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Microclimate and matter dynamics in transition zones of forest to arable land

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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 50–80 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.

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 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 Bebber, 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

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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:

1 The width of the transition zone from arable land to forest depends on the measured variable.
2 The abiotic environmental gradients are non-linear across ecosystem boundaries.
3 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).

2. Methods

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 $n = 26,657$ and $n = 32,014$), 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 (Fig. 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. In our east- and west-facing study design, we wanted to detect environmental gradients for these opposing cardinal directions rather than compare extremes like in north and south direction.

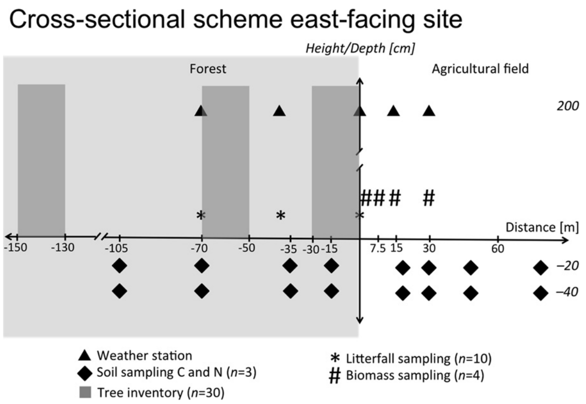

Fig. 1. Cross-sectional scheme of the measurement design on the example of the east-facing site. The heights and distance are not scaled. Adapted from Schmidt et al. (2018).

Aboveground biomass ($n = 4$) of oilseed rape, wheat, pea and barley was measured at four 1 m^2 plots at different distances (0–1 m, 7.5 m, 15 m, 30 m) from the zero line on the arable land. The aboveground parts of the plants were harvested, oven-dried and weighed. In the forest, the diameter at breast height (DBH) and the height of trees as proxy for aboveground biomass of pine and larch were measured at three plots ($n = 30$ trees per plot; 0 to -20 m, -50 to -70 m and -130 to -150 m from the zero line with a width of 80 m). Litterfall was measured at 0 m, -35 m and -70 m in the forest. At each distance, ten litterfall traps were arranged parallel to the zero line with a distance of 1 m towards each other to account for the forest heterogeneity. Soil was sampled at two depths (approx. 20 cm and 40 cm) at the transects (60 m, 30 m, 15 m, 0 m, -35 m, -70 m, -105 m) and analysed for total nitrogen and carbon content ($n = 3$).

2.2. Data analysis

The goal of the analysis of the time series of meteorological and soil parameters was to identify effects that could be ascribed to the position along the transect and separate them from other effects, like e.g. measurement imprecisions. To do this, each set of five time series of the same variable measured at different positions along the single transects underwent a principal 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

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

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 A).

We carried out a Bonferroni-adjusted post-hoc analysis to compare the data on trees (Fig. 4), litterfall (Fig. 5), soil (Table 1, Appendix B) and above-ground biomass (Fig. 6) 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 C).

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

3. Results

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 (Fig. 2A). 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 (Fig. 2B; 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 (Fig. 2C). These environmental gradients were asymmetric and S-shaped. The transition zone according to 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 (Fig. 2D).

The main wind direction for this region is southwest (SW; Fig. 3). 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–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–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.

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 (Fig. 4) with an average height of

Table 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		
Soil nitrogen content	Forest		−105 m	−70 m	−35 m	0 m	15 m	30 m	60 m
		−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			

