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UNIVERSITY OF
HOHENHEIM

M A S T E R   T H E S I S

as part of the Master studies

Environmental Sciences
– Soil, Water and Biodiversity

Development of the vegetation and changes in flora in landscape elements of an agricultural landscape section in Brandenburg since 1992

submitted by:

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April 1st 2020

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I hereby declare that I am the sole author of this work. No assistance other than that which is permitted has been used. Ideas and quotes taken directly or indirectly from other sources are identified as such. This written work has not yet been submitted in any part.

Berlin, April 1st 2020

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Acknowledgement

I would like to take this opportunity to thank Harald Meimberg and Klaus Schmieder, who supported me in writing my thesis in Brandenburg, far from my universities in Vienna and Hohenheim.

My special thanks goes to my on-site supervisor Jörg Hoffmann, who made my work at the Julius-Kühn Institute possible. He contributed with his extensive experience, took the time for mutual exchange and was always available for my questions. I am grateful to him for his constructive criticism and his support.

I would like to thank my colleagues Jörg Sellmann and Tim Wahrenberg, who helped to make data available to me. Many thanks also to Holger Pfeffer, without whose previous work this master's thesis would not have been possible.

Last but not least, I would like to thank Arjun Castelino, who never tires of improving my English and who proofread this work at the end.

Abstract

This master's thesis is part of the project *BioZeit.* It is a comparative analysis of vegetation and flora in an agricultural landscape between 1992 and 2019. The biotope structures of the landscape elements were mapped in a 400 ha large section of landscape in Hasenholz in the Naturpark Märkische Schweiz in Brandenburg. Uniform biotope structures were mapped and defined as sampling units in which the flora was recorded in detail. This was done on the basis of the work and methods of the diploma thesis of Holger Pfeffer written in 1992. It was repeated for 83 of the linear landscape elements, which had been analyzed at two different points in time in the year 1992. The biotope types of the elements were further aggregated into three classes: Grass margins, hedges and semi-open elements. Subsequently the 83 relevés analyzed in 2019 were compared by means of a Wilcoxon signed rank test with those in 1992. Species richness; species turnover; mean Ellenberg indicator values; nectar and pollen provision as ecosystem service; the constancy of plants throughout the survey area and diversity, characterized by the Shannon Index were compared.

The analysis showed that grass margins have declined in area and species number, while hedges have increased at the cost of grass margins and semi-open elements. The latter overcompensated for their loss at the cost of grass margins. They hold the highest species numbers and value for honey bees among the three classes.

219 plant species were found in 2019, 22 more than 27 years ago. Neglecting single observations as a likely source of errors reduces this number to 161 species; seven more than 1992. The species turnover largely varies between 30% and 79% in the relevés, and amounts to 26% in the total sample. A large share of the plant diversity was found in the biotope class of semi-open elements, which represents only approximately 12% of the landscape elements analyzed. They show improvements in diversity, while grass margins decline and deteriorate in terms of area, diversity and ecosystem services.

The plant community experienced a disproportionate increase in the seven most abundant species. Despite these being nitrophilous plant species, no change between the two years was detected for the mean Ellenberg indicator values (MEIV) for N. Neither was this observed for the T-value, despite the climatic development towards higher temperatures. The MEIV were calculated in two different ways: Based on the indicator values of equally weighted plants ($MEIV_e$) and based on the indicator value multiplied by the plant coverage in order to adjust for the species abundance (weighted, $MEIV_w$). A significant decrease for moisture (F-value) was detected for the $MEIV_w$, but not for the $MEIV_e$. The K-value only showed a significant decrease in $MEIV_w$, but not in $MEIV_e$, while for the L-value (light) the opposite was found. These results may be a consequence of plants competing for light and of climatic changes over the past three decades, which lead to dry summers (F-value). However, the ordinal nature of the EIV and the multidimensional nature of the K-value limit the explanatory power of the MEIV method as a means of comparing relevés over a long period of time.

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1. Introduction

Worldwide as well as locally in central Europe biodiversity is declining. Plant species in Germany have encountered severe losses, predominantly in agricultural landscapes. (Wesche et al., 2012) The consensus of the United Nations is that this development is not desirable. (United Nations, 2015) (United Nations, 2018) At the grassroots level, bottom-up movements of citizens are urging their governments to take action. Recent examples are the extinction rebellion movement, which spread rapidly across about 70 nations since its formation in 2018 (Kuhlmann, 2019), and the referendum for biodiversity in Bavaria, which was the most successful of its kind in the history of the state. (Bündnis Artenvielfalt, Bayern, 2019)

A major target for policy changes in favor of biodiversity are the farming systems, which are subject to decisions by the federal states of Germany and the framework negotiated at EU level. The aspiration of the EU common agricultural policy in the next period 2020-2027 is to dedicate subsidies to the delivery of ecosystem services by the farmers – with the maintenance and development of biodiversity as one of nine major goals. (European Commission, 2018)

Supporting diversity by the means of universal legislation poses a challenge. Conservation efforts must take into consideration, that ecosystems are subject to and dependent on dynamic development. Many of them thrive on constant change and regular disturbance while being subject to global change. The subject of this thesis is the identification of changes in the elements of agricultural landscapes and their species composition over long time periods. Along with the underlying causes, their knowledge is crucial to understanding the potential and limitations of conservation measures. This thesis is a contribution to answer the question of how local and global changes manifest in the plant societies of agricultural landscapes on the basis of a survey in Brandenburg.

1.1 Scope and objectives

This master's thesis shall contribute to a better understanding of the development of the ecosystems in agricultural landscapes subject to local, regional and global changes. Their resistance to change requires long investigation periods. The complexity of ecosystems and the number of potential influence factors creates the need for sufficiently large data sets in order to analyze these long term effects.

Owing to a botanical survey conducted by Pfeffer (1992) it was possible to study the changes in the landscape elements of an agricultural area in Brandenburg over a time period of 26-27 years. Within the scope of this thesis, the study was replicated and allowed for the comparison of the inventory of plant species in the year 1992 with the status quo in 2019.
This work focuses on the development of wood structures, grass margins and semi-open elements. Semi-open elements are mosaics consisting of woods and grass elements. The focus lies on their value as biotopes and the potential for their protection and development. The direct comparisons of biotope elements in agricultural landscapes holds the potential to identify the major influence factors on the development of the species composition.

1.2 Project *BioZeit*

This thesis is part of the Project *BioZeit*. Soon after the reunification of Germany, an extensive study of vegetation and flora in six landscape sections of Brandenburg was conducted and is now repeated within the scope of the project. *BioZeit* started in October 2018 and is realized by Julius Kühn-Institut (JKI) and Leibniz-Zentrum für Agrarlandschaftsforschung (ZALF). Its aim is to identify the long term changes of biodiversity in agricultural landscapes with focus on arable farmland. It covers a time span of 25 years and builds on the work of Pfeffer (1992) and Kretschmer et al. (1995). In six different sections of the landscape of Brandenburg, an extensive species inventory was conducted shortly after the fall of the Berlin wall. Aside from vegetation, it covered the species groups of birds, carabids, butterflies, amphibians and reptiles. In the same sections the development of the species inventory was analyzed again in the context of land use and its changes. One challenge of this work is bridging the gap between the use of modern technology and methods while maintaining the comparability with the available data.
The goal of the research project is to gain a better understanding of the key factors which improve biodiversity in agricultural landscapes. (Hoffmann & Glemnitz, 2018)
Part of the project was the work of Holger Pfeffer in the early nineties . As a diploma student, he conducted an extensive vegetation study of landscape elements in the survey area– *Hasenholz* – and recorded the associated flora within the scope of his diploma thesis, with the support of colleagues at the time. Landscape elements are biotopes embedded in a landscape mainly used for arable farming. These can be small biotopes such as grass margins, as well as water bodies or small wetlands, temporarily used tracks and wood structures like hedges as categorized by Kretschmer et al. (1995) and elaborated in

Chapter 2.2. From this categorization, mapping guidelines were derived in order to obtain a uniform method for biotope mapping.
Pfeffer selected a landscape section for analysis, which is exceptionally rich in structure. His goal was to derive information on the value of hedges and recommendations for recovering these structures in other parts of the country. (Pfeffer, 1992)
The focus of the project *BioZeit* is on the analysis of the development of butterfly populations and soil-borne insects , since flying insects have encountered tremendous losses of biomass over the past decades (Hallmann et al., 2017) which likely affects their predators such as birds, amphibians and reptiles. This thesis covers the analysis of temporal changes of the vegetation in the landscape elements of the survey area around the small village of Hasenholz and the associated flora.

The work of Pfeffer encompassed the following tasks:

1. Mapping of the landscape elements around Hasenholz and Liebenhof in the survey area of 400 ha following the mapping guidelines by Kretschmer et al. (1995)
2. Calculating the area of the most important element types
3. Recording and evaluation quantitative and qualitative characteristics of all elements
4. Survey and assessment of the floristic composition in all landscape elements
5. Drawing conclusions and deriving recommendations for the design of biotopes in the future
6. Creating a map of the landscape elements in the survey area

These steps were repeated, discussed and improved in this master thesis, by:

1. Mapping of the landscape elements in order to identify changes in biotope structure
2. Repeating the survey of floristic composition and coverage where the complete set of data from 1992 was available (including sampling date)
3. Digitizing of the map created by Pfeffer (1992)
4. Pairwise statistical analysis of differences in group means to compare for
    - species number
    - biotope area and width
    - mean Ellenberg indicator value (MEIV)
    - pollen- and nectar supply
5. Comparison of constancy

6. Discussion of the method and its improvement
7. Creating a map of the landscape elements in the survey area 2019

1.3 Plant diversity in Central Europe

The situation in Central Europe is special with respect to how the development of biodiversity took place. While in most regions in the world the loss of species is due to the loss of pristine habitats, in Central Europe a co-evolution has taken place among human culture and wild species. (Kunz, 2016) Many species which originate from the steppes of Asia are dependent on the anthropogenic open landscapes which agriculture provided naturally over centuries. (Kunz, 2016)

The Red Lists for Germany show a decline of biotope types, as well as species throughout these different biotope types, predominantly in the open landscapes. (Finck et al., 2017), (Metzing et al., 2018) While forest species are often bound to natural dynamics, nowadays many species of the open landscapes depend on human cultivation. Without diverse land use systems like agriculture, meadows, pastures and others, such as coppice culture, many species would not find livable conditions in the otherwise thickly forested central European landscapes. However, many of these land use systems are no longer economically feasible and on the verge of extinction. (Kunz, 2016), (Poschlod, 2017) The use of forests for firewood and their use as range land contributed largely to keeping the landscapes free of thick shrub and tree cover. Also the husbandry of small animals such as rabbits, sheep and goats was a lot more prevalent only a few decades ago, leading to heterogenous land use through harvesting of fodder for the animals. (Poschlod, 2017)

Furthermore, the agricultural structure changed over the years with a tendency towards larger fields. (Meyer, 2013) Until today, the prevalent agricultural structure in Brandenburg presented a competitive advantage on a national scale regardless of comparatively low yield potential of the soils (Martens, 2010).Due to more intensive use through mechanization and homogenization of croplands, habitats for specialists or their competitive advantage are lost. Efficient crop protection exerts additional pressure. Another factor for the decrease of diversity is the clear and legally defined division between forest and open landscapes and the disappearance of large herbivores. The consequence is a succession of forest species at the cost of light dependent species. (Finck & Rieken et al., 2007), (Kunz, 2016)

The transition zones between forest and open landscapes are especially diverse. (Ruthsatz, 1984) This holds true for α-diversity, the number of species, as well as the β-diversity, which refers to the sequence of different plant societies along ecological gradients. (Dierschke,

2000) A majority of 90% of the species in agricultural landscapes is found on these margins by Oppermann (1998) while Ruthsatz & Otte (1998) found 50% and more. The destruction and disappearance of ecotones and special habitats at a micro scale is a major factor for the present risk of extinction. (Korneck et al., 1998)

Besides the direct influence on the landscape by humans, another driver for change are influence factors from the air: Nitrogen precipitates and leads to eutrophication of biotopes (Kunz, 2016), and greenhouse gases result in the present change of the climate (IPCC, 2013). Adjustments can be made with regard to the use of excess biomass from fertilized ecosystems and with regard to the micro climate through soil and water management. Moreover, the biotope quality is threatened by herbicides, plant residues after harvest, stones, particle run-off and mechanical disturbance through heavy machinery and tillage. (Ruthsatz & Otte, 1987)

For ecosystems covering large areas, often a connection between diversity and stability has been postulated. This does not hold for landscape elements. They are diverse due to different site conditions on a micro level. Factors like aspect play a role, which is also taken into account in this thesis. However, landscape elements are often lacking buffer zones and are very susceptible to deterioration caused by outer influence despite their diversity (Dierschke, 2000).

The theory of island biogeography, proposed by Mac Arthur & Wilson (2001) suggests, that at equilibrium, the species richness remains constant. The rate of species immigration determines the extinction of other species and the changes in the species compositions can be quantified by the species turnover as a quantitative unit of measure. (Nilsson & Nilsson, 1985)

Relevant long-term studies of the flora in agricultural areas in Germany were conducted by Wesche et al. (2012): Surveying grassland communities in floodplains and drawing comparisons between the 1950ies and 2008. They discovered a 30-50% loss in species richness on a plot level and a decrease in nectar-producing grassland herbs. A causal relationship between this development and the increase of the Ellenberg indicator value for nutrients (N), rather than climate change is considered likely. This was not confirmed by Diekmann et al. (2014) who analyzed long term changes in calcareous grassland vegetation, where P and water are the limiting factors, due to the high pH. For arable flora, Meyer (2013) showed that the plant communities were subject to drastic alterations and associations were replaced by fragmented communities. The arable flora however, is often not present in the field margins and hedges due to their edge effects. (Meyer, 2013)

The importance of landscape elements for biodiversity is undisputed. However, the connection of biotopes by simply supporting linear structures as exercised presently is subject to debate. As pointed out by Link (2004) there is a risk in considering the conservation of linear structures without an extensification of the surrounding landscape, because the increase in eutrophilic species will eventually push back residual populations characteristic to the biotopes at risk.

Jüttersonke et al. (2008) surveyed the development of fallows over 17 years and found that the diversity in plant communities is mostly dependent on the site conditions and soil fertility as well as the potential natural vegetation.

1.4 Survey area

The survey area is situated in the district of Märkisch-Oderland in Brandenburg, east of Berlin and in the nature park Märkische Schweiz as displayed in Figure 1.

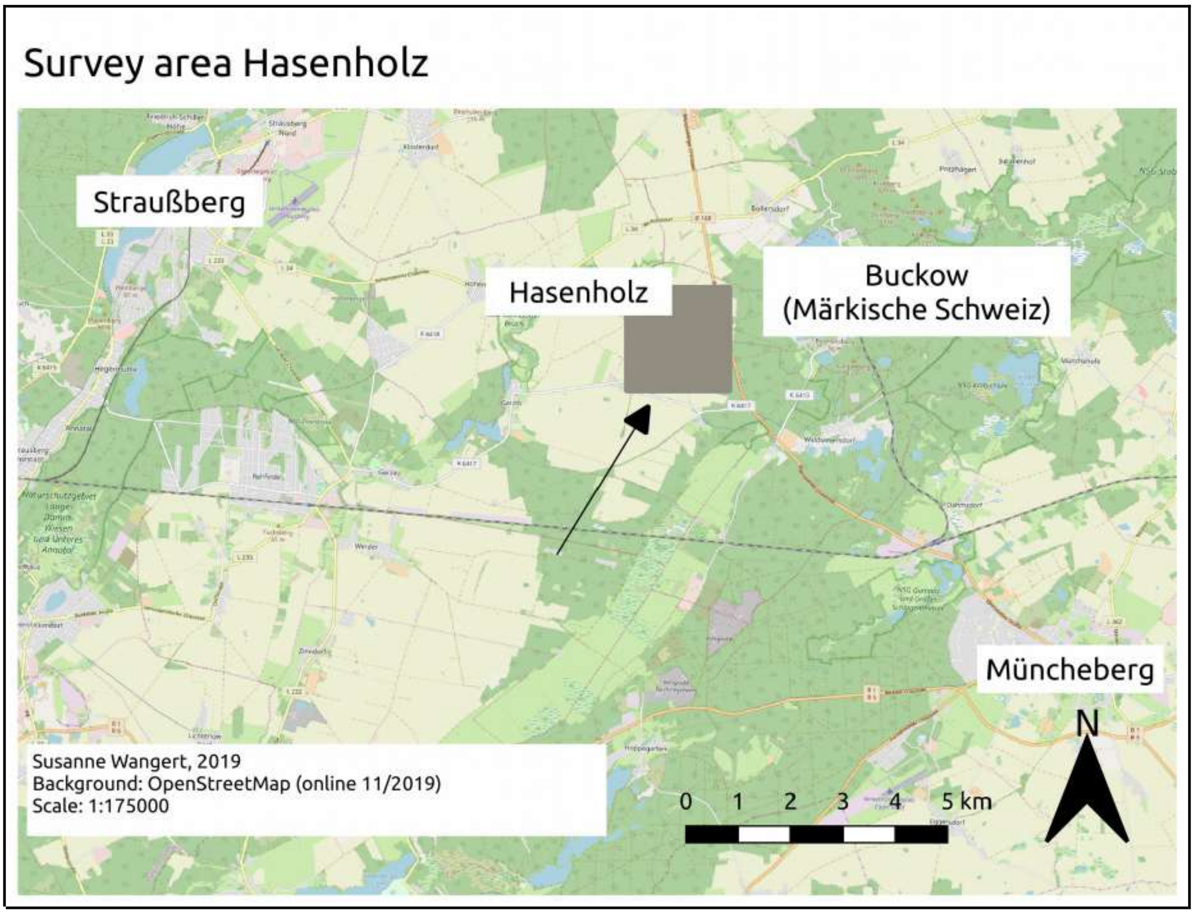

Figure 1: Survey area of Hasenholz between Straußberg and Buckow (grey square)

The survey area *Hasenholz* is located between Buckow and Strausberg. It comprises the villages of Hasenholz and parts of Liebenhof. Hasenholz is a farmers village with approximately 70 inhabitants and a history dating back into the 14th century. The village is now farmed by one farmer who lives in the village. (Förderverein Evangelische Dorfkirche zu Hasenholz e.V., undated) Liebenhof is situated in the southern part of the survey area with an organic farm and a shepherd farming the surrounding agricultural land.
The village structure of Liebenhof and Hasenholz is typical for old farmers villages of Mark Brandenburg. (Pfeffer, 1992)

1.4.1 Agricultural structure

It is often assumed that the agronomic structure of Brandenburg as part of the former GDR is solely a result of the collectivization (MLUL, undated) which took place since 1949 and was finalized in the 'Socialist Spring' in 1960. (Wolz, 2013) After World War II, in East-Germany as well as in West-Germany 90 % of all farms cultivated an area of less than 20 hectares and one percent more than 100 hectares. However, while in West-Germany seven percent of the land area were farmed by large farms, in the East it was already 30 % at the time. (Wolz,

Figure 2: Typical aspect of the survey area Hasenholz
- hedges rich in structure surrounding the agricultural fields

2013) In the GDR an expropriation of the land of large landowners, resulted in the division and redistribution of the land. The collectivization in the course of which cooperatives (Landwirtschaftliche Produktionsgenossenschaften, LPG) were founded (Martens, 2010), conducted a more rigid industrialization of agricultural field structures: While in West-Germany the average field size was six hectares, the GDR consolidation of farmland resulted in average field sizes of 38 hectares. (Spindler, 1989) The survey area Hasenholz represents an agricultural landscape which is used intensively. The three farms are of small to moderate size compared to the average full-time operated farm in Brandenburg.

Figure 3: Landscape element in the Hasenholz survey area, possibly an overgrown dead ice hole

The area was chosen by Pfeffer (1992) for his diploma thesis and as part of the project due to its richness in landscape elements. It represents a contrast to the large field structures described above. The system of hedges, which was planted in the 1920-30ies and which is no longer used since 1945 is not commonly seen across Brandenburg (Figure 2). Among the six landscape sections surveyed in the project *BioZeit*, it is the section richest in structure. The total area, excluding settlement area, surveyed in 1992 holds a share of 8.2 % landscape elements (Pfeffer, 1992). Of the total agricultural area this is 8.4 % (Kretschmer et al., 1995). The present share is subject to this study. The hedges contain elements, such as fruit trees and were formerly used more intensively than today. Dead ice holes, often overgrown by trees are commonly found in the area (Figure 3). Southwest of the survey area, the landscape elements are still fenced up to the roadside and used for sheep grazing as shown in Figure 4. The photo was taken outside of the survey area.

The area belongs to the protected landscape (Landschaftsschutzgebiet), 'Naturpark Märkische Schweiz' and is declared bird protection area. (LfU Brandenburg, 2019)

Figure 4: Fencing of landscape elements for sheep grazing, Garzau-Garzin 2019

1.4.2 Geological development

Neighboring the Baltic Sea and heavily influenced by the glaciers during the ice ages, the soils of Brandenburg are often sandy soils, terminal moraines and sandurs. (Hofmann & Pommer, 2005). With major development in the pleistocene and holocene, the landscapes are relatively young.

The survey area belongs to the physiographic unit Barnim-Platte as part of the Ostbrandenburgische Platte. (Strehmel, undated) The altitude of the slightly hilly area ranges between 72 and 97m above sea level. The landscape section is lightly slanted south west towards the Buckow Canyon (Buckower Graben). (Pfeffer, 1992)

Characteristic are the lakes and which are remainders of the glaciers of the ice age. (Hofmann & Pommer, 2005)

The survey area does not host large lakes, but various dead ice holes, depressions which retain water as shown also in the following images (Figures 5 and 6).

Figure 5: Dead ice hole, natural pool within survey area (biotope id: 50.2, 49.2, 49.4)

Figure 6: Dead ice hole, natural pool within survey area (biotope id: 28.1, 27.1, 27.2)

The area was affected by N-Emissions classified as medium and strong to medium Ca-immissions. (Hofmann & Pommer, 2005) The latter has been declining since 1980, when modern filter technology was installed in the local cement factory in Rüdersdorf. (Zimmermann, 2011)

While the main cultivars in the early nineties were corn, winter rye, winter wheat, summer barley, potato, winter rape seed, and feed such as Lucerne, the main crops today are rape seed, wheat, rye and corn. (Kretschmer et al., 1995)

The soils in Naturpark Märkische Schweiz are mainly Umbrisols, where the carbonate components are translocated into soil horizons where they are no longer accessible by plants. (Hoffmann, 2006) The soils in the survey area are deep loams and sands in the east as well as loams and deep loams in central parts of the area. Eutric Retisols (Geoabruptic, Arenic, Aric) and rarely Geoabruptic Luvisols (Arenic, Aric, Cutanic) are prevalent as documented in the soil survey map of Brandenburg (Bodenübersichtskarte, *BÜK 300,* by (LBGR Brandenburg b, undated)

The soils are under the influence of percolating water. (Kühn, 1997) The soils show an average yield potential of 50-30 and below. (LBGR Brandenburg, undated a) The fields used to be farmed by two cooperatives and one 'Volksgut' (state farm) yielding an average of 30-45 dt GE/ha (grain equivalent unit). (Pfeffer, 1992)

The area is classified as high plains with a low to very low soil moisture. The southern part of the survey area shows low soil moisture, while the area around the village of Hasenholz and the area east of it is marked by very low in soil moisture and not classified as area for water retention. (Strehmel, undated)

1.4.3 Climate

The climate in the region is cold with no dry season and warm summers, Dfb according to Köppen-Geiger classification. It is developing towards temperate climate with no dry season and warm or hot summers (Cfb or Cfa) by the end of the century under climate change as projected by Beck et al. (2018).

The average January temperature from the year 1990 to 2018 – the years for which data was available – calculated from daily averages is 0.4 °C; the mean temperature in July was 19.0 °C. (JKI, 2019) This deviates by +1.8°C and +1.3°C from the climate data provided by the weather station of Müncheberg and cited by Pfeffer (1992). The 30-year average January temperature from 1951 to 1980 was -1.37°C; the average in July was 17.72°C .

The trend over the latter time frame can be found in the following Figure 7.

The average precipitation is 565 mm per year between the years 2006 and 2018. Only for this time frame was data available. However the annual precipitation underlies large annual changes and ranges from 368 to 758 mm p.a. (JKI, 2019) as depicted in the following Figure 8.

