer-reviewed journals (Section 3 and 4) and the data publication is published in an open access peer-reviewed journal (Section “Accompanying Data Article”). All articles are related to each other, with later articles building on the concepts and knowledge gained in earlier articles (Figure 1.1). I have written all above-mentioned publications as first author. The detailed contributions follow below.

Table 1.1 Overview on research questions (RQ) for this thesis and the corresponding research articles that answer them.

	First RQ	Second RQ	Third RQ	Fourth RQ
First Article	█			
Second Article			█	
Third Article			█	█

1.2.1. First article

Schmidt, M., Jochheim, H., Kersebaum, K.-C., Lischeid, G., Nendel, C., 2017. Gradients of microclimate, carbon and nitrogen in transition zones of fragmented landscapes – a review. *Agricultural and Forest Meteorology* 232, 659–671.
<https://doi.org/10.1016/j.agrformet.2016.10.022>

The first and second research questions were answered by conducting a literature review. In this first article (Section 2), a framework to quantitatively describe environmental gradients in transition zones is proposed. I was the lead author and conducted the literature review. The framework was discussed with colleagues that are either experts in agroecosystem modelling (K.-C. Kersebaum) or forest ecosystem modelling (H. Jochheim). The general concept for the article was developed with the help of and supervised by C. Nendel and G. Lischeid. All authors reviewed and commented on the final draft of the article.

1.2.2. Second article

Schmidt, M., Lischeid, G., Nendel, C., 2019. Microclimate and matter dynamics in transition zones of forest to arable land. *Agricultural and Forest Meteorology* 268, 1-10.
<https://doi.org/10.1016/j.agrformet.2019.01.001>

The third research question was the focus of the second article (Section 3). With the insights developed from the review (Section 2), experiments were designed and conducted. After more

than one year of measurements, microclimate time series were analysed by G. Lischeid using principal component analysis. G. Lischeid also supported all other statistical analyses. C. Nendel and G. Lischeid helped to develop the measurement design as well as reviewed and commented on the final draft. I was the lead author, conducted all measurements and was responsible for all analyses except the principal component analysis. All authors reviewed and commented on the final draft of the article.

1.2.3. Third article

Schmidt, M., Nendel, C., Funk, R., Mitchell, M., Lischeid, G., 2019. Modeling Yields Response to Shading in the Field-to-Forest Transition Zones in Heterogeneous Landscapes. *Agriculture* 9, 6. <https://doi.org/10.3390/agriculture9010006>

To answer the fourth research question, the effects of changes in solar radiation and soil moisture in transition zones on wheat and maize yield were simulated. The results were analysed in the third research article (Section 4). During a research stay at the University of British Columbia in Vancouver (Canada) with M. Mitchell, I developed the modelling approach for solar radiation.

The virtual landscape simulated for this article was developed and analysed by R. Funk and me. Crop yields were simulated with the help of C. Nendel using the MONICA crop growth model.

I was the lead author for the third article and was responsible for the development of the modelling framework (based on Section 2), the final analyses and data provision. The general concept for the article was developed with the help of and supervised by C. Nendel and G. Lischeid. All authors reviewed and commented on the final draft of the article.

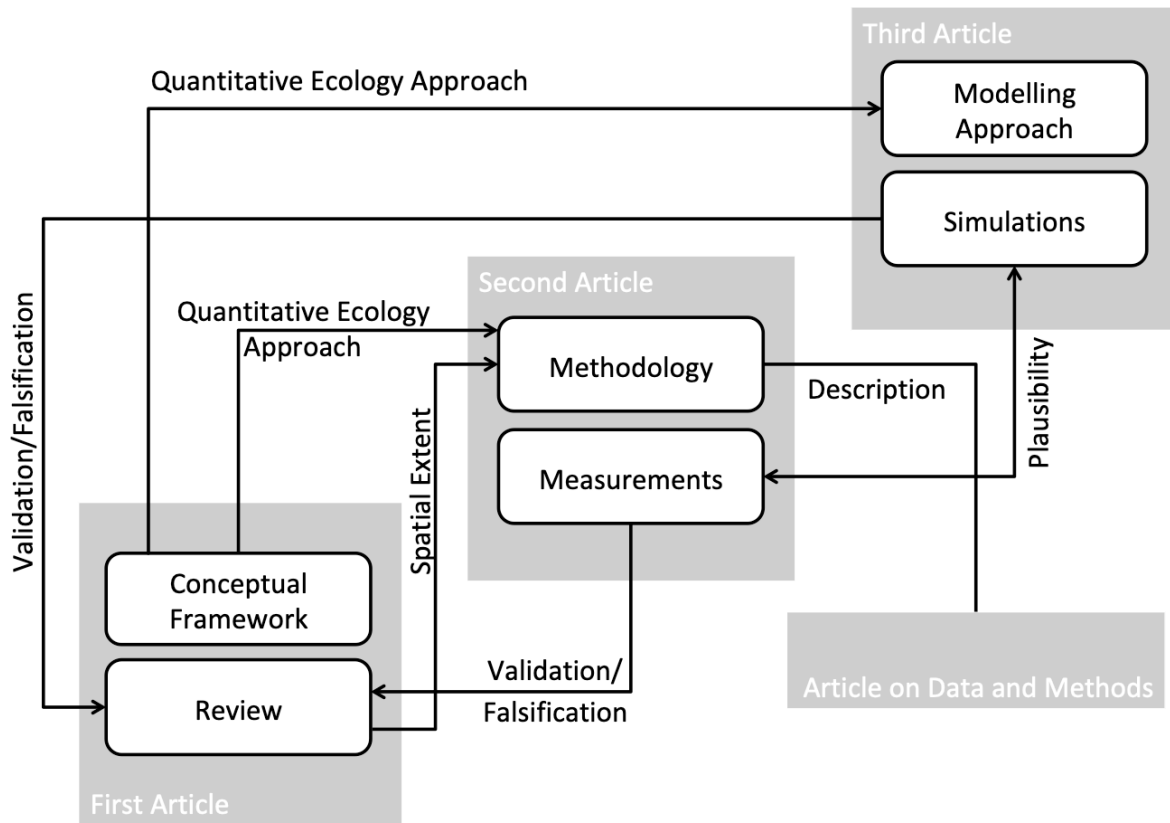


Figure 1.1 Overview on connections and correlations between the research articles for this thesis.

1.2.4. Data article

Schmidt, M., Lischeid, G., Nendel, C., 2018. Data on and methodology for measurements of microclimate and matter dynamics in transition zones between forest and adjacent arable land. *One Ecosystem* 3: e24295. <https://doi.org/10.3897/oneeco.3.e24295>

The data article (Section 8) is an accompanying article that openly publishes the data from the second article (Section 3). It includes a detailed description of the methodology for the second research article. Moreover, it contains the R code that was used to clean the raw data to be most transparent and enable reproducibility of the results. I was the lead author for the data publication. G. Lischeid and C. Nendel reviewed and commented the final draft. They also supervised me in conducting the experiments and in methodology.

2. Gradients of microclimate, carbon and nitrogen in transition zones of fragmented landscapes – a review³

Abstract

Fragmentation of landscapes creates a transition zone in between natural habitats or different kinds of land use. In forested and agricultural landscapes with transition zones, microclimate and matter cycling are markedly altered. This probably accelerates and is intensified by global warming. However, there is no consensus on defining transition zones and quantifying relevant variables for microclimate and matter cycling across disciplines. This article is an attempt to a) revise definitions and offer a framework for quantitative ecologists, b) review the literature on microclimate and matter cycling in transition zones and c) summarise this information using meta-analysis to better understand bio-geochemical and bio-geophysical processes and their spatial extent in transition zones. We expect altered conditions in soils of transition zones to be 10–20 m with a maximum of 50 m, and 25–50 m for above-ground space with a maximum of 125 m.

Keywords: Edge effects, Ecological boundaries, Matter cycling, Matter dynamics, Framework quantitative ecology, Ecotone hierarchy

2.1. Introduction

Most landscapes are composed of different kinds of ecosystems, which are nested but also often physically separated into fragments (Ries et al., 2004b). Fragmented forested and agricultural landscapes are characterised by the occurrence of discontinuities or variations in prevalent or native land cover and habitat properties (Strayer et al., 2003).

In quantitative terms, they differ from other landscapes by having a lower average size of the fragment, a lower interior-to-edge ratio (see Section 2.2. for definitions) and an increase in isolation and distance to each other for patches of similar properties (Mitchell et al., 2014; Saunders et al., 1991).

Fragmented landscapes are not static per se but are rather in a continuous natural process of fragmentation. Drivers of fragmentation act on various spatio-temporal scales: geogenic (e.g. differing parent rock), topographical (relief), geomorphological (e.g. kettle holes), pedogenic (e.g. climate), hydrological (e.g. groundwater or rivers), and phytological (e.g. seed dispersal or

³ This section is an Open Access publication: Schmidt, M., Jochheim, H., Kersebaum, K.-C., Lischeid, G., Nendel, C., 2017. Gradients of microclimate, carbon and nitrogen in transition zones of fragmented landscapes – a review. *Agricultural and Forest Meteorology* 232, 659–671. doi:10.1016/j.agrformet.2016.10.022

succession) (Cadenasso et al., 2003a; Wu and David, 2002). Moreover, landscapes are fragmented by sudden events, such as wind throw, erosion (water or wind), volcanic eruptions, earthquakes, pests and diseases, fires or floods (e.g. Braithwaite and Mallik, 2012; Laurance and Curran, 2008).

The total area of forest has been decreasing for millennia (probably for more than 6000 years) due to deforestation and the intrusion of agricultural land (FAO, 2012; Williams, 2006); currently, the area of contiguous intact forest is decreasing twice as quickly as the total area of forest (Riitters et al., 2015). Fragments of native vegetation are often surrounded by managed land (Saunders et al., 1991). This anthropogenically driven fragmentation of landscapes largely changes the land's properties and functioning by mixing zones of different habitat quality and ecological features. The main man-made drivers are agriculture and forestry (e.g. horizontal expansion, logging), urbanisation (Liu et al., 2016), rural development (e.g. road construction) and energy production (e.g. dams). In addition to natural sudden events, man-made disasters such as fires or pollution (e.g. chemical spill, nitrogen deposition, acid rain) also cause fragmentation.

Fragmentation leads to biome patches with zones of transition in between them. These transition zones are characterised by active and passive exchange of matter, energy and information – their properties differ from native forest, plain pasture and agricultural land (Gosz, 1992; Wiens et al., 1985). In fact, 74% of the total forest area in England (Riutta et al., 2014), 74% of semi-deciduous savanna forest in north-east Ivory Coast (Hennenberg, 2005; Hennenberg et al., 2008), almost 50% of all Brazilian Atlantic rainforests (Ribeiro et al., 2009), 44% of continental United States forest (Riitters et al., 2002) and 40% of the total forest area in Bavaria (Germany) (Spangenberg and Kölling, 2004) have been defined as being located within a transition zone of 90–100 m from the forest edge. Globally, Haddad et al. (2015) calculated that 20% of forested land was located in a 100 m transition zone within forests.

Fragmentation affects the local climate. For example, the air within and above cropland is warmer and drier than the moister and cooler air in adjacent forests (Ewers and Banks-Leite, 2013; Laurance et al., 2011). The different microclimate which evolves within the fragments fosters the establishment of differently adapted plant communities, which in turn also influence the microclimate (Chen et al., 1992; Laurance et al., 2011; van Rooyen et al., 2011; Saunders et al., 1999). Some taxa clearly respond positively or negatively to changes in microclimate caused by fragmentation (Godefroid et al., 2006; Heithecker and Halpern, 2007; Magnago et al., 2015). Research on edges conducted in recent decades mainly described them as hot spots for biodiversity and evolutionary processes (Kark and van Rensburg, 2006; Lidicker, 1999 see Ries et al., 2004), which will not be addressed in this review.

Within transition zones, microclimate alters matter cycling (Laurance et al., 2007, 2011; Nascimento and Laurance, 2004). In forested transition zones, above-ground carbon storage

capacity has been found to be as little as half that of the forest interior (Paula et al., 2011). Pütz et al. (2014) calculated a total of 200 Tg carbon gas emissions per year due to forest degradation (fragmentation) in tropical forests; this is one-fifth of all emissions caused by deforestation. Moreover, in addition to the carbon gas emissions caused by deforestation, simulations by Laurance et al. (1998) suggest that another 22–149 Tg C loss per year is caused by fragmentation of tropical forests worldwide. Due to altered decomposition rates and primary production (Chen et al., 1992) within these transition zones, Ewers and Banks-Leite (2013) hypothesise that, as global climate change take place, transition zones will increasingly gain in importance.

The relevance of transition zones is thus substantially increasing. However, up to this point, there is no consensus among scientists with respect to definitions and investigation strategies. A synthesis of the existing knowledge on matter dynamics and the connection to microclimate in transition zones is currently lacking. This review provides a first attempt to fill this gap.

The aim of this review is to a) address the various definitions of ‘edge effects’, b) review the literature on microclimate and matter cycling in transition zones and c) summarise this information using meta-analysis to better understand bio-geochemical and bio-geophysical processes in transition zones (Figure 2.1).

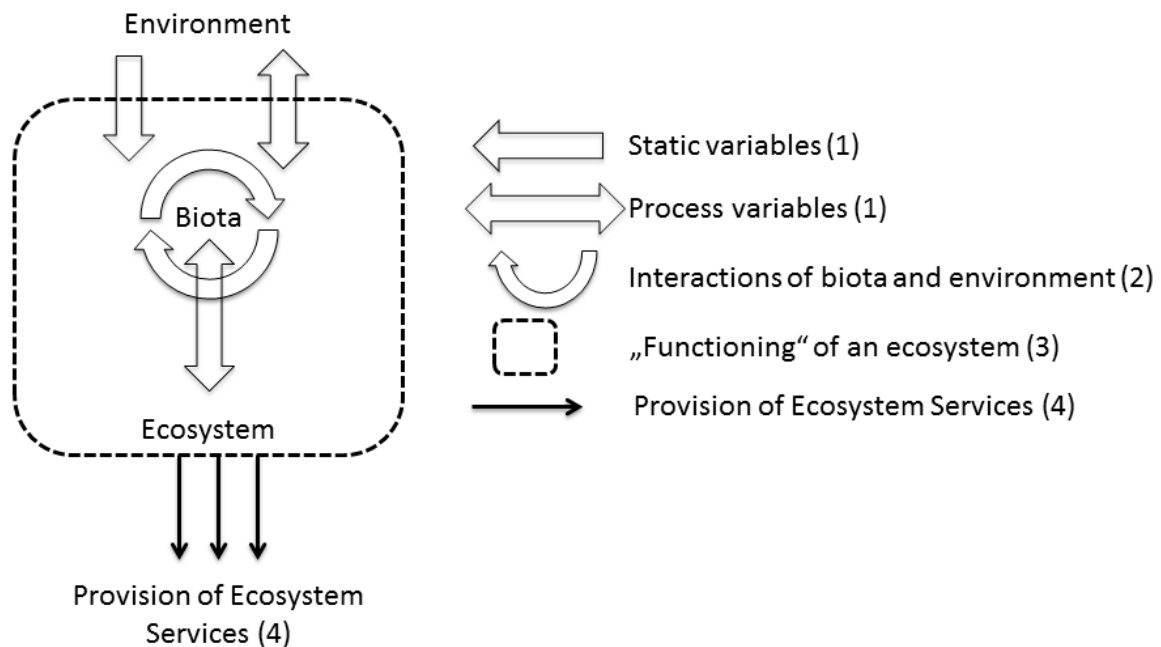


Figure 2.1 Breakdown of functions concerning ecosystems according to Jax (2005).

The meta-analysis consisted of a literature search for the expressions ‘edge effect’, ‘forest’, ‘microclimate’, ‘ecotone’, ‘transition zone’, ‘pasture’, ‘agriculture’, ‘carbon’, ‘nitrogen’, ‘matter and

nutrient dynamics' and 'cycling'. To define the spatial extent of the influence of transition zones, the maximum distance had to be stated as measured from the zero line (see Figure 2.2 or Table 2.1) perpendicularly in one direction. If a range was given, both values were used. Although the magnitude of variables has not been taken into account, studies that reported no significance were omitted.

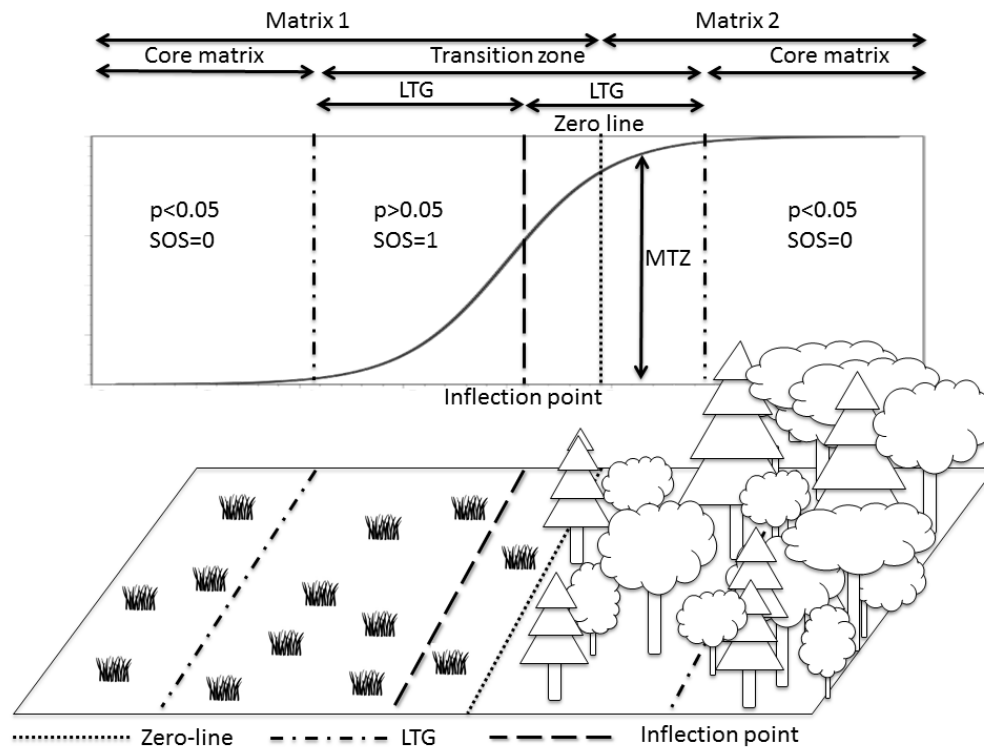


Figure 2.2 Example of the application of the definitional toolbox for the quantitative distinction of components of fragmented landscapes; here, for an agricultural field (left) and a forested area (right).

Table 2.1 Toolbox for a quantitative description of transition zones in fragmented landscapes.

Term	Definition	Synonyms used in literature
Landscape	A scale- and system-neutral conglomeration of matrices and their transitions, differentiated by land use and type of vegetation (Cadenasso et al., 2003a)	
Matrix	"Spatial domain where processes, properties or magnitudes" of physical, chemical or biological "variables are sufficiently distinct from those of its neighbors to warrant their segregation" (Woo, 2004)	Patch, fragment, ecosystem, biome, habitat (Fagan et al., 2003), exterior, environment, borders, biome ecotone, 'island'
Core matrix	Area in which biotic and abiotic properties do not change significantly over mesoscale (relative homogeneity; depends on research question)	Interior, 'end states' (Peters et al., 2006), core area (Fagan et al., 2003), remnant area, climax state
Transition zone ^a	Spatio-temporal variable entity with functional and structural gradients in between adjacent core matrices	Boundary, edge, corridor, ecotone, ecocline, ecological ecotone, buffer zone, interference zone, hybrid zones, space-segment, (see Hufkens et al. (2009) for an overview of ecocline and ecotone and Kark and van Rensburg (2006) for a history of ecotones)

Term	Definition	Synonyms used in literature
Solitary matrix	Matrix which – owing to its small size – consists of only a transition zone without a core matrix	Solitary fragment
Zero line ^a	The structural boundary of matrices; a point or line of edge creation and edge maintenance (Murcia, 1995), or land use change and its maintenance (“last unharvested tree trunk” Baker et al. (2016)); static or dynamic	Boundary, edge, barrier, delimitation, interface, border, demarcation line, delineation, borderline
Inflection point ^a	The functional boundary of matrices; defined as the line of maximum gradient in a transition zone	
Magnitude of variables in the transition zones (MTZ)	Physical property of a physical object, state variable, process variable or system which can be quantified (measured)	Magnitude of edge influence (MEI), steepness, intensity, degree, contrast
Transitional gradient (TG)	Vector of physical quantities (e.g. concentration of matter or density of population) in space describing the direction and magnitude of change in physical quantities for every point in a vector field $TG = \frac{\Delta MTZ}{L}$, where L is the distance perpendicular to the zero line	Edge influence (EI) according to Harper et al. (2005), edge effect, interference, transition, causal ecotone, complex gradient, factor-gradient (see Erdős et al. (2011) for a distinction between environmental gradients and community gradients)
Significance in slope (SOS)	Significant difference ($p > 0.05$) of the slope of the transitional gradient compared to the related core matrix in the same matrix	Significance of edge influence (Chen et al., 1995)
Length of significant transitional gradient (LTG)	Linear spatial extent (distance) perpendicular to the zero line where SOS is given	Depth of edge influence (DEI), extent, distance, edge-effect penetration distance
Permeability ^a	Reciprocal rate of space-filling vegetation Vertical: sparse (20–10% to 1% canopy cover), open (70–60 to 20–10%), closed (>70–60%) Horizontal (stratification): open (only tree layer), semi-open (dominant herb layer, less shrub), semi-closed (dominant shrub layer), closed (fully developed stratification)	According to the Land Cover Classification System by FAO (Di Gregorio, 2005)

^a Further explanations of these definitions are given below.

2.2. Definitions – gradients in fragmented landscapes

2.2.1. Structural traits in fragmented landscapes

Ecosystems are usually understood as complex systems: they are nonlinear, emergent, self-organised and self-regulated, interrelated, open and agent-based; they also have attractors (Gosz, 1992; Müller and Kroll, 2011; Wu and Loucks, 1995). In order to understand them better, humans tend to structure things when investigating units of a system. In ecology, patches are often used as such a concept for structuring a system (see Wu and Loucks, 1995 for a review). The characteristic feature of patches is a delineation from their environment in which patches can be seen as physical systems. As such, a system boundary must be identified, which is a question of definition and scale. Delineation is usually considered to be worthwhile when within-patch heterogeneity is substantially less than that of between patches. The scale is always a challenge, as a patch can be a leaf, a group of plants, an ecosystem, a landscape or a continent (Wu and David,

2002; Wu and Loucks, 1995; Yarrow and Salthe, 2008). Scale is apparently also a problem in transition zones: whereas both Gosz (1993) and Peters et al. (2006) suggest plants, populations, patches, landscapes and biome levels with transition zones, Erdős et al. (2011) exclude elements such as hedgerows, fences and roads from being 'landscape elements'. Despite in-depth discussion, the tenor in the literature is a multiple scales approach (Kark and van Rensburg, 2006).

Another approach to the structuring of complex ecosystems is the hierarchy theory (Wu and Loucks, 1995). This concept assumes that higher levels involve larger entities and bigger units, which makes them slower. Thus they can be seen as static for subsystem investigations. In contrast, the high-frequency processes of subsystems can be averaged at higher levels, with the exception of highly non-linear systems (Wu and David, 2002; Wu and Loucks, 1995). Depending on the scale of the research question, variables at higher levels or lower levels can be more manageable for the purpose of analysis and interpretation.

In the hierarchy patch dynamics paradigm, both concepts are merged (Wu and Loucks, 1995). Landscapes can be seen as hierarchical mosaics of nested patches (ecosystems), while these ecosystems "correspond to land cover types" with "homogenous vegetation-soil complex" (Wu and David, 2002). This is in line with Yarrow and Salthe (2008), who defined land cover type as "surface-type". Examples of application are classification into biomes (e.g. temperate broadleaf forest), biographic regions (e.g. continental or boreal) or soil types (e.g. Podzol or Stagnosol). Cadenasso et al. (2003a) also distinguished patches "compositionally and structurally". A broadly applicable approach to define structural traits for vegetation is the Land Cover Classification System (Di Gregorio, 2005). Besides structural traits for identifying patches, an article by Wu and David (2002) and a study by Cadenasso et al. (2003a) also named functional units, which can be problematic (see Section 2.2).

2.2.2. From functional traits to functional gradients in fragmented landscapes

The word function has several implications. Jax (Jax, 2005) suggested differentiating between at least four kinds of functions: "1) Processes and the causal relations that give rise to them, 2) the role of organisms within an ecological system, 3) overall processes that sustain an ecological system (functioning), and 4) services a system provides for humans and other organisms." We suggest using the following terms to make a precise distinction in functions:

Point 1 by Jax (2005) is split into static variables (e.g. energy, population size) and process variables (e.g. heat, work). Static and process variables are thus the functional traits of a certain system. Process variables as such are mathematical functions, which would be another function. For quantitative analyses, it is important to differentiate between both, as well as other functions.

In point of fact, functional traits are gradients and “symbolize the spatial, functional, or temporal differences of structures or energetic and material units in ecological systems or subsystems” (Müller, 1998). In sum, functional gradients are based on functional traits (static and process variables) and are influenced by structural traits. A combination of hierarchy theory and a functional and structural nesting leads to a hierarchical “system of gradients” (Müller, 1998).

Functional gradients are measureable and quantifiable, and are therefore a better basis for the understanding of interactions in and the functioning of ecosystems, as well as for evaluations of ecosystem services.

2.2.3. Quantification of structural and functional nesting in landscapes – the transition zone

In most papers, the line between at least two adjacent types of land or land use (structural traits) with a certain difference has been referred to as the edge (Murcia, 1995). However, the term edge implies a sharp and defined structure, which in many cases is only an adequate description for structural traits (Kark and van Rensburg, 2006). Cadenasso et al. (2003b) used the term ‘ecological boundary’, but this tended to describe an ecosystem boundary. In their review, Yarrow and Marín (2007) found boundaries described as two- or three-dimensional with a bordering line (the real ‘edge’; abrupt change in land cover) and an edge (the patch area; influenced zone). Dialectically, none of the three terms – edge, ecocline and ecotone – are broadly applicable.

In addition, a number of terms in articles on transition zones were used synonymously or were applied without an explicit definition (Erdős et al., 2011). For this reason, we (and others: Hufkens et al., 2009) feel that there is a need to propose a set of terms and definitions related to fragmented landscapes so as to establish a well-founded basis for further research on these increasingly important transition zones (Table 2.2).

