MEASURING COMPLEXITY: BIODIVERSITY IN THE WESTERN DUTCH PEAT MEADOW AREA

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ABSTRACT

Biodiversity is rapidly declining in the Netherlands, most notably because of agricultural intensification. The western peat meadow area experiences additional problems such as soil subsidence and greenhouse gas emissions. Strategies from multiple organizational levels are being implemented as the decline in biodiversity is increasingly recognized as a problem. Monitoring levels of biodiversity is important for formulating targets and tracing developments. Since measuring total species richness is expensive and time-consuming, meaningful and accurate indicators are necessary for monitoring biodiversity on a large scale. This study investigates the spatial distribution of biodiversity in the western peat meadow area of the Netherlands. Biodiversity indicators in general, for agricultural areas, and peatlands have been reviewed by studying existing literature. Subsequently, the most appropriate indicators for peat meadow areas have been determined by performing a multi-criteria analysis. This analysis resulted in the choice of the following three indicators: a single species indicator, a multi-species index, and remote sensing images of grassland use. The single species and multi-species indicators have been applied to the *Ronde Hoep* to develop spatial distribution maps of biodiversity in the area. For the single species indicator, the distribution of the godwit was used to represent overall biodiversity. The multi-species distribution was based on species types used for the Living Planet Index. The potential use of remote sensing images has been outlined through an example of a data layer developed by Ellipsis Earth of recent mowing, as a layer on herb richness is expected to be released towards the end of 2019. The resulting distribution maps of the species indicators show a concentration towards the north and center of the study area with extreme values in three areas of one square kilometer throughout all years. However, the prospective use of remote sensing suggests that this indicator is the most suitable in terms of data accessibility, resolution, and potential applications.

KEYWORDS: BIODIVERSITY, MONITORING, BIO-INDICATORS, PEAT MEADOW AREA, MULTI-CRITERIA ANALYSIS

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1. INTRODUCTION

Intensive agriculture is one of the main causes of biodiversity loss (Hole et al. 2005, Haveman and Stortelder 2006, Díaz et al. 2019). In the Netherlands, efforts from both the government and NGOs are directed towards restoring or at least stabilizing biodiversity by, among other things, increasing the sustainability of agriculture (Zijlstra et al. 2015, Nijpels et al. 2018). Organic farming is an example of how the sustainability and biodiversity of agricultural practices can be increased, as no pesticides, herbicides, and synthetic fertilizers are used, an efficient and closed nutrient cycle is required, and the amount of livestock per hectare is limited (Sundrum 2001; Bengtsson et al. 2005).

In addition to the intensification of agriculture (Wereld Natuur Fonds 2015), peatlands experience soil related issues. The peat underground, formed millennia ago in swamps that covered almost half of the Netherlands, nowadays makes for wet polders that are usually unsuitable for crop cultivation, and consequently dairy farming on grasslands is the dominant land-use type (Woestenburg 2009). The lowering of the water table, which enables faster grass growth and improved accessibility of machinery, affects the species dependent on water richness and causes the soil to subside (Woestenburg 2009). For these reasons, biodiversity is severely impacted on peatlands. Moreover, greenhouse gases such as carbon dioxide and nitrous oxide are formed through oxidization and emitted when the soil is exposed to the air after drainage (Verhoeven and Setter 2010; Bos et al. 2013; Van den Born et al. 2016). Therefore, unsustainable land use of peat meadows also contributes to climate change

Monitoring biodiversity is important as it facilitates evaluating the current state, specifying targets, measuring responses to disturbances, and assessing the effectiveness of policies (Secretariat of the Convention on Biological Diversity 2010). Mapping the spatial distribution of biodiversity in agricultural areas allows for the development of best management practices and thus the restoration of biodiversity. However, few examples or methods exist for measuring biodiversity in this area which do not require field work. Therefore, this study aims to do this by firstly analyzing appropriate ways of measuring the spatial distribution of biodiversity in the western Dutch peat meadow area through a multi-criteria analysis and subsequently applying the selected indicators.

The following research question is answered throughout this research:

> What is the spatial distribution of biodiversity in the western Dutch peat meadow area?

The following sub-questions have been formulated:

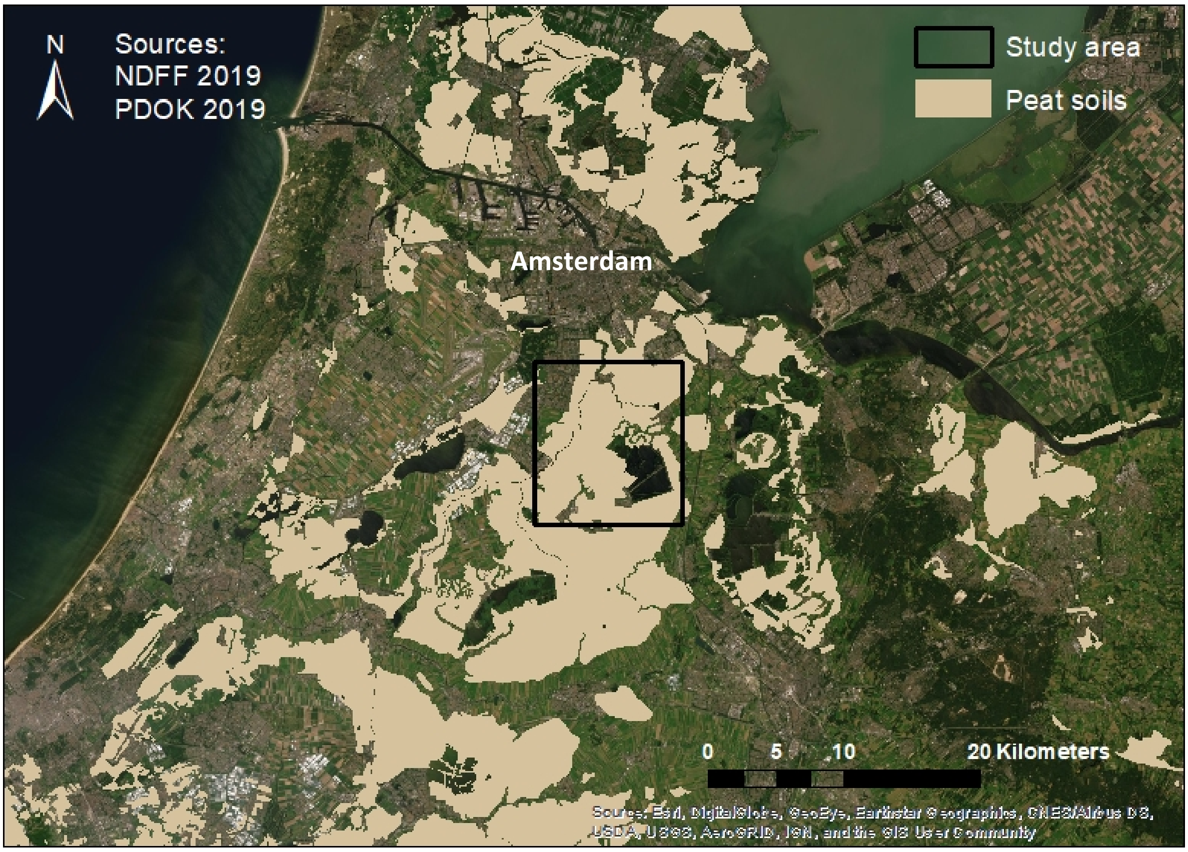
1. Which approaches are commonly applied to measure biodiversity?
2. What are typical indicators of biodiversity in agricultural and peat meadow areas of the Netherlands?
3. What are the most suitable indicators of biodiversity for peat meadow areas of the western Netherlands?
4. What are the distributional patterns of species-based biodiversity indicators in the *Ronde Hoep*?
5. How could remote sensing be used to spatially measure biodiversity in the western peat meadow area?

The first and second question have been answered by performing a literature review on indicators of biodiversity in general and how these have been used in agricultural areas and peatlands. With the use of a multi-criteria analysis (MCA), the three most suitable indicators have been determined in order to answer the third research question. Single species indicators, multi-species indices, and remote sensing images of grassland use have been found to be most suitable for the study area. Question four was answered through application of the single and multi-species indicators to a small area in the western part of the Dutch peat meadow areas. For this, data from the Dutch National Database of Flora and Fauna (NDFF) about the spatial distribution of species were used. The potential use of remote sensing to spatially measure biodiversity has been outlined with an example of preliminary data by Ellipsis Earth (2019) to answer question five.