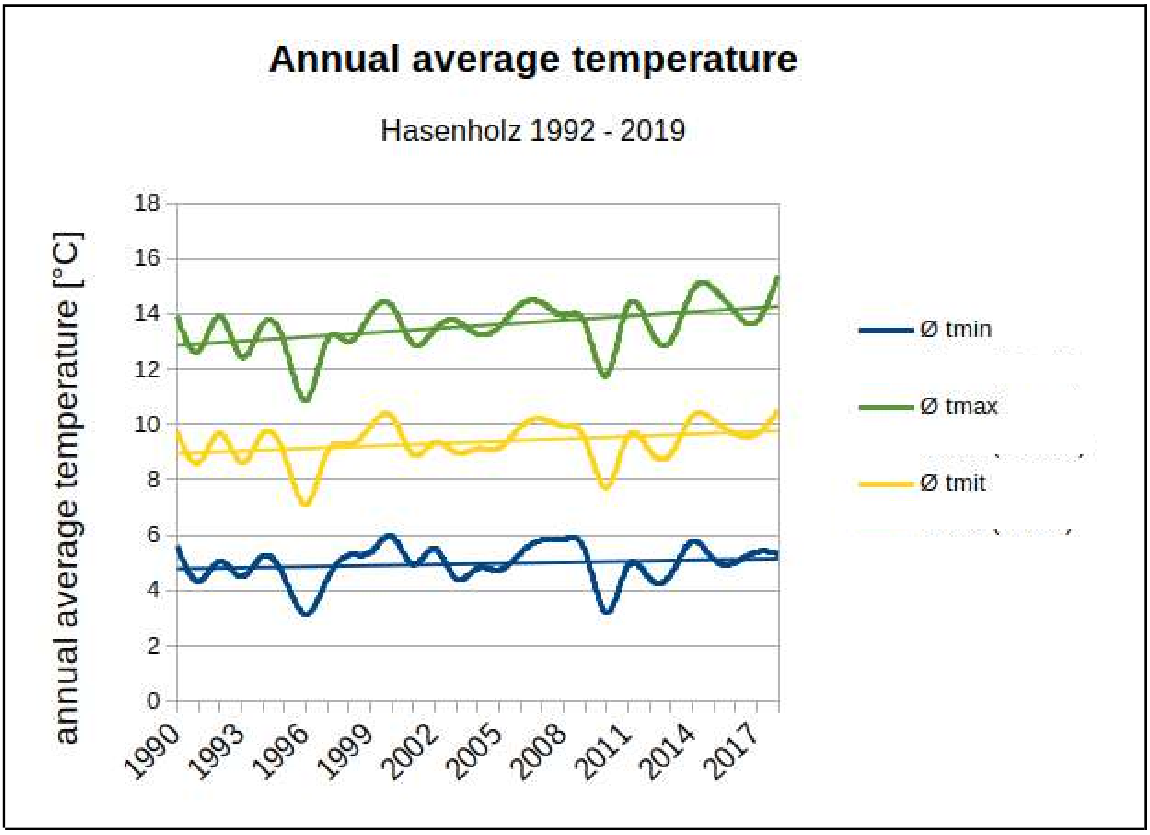

Figure 7: Annual average derived from daily average temperatures (data: JKI, 2019), minimum (tmin), maxium (tmax) and average (tmit) daily minimum air temperature at 2 m height

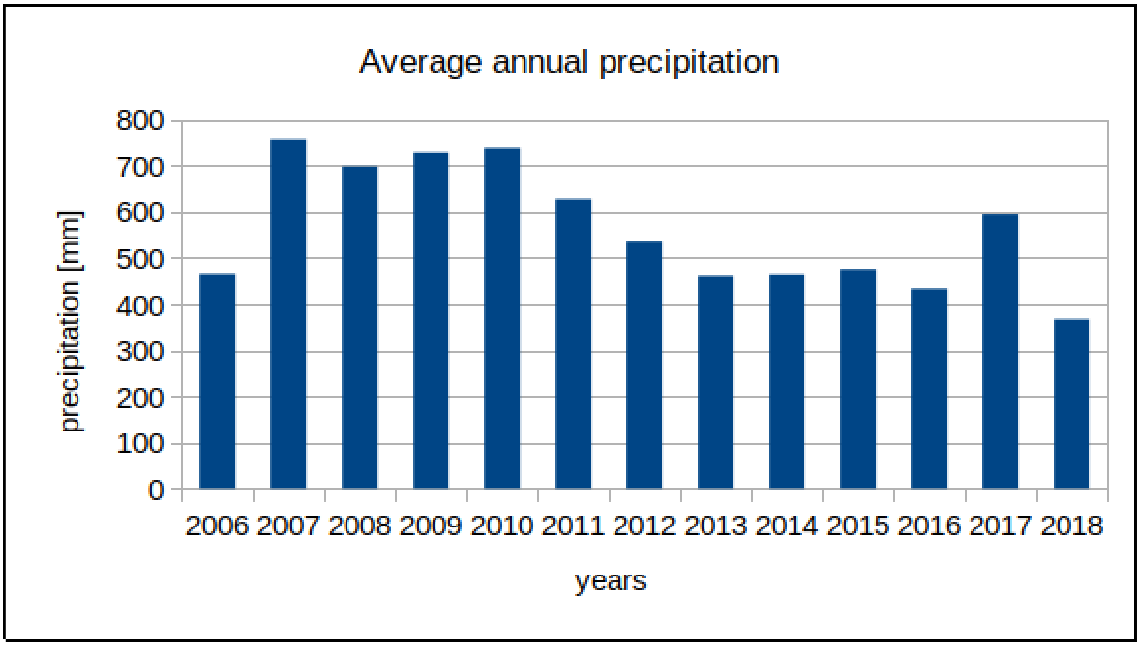

Figure 8: Average annual precipitation 2006 – 2018 (data: JKI, 2019)

1.4.4 Demarcation of the survey area.

The square area of 400 ha (2x2 km) was chosen in accordance with the remaining five landscape sections. The raster lines of the topographical map 1:10,000 (0810-131 Bollersdorf, Ausgabe für die Volkswirtschaft) served as demarcations in order to avoid a biased selection. (Pfeffer, 1992)

1.5 Research questions and hypotheses

This master thesis uses the data collected in the nineties to answer the question of whether and how the landscape elements of the agricultural landscape, its biotopes and flora have changed. If the biotopes as well as the farming structure have remained constant, other factors can be identified as drivers for changes in species composition. The aim of this study is to identify the role of global changes like nitrogen deposition and climate change.

While not much comparative research is available for landscape elements which covers a large time span of more than 27 years, within this thesis it was possible to detect long term trends.

The overarching question is: How did the landscapes and its biotopes change since 1992?

For detailed answers the following hypotheses were formulated:

Biotope extent - area and width

The total extent of biotopes has decreased. Area was lost throughout the years, while different biotope classes show differences in their development. Semi-open elements and grassland margins have reduced in width, while the hedges have increased in width.

This assumption is derived from subjective impressions by colleagues involved in the project as well as the tendency of woods to encroach into biotope types dominated by grass, forbs and herbs.

Biotope structure

The biotope structures and thus biotope types of the landscape have changed. Shrub encroachment does not only alter the extent of biotopes, it also alters biotope structures. Semi-open landscape elements with scattered bosks have grown to become dense woods.

Species number (richness)

The number of species has declined

- in the biotopes not facing changes in area

- in the biotopes not facing changes in structure
- in all biotopes
- in all biotope classes (defined in Chapter 2.2.3)

A decrease in diversity is hypothesized due to the overall trend described by Metzing et al. (2018). This trend is related to the loss of biotope types as described by Finck et al. (2017), who identified a positive development only for 3 % of the biotope types. Hasenholz allows a comparison without a major loss of biotopes.

Species turnover

A species turnover has occurred

- in all biotopes
- in all biotope classes

As a consequence of the previous hypothesis, it is hypothesized, that if the species number is subject to change, so is the species turnover.

Species occurrence frequency (constancy)

The constancy of plants has changed. Nitrophilic and frequently occurring species are expected to increase in constancy, rare species and specialists are expected to decrease. This is due to shifts in site conditions which are expected and further described in the following.

Mean Ellenberg indicator values (MEIV)

The mean Ellenberg indicator value shows different results for

- T-value (temperature)
- F-value (moisture)
- K-value (continentality)
- N-value (nitrogen, nutrients)
- L-value (light)

Due to the changes in climate as derived from weather data (JKI, 2019), illustrated in Chapter1.4.3, the temperature (T) and moisture (F) have changed, as well as the climate (K). For T higher values are expected, following the trend shown in 8. As a consequence, the opposite is expected for F. The climate is expected to shift towards a more continental climate, since the amplitude of temperature is increasing as shown in 7. The continuous use of nitrogen fertilizers is expected to lead to a plant composition with higher average N-values. As a result of vegetation growing more dense, a decrease in light demanding species and subsequently the L-value is expected to decrease as suggested by Kunz (2016).

The soil acidity is dependent on soil substrate, organic content, liming and fertilization. It is expected to remain constant, due to the buffer capacity of soils and was not tested, neither was the S-value for salinity, because due to the long-term positive water balance an accumulation of salts is unlikely.

Ecosystem services: Pollinator value

The functionality of the plant communities has changed with respect to the ecosystem service of pollen and nectar provision. A decrease of pollen and nectar is expected, due to a decrease in abundance of flowering plants. This is hypothesized as one potential cause of the decline in insects described by Hallmann et al. (2017).

Shannon index as diversity indicator

The Shannon index of the plant associations has changed as a result of the development of the previously hypothesized development of species richness and frequency of occurrence.

Research design

The research design was appropriate with respect to the size of sampling units and sample size.

2. Methodology

The following chapter gives an overview over the terms used (Chapter 2.1), the methods used in the field in the nineties and today (Chapters 2.2 - 2.3), the indicators chosen for comparison (Chapter 2.4) as well as the methods of data analysis (Chapter 2.5)

This master's thesis is classified as a scientific succession study, identifying and describing the changes in the local flora and seeking an explanation. (Traxler, 1997) Since the major goal is a comparison and interpretation of changes in the plant composition between 1992 and 2019, the methods formerly employed by Pfeffer (1992) and Kretschmer et al. (1995) were used for comparability.

2.1 Definitions

Prior to the methods, the terms used and the boundaries of the survey area shall be defined.

2.1.1 Terms and translations

Since the basis for this work was written in German, the most frequently used terms are clarified in the following:

Landscape elements
– *Strukturelemente, Kleinstrukturen*

The term landscape elements or structures refers to biotopes in agricultural landscapes, such as groves, hedges, grass margins and unused grassland to name a few.

Biotope structures, biotope types, landscape element types
– *Biotoptypen, Kleinstrukturtypen*

Biotope types are classified in various mapping guidelines (biotope mapping). In this thesis the focus lies on certain types of landscape elements in agricultural landscapes only, which are further differentiated in Chapter 2.2 on the basis of the definition by Kretschmer et al. (1995). The term 'types of landscape elements' and the broader term 'biotope type' are used interchangeably in this thesis.

Biotope classes
– *Biotopklassen, Kleinstrukturklassen*

Later on, these biotope types are aggregated into the three classes of hedges, semi-open elements and grass margins as described in more detail in Chapter 2.2.3.

Biotopes, sampling units
– Biotop, Kartiereinheit

This term refers to a uniformous unit of one biotope type, which was identified by Pfeffer (1992) and used as sampling unit throughout the study.

Relevé
– Vegetationsaufnahme

Survey of one sampling unit, yielding a list of the present flora and its coverage.

Tree strip
– Baumholzstreifen

A linear biotope type consisting of mainly trees (details in Chapter 2.2)

Wood strip
– Gehölzstreifen

A linear biotope type consisting of trees and shrubs (details in Chapter 2.2)

Shrub strip
– Gebüschstreifen

A linear biotope type consisting of mainly shrubs (details in Chapter 2.2)

Semi-open (tree / wood / shrub) strip
– (Baumholz-/Gehölz-/Gebüsch-) Streustreifen

Mosaic of tree / wood / shrub structures and grasslands, with scattered bosks on grassland. (details in Chapter 2.2)

Grass strips / grass margins
– Säume frischer Standorte

Margins on sites with medium available moisture, habitat of forbs, grasses and herbs. (details in Chapter 2.2)

2.1.2 Demarcation of the survey area

For demarcation of the survey area the grid lines of the topographical map 1:10.000 (0810-131 Bollersdorf, Ausgabe für die Volkswirtschaft, (Pfeffer, 1992)) were used in order to yield a random and comparable survey area. The total area represents a square of 2 x 2 km.
Within the *BioZeit* project, a core survey area of 100 ha was differentiated. This differentiation however, did not play a role in this study.

2.2 Biotope mapping of landscape elements

In the course of the biotope mapping, the vegetation in the landscape elements of the survey area were categorized into defined types of landscape elements. The data available from the mapping of 1992 was checked against the present reality in terms of biotope structure. For the biotopes which served as sampling units in the subsequent floristic survey, the size (length and width) was recorded.

For the floristic survey, the types of landscape elements were chosen, which occur frequently throughout the survey area. This means that the biotope type and its association with other biotopes occurs more than once. This selection was made, in order to reduce the likelihood of effects specific to the biotope type or biotope constellation. Another criterion was that the survey data from 1992 was complete, with respect to the frequency and date of sampling as described in Chapter 2.3.2.

2.2.1 Biotope structure / landscape element types 1991-1992

The mapping of biotope structures is a standardized procedure to characterize the habitats of a region. Kretschmer et al. (1995) evaluated biotope mapping guidelines from five different federal states. They were found not to be detailed enough with regard to the landscape elements of agricultural landscapes, such as farm tracks, field margins or waterlogged spots in the field for example. Therefore mapping guidelines were developed, to cater to the requirements for further research, especially of the research in entomology. While doing so, the aim was to stay in accordance with the biotope mapping standards of Brandenburg (Zimmermann, 1992, cited by Kretschmer et al., 1995) as much as possible, which were also still unpublished at the time. Furthermore, the method was influenced by instructions given by the Bayrische Landesanstalt für Bodenkultur und Pflanzenbau (anonymous, 1989) and the classification of habitat structures for the agricultural landscapes of the regions under loess in central Germany by Schnurrbusche et al. (1986), cited by Kretschmer et al. (1995). The outcome was different from the mapping standard of today, which was updated in the meanwhile. (Zimmermann, 2007) However, the mapping standard encompasses all biotope types on the list of ecologically relevant biotopes defined by Hille (1989), cited by Pfeffer (1992).

The main biotope types present in the area of Hasenholz are a selection of the overall catalogue by Kretschmer et al. (1995). It can be found in the following Table 1 and includes linear structures and wood structures.

Table 1: Mapping instruction for a selection of landscape elements (Kretschmer et al., 1995) (Translation from German)

ID	Biotope type	Criteria
F . . .	woods	outside of enclosed forests, surrounded by open landscapes, maximum 4 ha, no silvicultural use, >80% area coverage with woods
F1 . .	areal elements	ratio of length to width < 4:1 and/or width >20 m, natural or artificial woods, minimum area 0.025 ha
F10 .	areal woods	areal, enclosed woods with >80% coverage, little undergrowth by annual or perennial plants
F101	shrub area	mainly shrubs, maximum 10% trees overarching
F102	woods area	woods consisting of shrubs and trees, >10% coverage by trees
F103	trees area	mainly trees, maximum 10% coverage by shrubs
F11 .	semi-open areal elements	areal, 20 – 80% coverage, moderate or strong undergrowth by annual or perennial plants
F111	shrub meadow	mainly shrubs, maximum 10% trees overarching
F112	wood meadow	woods consisting of shrubs and trees, >10% coverage by trees
F113	tree meadow	mainly trees, maximum 10% coverage by shrubs
F2 . .	hedges / wood strips	linear wood structures, ratio of length to width < 4:1, <20 m, natural or artificial woods, minimum length 50 m
F20 .	hedges / wood strips	linear, mostly enclosed wood structures, with wood coverage >80% and scarce to moderate undergrowth by annual or perennial plants
F201	shrub strips	mainly shrubs, maximum 10% trees overarching
F202	wood strips	woods consisting of shrubs and trees, >10% coverage by trees
F203	tree strips	mainly trees, maximum 10% coverage by shrubs
F21 .	semi-open hedges / wood strips	linear wood structures, ratio of length to width < 4:1, <20 m, natural or artificial woods, 20 – 80% coverage, moderate or strong undergrowth by annual or perennial plants
F211	semi-open shrub strips	mainly shrubs, maximum 10% trees overarching
F212	semi-open wood strips	woods consisting of shrubs and trees, >10% coverage by trees
F213	semi-open tree strips	mainly trees, maximum 10% coverage by shrubs
F303	solitary shrubs	woods consisting of shrubs without trees, up to 0.01 ha
G130	grass margins	dominated by grass, herbs and forbs, often along tracks, roads or hedges, minimum width 0.5 m, no woods overarching

Linear structures were differentiated from areal structures. Another difference was made between the densely grown wood structures and semi-open elements with only 20-80% of coverage by woods. The separate mapping of semi-open elements was of particular interest for the entomological studies of the project. The succession stages shrubs, woods and trees are three biotope type classes which reoccur among linear as well as areal, semi-open as well as dense wood structures.

Further special biotope types such as shadeless minor water bodies, local wetlands and dystrophic meadows (Molinion) occurred within the survey area. However, they were not covered within the scope of this thesis.

All uniform units >100 m² were mapped by Pfeffer (1992). The residential areas, including the garden areas, were not considered. For the remaining agricultural area, full coverage was obtained. For cases where no margin was registered, no margin was present.

The following images show examples of the different biotope types present, starting with the rarest kind of biotope type, the semi-open elements (Figure 9).

Figure 9: Biotope type F111, shrub meadow - mainly shrubs, maximum 10% trees overarching (biotope-ID left: 20.2, right: 20.5)

The following image shows the relatively rare tree strips, which only show little undergrowth in the form of hedges, but more grass and herbal green cover than dense hedges themselves (Figure 10).

Figure 10: Biotope type F203, tree strips - mainly trees, maximum 10% coverage by shrubs (biotope-ID left: 25.7)

The type of hedges with only very little undergrowth in form of grass and herbs is shown in the following image (Figure 11).

Figure 11: Biotope type F201, shrub strips - mainly shrubs, maximum 10% trees overarching; Biotope type G130, rudimentary grass margin

The biotope elements are mostly linear, but also elements with areal characteristics occur as displayed in the following image (Figure 12).

Figure 12: Biotope type F101, shrubs - mainly shrubs, maximum 10% trees overarching

An example of a grass margin is shown in the next image (Figure 13).

Figure 13: Biotope type G130, grass margins - dominated by grass, herbs and forbs, often along tracks, roads or hedges, min width 0.5 m, no woods overarching. (road leading to Hasenholz from eastern direction)

However, the grass margin shown in Figure 13 is an example of a relatively wide margin. Most margins found in the survey area are marginal strips between tracks, wood structures and the agricultural field (Figure 14).

Figure 14: Biotope type G130, grass margins - dominated by grass, herbs and forbs, often along tracks, roads or hedges, min width 0.5 m, no woods overarching – example of narrow space for G130

The width and length of the margins were measured in the field and the area derived from this information by multiplication. (Pfeffer, 1992)

2.2.2 Biotope structure / landscape element types 2019

While in the early nineties, the measurement had to be done manually in the field, nowadays, geoinformation systems (GIS) are available. The information recorded by Pfeffer (1992) was digitized using a GIS by Pfeffer and Wahrenberg (2019). Based on this line-shapefile and the width of the linear elements which was recorded in 1992, a polygon shapefile was created within the scope of this thesis. The polygon was created using the buffer function of QGIS and manual adjustment according to the available orthophotos from 1994 (Geo-Basis-DE / LGB 2019).

A map on the basis of recent orthophotos and a device using GPS (global positioning system) was used during fieldwork to determine the present location of the biotopes. This was possible due to a high accuracy of the images presently available. However, the width of narrow linear elements could not be determined remotely, using GIS only, neither in case of the 1992 situation nor in the case of 2019. In order to determine location and area of all biotope sampling units, a combination of remote investigations and inspections in place was chosen. While the average width of grass margins could best be determined by a visual inspection in the field, in case of dense hedges the measurement of their width using orthophotos proved to be the better choice. However, overarching trees and shadow cast by them were sources for errors which could be reduced due to the knowledge of the conditions on site.

The level of accuracy of the earlier survey lead to a conservative approach to re-mapping the area in order to avoid errors due to the deviation in accuracy of the two orthophotos available. This means, that the width of biotopes was recorded only where a change in width was obvious and certain. Small changes could therefore not be detected. More accurate measurements conducted today, regardless of the accuracy previously obtained would have yielded a systematic error.

The grass margins were mapped starting at a width of ≥0.5 m. Structures without wood stands, which have not been subject to tillage classify as margins. (Kretschmer et al., 1995)

The margins which were tilled but not covered by a crop are not classified within this system and are not further investigated within the scope of this thesis.

The area of linear biotopes was calculated through multiplication of width and length as done by Pfeffer (1992). The biotope structures present were compared with the biotope types recorded by Pfeffer (1992), according to the mapping instructions by Kretschmer et al. (1995). Deviations were recorded. A mere change in size was not recorded as a change in structure.

2.2.3 Biotope classes 2019

The biotope mapping following the methods of 1992 was repeated in 2019 as described in Chapter 2.3.2. The mapping is expected to reveal differences where the biotopes have changed in structure. For further analysis, the biotopes were divided into three major classes: grass margins, semi-open biotopes and hedges.

Grass margins

biotope type: G130, abbreviation: ‘gras’

This biotope class encompasses low-growing, non-woody species, such as grass, herbs and forbs. The biotope type is maintained by grazing, mowing, occasional soil disturbance by animals or machinery and subsequent regrowth. An example is shown in the following Figure 15.

Figure 15: Biotope class 'grass margins'

Semi-open biotope types

biotope types: F111, F112, F113, F211, F212, F213, abbreviation: 'semi'

The semi-open biotope types can be anthropogenically influenced systems in which shrubs and trees are established in combination with low-growing, non-woody species, like grass, herbs and forbs. Like the biotope class of grass margins, they rely on disturbance or care by humans or animals, for their existence. Often they are transition stages between grass margins and hedges.

The German denotation of ‘Streustreifen/Streuflächen’ is likely due to the structural similarity with fruit orchards (‘Streuobstwiesen’), which are characterized by a loose strand of scattered (‘verstreut’, in Streulage’) tall fruit trees.

Figure 16 shows an example of the biotope class of semi-open elements.

Figure 16: Biotope class ‘semi-open elements’ (biotope ID: 25.4)

Hedges

biotope types F101, F102, F103, F201, F02, F203, F303, abbreviation: ‘wood’

Hedges may have derived from semi open elements from which no biomass was removed or they may have been planted intentionally for the purpose of wind protection or demarcation, in case of fruit trees for the respective harvest. They consist mainly of woody species and hardly show undergrowth in form of low-growing, non-woody species, such as grass, herbs and forbs.

A hedge in which sections are dominated by shrubs and other sections are dominated by trees is shown in the following Figure 17. It would be classified as a wood strip (F202).

Figure 17: Biotope class ‘hedge’ (F202)

For a detailed description of the biotope types consult Table 1 in Chapter 2.2.1 for reference.

2.3 Floristic survey

Aim of the floristic survey was to obtain a comprehensive list of the flora and its maximum coverage throughout the inspections for each sampling unit.

2.3.1 Floristic survey 1991-1992

Subsequent to the biotope mapping, the vegetation was further investigated by Pfeffer (1992). In the nineties, the survey was conducted mainly for linear structures which are most prevalent in the area, but also for the areal elements present.

The qualitative and quantitative assessment of the vegetation was conducted following the Braun-Blanquet method (Braun-Blanquet, 1964) in order to capture the coverage of the single species in a sampling unit. Unlike in other vegetation studies using the Braun-Blanquet method, no sampling units of defined size were used. Instead, the attempt was made to obtain a full species inventory for each uniform biotope unit identified in the field. The plant species list was provided with estimates of the total coverage of the plant species in each biotope.

As sample plots or sampling units for the floristic vegetation survey, the biotopes as defined in the biotope mapping were used. They are of different sizes, which is unusual for vegetation surveys. (Chytrý & Otýpková, 2003)

Due to the late start of the floristic survey in 1991, the survey was conducted over the course of two years 1991 and 1992. For simplicity, the two years are referred to as 1992 for the rest of this thesis. For the most part, two iterations took place during which the flora was recorded. In the field, forms on paper sheets were used to enter the data. The results of the survey were summarized in a spreadsheet as a synoptic list, which was completed over the course of the two investigations. The value in the synoptic list therefore represents the maximum coverage of a plant found in a particular place. The coverage values from former investigations were corrected in favor of a higher value, but not revised down. The aim was to exhaustively capture the flora of the biotopes as sample units. (Pfeffer, 2019)

2.3.2 Floristic survey 2019

Compared to the entire survey area only a small proportion of the landscape structures was analyzed, due to the requirement of having a set of uniform samples at hand and limited availability of original data meeting all criteria. In order to decrease the likelihood of random effects, the comparison was done without the consideration of single special sites such as water bodies, waterlogged sites, or wet meadows and the focus laid on grass margins, wood strips/ hedges and semi-open elements as a blend of the two as listed above in Chapter 2.2.

For the vegetation survey, the spreadsheet with the synoptic survey data from the nineties was accessible for reference. For detailed information on the exact time of the year in which the species list was created and the number of investigations, the original species lists were consulted.

The repetition of the floristic vegetation survey was done for 83 sample biotopes, for which the original paper records were available, including the date at which they were conducted and for which two inspections were done. This way it was ensured that the survey was conducted at approximately the same time of the year and that the error due to a time lag was minimized to 2-3 weeks.

Again, paper forms for use in the field were chosen and the results of the former investigation from the nineties pre-entered. They were only taken for reference during the field work and were not used as a checklist, in order to avoid bias towards the results obtained in the 90s.

Not only do the biotopes differ as units for the floristic vegetation survey in size among each other, the size may also vary throughout the years.

Before starting the field work it was decided to repeat the survey within the bounds of the biotope unit which was present and identifiable in 2019 instead of repeating the survey within the bounds of the original demarcations. Since the actual location of the biotopes may have shifted, different biotope types may be present in the original location. For example a margin may have been shifted and a crop may have taken its place. Assessing the crop area instead would not yield comparable results. Furthermore, the location of the original sampling units could not be determined precisely, due to the low accuracy of mapping without the use of a

GIS in the early nineties. Therefore, the definition of today's uniform biotope sample area proved to be the only practical approach.

2.4 Indicators and variables for data analysis 2019

In order to compare the biotopes and their plant inventory of 2019 with 1992, the above mentioned mean Ellenberg indicator values (MEIV) were used, as well as other indicators described in the following. (Δ = difference of)

1. Δ area and Δ width (Chapter 2.2.2)
2. Δ biotope type (Chapter 2.2.2), summed up in three biotope classes (Chapter 2.2.3)
3. turnover in the sampling units (biotopes) (Nilsson & Nilsson, 1985)
4. Δ species number (species richness) in the sampling units (biotopes)
5. Δ Shannon index (Shannon, 1949)
6. Δ MEIV
7. Δ pollinator value
8. Δ constancy of species

2.4.1 Species number / species richness

The species number was assessed for each sampling unit for both years and the difference Δ species richness calculated.

To reduce the likelihood of a systematic error originating from the differences in mapping by two different persons, the Δ species richness was calculated a second time without the single observations. Species which were only encountered once were neglected.

2.4.2 Species turnover

The turnover is a measure of the change in species composition. It indicates, how many species have disappeared and how many have newly emerged. It was defined by (Nilsson & Nilsson, 1985) as follows:

$$turnover = \frac{A_{ex} + B_{ex}}{A_{total} + B_{total}}$$

A_{ex} = number of species occurring exclusively in group A

B_{ex} = number of species occurring exclusively in group B

A_{total} = total number of species occurring in group A

B_{total} = total number of species occurring in group B

2.4.3 Constancy of species

The changes in species composition in the plant communities present the cause for the changes in their mean indicator values. The occurrence of species is analyzed in depth in order to explain potentially observed changes. Furthermore, qualitative information can be derived from the plant composition, for example on changes in management of the biotopes.
The term constancy describes the frequency of the occurrence of one species throughout all 83 sampling units. The constancy was calculated for all species and the situations of 1992 and 2019 were compared. The plants which have increased in their frequency of occurrence and those which have decreased were identified for the different biotope classes. In the rest of this document they are termed as winners and losers for ease of use. These terms shall be used as objective descriptive terms with no valuation attached.

2.4.4 MEIV

On the basis of the plant list from the floristic vegetation survey, Pfeffer (1992) sought to derive information on the ecological quality of the plant composition. For this purpose, the method „mean Ellenberg indicator values“ (ökologische Bestandszahlen) based on Vollrath, 1979, cited by Pfeffer (1992) was used. It builds on the ecological indicator values defined by Ellenberg (1974) and refined later on and until after Ellenberg's death in 1997. In this method, the Ellenberg indicator values attributed to each plant are multiplied by the mean coverage value of each plant in order to reflect the abundance of each species.
The mean coverage values are displayed in the following Table 2.

Table 2: Braun-Blanquet (B.-B.) scale and median values for the calculation of mean Ellenberg indicator values, chosen by Pfeffer (1992)

B.-B. **Scale**	*Coverage values (Artenmächtigkeiten)*	*Median coverage*	*Min*	*Max*
r	*sparsely, low coverage*	*0.01%*		
+	*<1%*	*0.1%*	*0.0%*	*1.0%*
1	*1-10%*	*5.0%*	*1.0%*	*10.0%*
2	*10-25%*	*17.5%*	*10.0%*	*25.0%*
3	*25-50%*	*37.5%*	*25.0%*	*5.0%*
4	*50-75%*	*62.5%*	*5.0%*	*75.0%*
5	*75-100%*	*87.5%*	*75.0%*	*100.0%*

This method was repeated and the results compared in 2019.