Table 2.2 Generic classification of transition zones.

Prefix	Second prefix and recommended application	General term	Sample suffixes
Terrestrial	Biotic (according to Peters et al., 2006; e.g. abundance and diversity of fauna and flora)	Transition zones	Vegetation zones according to Whittaker (1970)
	Abiotic (according to Peters et al., 2006; e.g. microclimate, matter dynamics, geology)		<ul style="list-style-type: none"> • of tropical rainforests • of temperate deciduous forest • of cool-temperate sphagnum bog • of savanna • of temperate grasslands
Aquatic	Biotic (abundance and diversity of fauna and flora)		Land use type according to Anderson et al. (1976)
	Abiotic (e.g. microclimate, matter dynamics, hydrology)		<ul style="list-style-type: none"> • of urban areas • of agricultural land

The following table combines ecological features with algebraic and geometric components to summarise existing definitions. This appears to be necessary because most frameworks focused on biotic factors, but neglected the role of microclimatic properties and matter cycling. Moreover, functional and structural traits were often investigated separately (Wu and Loucks, 1995). We believe that the revised definitions we propose in Table 2.1 are more suitable for quantitative studies and metric assessments.

These definitions and terms represent a basic toolbox for the quantitative description of transition zones in fragmented landscapes. The intention is to establish a relatively straightforward general system of concepts that quantitative ecologists can use; as a result, it will be broadly applicable as well as unambiguous (according to Erdős et al., 2011). The following section depicts a sample area (Figure 2.2), introduces a workflow chart (Figure 2.3), and explains some parts of the toolbox in greater detail to more clearly describe the terminology.

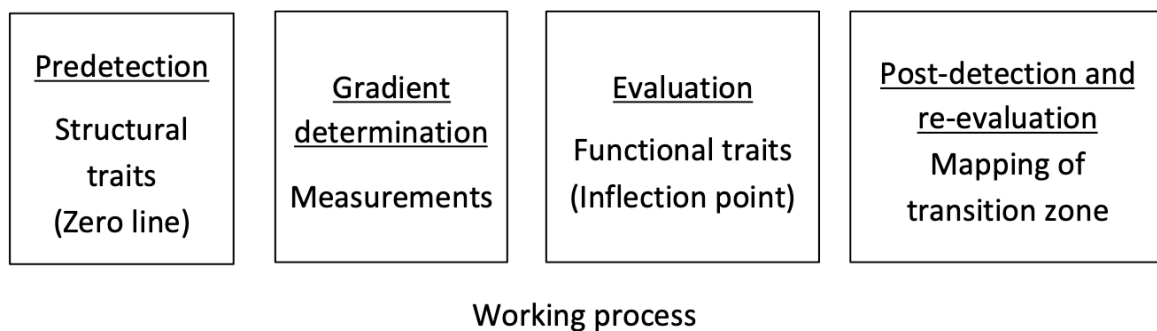


Figure 2.3 Workflow chart for mapping transition zones in fragmented landscapes (also works for other approaches such as populations, see Kolasa, 2014).

Transition zones include other concepts, such as 'ecotone', 'ecocline', 'interface', 'edge', 'system of gradients', 'ecological boundary' and 'border' (Cadenasso et al., 2003a; Müller, 1998; Yarrow and Marín, 2007). The biotic transition by Peters et al. (2006) can also be adopted, but without taking matrices as 'end states'. Furthermore, taking transition zones and their gradients as autonomous entities in landscapes emphasises their importance and makes them quantifiable (Müller and Kroll, 2011; Yarrow and Salthe, 2008). As such, they fit into the concept of hierarchy theory as well as the patch dynamics paradigm (Wu and David, 2002). The twofold approach – using structural and functional traits – may help tackle the problems due to the larger number of variables with a lower scale in modelling (Gosz, 1993).

The zero line is a result of a structural distinction of matrices, whereas the inflection point is the result of a functional analysis. In other words: the zero line exists for the detection of boundaries in fieldwork or on maps (visible discontinuity), while the inflection point is a result of measurements and mathematical analysis (Post et al., 2007; see Hufkens et al. (2009) for an overview of methods of detection). This is in line with the idea proposed by Kolasa (2014) for boundary detection recognising: a) "steepness of a gradient and a variable", b) "the amount of contrast between adjacent patterns", and c) "entities as 'owners' of boundaries". This differentiation is necessary to enable an initial, easy and practical solution to be found for structural matrix distinction, while leaving open the possibility to predict the extent and magnitude of transition zones.

Boundaries are "signal processors" (Yarrow and Salthe, 2008). Wiens et al. (1985) describe boundaries as membranes, Naiman and Décamps (1997) compare them with semi-permeable membranes of cells. Their permeability (or their resistance, reciprocally) depends on the characteristics of the patches (structural traits) and of the observed gradients (functional traits) (Gosz, 1992). Out of 52 studies considered to review the spatial extent (length) of gradients in transition zones, 30 used the terms 'open' (26), 'closed' (13) or both to describe structural characteristics. Thirty studies referred to canopy cover, 14 to land use, eight to age of vegetation, and three to history of management. In addition to underlining the need for a common definitional framework, it became the basis for the framework according to the appearance of authors' terms. We therefore decided to use the Land Cover Classification System (Di Gregorio, 2005) to define structural traits for vegetated areas. Horizontal permeability (e.g. horizontally open) is described by vertical stratification of herb, shrub and tree layers and their relative quantities. It affects physical processes that are vectored horizontally, such as wind. The higher the manifestation of stratification (e.g. a fully developed shrub layer and herb layer in addition to trees), the lower the horizontal permeability for a certain distance. For example, a forest with no shrub and herb layer has a higher depth of penetration of wind than a forest with full stratification. This is critical if the kinetic energy of the wind, which has to be processed, is the same, but needs to go a longer

distance into the forest to be transformed (Maurer et al., 2013). Vertical permeability (e.g. vertically open) also depends on stratification. The permeability for solar radiation, for example, depends on the development of the stratification: if the tree layer and the shrub layer are fully developed, the herb layer receives less radiation, which affects its biomass or ecological strategy (florescence), for instance. Moreover, the temperature of the soil is mainly driven by radiation and is therefore also influenced by vertical permeability, which influences soil microbial activity (see Sections 2.3.1.1 Solar radiation, 2.3.2 Carbon compounds and cycling and 2.3.5 Correlation of matter cycling and microclimate in transition zones).

To achieve interdisciplinary conformity, we further suggest the following expressions and definitions to enable a comparison of different transition zones: transition zones are four-dimensional with respect to time and their occurrence as three-dimensional physical bodies (Hufkens et al., 2009). They are clearly temporally variable (e.g. Chen et al., 1995; Saunders et al., 1999; Young and Mitchell, 1994). The general term transition zone can be specified by prefixes such as terrestrial, aquatic, and so on. Following Hufkens et al. (2009), Jax (2005), Yarrow and Marín (2007) and , this approach helps to a) include all terms used in the past, b) encourage further development of operational terms, c) satisfy policymakers' needs for one simple term, d) satisfy the need for precision in science by using prefixes and e) bring ecological approaches in line with modelling. Finally, suffixes specify the ecological or local conditions or the reference system itself.

2.3. Gradients of matter cycling and microclimate in forested transition zones

In addition to soil and hydrology, other key drivers influence processes and conditions of transition zones in forests. These include age, structure, fragment size, distance to next fragment, forest type, weather, climate and latitude. These transition zones are located in the forest, but have an adjacent matrix of different land use or cover. Most measurements were only conducted for gradients into forests – a fact which is reflected by the literature in this section.

The structure of transition zones depends on the age of the vegetation (Camargo and Kapos, 1995; Chabrierie et al., 2013; del Castillo, 2015; Matlack, 1993). Successive stages and the degree of maintenance lead to more open or closed transition zones. While aging, gradients weaken (Camargo and Kapos, 1995; D'Angelo et al., 2004; Didham and Lawton, 1999; Wicklein et al., 2012) but may increase when aging proceeds (Laurance, 2004). If they are not maintained, transition zones exhibit higher spatial variability (Saunders et al., 1999). As proposed by Chabrierie et al. (2013), the age of a transition zone can be indexed by comparing old and new maps. Didham and Lawton (1999) found that properties change if the character of the fragment remains the same, but the size of the fragment changes. Moreover, the interconnection of fragments plays a crucial role, as the impacted areas overlap (Porensky and Young, 2013).

2.3.1. Microclimatic factors

The predominant vegetation (forest type) of a given area has an influence on the spatial extent and magnitude of effects in transition zones. Values for the spatial extent of altered microclimate in transition zones have been found for boreal (Redding et al., 2003), temperate (Chen et al., 1995; Didham and Ewers, 2014; Dovčiak and Brown, 2014) and tropical forests (Hennenberg et al., 2008; Kunert et al., 2015; Patten and Smith-Patten, 2012). The microclimatic patterns established by Young and Mitchell (1994) for more closed transitions differed to those found by Chen et al. (1995) for more open transition zones. Didham and Lawton (1999) found the spatial extent of altered microclimate in transition zones to be two to five times higher at open transition zones compared to closed ones, suggesting the following rank order for the spatial extent of the influence of transition zones: closed continuous < closed fragmented < open continuous < open fragmented forests.

Microclimatic effects were highest on sunny and windy days (Wicklein et al., 2012), so there is a direct dependence on weather, but also on the time of the day (Chen et al., 1995; Davies-Colley et al., 2000; Meyer et al., 2001). Orientation perpendicular to the zero line is reported to be influential in most studies (Cadenasso et al., 1997; Dignan and Bren, 2003; Gehlhausen et al., 2000; Heithecker and Halpern, 2007). In contrast, Voicu and Comeau (2006) found air temperature to be independent of orientation. Furthermore, altitude is reported to have less influence on the magnitude of alteration of microclimate in transition zones (Lippok et al., 2014). This might be in contrast to Wicklein et al. (2012) because wind speed is altered in areas with hills and mountains, which function as obstacles that cause upwind and downward areas.

Obviously, the corresponding latitude of the site also has an impact in terms of climate (Matlack, 1993; Murcia, 1995; Williams-Linera, 1990; Young and Mitchell, 1994). In higher latitudes, seasons influence the magnitude of the effects in transition zones (Chen et al., 1995; Ewers and Banks-Leite, 2013; Kunert et al., 2015; Ritter et al., 2005). In this context, north-facing transition zones in the Southern Hemisphere are comparable to south-facing transition zones in the Northern Hemisphere, which is why Dignan and Bren (2003) deem the expression 'towards the equator' to be more coherent.

Solar radiation

Solar radiation is a key driver of altered microclimates in transition zones. Different wavelength ranges were used in the literature, depending on the research question. Nevertheless, radiation from the most influential spectrum for microclimate (250–3000 nm) decreased rapidly (Figure 2.4) within 10–60 m (Chen et al., 1995; Davies-Colley et al., 2000; Young and Mitchell, 1994) and nearly vanished within 100 m (Dignan and Bren, 2003). Denyer et al. (2006) highlighted the fact that different intensities of solar radiation had a shorter penetration distance in transition zones than

was the case with temperature. The intensity of solar radiation penetration was also influenced by vertical density of foliage (Mourelle et al., 2001; Parker et al., 2004). In light of this finding, Didham and Ewers (2014) therefore divided the space into bright, transition and dim zones. This enabled them – and Dignan and Bren (2003) – to detect a vertical gradient. The orientation and canopy height of transition zones also affected the penetrability of solar radiation (Dignan and Bren, 2003). The biggest effects were reported for equator-facing transition zones (Dignan and Bren, 2003).

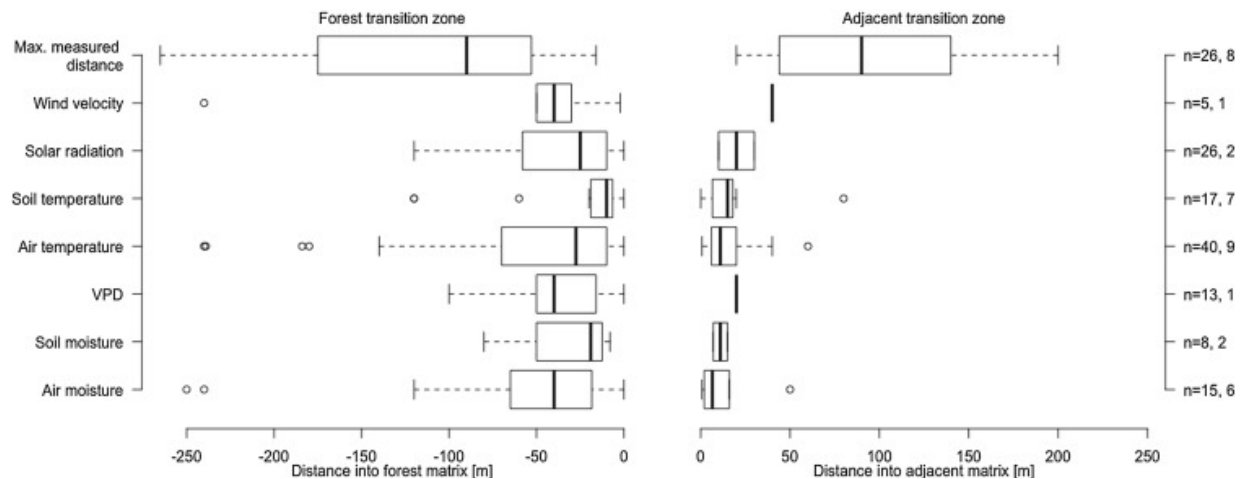


Figure 2.4 Quartiles of influence in transition zones from the zero line between adjacent matrix and forest (left), and vice versa (right). Out of 76 studies concerning microclimate, 33 stated a distance. n = number of values reported for forest transition zones and adjacent transition zones. VPD = vapour pressure deficit. The box denotes 25–75% of the values with the median (bold) in it. The dashed bars are areas without outliers (small circles).

Wind

Wind velocity was higher in transition zones (Cienciala et al., 2002). It decreased to about 20% of the wind in a non-forested matrix within approximately 60–240 m (Figure 2.4), and changed directions (turbulences) (Chen et al., 1995; Davies-Colley et al., 2000; Raynor, 1971).

Temperature

Most authors argued that temperature effects penetrated 50–100 m into the forest (Figure 2.4; Heithecker and Halpern, 2007; Meyer et al., 2001; Newmark, 2001), while heat flux was modelled to reach 100–200 m into the forest core matrix (Malcolm, 1998). Air and soil temperatures increased at night and decreased during the day from the zero line to the forest core matrix (Chen et al., 1995). There was also a significant vertical gradient of temperature (Didham and Ewers, 2014). Ritter et al. (2005) suggested that soil temperature is influenced by shading (Wright et al., 2010), higher evaporation and the isolating effects of a lower canopy height. This is in line with the findings by Giambelluca et al. (2003), which suggest that evapotranspiration is greatest when high positive heat flux is induced by high heat advection from clearings.

As air temperature was often lower in forests (Davies-Colley et al., 2000), Ewers and Banks-Leite (2013) argued that tropical forests reduced the surrounding temperature (if the maximum temperature outside the forest increased by 1 °C, temperature inside the forest increased by just 0.38 °C or 0.69 °C for the minimum temperature). Due to a higher heat capacity of forest and soils compared to air, transition zones and forest core matrices typically had a microclimatic lag time compared to non-forested matrices (Ewers and Banks-Leite, 2013).

Humidity and vapour pressure

Humidity increased from the zero line into forest core matrix (Figure 2.4; see also Wicklein et al., 2012; Williams-Linera et al., 1998). Dodonov et al. (2013) found the same at some sites, but also a decrease at other sites. Chen et al. (1995), Heithecker and Halpern (2007) and Mendonça et al. (2015) found no significant relationship. A vertical gradient in the vapour pressure deficit was found by Camargo and Kapos (1995) as well as by Didham and Ewers (2014), although its magnitude does not seem to be generalisable, as the figures were contradictory and were measured in different regions of the world. Didham and Ewers (2014) argued that vertical stratification of air layers was disrupted in transition zones.

Compared to forest core matrix, a higher wind velocity in transition zones increased conductivity for heat and gases and therefore, again, transpiration was higher (Cienciala et al., 2002).

Soil moisture

Tree water use was greater in forest transition zones than in forest core matrix (Cienciala et al., 2002; Herbst et al., 2007; Kapos, 1989; Taylor et al., 2001); this is because advection (Giambelluca et al., 2003) and convection (Klaassen et al., 1996) were higher. Gehlhausen et al. (2000) postulated that the spatial extent of soil water showing changes was greater than that of canopy openness in transition zones, which means that wind could also affect soil moisture. Farmilo et al. (2013) found that an increased canopy cover and decreased air temperature were responsible for a higher level of soil moisture in small forest fragments compared to continuous forest, in contrast to the results of Kapos (1989) and Gehlhausen et al. (2000). The reason might be a problem of scale, as the fragments analysed by Farmilo et al. were solitary, having no core matrix (Farmilo et al., 2013). Kapos (1989) determined lower soil matric potential (up to -1.5 MPa) within 20 m of a small patch of rainforest (Figure 2.4). Others found the spatial extent of changes in soil moisture in transition zones to be between 20 and 40 m to the zero line (Davies-Colley et al., 2000; Ewers and Banks-Leite, 2013). In winter, more open stands (Mellander et al., 2005) and lee sides of forests (Hiemstra et al., 2006) can facilitate a deeper layer of snow in transition zones. Zakrisson (1987) reported snow accumulation in non-forested transition zones of up to 40 m, with less snow in forested transition zones – up to 15 m. This might lead to changes in soil moisture and soil temperature as well as in carbon and nitrogen dynamics (Groffman et al., 2001). This phenomenon is attributed to

possible changes in water uptake and carbon assimilation of trees (Mellander et al., 2005). Otherwise, it is almost impossible to distinguish between measurable parameters leading to a given desiccating microclimate because microclimatic effects in transition zones tend to be cumulative (Godefroid et al., 2006; Laurance et al., 2011).

Spatial extent of altered microclimate in transition zones

In a review for forest microclimate ($n = 35$), Broadbent et al. (2008) determined a mean distance of alteration in transition zones into the forest core matrix of 191 m and a median of 60 m. Dodonov et al. (2013) recommended considering at least 60 m for transition zones in microclimate for savanna; this is similar to the average of 50 m reported by Hennenberg et al. (2008). Mosquera et al. (2014) recommended considering 10–20 m.

To our knowledge, three-dimensional (vertically and horizontally) studies have only been conducted by Camargo and Kapos (1995), Delgado et al. (2007), Didham and Ewers (2014), Dignan and Bren (2003) and Ewers and Banks-Leite (2013); findings from these studies showed that effects were higher in elevation, suggesting that near-ground measurements underestimate the influence of transition zones.

2.3.2. Carbon compounds and cycling

In tropical forest transition zones, mature stands of trees are replaced by pioneer trees (Laurance et al., 2006) within 300 m into the interior (Laurance et al., 2000). Dantas de Paula et al. (2016) found a lower tree cover within 50 to 100 m five years after fragmentation. In contrast, Williams-Linera (1990) reported a value of only 15 m. This replacement led to a decrease in biomass (Nascimento and Laurance, 2004), as well as a decrease in above-ground carbon storage (Laurance et al., 2007, 2011), although Ziter et al. (2014) argued that this is not valid for temperate forests. As bigger and older trees die faster after fragmentation (Laurance et al., 2000), they are displaced and replaced by younger trees that have a lower carbon storage capacity (Laurance et al., 2006). In contrast, Voicu and Comeau (2006) found that higher light transmittance was positively related to annual stem increment. Furthermore, Remy et al. (2016) found that stem density, wood volume and C stock of wood are lower towards the forest interior. It is hypothesised that less biomass production is directly connected with less leaf litter production (Farmilo et al., 2013), speeding up organic carbon decomposition (Nascimento and Laurance, 2004). Other than this, Remy et al. (2016) found no differences in C sequestration in transition zones.

Stanton et al. (2013) reported a higher soil total carbon stock in forest transition zones, while Johnson and Wedin (1997) detected a 17% lower soil organic matter content in a transition zone compared to in the core matrix. Remy et al. (2016) also reported a higher below-ground C stock for the transition zone (see Figure 2.5).

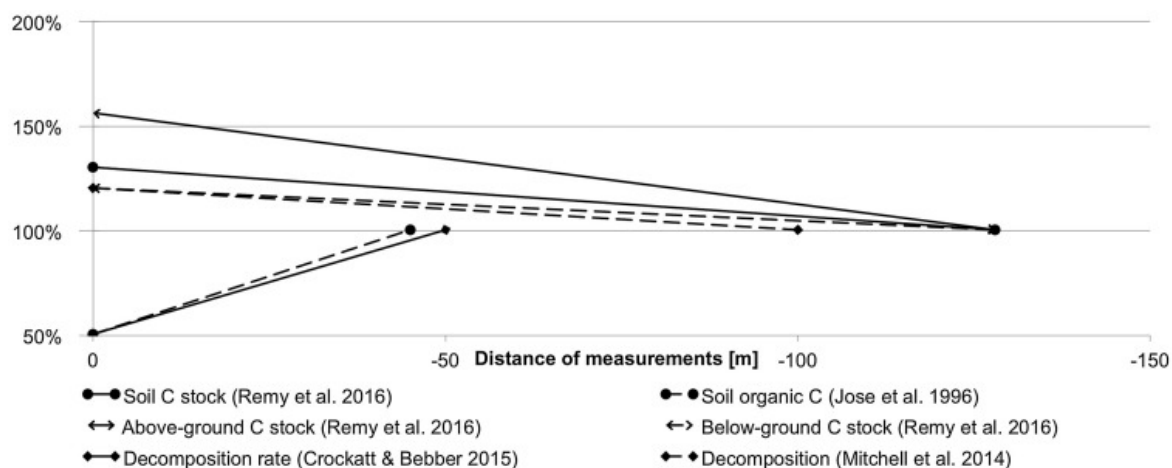


Figure 2.5 Relative changes in the amount of carbon compounds and rates of related processes with respect to distance to the zero line (0 m) into forest transition zones.

Decomposition is driven by microorganisms, climate (temperature and moisture) and litter quality (Coûteaux et al., 1995), and declines with fragment size independent of location in a fragment or the interactions among fragments (Moreno et al., 2014). Decomposition was found to be faster in the forest core matrix than in the transition zone (see Figure 2.5). Riutta et al. (2012) ascribed this to higher soil moisture, but see Section 3.3 Soil moisture on that topic. In contrast, neither Rubinstein and Vasconcelos (2005) nor Vasconcelos and Laurance (2005) found any differences. Nevertheless, others even reported distances of the influence of transition zones, as Figure 2.5 shows.

2.3.3. Nitrogen compounds and cycling

Forested transition zones have been described as ‘hotspots’ for nitrogen deposition and acidification (see Figure 2.6) because of local advection, turbulent wind flow and inflow (De Schrijver et al., 2007; Devlaeminck et al., 2005). Atmospheric deposition has been reported to be higher in transition zones (Wuyts et al., 2008) and can reach approximately 100 m into the forest (Ould-Dada et al., 2002). Weathers et al. (2001) measured 50% higher concentrations of ammonium and nitrate in throughfall compared to the core matrix. Ion deposition was three times higher (up to 15 times) in transition zones (Weathers et al., 1995). Stanton et al. (2013) reported higher total soil nitrogen contents for transition zones. Dissolved organic nitrogen leaching was also found to be higher, as well as nitrogen stocks (Wuyts et al., 2011). Remy et al. (2016) detected higher N stocks in the wood as well as in the mineral soil of transition zones. In contrast, Wicklein et al. (2012) argued that transition zones had no significant effect on nitrate and ammonium concentration in soil. Furthermore, Johnson and Wedin (1997) found that mineralised nitrogen in transition zones was one-third of that at the core matrix. Net nitrogen immobilisation and

microbial nitrogen were lower in forested transition zones (Toledo-Aceves and García-Oliva, 2007).

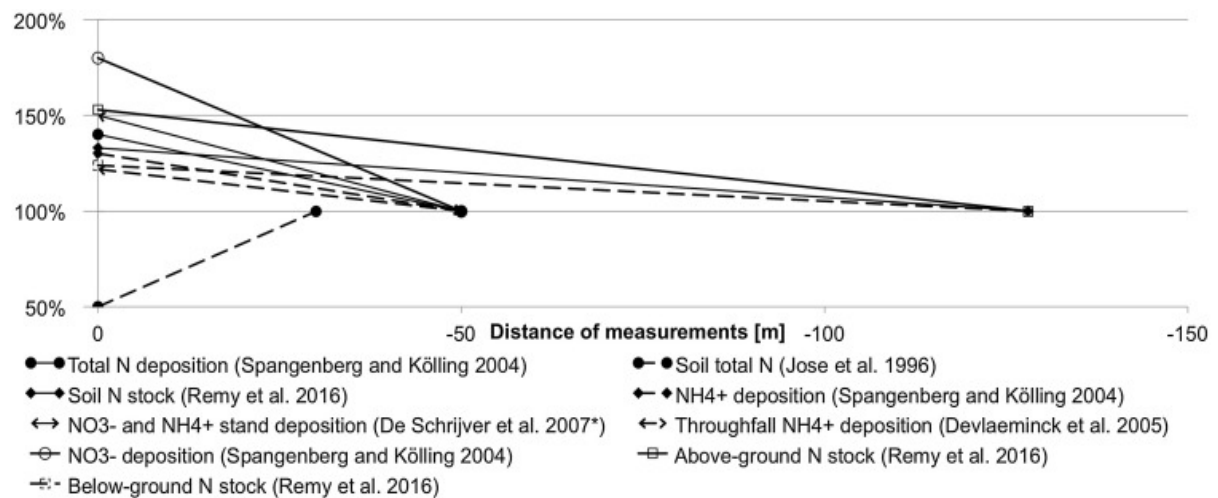


Figure 2.6 Relative changes in the amounts of nitrogen compounds in soils in forest transition zones with respect to distance to zero line (0 m). *Median of reviewed articles.