The literature review of the first two research questions can be of informative value to stakeholders in agricultural areas. The methods of the indicator application as described for research questions 3 and 4 are useful for monitoring policies related to biodiversity in the study area, such as the Habitats and Birds Directives which together form the Natura 2000 network (Habitats Directive 1992; Birds Directive 2009). The results of research question four can be used for future studies in which forms of land management are correlated to the distribution of biodiversity. This can lead to best management practices to be determined and thereby an increase of agricultural biodiversity. Also, the distribution maps resulting from sub-question four can be useful for farmers and other stakeholders in the *Ronde Hoep*. Finally, the section in which question five is answered forms a practical example of a future use of remote sensing data. Since no formal documentation is yet available of this, this section can lead to insights for anyone affiliated with this field.

Because of data limitations, the analyses regarding sub-questions four and five are performed on an area of around 100 km^2 in the *Ronde Hoep*. This area was selected because it is almost exclusively situated on peat soils, several organic farms are located in this area, and it is an important area for meadow birds such as the godwit (Van 't Veer et al. 2010; Van Paassen 2016; PDOK 2019b). The following map shows the extent of the study area above a map of areas around the city of Amsterdam with peat soils.

Figure 1: Map of study area and peat soils around Amsterdam



2. RESEARCH CONTEXT

The research context is comprised of the literature review conducted to substantiate the current study. The definition of biodiversity used throughout this research is outlined. Next, methods of measuring biodiversity are explained for an increasingly specific focus: in general, for agricultural areas, and peat meadow areas.

2.1 BIODIVERSITY

A high level of biodiversity is a greatly sought-after property of agricultural landscapes, as it provides important ecological functions such as resilience against both biotic and abiotic disturbances as well as positive effects on nutrient and water uptake (Brussaard et al. 2007). Moreover, a diverse grazing area and thus diet of cattle has positive health effects. According to some studies, this leads to small but noticeable differences in the quality and production of milk, but this is still a contested view (Wagenaar et al. 2017). Because of these advantages, the issue of declining biodiversity is widely recognized, and restoration strategies are being implemented from many levels of political organization (e.g. Habitats Directive 1992; Birds Directive 2009; Dijksma 2014).

The term biodiversity can refer to the variation of species on many different scales. In an extensively cited paper, Noss (1990) distinguishes four levels of biodiversity: regional landscape, community-ecosystem, population-species, and genetic. In this research, biodiversity is considered at the population-species level and is defined as “species richness”. Thereby, functional roles of species are not considered.

2.2 QUALITIES OF BIO-INDICATORS

Because of the intrinsic complexity of biodiversity, it is nearly impossible to comprehensively measure it on even a small area, let alone on large spatial and temporal scales (Duelli and Obrist 2003). Therefore, it is necessary to determine suitable indicators for biodiversity. Unfortunately, a standardized system of measuring biodiversity and producing statistics in order to judge the accuracy of indicators is lacking, as Gregory et al. (2005) proclaim.

Environmental indicators are used to provide critical information about the environment (Hammonds et al. 1995). Bibby (1999) names the following nine qualities which biodiversity indicators must have: quantitative, simplifying information, user driven, policy relevant, scientifically credible, responsive to changes, easily understood, realistic to collect, and susceptible to analysis. This set is based on a paper by Hammond et al. (1995), in which a systematic approach to sustainable development is proposed. Hammond et al. (1995) highlight that a bio-indicator must be quantitative in order to measure biodiversity without ambiguity and with clear significance. Secondly, indicators must imply a set of assumptions or a model which relates them to biodiversity, and in this way simplify information. Moreover, according to Hammond et al. (1995) it has been historically shown that successful indicators are “useful to their intended audience [and] crafted to reflect the goals a society seeks to achieve”, which is summarized as “user driven”. Policy relevance is necessary for an indicator to be easily connected to national or regional policy implementations. Furthermore, scientific credibility is required to show that the underlying assumptions of the indicators are valid, such that they accurately represent biodiversity. An indicator must also be responsive to changes, to monitor the effectiveness of efforts taken by society to increase biodiversity in addition to other types of pressures. Additionally, it is required that indicators are easily understood, to improve communication to non-expert stakeholders and policy makers. Moreover, it is important for indicators that there is a balance between completeness and simplicity in the data collection, which ensures their use and collection remains realistic and feasible. This quality is referred to as “realistic to collect”. Lastly, although the indicator is ideally explained as a simple measure, it is valuable if it can be disaggregated into different components, to enable the search for possible trends and causal relationships with other factors.

2.3 General biodiversity indicators

Siddig et al. (2016) summarized insights from fourteen years of the journal Ecological Indicators to name three categories of indicators: Single species, groups of closely related species, and cross-taxa indicators. Single species indicators are important indicators of biodiversity because of the monitoring advantages (Noss 1990; Lindenmayer 1999). Flagship, umbrella, or keystone species are sometimes selected as indicators. This is either because their role in an ecosystem is expected to affect other species or because they are charismatic enough to trigger widespread attention to their conservation, and thereby to that of related other species (Linnell et al. 1999). Rare species are also often targeted for conservation purposes (Duelli and Obrist 2003), but it appears that usually their abundance is not correlated with overall species richness (Bonn et al. 2002). Important for the use of single species indicators is a clear definition of the proposed use and the area in which it functions so that false conclusion and assumptions for further use are avoided (McGeoch 1998).

Indicators which are formed by a certain taxon can be found quite frequently in current literature. For example, Sahlén and Ekestubbe (2001) found the number of dragonfly species in boreal forest lakes to be significantly connected to the overall species richness. Moreover, multiple papers have provided evidence that the species richness of beetles is correlated to the richness of other species in multiple areas in the world (Pearson and Cassola 1992; Rodríguez et al. 1998; Rainio and Niemelä 2003). Fleishman et al. (2005) experimented with models of different species from two taxonomic groups and found that the two best-fitting models explained approximately 80% of deviances in wider species richness. Those models only included indicator species from the same taxa. However, more research is needed to determine whether species from one taxonomic group can function as surrogates for species from other groups (Fleisman et al. 2005).

In the third category, biodiversity is measured by monitoring the population counts of a large group of species (Siddig et al. 2016). This indicator type most closely approaches an actual measurement of biodiversity. Many different indices have been developed to monitor large amounts of species. The Shannon-Wiener index and Simpson index, both widely used and developed in 1949, combine the species richness with the relative abundance (Shannon and Weaver 1949; Simpson 1949). The two are similar but take a different approach to the role of dominant species. A critique is that, although it is useful to include both richness and abundance, information is lost by combining those aspects into one value (Noss 1990). In a study by Albrecht (2003), the Shannon-Wiener index is used for assessing the use of arable weeds as a bio-indicator. Additionally, Rao's quadratic entropy index (1982) incorporates the expected differences between species. As the European Commission combines many different indices and data types, they make use of the Monte Carlo method for sampling error, in order to decrease uncertainty (Soldaat et al. 2017; Eurostat 2018). In the Netherlands, the Living Planet Index (LPI) is used every other year to measure biodiversity of provinces or the whole country (Wereld Natuur Fonds 2015; Jonker et al. 2017). For the LPI, the populations of a predetermined group of representative animal species are tracked in time to determine average rates of change (Loh et al. 2005). The exclusion of invertebrate and plant species is related to low availability of time-series data of these groups. Thus, the LPI measures total biodiversity only as far as vertebrate species are representative of overall trends (Loh et al. 2005). Another method is to create a mean index, which simply portrays the average population trends of the included species (Gregory et al. 2003). Multi-species indices are commonly applied to bird population counts (e.g. Louette et al. 1995; Bradford et al. 1998; O'Connell et al. 2000; Gregory and Van Strien 2010). Some advantages of birds are that they are easy to detect, their taxonomy is well-understood, they have wide-ranging habitats, and respond to environmental changes at moderate temporal and spatial scales (Gregory et al. 2005). However, Temple and Wiens (1989) proclaim birds are unsuitable for detecting triggers on overall biodiversity as there are too many influences and variables to determine causes of changes in their populations. Van Strien et al. (2009) also describe the use of Red Listed species for these kinds of indices as indicators of biodiversity. Otherwise, the indices consist of a list of species considered appropriate for the intended study area (e.g. Jonker et al. 2017) or a compilation of all possible species data which fit the study methodology (e.g. Gregory et al. 2003). As such, bird indices cover one class of organisms, while the LPI incorporates representatives of several different types of vertebrates. For example, the province of North-Holland includes mammals, breeding birds, reptiles, amphibians, fish, dragonflies, and butterflies in the LPI calculations (Jonker et al. 2017).