Out of the seven Ellenberg indicator values, moisture (F), light (L), continentality (K), nutrients (N) and temperature (T) were chosen for analysis. Salinity (S) and reactivity (R) were assumed to be constant and considered.

The mean Ellenberg indicator values were calculated in two ways.

One method is the calculation of a $MEIV_w$ (w = weighted). It is derived through multiplication of the coverage values of each plant with the indicator value in order to give more weight to the species with higher abundance.

$$MEIV_w = \sum_{i=1}^{n} (plant\ coverage_i * indicator\ value_i)$$

The other method to calculate a $MEIV_e$ based on equally weighted plant individuals simply consists of calculating an average of all plant indicators attributed to the sample:

$$MEIV_e = \sum_{i=1}^{n} indicator\ value_i * \frac{1}{n}$$

For the weighted method, several steps had to be undertaken in order to prepare the gathered data for the calculation.

1. Conversion of the Braun-Blanquet scale

   The Braun-Blanquet scale was converted into decimal numbers in order to allow further processing and calculations. The median of the defined classes were chosen as shown in Table 3 in Chapter 2.4.4. (Pfeffer, 1992)

2. Normalization of coverage values

   The Braun-Blanquet method is a subjective method with respect to the estimation of plant cover. Therefore the sum of coverage value usually deviates from 100% in most biotope mapping units. It may lie above or below, which would lead to a distortion when calculating weighted mean ecological indicator values by multiplication of the percentages with the indicator values of the plants. The mean sum of coverage fluctuates between 30% and 144%. Even when the minimum and maximum of the respective coverage classes are used in the calculation, in some cases the total coverage of 100% cannot be reached. Even when the maximum of the classes is summed up, 21 of Pfeffer‘s 192 plant lists remain below 100% coverage. The mean indicator value would be biased accordingly.

   Therefore the species coverage values in each relevé, multiplied with their respective indicator values were divided by the sum of the coverage values (accumulated or total coverage) in the relevé before summing them up. This way the sum of the coverage values was normalized to 100% coverage.

$$coverage_{norm}=\sum_{i=1}^{n}\left(\frac{plant\ coverage_i}{\sum_{i=1}^{n} plant\ coverage_i}\right)$$

3. Adjustment for missing values

   Another source of bias is the fact that not all plants have an indicator value attributed. Whenever a biotope holds plants with no indicator value attributed, the product yields zero and leads again to a distortion of the result. Therefore another correction step was conducted in the same manner. Like in step two, the final result ($coverage_{norm}$) was divided by the sum of plant coverage values which were successfully multiplied with the ordinal number of the respective indicator value. (Indicator available: $coverage_{norm.av}$) This way, the coverage values were adjusted for those values for which an EIV was missing ($coverage_{norm.adj}$) so the sum of the coverage values would again yield 100%.

$$coverage_{norm}.adj=\sum_{i=1}^{n}\left(coverage_{norm}.\frac{i}{\sum_{i=1}^{n} coverage_{norm}}.av.i\right)$$

4. Multiplication of coverage with indicator value.

   Finally in the next step, each adjusted and normalized coverage value was multiplied with the indicator value of the appertaining species.

$$MEIV_w=\sum_{i=1}^{n}\left(coverage_{norm}.adj*EIV\right)$$

Furthermore the lists of winners and losers and the average of their respective EIV were compared.

In order to visualize a potential connection between the occurrence and the indicator values, the average indicator value was calculated for the top 10 winners as well as for the top 10 losers. The same was done for the subsequent top 11 and top 12, until the top 52. Each result of top losers was deducted from the average of top winners. The difference was shown in a plot in order to visualize, whether the plants’ indicators show a tendency to having decreased or increased.

2.4.5 Ecosystem services: Pollinator value

The floristic database at the former Institut für agrarrelevante Klimaforschung Müncheberg (AKF, institute for climate research relevant to agriculture), as part of the Bundesforschungsanstalt für Landwirtschaft Braunschweig-Völkenrode (FAL, federal research institution for agriculture in Braunschweig-Völkenrode), contains data, which allows further evaluation of changes in species composition. The database covers the flora of Germany based on primary data on the indicator values among others (Ellenberg, 1991) and values and categories of current and potential usage which were obtained from Schlosser et al. (1991). The floristic database is now maintained by Julius Kühn-Institut, which was founded in 2008, combining the former Biologische Bundesanstalt für Land- und Forstwirtschaft (BBA, federal biological institute for agriculture and forestry), the former Bundesanstalt für Züchtungsforschung an Kulturpflanzen (BAZ, federal institute for research on plant breeding) and the FAL. (Ordon, undated)

This species list, however, only gives an overview over the species present in the area of Märkische Schweiz at the time. The plants have a value attributed based on their value for pollinators. Separately for nectar and pollen the values 0, 1 and 2 were attributed to indicate this ecological function. For the category pollen and nectar producing plants '1' is equivalent to a honey yield value of 3-4 (Trachtenwert), indicating a high yield, '2' represents a value of 1-2, which stands for lower yields.

Presently, no general methodology exists for floral resource availability estimates. (Szigeti et al., 2016) Similar to the calculation of mean Ellenberg indicator values, the value of a plant with respect to nectar and pollen delivery was multiplied by the plant coverage and the results summed up for the plant community of the respective biotope unit. Again the results were divided by the sum of the total coverage to adjust for the bias due to accumulated coverage values deviating from 100% as described in the previous chapter.

2.4.6 Shannon index as diversity indicator

The Shannon index is an indicator developed by Claude Shannon as a measure for diversity, originally developed to measure entropy in strings of text. (Shannon, 1949) The term diversity is often used as a synonym for species richness as criticized by Spellerberg & Fedor (2003). The index is defined as:

$$H' = -\sum_i p_i \ln p_i \, with \, p_i = \frac{n_i}{N}$$

with

N = total number of individuals in a sampling unit

n_i = number of individuals of the i^{th} species

p_i = species coverage

In this thesis, the calculation was done using the estimated species coverage instead of actually counting individual plants.

2.5 Methods of data analysis 2019

The biotopes were divided into 192 sample units by Pfeffer, (1992) and described qualitatively with respect to their biotope structure (mapping of biotope types) and quantitatively with respect to their width and area (Chapter 2.2). For 83 of these, plant communities were analyzed qualitatively with regard to species composition and coverage (Chapter 2.3.2).

In order to support the hypotheses, the data was first processed and indicator values calculated for later comparison. Pairwise comparisons using classical statistical tests were conducted.

The interpretation of the collected data required processing of the vegetation tables which was done, using R, the free software environment for statistical computing and graphics. All operations were done using R-Studio (RStudio, Inc., 2019).

Geodata was processed using the open source software QGIS.

2.5.1 Classical statistical tests

The biotope types where grouped into different biotope structure classes with the status of the biotope in 2019 as the determining factor: Hedges, semi-open landscape elements and grassland ecosystems in form of grass margins.

Table 3: Sample sizes in subsets of total sample (83 relevés)

Number of samples	all	unchanged (type)	changed (type)	constant in area	increased in area	decreased in area
all biotope classes	83	69	14	57	13	13
woods (hedges, semi-open)	46	32	14	33	10	3
hedges	36	24	12	24	9	3
semi-open	10	8	2	9	1	0
grass margins	37	37	0	24	3	10

The sample size varies depending on whether the entire data set was analyzed or – for differentiation between different biotope types – subsets of the same as shown in Table 3.
The 1992 and the 2019 data set were compared to see whether the biotopes were significantly different with respect to their size, number of species and mean ecological indicator value (L, T, F, R, K, N). For pairwise comparison the Wilcoxon rank test was chosen. The number of species in a data set are count data. A normal distribution cannot be expected, because of the nature of count data, where due to low count numbers occurring more frequently, the curve may lean to the left. (Dormann & Kühn, 2011) For the comparison between 2019 and 1992 the Wilcoxon signed rank test was chosen. It is to be used under the condition that the variables in the two data sets are symmetric. (Dalgaard, 2008) This is given since the values of each sampling unit in 1992 corresponds with the value of 2019.
Since the count numbers are discrete, the procedure in R undertakes continuity correction. For the given application it is not necessary to quantify the relationships between the data sets. This would be required to make predictions, for example. (Dormann & Kühn, 2011) Therefore a non-parametric test like Wilcoxon is suitable.

2.5.2 Experimental design

In order to derive information about the experimental design and its suitability to yield reliable results, the relationship between the size of the mapping unit and species number was analyzed as well as the number of sampling units.
In order to visualize the influence of the size of the sampling unit on the species number as described by Lomolino (2000), these two variables were plotted against each other. The three biotope classes are displayed in different colors to detect structure-related differences.
Due to the restrictions by the data previously available, which was sampled at two points in time and the decision to exclude biotope types which rarely occur, the maximum sample size was not larger than 83 samples. Instead of choosing the sample size according to the desired accuracy of results and the required reliability, the same is determined by the given sample size. (Traxler, 1997) Given that the data is not normally distributed and the variance was not known beforehand, the determination of the optimum sample size prior to the study is not a trivial thing to do, as it would require running a simulation. (Dormann & Kühn, 2011)
Therefore an analysis was done in retrospect in order to assess the sufficiency of the given sample size. The relationship between area and species number is expected to yield a sigmoidal curve. (Rosenzweig, 1995), (Lomolino, 2000) The marginal benefit of increasing the sample number was quantified by first randomizing the species lists. Then, the samples were drawn one by one and the number of new species on each list added to the number of

the previous list. The number of new species accumulated was plotted against the number of lists, yielding a saturation curve.

When the slope has reached a plateau, no new species are expected to be discovered by the analysis of additional samples.

3. Results

The results are displayed in this chapter based on the hypotheses and research questions introduced in Chapter 1.5 and the indicators and methods elaborated in Chapter 2.

3.1 Biotope extent - area and width

The width and area did not change significantly for the sampling units overall. The difference between the sum of biotope area is visible in the graph in Figure 18. Statistically, the change in total biotope area was not significant.

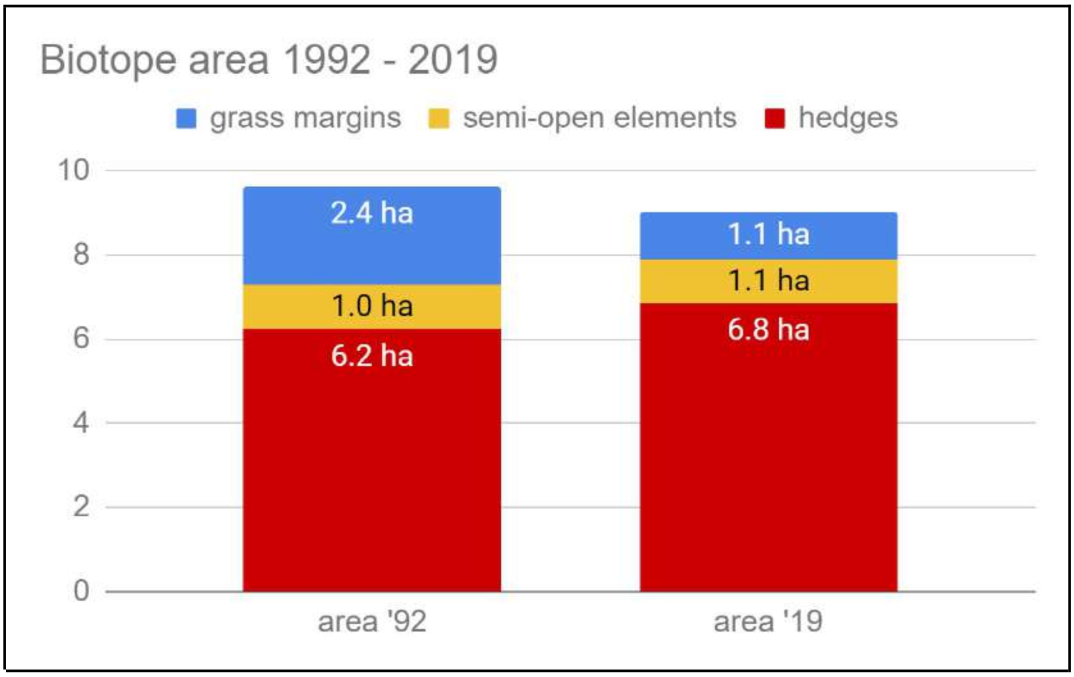

Figure 18: Biotope area 1992 and 2019, subdivided in the biotope classes of grass margins, semi-open elements and hedges

However, when the biotope classes grass margins, semi-open landscape elements and hedges are differentiated, a significant decrease was shown for grass margins (p-value ≈ 0.014) as shown in Figure 21, while the hedges increased significantly (p-value ≈ 0.017) as shown in Figure 20.

The comparison of the subset of grass margins which changed in structure did not yield significant results. The subset of grass margins which changed in structure was too small to apply the Wilcoxon test and yield significant results. The subset of unchanged margins also proved to be significantly different between the years (p-value ≈ 0.013) in terms of area. This, however could not be shown for subsets of hedges, neither for those which changed nor for those which did not change in structure.

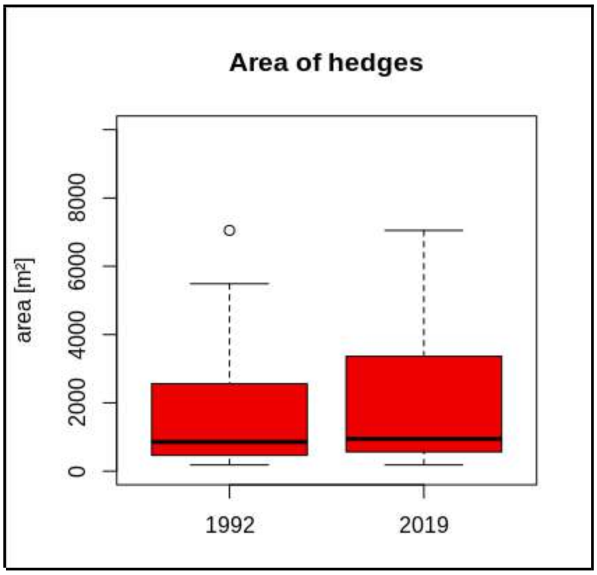

Figure 20: Changes in area of sampling units of the class hedges (n=36)

The width shows a significant change in favor of hedges (p-value ≈ 0.030). This biotope class has increased in width as shown in Figure 23. The same could not be detected for grass margins, Figure 22.

The data used in order to determine changes in width and area shows gaps as can be seen in Figure 24 and Figure 25. The biotope samples which show a change of zero, were below the detection level of the visual assessment.

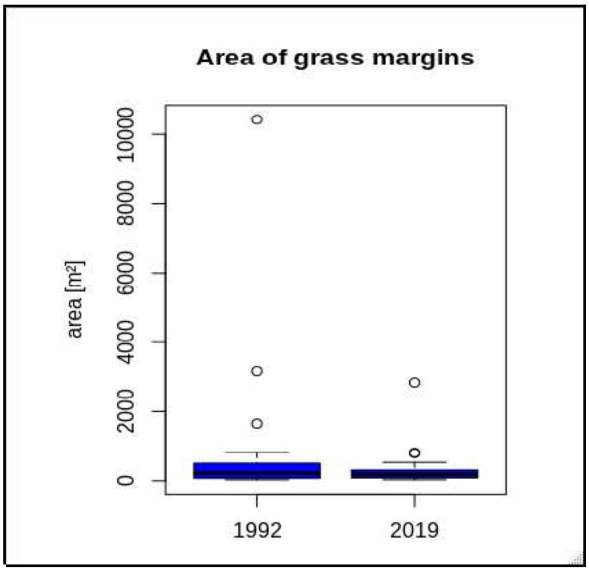

Figure 21: Changes in area of sampling units of the class grass margin (n=37)

The results presented in Figure 18 suggest that the area of semi open elements has slightly increased.

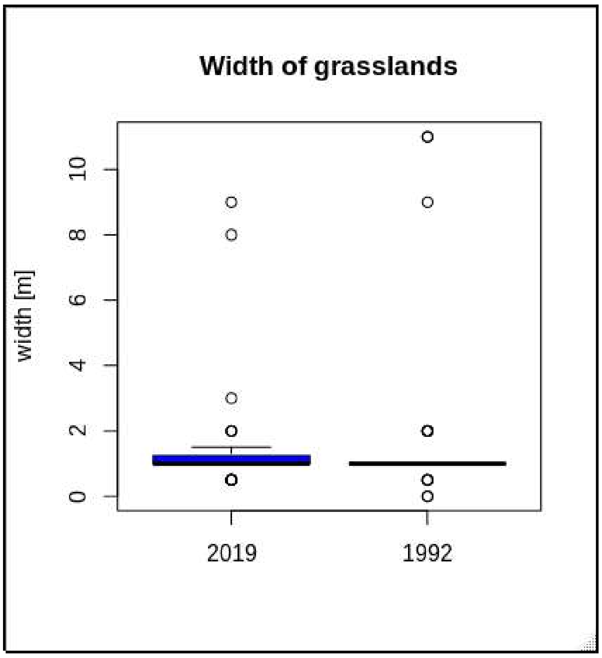

Figure 22: Changes in width of sampling units of the class grass margins (n=37)

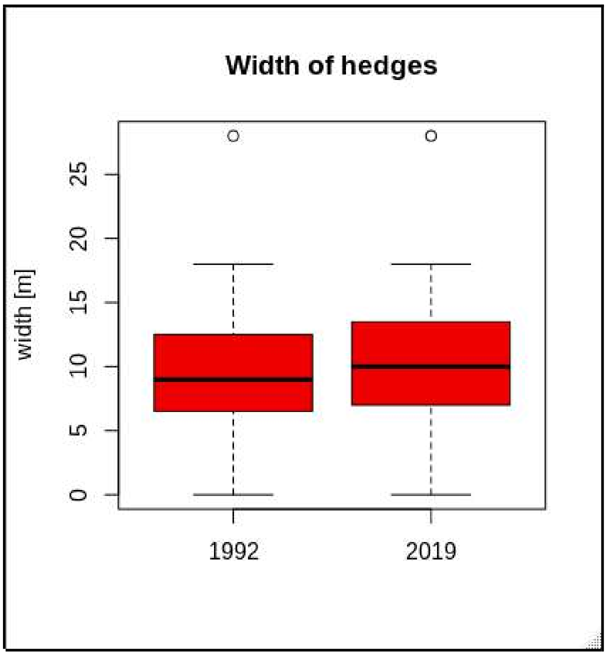

Figure 23: Changes in width of sampling units of the class hedges (n=36)

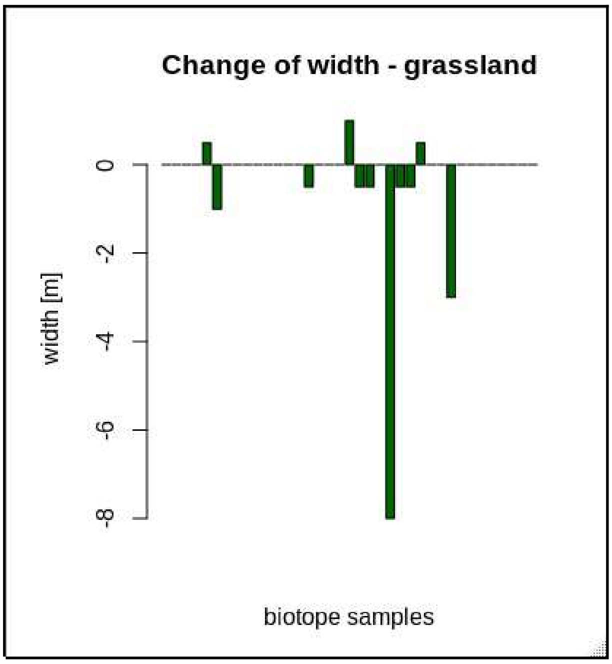

Figure 24: Sampling units in the class grass margins. Change in width (n=37)

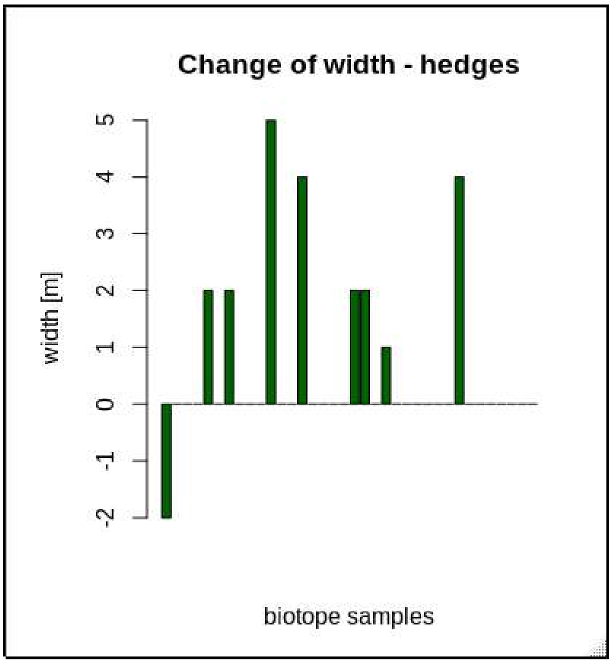

Figure 25: Sampling units in the class hedges. Change in width (n=36)

3.2 Biotope structure

The results of the previous section 3.1 show the combined effect of area loss and changes in biotope structure, which resulted in an expansion or decrease of biotope area.

In order to differentiate between the two, it is useful to look at the qualitative change in biotope structure separately as done in this chapter.

17% of all sample biotopes have encountered changes in their structure throughout the 27 years which have passed. Three types of changes have been identified: Densification, aging and shrub encroachment, depicted in Figure 26.

The term densification is used to describe the development of semi-open elements into dense hedges without considerable undergrowth in the form of grass, herbs and forbs.

Aging occurs in hedges, when a dense cover of shrubs grows into a biotope type where trees are more dominant. Shrubs are pushed back and the dense wood structures in the lower vegetation layers is broken up again. Shrub encroachment affects the grass margins and landscape elements where annual, low-growing, non-woody species, such as grass, herbs and forbs are dominant, turning them into semi-open landscape elements.

Shrub encroachment occurs when wood species start competing against the species in a grassland biotope.

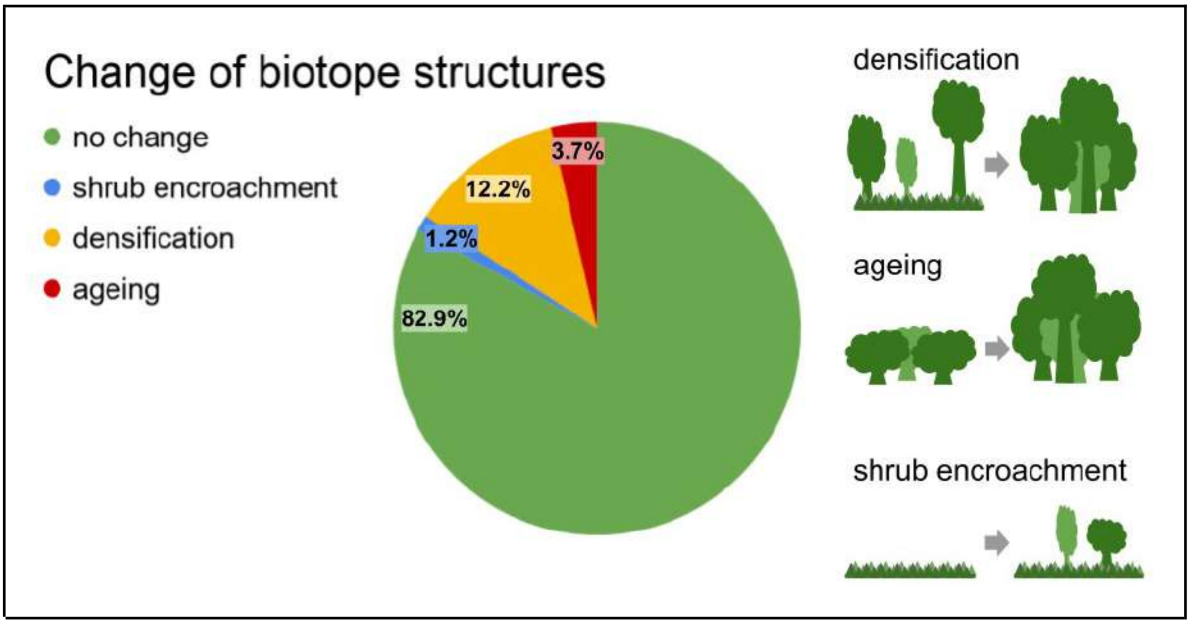

Figure 26: Changes in biotope structure due to succession and ageing which occurred between 1992 and 2019

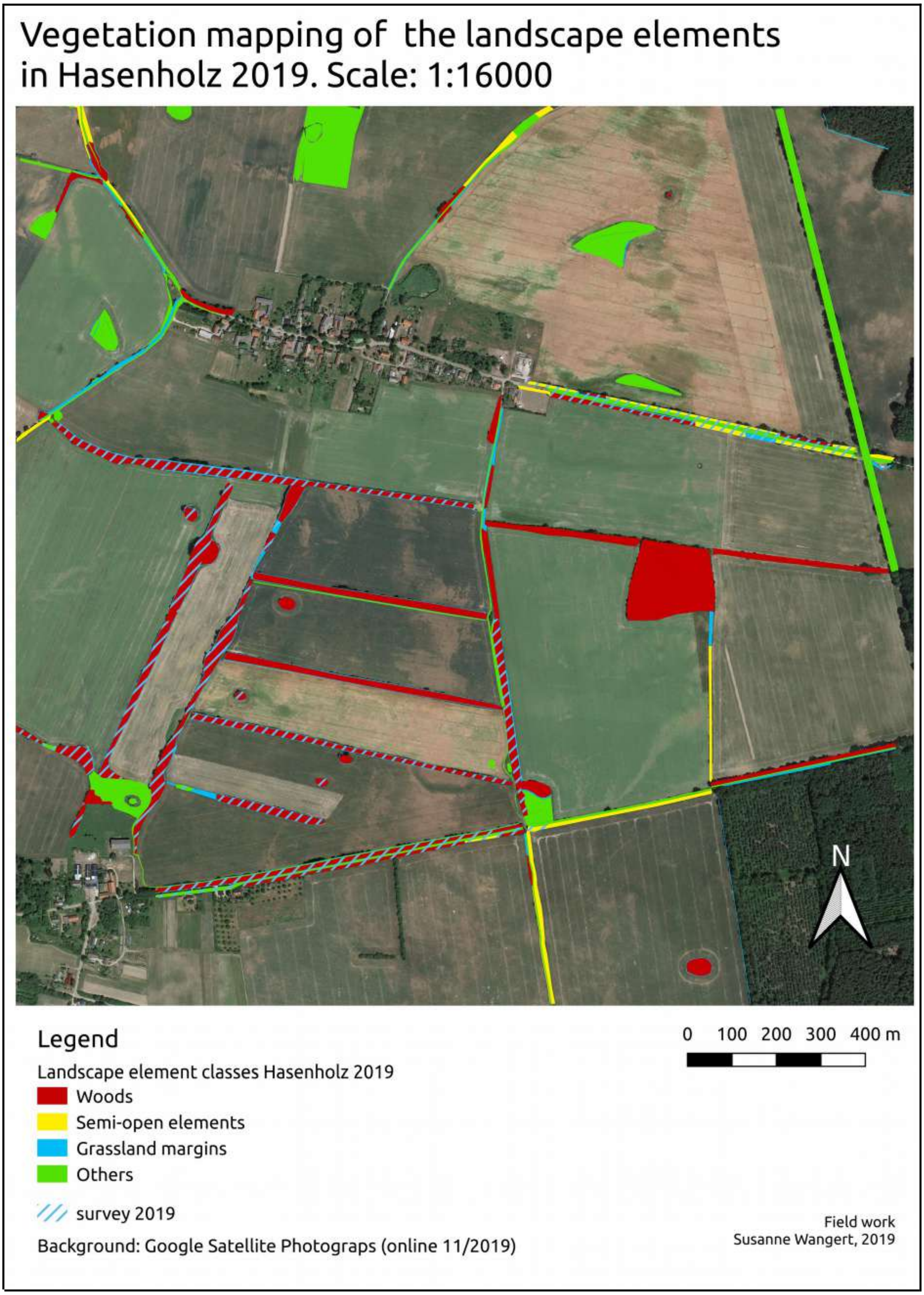

Figure 27: Overview map of landscape elements, their biotope classes and coverage of the detailed survey of flora in 2019

For 80 out of 83 sampling units area and width were fully documented by Pfeffer (1992). Out of the 80 samples, 30% showed a change in area, while 26% showed a change in area but no structure change. Therefore the change in area is regarded as independent from a change in structure. This is despite the fact that the increase and decrease in area results in an expansion and shrinkage of types of biotope structures (biotope types).