2.3.4. Gradients of matter cycling and microclimate in non-forested transition zones

Although Tuller (1973) maintained that the adjacent matrix to forest is also a zone of transition, only a few researchers have investigated effects for both the forest and the adjacent matrix (e.g. Baker et al., 2014; Davies-Colley et al., 2000; Dodonov et al., 2013). The evaluated literature includes studies on pasture land (Davies-Colley et al., 2000; Didham and Lawton, 1999; Williams-Linera et al., 1998), cropland (Hernandez-Santana et al., 2011; Williams-Linera, 1990), recently harvested forest or clear cuts (Baker et al., 2014; Dovčiak and Brown, 2014; Heithecker and Halpern, 2007; MacDougall and Kellman, 1992; Redding et al., 2003), savanna (Dodonov et al., 2013; Hennenberg et al., 2008) and plantations (Denyer et al., 2006; Farmilo et al., 2013). Studies have also been conducted on linear elements, such as roads, power lines and similar anthropogenic structures (Delgado et al., 2007; Kunert et al., 2015).

In agricultural transition zones with adjacent forest, the matrix is shaded by trees. The shading effect might cause lower rates of evapotranspiration (Laurance et al., 2011) and lower temperatures for both air and soil, which depend on incoming direct radiation (Gray et al., 2002). Voicu and Comeau (2006) found a spatial extent of shading of 0.3 times the height of aspen on adjacent spruce. The magnitude of alteration of microclimate in transition zones decreases as the age of the adjacent regenerating forest increases (Farmilo et al., 2013) and depends on the distance from the adjacent forested matrix, short-term and medium-term time scales, and climatic scales (Baker et al., 2014). Clearings – interpreted here as temporally non-forested to stress initial fragmentation effects – were usually hotter and drier compared to forest core matrix (Laurance et al., 2011), but this only seems to be true for tropical forests. Mixing of air led to lower air temperatures in clear

cuts (Chen et al., 1993). The centres of gaps have been reported to have higher soil moisture than the transition zones in adjacent forest, at least initially (Gray et al., 2002). In a temperate forest gap, soil water content reached the level of the adjacent beech forest within two years (Ritter et al., 2005).

For the adjacent matrix (pasture land), lower total carbon stocks in soil and litter have been reported (Stanton et al., 2013; Toledo-Aceves and García-Oliva, 2007), although Farmilo et al. (2013) determined no significant differences. Johnson and Wedin (1997) did not differentiate between quantitative differences, but an altered quality of carbon compounds. These differences are likely to occur because of a lower rooting depth of plants and a lower leaf area index (Laurance et al., 2011), resulting in less leaf litter mass (Farmilo et al., 2013). Stanton et al. (2013) found lower total nitrogen levels in the adjacent matrix, which is in line with Toledo-Aceves and García-Oliva (2007), who reported lower total nitrogen and soil microbial nitrogen levels in pasture land.

2.3.5. Correlation of matter cycling and microclimate in transition zones

Both Hastwell and Morris (2013) and Simpson et al. (2012) found a correlation between microclimate and matter cycling in transition zones; Jose et al. (1996) was unable to detect any regularities; and Didham (1998) found no correlation whatsoever. Crockatt and Bebbier (2015) reported that altered microclimate in transition zones of forests hampers decomposition. The findings of Riutta et al. (2012) outlined a correlation with soil moisture and temperature as key drivers influencing the metabolism of microorganisms, increased soil erosion and lower productivity (Trnka et al., 2013). As temperature is driven by radiation, Hastwell and Morris (2013) argued that canopy light transmission has a greater influence on litter decomposition than fragmentation-related features. The importance of microorganisms for matter cycling is well known and the correlation with temperature is evident (Moyano et al., 2008). The comparison of Figures 2.4–2.6 does not negate the idea that there might be a general correlation, but it does suggest that there is a site-dependent relationship of matter cycling to microclimate. Furthermore, the matter cycling system reacts much more slowly to microclimatic changes. Simple and short measurements that did not find a correlation may be inaccurate as the correspondence is time-shifted. Microclimate, especially radiation with soil moisture as the thermal storage system and temperature as its expression, was correlated with the activity of microorganisms. Hence, conditions for altered matter cycling in transition zones – as radiated areas – change temporally and spatially.

Despite radiation, wind shadows and vertical and turbulent wind dynamics foster the penetration of fertilisers in forest transition zones (Draaijers et al., 1988). Higher nitrogen availability enhanced wood and leaf litter decomposition (Bebber et al., 2011). On the other hand, depending on the orientation of the transition zone, wind can blow out the litter, which leaves less biomass for soil

carbon sequestration (Hastwell and Morris, 2013) in the forest stand, but creates an additional input in the adjacent land use system. Thus, it has an effect on soil water storage capacity, and therefore heat storage capacity, which again influences the activity of soil microbial biomass. Transpirational stress increases by one-third in transition zones caused by radiation and wind (Riutta et al., 2014), which may lead to less leaf litter production and therefore less source material for decomposition.

2.4. Conclusions

It can be gathered from the considerations above that the rapidly increasing total area of forested transition zones (Riitters et al., 2015) may be so relevant that it influences processes at the global scale. Tropical rainforests take up the largest amount of atmospheric carbon over the course of a year, followed by savannas (Beer et al., 2010). At the same time, these are the areas that are most threatened by deforestation and degradation, causing the formation of new transition zones. Fragmenting these highly vulnerable ecosystems – as a form of degradation – will increase the rate of carbon dioxide emissions, and therefore accelerate global warming (Ewers and Banks-Leite, 2013; Haddad et al., 2015). For boreal forests, Baltzer et al. (2014) reported a higher fragmentation caused by thawing, which possibly adds to the emissions. This is a correlating feedback to global warming and affects an even bigger storage of carbon: frozen soils in boreal forests thaw and increase respiration; this releases large amounts of greenhouse gases (Koven et al., 2011). Hence, further fragmentation of landscapes leads to an additional acceleration of global warming. Moreover, the accompanying feedback effects foster fragmentation. Together, climate change and fragmentation decrease actual net carbon sequestration, and thereby endanger one of the most important regulating ecosystem services (Riutta et al., 2012).

The relevance of transition zones is not only justified by their global importance and extent: the current imbalance of research on forested versus non-forested transition zones is reflected in this review. A much larger section addresses forested areas, revealing a noticeable knowledge gap with respect to non-forested transition zones. However, with respect to microclimate, these transition zones in ecosystems also influence each other: higher temperatures in forested transition zones compared to forest core matrices, for example, lower the soil's moisture content, but increase the rate of chemical processes. With respect to organic matter decomposition, these effects act antagonistically, and it is up to simulation models and field observations to determine whether decomposition is slowed down or accelerated at specific locations. The opposite may then occur in the adjacent non-forested area, where the cooling effect of forest on adjacent non-forested areas might reduce evapotranspiration and hence increase soil moisture. This example demonstrates the mutual dependencies of ecosystems' transition zones, and almost suggests addressing them as ecosystems in their own right (according to the definition by Jax, 2006).

Improved insights into the complexity of ecosystems' transition zones could emphasise the hot spot character attributed to them – not only in terms of biodiversity: since forest transition zones are often subject to higher deposition by winds and surface water, such as of nitrate (De Schrijver et al., 2007; Devlaeminck et al., 2005), they could serve as an “early warning system” for critical loads (Kark and van Rensburg, 2006).

Modelling ecosystems' transition zones and the effects of fragmentation in landscapes could provide more insights: for example, linking adjacent matrices or landscape elements via the soil water fluxes within and between them may reveal different mechanisms to explain observations, rather than simply comparing the soil water regimes of two ecosystems. The use of plant growth models for different adjacent matrices and their transition zones may change the accuracy of predictive models for large-scale evapotranspiration, which could then refine watershed models for fragmented landscapes (Wright et al., 2012). To facilitate this approach, state-of-the-art remote sensing should be used to image transition zones: for example, the resolution of satellite images of 30 m some years ago was improved to less than 5 m. This now enables transition zones to be detected that are most likely to be smaller than 30 m. The possibilities offered by state-of-the-art computing – for example, the ability to realise non-linear and high-dimensional modelling in a reasonable time – can be used to analyse and upscale information from these combined landscape models with their transition zones to a global level. The theoretical techniques for detecting transition zones already exist: the most common ones are wombling (identifying zones of rapid changes, Fitzpatrick et al., 2010) and moving split windows (see Hufkens et al., 2009 for an overview). Nevertheless, a common framework must be established to enable a comparison of results – this review offers such as framework.

In order to increase our knowledge of ecosystems' transition zones, we evaluated the literature concerning the significance of the values under review: it was not possible to validate the spatial extent of altered conditions of 100 m perpendicular to the zero line, which is suggested (see Section 1) as being universally applicable. It is most likely that transition zones have spatio-temporal differences and must therefore be adjusted for the research question and the region under investigation. However, they are important and should be considered. Our review of the literature suggests that we can expect altered conditions in soils of transition zones to be 10–20 m with a maximum of 50 m, and 25–50 m with a maximum of 125 m for above-ground space. Nevertheless, further insight is necessary in order to enable us to understand the global influence of fragmented landscapes, especially for non-forested matrices and in terms of ecosystem services to humans (Mitchell et al., 2015). Furthermore, the difference – if any – between natural and anthropogenic transition zones deserves a thorough investigation in this context (Kark and van Rensburg, 2006).

3. Microclimate and matter dynamics in transition zones of forest to arable land⁴

Abstract

Human-driven fragmentation of landscapes leads to the formation of transition zones between ecosystems that are characterised by fluxes of matter, energy and information. These transition zones may offer rather inhospitable habitats that could jeopardise biodiversity. On the other hand, transition zones are also reported to be hotspots for biodiversity and even evolutionary processes. The general mechanisms and influence of processes in transition zones are poorly understood. Although heterogeneity and diversity of land use of fragments and the transition zones between them play an important role, most studies only refer to forested transition zones. Often, only an extrapolation of measurements in the different fragments themselves is reported to determine gradients in transition zones.

In this article, we analyse environmental gradients and their effects on biota and matter dynamics along transects between managed continental temperate forests and agricultural land for one year. Accordingly, we found S-shaped microclimatic gradients in transition zones of 50 to 80 m between arable lands and forests. Aboveground biomass was lower within 65 m of the transition zone, 30 m in the arable land and 35 m in the forest. Soil carbon and nitrogen contents were elevated close to the transition zone's zero line.

This paper contributes to a quantitative understanding of agricultural landscapes beyond individual ecotopes, and towards connected ecosystem mosaics that may be beneficial for the provision of ecosystem services.

Keywords: Edge effects, Environmental gradients, Fragmentation, Ecosystem Services, Carbon, Nitrogen

3.1. Introduction

In ecology, fragmentation is defined as the occurrence of discontinuities in prevalent or native land cover and habitat properties (Strayer et al., 2003). Although it is a natural process, fragmentation as we observe it today is mainly caused by humans (Haddad et al., 2015). As fragmentation occurs, it substitutes diverse and biomass-rich ecosystems with intensively used, man-made ecosystems, e.g. agricultural land (Tuff et al., 2016). Between these ecosystems, i.e. at

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their edges, transition zones occur through fluxes of matter, energy and information (for definitions, see Schmidt et al., 2017).

The processes and effects that occur have been categorised by Murcia (1995) into abiotic, direct biological and indirect biological effects of transition zones. Abiotic conditions – such as temperature – affect biological processes (Tuff et al., 2016) and thus habitat functions (Baker et al., 2016). In the literature, there is evidence that microclimatic gradients alter processes in transition zones, e.g. litter decomposition (Crockatt and Bebbler, 2015; Remy et al., 2017b; Schmidt et al., 2017). Altered soil and air moisture and temperature in transition zones (Baker et al., 2016) influence the metabolism of microorganisms, and with that matter dynamics (Riutta et al., 2012). Wind blowing into transition zones of forests carries nutrients that trees and bushes comb out of the air (Draaijers et al., 1988). This leads to higher nitrogen availability in the transition zone, which enhances wood and leaf litter decomposition (Bebber et al., 2011). Higher nitrogen deposition might be beneficial for above- and belowground carbon stocks and sequestration (Remy et al., 2016) in the transition zone, but on the other hand trees are reported to have less wood volume (Veselkin et al., 2017).

Fragmentation-related habitat loss is likely to be the most important threat to biodiversity and one reason for the continued extinction of species (Arroyo-Rodríguez et al., 2017; Ibanez et al., 2017). Fragmentation is most often caused by an expansion of arable land and increases the ratio of edges to forest interior. Magura et al. (2017) have argued that these managed edges with an intensive human impact offer a rather inhospitable habitat in addition to habitat loss caused by fragmentation alone. However, the hospitability of transition zones greatly depends on the species that are investigated. Kark and van Rensburg (2006) as well as Lidicker (1999) have argued that transition zones can be hotspots for biodiversity and even evolutionary processes as novel niches (see Ries et al. 2004 for a review). Edges caused by roads or with adjacent managed areas can favour exotic species compared to native species (Gehlhausen et al., 2000; Watkins et al., 2003). In a review, Fahrig (2017) argued that fragmentation has a positive effect on biodiversity. On the other hand, Fletcher et al. (2018) argued that this perspective is too onesided and that in fact negative effects on biodiversity occur.

Nonetheless, the general mechanisms and influence of processes in transition zones are poorly understood. As Ries et al. (2017) have noted, scientists have often merely described the edge effect of a single matrix (a “spatial domain where processes, properties or magnitudes” of physical, chemical or biological “variables are sufficiently distinct from those of its neighbors to warrant their segregation,” see Woo, 2004) and then they have extrapolated between matrices. Moreover, many studies focus on the fragment, but Ferrante et al. (2017) argue that the character (land use) of the matrices plays a more important role. In addition, most studies only refer to forested

transition, considering it to be 100 m perpendicular to the zero line (Riitters et al., 2002; Riutta et al., 2014; Spangenberg and Kölling, 2004). Among those studies, few measurements exist for temperate forests (Wright et al., 2010). For arable land, Cleugh (1998), Kort (1988) and Nuberg (1998) reviewed literature on the windbreak effect of forested areas on microclimate, soil conditions and crop productivity. Cleugh and Hughes (2002) also provide models based on wind tunnel experiments and analyses of field experiments. Another article by Bird (1998) highlights similar positive effects of windbreaks and shelter on pasture.

We measured microclimate along different transects between managed continental temperate forests and agricultural land for one year. In addition, we measured soil nitrogen and carbon content as well as litterfall. In this paper, we analyse environmental gradients and their effects on biota and matter dynamics based on the following hypotheses:

- The width of the transition zone from arable land to forest depends on the measured variable.
- The abiotic environmental gradients are non-linear across ecosystem boundaries.
- Biotic effects are the consequences of abiotic environmental gradients in the transition zone.

The terminology in this article follows our concept of transition zones in quantitative ecology (Schmidt et al., 2017).

3.2. Methods

3.2.1. Experimental design

The measurements for this study were conducted in northeast Germany in the Federal State of Brandenburg in 2016 and 2017. For a detailed description of methods and data, see Schmidt et al. (2018). For hourly microclimatic measurements (air and soil temperature, air and soil moisture, wind speed and direction, air pressure, precipitation and solar radiation with a repetition between $n=26657$ and $n=32014$), an east-facing (the arable land is east of the forest) and a west-facing site were equipped with one transect of five weather stations (Figure 3.1) each – one weather station at the zero line, two within the arable land (15 m and 30 m) and two within the forest (-35 m and -70 m). For the sake of brevity, positive values are used for distance from the zero line for the arable land, and negative values for the forest. The distances were chosen according to the results of our literature review (Schmidt et al., 2017). At greater distances no significant effects were expected.

component analysis. The principal component analysis of time series is meant to decompose the total variance of multidimensional data sets. It yields a set of independent principal components that explain most of the variance of the time series (Hohenbrink and Lischeid, 2015). In terms of microclimatic time series this analysis is done, as the variance can be high and might result in misleading interpretations. In mathematical terms, the principal component analysis performs an eigenvalue decomposition of the covariance matrix of the respective time series. Usually the first principal component is very close to the time series of spatial mean values from all considered sites, and depicts the largest fraction of variance of the total data set (Hohenbrink et al., 2016; Lischeid et al., 2017). Each of the remaining principal components then describes deviations from that mean behaviour, which can be ascribed to a specific effect (Hohenbrink et al., 2016). Identification of that specific effect, however, requires additional background data and a sound understanding of the relevant system. Our analysis aimed to identify the principal component that would reflect the effect of position along the transect rather than, e.g., the effect of local soil heterogeneities. We identified the respective component by checking the time series of the relevant principal components for monotonic decrease or increase along the transect.

In cases where such a relationship existed, correlation of the single observed time series x with the time series of the relevant principal component PCy was used as a quantitative measure of the strength of the effect. The correlation coefficients $r_{x,PCy}$ were then normalised in such a way that $+1$ denotes typical time series of the inner forest position, -1 typical time series of positions in the arable land, and any value $-1 < x < 1$ describing the degree of similarity to either the typical forest or typical arable land time series of the relevant variable.

To do this, the correlation coefficients were transformed in a way analogous to the damping coefficient defined by Hohenbrink and Lischeid (2015), below,

$$D_{x,PCy} = \arctan \frac{r_{x,PCy}}{r_{x,PC1}}$$

and were then normalised

$$D'_{x,PCy} = s \times \frac{D_{x,PCy} - \min(D_{PCy})}{\max(D_{PCy}) - \min(D_{PCy})}$$

where

$$s = \begin{cases} 1, & \sum_{forest} D_{x,PCY} > \sum_{arable} D_{x,PCY} \\ -1, & \sum_{forest} D_{x,PCY} < \sum_{arable} D_{x,PCY} \end{cases}$$

and $\sum_{forest} D_{x,PCY}$ and $\sum_{arable} D_{x,PCY}$ is the sum of the coefficients for the two positions within the forest or arable land, respectively.

We also state the cumulative fraction of variance for the first (σ_1^2) and second (σ_2^2) principal component (Appendix A⁵).

We carried out a Bonferroni-adjusted post-hoc analysis to compare the data on trees (Figure 3.5), litterfall (Figure 3.6), soil (Table 3.1, Appendix B⁵) and above-ground biomass (Figure 3.7) with respect to their position in the transect. To verify whether samples originated from the same distribution, we performed Kruskal-Wallis one-way analyses of variance (see Appendix C⁵).

The R programming language (R Core Team, 2017) was used to perform all statistical analyses. The data is available in the accompanying method paper (Schmidt et al., 2018).

3.3. Results

3.3.1. Microclimate

At the west-facing site, soil moisture ($\sigma_1^2=.71$, $\sigma_2^2=.2$) and soil temperature ($\sigma_1^2=.97$, $\sigma_2^2=.03$) as well as the average ($\sigma_1^2=.85$, $\sigma_2^2=.09$) and maximum wind speed ($\sigma_1^2=.86$, $\sigma_2^2=.07$) and wind direction ($\sigma_1^2=.63$, $\sigma_2^2=.19$) exhibited a close to monotonic transition from the forest to the arable land (Figure 3.2, A). These environmental gradients were asymmetric and S-shaped. The transition zone according to the first principal component of these variables was approximately 50 m wide (from 15 m in the arable land to -35 m in the forest).

The measured variables of air pressure, air temperature, precipitation, relative humidity and solar radiation did not follow distinct patterns of a transition zone from arable land to forest at the west-facing site (Figure 3.2, B; Appendix A). In the forested transition zone, the relative similarities were rather stable, except for solar radiation.

At the east-facing site, the similarities of air temperature ($\sigma_1^2=1$), average wind speed ($\sigma_1^2=.65$, $\sigma_2^2=.19$), relative humidity ($\sigma_1^2=.98$, $\sigma_2^2=.01$), soil temperature ($\sigma_1^2=.99$, $\sigma_2^2=.01$) and wind direction ($\sigma_1^2=.63$, $\sigma_2^2=.2$) reflected the patterns of a transition zone from forest to arable land (Figure 3.2, C). These environmental gradients were asymmetric and S-shaped. The transition zone according to

⁵ Appendices are available in the online version at <https://doi.org/10.1016/j.agrformet.2019.01.001>

the first principal component of these variables was approximately 65 m wide (from at least 30 m in the arable land to -35 m in the forest). For the average wind speed, the transition zone was 85 m.

Wind direction, air and soil temperature tended to be more similar to forest patterns; average wind speed was more similar to arable land. Air pressure, maximum wind speed, precipitation, soil moisture and solar radiation did not exhibit a clear pattern along the transect (Figure 3.2, D).

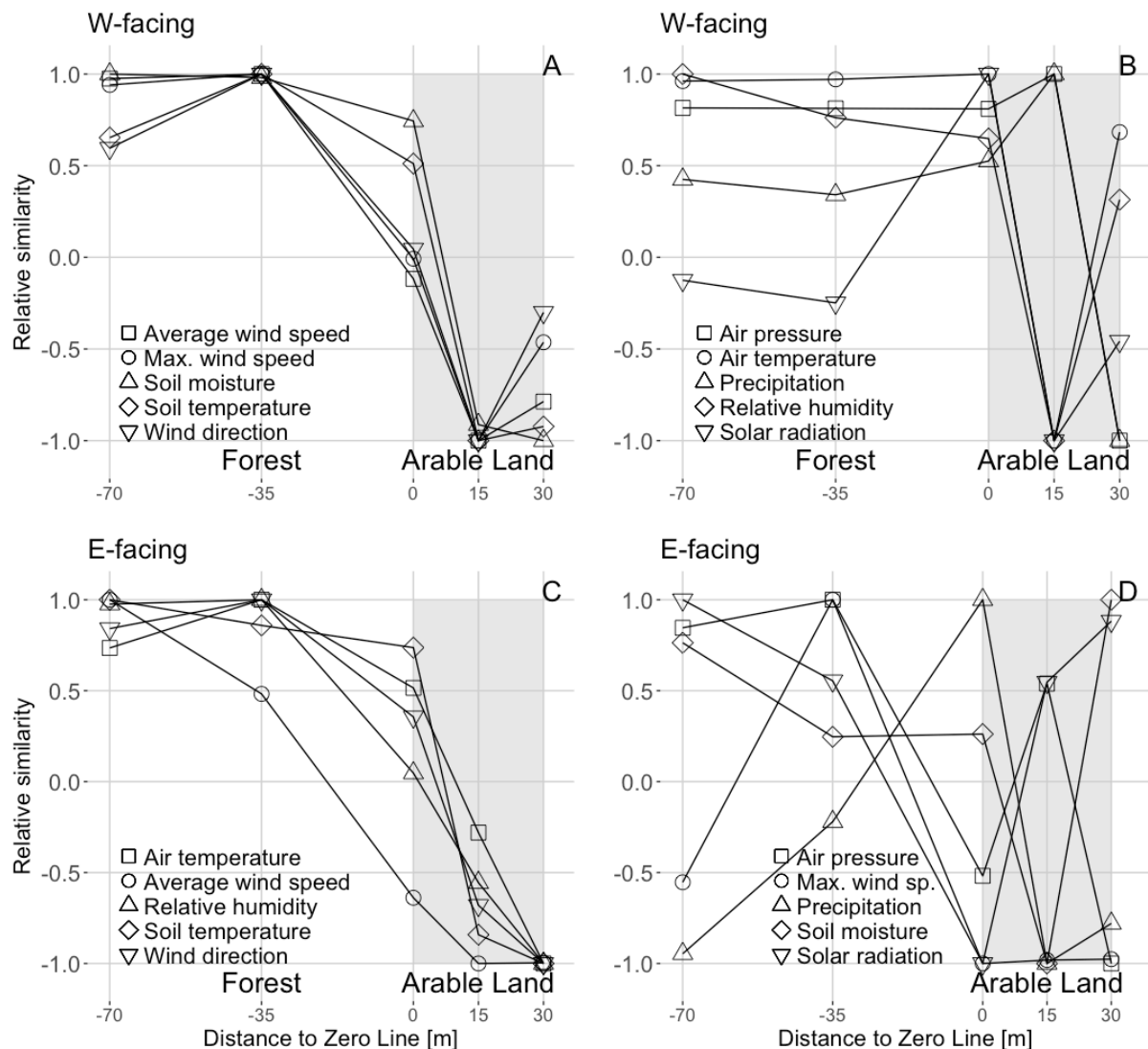


Figure 3.2 Shape of the transition zone between forest and arable lands for time series of various variables. Left: Variables that exhibit an approximately monotonic transition. Right: Variables that do not exhibit a clear transition. The points were connected using locally weighted scatterplot smoothing (loess).

The main wind direction for this region is southwest (SW; Figure 3.2). At the west-facing site at 0 and 30 m, the main wind direction tends towards the west, while at 15, -35 and -70 m the direction

is south. At the east-facing site, the main wind direction at 15 m is more westerly than the main wind direction of the region. At -35 m, it is the same as for the region as a whole. At 15 and 0 m, the wind direction is more to the south, and is to the south at -70 m.

Comparing results from the two transects, only average wind speed and direction as well as soil temperature exhibited roughly monotonic patterns along both transects, while solar radiation and precipitation as well as air pressure did so in only one out of the two transects.

In terms of absolute values, soil temperature was 2–5 °C higher on average in the arable land of the west-facing site compared to the forest interior in June and July 2016 as well as from March to July 2017 (see Schmidt et al. 2018 for data). In winter, the forest soil tended to be warmer. Except for January, February and July 2017, soil moisture was lower on average in the forest. Maximum (approx. 1.5 to 3 m/s) and average wind speeds (approx. 0.2 to 1 m/s) were higher in the arable land compared to the zero line as well as to the forest interior.

At the east-facing site, average soil temperature was approx. 2 °C to 4 °C higher on average in the arable land compared to the zero line and the forest interior, except in autumn and winter (October to February). The average air temperature tended to be slightly higher in the arable land, except for the period June to September 2016, when arable lands were considerably warmer than the forest interior, by 0.5 °C to 2 °C. The average relative humidity was lower in the arable land, while the average wind speed was higher (up to 1.5 m/s) in all months of measurement.