2.4 Biodiversity indicators for agriculture

In addition to the three categories distinguished by Siddig et al. (2016), several papers explain specific measures of biodiversity for agricultural areas. These indicators often incorporate human influences which are assumed to indicate biodiversity, instead of focusing solely on the influences of wild species. For example, landscape characteristics can be used as indicators of biodiversity. Dauber et al. (2003) found that landscape diversity and percentage cover of certain land-use types can be used as appropriate indicators for species richness when a variety of taxa is included. Noss (1990) also affirms that “landscape features such as patch size, heterogeneity, perimeter-area ratio, and connectivity can be major controllers of species composition and abundance”. Furthermore, Wascher et al. (2010) determined the spatial distribution of biodiversity of European farmlands by using datasets about land use and land structure in combination with farmland birds.

Several monitoring schemes in the Netherlands and Western Europe derive biodiversity from farm management strategies (e.g. Guijt et al. 2002; Elferink et al. 2012; Birrer et al. 2014; Stortelder et al. 2014). These share the method of combining registered farm data with survey answers in order to form an assumption of the on-site level of biodiversity (Zijlstra et al. 2015). A recently developed example is the Biodiversity Monitor for the Dairy Farming Sector (Van Laarhoven et al. 2018). This monitor is based on four pillars, each representing a different type of biodiversity: functional agrobiodiversity, landscape diversity, diversity of species, and regional diversity. The seven Key Performance Indicators (KPIs) range over these pillars and are related to several aspects of farm management. It includes, for example, ammonia or nitrogen emissions from manure or in the stable, and total greenhouse gas emissions. Also, it considers the percentage of permanent grassland, herb richness, and other types of landscape biodiversity which can be introduced to the farm management.

The concentration and richness of herbs in grasslands is also in several other studies used to indicate biodiversity in agricultural regions. The herbs can grow naturally but are also deliberately distributed throughout grasslands, as a high diversity of herbs has a positive effect on the livelihood of insects, meadow birds, and surrounding vegetation (Sanders and Westerink 2015). Kentie et al. (2013) showed significant differences between chicks of the godwit hatched in herb-rich meadows versus monocultures, namely survival rates which increased 2.5 times, a 14-16% heavier weight at fledging, and 4% bigger bills. Despite these advantages, little formal biodiversity goals or regulations have been formulated around herb richness (Sanders and Westerink 2015). The impact of mowing on farmland biodiversity is more frequently addressed and included in policy plans (Hammers et al. 2014). Frequent and early-spring mowing disturbs wildlife in grasslands as it eradicates nests and further habitat, thereby especially harming meadow birds. Furthermore, on average only 2.1% of agricultural lands in the Netherlands is used as a semi-natural area, such as ditch sides, hedges, tree edges, and dikes, compared to 96,2% which is used for tillage (Sanders and Westerink 2015). Therefore, a large percentage of overall biodiversity in agricultural areas is concentrated in and dependent on these areas. Preserving the size and sustainable management of these areas can thus increase biodiversity and connect habitats. Although these areas are irregularly distributed throughout a landscape, the quality of management can be used to depict the nature inclusiveness of farms and thus indicate a potential level of biodiversity (Van Laarhoven et al. 2018).

Since large-scale data on the indicators mentioned in the previous paragraph are lacking, they are difficult to monitor. However, the potential of remote sensing for determining the intensity of grassland use is now increasingly being studied (Franke et al. 2012; Dusseux et al. 2014; Asam et al. 2015; Sibanda et al. 2017; Bekkema and Eleveld 2018). Extensive or semi-natural use of grasslands can maintain biodiversity and plays an important role in nature conservation (Asam et al. 2015). Therefore, this measure can function as an indicator for biodiversity in grasslands. High resolution satellite constellations like Sentinel-2, RapidEye, and WorldView-3 have greatly increased the potential of grassland use classification in recent years because of their high spatial and temporal resolutions (Dusseux et al. 2014; Ali et al. 2016; Sibanda et al. 2017). Remote sensing can be applied in the context of grasslands for characterizing grass types and vegetation change, measuring productivity and other biophysical properties, assessing plant species composition, and mapping habitat and grazing intensity (Franke et al. 2012). These aspects can be derived from vegetation indices such as the Normalized Difference Vegetation Index (NDVI), Leaf Area Index (LAI), or fraction of Vegetation Cover (fCOVER) (Dusseux et al. 2014). By monitoring these indices, conclusions can be formed about the management of the grassland. For example, visible soil or steadily decreasing biomass can signal (over)grazing, and rapid decreases in biomass and crop height can be used to identify a mowing event (Dusseux et al. 2014). Although methods differ throughout academic papers, often the vegetation indices are used to classify pixel cells as one of several land use categories, such as 'Semi-natural, 'Extensively used, or 'Intensively used' (Franke et al. 2012). Larger areas can then be assessed by the percentage cover of the respective classes (e.g. Ellipsis Earth 2019). Studies by Franke et al. (2012) and Sibanda et al. (2017) have combined classifications based on remote sensing with field observations and have already reached accuracies above 80%. Although some papers mention herb richness or grass species diversity (e.g. Ali et al. 2016), mowing frequency is most often used as an indicator for intensity of land use, which might be connected to the focus of policies on the latter aspect (Sanders and Westerink 2015).

Two projects are currently using grassland intensity measurements to indicate biodiversity for grasslands in the Netherlands, in cooperation with, among others, the World Wildlife Fund and the Biodiversity Monitor. Tim Visser, a current researcher at the Wageningen University and Research is working on a web application called *Beheer op Maat* ("Customized Management") (Visser 2018). Based on soil humidity, species disturbances, landscape openness, and heaviness of vegetation, several layers are produced with the use of remote sensing techniques. These layers are used to calculate the potential for meadow birds as an indicator of the land-use sustainability. Layers with potential for herb richness can also be produced from the used data, but this is not yet available on the web application. The tool is intended to support national policy makers on the topic of agricultural sustainability once its accuracy has been verified with field research. A similar web application has been launched by Ellipsis Earth Intelligence (Ellipsis Earth 2019). In their web viewer, layers for instance indicate how recently grass has been mowed or where tree edges are located. In a telephone interview conducted on 27 May 2019, R van der Maas , the CEO of Ellipsis Earth, explained how three sources of information are combined to create indicators of grassland herb richness: the mowing frequency, the NDVI, and an artificial intelligence model which measures the heterogeneity of the grassland. Currently, the accuracy of this method is still being verified through crowdsourcing. Therefore, these details have not been outlined in a published document. All layers, both existing or still in development, are or will be accessible through the internet, both on the web viewer and as downloadable data through an application programming interface (API) (Ellipsis Earth 2019).

2.5 Biodiversity indicators in peatlands

Because of the unique soil type, specific bio-indicators for peatlands have been formulated. For example, Deru et al. (2012) determined that certain soil characteristics are positively correlated with biodiversity of peat meadow areas. For instance, they showed that on both agricultural and natural parcels an increased nitrate content positively influences the biodiversity of fauna in the soil. This same influence was measured for the amount of aggregates in the soil on natural parcels. Moreover, Dutch reports about peatlands and the decline of biodiversity often use meadow birds as characterizations of the biodiversity, as they have adapted to this type of landscape after centuries of agricultural use (Woestenburg 2009; Sanders and Westerink 2015; Van den Born et al. 2016). Woestenburg (2009) explains that most meadow birds are rare, and often a large part of the total European population resides in the Netherlands, since peat meadows form such a specific type of landscape. Especially the godwit, or *grutto* in Dutch, is frequently used as an indicator species for biodiversity, although the redshank (*tureluur*), lapwing (*kievit*), oystercatcher (*scholekster*), and skylark (*veldleeuwerik*) are used as indicators as well (Woestenburg 2009; Kentie et al. 2013; Sanders and Westerink 2015; Heijligers 2016; Van den Born et al. 2016). Sustainable agriculture is essential to the conservation of these species and thus these species are also part of the Bird Directive of the European Commission (Hammers et al. 2014). Especially the godwit is considered a species that is critical of its living environment and sensitive for disturbances (Heijligers 2016). Because of these qualities, the species is considered to reside in areas with a high level of biodiversity, which makes it a suitable indicator for biodiversity in peat areas.