Of all structure changes shown in Figure 26, densification is the most frequent process with a share of 12.2%. It is followed by aging processes. 3.6% of the sample biotopes are affected by this process. 1.2% of the biotopes are grass margins which encounter shrub encroachment.

Figure 27 provides an overview over the biotope classes detected and analyzed in the survey area. The comparison of width, area and structure only took place in the striped area.

3.3 Species number (richness)

The species number recorded per biotope ranges between seven and 68 in 1992 and between eight and 82 in 2019. The outlier in area of grass margins in Figure 21 in Chapter 3.1 represents the highest species number found in the survey area. Excluding the single observations of plant species, which were found only once throughout the entire sample, the range comes down to seven to 64 in 1992 and eight to 75 in 2019.

In the sum of the samples, the average species numbers barely varied between the two years 1992 and 2019. However, the differentiation between biotope classes and those biotope samples which have changed in structure and area yields more detailed information.

The rounded average change in species number is displayed in Table 4:

Table 4: Mean change in species number in biotope classes and subsets.
Use of different methods for all biotope classes: All observations / single observations excluded

change in species number	all	unchanged (type)	changed (type)	constant area	increased in area	decreased in area
all biotope classes	0.5 / 0.3	0.8 / 0.7	-1.4 / -1.8	0.7 / 0.4	5.7 / 5.6	-5.4 / - 5.2
woods (hedges/semi-open)	3.5	***5.7**	-1.4	2.7	5.1	7.0
hedges	2.5	***5.7**	-3.8	1.3	4.2	7.0
semi-open	7.2	5.6	13.5	6.6	13.0	-
grass margins	-3.4	-3.4	-	-2.1	8.5	-8.2

*** = confidence level above 95%**

The analysis shows that overall, in all biotope types which hold wooden species, the average species number is increasing, unless biotope types have changed into such a biotope type. In contrast to this development the grass margins face losses, with the exception of the biotopes which have increased in area. Generally, an increase in area resulted in an increase in species.

Whenever the biotope structure changed, grass margins converted to semi-open elements host a higher species number. The further development of semi-open elements to hedges then would result in a loss of species.

Not all of these figures are statistically significant. Results with significance levels above 5% are printed in bold letters in Table 4.

The category of woods, which represent hedges and semi-open landscape elements combined, show significant changes of mean species numbers for those biotope samples for which the biotope type did not change over the years. This holds for hedges as well. The intuitive circumstance, that biotope samples which increased in area show higher species numbers could be shown for the total sample with sufficient significance of 95%.

The result is visualized in the following box plot, Figure 28:

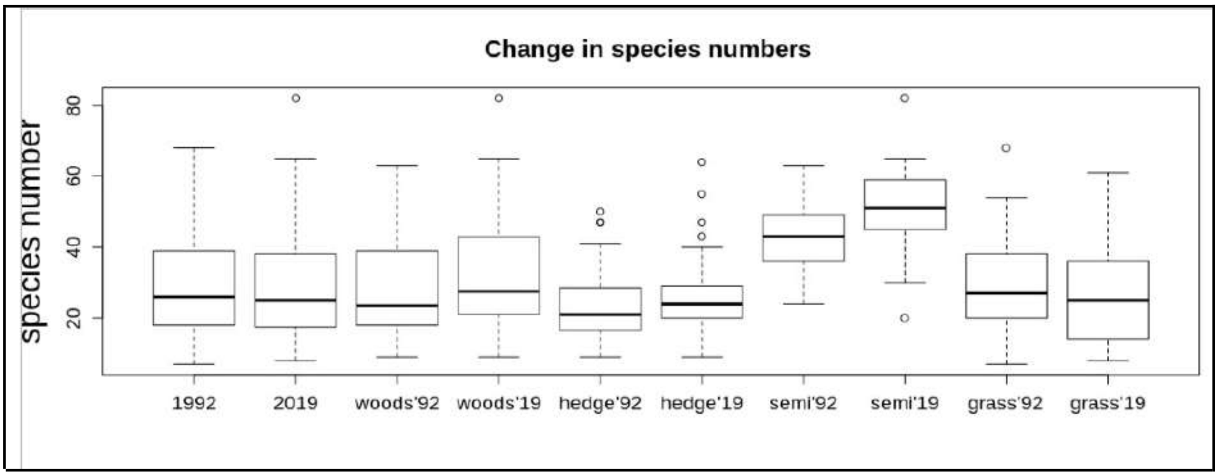

Figure 28: Change in species number in the sampling units overall (n=83) and in the biotope classes woods (semi-open elements + hedges, n=46), hedges (hedge, n=36), semi-open elements (semi, n=10), grass margins (grass, n=37).

The species numbers are displayed in the following graph, Figure 29. The species number for each year consists of the number of species exclusively found in each year and the species found in both years. The tall yellow bar shows the number of species found in both years. In order to reduce a potential error due to overlooked rare species, the same is displayed to the right without observations which were only made once, only for plants with a constancy above one („>1“).

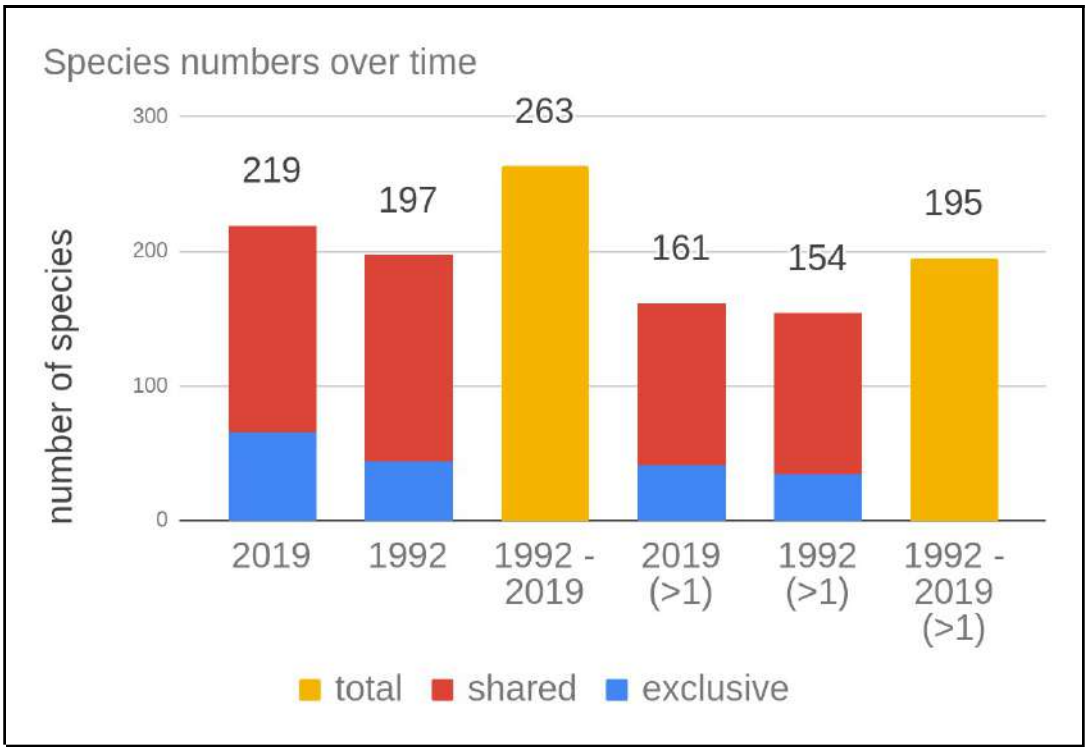

Figure 29: Species number observed in the sampling units (n=83) in total and excluding single observations (>1)

Based on these figures, the turnover was calculated as shown in the following.

3.4 Species turnover

The species turnover ranges between 30% and 79% in the individual biotopes. The average is 50% turnover. The turnover was calculated twice, once with the entire data set, once without the single observations in order to decrease a potential error due to different persons conducting the survey in 1992 and 2019. When single observations are neglected, the average turnover comes down to 48% as shown in Table 5:

Table 5: Average turnover derived from individual samples (n=83)

	total sample			>1x found		
	min	**max**	**ave**	**min**	**max**	**ave**
turnover	30%	79%	50%	29%	79%	48%
plants species on found in '92	0	35	14	0	31	13
plants species only found in '19	3	40	14	3	32	13
plants species number '92	7	68	29	7	64	28
plants species number '19	8	82	29	8	75	29
difference in plant species number	-21	29	0	-21	28	0

The overall turnover is 26% for the overall data set and 24% excluding the single observations, as calculated with the information shown in Table 6:

Table 6: Overall turnover within the total sample (n=83)

plant species found...	**species number**	**species number >1**
...only in '92	44	34
...only in '19	66	41
...in both years	153	120
...in total '92	197	154
...in total '19	219	161
...in total both years	263	195
total turnover	**26%**	**24%**

The following graph shows the different turnover rates for the three biotope classes grassland ecosystems/grass margins, semi-open elements and hedges.

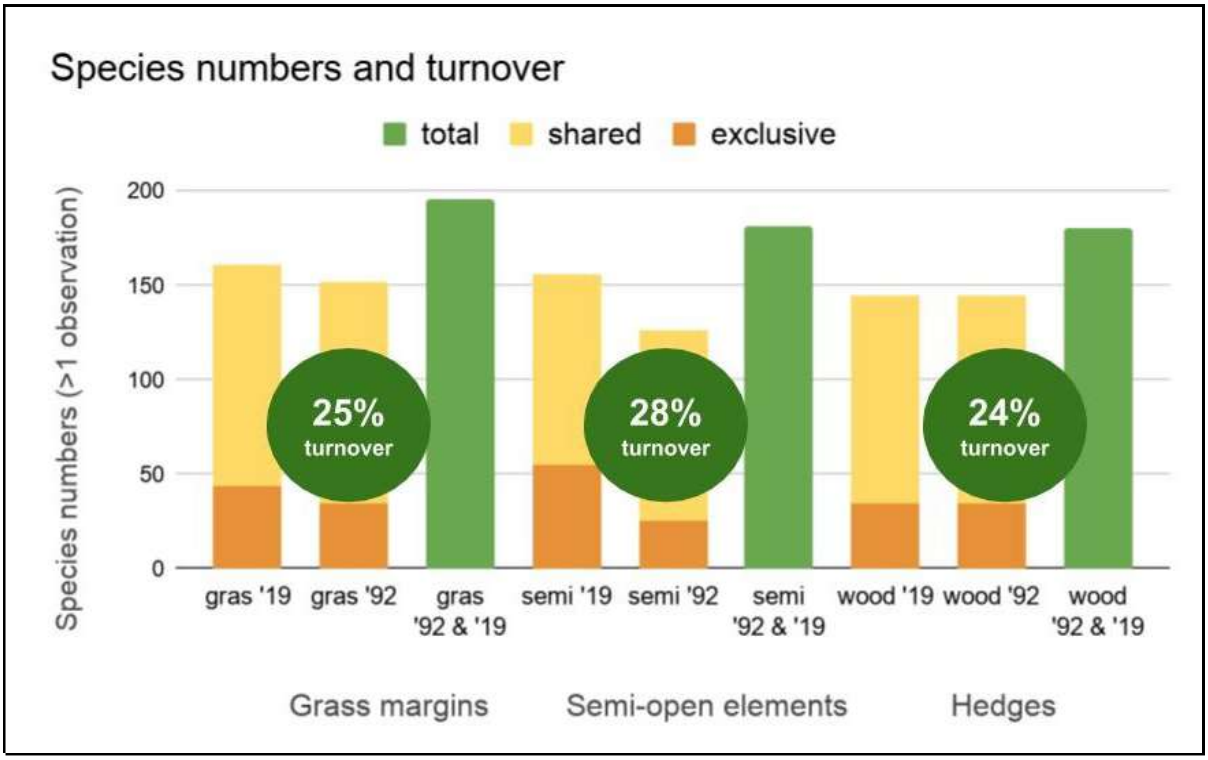

Figure 30: Turnover for different biotope classes, single observations excluded

The turnover was highest for the semi-open elements with 28%, the turnover for grass margins and hedges ('wood') was slightly lower with 24 and 25%. (Figure 30)

3.5 Species occurrence frequency (constancy)

The turnover shows that the spectrum of species has changed. More detailed information on the winners and losers is given in the following. The constancy has decreased for more than half the species. The highest decrease is more than 20 occurrences throughout the 83 sampling units, including those which have changed in area. For less than half the species, the constancy has increased up to 33 additional occurrences, for seven of them in particular.

The graph in Figure 31 shows the changes in constancy for all species arranged from losers to winners.

Top-15-lists for species constancy were created for all biotope units and separately for the three biotope classes, grass margins, semi-open areas and hedges, in order to eliminate the effect of their different coverage in area during the comparison.

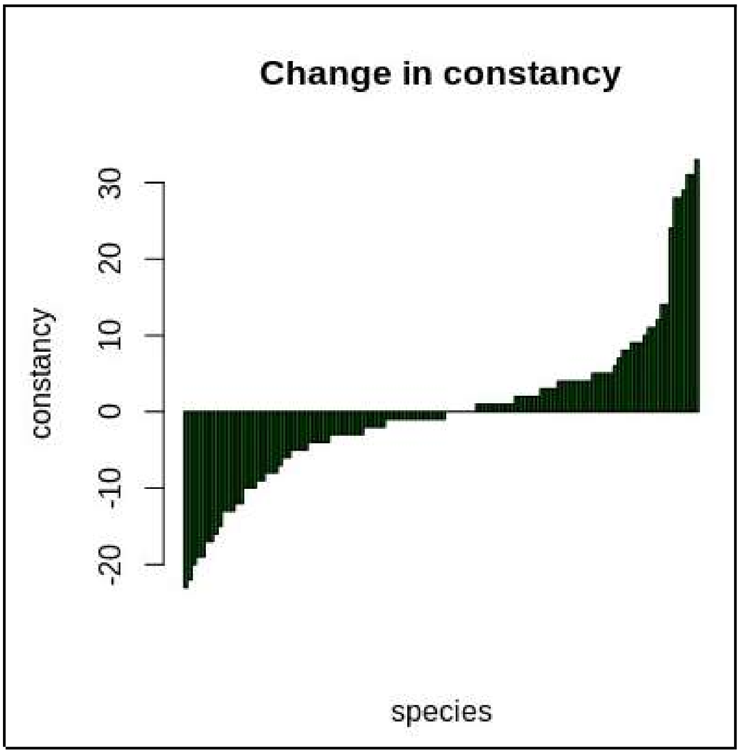

Figure 31: Difference in frequency of occurrence of the species in all sampling units in 1992 and 2019

Most frequently occurring plant species

New among the top 15 most frequently occurring plant species are *Fallopia convolvulus L., Geum urbanum L., Arctium lappa L., Rubus L. sp. and Alliaria petiolata (M.B.) Cav. et Gr*, while the species *Stellaria media (L.) Vill., Veronica hederifolia L., Rosa canina L., Convolvulus arvensis L., Lamium purpureum* L. are missing among the top 15 in 2019.

For **grass margins** *Capsella bursa-pastoris (L.) Med., Taraxacum sp., Achillea millefolium L.* and *Stellaria media (L.) Vill.* are now missing among the top-15 while *Silene latifolia Poiret subsp. alba (Mill.) Greuter et Burdet, Arctium lappa L., Fallopia convolvulus L.* and *Hypericum perforatum* L. took their place.

While for grass margins and hedges, the constancies of the top 15 species range between 15 and 35 occurrences because of the higher number of sampling units of that biotope type, for **semi-open elements** the counts reach 7-10 species. *Gagea pratensis (Pers.) Dumort, Rumex acetosa L., Dactylis glomerata L., Cichorium intybus L., Sambucus nigra L.* and *Anthriscus sylvestris (L.) Hoffm.* are missing in 2019 while *Geum urbanum L., Chenopodium album L., Plantago lanceolata L., Taraxacum sp., Prunus spinosa L.* and *Hypericum perforatum L.* did not occur in this upper range in 1992.

Artemisia vulgaris L., Elymus repens L. (Gould), Stellaria media (L.) Vill., Veronica hederifolia L., Euonymus europaeus L., Ballota nigra L., are species of the **hedges** which have disappeared from the top-15-list of species while *Alliaria petiolata (M.B.) Cav. et Gr, Rubus L. sp., Geum urbanum L., Geranium robertianum L., Galeopsis tetrahit L.* and *Lamium purpureum L.* have become more frequent.

Changes in constancy - 'losers'

Despite their status as one of the most frequently occurring species in the survey area, *Elymus repens L. (Gould)* and *Artemisia vulgaris L.*, have encountered the largest absolute losses. *Veronica hederifolia L.* which has disappeared from the top-15-list of most frequently recorded species is also one of the species which encountered largest absolute losses in species constancy.

Typical grassland species such as *Medicago sativa L., Dactylis glomerata L., Achillea millefolium L.* and *Knautia arvensis (L.) Coult. have shown major decline, as well as* ruderal species such as *Daucus carota L.*.

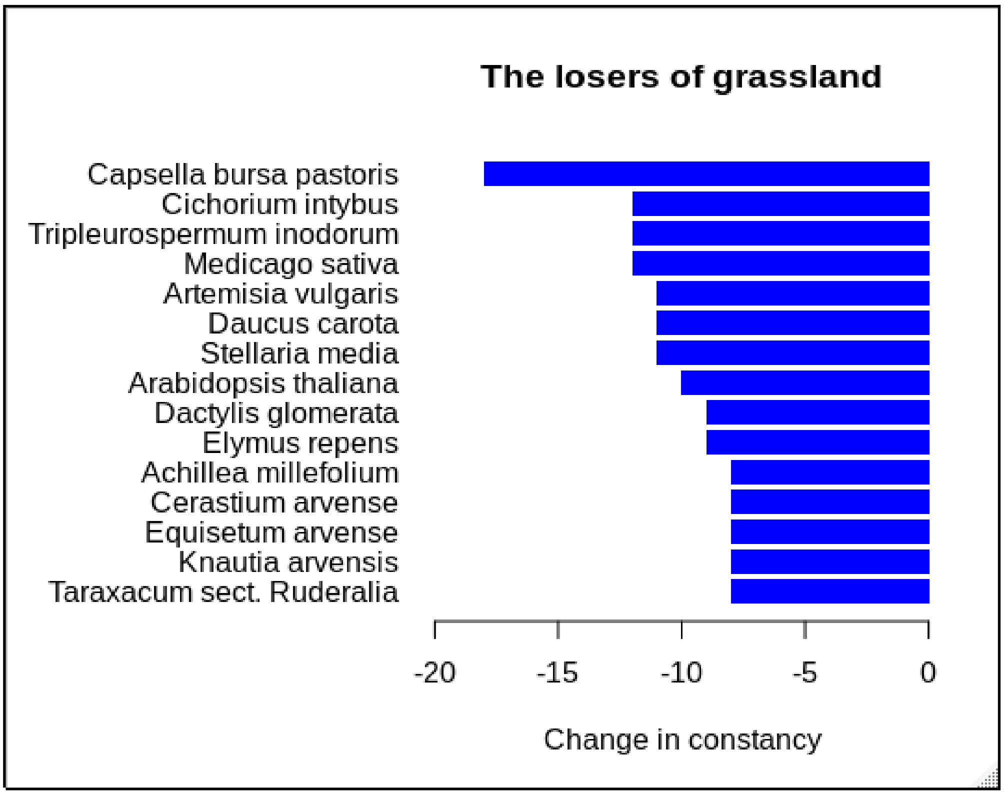

Figure 32: 15 plant species with the highest decreases in frequency of occurrence (constancy) in grassland biotopes

Elymus repens L. (Gould) encountered the largest losses in the biotope class of hedges by far, followed by light-dependent herbal species such as *Lamium album L., Medicago sativa L.* and *Cichorium intybus L.*.

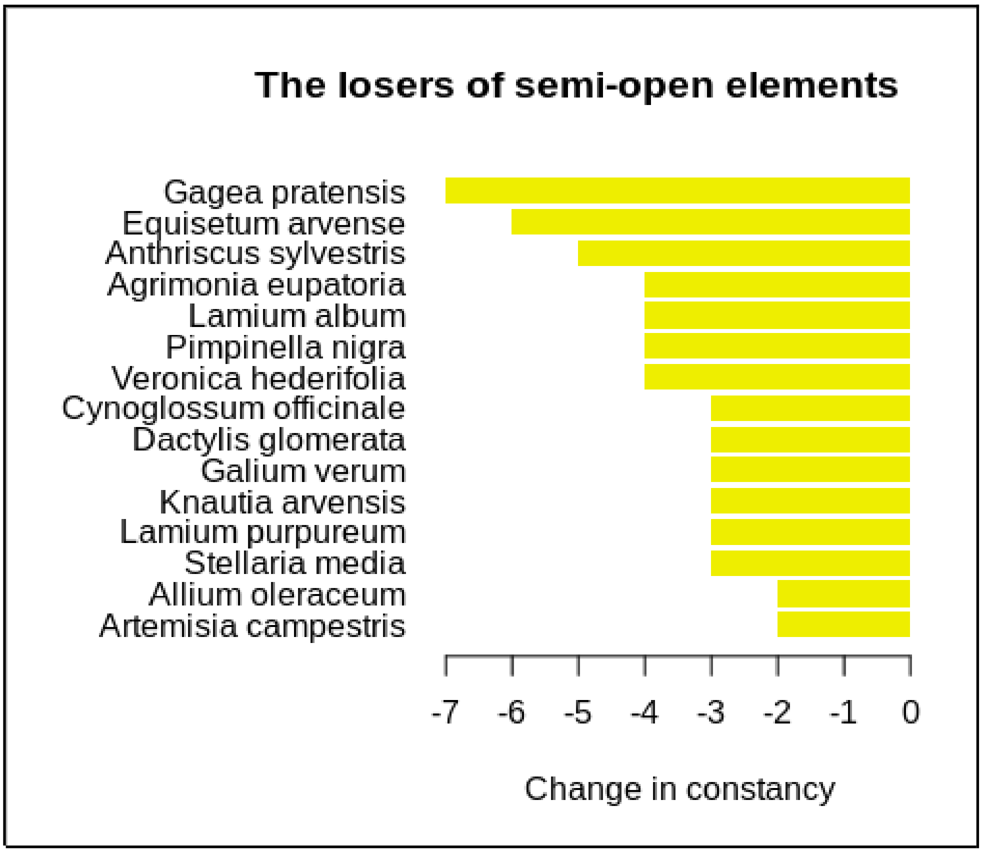

Figure 33: 15 plant species with the highest decreases in frequency of occurrence (constancy) in semi-open elements

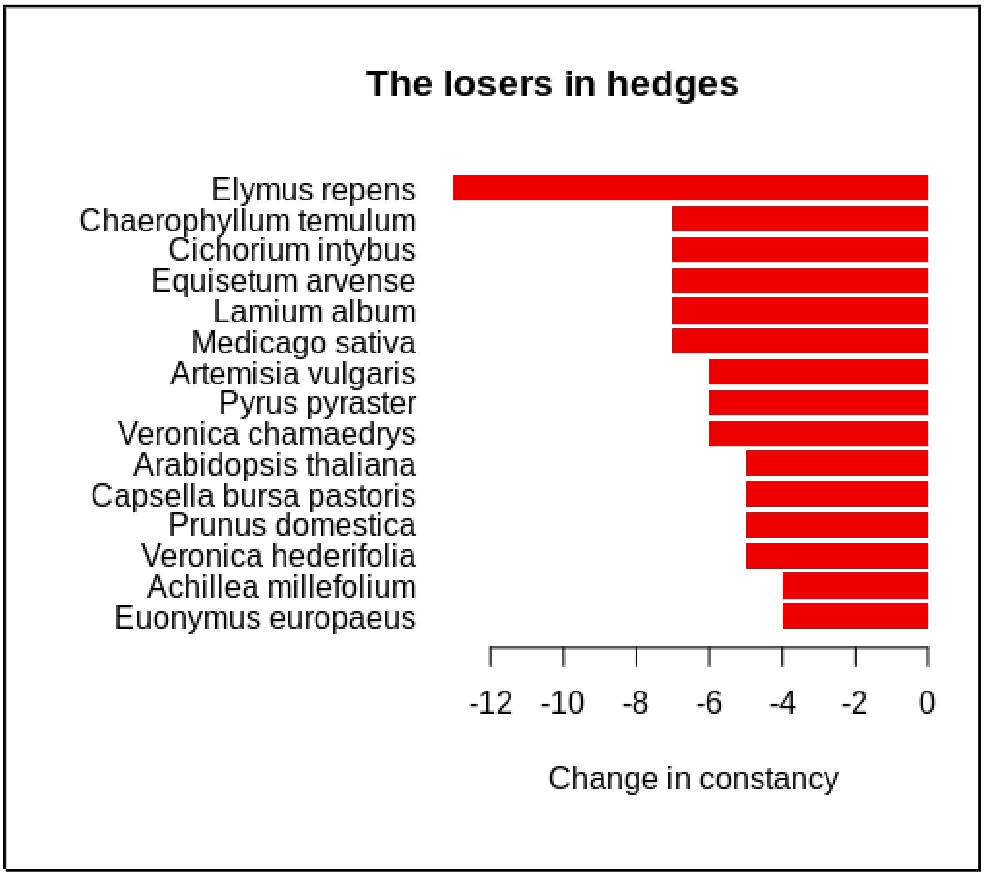

Figure 34: 15 plant species with the highest decreases in frequency of occurrence (constancy) in hedges

The bar plots in Figure 32, 33 and 34 show the 15 plant species that have decreased in frequency of occurrence the most in the three biotope classes.