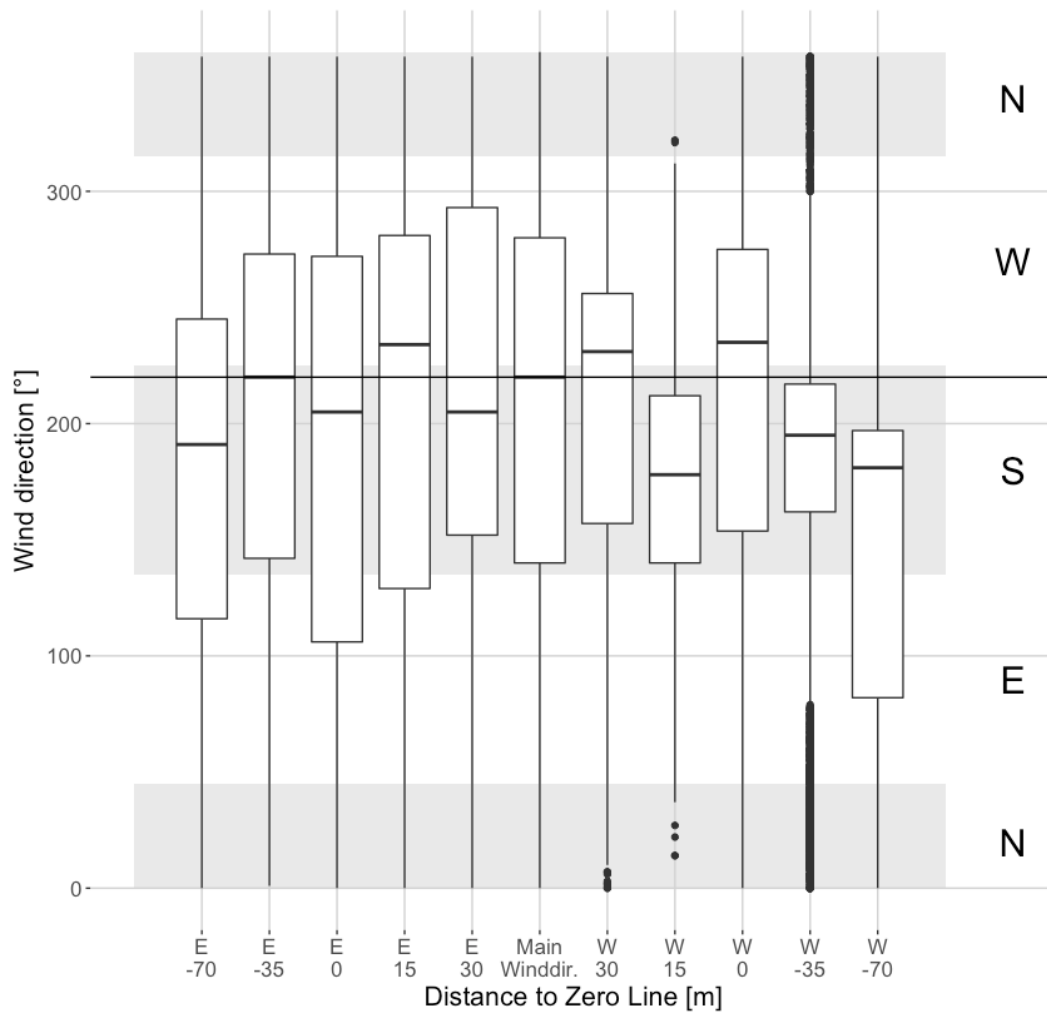


Figure 3.3 Quartiles of the wind directions of a west-facing site (W, right boxplots), an east-facing site (E, left boxplots) and the main wind direction for Müncheberg (coordinates: 52°30'57.8"N 14°07'20.9"E; the official weather station between both sites, with its median depicted as a horizontal line). The boxes denote 25–75% of the values with the median (bold) in it. The bars are areas without outliers (small dots). The greyish areas indicate the areas of the winds' cardinal direction (right).

3.3.2. Tree height and diameter

The height of the trees per plot ($n=30$) is significantly lower at the zero line (0–20 m) at both sites (Figure 3.4) with an average height of 18.98 m (east-facing, 50–70: $p=1.1 \cdot 10^{-6}$, 130–150: $p=8.7 \cdot 10^{-10}$) and 20.52 m (west-facing, 50–70: $p=1.1 \cdot 10^{-6}$, 130–150: $p=2.4 \cdot 10^{-5}$) compared to the interior plots. This figure does not differ significantly between the plots from 50 to 70 m and 130 to 150 m (west-facing: 21.95 m and 22.82; east-facing: 25.4 and 24.73 m). The diameter at breast height (not shown) was not significantly different except for the east-facing site in the 0 to 20 m plot (zero line; $p=0.039$) with 24.94 cm compared to 27.8 cm (50 to 70 m) and 25.78 cm (130 to 150 m).

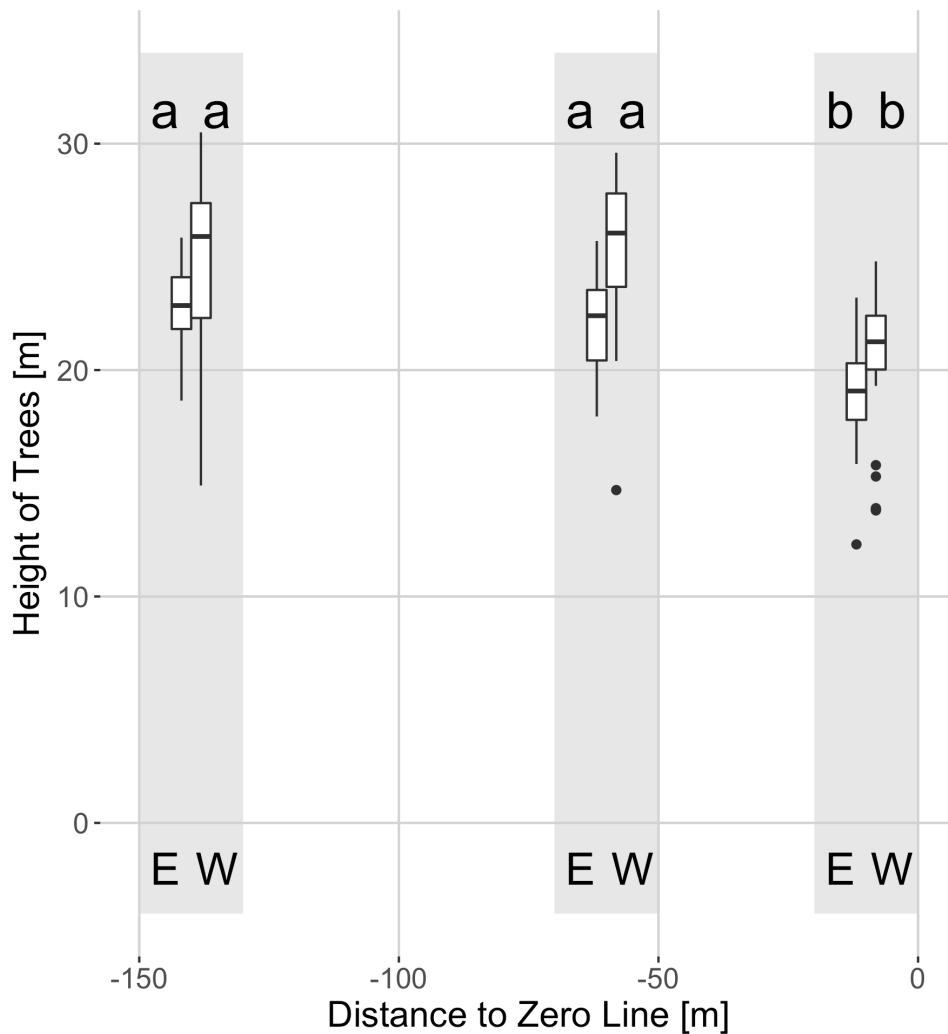


Figure 3.4 Boxplots of the heights of trees of the east-facing site (E, left boxplots) and of the west-facing site (W, right boxplots) in three plots (light grey; 0-20 m, 50-70 m and 130-150 m). $n=30$ trees were measured per plot. The boxes denote 25–75% of the values with the median (bold) in it. The bars (whiskers) either denote the range as long as it is less than 1.5 times the interquartile range or 1.5 times interquartile range, while the small dots are values outside that range. The small letters indicate differences in the means between plots per site according to Bonferroni-adjusted post-hoc analysis.

3.3.3. Litterfall

At the east-facing site, the mean dry mass of litterfall of pine (*Pinus sylvestris* L.) was not significantly different with respect to distance to the zero line (Figure 3.6). The mean dry mass of the litter of larch (*Larix decidua*) at the west-facing site was significantly lower ($p=8.2 \cdot 10^{-5}$) in the plot at the zero line (6.1 g) compared to 35 m (8.9 g) and 70 m (12.2 g) towards the forest core matrix. It is not pertinent to compare both sites because of their different tree species and tree ages.

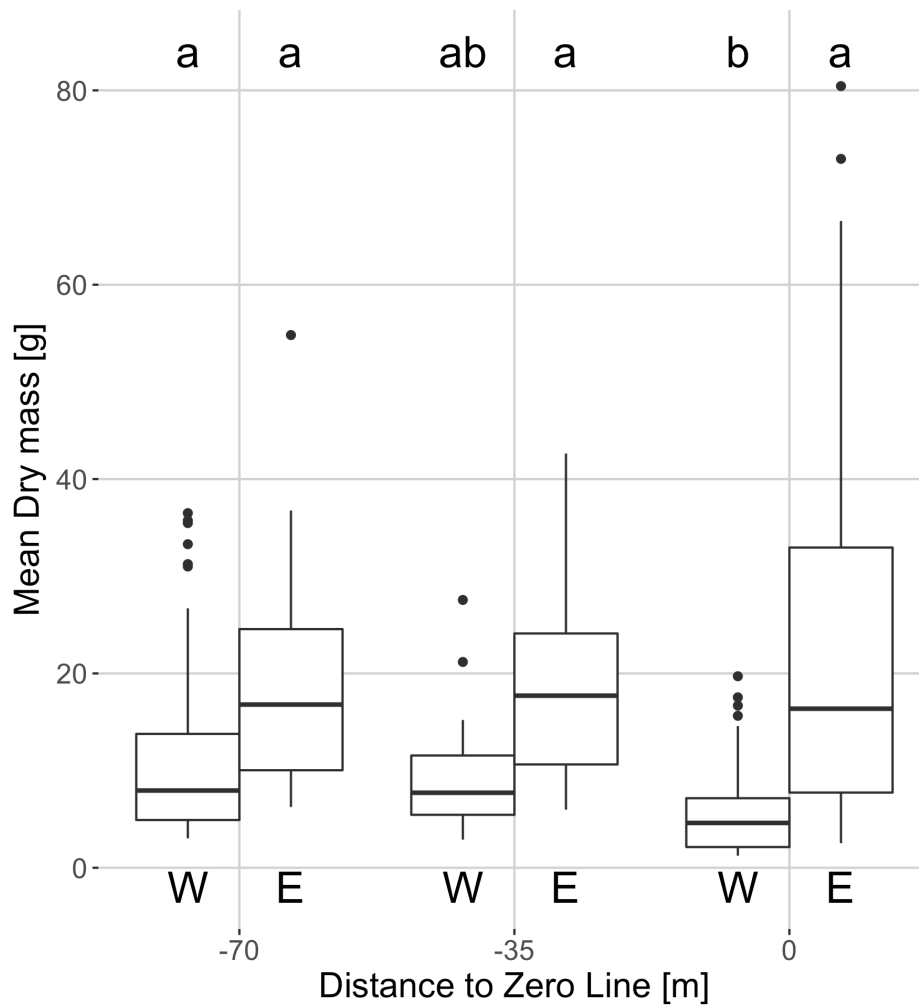


Figure 3.5 Boxplots of litterfall of a west-facing site (W; left boxplots; *Larix decidua*) and an east-facing site (E, right boxplots; *Pinus sylvestris* L.) at three sample points (0 m, 35 m and 70 m from the transect). $n=10$ litterfall traps were used at each sample point. Sampling was conducted on five days between September 2016 and May 2017. The boxes denote 25–75% of the values with the median (bold) in it. The bars (whiskers) either denote the range (as long as it is less than 1.5 times the interquartile range) or 1.5 times the interquartile range, while the small dots are values outside that range. The small letters indicate differences in the means between plots per site according to Bonferroni-adjusted post-hoc analysis.

3.3.4. Aboveground biomass in the arable land

For barley, the mean dry biomass was significantly higher at 7.5, 15 and 30 m ($p=3.5 \cdot 10^{-7}$, $p=7.9 \cdot 10^{-9}$, $p=1.2 \cdot 10^{-7}$) compared to the zero line. At 7.5 and 30 m, mean dry biomass of barley was not significantly different, while at the 15 m mean, the dry biomass was significantly higher (7.5: $p=.0031$, 30: $p=.0269$).

Pea had significantly higher mean dry biomass at 7.5 and 30 m ($p=.0052$, $p=.0092$) compared to the plot at the zero line. At 15 m, the mean dry biomass of pea was significantly lower than at 7.5 m and 30 m ($p=.0233$, $p=.0422$).

The mean dry biomass of oilseed rape was significantly higher at 7.5, 15 and 30 m ($p=.0074$, $p=.0001$, $p=.0005$) compared to the zero line. The mean dry biomass at all other distances was not significantly different.

Wheat had the statistically highest mean dry biomasses at 15 m, but not different at 30 m. However, the mean dry biomass was lowest at the zero line ($p=.001$, $p=5.1\cdot 10^{-8}$, $p=5.9\cdot 10^{-7}$). At 7.5 m, it was also significantly lower than the figures observed at 15 and 30 m ($p=1.0\cdot 10^{-5}$, $p=4.4\cdot 10^{-4}$).

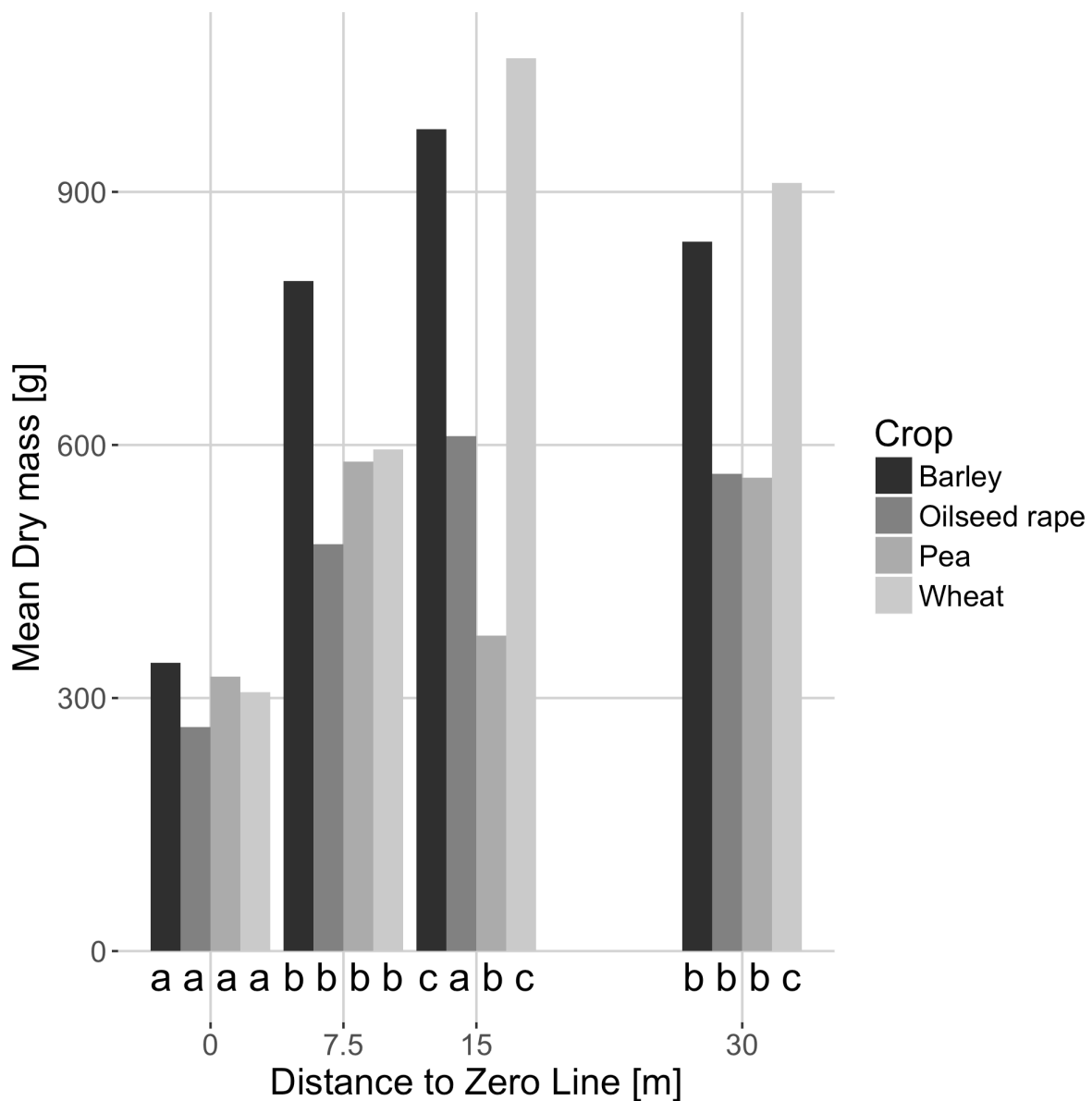


Figure 3.6 Mean dry matter aboveground biomass for two growing seasons in 2016 (pea, *Pisum sativum* L., and oilseed rape, *Brassica napus* L.) and 2017 (barley, *Hordeum vulgare* L., and wheat, *Triticum aestivum* L.). Each bar represents the mean dry biomass of one-square-metre plots ($n=4$) of aboveground biomass at a different distance to a forest edge at an east-facing site (barley and pea) and a west-facing site (wheat and oilseed rape). The letters represent the results of Bonferroni-adjusted post-hoc analysis comparing the distances per crop.

3.3.5. Soil carbon and nitrogen content

The highest mean values for total soil carbon content (C_t) were found at the zero line, with 1.56% at the east-facing site and 1.67% at the west-facing site at a 20 cm depth (Appendix B). These values are significantly higher than all other distances except 70 m in the forest (Table 3.1). The same holds true for the samples from the 40 cm depth, except for 35 m from the transect in the forest. The lowest values for C_t were found in the arable land, with less than 0.2%. Additionally, C_t was significantly different between 15 m in the arable land and 35 m in the forest at 40 cm depth as well as between 60 m in the arable land and 70 m in the forest at 20 cm depth. In terms of N_t , the highest values were also at the zero line, with 0.13% at both sites. Here, the zero line differs significantly from all other distances (Table 3.1). The ratio between total soil carbon and nitrogen content (C:N) was – with values between 4.17 and 6.12 – the lowest at a depth of 40 cm and in the arable land, except for 105 m in the forest on the west-facing site, where it was 5.13 (Appendix B). The widest C:N relationship was found at the 20 cm depth in the forest at both sites, with values between 13.35 and 16.07.

Table 3.1 Significant ($p < 0.05$, Bonferroni-adjusted post-hoc analysis) differences in total soil carbon (indicated by C; above the diagonal) and total soil nitrogen (indicated by N; below the diagonal) merged contents for 20 ± 3 cm and 40 ± 3 cm depth for different positions along the transects. If a capital letter is given with a number, there is only a difference at that depth; otherwise at both depths. ↓ indicates lower and ↑ higher levels at the distance given in columns compared to those in rows.

		Soil carbon content								
		Forest			Zero line	Arable land				
		-105 m	-70 m	-35 m	0 m	15 m	30 m	60 m		
Soil nitrogen content	Forest									
		-105 m			↑C					
		-70 m			↑C at 40				↓C at 20	
		-35 m			↑C at 20	↓C at 40				
	Zero line	0 m	↓N	↓N	↓N		↑C	↑C	↑C	
		Arable land	15 m			↓N				
			30 m			↓N				
	60 m				↓N					

3.4. Discussion

3.4.1. Properties of environmental gradients in transition zones

We hypothesised that the width of the transition zone from arable land to forest depends on the measured variable. We found that it is smaller for some microclimatic gradients according to the shape of the correlation coefficients of the first principal components (approx. 50 or 85 m) compared to other authors (e.g. Haddad et al., 2015). This is in line with other authors (e.g. Hennenberg et al., 2008). In most cases, the forested transition zone was approx. 35 m, which is only one-third of the extent other authors have assumed (Riitters et al., 2002; Riutta et al., 2014; Spangenberg and Kölling, 2004). In the arable land, the spatial extent was approximately 15 m at the west-facing, and up to 30 m at the east-facing site. The widths we report here coincide with transition zones of 25 to 50 m for the aboveground space with a maximum of 125 m we reviewed earlier (Schmidt et al., 2017). Differences in the spatial extent compared to other authors might occur due to the physical structure of edges. Moreover, our study comprises measurements for more than one year and covers all seasons. Seasonal differences might be not covered in other studies due to shorter measurement periods. The cardinal direction of measurements in transition zones plays an important role (Matlack, 1993), e.g. for solar radiation. Therefore, results may vary between transition zones for i.e. north- and south- as well as east- and west-facing edges. In our study, we wanted to avoid too strong effects of cardinal directions north and south and use opposing transition zones instead. This might be a reason for differences in microclimatic gradients to other studies.

The width of transition zones we report in this article is based on the assumption that the maximum extent of the transition zone in general is not wider than in our measurement design including all other spatial conditions. The first principal component depicts the mean temporal pattern averaged over all positions along the transect. It indicates whether a measurement point is within the assumed maximum transition zone. Although this approach allows separating the spatial effect from other effects, it does not account for the width of microclimatic gradients at the respective positions in the transect and beyond per se. However, the similarities in Figure 3.2 reflect the strength of the spatial effect and a correlation between observed time series and the relevant principal component. Therefore, the monotony of the similarities and its S-shape are the explanatory approach and can be assumed as an approximation to the microclimatic gradients. The strength in our study is therefore not a spatial repetition, but rather a high temporal resolution and the seasonality. The variance is disentangled by the principal component analysis and assigned to the spatial position in the transect. The S-shape and its width figures the similarity of the relationship between measured values and the main behaviour and assigns it to values that are typical for the forest or arable land based on our data.

Some of the evaluated microclimatic gradients are S-shaped. On the other hand, for solar radiation, precipitation and some other microclimatic variables, the graphs go up and down and the similarities are not specific to their position in the transect. We especially expected S-shaped gradients for solar radiation in the transition zone (Schmidt et al., 2017). Other authors like Erdős et al. (2013) and Wicklein et al. (2012) report significant gradients in solar radiation for north- and south-facing transition zones. However, the lacking S-shape of gradients does not mean that there are no relevant gradients per se. In terms of precipitation, the measurement tools tended to be dirty in the forest which might have made some measurements inaccurate. For air pressure, there might be no gradient on the measured scale. Shading of trees to a higher distance and the intensity of solar radiation might have influenced gradients in solar radiation.

The shape of the gradients may also be inverted over the course of the year: in summer, soil temperature was higher in the arable land compared to the forest (Schmidt et al. 2018). In winter, soil temperatures were lower in the arable land. Ewers and Banks-Leite (2013) argued that this is a buffering effect of temperature in the surrounding area of forests. Although they made their argument for tropical forests, we can support this for temperate forests. Another aspect is that the soil in the arable land is bare and unprotected to air temperatures during winter. Like others, we measured higher soil temperatures at the zero line compared to the forest interior (e.g. Chen et al., 1993; Remy et al., 2016).

The air temperature was only slightly different over the course of the year. Comparing air temperature gradients for summer months with the results of Erdős et al. (2013) or Heithecker and Halpern (2007), we came to similar results: forests are colder when compared to arable or grassland. A change in magnitude over the course of the year was also measurable (Wright et al., 2010). This is most probably due to changing foliage and plant cover. For summer months, we can give support for the correlation between distance to the zero line and lower air temperature presented in the meta-analysis by Arroyo-Rodríguez et al. (2017) and a review by Tuff et al. (2016). Although temperature should be closely related to solar radiation, we were unable to find monotonic patterns along the transects in these time series.

At the west-facing site, soil moisture was slightly lower in the forested transition zone relative to arable land and the zero line. This contrasts with the findings of Remy et al. (2016) as well as Riutta et al. (2012, 2016), who have reported drier zero lines. However, Farmilo et al. (2013) reported higher soil moisture for small fragments in contrast to continuous forest, which is comparable to a transition zone. The problem with these measurements is that they are difficult to compare accurately, as the two studies from Riutta et al. (2012, 2016) only measured soil moisture occasionally, and Farmilo et al. (2013) only four times, while we measured continuously for more than one year. The lack of comparability is problematic, as soil moisture influences the activity of

soil biota, which in turn is an important factor for matter dynamics and possible greenhouse gas emissions (Riutta et al., 2012). Moreover, it was not possible to show precipitation to be a main influencing factor for an altered soil moisture regime, as we did not find clear monotonic shifts along the transects for precipitation.

Another microclimatic generalisation can be derived from our results for direction of wind. The wind direction in the transition zone changes due to turbulences caused by obstacles (Figure 3.2). This is in line with other authors (see Schmidt et al., 2017). It is also supported by the average wind speed that changes at both sites and for nearly every month as we report in Schmidt et al. (2018): wind speed at 70 m in the forest was half that of 30 m in the arable land. This penetration distance, the spatial extent of higher wind speed in the forested transition zone compared to forest interior, is also in line with other authors (see Schmidt et al., 2017).

3.4.2. The significance of biotic effects in transition zones

A transition zone between forest and arable land of altered aboveground biomasses has a width of up to 65 m perpendicular to the zero line. Because of the distances between the plots, this is just an approximation. Nevertheless, the extent appears to be in line with the approximated extent of altered environmental gradients. Considering the whole transition zone, aboveground biomass has an inverted bell shape.

With respect to tree height and diameter as an indicator, we found lower aboveground biomass in the forest at the zero line. This was also reported for decreased tree heights at distances of 25 to 30 m by Ibanez et al. (2017) and for an urban pine forest by Veselkin et al. (2017). Wright et al. (2010) found the basal area to be lowest at the zero line but then stabilised at 20 m from the zero line. More generally, Islam et al. (2017) have found trees next to the zero line to be smaller and lower in diameter in fragmented forests, which could mean reduced carbon storage or wood volume (Veselkin et al., 2017). This is contrary to Hernandez-Santana et al. (2011) and Dodonov et al. (2013), who reported an increase in height towards the zero line. Remy et al. (2016) argued that wood volume was higher towards the zero line due to increased atmospheric N deposition (Remy et al., 2017a) and favourable light conditions compared to forest interior (Chen et al., 1993; Dodonov et al., 2013; Schmidt et al., 2017). Similar results are reported by Wicklein et al. (2012) who, in addition, found higher sapling density in north and south-facing transition zones. Most studies like ours only took trees into account, but not the bush and shrub layer. Islam et al. (2017) have described this as a problem, albeit a minor one. However, Erdős et al. (2014) report the highest vegetation cover in the transition zone between forest and steppe. In the light of this, height and diameter as proxies for lower aboveground biomass in forested transition zones might be not sufficient as shrubs, higher sapling density and herb biomass are not accounted for. These measures should be

considered when calculating biomass in transition zones. The influence of this, however, might be case specific.