In addition to meadow birds, Sanders and Westerink (2015) name certain insects and mammals as important to the biodiversity in agricultural grasslands. Herb-rich meadows are especially important for several farmland butterflies (Van Diepen-Loos et al. 1998; Sanders and Westerink 2015). Because of the agricultural intensification, these species are also in decline and are therefore included in species protection laws and the Red List (Heijligers 2016). Similarly, hares are affected by frequent mowing and increasing parcel sizes leading to smaller semi-natural edges (Sanders and Westerink 2015). Moreover, many species of dragonflies appear in the peat meadows, especially around lakes, along ditches, or in the swamp-like parts of the peat meadow areas (Beltman et al. 2012). Because of their decline, the foundation for Dutch Butterfly Conservation monitors butterflies and dragonflies and publishes yearly reports on the population trends (Van Swaay et al. 2018).

3. METHODOLOGY

Based on the indicators and qualities found through studying existing literature, a multi-criteria analysis (MCA) has been performed as the first part of the methods section. The guidelines of the paper "Multi-criteria analysis: a manual" by Dodgson et al. (2009) have been used to structure this process. The second section describes the elaboration on the three most suitable indicators resulting from the MCA. Two of these indicators have been applied to the *Ronde Hoep* to test their effectiveness and to produce an indication of the spatial distribution of biodiversity. For the third indicator, an example has been given of potential use for different data, as the required data are expected to be released after the submission of this research.

3.1 INDICATOR SELECTION WITH A MULTI-CRITERIA ANALYSIS

In the literature review section, the most important types of indicators found in literature have been introduced. As mentioned earlier, indicators for biodiversity intrinsically simplify reality, especially the ones based on a single feature. Ideally, all indicators would be tested in terms of reliability on the study area over a long period of time. This would then be done by comparing trends of the indicators with trends of more complex measurements of biodiversity which have been already verified as reliable (e.g. Rodríguez et al. 1998). A (statistical) analysis could then confirm whether the studied indicator is suitable. Because of limitations regarding the availability of time and data, this type of analysis goes beyond the scope of this study. Alternatively, a multi-criteria analysis (MCA) is performed to determine suitable indicators from the ones found in academic sources in a structured yet time-effective manner.

According to Dodgson et al. (2009), the role of MCA techniques is "to deal with the difficulties that human decision-makers have been shown to have in handling large amounts of complex information in a consistent way". Using an MCA thus provides structure and transparency about a decision-making process when many factors must be considered. A performance matrix is a standard form of an MCA, in which each row describes an alternative and each column describes the performances of the alternatives for a criterion (Dodgson et al. 2009). Calculating the weighted sum of these scores leads to a ranking of alternatives and can thus lead to a calculated decision.

Dodgson et al. (2009) name eight steps for creating a multi-criteria analysis, as shown in Table 1. These steps are executed in the following sections with corresponding numbering. In this study, options are referred to as alternatives.

Table 1: Steps in a multi-criteria analysis (Dodgson et al. 2009)

1. Establish the decision context.
2. Identify the options.
3. Identify the objectives and criteria that reflect the value associated with the consequences of each option.
4. Describe the expected performance of each option against the criteria.
5. 'Weighting'. Assign weights for each of the criteria to reflect their relative importance to the decision.
6. Combine the weights and scores for each of the options to derive and overall value.
7. Examine the results.
8. Conduct a sensitivity analysis of the results to changes in scores or weights.

3.1.1 DECISION CONTEXT OF THE MULTI-CRITERIA ANALYSIS

The objective of the MCA is in line with that of this research, which is to provide guidance to the process of determining best management practices for farmers to increase agricultural biodiversity. It is necessary to select appropriate indicators so that biodiversity can accurately and effectively be measured. Only then, it will be possible to connect levels of biodiversity to land use and thus farm management.

3.1.2 Identification of alternatives

In addition to the three categories of indicators identified by Siddig et al. (2016), four indicator groups have been formulated based on the review of studies about biodiversity indicators for agriculture and peatlands. This has resulted in the following list of indicators as the seven alternatives for the multi-criteria analysis:

- Single species
- Taxon
- Multi-species index
- Soil components
- Landscape organization
- Management index
- Remote sensing

Some alternatives encompass multiple, comparable types of indicators, as these would receive the same scores because of their similarities. For example, landscape organization includes land-use cover as indicators for biodiversity, in addition to heterogeneity, patch size and other characteristics of non-natural areas. Moreover, the Single species alternative represents indicators for flagship species, keystone species, umbrella species, and rare species, in addition to species whose population counts have been determined to be correlated to overall biodiversity. Lastly, multi-species indices can consist of many different assemblages of species.

3.1.3 Identification of criteria

The following ten criteria have been formulated for the MCA:

1. Quantitative
2. Simplifying information
3. User driven
4. Policy relevant
5. Scientifically credible
6. Responsive to changes
7. Easily understood
8. Realistic to collect
9. Susceptible to analysis
10. Spatially specific

The first nine criteria are derived from the nine properties of effective indicators for biodiversity which Bibby (1999) formulated based on a paper by Hammonds et al. (1995), as explained in the literature context section. *Spatially specific* has been added as the tenth criterion to this list. This quality is important for answering the current research question but is not required for detecting the overall level of biodiversity in a defined area. This can explain why this quality is not always necessary for general bio-indicators. Bibby (1999) points out, after listing the nine criteria, that finding an indicator which possesses all qualities is challenging. Therefore, this research does not intend to find indicators with a perfect score, but to combine several of the most appropriate alternatives. Moreover, it will be considered that the three indicators chosen to be applied compensate for respective shortcomings by analyzing absolute differences in scores.

3.1.4 Score assignment

Because the criteria are not quantitative in nature, the assigned scores are based on an ordinal scale. The criteria are all described in the same ordinal scale, so there is no need for normalization of the scores. Lastly, the scores vary between 1 and 3, where 1 stands for the lowest score and 3 for the highest. A larger range of scores would provide an unjust indication of precision, as they represent only estimations of reality.

Table 2 shows the scores that have been assigned to all alternatives based on the criteria. In the following sections, the scores of each criterion will be discussed.

Table 2: Score assignment of biodiversity indicators

	Quantitative	Simplifying information	User driven	Policy relevant	Scientifically credible	Responsive to changes	Easily understood	Realistic to collect	Susceptible to analysis	Spatially specific	Total:
Single species	3	2	3	3	1	3	3	3	1	3	**25**
Taxon	3	2	3	2	3	3	2	2	1	3	**24**
Multi-species	3	1	1	3	3	3	2	2	3	3	**24**
Landscape organization	1	3	2	3	2	1	1	2	1	2	**18**
Soil components	3	3	1	3	2	2	3	1	3	3	**24**
Management index	1	3	3	2	2	2	3	1	3	2	**22**
Remote sensing	3	3	3	3	3	2	2	2	2	2	**25**

3.1.4.1 Quantitative

The first criterion in the MCA is the level to which the indicator is quantitative. As this is criterion can either be fulfilled or not, the score of 3 stands for the indicator being quantitative, and a 1 means it is not. Most of the alternatives have received a score of 3 for this criterion, as the species-based indicators are described in concentrations, and remote sensing images are translated to index values or to concentrations of land-use classes. The *management index* is described as a number which represents a quality instead of a quantity (Zijlstra et al. 2015). Moreover, *landscape organization* is a qualitative measure of how well it supports biodiversity which is not easily described numerically. Therefore, these two indicators have received a 1.

3.1.4.2 Simplifying information

Second is the property of simplifying information. Here, the indicators are scored based on whether they are (3) very simple, (2) slightly complex, or (1) complex. *Multi-species indices* convey a portion of the biodiversity in a region, but still combine the population count of a lot of species. Therefore, it is barely any less complex than biodiversity itself and thus it received the score of 1. The *management index* also combines many factors to produce a score of biodiversity. In this sense, biodiversity is not translated into a simpler measure either, which is why it has a score of 1 as well. The *taxon* indicator is simpler than the multi-species indicator because of the significantly smaller amount of species included. Still, it is more complex than a *Single species* indicator, and therefore it received a score of 2. The *Single species, soil components,* and *remote sensing* indicators indicate biodiversity with a concentration or index value of a single aspect of the landscape, resulting in the high score. *Landscape organization* also portrays a single aspect, so forms a significant simplification of biodiversity. These indicators have therefore received a score of 3.