Changes in constancy - ‘winners’

Species which have increased disproportionately in frequency of occurrence across all biotope classes are *Geranium robertianum L., Fallopia convolvulus L., Rubus L. sp., Arctium lappa L., Galeopsis tetrahit L., Geum urbanum L.* and *Alliaria petiolata (M.B.) Cav. et Gr.* They are all found among the top-15 winners as shown in Figure 35.

Ballota nigra L., Bromus sterilis L. and *Potentilla reptans L.* already occur frequently in grassland and have increased in constancy.

For semi-open elements the constancy values are lower, but spread across species more evenly: The dominance of individual plant species is lower. The species *Geum urbanum L.* and *Chenopdium album L.* are the only winners in the class of semi-open elements, which have increased in constancy while already occurring among the top-15 most frequent plants in semi-open elements.

In the biotope class of hedges, among the top-15 winners are *Geranium robertianum L., Alliaria petiolata (M.B.) Cav. et Gr, Galeopsis tetrahit L., Geum urbanum L.* and Rubus L. sp. which are already present in the top-15 on the list of highly frequent species. Other species like *Hedera helix L.* newly occur.

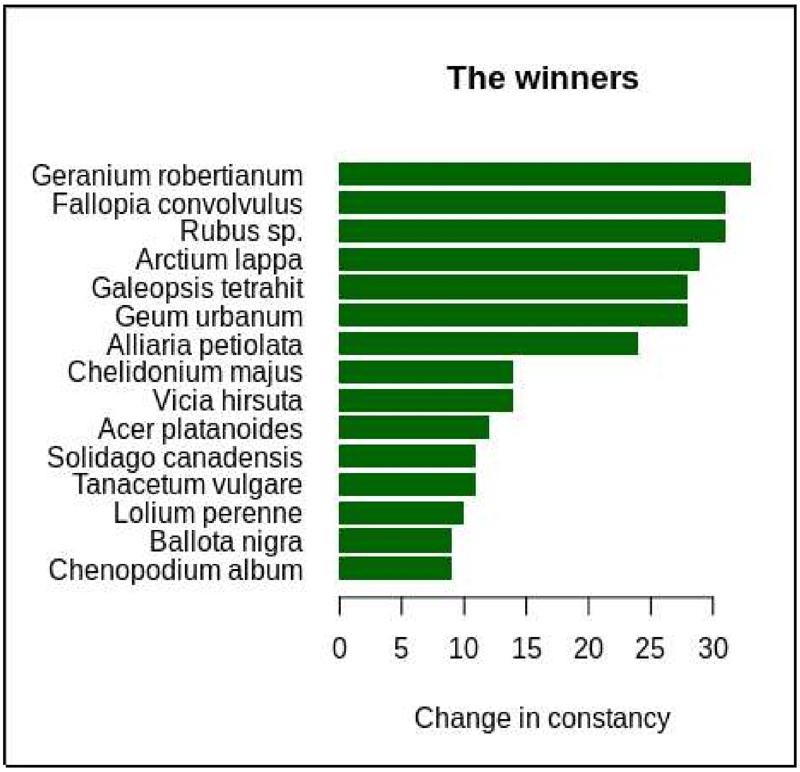

Figure 35: 15 plant species with the highest increases in frequency of occurrence (constancy) in all biotope classes

The newcomers and lost species are listed in the following Table 7.

Table 7: List of newly occurring (newcomer) and lost species.
*(?) = Determination uncertain * = Single observation*

Newcomer species	Lost species
Acer campestre L.	*Alopecurus geniculatus L.**
Acer negundo L.	*Amaranthus retroflexus L.**
*Allium sp ('schoenoprasum L.') (?)**	*Arabidopsis thaliana (L.) Heynh.*
*Anthemis arvensis L.**	*Armeria maritima subsp. elongata (Hoffm.) Koch**
*Aphanes arvensis L.**	*Betula pendula Roth*
*Apiaceae (?)**	*Campanula trachelium L.**
*Aquilegia vulgaris L.**	*Carduus crispus L.*
*Arctium tomentosum Mill.**	*Carex hirta L.*
*Arenaria serpyllifolia L.**	*Centaurea jacea L.*
*Asparagus officinalis L.**	*Cirsium oleraceum (L.) Scop.**
*Centaurea stoebe L.**	*Descurainia sophia (L.) Webb**
Chenopodium praticola Rydb.	*Echinochloa crus-galli (L.) P.B.**
Clematis vitalba L.	*Epilobium hirsutum L.**
Crepis capillaris (L.) Wallr.	*Festuca gigantea (L.) Vill.**
Dactylis polygama Horvatovszky	*Festuca ovina L.**
*Erigeron annuus L.**	*Festuca pratensis Huds.*
*Fallopia japonica (Houtt.) Ronse Decr.**	*Ficaria verna Huds.*
Festuca rubra L.	*Gagea pratensis (Pers.) Dumort*
*Fragaria viridis Duch.**	*Gagea villosa (M. Bieb.) Sweet**
*Galeobdolon argentatum Smejkal**	*Helianthus annuus L.**

Geranium dissectum Jusl.	*Hieracium lachenalii C. Gmel.**
Hedera helix L.	*Holcus mollis L.**
Helianthus tuberosus L.	*Knautia arvensis (L.) Coult.*
*Helichrysum arenarium (L.) Moench**	*Lamium amplexicaule L.*
Hylotelephium maximum (L.) Holub.	*Leucanthemum vulgare (Vaill.) Lam.**
*Hypochaeris radicata L.**	*Lolium multiflorum Lam.*
Impatiens parviflora L.	*Persicaria amphibia mod. terrestris (L.) Delarbre*
*Iris sp.**	*Peucedanum oreoselinum (L.) Moench**
*Juglans regia L.**	*Phalaris arundinacea L.**
*Lathyrus latifolius L.**	*Poa trivialis L.*
*Lathyrus pratensis L.**	*Potentilla anserina L.*
Ligustrum vulgare L.	*Primula veris L.**
Malva neglecta Wallr.	*Prunus serotina Ehrh.**
*Medicago lupulina L.**	*Scorzoneroides autumnalis (L.) Moench**
Medicago x varia	*Senecio vernalis W. et Kit.*
*Melilotus albus Medik.**	*Senecio vulgaris L.*
Moehringia trinervia (L.) Clairv.	*Solanum dulcamara L.*
Myosotis arvensis (L.) Hill.	*Sonchus arvensis L.**
Oenothera parviflora L.	*Sorbus aucuparia L.**
Ononis repens L.	*Trifolium medium L.*
*Papaver dubium L.**	*Tripleurospermum inodorum (L.) Sch. Bip.*
*Parthenocissus inserta (Kern.) Fritsch**	*Turritis glabra L.*
Picris hieracioides L.	*Viburnum opulus L.**
*Plantago major subsp. intermedia (DC.) Lange**	*Vicia villosa Roth**
*Poa nemoralis L.**	
*Pteridium aquilinium (L.) Kuhn**	
Reseda lutea L.	
*Rumex acetosella L.**	
*Salix caprea L.**	
Saponaria officinalis L.	
*Scilla siberica Haw.(?)**	
Setaria viridis (L.) P.B.	
*Sonchus asper (L.) Hill.**	
Sonchus oleraceus L.	
*Spergularia rubra (L.) Presl.**	
*Symphytum L. sp.**	
*Taxus baccata L. **	
Tilia cordata Mill.	
Trifolium campestre Schreb.	
*Trifolium dubium Sibth.**	
Verbascum lychnitis L.	
Verbascum nigrum L.	
Veronica arvensis L.	
*Vicia angustifolia L.**	
Vicia parviflora Cav.	
*Vitis vinifera L.**	

3.6 Mean Ellenberg indicator values (MEIV)

The MEIV were tested against the null hypothesis H_0, that 1992 and 2019 show no differences.

Mean Ellenberg indicators taking into account the plant coverage by weighting ($MEIV_w$) were mostly stable over time as H_0 could not be rejected for L, T, N and R. The F-value for moisture and K-values for continentality however, pose an exception, both have significantly decreased. The $MEIV_e$ were calculated by taking the average of all indicator values attributed to the individual plants with each of them given equal weight as described in 2.4.4. This qualitative assessment, showed significant change for moisture (F) and light (L). For the average continentality (K)-value the change was below the significance level of 95% for the MEIV taking only plants into account, which occurred more than once in the survey area.

The significance of the results are displayed in the following Table 8.

Table 8: Average change of MEIV and significance level as results of a comparison between $MEIV_w$ and $MEIV_e$ of 1992 and 2019, using the Wilcoxon signed rank test.

	$MEIV_w$	**$MEIV_e$**	**$MEIV_e$, no single observations**
F	-0.18, 99.5%	no significant change	no significant change
K	-0.25, 97.4%	no significant change	(-0.06, 93.5%)
L	no significant change	-0.09, 99.1%	-0.09, 99.5%
N	no significant change	no significant change	no significant change
T	no significant change	no significant change	no significant change

Across all indicator values, $MEIV_w$ show a larger variance than the $MEIV_e$.

3.6.1 F-value (moisture)

The $MEIV_w$ for moisture (F, German: ‘Feuchte’) of 1992 and 2019 are significantly different from each other, with a confidence level of 99.5%. This however, does not hold true for a

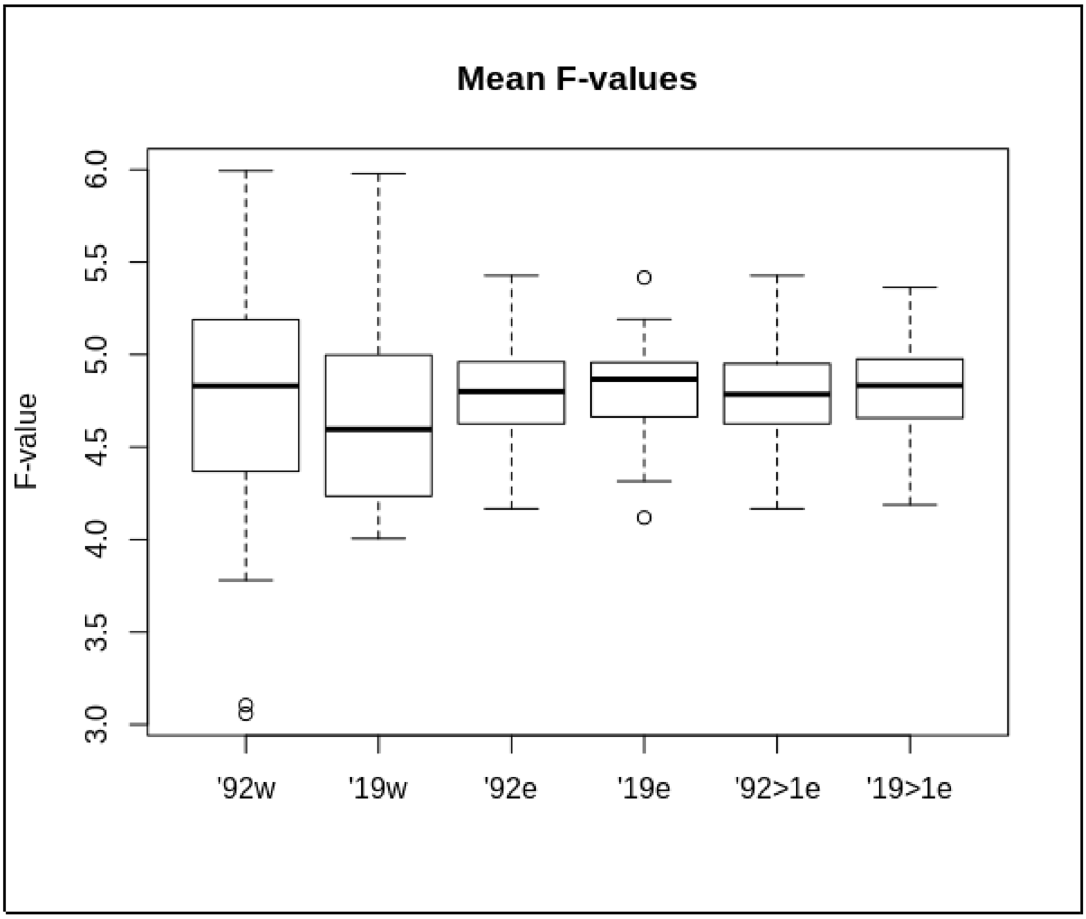

Figure 36: Mean F-values calculated as $MEIV_w$ (‘92w, ‘19w), $MEIV_e$ (‘92e, ‘19e) and $MEIV_e$ for plants with a constancy above 1 (‘92>1e, ‘19>1e)

comparison between the $MEIV_e$ of 1992 and 2019, no matter whether single observations are taken into account or not as shown in the box plot below (Figure 36).

The values range between four and six, as can also be seen in the scatterplot below (Figure 37). It shows the $MEIV_e$ for F of 2019 plotted against 1992 and shows the weak linear correlation between the values of 1992 and 2019.
The distribution in this case is independent from the biotope class.

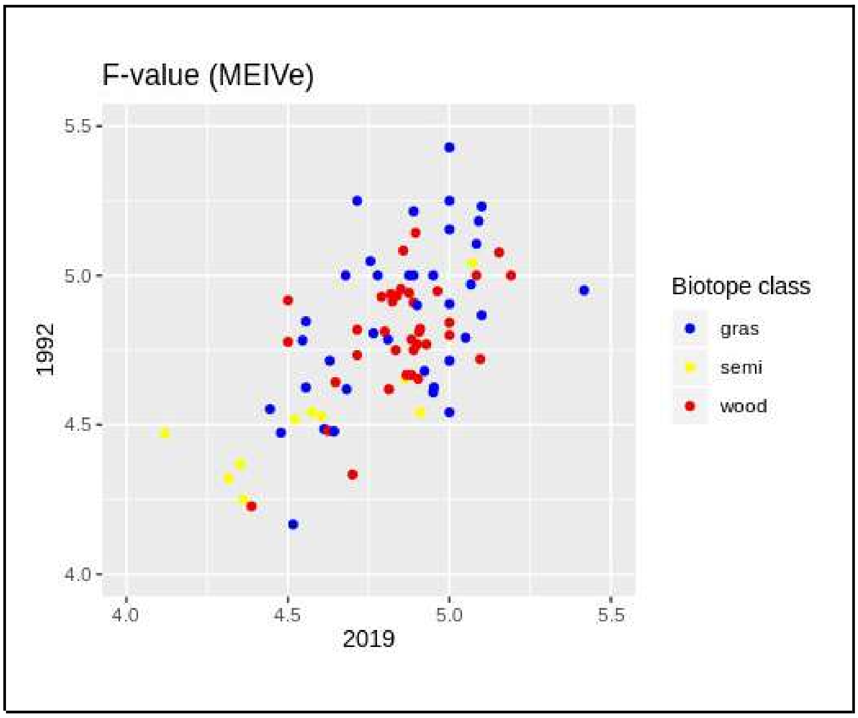

Figure 37: Mean Ellenberg F-value of biotope samples of 2019 plotted against 1992 ($MEIV_e$)

3.6.2 K-value (climate, C-value)

The accumulated K-value (or C-value, German: ‘Klima’) changed significantly between the years 1992 and 2019, when calculated by weighting the indicator values according to the plant coverage ($MEIV_w$). This could be shown with a confidence level of 97.4%.

The values for the $MEIV_w$ are higher than those of the $MEIV_e$.as displayed in Figure 38.

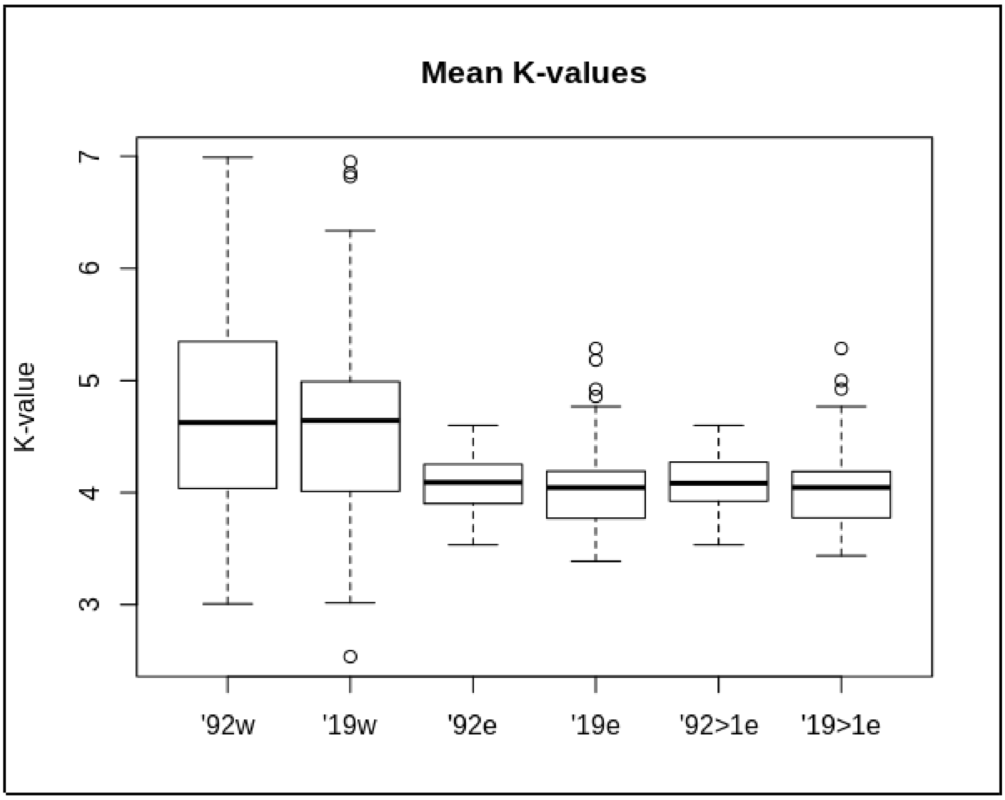

Figure 38: Mean K-values calculated as $MEIV_w$ (‘92w, ‘19w), $MEIV_e$ (‘92e, ‘19e) and $MEIV_e$ for plants with a constancy above 1 (‘92>1e, ‘19>1e)

While the data shows a large variation overall, the $MEIV_w$ in the upper range have decreased, which can also be detected visually in the scatterplot, in Figure 39. In contrast to the findings in the calculation by weighting the plants equally, the average indicator value decreased by 0.25 units to 4.63 in 2019.

The Wilcoxon test, comparing the $MEIV_e$ however, did not reveal a significant difference between the two groups of 1992 and 2019.

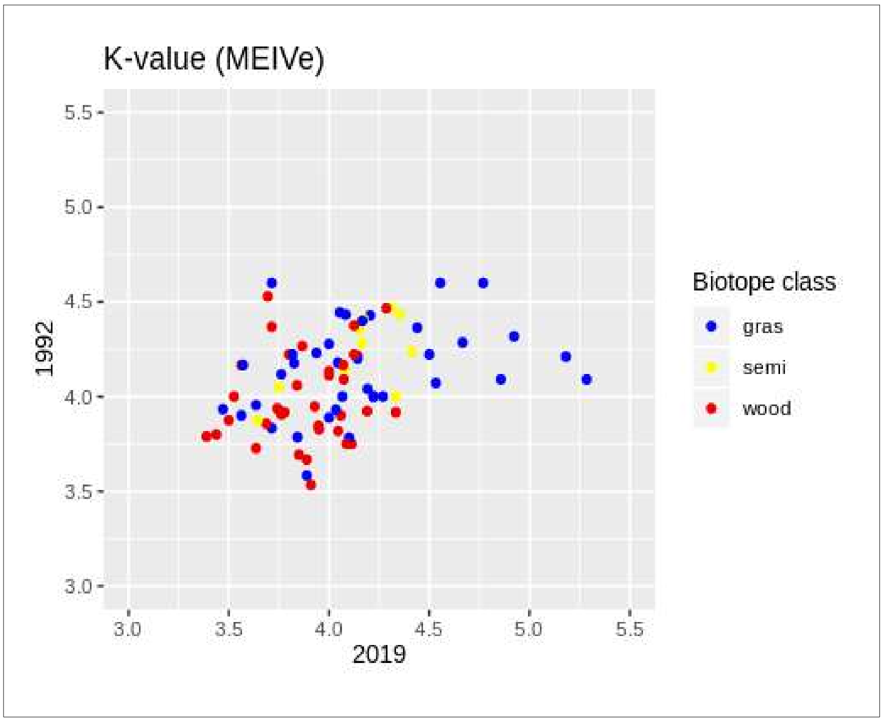

Figure 39: Mean Ellenberg K-value of biotope samples of 2019 plotted against 1992 (MEIV$_e$)

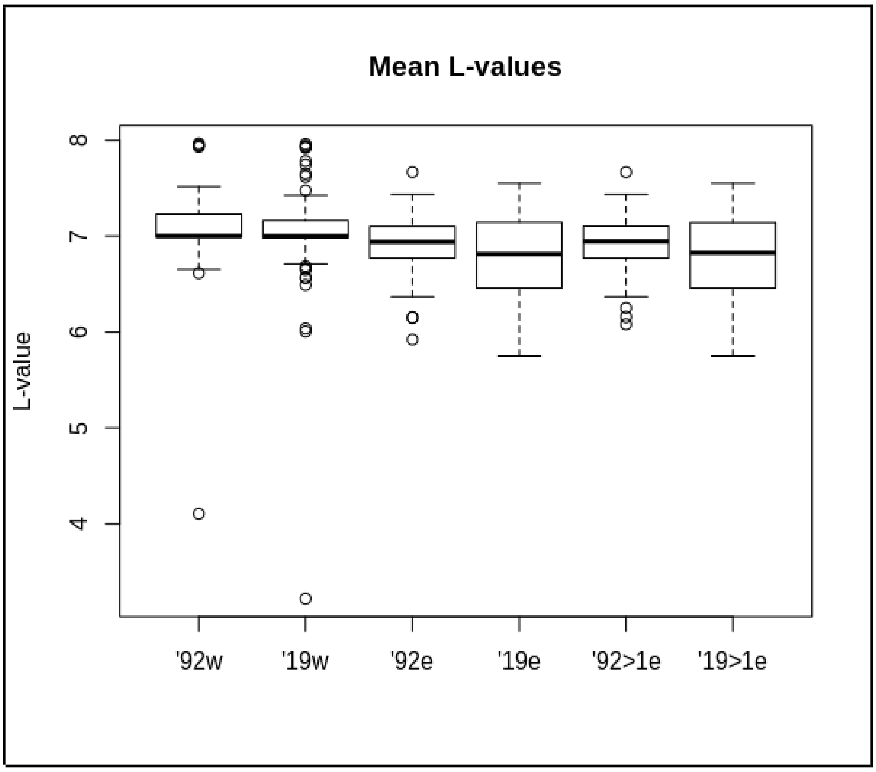

Figure 40: Mean L-values calculated as MEIV$_w$ ('92w, '19w), MEIV$_e$ ('92e, '19e) and MEIV$_e$ for plants with a constancy above 1 ('92>1e, '19>1e)

3.6.3 L-value (light)

While for the MEIV$_w$ for light no significant difference was found, for the MEIV$_e$ a decrease was detected with a significance level of 99.0%.

The scatter, visualized in Figure 40 is substantially smaller than for the MEIV$_w$ previously described. Like for the T-value the light value is different depending on the biotope class: While more light dependent species live in the grass margins, the less light dependent species are found in hedges (Figure 41).

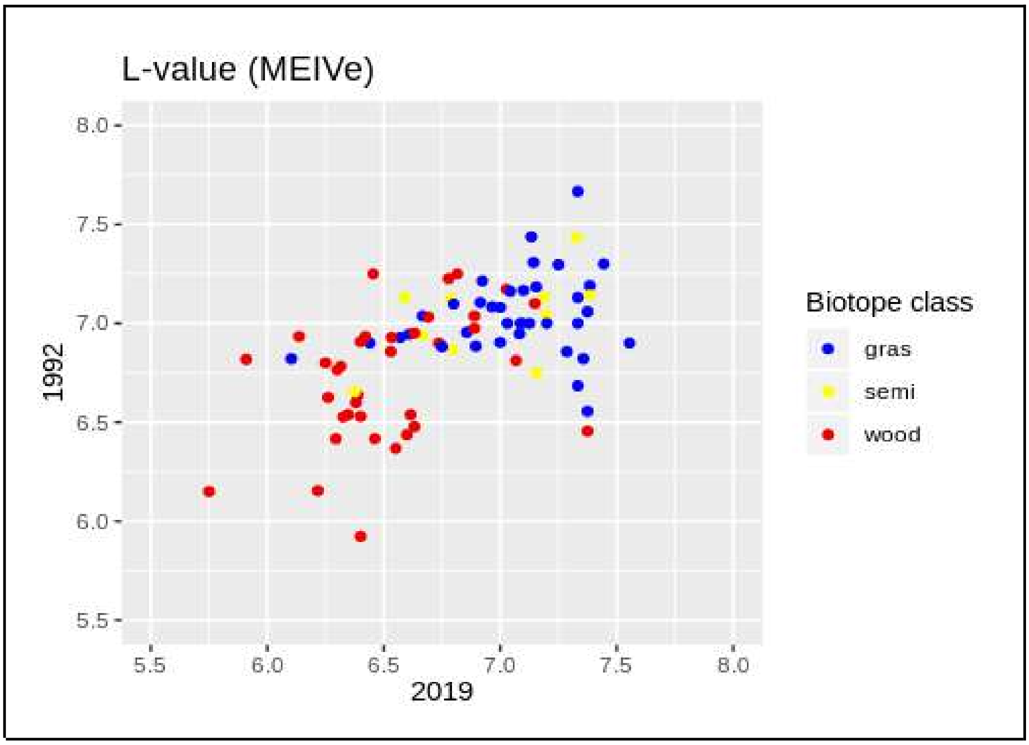

Figure 41: Mean Ellenberg L-value of biotope samples of 2019 plotted against 1992 (MEIV$_e$)

3.6.4 N-value (nitrogen, nutrients)

No significant difference could be detected between the years 1992 and 2019.

However, the scatterplot shows that compared to the sampling units in the class wood and grass margins, the semi-open elements show lower mean N-values (Figure 42).