Litterfall was lower at the west-facing site. One reason might be the windward direction of this site (Figure 3.2), as wind can carry litter into the forest and away from the zero line. In addition, the two to threefold higher average wind speed compared to the interior forest would substantially enhance litter removal in the forested transition zone. Lower litter cover and litter depth was also found by Watkins et al. (2003) close to roads compared to the forest interior.

The biomass in the cropped transition zone increased as distance from the zero line increased. This was also found by Mitchell et al. (2014) for soybean, with an increase of 55% to 117% from the zero line to 100 m in the arable land. Mitchell et al. (2014) argued that pest regulation has an influence on crop growth, and vice versa. On the other hand, pest regulation is influenced by the distance to forest as well as the general landscape structure (maximum pest regulation near the forest fragment; Mitchell et al., 2014).

Lower air and soil temperatures and altered solar radiation, as reported by Gray et al. (2002) for forest gaps, may cause these effects. Especially, dimmed solar radiation is reported to have a negative influence on crop growth in transition zones (Dufour et al., 2013; Malik and Sharma, 1990; Nuberg, 1998), but also affects species composition (Erdős et al., 2014). Another possible reason for these effects could be an altered soil water regime in the transition zone, e.g. drier transition zones as described in the discussion on microclimate (see our review for more explanations, e.g. altered evapotranspiration). Kort (1988) argued that decreased crop production within 50 m is due to competition between crops and trees for water and solar radiation. In addition, manoeuvring heavy agricultural machinery at field edges (headland) might have compacted the soil, which would reduce crop growth (Hamza and Anderson, 2005).

Since we only measured biomass, we cannot make predictions about actual yield. However, we found visible proof that crop anthesis lags behind in the cropped transition zone to up to 15 m from the zero line (Figure 8.5; Schmidt et al. 2018). That most likely affects the degree of ripeness of crops in the transition zone, and might cause lower yields there, as the harvest is on a fixed date. On the other hand, Ricketts et al. (2008) reported increased pollination in the transition zones. In our case, shading by the trees most probably caused delayed flowering.

Crop growth in transition zones adjoining forest fragments is influenced by several biotic and abiotic variables. Moreover, the landscape structure (connectivity, composition, configuration) plays an important role (Seppelt et al., 2016). However, according to Kort (1988) and Mitchell et al. (2014) the spatial extent perpendicular to the zero line of decreased crop growth appears between 15 and 50 m.

3.4.3. Feedback from abiotic and biotic effects

The content of soil carbon and nitrogen was primarily elevated at the zero line. An explanation might be an accumulation of nitrogen from fertilisation and higher atmospheric N deposition (Remy et al., 2017a). In terms of carbon, a strip of approximately two to three meters with a grassland character directly at the edge (see photos in Schmidt et al., 2018) might have accumulated carbon in the soil over the years. Therefore, a transition zone can have a maximum width of 50 m perpendicular to the zero line in our experiment. This width is in line with our findings that altered conditions in soils of transition zones occur within 10 to 20 m with a maximum of 50 m (Schmidt et al., 2017). In general, the levels of soil carbon and soil nitrogen were low, most likely due to the sandy soils (Schmidt et al., 2018) in this region. This and the rather intensive use of N mineral fertilisers leads to low C:N ratios in the mineral soil. The gradients for C and N levels are most likely bell-shaped, because there was no statistical difference between the arable land and the forest – in spite of what we generally expected and in part due to findings by other authors regarding soil and litter deposition (Stanton et al., 2013; Toledo-Aceves and García-Oliva, 2008) – but there were higher values at the zero line.

Higher C and N content levels cannot be ascribed to reduced litter input, as Remy et al. (2016) found no effect of distance for C and N in needles and leaves. In addition, we only found significantly less litterfall at one site. However, C and N stocks in the mineral soil were higher at the zero line by approximately one-third (Remy et al., 2016), which is in line with our findings. For N, the reason might be higher atmospheric N deposition at the zero line (Remy et al., 2017a; Wuyts et al., 2008), and N being released more quickly from litter and wood (Bebber et al., 2011; Didham, 1998; Remy et al., 2017b). On the other hand, Moreno et al. (2014) as well as Vasconcelos and Laurance (2005) reported no difference in litter decomposition rates at the zero line relative to the forest interior. It is still unclear what role soil moisture plays in this context. Didham (1998) and Remy et al. (2017b) also found no effect for air temperature (Didham, 1998). However, Riutta et al. (2012) and Simpson et al. (2012) reported a correlation between soil moisture, microbial activity and litter decomposition. It could be that the effect of single trees on litter decomposition is underestimated (Hastwell and Morris, 2013), which makes processes even more complex.

3.5. Conclusions and outlook

Like other authors, we report spatially explicit environmental gradients, their biotic effects and feedback relations. For deeper understandings of landscape processes, researchers often apply mechanistic modelling (Ries et al., 2017). In most of the modelling studies that include more than just one ecotope, different ecosystems are modelled independently, without consideration for any lateral connections. Some habitat models have considered at least biotic exchange through individual movement (Fletcher et al., 2016), and hydrological models at watershed level have also

included lateral water flows (Hwang et al., 2012). However, cross-ecosystem relations are rare in models for biomass growth and ecosystem service assessment. Depending on the goal of the model, it may be necessary to account for transition zone gradients and their effects, e.g. when applying forest and crop growth models or biogeochemical models on the landscape scale. Some of the feedback relations seem obvious: soils close to the zero line may contain higher soil carbon content due to litterfall from adjacent trees, while trees are smaller towards the zero line and may store less carbon. Crop yield depressions in the transition zone might result from shading or from competition for water. Higher air humidity at the edge of the forest could decrease evapotranspiration and thus increase the risk of fungal infections, which could consequently affect yields and the quality of agricultural products. These effects – and probably many more – all affect the provision of ecosystem services and hence human wellbeing. With deeper insights into transition zones, we may be able to connect up forest and crop growth models at their ecological boundaries and explore more of these assumed feedback patterns, disentangling some of the complexity. This would be an important step towards a holistic understanding of processes on the landscape scale.

4. Modelling yield response to shading in field-to-forest transition zones in heterogeneous landscapes⁶

Abstract

In crop modeling and yield predictions, the heterogeneity of agricultural landscapes is usually not accounted for. This heterogeneity often arises from landscape elements like forests, hedges, or single trees and shrubs that cast shadows. Shading from forested areas or shrubs has effects on transpiration, temperature, and soil moisture, all of which affect the crop yield in the adjacent arable land. Transitional gradients of solar irradiance can be described as a function of the distance to the zero line (edge), the cardinal direction, and the height of trees. The magnitude of yield reduction in transition zones is highly influenced by solar irradiance—a factor that is not yet implemented in crop growth models on a landscape level. We present a spatially explicit model for shading caused by forested areas, in agricultural landscapes. With increasing distance to forest, solar irradiance and yield increase. Our model predicts that the shading effect from the forested areas occurs up to 15 m from the forest edge, for the simulated wheat yields, and up to 30 m, for simulated maize. Moreover, we estimated the spatial extent of transition zones, to calculate the regional yield reduction caused by shading of the forest edges, which amounted to 5% to 8% in an exemplary region.

Keywords: Edge effect, transition zone, solar irradiance, crop growth, maize, wheat

4.1. Introduction

Food provision is a fundamental ecosystem service with an emerging importance. Model-based projections of agricultural yields, as an indicator for food availability, have developed into an essential tool to derive strategies for a sustainable food supply, to meet the demands of an increasing world population. Crop models that are developed for this purpose often simulate single plants that represent a certain area. These models usually incorporate management activities, climate, and soil conditions. However, these represented areas are often not put into the landscape context, and it is not clear, until now, which important feedback mechanisms the models are not able to capture, for this reason. The heterogeneity of agricultural landscapes, in terms of land cover, as well as landscape structures and elements (e.g., forested areas, trees, and hedges), impose different effects on the plants that grow there. These landscape elements differ

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All supplementary materials (Table S1 and S2) are available online at www.mdpi.com/2077-0472/9/1/6/s1

from arable land, in their physical or biological nature, as they host different species or receive different types of management. One obvious difference is the height of trees and shrubs, compared to smaller crops of arable land, a feature that dominates the visible appearance of such landscape elements. The height difference affects the physical conditions of the habitats, at the edge between such landscape elements and the surrounding agricultural land, in which crops grow to a maximum of 2 to 3 m height. These transition zones (Schmidt et al., 2017) and their specific environment, are expected to play a vital role in the productivity and biodiversity of agricultural landscapes. Hence, crop models need to include these transition zones if they want to accurately model crop yields across space.

Forested areas, trees and shrubs generate temporally shaded areas, reducing the solar irradiance input for those plants that grow in the shade and their potential biomass accumulation. Depending on the definition used (Schmidt et al., 2017), the area of these transition zones from forested to non-forested matrices, makes up one fifth of the global forested area (Haddad et al., 2015). Shading of the adjacent non-forested areas in the transition zones, has various effects (Schmidt et al., 2017; Schmidt et al., 2019 (submitted)) the reduction of solar irradiance has direct consequences on temperature, evapotranspiration (Laurance et al., 2011; Nuberg, 1998), and, as a consequence of reduced plant growth and evapotranspiration, it also has an effect on the soil moisture (Gray et al., 2002; Kort, 1988). Spatial gradients of solar irradiance change drastically from the zero line of the forest (the forest edge), towards the more open space of the adjacent cropland areas (Kapos, 1989; Matlack, 1993). The spatial extent of the shading is influenced by the height of the trees, whereas the transmittance is influenced by the species composition, foliage density, and the type of foliage (Kort, 1988; Gholz et al., 1991; Voicu & Comeau, 2006). Moreover, the cardinal direction of the edge and azimuth of the sun play an important role, by determining the relative position of the shading element, with respect to the course of the sun (Matlack, 1993).

The reaction of plants to increased solar irradiance follows two main pathways: (1) A higher radiation turn-over on plants and soil surface leads to higher temperatures of the plant tissue, which directly affect all physiological processes in plants, including photosynthesis and respiration. (2) Higher temperatures at the leaf level, increase the water vapor deficit of the surrounding air volume and, thus, increase the water loss through the plant's open stomata. Together with an accelerated photosynthesis and the resulting higher water consumption and nutrient demand, the consequence is a higher transpiration rate and water uptake from soil. A sun-lit plant will, therefore, deplete its soil water reservoir, much quicker than a shaded plant, with an added, elevated evaporation taking place in soil that is exposed to sunshine. As transpiration is limited to the availability of soil moisture, plants close their stomata to reduce transpiration as soon as soil water resources fall below the soil's permanent wilting point. A reduction of photosynthesis and growth, including leaf area and yield formation, is the

consequence. A reduced leaf area will also affect the growth and yield formation at later stages, when water supply may eventually have returned to sufficient levels again, just by reducing the potential area for the interception of sunlight. This interplay between the solar irradiance, soil moisture availability, evapotranspiration, and plant growth, is complex (Figure 4.1). However, a detailed understanding of the balance between a sufficient solar irradiance for reaching full growth potential, and the limitations that keep the plant below its potential, is essential, in agricultural systems. As many of these limiting factors have a landscape dimension, a precise knowledge about how limiting factors emerge at the landscape scale and the ecophysiological functioning of these processes, and how and where forested areas and transitions zones may influence these processes, is critical, to upscale the predictions of biomass production, and yields to a landscape level and beyond.

Most crop growth models do not account for the effects of shading at the interception of trees and crops (Dufour et al., 2013). Some consider management techniques (Ghaffari et al., 2001) or the impact of climate change (Juin et al., 2004). However, the reactions of crops on the limited light resources, are diverse, for the different crop varieties and have to be considered, to model crop growth in these mixed agro-ecosystems (Jackson, 1987). Clearly, there is a competition for light, as reported for wheat (Chirko et al., 1996; Li et al., 2008), maize (Huxley et al., 1994), and soybean (Reynolds et al., 2007), but these results are often based on studies in agroforestry systems and do not consider the forest edges.

Almost all process-based simulation models for crop growth and development that are employed for impact assessments of food productivity, are one-dimensional. While soil moisture and solar irradiance are essential inputs to drive their simulations, solar irradiance is usually considered to be only dependent on the time of the year and the latitude, and water availability is typically implemented in a way to reflect the small-scale heterogeneity, for example, per square meter. However, at the landscape scale, solar irradiance is affected by terrain and other features of landscape composition—hills and slopes shade cropped areas (Reuter et al., 2005), as do forests and hedges (Jackson, 1987; Reynolds et al., 2007). Shading of plants is implemented in some crop agro-ecosystem models to simulate the simultaneous growth of different plants, for example, in intercropping systems (Brisson et al., 2004; Corre-Hellou, et al., 2009; Knörzner et al., 2011). However, to our knowledge, shading parts of the arable land from the adjacent trees or shrubs, is not implemented in any agro-ecosystem models (Dufour et al., 2013) and, hence, is not considered in large-area yield predictions, except for agro-forestry systems (Artru et al., 2017; Artru et al., 2018). Additionally, some agro-ecosystem models are coupled to the hydrological models to reflect the hydrological and soil-specific heterogeneities, when simulating processes at the watershed level (Bithell & Brasington, 2009; McNider et al., 2015; Srinivasan et al., 2010). However, to our

knowledge, none of them account for the altered soil moisture conditions, due to shading in the transition zone between the agricultural land and the forest.

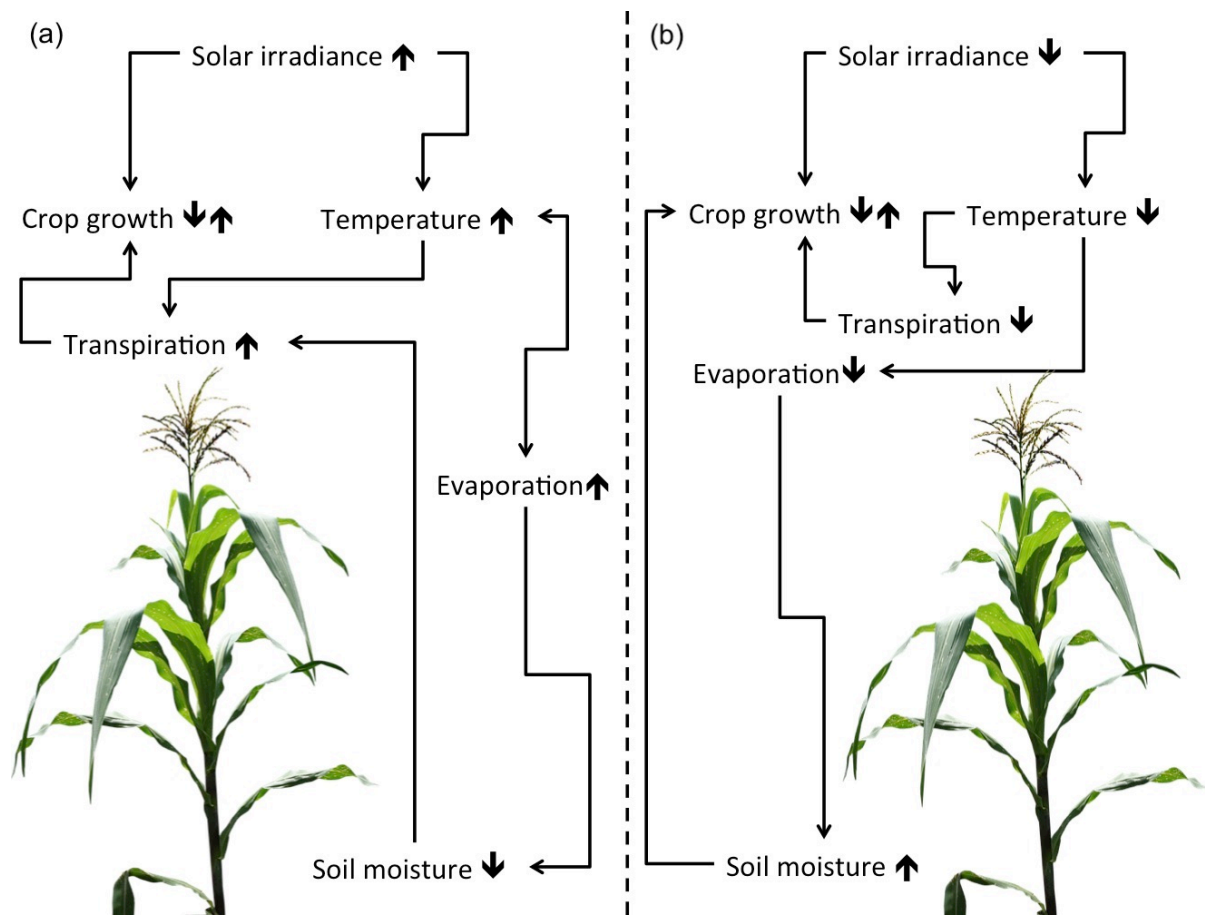


Figure 4.1: Relationship of processes according to higher (a) and lower (b) solar irradiance that affect the plant growth.

In this study, we simulated the effects of shading by a forest edge on the crop yield and soil moisture of an adjacent field, and investigated the direct effects of reduced irradiance against the indirect effects of the soil moisture. We hypothesized that (i) crop yields within the transition zone is significantly reduced through a reduced solar irradiance, and that (ii) soil moisture feedback plays a significant role in the yield formation in this zone. Generally, yield reduction in transition zones are suggested to be significant in agricultural landscapes. Therefore, we (iii) exemplarily calculated the impact of shading on yield, on the landscape scale for the federal state Brandenburg, Germany, to validate this major hypothesis.

4.2. Methodology

For our approach, we used a virtual landscape model with a forest matrix and surrounding arable land that we simulated in the ArcGIS (ArcGIS Desktop 2011, ArcMAP 10.4.1, ESRI Inc., Redlands, CA, USA). To model solar irradiance, data on the sun angle, geolocation, day of the year, time of the day and height of the shading element were necessary (Brandle et al., 2004). This simulation

included a calculation of the course of the sun for Berlin, Germany (52.500° N, 13.405° E), for one year. The intensity of solar irradiance was corrected to the intensity of the reduced solar irradiance, in the shaded areas. Climate data were combined with the simulated solar irradiance for different distances and cardinal directions from the forest, to simulate the possible yield reductions along the transects, in different directions, from the forest edge into the cropland areas.

4.2.1. Shadow modelling

Calculation of azimuth and altitude

To model the course of the sun, daily information of the azimuth and altitude of the sun were necessary. This could be solved with astronomical equations, according to the NOAA (National Oceanic and Atmospheric Administration, MD, USA) Solar Calculator (ESRL, 2018). First, the fractional year (n) was calculated:

$$n = \left(\frac{\pi}{180}\right) \times \frac{(x - 1 + \left(\frac{(\frac{hour + min}{60}) - 12}{24}\right))}{365}$$

where x is the day of the year. Second, we needed to define the solar declination angle ($declin$ as radians) and estimate the equation for time ($timeeq$ in min):

$$declin = 0.006918 - 0.399912 \times \cos(n) + 0.070257 \times \sin(n) - 0.006758 \times \cos(2n) \\ + 0.000907 \times \sin(2n) - 0.002697 \times \cos(3n) + 0.00148 \times \sin(3n)$$

$$timeeq = 229.18 (0.000075 + 0.001868 \times \cos(n) - 0.032077 \times \sin(n) \\ - 0.014615 \times \cos(2n) - 0.040849 \times \sin(2n))$$

Next we calculate the solar hour angle ($hourangle$) by

$$hourangle = 15 \times \left(hour + \frac{min}{60} - \frac{(15 - long)}{15} - 12 + \frac{timeeq}{60} \right)$$

where $long$ is the longitude (°). Then, we calculated the altitude by:

$$\begin{aligned} altitude = \sin(x) = & \sin(K \times lat) \times \sin(K \times declin) \\ & + \cos(K \times lat) \times \cos(K \times declin) \times \cos(K \times hourangle) \end{aligned}$$

where

$$K = \frac{\pi}{180}$$

and *lat* is the latitude (°). The solar zenith angle (*azimut*) could then be calculated from the hour angle (*hourangle*) and the solar declination (*declin*), using the following equation:

$$azimut = \cos(y) = \frac{-[\sin(K \times lat) \times \sin(K \times h) - \sin(K \times declin)]}{[\cos(K \times lat) \times \sin(\arccos(\sin(K \times h)))]}$$

where

$$h = \frac{\arcsin(x)}{K}$$

Both, altitude and azimuth were calculated with a two-digit accuracy. For the altitude, the mean absolute variance was 0.22° with a maximum of 0.4°, from 8 am to 4 pm. For the azimuth, the mean absolute variance was 0.14° with a maximum of 0.35°, from 8 am to 4 pm.

Solar irradiance and canopy transmittance for modelling

We used the Photovoltaic Geographical Information System (Joint Research Centre Photovoltaic Geographical Information System (PVGIS)) provided by the European Commission, to obtain data on the solar irradiance. We used Berlin, Germany (52.500° N, 13.405° E) as an exemplary location, with an elevation of 35 m.

As the solar irradiance was scattered through the foliage absorption, we used an approximated 0.3 transmittance factor, according to Gholz et al. (1991), to simulate the shading from trees. The values they reported for slash pine (*Pinus elliottii*) were a suitable proxy, as trees in the Berlin area were mostly scots pines (*Pinus sylvestris*). This value for reduced solar irradiance was constant for the whole simulation period.

Simulation of shadows using geographical information systems

To simulate the shadowing effects with ArcGIS, we used the azimuth, altitude, solar irradiance and reduced solar irradiance, as described above. A block representing a forest with an area of 30 × 30

meters and a height of 20 m was implemented into a surrounding flat area of 100 m, in each direction that represented the arable land (Figure 4.2). The course of the sun was calculated at hourly intervals, for each day of the year, with the ArcGIS (ArcGIS Desktop 2011, ArcMAP 10.4.1, ESRI Inc., Redlands, CA, USA) commands “hillshade” and “shadow”. Shadows were simulated pixel-wise (1 × 1 meters), writing 1 for the shaded and 0 for the sun-lit areas. In total 4402 separate grid files were calculated and saved. The shadowed areas were multiplied, with an average transmittance factor of 0.3 (see above). Finally, the command “Zonal statistics” was used to generate a query of the hourly reduced solar irradiance, at certain points, around the virtual forest. The points were located in the northern (N), eastern (E), southern (S), and western (W) directions at a distance of 1, 2, 5, 10, 15, 20, 25, 30, 40, 50, 60, 70, 80, 90, and 100 m from the zero line of the forest. The data were stored in a table and were used as the inputs into a crop model.

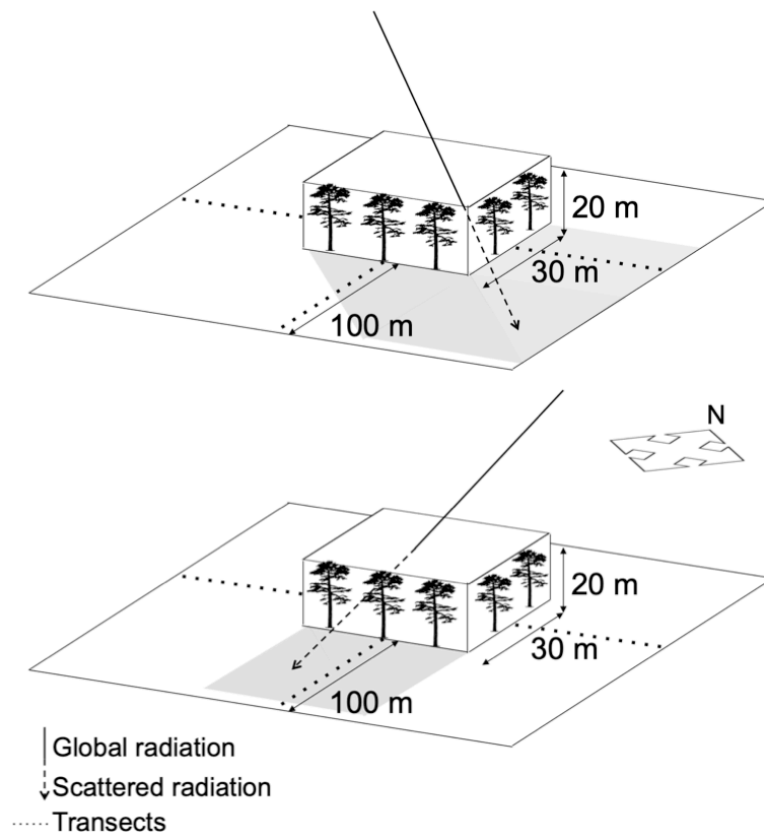


Figure 4.2: Sketch for two exemplary angles of penetrating solar irradiance, through the forest, in a virtual landscape and the shadows it casts. The drawn-through line represents the solar irradiance for Berlin, Germany (52.500° N, 13.405° E). The dashed line denotes the reduced solar irradiance (shadow). The dotted lines represent the directions of the transects for which crop yield was simulated.

4.2.2. Crop modelling

Simulations

Simulations were conducted using the MONICA (Model for Nitrogen and Carbon in Agroecosystems) crop growth simulation model (Nendel et al., 2011). MONICA works in a one-

dimensional way and calculates the values for a minimum spatial resolution of 1 m² surface area and 2 m depth. On daily time-steps, the crop growth and yield of the silage maize (*Zea mays* L.) and the winter wheat (*Triticum aestivum* L.) was simulated, using thirty years of climate data (see below), each. Every single year was simulated, independently, without any carry-over or crop rotation effects. This led to $n = 29$ simulations for the winter wheat, with sowing in the first year (1988) of the climate data. For maize, there were $n = 30$ simulations.

Simulations were performed for the plots at different distances (1, 2, 5, 10, 15, 20, 25, 30, 40, 50, 60, 70, 80, 90, and 100 m) from the forest edge, along the cardinal directions (north, east, south, west). This resulted in a total of 3540 observations.

For sowing and harvesting, the implemented functions were used, with an earliest date of sowing for maize, between 4th and 20th of May, and a latest harvest date of 30th of September. For the winter wheat, the sowing period was set from the 5th of October to 8th of November. For both crops, the sowing date of the respective year depended on a minimum temperature of several subsequent days in a row. The sowing dates were based on the farmers' knowledge, and were typical for this region. The parameterization of the MONICA model for wheat and maize was based on several previous studies (Asseng et al., 2013; Bassu et al., 2014; Pirttioja et al., 2015).