3.1.4.3 User driven

Thirdly, an indicator must be user driven. This is interpreted in the current case study as: *how readily can a farmer adapt his management to improve their performance on this criterion.* This thus depends on the costs and other factors related to the indicator. Therefore, the indicators are scored based on whether they are easily adapted to (3), somewhat difficult to adapt to (2), or very difficult to adapt to (1). Because the requirements for increasing the survival chances of a *single species* are often well-defined, it is relatively easy to adapt to these needs. Although *taxa* encompass multiple species, these are very similar are would likely require the same adaptations as individual species within the taxon. Therefore, both these indicators have been scored a 3. The *management index* is developed to relate management from a farmer to biodiversity. Similarly, the *remote sensing* images mostly analyze management strategies such as mowing frequencies. Thus, these two indicators have received a 3 as well. As a farmer, it is possible to adapt the *landscape organization*, for example by preserving semi-natural areas and building connections between them. Moreover, more landscape diversity could be introduced by varying the land use of adjacent areas. These measures, however, are costly and would require a transition time, so are likely not to be welcomed by conventional farmers. Therefore, this indicator has received a score of 2. Lastly, *multi-species indices* and *soil components* have been scored with a 1. Since *multi-species indices* incorporate many species, it is very difficult to adapt to all the species' varying requirements. Finally, because of the many factors which influence a soil next to the deliberate input of a farmer, s*oil components* are difficult to alter such that the whole equilibrium shifts.

3.1.4.4 Policy relevant

An indicator for biodiversity should also be policy relevant. For this criterion, a 1 stands for the indicator not being represented in regional or national policies, a 2 for the indicator being represented in policies occasionally or indirectly, and a 3 means that the indicator is well-represented in policies. The latter is the case for most indicators (e.g. Ministerie van VROM 2006; Meerburg and Korevaar 2009; Provincie Noord-Holland 2010; Sierdsema et al. 2013; Van Dam 2017), except *taxa* and *management indices*. In these policies, usually either a single indicator species or a large group of species is addressed, not a group of closely related species. Because of the obvious relation to the other two species-based indicators, this indicator receives a score of 2. The management index is used to measure biodiversity based on many different components. These components are often represented in policies related to increasing biodiversity, for example by switching to more extensive methods, but the index itself is not (yet) used in policies. Therefore, this indicator has been scored with a 2 as well.

3.1.4.5 Scientifically credible

Scientific credibility is required for the acceptance of a biodiversity indicator. This criterion is judged based on whether, according to existing studies and reports, the indicator and biodiversity are (3) very (cor)related, (2) somewhat correlated, or (1) only slightly correlated. Out of all the remaining indicators, *multi-species* indices come closest to an actual measure of biodiversity, as they incorporate the populations of many species. In literature, it is often used as the measure for biodiversity to which other indicators are compared (e.g. Rodríguez et al. 1998; Fleishman et al. 2005). For this reason, this indicator type has received a high score (3). Moreover, several *taxa* have been scientifically demonstrated to be correlated with biodiversity and to form an accurate bio-indicator (Pearson and Cassola 1992; Rodríguez et al. 1998; Sahlén and Ekestubbe 2001; Rainio and Niemelä 2003). Therefore, this indicator also received a high score (3). Lastly, mowing frequencies and herb concentrations have been widely accepted in Dutch reports to have a strong relation to biodiversity (Hammers et al. 2014; Sanders and Westerink 2015). Therefore, *remote sensing* has also received a score of 3. Although the importance of the use of *Single species* indicators is underlined throughout academic literature, it is also widely acknowledged that their use is controversial and has not yet proven to be accurate, which is why they received the lowest score (1) for this criterion (Noss 1990; Lindenmayer 1999). Certain studies have related the remaining indicators to biodiversity (e.g. Dauber et al. 2003; Van Eekeren et al. 2010; Deru et al. 2012; Van Laarhoven et al. 2018), but more scientific research is required to support these claims. For this reason, these indicators have been scored with a 2.

3.1.4.6 Responsive to changes

An indicator of biodiversity needs to be responsive to changes, so that changes and trends can be picked up quickly enough for appropriate responses to be effective. This responsiveness can also apply to spatial changes, meaning that small differences in the values more accurately translate to an actual difference in biodiversity. The indicators have been categorized based on whether they can respond (3) almost immediately, (2) within a year or season, or (3) after a year or season. Living organisms are very directly dependent on the resources in the landscape. For example, Reice (1985) found that the population densities of macroinvertebrates declined immediately after experimental disturbances. Fernández-Juricic (2000) also found almost immediate responses from bird populations after disturbances from pedestrians. *Landscape organization* is dependent on human choices which do not have to be related to the level of biodiversity. Therefore, biodiversity responds to changes in the indicator instead of the other way around. Although extensive research on response times is lacking, Dale et al. (1994) show that it takes multiple years for faunal populations to reach a new equilibrium after land-use change in the Central Amazon. Moreover, Metzger et al. (2009) discuss the time-lag in biological responses to landscape changes and conclude that long-term effects must be considered to avoid conservation decisions. Therefore, this indicator has received a low score (1). This same logic applies to the *management index* and *remote sensing*. However, since the management strategies are more directly applicable to the entire land the biodiversity is likely to respond more quickly than after a change in the landscape organization. Therefore, these alternatives have been scored with a 2. Lastly, soil components are more directly related to a region as well. For this reason, the biodiversity of the area is likely to respond within a year to significant changes to the soil, resulting in a score of 2.

3.1.4.7 Easily understood

Bibby (1999) explains that it is important for indicators to be easily understood, so that non-expert stakeholders and policy makers “have a sense of ownership and sympathy”. This criterion is scored based on whether the meaning of the indicators (3) are easy to understand for non-experts, (2) require some expert knowledge, or (1) are difficult to understand for non-experts. *Landscape organization* is the only indicator to have received a score of 1, because the values, concepts, and relation with biodiversity can be difficult to grasp. Next, *taxon* and *multi-species index* have received a score of 2, because they both incorporate multiple species which result in an index value of the combined trends. This value is understandable with some explanation, but not completely straight-forward. *Remote sensing* has received a score of 2 as a compromise between the complex index values and easily understood percentage cover. The *Single species indicator* and *soil components* have received a high score, because the units of concentration or frequency are easily understandable. Moreover, the score for biodiversity as is given for the management index is very intuitive, so this indicator has received a high score as well.

3.1.4.8 Realistic to collect

Considering the scope and time span of the current research, it is crucial that the required data have already been produced and is publicly available. For this criterion, a score of (1) stands for data which are nearly impossible to obtain, a (2) means that the data are available but costly and/or timely to obtain and process, and a (3) means the data are easy to obtain. Because of national and supranational monitoring policies that are already in place, the populations of many species are counted or estimated on a yearly basis (CBS 2019). Therefore, *Single species* indicators have received a high score. *Landscape organization* is generally monitored in the Netherlands, for example in PDOK, the online geo-database of the Dutch government (PDOK 2019a). Therefore, *landscape organization* also received the score of 3. Detailed species population data are, however, not publicly accessible. In order to obtain population data from the National Database of Flora and Fauna in the Netherlands, a payment is required per square kilometer, which decreases the ease with which it is collected. For the *taxon* and *multi species* data multiple species are required, so these have received a score of 2. Moreover, *remote sensing* images are frequently updated and freely available, but require a lot of processing before usable data are produced and are dependent on the absence of clouds. This results in a score of 2 for this indicator. Finally, *soil components* and *management strategies* require very specific data that need to be measured on each plot in the study area. Much of these data are private and can only be accessed by the landowners and would otherwise be very time-consuming and costly to obtain for a large area. For this reason, these two alternatives received the lowest score.

3.1.4.9 Susceptible to analysis

The last criterion listed by Bibby (1999) is susceptibility to analysis. The indicators are differentiated based on whether they are connected to (1) only one component related to biodiversity, (2) a small number of components related to biodiversity, or (3) a large number of components related to biodiversity. *Single species indicators* and *landscape organization* have received a score of 1 for this criterion, because they only related to one component of biodiversity, namely the indicator itself. *Taxa* do consist of more species than Single species indicators, but because these organisms are so closely related, they are given the same score as Single species indicators. *Remote sensing* images are related to both the management of the land and the response of the vegetation, which is why it has received a score of 2. The remaining three indicators have received a high score. For *multi-species indices*, this is the case because they nearly measure the actual level of biodiversity. *Soil components* can be affected by influences from many different sources, like land management and use, precipitation, and water table. Lastly, a management index inherently considers many different components of biodiversity. Therefore, it has received a score of 3.

3.1.4.10 Spatially specific

As this study researches the spatial distribution of biodiversity, it is required that the indicators are spatially specific. Three categories have been formulated, namely (1) high resolution (<1 km) and flexible locations, (2) high resolution (<1 km), but dependent on predefined borders, and (3) low resolution. All the considered bio-indicators have a high resolution. However, the *landscape organization* and *management index* are largely dependent on existing plot borders. The *remote sensing* results, despite the high resolution, are likely to be the same throughout plots because of the same land management. For this reason, these three alternatives have received a score of 2 instead of 3.