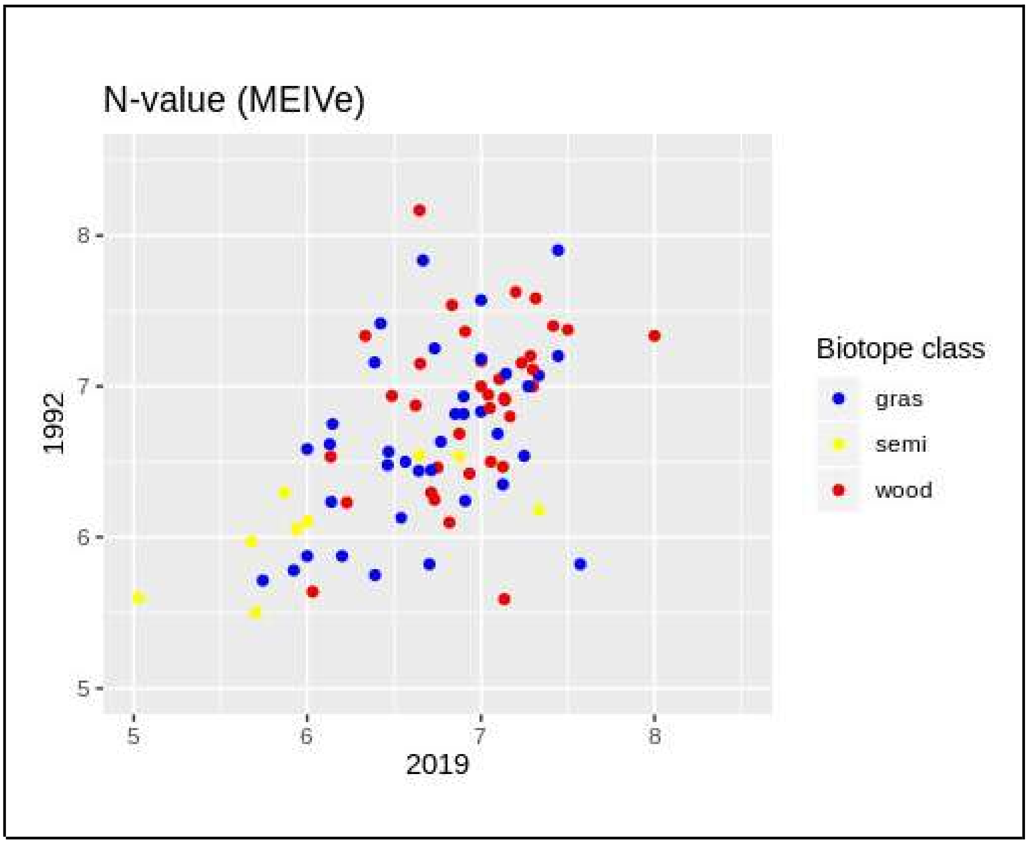

Figure 42: Mean Ellenberg N-values of biotope samples of 2019 plotted against 1992 (MEIV$_e$)

3.6.5 T-value (Temperature)

Figure 43 shows the distribution of mean T-values in the year 1992 and 2019 plotted against each other. The plot shows a tendency of higher average T-values for the grassland biotopes and lower values for the plants associated with wood structures.

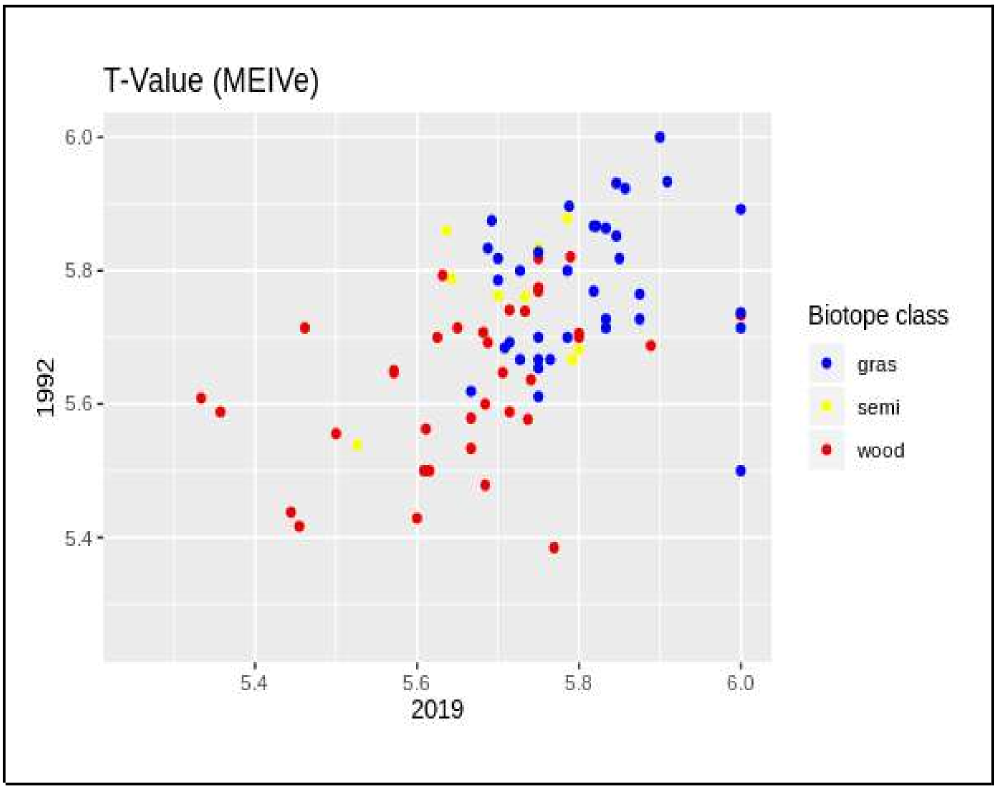

Figure 43: Mean Ellenberg T-values of biotope samples of 2019 plotted against 1992 (MEIV$_e$)

The null hypothesis, that the findings in 1992 are not different from those in 2019 could be not be rejected either for the MEIV$_e$ nor for the MEIV$_w$. Temperature was not identified as significantly different.

3.6.6 MEIV of winner and loser plants

Based on the species list of 'winners' and 'losers' of constancy, another comparison was done between the MEIV of these two groups. The detailed results on the constancy in the different biotope classes are elaborated in Chapter 3.5. Using the Wilcoxon test once again, an unpaired test was run, in order to compare the group of winners with the group of losers with respect to the indicator values. The assumption: When significant differences between the year 2019 and 1992 were shown, using the Wilcoxon test as described in 3.6, the same must yield a similar outcome in a comparison between the species which have increased in constancy and those which have decreased. It could not be shown, that there are significant differences between winners and losers with regard to all MEIV. The result, that the K-value has changed significantly could not be reproduced, neither could the same be shown for L and F.

The number of data points for the comparison of winners and losers for the K-value were 52 and 61 data points of which up to one third of the values was not available ('NA'). Therefore for K, the data set of winners and losers which could be used for the analysis had the size of n=44 and n=39. For F the size was n=40 and n=50, for T n=44 and n=33 and for N n=42 and n=49.

The comprehensive list of winners and losers and their EIV is accessible in appendix 9.1.

The plotting of the difference between averages of top winners and top losers as described in Chapter 2.4.3 yielded visible results for some indicator values as shown in Figure 44. For R,

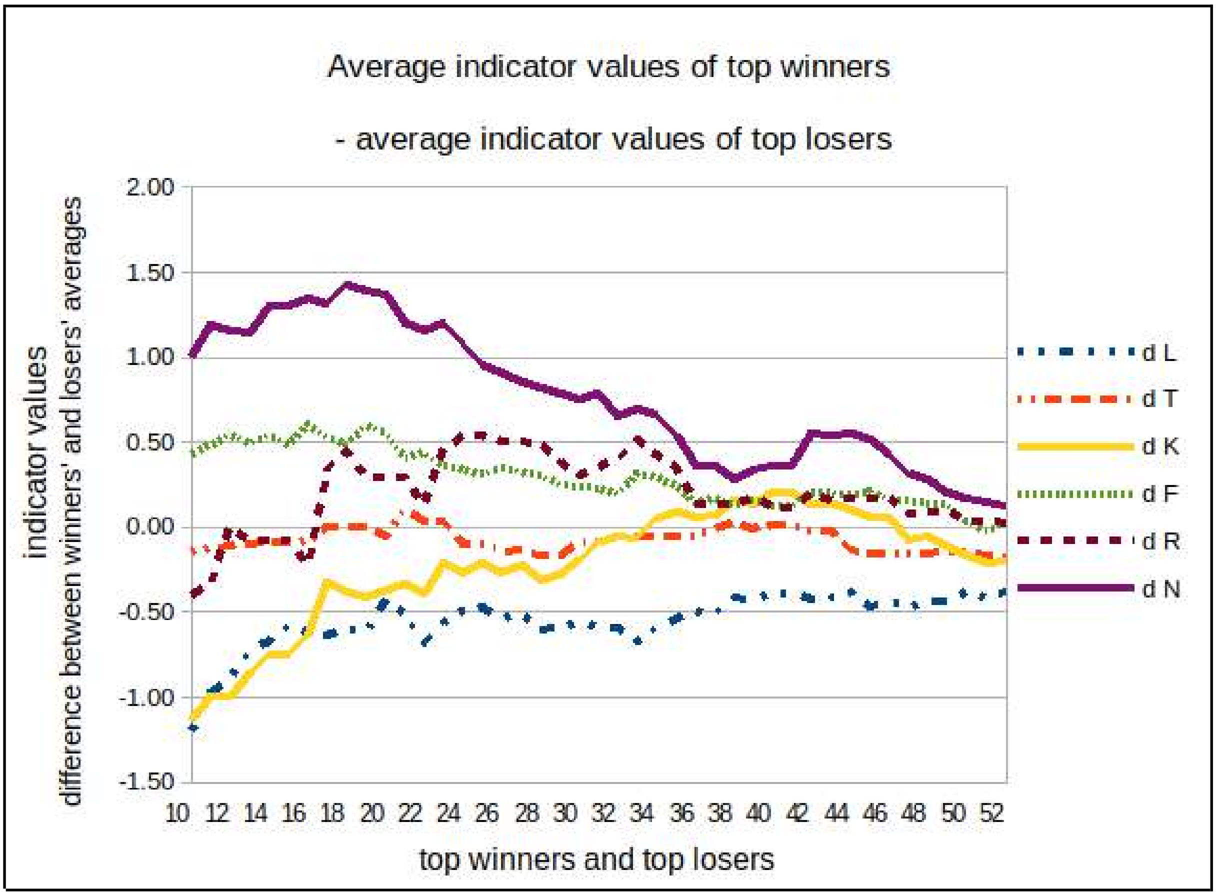

Figure 44: Differences between averages of MEIV of top 10 to top 52 winners and averages of MEIV of 10 to 52 top losers

T and K the curve showing the difference between the averages of top 10 to top 52 winners and losers oscillated around zero and no clear direction is visible.

While for K the difference between the MEIV of the top 10 winners and the MEIV of the top 10 losers shows a difference of one unit, it drops to zero for the MEIV of the top 30 winners. For the L value, the difference between the averages is constantly below zero, while for N and F the opposite holds true and both are above zero, indicating that the winners show higher averages in indicator values than the losers groups from 1-10 up to 1-52.
The average indicators of newcomers and disappearances are shown in the following graph (Figure 45).
A decrease of the F- and the N-value by half a unit is shown and a small increase for L, K and R.

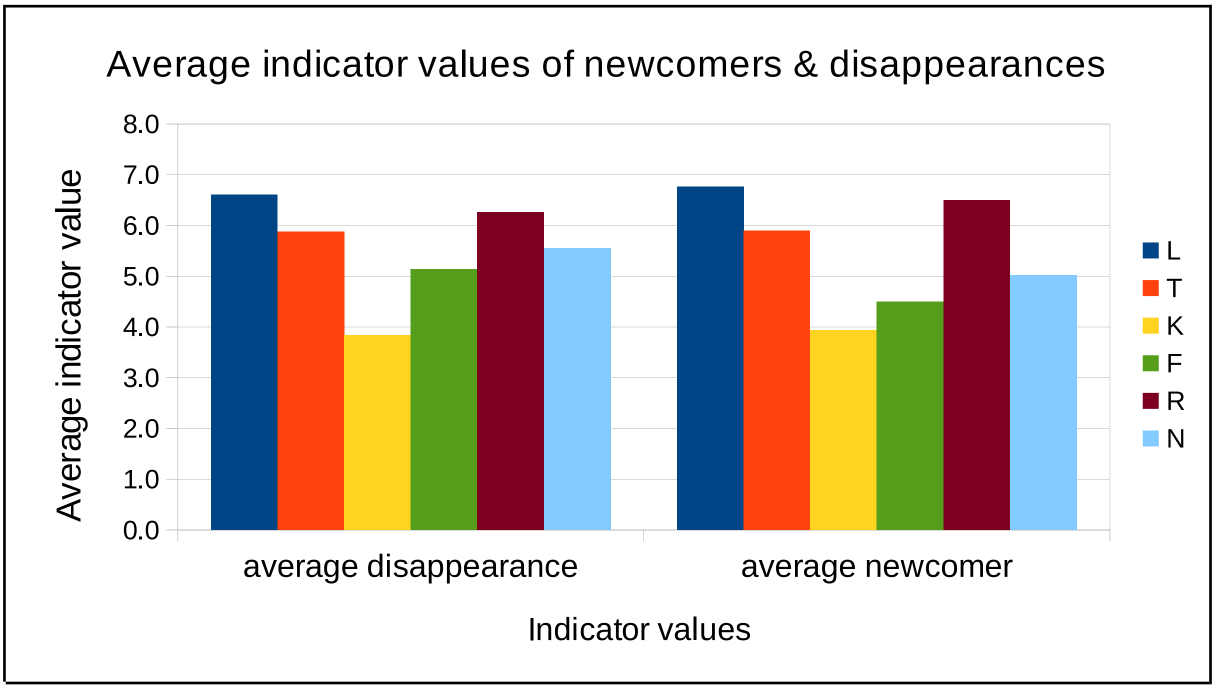

Figure 45: Average indicator values of newcomers and plants exclusively found in 1992

The detailed results on the winner's and loser's group as well as on the newcomer and lost species can be found in Chapter 3.5.

3.7 Ecosystem services: Pollinator value

The comparison between the abundances of nectar and pollen producing plants in the biotope samples yielded varying results. Counter intuitively, in the floristic database the lower value of 1 stands for a higher, the value of 2 for a lower value of pollen and nectar supply. Thus a lower value in the following evaluation stands for a higher value for honey bees. (Hoffmann et al., 1998)

3.7.1 Mean nectar value

Nectar production shows a significantly higher value in 2019 as compared to 1992, as shown in the following box plot, (Figure 46) indicating a decrease in value for the pollinators.

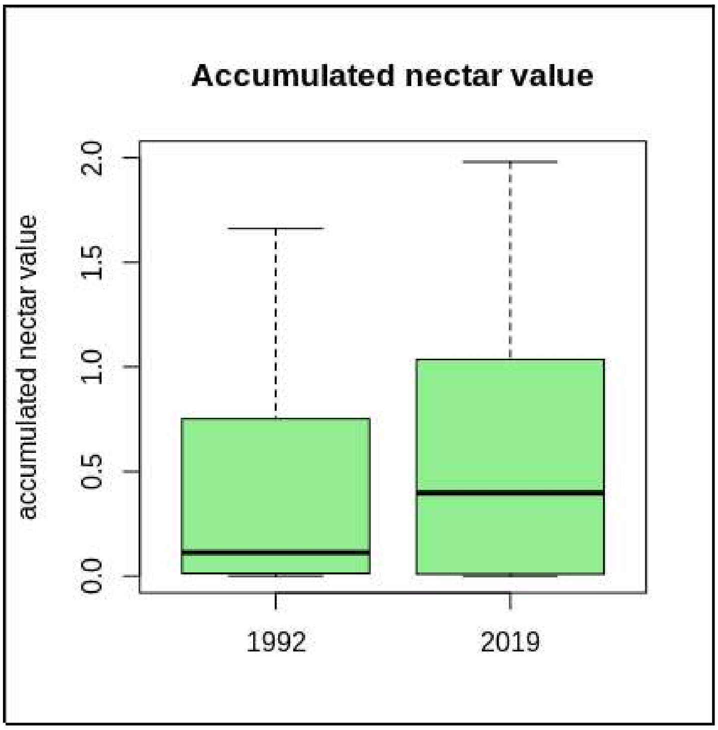

Figure 46: Mean nectar value of all samples
nectar value 1 = honey yield of 3 or 4 (high yield)
pollen value 2 = honey yield of 1 or 2 (low yield)

The following scatter plot shows an expected linear relationship between the years with the exception of grass margins, which often yield values close to zero in either 2019 *or* in 1992. (Figure 47) The values for the hedges ('wood') are inclined towards 2019, while in contrast the data points of semi-open elements show higher values in 1992, indicating an improvement of the situation for pollinators.

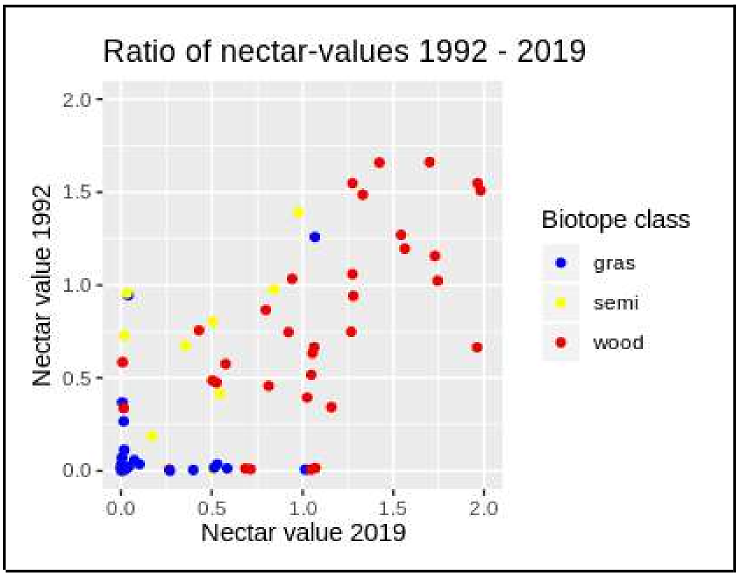

Figure 47: Nectar values - 1992 plotted against 2019 for biotope classes grass margins (gras), semi-open elements (semi) and hedges (wood)

However, as with the calculation of width and area in Chapter 3.1 as well as species numbers in Chapter 3.3, different results were obtained with differentiation. Biotope classes, sampling units which remained unchanged and those which encountered changes in structure and area were differentiated.

The results in detail are listed in the following Table 9. Significant changes are printed in bold letters and marked with an asterisk.

Table 9: Change in average nectar value 1992-2019.
** = statistically significant, confidence level >95%*

Change in average nectar value	**all**	**unchanged (type)**	**changed (type)**	**constant area**	**increased in area**	**decreased in area**
all biotope classes	***+0.11**	+0.08	+0.26	+0.13	+0.21	- 0.04
hedges	***+0.29**	+0.23	+0.39	+0.28	+0.39	- 0.01
semi-open	- 0.22	- 0.14	- 0.53	- 0.21	- 0.32	-
grass margins	+0.04	+0.04	-	+0.09	- 0.15	- 0.04

This means, that on average in all biotope classes and even more so in hedges, the nectar provision has declined. For grass margins the nectar value has increased, suggesting a deterioration for the honey bees, except for those grass margins which have increased in width. In semi-open elements the situation has improved overall.

3.7.2 Mean pollen value

For the mean pollen value only semi-open elements which have remained constant in area showed significant changes, namely a decreasing mean pollen value, thus improving for insects feeding on them. (Table 10)

Table 10: Change in average pollen value 1992-2019.
** = statistically significant, confidence level >95%*

Change in average pollen value	all	unchanged (type)	changed (type)	constant area	increased in area	decreased in area
all biotope classes	- 0.06	- 0.06	- 0.10	- 0.07	+ 0.04	- 0.07
hedges	- 0.01	- 0.02	+0.01	- 0.03	+ 0.09	- 0.16
semi-open	- 0.49	- 0.42	- 0.74	***- 0.57**	+ 0.29	-
grass margins	+0.001	+0.001	-	+0.07	- 0.33	- 0.05

The distribution of mean pollen and nectar value in the different biotope types shows similar patterns. This becomes evident, comparing the following graph in Figure 48 with the scatterplot showing the nectar values in the different biotope classes (Figure 47). While the majority of nectar values for hedges showed lower values in 1992 as compared to 2019 (better for honey bees in the past) the pollen value is shifted more towards higher values in 1992 (better or equivalent for honey bees in the present).

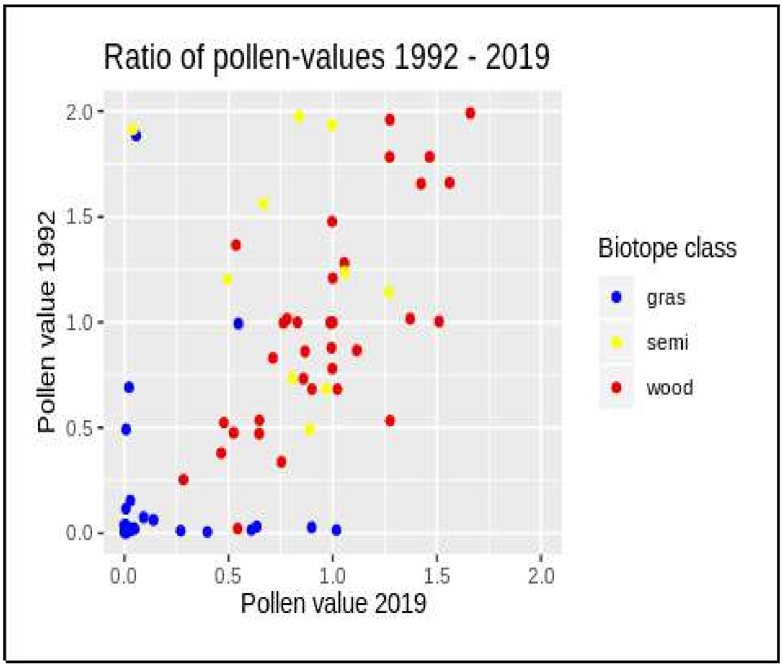

Figure 48: Pollen values - 1992 plotted against 2019 for the biotope classes grass margins (gras), semi-open elements (semi) and hedges (wood)

3.8 Shannon index as diversity indicator

The Shannon index as an indicator for diversity shows widely scattered values displayed in Figure 49. The comparison between the years 2019 and 1992 using the Wilcoxon signed rank test revealed a significant difference between the years with a confidence level of 97.3%.

The differences in the subsets of grass margins and wood biotopes were not significant, while with a confidence level of 98.6% the semi-open elements showed a significant change in their Shannon index. Neglecting the single observations yielded the same p-value.

While the average Shannon index for all biotopes changed from approximately 1.07 in 1992 to ca. 1.27 in 2019, the semi-open elements show Shannon indices of approximately 1.21 in 1992 and a higher value of 1.61 in 2019.

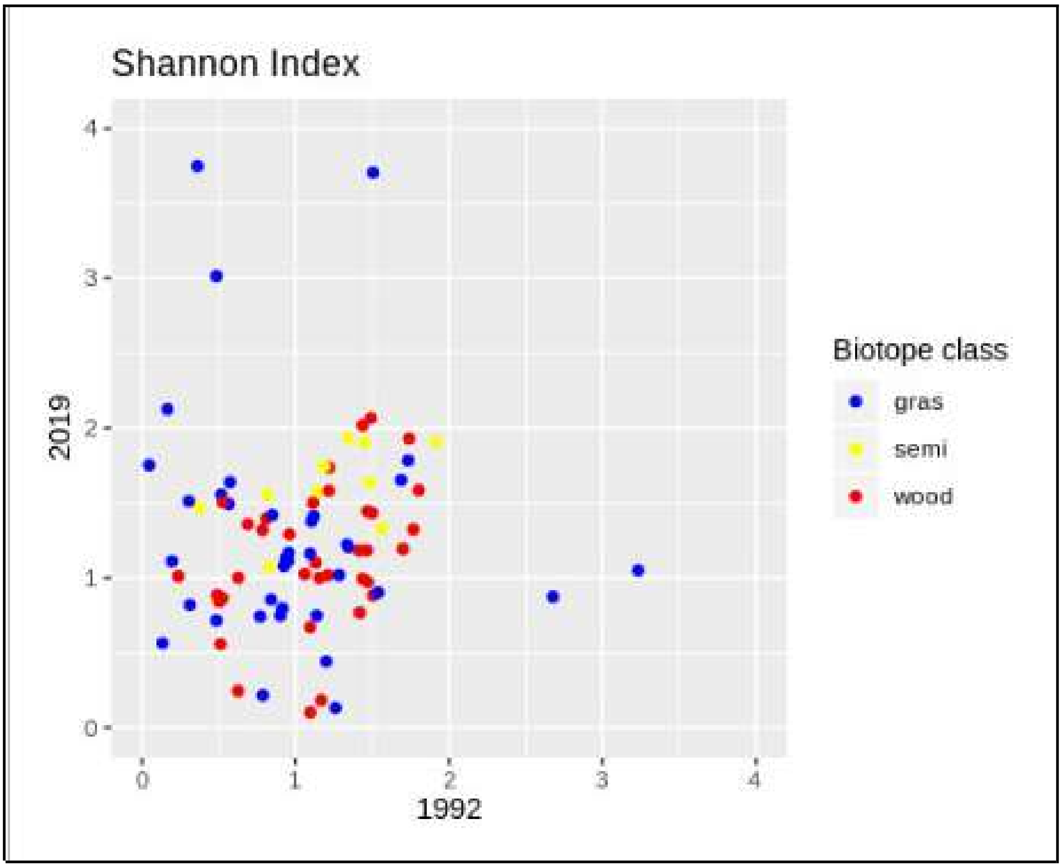

Figure 49: Shannon index of sampling units 1992 plotted against 2019

3.9 Research design

The relationship between area and species number determines in this case, whether the sample size is sufficient. This question is approached in the following.

Area-species relationship

The relationship between area and species number is expected to be sigmoidal, however, the biotope data shows different curves for each biotope class as shown in Figure 50.

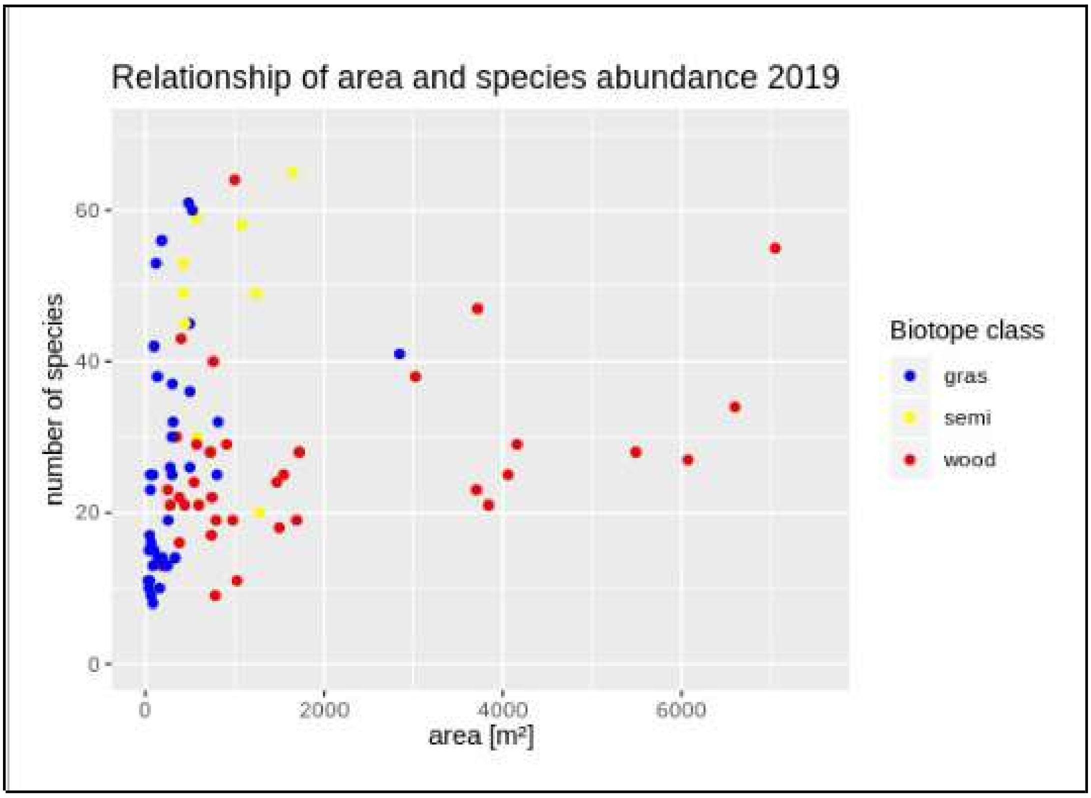

Figure 50: Relationship between area and species number

The number of species found in each sampling unit in 2019 was plotted against the area of the individual biotope sample.