For simulations of the soil moisture, an initial value of 100% of the field capacity was set. Evapotranspiration was affected by the shading effects, in terms of the reduced solar irradiance, during the simulations. MONICA separated the soil into 10 cm layers. For our simulations we used soil characteristics, as described below, defining three soil horizons (Table 4.1). The maximum rooting depth for the winter wheat was set to 150 cm, and for the silage maize to 200 cm.

The site was simulated, using the example data implemented in the MONICA. This comprised a latitude of 52.8093° N, no slope, and 0 m above the sea-level. No virtual fertilizers were added and the response to nitrogen was switched off, to avoid it to be limiting. The soil profile had the following characteristics, which were representative of the soils in the region from which the climate data were retrieved—the Federal state of Brandenburg.

Table 4.1 Soil characteristics implemented in the MONICA (Model for Nitrogen and Carbon in Agro-ecosystems). The texture class is given according to the World Reference Base (WRB) for Soil Resources.

Depth [cm]	Soil organic carbon [%]	Texture class	Raw density [kg m ⁻³]
0 to 30	0.8	loamy sand	1446
30 to 40	0.15	loamy sand	1446
40 to 200	0.05	loamy sand	1446

Climate data

Climate data was retrieved from the German Weather Service (DWD) for Berlin-Tegel (station number 430) for the years 1988 to 2017 (30 years). We used average daily wind speed, air temperature, humidity, precipitation as well as the minimum and maximum air temperature. This fulfilled the minimum requirements for the MONICA model. Solar irradiance was added to the climate data, according to our approach, as described above.

To disentangle the effects of soil moisture and solar irradiance, we simulated the yields for the driest and the moistest years. We compared the total annual precipitation for the three driest (1999, 2003, and 2016) and moistest (1993, 2007, and 2017) years, in the respective time periods.

Spatial Extent of the Transition Zones

For the calculation of the transition zones in Brandenburg, we used the German Biotope Map. According to the code of land use, the area of transition zones of forest and non-forest land use were calculated using the buffer tool ArcGIS (ArcGIS Desktop 2011, ArcMAP 10.4.1, ESRI Inc., Redlands, CA, USA). The buffered areas were created with different distances, perpendicular to the zero line (20, 50, and 100 m), in both directions, into the interior of either the forest or the non-forest land, and to the outside. After that, these areas were overlaid and clipped. Finally, the share of the transition zones, compared to the whole area of the respective land-use type (forest: 11045 km², non-forest: 18799 km²), was calculated.

4.3. Results

4.3.1. Solar irradiance

Solar irradiance increased with increasing distance to the zero line (Figure 4.3, Table S1). It was least for the north-facing transition zone. Compared to the south-facing transition zone (12.1 MJ m⁻² d⁻¹), which can be considered as fully irradiated; only one-third (4.3 MJ m⁻² d⁻¹) of the irradiance is potentially available close to the zero line in the north. At 30 m, the difference between the north and south, compared to the zero line is only 8%, and almost vanishes from 50 m on.

West- and east-facing transition zones are nearly equal in their average daily solar irradiance. Close to the zero line it is, approximately, two-third (8.5 to 8.6 MJ m⁻² d⁻¹) of the solar irradiance in the south-facing transition zone. At 30 m, the difference to the south-facing transition zone is only 5 to 6% and almost vanishes from 50 m on. An irradiance gradient, with respect to distance to the zero of shading in the south-facing transition zone, is hardly distinguishable.

4.3.2. Soil moisture

The simulated average soil moisture was highest for both crops, up to ca. 15 m from the zero line, at the north-facing side (Figure 4.4). Yet, the highest difference between the cardinal directions is

at 1 m from the zero line, with ca. $0.19 \text{ m}^3 \text{ m}^{-3}$ (north), $0.18 \text{ m}^3 \text{ m}^{-3}$ (west and east), and $0.17 \text{ m}^3 \text{ m}^{-3}$ (south) for wheat (Figure 4.4). For maize, the highest difference in soil moisture values were ca. $0.202 \text{ m}^3 \text{ m}^{-3}$ (north), $0.199 \text{ m}^3 \text{ m}^{-3}$ (west and east), and $0.196 \text{ m}^3 \text{ m}^{-3}$ (south). After not more than 50 m from the zero line, soil moisture values did not differ between the shaded and the non-shaded areas, for both crops, with ca. $0.17 \text{ m}^3 \text{ m}^{-3}$, for wheat and $0.195 \text{ m}^3 \text{ m}^{-3}$ for maize.

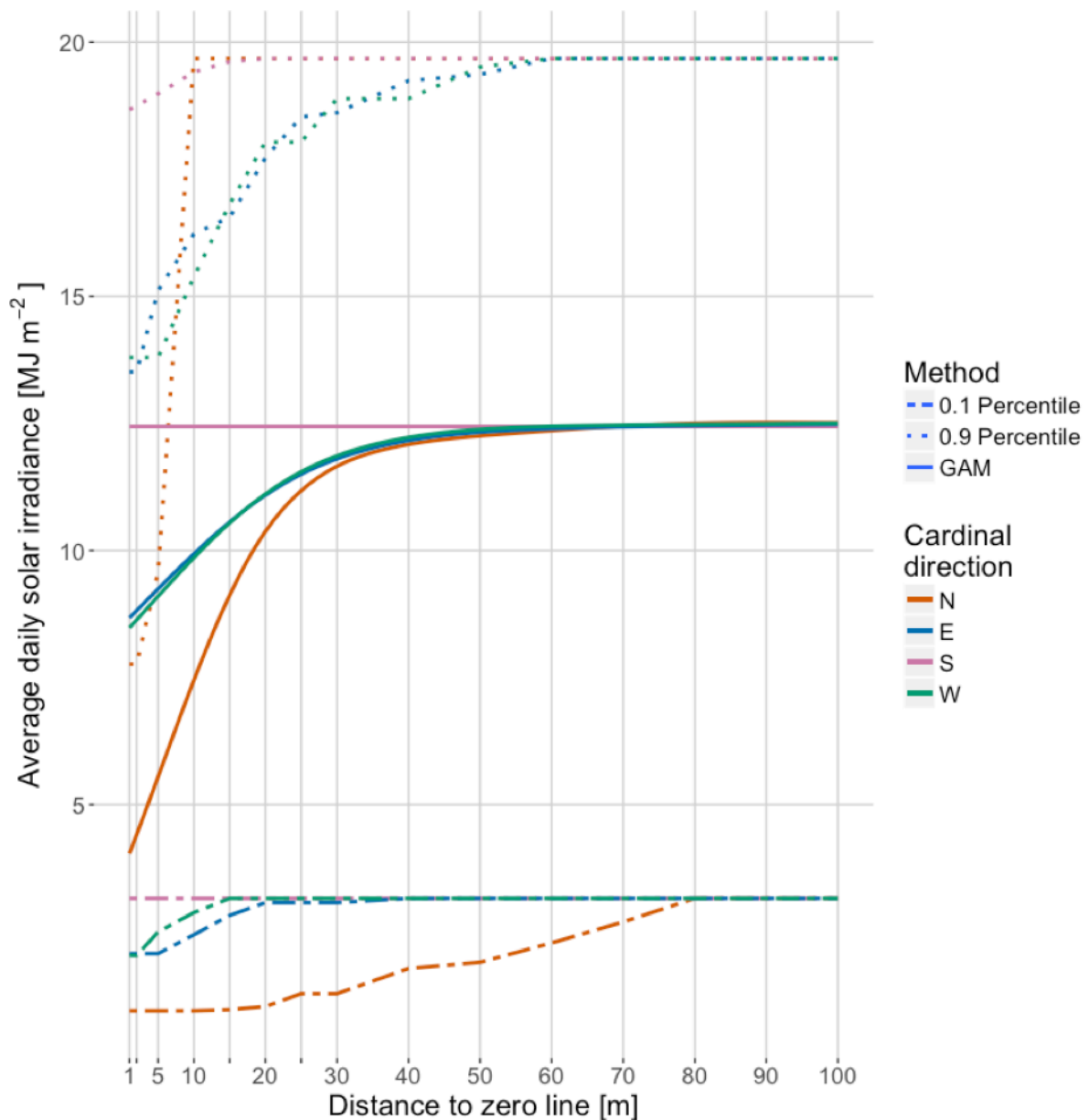


Figure 4.3 Simulated average daily solar irradiance for the four cardinal directions (north, east, south, and west) of the arable land, shaded by a forest, with respect to distance to the zero line. The solid lines represent the general additive models (GAM), fitted to the data, and the dashed lines represent the 0.1 and 0.9 percentile.

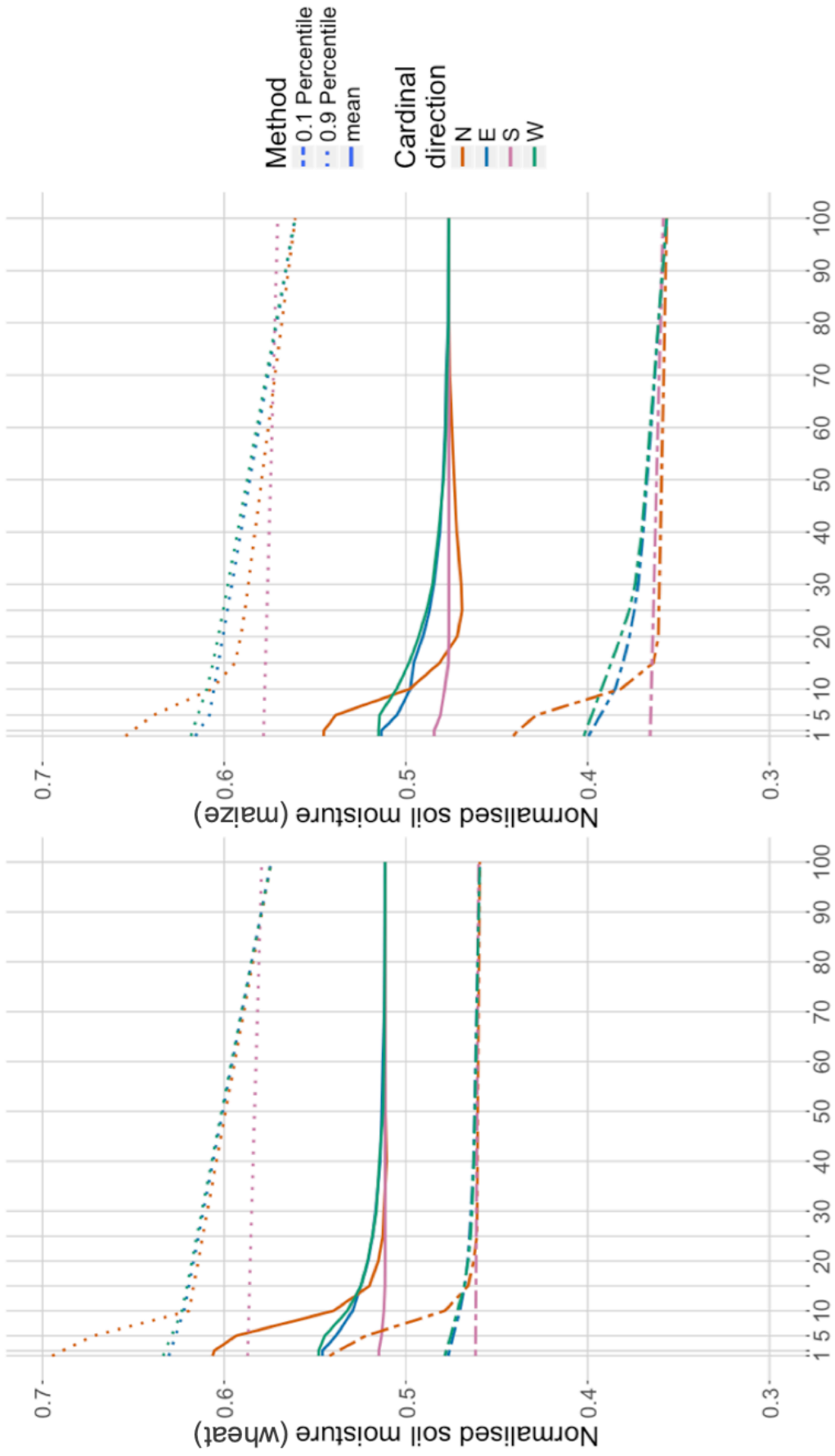


Figure 4.4 Simulated average daily soil moisture for wheat (a) and maize (b), with respect to the distance to the zero line, and the cardinal direction of a virtual forest matrix shading the adjacent arable land. The solid line represents the mean and the dashed lines represent the 0.1 and 0.9 percentiles.

4.3.3. Yield

Winter wheat yield increased with an increasing distance, up to 10 m from the zero line, in all cardinal directions (Figure 4.5, Table S2). Yield was 2% lower at 1 m, compared to 10 m, at the south-facing side and 8% lower at the east-facing side. At the west-facing side, the lowest yields were simulated at 1 m, with a reduction of 8%, compared to 15 m. The highest yield reduction of -37%, was simulated for the north-facing side, at 1 m compared to 15 m.

Maize yield increased with the increasing distance from the zero line, in all cardinal directions (Figure 4.5, Table S2). In the south-facing transition zone, the yield depression was least, with a reduction of 9% in dry mass, at 1 m compared to 15 m. Beyond that, there was nearly no difference in the average yield per year. The west- and the east-facing side had very similar results, where yields were lowest at 1 m. This was a reduction by one-third, compared to the values at 50 m, where the yield was not affected. Thus, a substantial yield reduction occurred at a 1 to 30 m distance (-31% to -4%, compared to 50 m). The highest reduction in yield (-54%), occurred at the north side, between 30 m and 1 m.

Separating the driest and the wettest years from the simulated 30-years time-series, we found that in the three driest years, the average wheat yield was 33% lower, at 1 m in the south, compared to the three wettest years (Figure 4.6, Table S2). In the east- and the west-facing directions, the difference at 1 m was -11 to -12%. In the north, the yield was 4% lower. In the three wettest years, the yields were 2% lower in the north, up to 10 m from the zero line. At around 50 m, there was no difference between the cardinal directions. We also found that along the north-facing transect, the yields peaked at a 10 m distance, in the dry years, producing 19% more than at the 50 m distance, where the effect ceased. Along the other transects, the yields were only slightly higher and the peak was closer to the zero line. In the wet years, the yield increased towards the interior of the field, along all transects.

At 1 m, the maize yields were 7% higher in the south, in the driest years, compared to the wettest years (Figure 4.6, Table S2). From 1 m to 10 m, the yields for the driest and the wettest years, were similar, at 10 m. At a greater distance, the yields in the driest years were up to 13% higher, compared to the wettest years in the north and south. From 25 m onwards, the north- and south-facing side reached the yield potential, in the driest years. In the east and the west transects, the yields in the driest years were approximately 4% (15 m) to 7% (70 m) higher, than in the wettest years. From 60 m onwards, the yields in the wettest years, and from 80 m onwards, the yields in the driest years, were all similar, with a difference of 13%, between treatments. For maize, we observed no yield peaks in the dry years, in the transition zone.

4.3.4. Upscaling the shading effects on the crop yield

The share of the non-forested area at short distances from the adjacent forests in Brandenburg, is presented in Table 4.2. Not all non-forested area, but 13,230 km², is agricultural land, according to the farmers' association (Landesbauernverband Brandenburg); 75% of this area is cropped land, which is about 9923 km². Maize is grown on 18% (1786 km²) of the cropped land and wheat on 13% of it (1290 km²).

Table 4.2 Calculated share of the transition zone (TZ), in the area of Brandenburg, for the different TZ lengths.

	10 m *	15 m	20 m	30 m *	40 m *	50 m	100 m
Non-forest	4.8%	6%	7%	9.6%	12%	15.2%	26.3%

Values with asterisks (*) were interpolated linearly between the modeled values of those without asterisks.

Considering the area of the crops in the Brandenburg, the simulated values for the yield from the MONICA model, the length and share of the transition zone and an, approximated yield reduction in the transition zone, we arrived at an overall yield loss of 5.4% for wheat and 8.5% for maize, due to the shading by the forest edges (Table 4.3).

Table 4.3 Calculation of the yield reduction due to shading of the forest edges, for the winter wheat (grain yield) and the silage maize (whole plant), in Brandenburg. Transition zone (TZ) shares were taken from Table 4.2, TZ length and yield reduction in TZ are the calculated values in this article. Yield reduction in Brandenburg was calculated, in comparison to no-shading in TZ.

Crop	Area (ha)	Average Yield (kg ha ⁻¹)	TZ (m)	TZ Share (%)	Yield Reduction in TZ (%)	Yield Reduction in Brandenburg (%)
Winter wheat	129000	6314	15	6	10.4	5.4
Silage maize	178600	2564	30	9.6	12.3	8.4

TZ = transition zone.

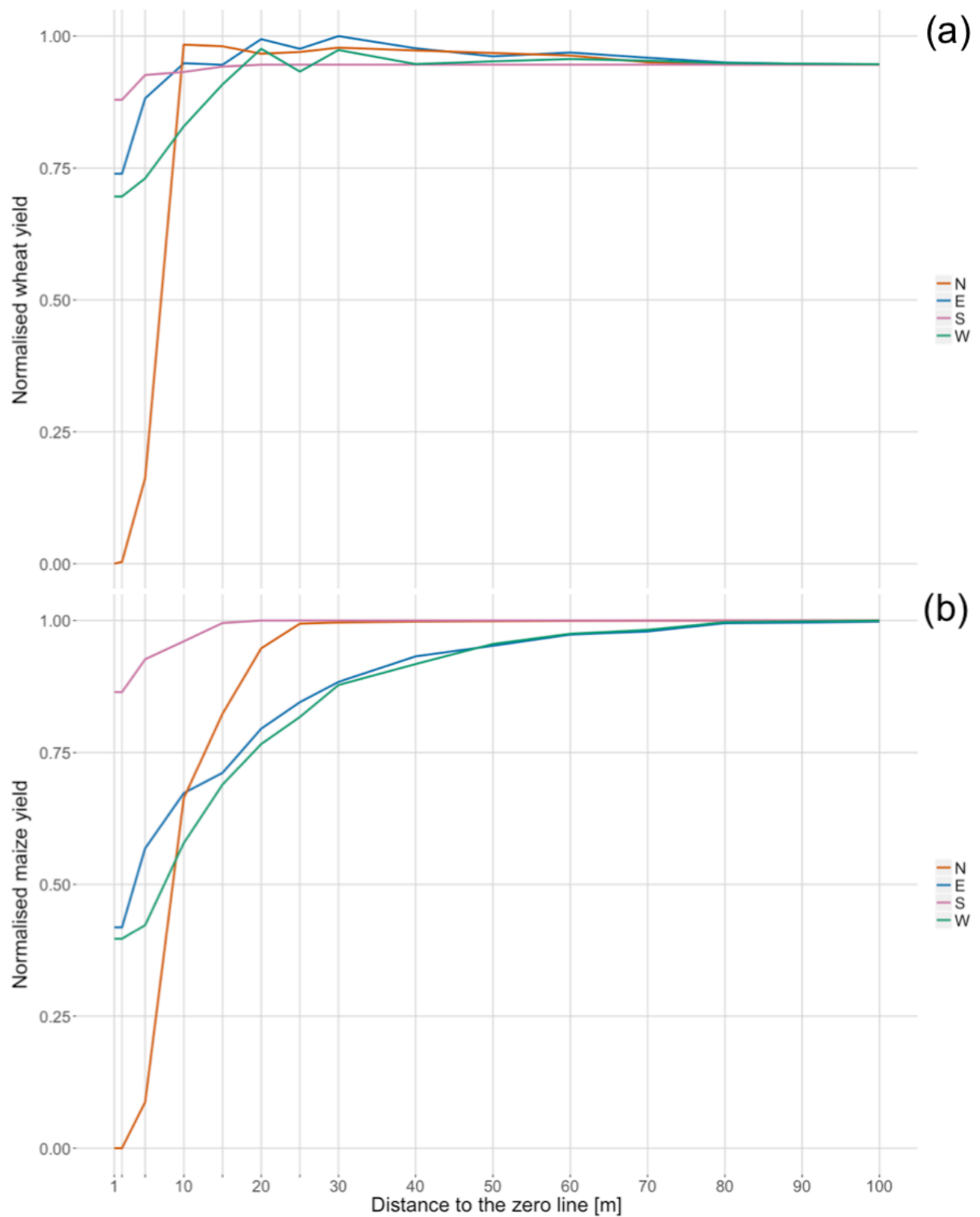


Figure 4.5 Simulated yields for wheat (a) and maize (b), in the four cardinal directions (north, east, south, and west), with respect to the distance to the zero line and the resulting shading of a forest matrix. The mean values of the annual simulations for 30 years of climate data are shown.

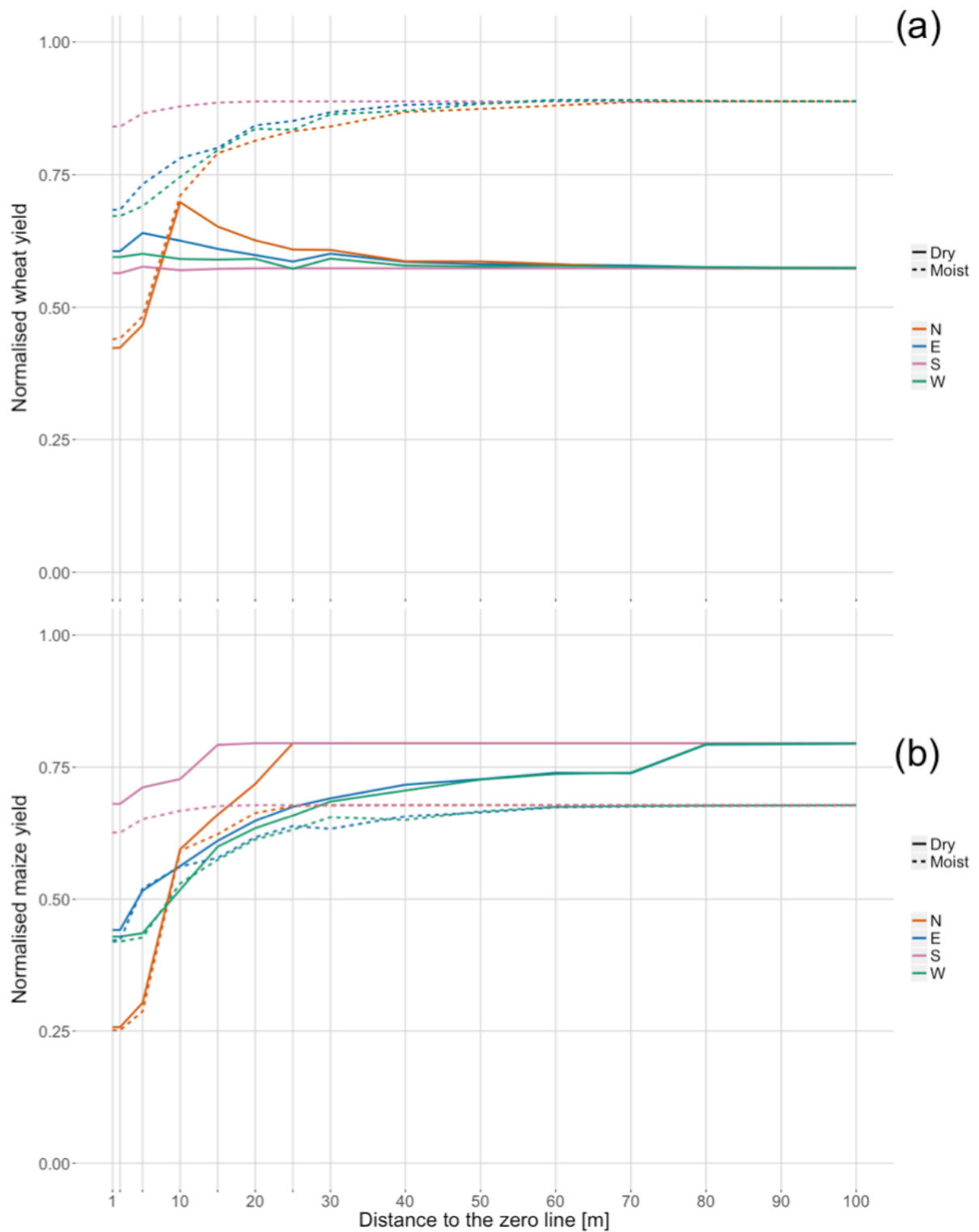


Figure 4.6 Simulated yields for wheat (a) and maize (b) in the four cardinal directions (north, east, south, and west), with respect to distance to the zero line and the resulting shading of a forest matrix. The solid line represents the mean for the three driest years and the dashed line represents the mean of the three wettest years, within the 30 years of climate data.

4.4. Discussion

4.4.1. Relation between yield reduction, soil moisture, and solar irradiance

In general, the gradients of yield reduction (Figure 4.5) correlated with the modeled solar irradiance (Figure 4.3). The reduction of solar irradiance was the strongest in the north, and the spatial extent of shading was similar in the east and west. The general shapes of the gradients for the simulated yields, as well as solar irradiance, were similar and the solar irradiance could be adduced as a major explanatory variable for the decreasing yields, in the transition zone. Although a gradient was also simulated for the soil moisture, it was very small.

To disentangle the effects of soil moisture and solar irradiance, we separated the three driest and three moistest years, from the simulations, under the assumption that in the wet years, crop growth would not be limited by water deficiency, but only by radiation. Indeed, we found that yields for wheat were reduced in the shaded transition zone, while outside, towards the field, simulated wheat yields remained stable. In the dry years, as a contrast, wheat was limited by water availability, along the transect, expressed by a lower yield level, in the simulations. However, in the northern transition zone, we observed higher simulated yields, at 5–40 m distance, from the zero line, as compared to the field matrix. A similar effect, but a less pronounced one was also observed for the western and the eastern transect.

Maize, obviously, grew better in drier than in the wetter years in the simulations, as the yields were higher in the drier years compared to the moistest years, which reflects the nature of maize, which is better adapted to warm and dry conditions than wheat (Nuberg, 1998). In the shaded transition zone, yields were equal to the moistest years, indicating that the yields, at no time, were limited by water, but were significantly limited by a reduced radiation input in the transition zone.

With a comparably simple set-up of tools and a minimum of climate data, it was already possible to model the effect of the forested areas on solar irradiance, soil moisture, and crop yields, in the transition zones, and the feedback loop that coupled soil moisture and crop physiology. However, there were further effects in the transition zones that we were not able to model, including wind turbulences, cardinal direction of wind shelter, rain shadow, or snow entrapment (Nuberg, 1998; Kort, 1988). As this shelter effect was reported to have a positive influence on the soil moisture content (Nuberg, 1998), future work should obtain data on, for example, the transitional gradients of wind speed. In addition, the yields in the transition zones might have been affected by allelopathy and nutrient deficiencies (Kort, 1988), which was not modeled as well. In a natural environment, these effects might have compensated the yield reductions through shading.