3.1.5 Weighting

Weighting is an important yet subjective step of an MCA. Many of the weighting methods that exist (e.g. Ranked Sum, Ranked Order Centroid, Rank Reciprocal; see Roberts and Goodwin (2002) for a summary) are based on a predetermined order of importance of all the criteria. Although there are methods by which the process of ranking can be structured, such as the swing weighting method (Dodgson et al. 2009), it remains dependent on personal preferences. Therefore, an example of an MCA by Van Herwijnen et al. (1993) is followed, in which criteria are divided into two groups which received collective weights. The criteria have been distinguished based on whether they are important for the measurement of biodiversity or for the application of such measurements, hereafter referred to as IM and IA, respectively. For the sensitivity analysis, the weight distributions have been compared to a scenario with equal weights and another in which the weights have been inverted between the two categories, similarly as done in the paper by Van Herwijnen et al. (1992). In addition, scenarios with larger differences between the two categories are considered. Lastly, the interdependences between scores have been calculated.

The following table shows the distinction of both groups:

Table 3: Distinction of indicator qualities into two groups

IM	IA
Quantitative	User driven
Spatially specific	Policy relevant
Realistic to collect	Easily understood
Scientifically credible	Susceptible to analysis
Responsive to changes	Simplifying information

Measuring the spatial distribution of biodiversity requires data which are quantitative, spatially specific, and realistic to collect for such a study to be completed. The indicators must be scientifically credible, such that the measurements are reliable. Finally, an indicator needs to be responsive to changes to ensure that the spatial and temporal differences can be detected and the data are up to date. These criteria have thus been identified as IM, important for measuring biodiversity.

The remaining five criteria have been marked as important for the application of results from the measurement of biodiversity (IA) as they are related more to further uses such as policy implementation or communication to (non-expert) stakeholders. Simple and easily understood indicators allow for faster and easier communication. An indicator which is policy relevant and user driven will be implemented more quickly in policies and in practice. Lastly, susceptibility to analysis is useful for establishing causal factors between biodiversity and management but not as much for the initial measurement.

The current study focuses on the measurement of the spatial distribution of biodiversity and does not explore further applications. Therefore, the IM indicators are prioritized and have received higher weights. The scenario with an emphasis on application will be explored as a sensitivity analysis.

The selected ratio between the weights of the groups Important for measuring biodiversity and Important for the application of results is 1.5 and 1. Although this ratio is selected somewhat arbitrarily, it forms a compromise between a significant yet not exaggerated difference between the categories. Since the groups are equal in size, this leads to the weights of 0.12 and 0.08 of all criteria in the respective categories.

3.1.6 Combining scores and weights

Multiplying the scores in Table 2 with the weights according to an IM, equal weights, and IA scenario lead to the rankings and scores shown in Table 4. The complete effects tables can be viewed in Appendix 1. The IM scenario is considered most realistic for this research, so has been used to guide the selection of indicators.

Table 4: Rankings of performance matrices of IM, equal weights, and IA scenario

IM	Score	Equal Weights	Score	IA	Score
1. Single species	2.52	1. Remote sensing	2.5	1. Remote sensing	2.52
2. Remote sensing	2.48	2. Single species	2.5	2. Single species	2.48
2. Taxon	2.48	3. Taxon	2.4	3. Soil components	2.44
2. Multi-species	2.48	3. Multi-species	2.4	4. Taxon	2.32
3. Soil components	2.36	3.Soil components	2.4	4. Multi-species	2.32
4.Management index	2.08	4. Management index	2.2	4. Management index	2.32
5.Landscape organization	1.76	5.Landscape organization	1.8	5. Landscape organization	1.84

3.1.7 Examining results

Resulting from the rankings, the indicator Single species comes out as best for the IM scenario. Remote sensing, Taxa and Multi-species indices share the second place. In section 3.1.3 it is explained how the indicators should complement each other by compensating for lower scores. In order to incorporate this, the sum of the absolute differences in scores between the Multi-species, Taxa, and Remote sensing indicators and the Single species indicator was calculated for the score assignments table. From the resulting differences (Single species – Taxon: 6; Single species – Remote sensing: 8; Single species – Multi-species: 10), it is concluded that the Multi-species and Remote sensing indicators are more appropriate to combine with Single species. This is supported by the fact that, between the three indicators with the same score, the largest difference exists between the Multi-species index and Remote sensing as well.

3.1.8 Sensitivity analysis

The last step of a multi-criteria analysis (MCA), following Dodgson et al. (2009), is a sensitivity analysis. In this step, the uncertainties for the scores or weights are considered, to see if the results are impacted. This is meant to reduce or at least portray the effect of the unavoidable subjectivity.

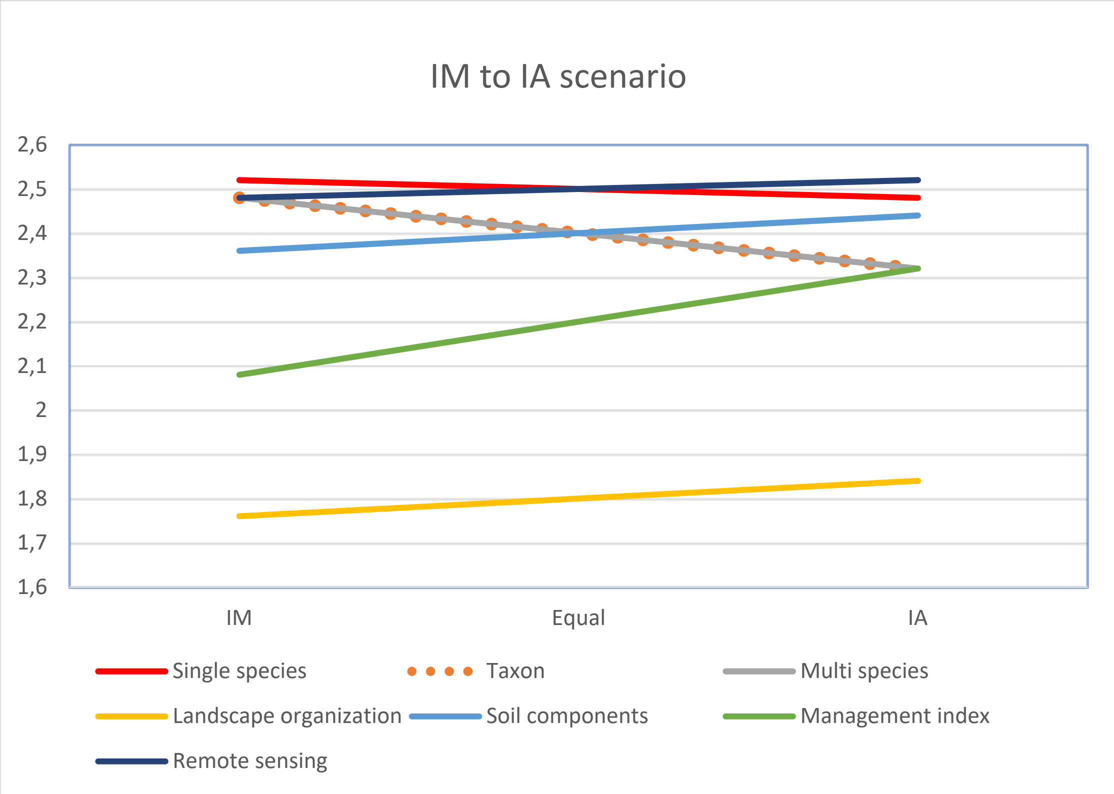
3.1.8.1 Prioritization of criteria with weights

The sensitivity analysis of the weights of the performance matrix consists of two variations, namely that of the distribution of the weights and the intensity. For the variation of weight distributions, the results are first compared to results from a scenario with equal weights. Secondly, the weights between the IM and IA category are inverted, such that IM criteria receive a weight of 0.08 and IA criteria weigh 0.12. A similar sensitivity analysis, where weights have been interchanged between two categories, has been performed by Van Herwijnen et al. (1992). The results of these scenarios are shown in Table 4 in section 3.1.6.

The differences of the three tables are best portrayed in a graph, as shown in Figure 2 below. The graph shows that the ranking of Soil Components compared to Taxa is the most significant difference for the various weights. Also, Remote sensing and Single species shift places, but the difference remains small. These shifts occur when the weights are equal. Moreover, Multi-species and Taxa receive the same score as Management indices in the inverted weighting scenario.