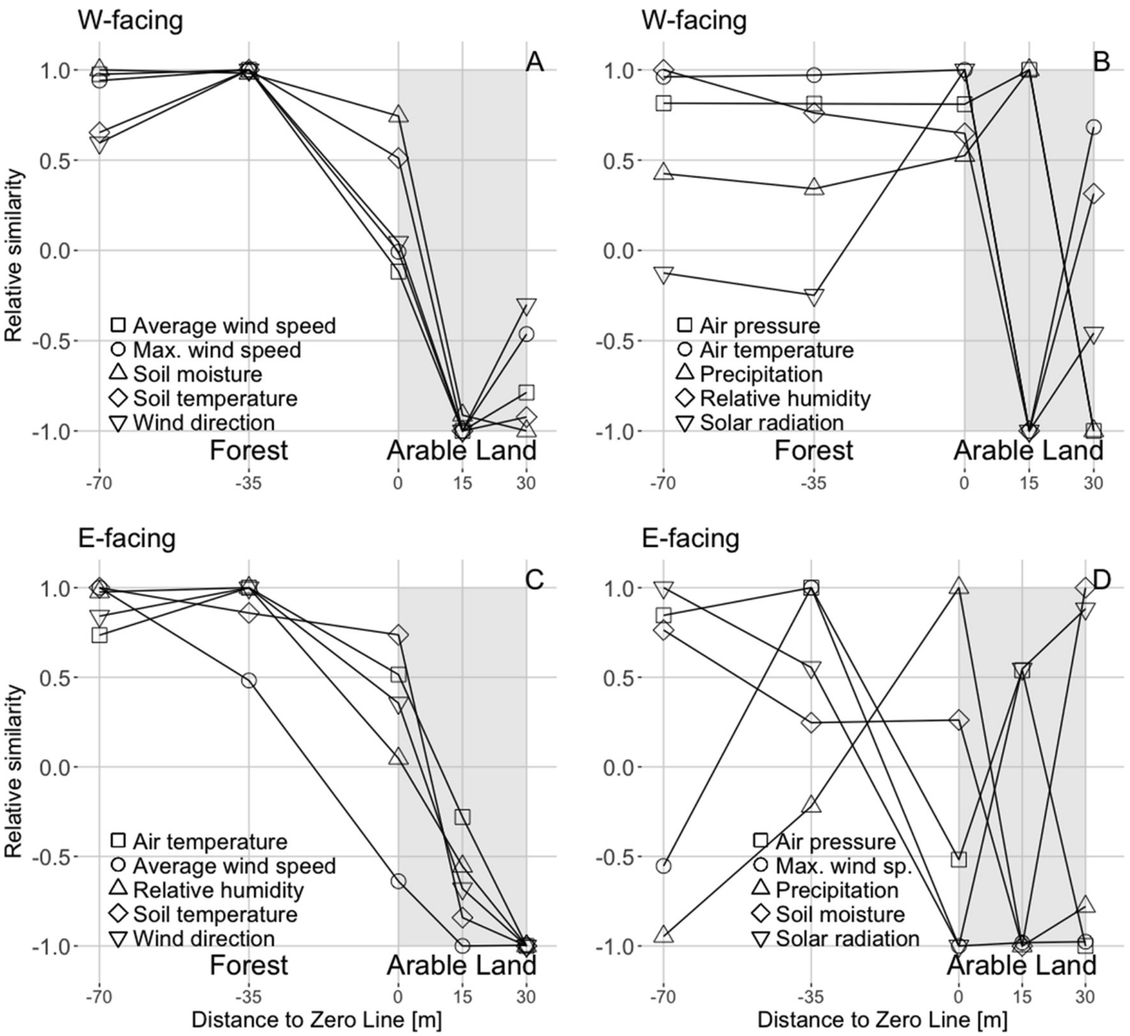

Fig. 2. Shape of the transition zone between forest and arable lands for time series of various variables. Left: Variables that exhibit an approximately monotonic transition. Right: Variables that do not exhibit a clear transition. The points were connected using locally weighted scatterplot smoothing (loess).

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).

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 (Fig. 5). 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.

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 ($p = 1.0{\cdot}10^{-5}$, $p = .4.4{\cdot}10^{-4}$).

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 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

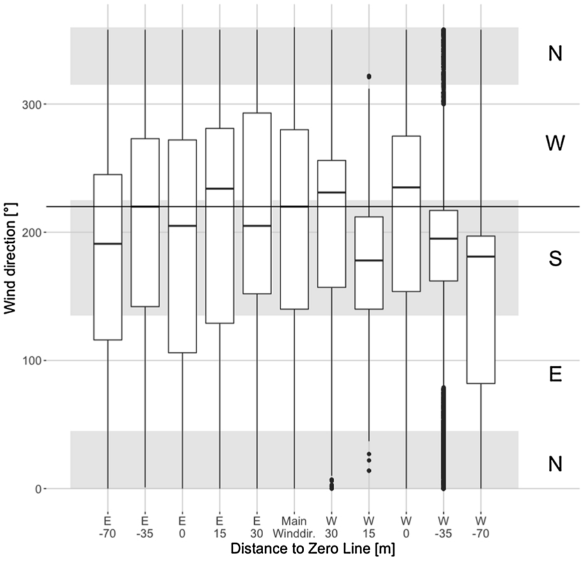

Fig. 3. Quartiles of the wind directions of a west-facing site (W, right boxplots), an east-facing site (E, left boxplots) and the main wind direction for Müncheberg (coordinates: 52°30'57.8"N 14°07'20.9"E; the official weather station between both sites, with its median depicted as a horizontal line). 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).