The transmittance of solar irradiance through the trees in the transition zone of forests, was approximated with 30% for pine (Gholz et al., 1991), in our approach. However, Voicu and Comeau

(2006) reported a 60% transmittance for aspen, close to the zero line and at least 80% after 1 H—an often used concept of tree heights—where 1 H equals the height of the element that shadows. Hence, the tree species composition should be considered when solar irradiance is modeled.

4.4.2. Magnitude of the shading effects on the yield

Considering the whole transition zone in all cardinal directions, the simulated yield reduction in transition zones was, approximately, 10% for wheat and 12% for maize. Simulated yields were lower, closer to the forest. In all cardinal directions, the yields were reduced by a reduced solar irradiance, compared to the areas with full irradiance. The yield reduction was highest in the north and lowest in the south. The simulated results were in line with measurements. Mitchell et al. (2014) measured very similar gradients for soybean, with respect to the distance to the zero line, as the ones simulated close to the forest with a decrease of ca. 30% to 50% within 100 m. The magnitude of the simulated yield reduction for wheat was similar to the 34% reported by Malik and Sharma (1990), and the 31% reported by Lyles et al. (1984). Dufour et al. (2013) reported a decrease of 50%, due to a reduced number of grains per spike and weight of grains. Nuberg (1998) measured similar, and even higher, yield reductions for wheat.

4.4.3. The spatial extent of the shading effects on the yield

Most of the yield reduction for the simulated wheat occurred within 15 m. The impact of shading on the maize yields occurred at least until 30 m from the zero line. The simulated solar irradiance reached maximum values at, approximately, 30 m. Correspondingly, the effects on soil moisture occurred until a distance of 30 m. Thus, our model predicted a correlation between the solar irradiance, soil moisture, and a reduced yield, to a distance of 15 m from the zero line, on average, and 30 m maximum, depending on the cardinal direction. These distances were in line with findings from other authors. They often use the concepts of tree heights H , which were as follows, in our approach: 15 m = 0.75 H and 30 m = 1.5 H . Voicu and Comeau (2006) reported a maximum of 1.5 H (aspen stand) for reduced solar irradiance, while Emmingham and Waring (1973) reported 1 H (Douglas-fir). Lyles et al. (1984) reported a decrease in the winter wheat yield, between 0.25 and 3 H . In a review by Kort (1988), values from 1 to 3.3 H were reported by several authors and for different crops. Bulir (1992, in: Nuberg, 1998) even reported a yield reduction up to a distance of 300 m (20 H), for wheat. Our results did not account for distances that went beyond 100 m.

The distances from the forest edge for reduced solar irradiance, soil moisture, and reduced yields depended on the cardinal direction. The lowest impact, with respect to distance, on solar irradiance and yield, was in the south of the forest and the shrub matrices, for the sites on the northern hemisphere of the globe. In the north, it was highest, while it was intermediate in the east and the west. This effect was also found by Matlack (1993), for soil moisture and solar

irradiance, by Voicu and Comeau (2006), as well as by Groot and Carlson (1996). With 30 m, the transition zone for our case study of modeled solar irradiance, was only one-third of the often referenced value of 100 m for the forested transition zones (Haddad et al., 2015; Riitters et al., 2002; Riutta et al., 2014). This suggests that this value is not universally applicable, especially, not for agricultural areas.

Accepting the approximation we made to simulate the shading effects of the forest edges on wheat and maize, our results showed a regional yield loss of 5% to 8%. These calculations helped to understand the importance of transition zones in agricultural landscapes. Although it is a simplified calculation, it combined the magnitude of yield loss, with the spatial extent for a whole region. However, these calculations did not account for the biological effects that are beneficial for plant growth, like pollination or pest control. The benefits and yield gains that were in relation to the distance to forests were likely to be much higher, compared to the loss through shading. Several aspects, like a higher biodiversity or the higher rate of ground water recharge of and through forests—just to mention some—were likely to have a higher importance, in terms of provision of the ecosystem services. Although the provision of food was one of the most important ecosystem services, a higher biodiversity in the transition zones, likely compensated the yield loss through shadows. With that in mind, management implications for farmers might be derived from the results, for the selection of crop varieties planted adjacent to the forests. Moreover, knowledge about the yield reduction might encourage farmers to reduce the management intensity adjacent to forests in favor of other ecosystem services rather than food provision.

4.5. Conclusion and outlook

We used a model to assess the yield reductions due to shading in the transition zones of agricultural landscapes, and used the simulation model to investigate the feedback loops that apply at the different distances from the zero line. With a minimum set of climate data, modeled solar irradiance, the crop growth model MONICA, and the basic functions of a geographical information system, the transitional gradients were modeled. In this paper, we presented and explained the procedure, analyzed the results, and compared them with the measured results from the literature. Moreover, we estimated the spatial extent of the transition zones in an agricultural landscape—Brandenburg in Germany—and calculated the yield loss on a regional level. We found the following, as stated below:

- Solar irradiance and yield have a strong correlation; with increasing distance to forest, solar irradiance and yield increase.
- The main influencing factors for the reduction of solar irradiance and the accompanying yield are tree height, distance to the forest, and cardinal direction.
- Crop varieties react differently, according to their physiological disposition.
- In dry years, the shading effects in the transition zones can be beneficial for the crop growth.
- On a regional level, a yield reduction of 5% to 8% can be considered to have been caused by shading in the transition zones.

Although our approach is satisfying to account for yield reductions due to shading in agricultural landscapes, it could also be further developed. The transmittance rates could be included as a logistic regression function of the distance to the forest, to make the results more precise. Moreover, functions for the transitional gradients of temperatures, wind speed, and soil moisture could be implemented to allow more precise simulations, on a larger scale. Further, a step from virtual to real landscapes would be to use light detection and ranging (LIDAR)-based tree maps, to simulate shadows in the agricultural landscapes. This would make the regional prediction of yield loss, more precise, especially, according to the cardinal directions. With respect to the ecosystem services approach, a comparison of trade-offs and benefits between the yield losses and, for example, biodiversity would help to understand the impact of the forest transition zones and to improve the management decisions.

5. Comprehensive discussion

5.1. Definitions for transition zones

In the review of existing literature it became evident that the current definitions for e.g. 'edges', 'ecocline', 'boundaries' or 'transition zones' are not consistent. This is a problem because it makes scholarly communication on transition zones difficult and inefficient. Historically, definitions of transition zones evolved from different fields including landscape ecology, ecosystem ecology and functional ecology (Section 2.2.; Yarrow and Marín, 2007). However, it is necessary to compare and contrast studies around the world and across different scientific disciplines to arrive at a comprehensive understanding of the processes of transition zones and their impact on ecosystem services. With the toolbox developed in Section 2 (Table 2.1), a definitional framework is proposed that helps set out a common language among scientists. It comprises synonyms used in the literature and links them to make it more comprehensive and comparable. In combination with the hierarchy patch dynamics paradigm (Wu and David, 2002; Wu and Loucks, 1995), and structural (Gregorio et al., 2005) and functional traits frameworks (Jax, 2005), a generic classification system for transition zones was proposed (Table 2.2). In comparison to others, this classification system links abiotic and biotic ecological features with quantifiable components. Due to these characteristics, it is highly suitable for quantitative studies and indicator-based assessments in ecology and related disciplines. To conduct mapping studies of transition zones that include a quantification of indicators a workflow was proposed (Figure 2.3) as well.

By combining schools of thought and approaches into a comprehensive framework older concepts in the field of quantitative ecology are explicitly not neglected. Hence, the definitional framework accelerates the knowledge gain on transition zones as a connector. However, the framework was intended to be applicable on a landscape scale and it is possible that it will be insufficient or ill-suited to other scales. However, for the studies that were performed (Section 3 and 4), the proposed framework was used and it was found to be highly suitable for measurements at the landscape scale. The definitions were also suitable for analyses with geographical information systems. In sum, the framework is flexible enough to adapt to different scenarios in the field but also allows a mathematical description of transitional gradients of different variables.

5.2. Spatial extent of transition zones across ecosystem boundaries

The spatial extent of transition zones across ecosystem boundaries is described utilising different methods in this thesis. First, articles were systematically reviewed according to a spatial extent and the variables observed. The values were assigned to either forested or non-forested transition zones. Second, field measurements of the spatial extent of the most common microclimatic variables that vary across transition zones were measured. Third, the impact on ecosystem

services of transition zones was modelled. This gives a comprehensive insight into the complexity of the topic, but also allows for a reduction of complexity with modelling where necessary.

As described in the introduction to this thesis and Section 2, most scientists that have quantified the spatial extent, the area or share of transition zones, used values between 90 and 100 m perpendicular to the zero line (e.g. Haddad et al., 2015; Riitters et al., 2002; Riutta et al., 2014). Most of these calculations have also been done for forested transition zones. This approach of using 100 m is reasonable and for some cases or variables it might hold true. It also gives an overall impression of the spatial extent of transition zones worldwide and hereby underlines their importance (Haddad et al., 2015). But for deeper insights into the processes of transition zones and their impact on ecosystem services a more specific evaluation according to observed and quantified changes is necessary. To calculate, for example, the area of forested transition zones where a reduction of carbon sequestration is possible, it is necessary to have a precise knowledge on the area and magnitude of transitional gradients where aboveground biomass is altered.

The review of 76 studies in Section 2 resulted in an average range of 25 to 50 m perpendicular to the zero line of an altered microclimate compared to the forest interior (Figure 2.4). This is less than half of the 100 m used by Haddad et al. (2015) – a highly cited study reporting a 20 % global share of transition zones in forests. For non-forested transition zones like cropland or pastures the average is most likely smaller than 25 m according to the review in this thesis (Figure 2.4). For temperature and moisture in soils an average range of only 10 to 20 m is reported in the literature. However, the number of studies reporting a length of transitional gradients is rather small and highly distributed across the world with most studies conducted in the Tropics. To provide required knowledge about the length of transitional gradients in other regions, measurements and simulations were conducted in temperate areas of Europe (Section 3 and 4, Schmidt et al., 2019a; Schmidt et al., 2019b).

To my knowledge of the author of this thesis and his co-authors little is known about microclimate, matter cycling and ecosystem services in transition zones in the temperate zone, at least in the forest and agriculture scientific communities. Measurements in Brandenburg, Germany (Section 3 and 8), resulted in an average length of transitional gradients from 50 to 85 m across the zero line for microclimate. In the forested transition zone, the length of transitional gradients for microclimate was approximately 35 m (see Section 3.3.1; Figure 3.2). This is in the range with the findings from the review (Section 2.3.1; Figure 2.4). However, it is only one third of the 100 m transition zone assumed for global or national-level studies. For aboveground biomass in the forest we found a length of 35 m (Section 3.3.2; Figure 3.4), which concurs with microclimate. A causal relationship could not be proved as not all microclimatic variables occurred as transitional gradients (Section 3.3.1; Figure 3.2). For example, solar radiation was not in S-shaped gradients as

expected although it is an important explanatory variable for biomass production as simulations show (Section 4.4). Microclimate and aboveground biomass in the transition zone of the arable land were altered within a range of 15 to 30 m, respectively. Similar to the forested transition zone, both variables are related to some extent, but it does not mean that there is causality.

For arable land, additional simulations were performed to gain insights into yield reduction in transition zones and possible mechanisms for this. Again, the yield reduction occurred within 15 to 30 m. Although observed variables (aboveground biomass vs. yield), the cardinal directions of the transition zones and crops were different in the simulations (Section 4) and the measurements (Section 3), the lengths of the transitional gradients are similar. In addition, modelled solar radiation also showed similar lengths of transitional gradients between 15 to 30 m. The reduction of solar irradiance in the simulations led directly to yield reductions in general. Here, causality is highly plausible as solar radiation is the most important energy source for plant growth.

Measured soil carbon and nitrogen contents were elevated at the zero line. Most often, soil is sampled in the interior of an ecosystem to avoid the edge effects. However, edges have to be considered in calculations regarding carbon and nitrogen contents in the soil on a landscape scale in the light of these findings. Due to the measurement design (Section 3.2. and 8) it is only possible to assume the length of this transitional gradient with a maximum of 50 m across the zero line, but not beyond 100 m. The observed values are supported by the literature (Section 2, Schmidt et al., 2017).

Especially in studies where the area of a transition zone is quantified, the observed variables that are referred to have to be reported to make results comparable. As described above, the length of transitional gradients differs for different variables or processes. Based on measurements in the temperate zone and accompanying simulations from this thesis, the length of forest-agriculture transition zones is likely to be smaller than 100 m (see Figure 5.2). However, the effects on various ecosystem services in transition zones might be more complex. Yields, for example, are influenced by biota besides abiotic variables, for example pollinators (Ricketts et al., 2008) and insect pests or predators (Tscharntke et al., 2005). Nguyen and Nansen (2018) described this as “edge-biased” distribution of insects as the edge is the main explanatory variable for the species composition. The altered microclimate in transition zones influences the abundance and diversity of fungi in the forest (Crockatt, 2012; Ruete et al., 2016). This could also be the case for fungi that affect agroecosystems, however little study of this side of transition zones has been performed. Another important influencing factor is the structure of the landscape: connectivity, composition and configuration (Seppelt et al., 2016) of fragmented agricultural landscapes also have an effect on ecosystem services. Hence, the transition zones should be explained and understood as a part of the hierarchical patch theory as was done in the theoretical framework in Section 2 (Wu and David,

2002; Wu and Loucks, 1995). Transition zones for different variables can be nested within another patch (Figure 5.1). For example, a forest patch with its forested and non-forested transition zones can be nested in a patch of cropland. These nested transition zones are nested in hierarchical patches themselves: the patches of cropland and the forest patch are part of a landscape and so forth.

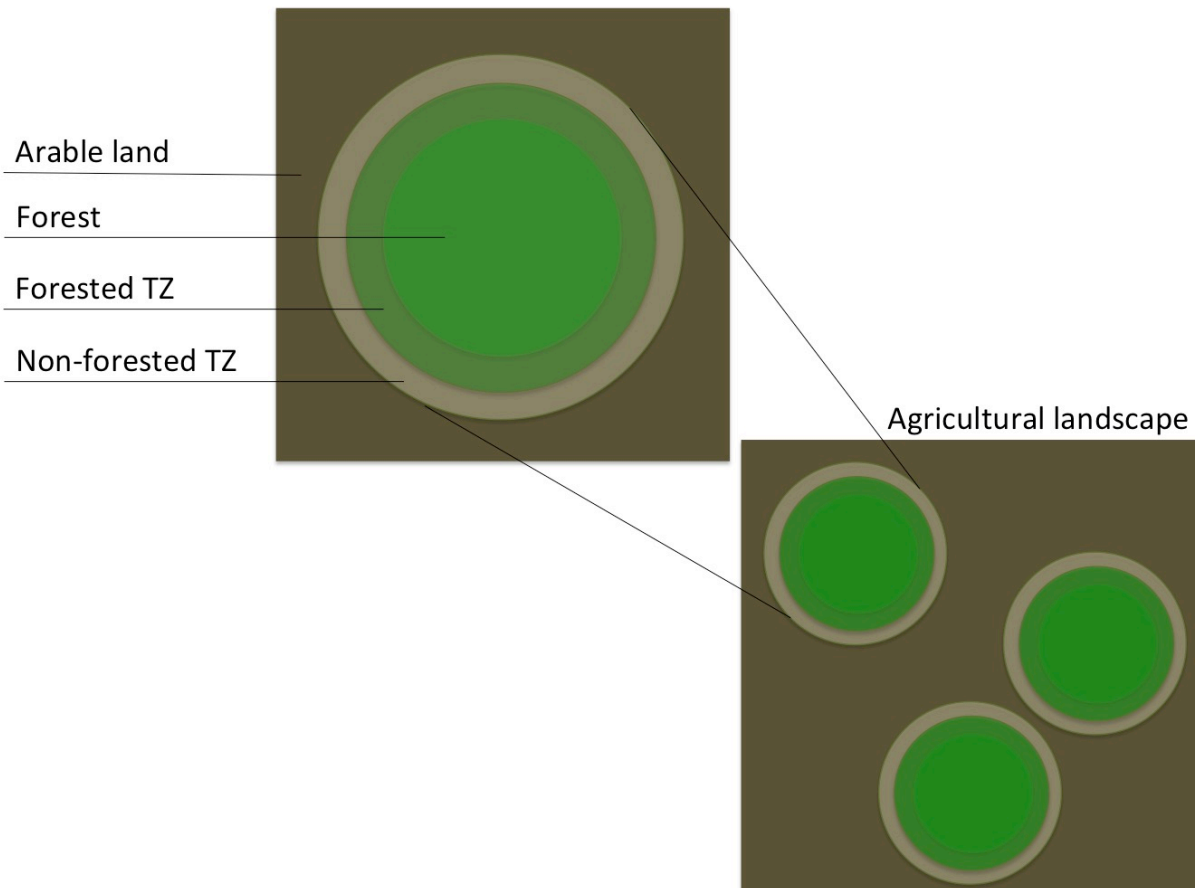


Figure 5.1 Schematic top view of an agricultural landscape according to the hierarchical patch dynamics model with nested transition zones.

5.3. Quantification of environmental gradients and ecosystem services

In addition to the spatial extent of transition zones, it is also desirable to have knowledge of the magnitude of transitional gradients and ecosystem services in transition zones. However, transition zones from forest to arable land develop and change over time (Chen et al., 1995; Young and Mitchell, 1994). With that in mind, our approach was to find patterns in the literature and add knowledge or prove results with continuously measured time series and modelling.

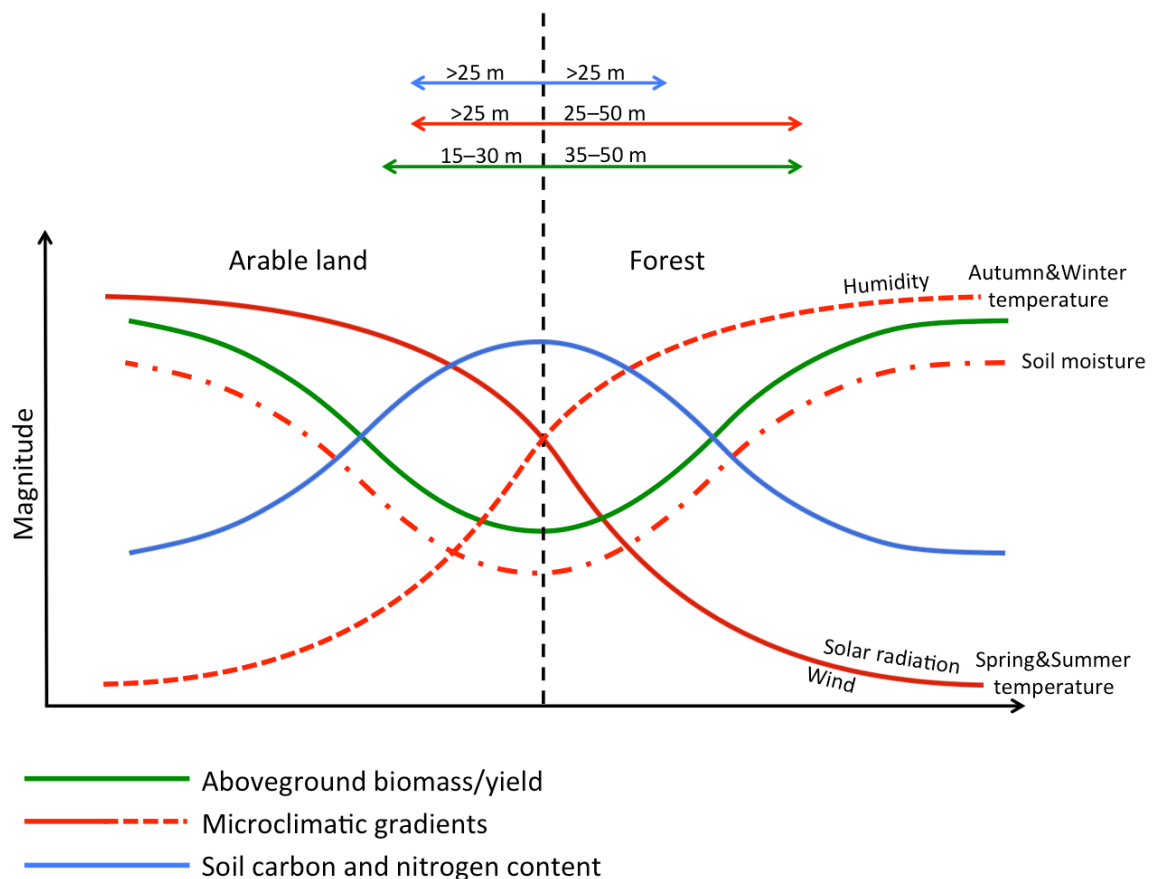


Figure 5.2 Cross-sectional scheme of transitional gradients of microclimate and ecosystem services in transition zones from arable land to forests according to measurements, simulations and a systematic review in this thesis. The arrows on top indicate a suggested and simplified spatial extent that has to be considered for these gradients.

Microclimatic gradients were found to be S-shaped in transition zones between arable land and forests. In Section 3.3., S-shaped gradients were shown for wind speed and direction, soil moisture and temperature as well as air temperature. The temperature for both soil and air, were higher in the arable land or the zero line in spring and summer compared to the forested transition zone. This was also reported by Ewers and Banks-Leite (2013), Heithecker and Halpern (2007) and Laurance et al. (2011). In autumn and winter, temperatures were higher in the forest. This is in contrast to the mentioned studies, but most likely due to the fact, that two of the studies report

results from the Tropics and the measurement period in the other study was only two to three weeks while the study in Section 3 comprises a whole year. However, the results here are in line with temporal changes in the course of the year reported by Young and Mitchell (1994). For the field measurements (Section 3), wind speed was both higher in the arable land but also in the forested transition zone compared to the forest interior. Other authors have reported a decrease in the wind speed of the non-forested matrix by one quarter towards the forest interior. In Section 3.3 a decrease of approximately 50% was observed, which is similar to that found by Chen et al. (1993). Humidity was lower in the arable land compared to the forest interior, which was also reported by Wicklein et al., (2012) and Williams-Linera et al. (1998). However, the opposite effect (Dodonov et al., 2013) or no change has also been found (Chen et al., 1995; Heithecker and Halpern, 2007; Mendonça et al., 2015). To derive and proof general patterns for these microclimatic gradients, measurements combined with modelling have to be done.

With the principal component analysis of time series described in Section 3.2.2, transitional gradients for solar irradiance could not be proved. However, a decline in solar irradiance from the arable land towards the zero line was simulated (ca. -33 to -66 %, Figure 4.3). Further towards the forest interior, Chen et al. (1995) report that solar irradiance was less than 15 % compared to the zero line. Young and Mitchell (1994) report a solar irradiance intensity of 5 % in the forest transition zone compared to the arable transition zone. Thus, transitional gradients of solar irradiance with a decrease of approximately 30 to 50 % from open space to the zero line and at least 70 to 90 % approaching the forest interior seem to be common.

Many authors report transitional gradients for soil moisture (e.g. Farmilo et al., 2013; Riutta et al., 2012, 2016) in transition zones. The findings in Section 4.3.2 agree with this using simulated data. However, it is uncertain whether the transition zone has a positive (higher soil moisture content) or negative (lower soil moisture content) effect. In the literature (Riutta et al., 2012, 2016), soil moisture has been reported to be lower for the forested transition zone. In the simulated results, soil moisture was slightly higher at the zero line compared to the arable land. One potential problem with the simulated soil moisture results is that the model does not account for soil water removal by the trees adjacent to the arable transition zone. Hence, the simulated results might be misleading. Soil moisture content can be predicted to decrease towards the zero line in both, forested and arable transition zones. As described above (Section 3.4.1), methods of soil moisture measurements in the literature are sometimes quite weak, with wide measurement intervals or insufficient repetition. This has to be considered when results are compared. Obviously, there are uncertainties in the patterns of soil moisture contents. Hence, more field measurements across ecosystem boundaries should be conducted and, again, combined with models.

Aboveground biomass and yields are reduced towards the zero line (Figure 3.6 and 4.5). Simulations of yield resulted in an average of 10 to 12 % reduction for the whole transition zone depending on the cardinal direction and distance to the zero line (Section 4.4.2). In field measurements, Mitchell et al. (2014) found a reduction crop yields by approximately 20 to 48 % within 100 m of the zero line. The aboveground biomass reductions in the measurements from Section 3.3.4 comparably span from one to two thirds. Lyles et al. (1984) also report a decrease in yield of 31%. As described in Section 4.4.2, other authors report similar (Malik and Sharma, 1990) or even higher values (Nuberg, 1998). However, the yield reduction depends on the crop and availability of water as showed in Section 4. The most critical portion of the transition zone for crop yields is the area that is shaded by trees that occurs from approximately 0 to 30 m from the zero line. Here, a yield reduction of up to 20 to 50 % can be expected with respect to the cardinal direction and crop variety according to the literature and simulated results.

The aboveground biomass of trees (with height as a proxy since no significant change in stem diameter was observed) declined by 18 % towards the zero line in the measurements conducted in this thesis (Section 3.3.2.; Figure 3.3). The same effect, no changes in stem diameter but higher trees with increasing distance was reported by Veselkin et al. (2017) for older forested transition zones in cities and by Cienciala et al. (2002) in a Swedish scots pine stand. However, an increase in stem diameter was reported by Ibanez et al. (2017). Thus, tree size and aboveground biomass appear to increase with increasing distance to the zero line. Veselkin et al. (2017) quantified this with a 25 % increase in stand density and timber volume. While the aboveground biomass of trees decreases towards the zero line, de Paula et al. (2011) calculated that the stored carbon is reduced by half compared to the forest interior. On the other hand, Hernandez-Santana et al. (2011) found higher trees and Remy et al. (2016) higher wood volume and stem density towards the zero line. Although Ziter et al. (2014) found a higher stem density close to the zero line, the aboveground carbon stocks were not affected. They suggest a too small fragment size for the experiments as an explanation. This has to be considered as smaller fragments tend to show no interior but can be assumed as a whole transition zone. Thus, the effects that occur in transition zone occur in the whole fragment. In our experiments, we made sure that the fragment size is larger than 10 ha and more than 500 m of forested area perpendicular to the zero line (edge) of the forest (Schmidt et al., 2018)

In the soil at the zero line, elevated carbon and nitrogen contents were found (Table 3.1). This was also found by Remy et al. (2016) for belowground carbon stocks and by Stanton et al. (2013) for soil total carbon stocks. On the other hand, Johnson and Wedin (1997) report a decrease of 17 % in soil organic matter compared to the forest interior. Weathers et al. (2001) found a higher throughfall of ammonium and nitrate in the transition zone. Remy et al. (2017a) as well as Wuyts et al. (2008) reported increased atmospheric N deposition in the transition zone. This possibly has an effect on

nitrogen stocks as they are reported to be higher (Remy et al., 2016; Stanton et al., 2013; Wuyts et al., 2011). The opposite was reported by Johnson and Wedin (1997) as well as Wicklein et al. (2012) who report a 66 % decrease of mineralised nitrogen in the transition zone or no effect on nitrate and ammonium concentration. The results of Johnson and Wedin (1997), however, have to be interpreted with caution as the experimental site was affected by grass invasions and annual fires. Thus, a majority of authors report an increase of carbon and nitrogen compounds stocks in the transition zone. This has to be considered in calculations on a landscape scale as most calculations are based on measurements where edge effects are avoided.