These results imply that biodiversity research for which the application of the results has priority over facilitating the measurement process has more use of Soil composition as a bio-indicator than a Multi-species index or Taxon indicator. Remote sensing and Single species indicators remain the most appropriate indicators, so would both be used in a selection of three indicators.

Figure 2: Sensitivity analysis: Inversion of weights Very Important and Slightly Important



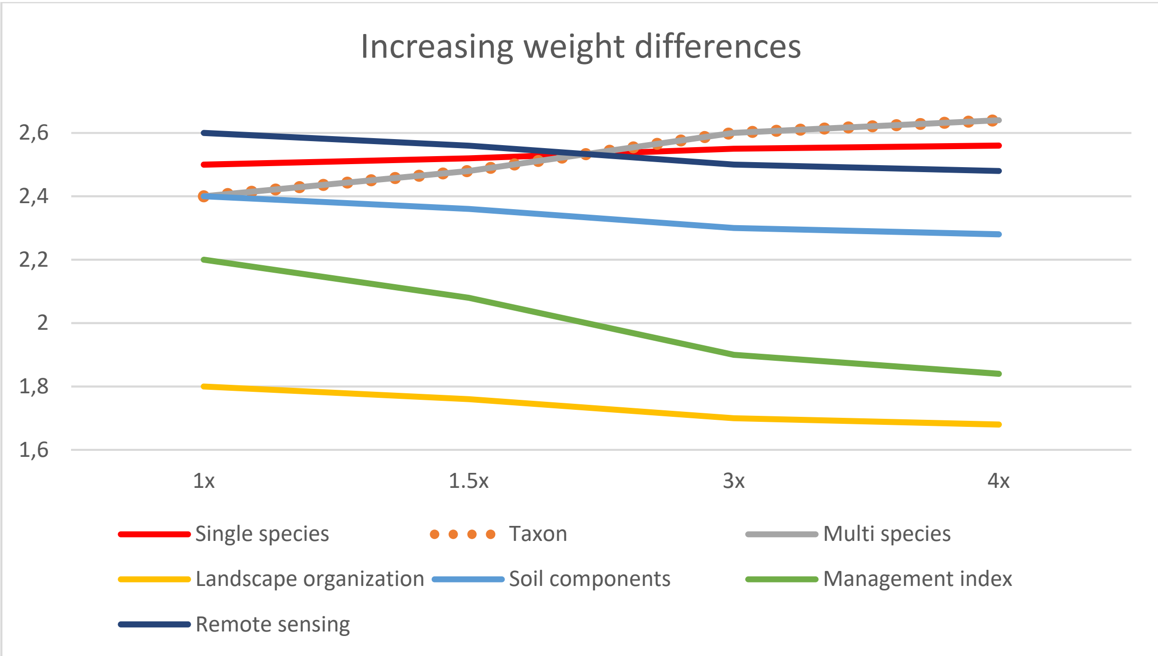
3.1.8.2 Weight difference between categories

Secondly, the weight difference between the IM and IA category are intensified, such that the ratios become 3 to 1 and 4 to 1. The following table and graph show these results.

Table 5: Rankings resulting from effects tables with increasing weight differences in IM scenario

Equal Weights	Score	1.5x	Score	3x	Score	4x	Score
1. Remote sensing	2.5	1.Single species	2.52	1. Taxon	2.6	1. Taxon	2.64
2. Single species	2.5	2. Remote sensing	2.48	1. Multi-species	2.6	1. Multi-species	2.64
3. Taxon	2.4	3. Taxon	2.48	2. Single species	2.55	2. Single species	2.56
3. Multi-species	2.4	3. Multi-species	2.48	3. Remote sensing	2.45	3. Remote sensing	2.44
3.Soil components	2.4	4. Soil components	2.36	4. Soil components	2.3	4. Soil components	2.28
4. Management index	2.2	5.Management index	2.08	5. Management index	1.9	5. Management index	1.84
5.Landscape organization	1.8	6.Landscape organization	1.76	6. Landscape organization	1.7	6. Landscape organization	1.68

Figure 3: Sensitivity analysis: Increasing weight differences between criteria



It is visible that the four alternatives with the highest scores, namely Remote sensing, Single species, Taxa and Multi-species, change order among themselves and that for each scenario several alternatives have received the same scores. Moreover, the difference between these alternatives and the three remaining ones increases. In the scenarios with increased weights the Taxon indicator comes out as best, and hence would have been selected over Remote sensing. However, because of the calculated similarities between Taxon and all three other indicators, it could still be argued that Remote sensing could be selected so that the indicators compensate for the low scores of other indicators.

3.1.8.3 Interdependence

Many academic studies in which an MCA is performed, use the Monte Carlo method to determine the sensitivity of the scores in the effects table (e.g. Van Herwijnen et al. 1992; Soldaat 2017). However, because of the ordinal scores used in this research, this method would not be useful. Namely, the scores do not represent actual numbers, so therefore it is not sensible to define the uncertainty in terms of a percentage of the scores. What can be calculated, however, is the amount to which the scores are interdependent. If many of the criteria are interdependent, this can cause over- or underestimations of scores and thus distort the resulting rankings (Van Herwijnen et al. 1995). The following correlation matrix and graph were produced by DEFINITE, a software for performing multi-criteria analyses (Van Herwijnen and Janssen 2004).

The figures below show that most criteria are not interdependent. There are no scores above (-)0.8 and only four pairs have a score above (-)0.7. Three of these include responsive to changes, namely the pairs with quantitative, simplifying information, and spatially specific. Moreover, quantitative and simplifying are shown to be somewhat correlated. However, because of this low number of correlating criteria, the criteria are considered to be fairly independent and thus the effects table is deemed robust.

Correlation

Effects	Quant.	Simpl. info.	User driven	Pol. rel.	Scient. cred.	Resp. to chan.	Eas.under st.	Real. to coll.	Susc. to anal.	Spat. spec.
Quant.		-0.487	-0.0199	0.3	0.1494	0.7279	0.3607	0.3318	0.0434	0.7303
Simpl. info.			-0.1811	-0.0752	-0.1558	-0.7121	-0.0341	-0.4961	0.0254	-0.6668
User driven				0.5448	0.1484	-0.3182	-0.3182	-0.1907	0.3148	0.2305
Pol. rel.					-0.2177	-0.2177	-0.2177	0.3318	0.0434	0.0913
Scient. cred.						0.0259	-0.3593	-0.0392	0.3417	-0.1306
Resp. to chan.							0.6148	0.2176	0.0854	0.7327
Eas.underst.								-0.4615	0.4386	0.3975
Real. to coll.									-0.68	0.1588
Susc. to anal.										-0.0793
Spat. spec.										