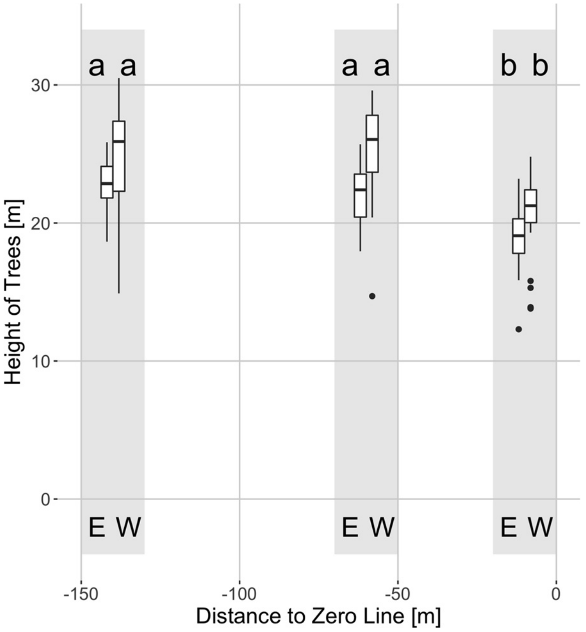

Fig. 4. Boxplots of the heights of trees of the east-facing site (E, left boxplots) and of the west-facing site (W, right boxplots) in three plots (light grey; 0–20 m, 50–70 m and 130-150 m). $n = 30$ trees were measured per plot. The boxes denote 25–75% of the values with the median (bold) in it. The 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.

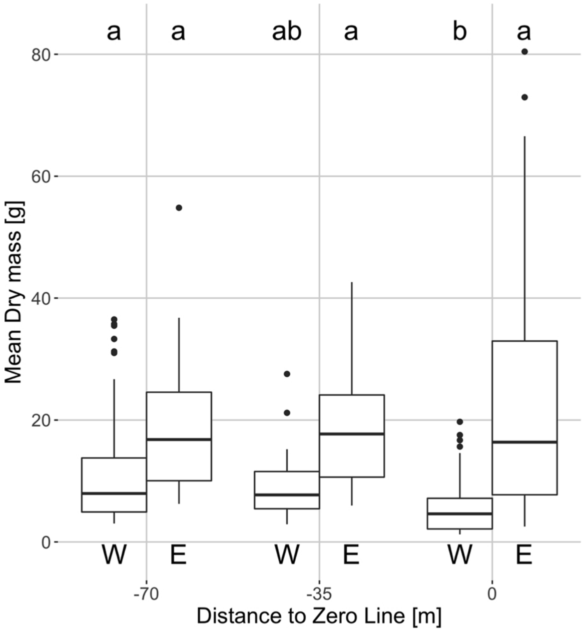

Fig. 5. Boxplots of litterfall of a west-facing site (W; left boxplots; *Larix decidua*) and an east-facing site (E, right boxplots; *Pinus sylvestris* L.) at three sample points (0 m, 35 m and 70 m from the transect). $n = 10$ litterfall traps were used at each sample point. Sampling was conducted on 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.

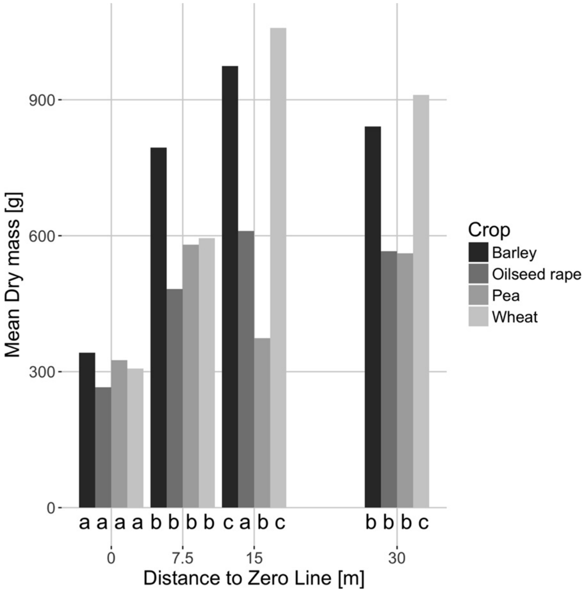

Fig. 6. Mean dry matter aboveground biomass for two growing seasons in 2016 (pea, *Pisum sativum* L., and oilseed rape, *Brassica napus* L.) and 2017 (barley, *Hordeum vulgare* L., and wheat, *Triticum aestivum* L.). Each bar represents the mean dry biomass of one-square-metre plots ($n = 4$) of aboveground 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.