6. Conclusion and outlook

In this thesis, transitional gradients of microclimate, matter dynamics and ecosystem services were measured and modelled across ecosystem boundaries. In addition, a structured literature review was conducted that led to the development of a definitional and quantitative framework to describe transition zones. The broad approach in this thesis gives a comprehensive overview of transition zones and significantly advances the understanding of the size of transition zones and the mechanism that underlie them. On one hand, the detailed measurements presented here provide insights into environmental gradients in temperate climates for transition zones across ecosystem boundaries. Further, these measurements were used to inform a modelling approach that simulated transition zones and their effects on yield. On the other hand, the results were contextualised with the help of a systematic review. This is an important step towards generalized statements on the functioning, magnitude and spatial extent of transition zones. Additionally, all measured data was published in an openly accessible article (Schmidt et al., 2018) or as supplementary material to reproduce and for comparisons with other studies.

The diverse analyses and measurements presented here lead to the following conclusions:

- Transition zones have a spatial extent perpendicular to the zero line of 50 to 100 m across agroecosystem and forest ecosystem boundaries. The often used value of 100 m for forested transition zones is likely to be too high.
- Yield is negatively influenced in arable transition zones within 30 m of the forest-field edge by approximately 10 % on average but up to 50 %.
- Aboveground biomass in the forested transition zone is negatively influenced within at least 35 m by 20 to 25 % or more.
- Microclimatic gradients are S-shaped and have a spatial extent of approximately 75 m aboveground and half of that belowground across ecosystem boundaries.
- Solar irradiance decreases by 30 to 50 % from open space to the zero line and at least 70 to 90 % from the zero line into the forest.
- Microclimatic gradients have positive or negative transitional gradients from the arable land to the forest interior (Figure 5.2). While wind speed as well as temperatures in spring and summer decrease into the forest, humidity as well as temperatures in autumn and winter increase.
- Soil moisture content follows an inversed bell-shape and is most likely lowest at the zero line.

This thesis contributes to a better quantitative understanding of agricultural landscapes beyond a simple dichotomy of agricultural land and neighbouring ecosystems. It adds knowledge to the spatial extent of transition zones and offers a framework for description and modelling. Further, it is suggested that agricultural landscapes and their ecosystem services should be thought of as nested areas of different land uses and land covers where various transition zones overlap. As described above, the area of transition zones worldwide is significant, and due to anthropogenic fragmentation it is likely to increase. Moreover, due to climate change the effects of transition zones and their feedback on climate might become even more relevant (Ewers and Banks-Leite, 2013). Baltzer et al. (2014), for example, argue that global warming causes more thawing in boreal areas, which accelerates fragmentation and forest loss due to waterlogging. This releases additional greenhouse gases which accelerates the process in a feedback loop (Koven, 2011).

The anthropogenic influence on transition zones and critical loads of pollutants should gain more scientific attention. As described, nitrogen contents are elevated in transition zones. Most likely, it holds true for other substances like pesticides as well. That changes the diversity of communities and the abundance of species. In that context, more light should be shed on microbial activity in transition zones. As the availability of nitrogen and microclimate are altered in transition zones, so is the correlating metabolism of microorganisms and other fauna (Knief, 2015; Kolb, 2009; Riutta et al., 2012). The importance of soil microorganisms for nitrogen cycling is accounted for by their role as sources (nitrous oxide; N_2O) and sinks for (e.g. methane, CH_4) of green house gases. This, again, is related to climate change.

Another issue that was mostly left out in this thesis is biodiversity in transition zones. Although there is a long history of many articles on that (Ries et al., 2004), the debate was sparked again as Fahrig (2017) argued in a review, fragmentation has a positive effect on biodiversity. This “per se”-dictum was too onesided for some of the most influential researchers in that field (Fletcher et al., 2018). They call for mechanistic modelling for a better understanding of the processes in transition zones. Hence, the scientific contributions in this thesis are a valuable puzzle piece in a recent scientific debate as the non-living environment always affects the living and vice versa.

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8. Accompanying Data Article – Data on and methodology for measurements of microclimate and matter dynamics in transition zones between forest and adjacent arable land⁷

Abstract

Ecosystems are often defined by visually perceived boundaries, while for many properties sharp boundaries are difficult to draw. Boundaries between terrestrial ecosystems have often been described with much emphasis on edge effects, which is the impact of the presence of one ecosystem on an adjacent ecosystem. At the boundary of forested and agricultural ecosystems, measurements of environmental variables exist almost only for the forested area, describing the extent of a transition zone and the rates of exchange of matter, energy and information from the zero line (edge) into the forest. The opposite direction has been nearly neglected so far. Microclimatic variables differ in magnitude in the transition zone between arable land and forest. They affect habitat properties, biotic activity, carbon and nitrogen stocks, as well as turn-over rates under the different input of organic matter.

We conducted microclimatic measurements in two 105 m long transects perpendicular to the boundaries in transition zones of forests to arable land for more than one year. In addition, we measured aboveground biomass, litterfall, soil carbon and nitrogen content. In this paper, we explain the measurement design and methodology as well as make the data openly accessible.

Keywords: Edge effects, Ecological boundaries, Matter cycling, Fragmentation, Ecosystem services, Carbon, Nitrogen

8.1. Context

In ecosystem ecology, the focus is most often on single ecotopes like forest ecosystems or agro-ecosystems. The edge effects are rarely accounted for. However, ecosystems are open and complex systems. They exchange matter, energy and information at their boundaries. The transition zones usually have steep environmental gradients and often have proved to be hotspots for biodiversity (Ries et al. 2004). Especially for microclimate, carbon and nitrogen stocks, numerous have been found having significant effects on biogeochemical and biophysical processes (Schmidt et al. 2017).

⁷ This appendix is an open access data publication: Schmidt, M., Lischeid, G., Nendel, C., 2018. Data on and methodology for measurements of microclimate and matter dynamics in transition zones between forest and adjacent arable land. *One Ecosystem* 3: e24295. <https://doi.org/10.3897/oneeco.3.e24295>
All supplementary materials in this article are available online.

Following the logic of a single ecosystem, most experiments and samples were conducted in only one ecosystem. We conducted measurements in transects across boundaries of forest to arable land to explicitly overcome these constraints as well as boundaries of scientific fields. These measurements were conducted for more than one year comprising two growing seasons and one season of leaf fall. The purpose of these measurements was to gain insight into magnitude and extent of environmental gradients in the transition zone as a base for subsequent modelling studies.

This data paper is meant to a) explain the measurement design and methodology in detail, b) make all measured data openly accessible for re-use and c) make the editing of raw data transparent.

8.2. Methods

8.2.1. Measurement site

The measurement sites are located in north-east Germany in the federal state of Brandenburg. Two sites were selected according to 1) the homogeneity of tree species composition within the forest stands, 2) the size of the ecosystem patches (fragment), 3) the cardinal direction of their zero line (edge), 4) the age and management of the forest, 5) the homogeneity of the surrounding landscape, 6) the management of the agricultural land, 7) the distance to Müncheberg, Germany and 8) the willingness of farmers, forest managers and land owners to cooperate. According to this framework, the following specific criteria were set and evaluated using geographic information systems (GIS) to prepare the selection:

- continental, temperate forest with coniferous trees (comprise ca. 75% in the region) resulting in pine (*Pinus sylvestris* L.) and larch (*Larix decidua*) (it was not possible to find sites with the same main species composition that fit all other criteria at the same time),
- >10 ha total area of the forest patch and more than 500 m of forested area perpendicular to the zero line (edge) of the forest,
- one east-facing and one west-facing edge to avoid too strong effects of the cardinal directions north and south,
- managed, planted forest with an age of trees between 40 and 60 years,
- a minimum of additional wind-breaking landscape elements and flat terrain within 500 m of the measurement sites,
- agricultural land cropped for more than 10 years prior to the study,
- less than 50 km distance to Müncheberg to remain operable,
- approval to conduct all measurements from June 2016 to September 2017.

We identified an east-facing (the arable land is east of the forest) site in Ihlow, Germany (52°37'23.8"N, 14°03'40.4"E) and a west-facing site in Elisenhof, Germany (52°29'37.1"N, 14°11'03.4"E). Both sites are located in the cold temperate area with summers colder than 22°C on average and 4 months where the temperature is above 10°C (Dfb; Peel et al. 2007). See Figure 8.3 for an exemplary picture of a forest edge. Besides the main tree species composition, the edge was characterised by grassland as well as scattered shrubs (bird cherry (*Prunus padus*) at the east facing site) and deciduous trees (birch (*Betula pendula*) at the west-facing site). The zero line is the abrupt change in ploughed arable land to the unmanaged forest soil.

Cross-sectional scheme

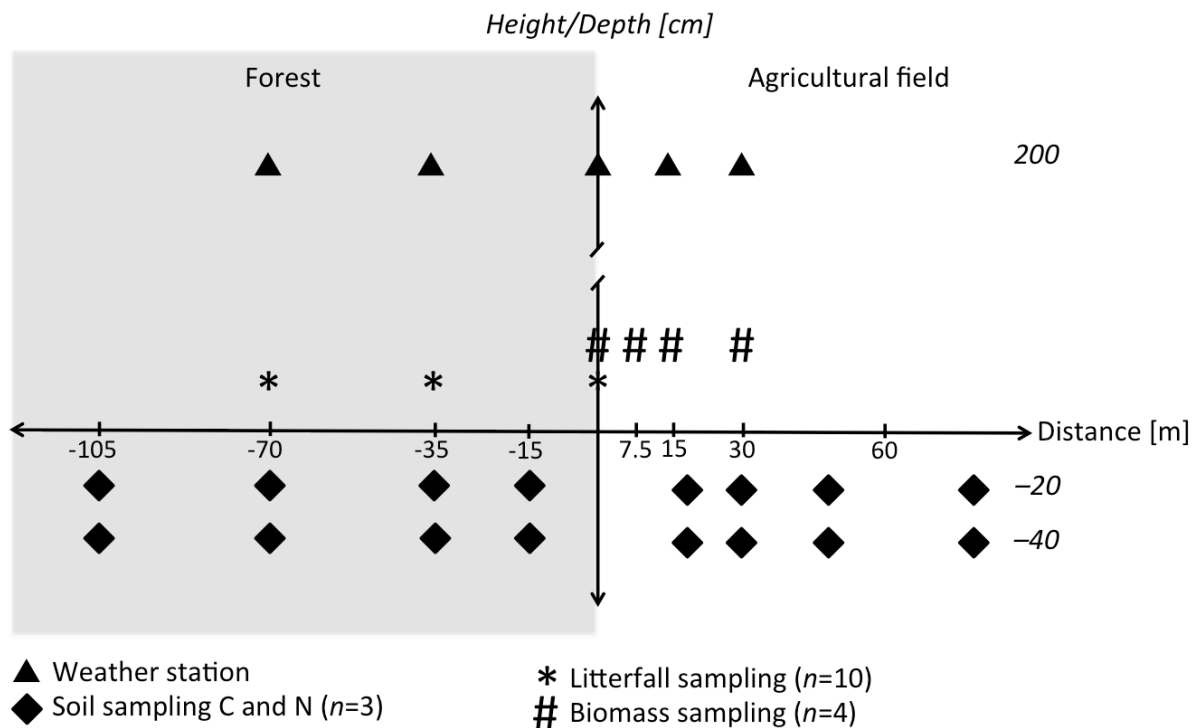


Figure 8.1 Cross-sectional scheme of the measurement design for both sites. The height for litterfall and biomass sampling is not scaled.

Topview

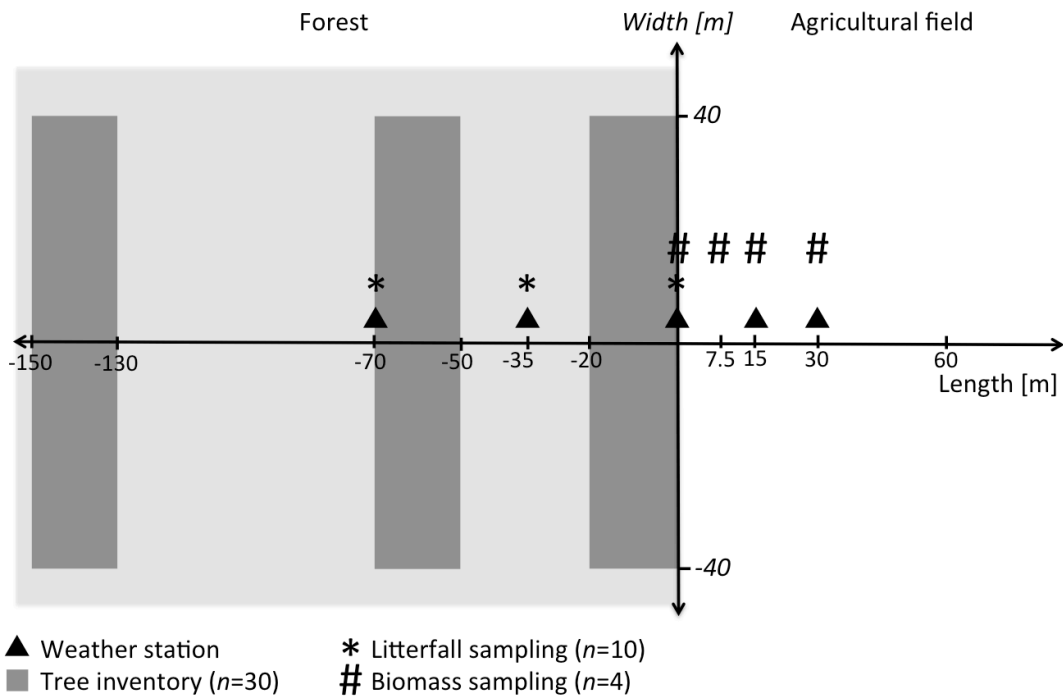


Figure 8.2 Topview on to the measurement design for east-facing site. The west-facing site is mirrored. The width is only scaled to the forest inventory plots. The symbols for weather stations, litterfall and biomass sampling are offset.



Figure 8.3 Exemplary picture of the forest edge at the west-facing site. View towards north.

8.2.2. Microclimate

The microclimate was measured in 105 m long transects with five weather stations – one at the zero line (edge), two in the arable land (see Figure 8.4 for the setting) and two in the forest (Figure 8.1, 8.2). The measured variables were air and soil temperature as well as moisture, wind speed and direction, air pressure, precipitation and solar radiation. The microclimatic data are given separately for the west-facing site (Suppl. material 1) and the east-facing site (Suppl. material 2).



Figure 8.4 Setting of weather stations in the arable land at the east-facing site. View towards south. Picture taken in autumn.

The measurements started on 15 June 2016 and ended on 17 July 2017. The distances were chosen based on a previous literature analysis (Schmidt et al. 2017).

The following sensors were used (see Table 8.1 for specifications):

Table 8.1 Information on the device name, accuracy, resolution and range of the sensors of the weather stations as well as the measurement height or depth and direction.

	Device	Accuracy	Range	Resolution	Direction of measurement	Height or depth of measurements
Vapour pressure	Decagon Devices VP-4	max. error: ± 0.5 kPa	0 to 47 kPa	0.001 kPa	N	ca. 2 m
Humidity	Decagon Devices VP-4	max. error: $\pm 5\%$	0 – 100% RH	0.1% RH	N	ca. 2 m
Temperature	Decagon Devices VP-4	max. error: $\pm 3^\circ\text{C}$	-40°C to $+80^\circ\text{C}$	0.1 $^\circ\text{C}$	N	ca. 2 m
Atmospheric pressure	Decagon Devices VP-4	0.4 kPa	49 to 109 kPa	0.01 kPa	N	ca. 2 m
Soil moisture (Volumetric Water Content (VWC))	Decagon Devices 5TM	± 0.03 m ³ /m ³	ϵ_a : 1 (air) to 80 (water)	0.0008 m ³ m ⁻³ from 0 to 50% VWC	-	ca. -20 cm
Soil temperature	Decagon Devices 5TM	$\pm 1^\circ\text{C}$	0.1 $^\circ\text{C}$	0.1 $^\circ\text{C}$	-	ca. -20 cm
Wind speed	Decagon Devices DS-2 Sonic Anemometer	0.30 m/s	0 to 30 m/s	0.01 m/s	-	ca. 2 m
Wind direction	Decagon Devices DS-2 Sonic Anemometer	$\pm 3^\circ$	0 $^\circ$ to 359 $^\circ$	1 $^\circ$	-	ca. 2 m
Precipitation forest	UMS KIPP100 (area: 2 m, Fig. 8.6)	1% at 1 l h ⁻¹	max. 5 l m ⁻¹	0.1 l	-	ca. 1 m
Precipitation arable land	Delta OHM HD2015 (area: 0,2 m, Fig. 8.4)	max. error 0.2 mm/tip		0.1 – 0.2 mm/tip	-	ca. 1 m, >50 cm above crops
Solar radiation	Decagon Devices PYR Solar Radiation Sensor	$\pm 5\%$	380 – 1120 nm 0 – 1750 W m ⁻²		S	ca. 2 m

8.2.3. Aboveground biomass

Biomass of crops

In 2016, at four plots of 1 m² each and at 0 m, 7.5 m, 15 m, 30 m distance from the zero line, the aboveground biomass was harvested manually (Figures 8.1, 8.2, 8.5) approximately one week before the harvest by the farmers (12 July). In 2016, at the east-facing site, peas were grown and at west-facing site oil-seed rape. In 2017, another biomass harvest was conducted with the same design. On 3 July, approximately 10 days before the official date, winter barley was harvested at the east-facing site and winter wheat at the west-facing site. After chopping the plants, they were oven-dried for 48 h at 65 $^\circ\text{C}$ and the dry mass was weighed thereafter (Suppl. material 3).



Figure 8.5 Delayed flowering of rape in the transition zone to the forest at the west-facing site. View towards south. Picture taken in summer.



Figure 8.6 Setting of weather stations in the forest (zero line, -35 m and -70 m) with rain gauges.



Figure 8.7 Setting of litter fall traps at the zero line (edge).

Stem diameter and height of trees

At the end of April 2016 within the east-facing site and at the beginning of May within the west-facing site, we measured the stem perimeter at 1.3 m height above the ground and the height of the trees (Suppl. material 4; hypsometer: Haglöf Vertex IV). Per site, we defined three plots with a width of 80 m parallel to the zero line and a depth (length) of 20 m perpendicular to the zero line. The plots were 0-20 m, 50-70 m and 130-150 m away from the zero line (edge). As tree density was different at both sites, we measured every fourth tree in a range of 5 m within the east-facing (A, Figure 8.8) and every third tree in a range of 7 m within the west-facing site (B, Figure 8.8) resulting in $n=30$ per plot.

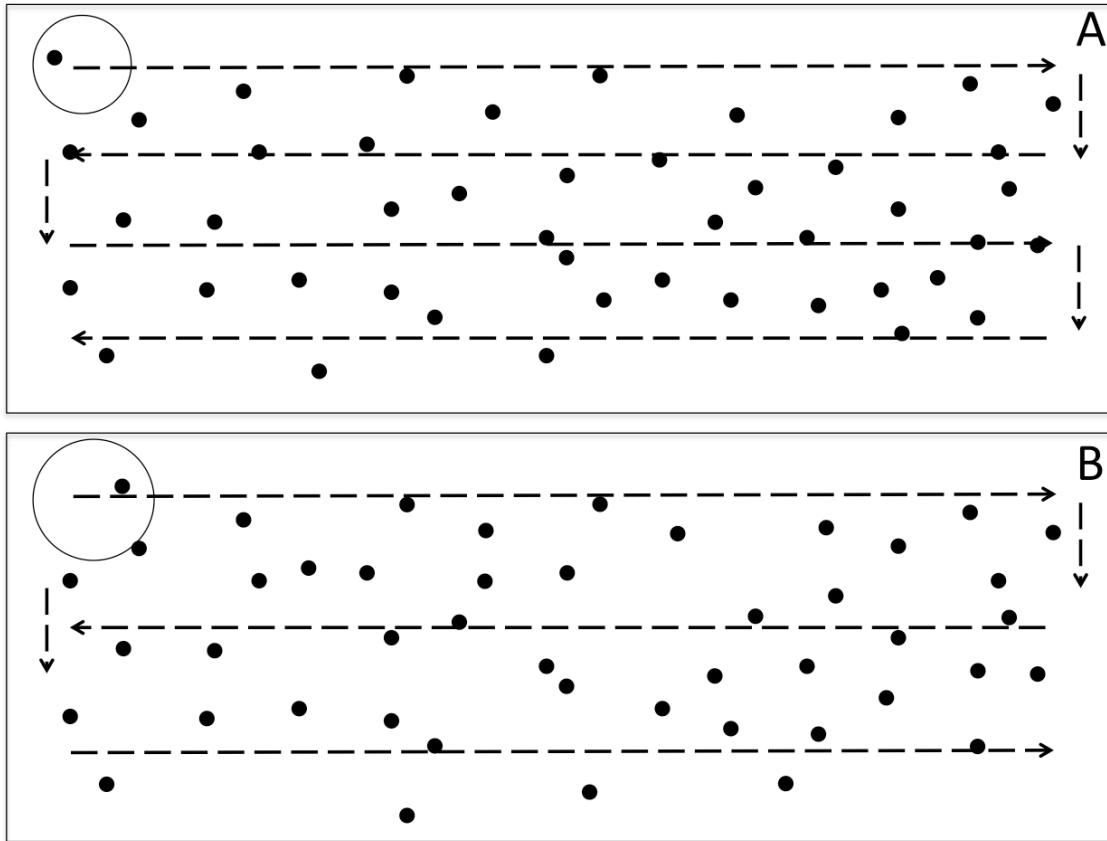


Figure 8.8 Design for tree inventory with the dots as exemplary trees, the arrows as walking direction during measurements and the circles as the range of counting the trees with 5 m range in A and 7 m range in B.

Litterfall

Litterfall was measured with buckets (see Figure 8.7) of 52 cm inner diameter resulting in an area of ca. 0.21 m² per bucket trap fall. Per distance (0 m, 35 m and 70 m; Figures 8.1, 8.2), $n=10$ litter fall traps were used. Sampling was conducted from September 2016 to May 2017 at five dates (13th Sept., 10th Oct., 7th Nov., 10th Jan., 22nd May). After each sampling, the needles of larch (*Larix decidua*) at the west-facing site and pine (*Pinus sylvestris* L.) at the east-facing site were oven-dried for 48 h at 65°C and, subsequently, dry mass was determined by weighing (Suppl. material 5). The samples also comprised small amounts of other biomass.

8.2.4. Soil sampling and analyses

The soil was sampled at two depths – 20 ± 3 cm and 40 ± 3 cm – along the transects (Figures 8.1, 8.2). The samples were sieved at 2 mm, air dried and analysed ($n=3$) for total soil carbon content and total soil nitrogen content (see Suppl. material 6). All samples were carbonate-free as tested in the field using hypochloric acid (see also pH in Suppl. material 6). Soil types were determined in the field according to WRB (FAO 2014; Table 8.2) using a soil drill (1 m length).

Table 8.2 Soil classification according to WRB (FAO, 2014) with soil textural classes for two different depth (20±3 cm and 40±3 cm) in a transect from forest (negative distances) to arable land (positive distances) and its zero line (edge; 0) at an east-facing site (E) and a west-facing site (W) in north-east Germany.

Site	Distance to edge [m]	Soil	Soil type in 20 cm	Soil type in 40 cm
W	60	Cambisol arenic aric	Loamy sand	Loamy sand
W	30	Protostagnic Cambisol loamic aric ⁸	Sandy loam	Sandy loam
W	15	Cambisol arenic aric	Loamy sand	Loamy sand
W	0	Protostagnic Cambisol arenic humic	Sandy loam	Sandy loam
W	-35	Cambisol arenic humic	Loamy sand	Sandy loam
W	-70	Protostagnic Cambisol loamic humic	Loamy sand	Sandy loam
W	-105	Cambisol arenic humic	Loamy sand	Loamy sand
E	60	Protostagnic Cambisol loamic aric	Loamy sand	Loam
E	30	Cambisol loamic aric humic	Loamy sand	Loamy sand
E	15	Cambisol arenic aric humic	Loamy sand	Sandy loam
E	0	Cambisol arenic humic	Loamy sand	Loamy sand
E	-35	Cambisol arenic humic	Loamy sand	Loamy sand
E	-70	Cambisol arenic humic	Loamy sand	Loamy sand
E	-105	Protostagnic Cambisol loamic humic	Sandy loam	Loam

8.2.5. Data editing

The microclimatic data from all ten weather stations were merged (see Suppl. material 7 for the R script). Moreover, the original header contained non-unicode symbols which had to be transformed. Data had to be rearranged as two weather stations were mixed up after ploughing and had to be replaced to their original positions. Further, data was cut according to the official starting and end points of the experiment and pauses due to management activities of the farmers (west-facing: 12/07/2016 till 25/10/2016; east-facing: 13/07/2016 till 19/08/2016 and 13/09/2016 till 24/10/2016).

We deleted two outliers for precipitation (error in measurements shortly after installation and the measurement date was wrong) and one for relative humidity (relative humidity of 4.8 not possible) from the whole data set.

The data logger (EM-50G) used the time zone of the plugged device. Due to that and the switch of standard time during the year, we had to unify the time to Central European Time (CET) manually.

⁸ This Cambisol was deeply eroded which cut off the cambic horizon and mixed it due to ploughing.