A repetition with the data from 1992 showed a similar pattern as in the scatterplot of Figure 50.

Adequacy of sample size

The minimum area calculation yielded a saturation curve which did not fully reach a plateau, as depicted in Firgue 51. The data set of species numbers was randomized and the number of new species subsequently summed up as described in Chapter 2.5.2. This yielded a logarithmic curve approaching a slope of zero.

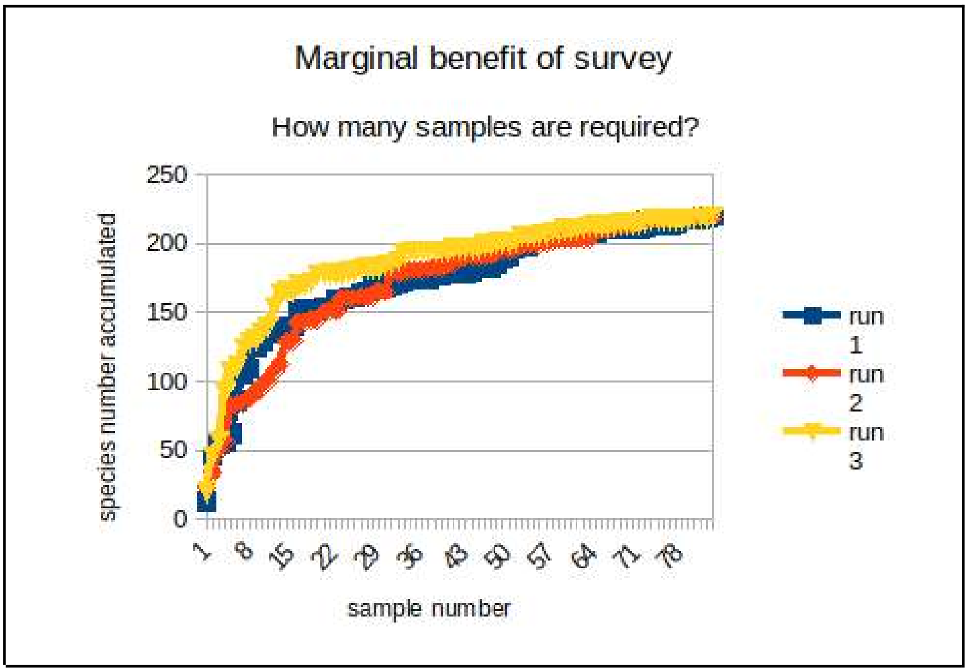

Figure 51: Accumulation of new species from randomized total sample - three iterations (runs)

4. Discussion

The following chapter will contribute to the interpretation of the results and to the discussion of the method.

4.1 Spatial and structural changes

The hypotheses – that total biotope area has decreased and the hedges have grown in area at the expense of grass margins – based on long-term observations in the field, could be confirmed for the 83 sampling units. The changes in structure from semi-open elements towards dense wood structures are in accordance with the usage of the landscape elements. Where the grass margins and the semi-open areas are not used to obtain feed for animals or where they are not cut within the scope of roadside maintenance, due to succession, the proportion of shrubs has increased.

Hedges are only cut occasionally in order to maintain them, while no incentives exist to use them productively. This is true even for the fruit trees in the landscape elements, which are hardly harvested and taken care of.

The biotope area has decreased and where it has increased, as in the case of semi-open elements, it was due to structural changes. Instead of a growth in biotope area at the cost of arable land, shrubs encroach into grass margins. A conversion from grass margins to semi-open elements is only shown for one of the biotope units. This was because shrub encroachment was counted as a loss and increase of area, not as a complete change in biotope type.

The outlier, mentioned in 3.3 is a developing grass margin which has decreased in area by more than half and is now only three meters wide. It was a marginal case, which could also be classified as a temporarily used track on arable land as it was tilled at some point in the past. It is a good example of how classification is not always black and white, as no clear threshold exists at which a fallow strip turns into a grass margin. Furthermore it shows that these special sites with moderate disturbance can contribute substantially to diversity and host a wide spectrum of plant species.

Method

The change in area was not recorded for all sampling units, due to the level of uncertainty during field work. Since a change in width and area was only recorded when it was clearly visible, and the change is not due to a different level of accuracy in the aerial photographs of the 90s and 2019, or bias in measurement in the field, changes in width of one meter and below could not be recorded. Furthermore it can be assumed, that the width estimated for the landscape elements in 1992 is subject to errors, since in some cases the sum of the width of parallel linear elements was not in accordance with the width of the total linear structure as derived from the aerial photographs.

The sample size in total was sufficient for the analysis conducted. However the size of the class of semi-open landscape elements was too small to be tested successfully with classical statistical methods for a change in area and width.

A non-representative survey among colleagues suggests that the classification of margins does not always yield the same results. Despite the mapping instructions, the outcome of the classification as margin substantially depends on the person who conducted the mapping. Differences in mapping exist mainly with respect to whether a grass margin has overarching trees or not. This was agreed on to be a criterion for exclusion of biotopes from the category of grass margins. However, when partly covered by trees at a height of approximately three meters and above, grass margins were probably still classified as such by Pfeffer (1992) as they were in this survey. For future surveys a more precise definition is recommended.

Therefore using the records of 1992 as a reference point and checking for evident changes proved to be the best method given the circumstances, as it would exclude additional bias by the author.

4.2 Species numbers / richness and turnover

The results show that species numbers barely varied between the years, while the hypothesis suggested it would decline. Despite plants which could no longer be found in the sampling units covered by the floristic survey, other plants were newly detected, adding to the comprehensive list provided by Pfeffer (1992).

A repetition of parts of the analysis neglecting single observations of plant species did not strongly affect the result for individual relevés, though it shows that for the overall number of species found, single observation do play an important role. However, this does not affect the overall trend of increasing species numbers. There is a possibility that the number of new species is slightly overestimated, because two species which were not determined at a species level were regarded as separate species.

Whether changes in structure and area affect the species number in a biotope, differs among the biotope classes. The sampling units which have changed in structure as well as those which decreased in area showed a decline in species number. The latter may be explained by the relationship between area and species number. For the sampling units overall, as well as for the classes of hedges and grass margins, a change in biotope class led to a decline, while only for semi-open elements (formerly grass margins) it resulted in a rise of species numbers. This suggests, that succession in grass margins first increases the phytodiversity before it deteriorates as the wood cover grows more dense. Unless they increased in size, on average grass margins have declined in species numbers. This suggests, that the conditions for plant diversity are deteriorating independently of the loss in area. For hedges and semi-open elements the opposite seems true. One potential explanation could be the distance to the field. It provides a larger buffer distance to the area where fertilizers and pesticides are applied. Apart from grass margins, on average, the sampling units which have not changed in structure and area have slightly increased in species number, suggesting that the situation is improving for biodiversity in these biotope classes.

The hypothesis that a turnover in plant species exists could be confirmed. It is surprisingly large, illustrating how volatile single plant communities in landscape elements can be in contrast to a landscape section as a whole. The latter can be seen as more resilient, where biotopes are well connected.

Other studies show different results for how fast species migrate in wood structures. While Reif & Aulig (1990) report a slow migration of wood species into an established hedge, even after two decades, Gruttke et al. (1998) cited in Dierschke (2000) have observed spontaneous propagation of wood species in newly established hedges within ten years. Dierschke (2000) refutes the idea that the herbaceous species are often neglected in the context of hedges. He reports that no spontaneous establishment of species typical for grass margins took place in the first years after planting. These findings stress the importance of a network of biotope complexes, consisting of different biotope types.

Method

Even though the development of species richness shows a clear pattern throughout all biotope classes, corresponding with changes in size and structure, the results are not statistically significant for the most part, due to the small sampling size of the subsamples.

With the number of single observations, the chance of obtaining an error by overlooking single plants increases. It can be assumed that more reliable information is obtained for the purpose of comparison, when all single values were neglected. This again leads to a loss of information. Ecologically, it is of greater significance to know whether a species has newly occurred or disappeared (Wildi, 1986). This forbids neglecting single observations. However, comparing the results from the full and from the trimmed sample gives a good estimate of how large an error may be between the surveyors. Nilsson & Nilsson (1985) have observed pseudo-turnover between teams of 11.4%. Pseudo-turnover is the sum of sampling errors which increase species turnover. Within teams it was found to be lower with 7.9%. Between 1992 and 2019 different teams were surveying. Therefore, the error can be assumed to be in this order of magnitude, both for within-team pseudo-turnover due to different surveyors who contributed to the generation of the species lists as well as for pseudo-turnover between the surveyors of 1992 and today.

Changes in coverage are expected to be subject to errors as well (Traxler, 1997), especially when large sampling units with varying size are involved as is the case. The errors occur due to different expertise, diligence and at random throughout the different mapping procedures.

Another source of bias which can not be quantified at present is the time that was taken to survey an area. The surveyor was walking along the length of the linear structures, entering the wider structures in short distances in order to get an overview. However, for narrow structures more time per area was spent on the survey in comparison to the wider structures. This increases the likelihood for plants in the grass margins to be detected over that of plants growing in wood structures.

4.3 Species distribution and constancy

The values for constancy are a measure of the occurrence of plants relative to the number of respective biotope units. Due to the smaller sample size – for example, for semi-open elements – lower values are obtained.

It was hypothesized that nitrophilic and frequently occurring species would increase in constancy while rare species and specialists would decrease. The circumstance that the top seven winner species *Geranium robertianum L., Fallopia convolvulus L., Rubus L. sp., Arctium lappa L., Galeopsis tetrahit L., Geum urbanum L.* and *Alliaria petiolata (M.B.) Cav. et Gr.* all show N-indicator values for N above average supports this hypothesis. More than the previous calculation of the MEIV, this shows that N remains a strong driver, altering the species composition in favor of a few well adapted species. Winners are also often species associated with woods, such as *Hedera helix L., Geranium robertianum L., Galeopsis tetrahit L., Geum urbanum L.* and *Rubus L. sp.,* corresponding with the increase of hedges. For *Hedera helix L.*, its current propagation is likely caused by the increased temperatures as stated by Dierschke (2005).

The species which have most declined in constancy are not necessarily highly specialized plant species and they are rare only relative to the plant inventory of the survey area. Examples are *Equisetum arvense L., Pimpinella nigra Mill.* and *Knautia arvensis (L.) Coult.*.

Most species of the top-15 species which decreased in constancy, such as *Capsella bursa-pastoris (L.) Med., Cichorium intybus L., Tripleurospermum inodorum (L.) Sch. Bip., Artemisia vulgaris L., Stellaria media (L.) Vill., Arabidopsis thaliana (L.) Heynh.* and *Cerastium arvense L.,* occur on cultivated fields where the soil is disturbed. Other winning species, however, which occur frequently, such as *Consolida regalis S.F. Gray* and *Papaver rhoeas L.* require soil disturbance as well. (Rothmaler & Jäger (ed.) et al., 2011) These herbs were often observed as single occurrences in grass margins, enough to be counted as present.

Other species such as *Dactylis glomerata L., Achillea millefolium L., Knautia arvensis (L.) Coult.* and *Daucus carota L.* are grassland species which occur in meadows. (Rothmaler & Jäger (ed.) et al., 2011) They are adapted to and rely on disturbance in form of cutting or grazing, which does not take place in the margins, except by deer. The loss of a species like *Medicago sativa L.*. is simply due to the fact that it used to be a fodder crop which is no longer cultivated in the area but persists in the grass margins.

Taraxacum ruderalia Kirschner et al. and *Elymus repens L. (Gould) (*formerly *Elytrigia repens)* grow where the soil was disturbed – for example by ploughing – and then abandoned. (Rothmaler & Jäger (ed.) et al., 2011) The decrease of *Lamium album L., Medicago sativa L.* and *Cichorium intybus L.* may partly due to a lack of light which is taken by nitrophilic competitors, as they have a high L-value indicating a high demand for light.

Elymus repens L. (Gould) grows over an extended period in fallows before its abundance decreases. (Jüttersonke et al., 2012), (Jüttersonke et al., 2008) Their decrease observed in this study can well be explained by the grass margins remaining unused for a long period of time. Jüttersonke et al. (2008, 2012) described a decline in species in long-term set-aside areas and field margins. However, unlike in the trial, no cutting regime was applied in the grass margins of Hasenholz.

Equisetum arvense L. is associated with water logging and soil compaction. (Oberdorfer, 2001) Its disappearance in some biotopes may be a result of the decrease in water supply throughout the year, corresponding with the deficient supply of rainwater in the recent past and the overall trend towards hotter summers (8 in Chapter 1.4.3).

Method

The fact that species are more or less frequently recorded in 2019 in comparison with 1992 (Chapter 3.5) can have various causes rooted in the method as well. One is the difference in the person conducting the survey and her/his estimates, experience and diligence and the fact that a perfect coverage of the entire survey area is unlikely to be obtained even with highly skilled personnel as described in the previous subchapter.

One reason for an increased number of segetal plants in the 1992 survey, such as *Cerastium arvense L., Arabidopsis thaliana (L.) Heynh., Stellaria media (L.) Vill., Tripleurospermum inodorum (L.) Sch. Bip., Cichorium intybus L.* and *Capsella bursa-pastoris (L.) Med.* may be differences in mapping. The fact that among the top 15 of the winners in grassland no segetal plants apart from *Papaver rhoeas L.* were present, could be due to a lack of disturbance since 1992, but also due to a classification of parts of the dynamic margins as grass margins in the earlier study. The "dynamic margins" are part of the arable land. They are cultivated but not sown in and thus habitat for species which rely on disturbance. However, transitional stages occur.

The apparent loss of *Veronica hederifolia L.* may be due to differences in timing while mapping the vegetation. Since *Veronica hederifolia L.* blooms and perishes early in the year, the two weeks of delay in the 2019 mapping season may have sufficed for the plant to disappear due to its natural annual cycle. An earlier start of the vegetation period may contribute to this as well.

4.4 Mean Ellenberg Indicator values

The results for the MEIV indicate potential causes for changes and adaptations of the plant communities.

For **moisture,** the F-value suggests, that the abundance of species has changed, resulting in a lower average F-value ($MEIV_w$). According to these findings, newcomers and lost species have a negligible effect on the $MEIV_e$ for moisture. Since this could not be shown for the $MEIV_e$ it may be concluded that the abundance of species is more sensitive to changes in moisture than the species occurrence in the overall plant community. The findings of Schaffers & Sýkora (2000), that the F-indicator is a good predictor of actual soil moisture condition supports these results.

The K-value for **continentality** showing significant changes for the $MEIV_w$, but not for $MEIV_e$ suggests the same: Changes in the overall climate may be reflected by shifts in the abundance of species within a plant community, but not or less by a shift in species composition. One explanation is that the migration of plant species occurs slowly, relative to a shift in species abundance. The process of a share of one plant species in a community growing to become dominant or minor may happen faster than the migration of new species into an area. The pace at which the process of migration takes place is dependent on the species' ability to spread and the fragmentation of the landscape. (Thuiller et al., 2005)

The decrease of the mean K-value suggests, that the climate has grown more oceanic. However, while moisture is a single variable which is relatively simple to measure and which can be related to rainfall data, the K-value, is more complex. It reflects the distribution of the respective plants in climate zones, based on which the ordinal scale was created by Ellenberg (1974) arbitrarily merging the categories subcontinental and suboceanic (Berg et al., 2017). Climate conditions are determined by different aspects, such as the temperature amplitude and total rainfall. Continental climate is characterized by a large temperature amplitude and dry weather conditions. Changes in these two factors may serve as a potential explanation for the results.

The temperature amplitude has increased in Brandenburg in the past decades, (JKI, 2019) contradicting the shift towards a more maritime climate as suggested by the result of the analysis. Furthermore the rainfall in the summer has diminished in recent years, while an oceanic climate typically shows more precipitation. (JKI, 2019) However, the available temperature and rainfall data are aggregated and do not give information on the daily amplitudes which may play a decisive role.

One potential explanation for the decrease in K-value could be the response of the plant community to the increased overall temperature (JKI, 2019) and a slightly higher rainfall in the winter (Gerstengarbe et al., 2003), (Schönthaler et al., 2015). This leads to a more even distribution of the rainfall throughout the year, but not to more temperate moisture conditions, since the elevated summer temperatures lead to higher evaporation.

Possibly some of the species which have increased in abundance are better adapted to the increased new temperature range, while others suffer and decrease in coverage. However, a closer look at the distribution of the data pairs shows, that a limited number of plant communities in class of grass margins is responsible for the shift in mean K-value. They all represent grass margins facing east or south. Climate conditions, however should affect all biotopes equally, unless micro climatic conditions play a role here.

The **L-value** for light significantly decreased for $MEIV_e$, while for $MEIV_w$ no change was detected. (Kunz, 2016) states the opposite to be true and relating it to dominant plant communities on eutrophic sites taking light from smaller plants adapted to dystrophic conditions. The scatterplot clearly shows that as expected the shade tolerant species grow in the hedges, those dependent on light on the grass margins only. However, the cloud in the scatterplot shows an uneven pattern which may be due to irregularities in the data.

While seven nitrophile species as top winners of plants with high **N-value** have increased in abundance, the mean N-value did not show a significant difference in any direction. This is surprising at first glance, however, Schaffers & Sýkora (2000) also found that the Ellenberg

N-value was only weakly correlated with available mineral N and soil parameters, but more so with biomass production. A higher biomass production however may not have been realized in this case, due to the decreased moisture.

The findings for the **temperature** indicator suggest, that the spectrum of species in the different sampling units did not shift towards a species composition with a higher mean T-value, neither for $MEIV_e$ and $MEIV_w$, despite the higher average summer temperatures. This is contrary to the hypothesis based on the climate development and observations by Litza & Diekmann (2016).

The difference between the average of the top ten to top 52 winners and the top ten to top 52 losers depicted as curves visualizes an inclination of the plant communities towards more nutrient rich plants, which was not detected by the Wilcoxon signed rank test. The factors light and moisture also suggest a clear tendency, while temperature oscillates around zero, in line with the result for the MEIV. The curve for the K-value is also relatively steep, approaching and crossing the y-axis, suggesting, that the topmost winners and losers may well be related to a more oceanic climate.

Methods

MEIV derived from observations which were weighted according to the plant's coverage ($MEIV_w$) and those calculated from equally weighted observations ($MEIV_e$) once including and once excluding single observations (>1) show a discrepancy which can be attributed to the different methods. Single observations may lead to a distortion of the mean indicator value, because of the possibility that a single observation was erroneously made in only one of the two years, due to its rarity. However, the results barely differed depending on whether the single observations were neglected or not, so it can be concluded that for this application, this type of error can be neglected. For $MEIV_w$, a distortion through single observations can be neglected, because the coverage value of single observations is too small to have a relevant impact on the result.

The results support the claim made by Möller (1992) that the use of mathematical operations with the arithmetic mean of indicator values is a questionable method. Since not all plants will show dominance in their abundance when they encounter ideal site conditions, the presence and absence values of the plants in a community may yield more relevant results, as stated by Wilson (2012).

It may seem surprising, that none of the indicator values attributed to the lists of winners show significant differences compared to the list of losers. The reason for the lack of significance may be due to the application of the Wilcoxon test to unpaired, discrete data in contrast to the comparison of the two groups of 2019 and 1992 previously. The unpaired test is less powerful, due to the inter subject variability playing a larger role. The comparison of newcomers and disappearances can therefore only weakly support a hypothesis and other findings. It is not significant, due to the small sample size and the fact that the numbers are integers which limit the ability of the Wilcoxon test to calculate exact p-values based on whole numbers.

The discussion about the best method and the use of mean indicator values is still ongoing among ecologists. While Böcker et al. (1983) named the identification of environmental changes as the most interesting field of application for the MEIV and Wildi (2015) encourages rehabilitation of the use of mean EIV, Zelený & Schaffers (2011) conclude that EIV related to other variables derived from plant composition should not be used for statistical inferences. In order to avoid biased results, they suggest a modified permutation test using mean randomized EIVs. However, in the present case, subsamples in form of classes were sometimes too small to yield statistically significant results.

Results contradicting the observed change in site conditions were reported by Litza & Diekmann (2016), when despite the measured lower pH an increase in the mean R-value was recorded.

The comparison between topmost winners and losers is not an established method, as far as is known. The visualization gives more weight to the winners and losers ahead in the ranks leading to visual results especially in the case of nitrogen for which it was shown that the majority of plants which gained the most were also highly nitrophilic. By taking into account the rank of winner and loser plants this method may be able to point towards the factors determining the winners and losers. The steeper the slope and the straighter the curves, the higher may be the relevance of the influence factor on the range of species gaining and losing in constancy.

4.5 Ecosystem services

The results show, that nectar and pollen do not correspond as expected, but show contrasting results. One reason for the unequal development of nectar and pollen could be the gain of species like *Prunus spinosa L.* and *Hedera helix L.* which are increasing in abundance, providing more pollen than nectar. (Hoffmann et al., 1998) Even though, *Prunus spinosa L.* does not belong to the top 15 winners, it has a large total coverage. Winner species like *Rosa canina L., Quercus robur L.* and *Ulmus laevis Pall.* also produce pollen, while not providing nectar to the pollinators. (Hoffmann et al., 1998)

For the grass margins, the results for the accumulated nectar and pollen value for 2019 and 1992 do not show a linear correlation between the sampling units as expected. The values lie very close to zero, inclined towards 2019. The higher values in 2019 suggest a lower supply in 1992, only for few cases this is the opposite. This may be due to a higher abundance of flowering plants valuable to *Apis mellifera L.* in 1992. One explanation could be the mass occurrence of grass species such as *Bromus sterilis L.* or *Elymus repens L. (Gould)* outcompeting the flowering plants which profit from soil disturbance and less competition of nitrophile species.

Furthermore the findings are in line with the findings of Phillips et al. (2018) who have observed a drop in available resources for pollinators under drought conditions in calcareous grassland. However, in the floristic survey, the phenomenon that drought is correlated with a lower production of flowers, would only be detected by a biased estimation of plant coverage towards lower coverage values, due to the lower flowering aspect.

In order to assess the nectar and pollen value of plants next to their presence, the phenology must also be taken into account, since in dense plant communities plants may not necessarily flower and only the flowering aspect turns a plant into a valuable food source for the honey bee and others pollinators (Hoffmann, 2019), (Timberlake et al., 2019).

A review of 158 pollination studies, Szigeti et al. (2016) revealed large differences in methodology and found that many studies were conducted over too few years and that vegetation sampling was presented insufficiently in many studies while sampling covered only a small section of the study sites. In this thesis, these weaknesses were not present.

4.6 Shannon index as diversity indicator

The entropy and thus the diversity has increased between 1992 and 2019 with semi-open elements contributing substantially to this development, while hedges and grass margins have not increased in diversity. This is likely due to the few species which have gained in dominance disproportionately in the latter, reducing the evenness of in the sample. One reason for the high Shannon indices could be a combination of light exposure and distance to the arable field, which may serve as a nitrogen source. These two factors lead to an increase in nitrophilic species. (Kunz, 2016)

The negative correlation between mean N-value and the Shannon index observed by Wesche et al. (2012) could not be reproduced in this study.

4.7 Experimental design and outcomes

The Braun-Blanquet method remains state of the art, while different ways of recoding the scale have emerged. (Camiz et al., 2017)

The strength of the experimental design and the entire project *BioZeit* is that the landscape and its biotope types from arable land up to hedges, woods or lakes are investigated intensively with regard to detail, frequency and extent in terms of area coverage. For this work in particular, the large data set was a precondition for obtaining significant results. The complexity of the ecological system of the plant communities in the hedges and margins requires sufficient data, as well as an adequately large time frame in order to capture long-term changes.

However, little can be said about the development of species numbers and composition in the 27 years in between and about annual changes. In succession studies despite the long term trend, short term variation between two years of up to 20% deviation from a reference scenario was observed by Köhler et al. (2005) for example. The results are therefore always a combination of annual effects due to weather or management and long-term trends.

Weaknesses of the setup are the different sizes of the sampling units. In comparable surveys, more commonly, plots of defined size are surveyed, often with exact localization as done by Walentowski et al. (2014) during long-term vegetation monitoring in Bavarian forests. The fact that the sampling units vary in size is problematic for the purpose of comparison between samples. (Chytrý & Otýpková, 2003) The sampling units in Hasenholz do not only differ in area among each other, but may also vary in size throughout the years. Therefore the results have to be interpreted as either biased due to the changes in area of the individual sampling units or as less significant due to the smaller number of samples that were compared in the statistical test.

Due to the difference in quality of aerial photographs available from the nineties and the photographs almost three decades later, there was no way to detect small systematic changes of area which are smaller than the potential error of the method. It can be assumed that the change in area and width is generally underestimated.

The results of the analysis of the area-species-relationship show that while for grass margins sampling units larger than 1000 m² will hardly have an effect on the number of species found, for semi-open areas an area twice as large may yield the same number of species with marginal benefits in case of an increase of the area of the sampling unit. For hedges, the number of species found increases up to 4000-7000 m² in the given sample. However, since the relevés of the biotope classes were of different sizes, it is not possible to compare the number of species of the different biotope classes, but only the respective change. The method of choosing sampling units of different sizes as done by Pfeffer (1992) increases the variance of the data.

The sample size however, may be regarded as appropriate for the scope of this thesis. The number of new species which may be detected by increasing the sample size is expected to increase only marginally with any additional relevé. This is suggested by the slope in 51 in Chapter 3.9, which approaches zero. However, the survey does not represent a full species inventory of the two times two kilometer survey area of Hasenholz.

5. Conclusions

In terms of species number and species diversity measured by the Shannon index, the biotopes in the survey area of Hasenholz have shown **resilience** with respect to diversity. Plant communities overall are adjusting to changes through shifts in their species composition and species abundance without facing losses in total species numbers. The fact that the **turnover rates** in the individual biotopes as well as in the whole of the biotope samples are substantial, suggests that the plant communities have not reached a steady state, but are subject to changes.