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 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.

4. Discussion

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 Fig. 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 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 (Fig. 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).

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 (Fig. 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 (Fig. 5 in 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.

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, b). In terms of carbon, a strip of approximately two to three meters with a grassland character directly at the edge (see a photo 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.

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.

Declarations of interest

None.

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Appendix A. Variance of the first and second principal component for Z-transformed variables at a west-facing and an east-facing site

Site	Measured variable	First principal component	Second principal component
West-facing	Air pressure	0.97	0.02
	Air temperature	0.99	0.01
	Precipitation	0.81	0.13
	Relative humidity	0.95	0.03
	Soil moisture	0.71	0.2
	Soil temperature	0.97	0.03
	Solar radiation	0.84	0.1
	Average wind speed	0.85	0.09
	Wind direction	0.63	0.19
	Maximum wind speed	0.86	0.07
East-facing	Air pressure	1	–
	Air temperature	1	–
	Precipitation	0.86	0.09
	Relative humidity	0.98	0.01
	Soil moisture	0.76	0.13
	Soil temperature	0.99	0.01
	Solar radiation	0.86	0.09
	Average wind speed	0.65	0.19
	Wind direction	0.63	0.2
	Maximum wind speed	0.7	0.15

Appendix B. Levels of total soil carbon (C) and total soil nitrogen (N) depending on the distance to the zero line of a forest (negative values) to arable land (positive values) at a west-facing (W) and an east-facing (E) site. Soil was sampled at two depths (± 3 cm)

Distance to zero line	Site	Depth of sampling	Mean C_t, % (n = 3)	Mean N_t, % (n = 3)	C:N
60	E	−20	0.39	0.05	8.57
60	W	−20	0.51	0.06	9.31
60	E	−40	0.13	0.03	4.17
60	W	−40	0.24	0.03	7.50
30	E	−20	0.50	0.06	8.43
30	W	−20	0.62	0.07	9.23
30	E	−40	0.28	0.05	6.12
30	W	−40	0.28	0.03	9.10
15	E	−20	0.43	0.06	6.66
15	W	−20	0.65	0.07	8.99
15	E	−40	0.12	0.02	5.42
15	W	−40	0.19	0.03	5.88
0	E	−20	1.56	0.13	12.28
0	W	−20	1.67	0.13	13.04
0	E	−40	0.51	0.07	7.49
0	W	−40	0.47	0.06	7.88
−35	E	−20	1.02	0.06	16.07
−35	W	−20	0.65	0.06	11.15
−35	E	−40	0.41	0.04	9.89
−35	W	−40	0.28	0.03	10.05
−70	E	−20	0.84	0.06	14.33
−70	W	−20	1.38	0.09	15.93
−70	E	−40	0.35	0.04	9.96
−70	W	−40	0.26	0.03	9.33
−105	E	−20	0.65	0.07	9.44
−105	W	−20	0.73	0.05	13.35
−105	E	−40	0.36	0.03	11.44
−105	W	−40	0.20	0.04	5.13

Appendix C. Results of Kruskal-Wallis one-way analysis of variance for different measured variables with respect to their spatial distribution in the transition zone. The right column is the *p*-value for the Kruskal-Wallis test

Variables	Kruskal-Wallis test (p)
Diameter at breast height at east-facing site (n = 30)	0.0658
Diameter at breast height at west-facing site (n = 30)	0.2887
Height at east-facing site (n = 30)	$3.5\cdot10^{-08}$
Height at west-facing site (n = 30)	$2.52\cdot10^{-07}$
Litterfall at east-facing site (n = 150)	0.9351
Litterfall at west-facing site (n = 150)	0.0002
Barley (n = 4)	0.0048
Pea (n = 4)	0.0122
Oilseed rape (n = 4)	0.0097
Wheat (n = 4)	0.0037
Soil total C at 20 ± 3 cm (n = 3)	0.0001
Soil total C at 40 ± 3 cm (n = 3)	0.0002
Soil total N at 20 ± 3 cm (n = 3)	0.0036
Soil total N at 40 ± 3 cm (n = 3)	0.0173

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