However, as the trends differ among the biotope classes, the adaptation may not be part of a **sustainable development** with regard to a maintenance of diversity: The trend is a shrub encroachment and widening of hedges. Thus the variety of biotope types is decreasing. While semi-open elements host the most species, this biotope class covers only one tenth of the area. Its area share grows slightly more than it decreases through shrub encroachment. However, it does so, at the cost of grass margins which have encountered substantial losses. Therefore the elements surveyed may all eventually develop into hedges.

The overall turnover of the survey area is substantially lower than the change in species inventory of single sampling units. This emphasizes the importance of a connected system of biotopes. Such a network enables plant species to migrate in order to spread in a more suitable neighboring biotope and therefore sustain its population in the area overall. Especially in the context of a changing climate this is of high importance to maintain the resilience of the landscape. Grass margins have not only declined in area, but also in plant diversity independently of area changes. This indicates worsening conditions for plant diversity in that biotope class.

Intuitively and following the typical **area-species-relationship**, the area and width of the sampling units positively influences the species number. For individual biotopes it can be stated that a larger area is in favor of a larger species number. For the overall survey area of Hasenholz, since the marginal benefit approaches zero, it can be concluded that a sufficiently large share of the plant species in the surveyed biotope type was covered to draw comparisons between the years. However, in order to obtain a full species inventory, the sampling size has to be increased. For future studies it is also advisable to choose sampling units of equal size, if a comparison of the species richness of the different biotope types is desired.

The **influencing factors responsible for changes in the plant community** independent from changes in biotope area and biotope type are multifaceted. **Nitrogen supply** is one explanatory factor which explains the increase of species which already showed the highest frequency of occurrence in 1992.

The method of calculating **mean Ellenberg indicator values (MEIV)** yields different results when calculated based on species abundance or based on the mere presence of plants. The results indicate that the plant communities have adapted to dryer conditions, less availability of light and a changing climate or aspects of it, since the K-value is not a one-dimensional value which shifts along a linear scale. The temperature amplitude, the shift in temperature extremes, and precipitation are multiple dimensions which may develop independently. This is not reflected by the K-value ranging from one to ten. A possible explanation for the shift towards lower K-values is the increased precipitation in the winter, while a larger average temperature amplitude and hotter summers indicate the opposite.

The different results for $MEIV_e$ and $MEIV_{w,}$, the unexpected results for N and T, suggest that the method of calculating MEIV may not always detect shifts in site conditions reliably, since despite the evident increase in dominance of nitrophilic species and the temperature rise, no shift in indicator value was detected.

As a potential consequence of the shift in species composition, the **pollen and nectar values** are subject to changes. Calculated analogous to the MEIV, the average pollen and nectar values may serve as an indicator providing information on the development of pollen and nectar producing plants. The value of a plant for pollinators, precisely for the domesticated honey bee, is dependent on other variables like the phenology and the distribution of flowering aspects over time.

6. Perspectives

For further research, the data generated in 1992 as well as 2019 holds the potential to serve as **reference for similar studies** on the development of the area and its vegetation in the future. Another study in future decades is advisable. To what extent the plant communities change on an annual basis relative to the change which is due to a long-term development since 1992, remains unanswered. Therefore even in shorter intervals a repetition of the study may be useful.

The issue of **ecosystem services by nectar and pollen provision** requires more attention. For future studies it is recommended to take the phenology into account, in addition to mere plant coverage. Vegetative growth adds to the plant coverage, however without flowering aspects it does not contribute to the value for bees and other pollinators.

During fieldwork the **dynamic zones between the crop and the grass margins**, which were cultivated but where no crop was sown afterwards, were partly documented but not analyzed within the scope of this thesis. Since these margins were not yet defined as separate biotope types and yet excluded from the survey on wild plants within the agricultural fields, they have hardly been studied. Their characteristics, at the transition zone between agricultural fields and margins makes them hard to grasp as a clearly defined category but interesting in terms of their species inventory and contribution to the overall diversity of the landscape. Future research could yield information on the contribution to the botanical as well as entomological diversity of the plants hosted by these dynamic zones. To what extent can these be habitat to segetal plants which are often efficiently managed in the field?

It is recommended to take into account **qualitative information** in future studies, such as adjacent land use, shading by trees, management of hedges, grass margins etc.. In that case, recommendations could be derived on an ideal management of these biotope structures.

This thesis shows the importance of semi-open areas as a stage of transition between hedges and grass margins with respect to the number of species they host. The maintenance of roadside greenery can play a major role in **management in order to maintain the diversity** of hedges, margins and intermediate structures. Furthermore the partial use of these wood structures – for example as fuel or grazing area – may be encouraged to create space for light dependent species. A temporary cultivation of parts of the structures, followed by another phase of succession is an option in order to maintain and create biotope diversity. Chances lie in a coordinated harvest and partial use of the hedges as part of a rural bioeconomy.

While this thesis focuses on the comparison and assessment of frequently occurring and thoroughly studied landscape elements and linear structures, a complementary survey on special sites and areal elements could be insightful. It is expected that these sites host species which are less frequently found and are at a higher risk of disappearance since they are bound to sites which are less common in the area.

7. Summary

The thesis is part of the project *BioZeit.* In the project, studies conducted in 1991-93 on the inventory of plant species of six different landscapes sections in Brandenburg and six species groups are repeated. This thesis focuses on the flora of the landscape elements in the landscape section Hasenholz and is based on work of Holger Pfeffer (1992), who wrote his diploma thesis about the subject, and on the work of his colleagues supporting the fieldwork.

As a first step, the vegetation survey and mapping of biotope structures was repeated for all landscape elements in the survey area of 2x2 km. The biotope types were further aggregated in to three classes: Grass margins, hedges and semi-open elements, representing a mix of grassland ecosystems and wood structures. A focus was laid onto linear structures, relevant for linking biotopes and thus for species migration. 192 relevés were formally identified and defined by Pfeffer (1992) and a detailed analysis of the flora was conducted twice in each sampling unit. For 83 of these sampling units, this survey was repeated. Furthermore it was recorded whether the sampling units had changed in structure or size. Based on the Braun-Blanquet method, the coverage of the plant species was recorded twice and the higher coverage value used for subsequent calculations as done in 1992.

The raw data from the species lists of 2019 as well as 1992 were compared by means of a Wilcoxon signed rank test. They were analyzed with respect to their species richness; species turnover; mean Ellenberg indicator values; nectar and pollen provision as an ecosystem service; the constancy of plants throughout the survey area and diversity, characterized by the Shannon Index.

The analysis showed that grass margins have declined in **area** while hedges have increased at the cost of semi-open elements which have overcompensated their loss minimally at the cost of grass margins. The **species richness** has slightly increased on average in all biotope classes, except for grass margins and those biotopes which have changed in structure. Semi-open elements showed the highest increase in species. However, only for hedges which did not change in size and structure, an average increase by five species was statistically significant.

In total, 219 species were identified in 2019 – 22 more than 27 years before. In order to minimize the error due to single observations which are assumed to be a source of error during the survey, they were left out. Doing so, still 161 species were found more than once; seven more than 1992.

The **species turnover** in the relevés largely varied between 30 and 79%. Rare plants for which only single observations were made and which are more likely to be overlooked hardly affected this result. The overall turnover for the 83 relevés was found to be 26%, which

means that about one quarter of the sum of species present in 1992 and today exclusively occurs either only now or only 1992.

The **frequency of occurrence** of the species throughout the entire survey area has decreased for some and increased for others. However, for the seven species *Geranium robertianum L., Fallopia convolvulus L., Rubus L. sp., Arctium lappa L., Galeopsis tetrahit L., Geum urbanum L.* and *Alliaria petiolata (M.B.) Cav. et Gr.* the frequency of ocurrence has increased overproportionately. These winner species hold in common that they are nitrophilous species mostly adapted to grow in the proximity of woods.

The mean **Ellenberg indicator values (MEIV)** were calculated for the purpose of comparison between 1992 and 2019. This was done in two different ways, by multiplying each indicator value with the coverage of the respective plant ($MEIV_w$) and by calculating the average while equally weighting the plants based on their occurrence ($MEIV_e$).

The results varied among the two methods. For moisture **(F)**, only the $MEIV_w$ shows a significant decline, while this could not be found for the $MEIV_e$. For the climate value **(K)**, only the $MEIV_w$ revealed significant changes towards an oceanic climate.

It is subject to discussion, whether it is appropriate to undertake such a calculation with values of ordinal nature such as the K-value (C-value for climate) which furthermore consists of multiple components and therefore can change in multiple dimensions. The temperature amplitude, the absolute temperatures and the moisture conditions on site may vary independently. Perhaps plant communities have responded to milder winters with an increase in abundance of plants with a lower K-value and thus better adaptation to more oceanic climate. Interestingly, the largest changes in $MEIV_w$ for climate have occurred in south and east-facing grass margins, which may point to effects of the micro climate.

Furthermore the $MEIV_e$ for light **(L)** showed a significant change, which might be explained by a lack of light due to the competition of plants which show increased vegetative growth under eutrophic conditions. However, the $MEIV_w$ did not show these changes.

Regardless of the increase in abundance of nitrophilc species and despite the temperature trend in the weather statistics, the MEIV for nutrients **(N)** and temperature **(T)** showed no increase, regardless of the method used.

Furthermore, results suggest that **ecosystem services** have improved with respect to pollen supply, but worsened in terms of nectar provision in the overall landscape.

The decrease in nectar provision was significant for hedges. Semi-open elements show an improvement for nectar with a significance level below 95% and an improvement above 95%

for pollen. Grass margins show a negative tendency for both pollen and nectar which also was not statistically significant.

The **Shannon index** shows an increase in diversity for the biotopes on average and for the class of semi-open elements in particular, while no significant changes were detected for the class of hedges and grass margins. Also in this case single observations – plants occurring only once throughout the entire sample – which were assumed to point to potential systematic errors, did not influence the result.

The sampling units defined by Pfeffer are of different sizes, due to their definition according to the identified biotope structures. The **area-species relationship** does not allow a comparison between the biotopes with regard to the number of species, only with regard to its change between the years. It shows that while for grass margins and semi-open elements a smaller sampling unit may be sufficient, hedges require larger areas in order to sufficiently detect the species present. The sample size chosen for the thesis was sufficient in order to draw a comparison between 1992 and 2019. A full species inventory of the area, however would require a larger survey area.

These results suggest that a large share of the plant diversity is present in the smallest and decreasing biotope class of semi-open elements. This biotope class shows improvements in diversity, while grass margins decline and deteriorate in terms of area, diversity and ecosystem services.

8. References

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9. Appendices

9.1 Constancy of winners and losers

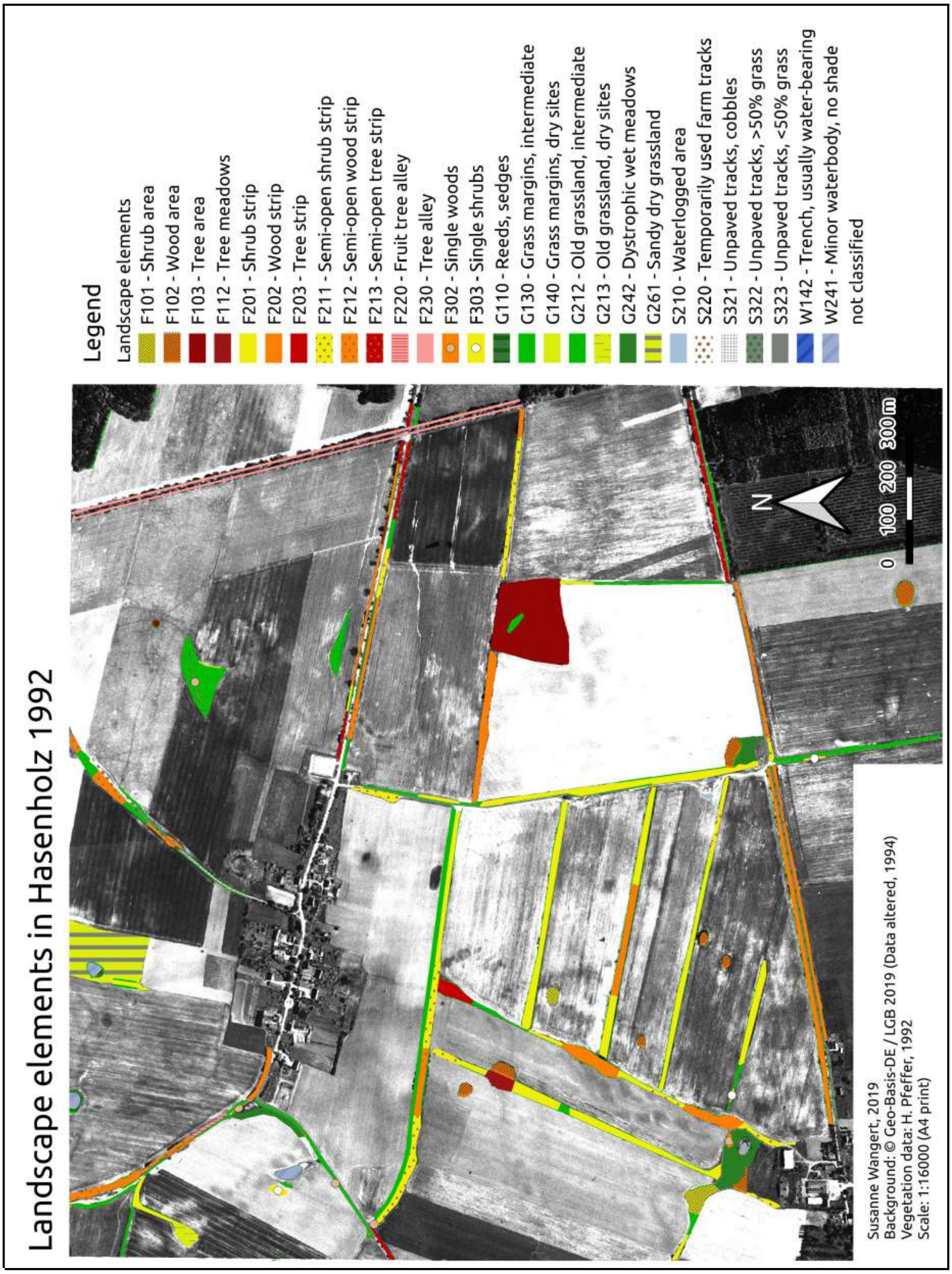
Species	constancy 1992	constancy 2019	Δ constancy	L	T	K	F	R	N
average winners				**6.7**	**5.7**	**4.0**	**4.7**	**6.8**	**6.0**
average losers				**7.1**	**5.9**	**4.3**	**4.6**	**7.0**	**5.8**
difference				**-0.4**	**-0.2**	**-0.2**	**0.0**	**-0.2**	**0.2**
Geranium robertianum L.	6	39	33	5		3			7
Polygonum convolvulus L.	10	41	31	7	6		5		6
Rubus sp.	18	49	31						
Arctium lappa L.	20	49	29	9	6	4	5	7	9
Galeopsis tetrahit L.	2	30	28	7		3	5		6
Geum urbanum L.	24	52	28	4	5	5	5		7
Alliaria petiolata (M.B.) Cav. et Gr.	21	45	24	5	6	3	5	7	9
Chelidonium majus L.	5	19	14	6	6		5		8
Vicia hirsuta (L.) S.F. Gray	15	29	14	7	6	5	4		4
Acer platanoides L.	5	17	12	4	6	4			
Solidago canadensis L.	4	15	11	8	6	5			6
Chrysanthemum vulgare (L.) Bernh.	4	15	11	8	6	4	5	8	5
Lolium perenne L.	5	15	10	8	6	3	5	7	7
Ballota nigra L.	43	52	9	8	6	5	5		8
Chenopodium album L.	9	18	9						
Papaver rhoeas L.	2	11	9	6	6	3	5	7	6
Consolida regalis S.F. Gray	3	11	8	6	7	6	4	8	5
Silene latifolia Poiret subsp. alba (Mill.) Greuter et Burdet	33	41	8	8	6		4		7
Potentilla reptans L.	8	15	7	6	6	3	6	7	5
Arctium minus (Hill) Bernh.	2	8	6	9	5	3	5		8
Allium vineale L.	12	17	5	5	7	3	4		7
Brachypodium sylvaticum (Huds.) P.B.	4	9	5	3	5	3	5	6	6
Bromus inermis Leyss.	5	10	5	8		7	4	8	5
Crataegus monogyna Jacq.	27	32	5	7	5	3	4	8	4
Quercus robur L.	18	23	5	7	6	6			
Aegopodium podagraria L.	2	6	4	5	5	3	6	7	8
Cyanus segetum Hill.	16	20	4	7	6	5			
Fraxinus excelsior L.	5	9	4	4	5	3		7	7
Fumaria officinalis L.	2	6	4	6	6	3	5	6	7
Galium aparine L.	62	66	4	7	6	3		6	8
Pyrus communis L.	22	26	4	6	6	5	5	8	
Rhamnus cathartica L.	10	14	4	7	5	5	4	8	4
Ulmus laevis Pall.	6	10	4	4	6	5	8	7	7
Berteroa incana (L.) DC.	6	9	3	9	6	7	3	6	4
Bromus sterilis L.	57	60	3	7	6	4	4		5
Potentilla argentea L.	7	10	3	9	6	3	2	3	1
Trifolium pratense L.	5	8	3	7		3	5		
Lactuca serriola L.	7	9	2	9	7	7	4		4
Populus tremula L.	5	7	2	6	5	5	5		
Prunus spinosa L.	40	42	2	7	5	5	4	7	
Rosa L. sp.	2	4	2						
Sambucus nigra L.	42	44	2	7	5	3	5		9
Trifolium repens L.	5	7	2	8			5	6	6
Arrhenatherum elatius (L.) P.B. ex J. et C. Presl	39	40	1	8	5	3		7	7

Species	**constancy 1992**	**constancy 2019**	**Δ constancy**	**L**	**T**	**K**	**F**	**R**	**N**
Prunus avium L.	4	5	1	4	5	4	5	7	5
Euphorbia cyparissias L.	4	5	1	8		4	3		3
Geranium molle L.	17	18	1	7	6	3	4	5	4
Linaria vulgaris Mill.	7	8	1	8	6	5	4	7	5
Malva moschata L.	2	3	1	8	6	3	4	7	4
Polygonum aviculare-Gruppe	11	12	1	7	6		4		6
Ribes uva-crispa L.	2	3	1	4	5	2			6
Rumex crispus L.	14	15	1	7	5	3	7		6
Apera spica-venti (L.) P.B.	12	12	0	6	6	4	6	5	
Bryonia alba L.	2	2	0	7	6	5	5	7	6
Calamagrostis epigejos (L.) Roth	10	10	0	7	5	7			6
Urtica dioica L.	74	74	0				6	7	9
Vicia cracca L.	14	14	0	7	5		6		
Viola arvensis Murr ssp. arvensis	15	15	0	6	5				
Viola odorata L.	15	15	0	5	6	3	5		8
Agrostis stolonifera agg.	16	15	-1	8		5	7		5
Anchusa arvensis L.	3	2	-1						
Echium vulgare L.	8	7	-1	9	6	3	4	8	4
Falcaria vulgaris Bernh.	21	20	-1	7	7	6	3	9	
Lamium purpureum L.	40	39	-1	7	5	3	5	7	7
Lotus corniculatus L.	3	2	-1	7		3	4	7	3
Medicago falcata L.	10	9	-1	8	6	7	3	9	3
Plantago lanceolata L.	18	17	-1	6		3			
Poa annua L.	4	3	-1	7		5	6		8
Persicaria maculosa Gray	3	2	-1	6	6	3	5	7	7
Ranunculus repens L.	8	7	-1	6			7		7
Salix alba L.	8	7	-1	5	6	6	8	8	7
Sisymbrium officinale (L.) Scop.	6	5	-1	8	6	5	4		7
Tragopogon pratensis L.	4	3	-1						
Artemisia absinthium L.	4	2	-2	9	6	7	4	7	8
Prunus cerasus L.	8	6	-2						
Erodium cicutarium (L.) L'Herit.	4	2	-2	8	6	5	4		
Malus pumila Borkh.	10	8	-2	7	8		5		6
Plantago major L. ssp. major	11	9	-2	8			5		6
Artemisia campestris L.	7	4	-3	9	6	5	2	5	2
Bromus hordeaceus-Gruppe	16	13	-3						
Euonymus europaeus L.	21	18	-3	6	5	3	5	8	5
Hypericum perforatum L.	34	31	-3	7	6	5	4	6	4
Poa pratensis L.	32	29	-3	6			5		6
Rosa canina L.	41	38	-3	8	5	3	4		
Rumex acetosa L.	19	16	-3	8					6
Torilis japonica (Houtt.) DC.	23	20	-3	6	6	3	5	8	8
Allium oleraceum L.	13	9	-4	7	6	4	3	7	4
Matricaria L. chamomilla Grey	16	12	-4	7	6	5	5	5	5
Cirsium arvense (L.) Scop.	50	46	-4	8	5				7
Galium album Mill. ssp. album	23	19	-4	7		3	5	7	5
Heracleum sphondylium L.	32	28	-4	7	5	2	5		8
Glechoma hederacea L.	8	3	-5	6	6	3	6		7
Phleum pratense L.	9	4	-5	7		5	5		7
Prunus domestica L.	10	5	-5	7	6		5		7
Taraxacum sect. Ruderalia *Kirschner et al.*	33	28	-5	7			5		8

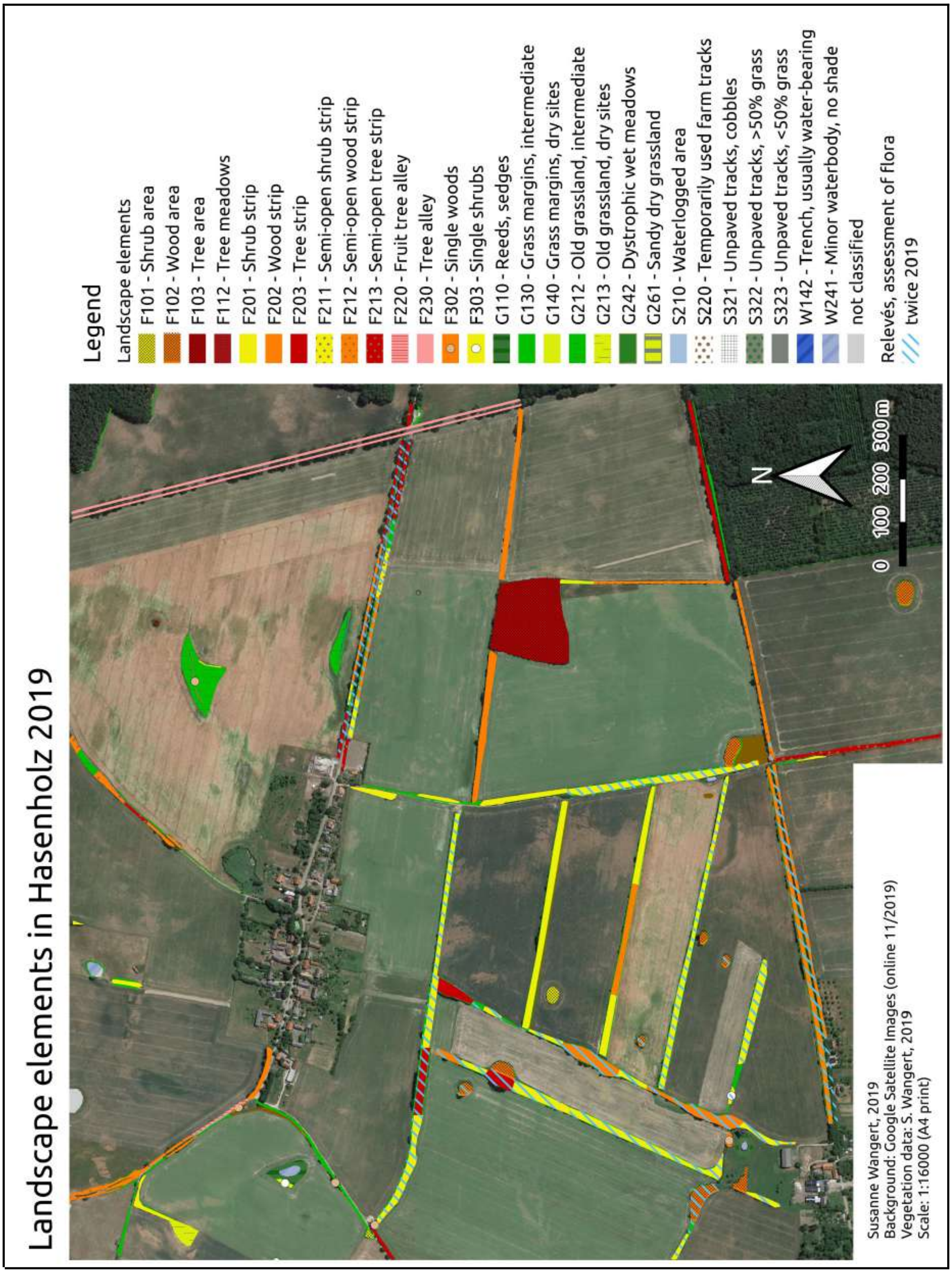
Species	constancy 1992	constancy 2019	Δ constancy	L	T	K	F	R	N
Anthriscus sylvestris (L.) Hoffm.	69	63	-6	7		5	5		8
Pyrus pyraster Burgsd.	15	9	-6	6	7		4		
Stellaria graminea L.	12	5	-7	6			5	4	3
Convolvulus arvensis L.	43	35	-8	7	6		4	7	
Rumex obtusifolius L.	15	7	-8	7	5	3	6		9
Veronica persica Poir.	11	3	-8	6		3	5	7	7
Agrimonia eupatoria L.	16	7	-9	7	6	4	4	8	4
Cerastium arvense L.	14	5	-9	8		5	4	6	4
Agrostis capillaris L.	17	7	-10	7		3		4	4
Centaurea scabiosa L.	17	7	-10	7		3	3	8	4
Veronica chamaedrys L.	23	13	-10	6			5		
Achillea millefolium L.	35	23	-12	8			4		5
Dactylis glomerata L.	38	26	-12	7		3	5		6
Daucus carota L.	16	3	-13	8	6	5	4		4
Galium verum L.	18	5	-13	7	6		4	7	3
Pimpinella nigra Mill.	18	5	-13	9	6	6	2	8	1
Lamium album L.	31	16	-15	7		3	5		9
Stellaria media (L.) Vill.	52	36	-16	6				7	8
Artemisia vulgaris L.	58	41	-17	7	6		6		8
Veronica hederifolia L.	52	35	-17	6	6	3	5	7	7
Cichorium intybus L.	32	13	-19	9	6	5	4	8	5
Medicago sativa L.	25	6	-19	8	6	6	4	7	
Equisetum arvense L.	26	6	-20	6					3
Capsella bursa-pastoris (L.) Med.	34	12	-22	7			5		6
Elymus repens L. (Gould)	76	53	-23	7	6	7			7

Table 11: Ellenberg indicator values of plants and their constancy in 1992 and 2019

9.2 Map of landscape elements / biotope types in 1992



9.3 Map of landscape elements / biotope types in 2019



9.4 Floristic survey data (Braun-Blanquet method)

Due to the large data set, the survey data is available in digital table format only, to be obtained from the author: susanne.wangert@gmail.com.