Figure 4: Correlation matrix of the ten criteria used for the MCA

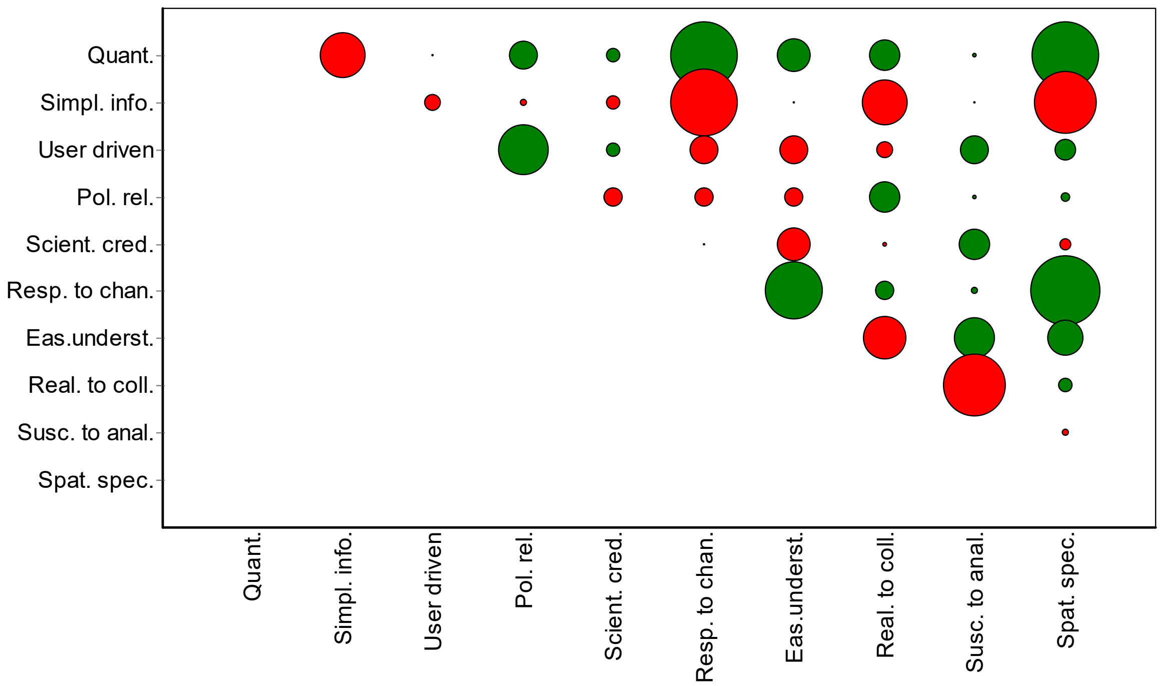


Figure 5: Correlation graph of the MCA criteria

3.1.8.4 Conclusion

Based on these sensitivity analyses it is concluded that the results are quite robust considering the assigned weights and interdependence among criteria. In the first analysis, the two most suitable alternatives, Single species and Remote sensing, remain the same in each set of results, although in the IM scenario Remote Sensing shares the second place in the rank order with Taxon and Multi-species indicators. The ranking of Soil Components forms a more significant change. However, considering that Soil components, Taxa and Multi-species indicator all received the same scores with equal weights, it is to be expected that weighting has a big influence on their respective order. Furthermore, the change in rank order of Soil Components only occurs when the weights are completely equal, so a similar distinction but with lower weights would not change the ranking. Because the weight difference between the more and less important criteria is more appropriate for this study than the other two, the original results are used for the selection of most appropriate indicators.

In the second sensitivity analysis, the highest-scoring alternatives shift in ranking, which results in Remote sensing becoming less appropriate and Taxon becoming the most appropriate alternative. However, in section 3.1.3 it was explained that the selected indicators should complement each other in terms of high and low scores on some criteria. Following this reasoning, it could still be argued to include Remote sensing, because of the clear similarities between the species-based indicators types. From these results it becomes clear that more certainty about the difference between the different weights groups would be advisable for a more accurate selection of the third indicator type. For the current research, the Taxon indicator is omitted from the selection to ensure mutual compensation between indicator groups.

3.2 Application of indicators

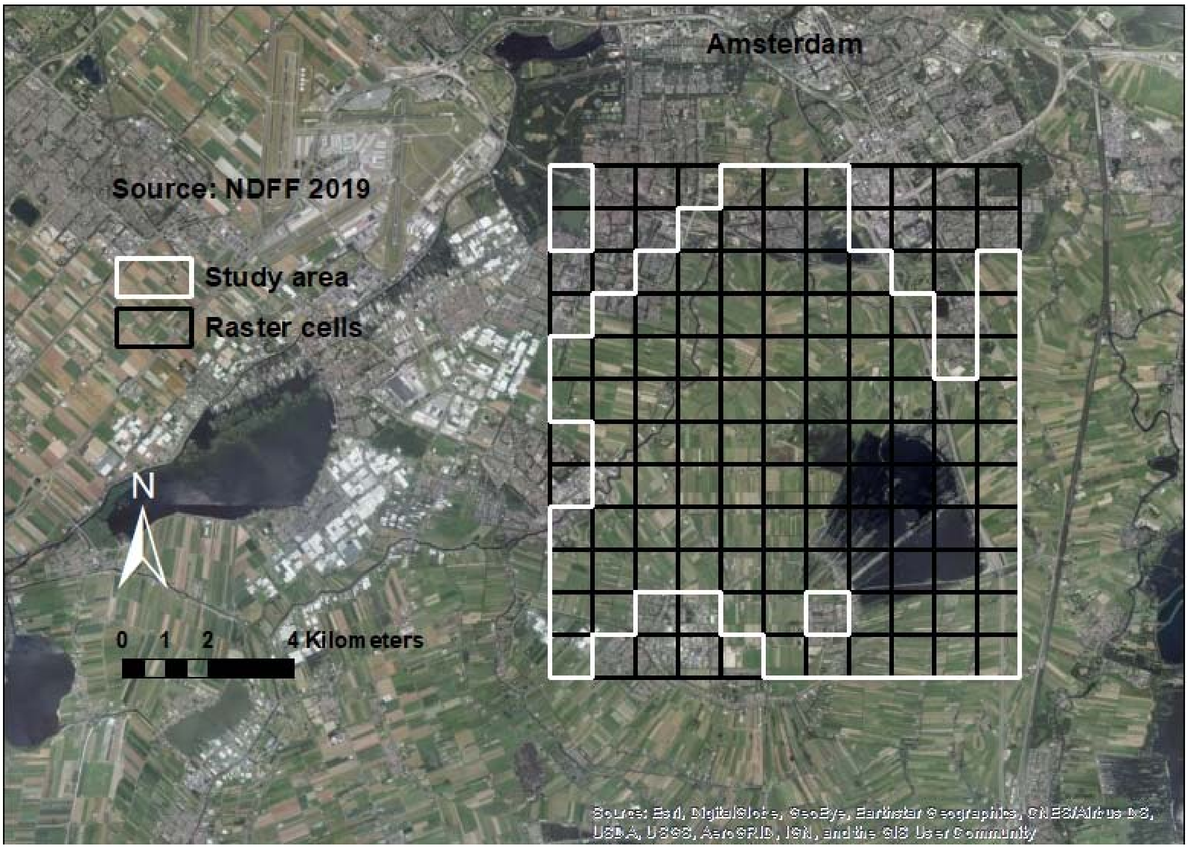
Resulting from the MCA in the previous section, a Single species indicator, a Multi-species index, and Remote sensing have been selected as the most suitable indicators for the current research. Thus, an elaboration on these indicators follows in the next section. Because of time and data constraints, the application of the Remote sensing indicator is limited to a practical example of potential use. The Single species indicator and Multi-species index have been applied to the *Ronde Hoep*.

3.2.1 Study area and data collection

3.2.1.1 Study area

Because of budget constraints, the bio-indicators have not been applied to the entire western peat meadow area. An area below the city of Amsterdam of approximately 100 square kilometers surrounding the *Ronde Hoep* has been selected as the study area. This polder area is almost exclusively situated on peat soils, accommodates several organic farmers, and is home to several types of meadow birds (Van 't Veer et al. 2010; Van Paassen 2016; PDOK 2019b). The extent of the study area encompasses 132 square kilometers (11 by 12 kilometers), divided into cells of 1 by 1 kilometer. The 29 cells which cover mostly urban or forest areas have been omitted, such that 103 square kilometers have been studied in total. The following map shows the study area and the raster cell division.

Figure 6: Map of study area and raster grid cells



3.2.1.2 Data collection and preprocessing single species and multi-species

The observation data for both species indicators has been retrieved from the National Database of Flora and Fauna (NDFF) (NDFF 2019). In this database, millions of observations of various species have been compiled, standardized, and validated. Observations have been obtained for four species types:

mammals, birds, butterflies, and dragonflies. The selection of first three species types was based on the species lists used to calculate the Living Planet Index for agricultural areas in the Netherlands (Wereld Natuur Fonds 2015; CLO 2018). Dragonflies have been included because of their occurrence in peatlands, as described by e.g. Stip (2013) and Beltman et al. (2012), and their extensive monitoring network (CBS 2019). All the collected observations have taken place within the years 1999 up until 2018. This period has been selected because most of the monitoring networks for these species types were established throughout the last decade of the 20th century, yet by taking a period 20 years it is still possible to look at time trends (CBS 2019). Although using 1999 as a starting year may suggest that the biodiversity levels of that year were optimal and should be regained, this is not the intention. By requirement of the NDFF, the observation data have been aggregated with a resolution of 1 kilometer. Furthermore, it is important to note that the species selected for these indicators are specific for the study area of this research and should not be applied to other areas without any adjustments.

The data for the single and Multi-species indicators have been preprocessed on the following aspects. Because of the data analysis with a resolution of 1 km^2 and a time span of one year, observations exceeding these values have been removed from the dataset. After this step, all observations were converted to points to facilitate the data processing. Moreover, species of which less than ten individuals have been observed in the study area throughout the twenty years have been omitted, as these occurrences could be coincidental and thus do not represent the actual biodiversity. Lastly, the observations represent varying numbers of observed individuals. To remove outliers from the dataset, single observations representing a higher number of individuals than one standard deviation above the mean of counts for that same species have been deleted as well. This has been done because some observations included up to multiple hundreds of individuals within a small area and time span. For the godwit, for example, a concentration of 6.6 pairs per 100 hectares, i.e. one square kilometer, was calculated to be average by Slaterus (2016). Therefore, these high counts were not considered realistic.

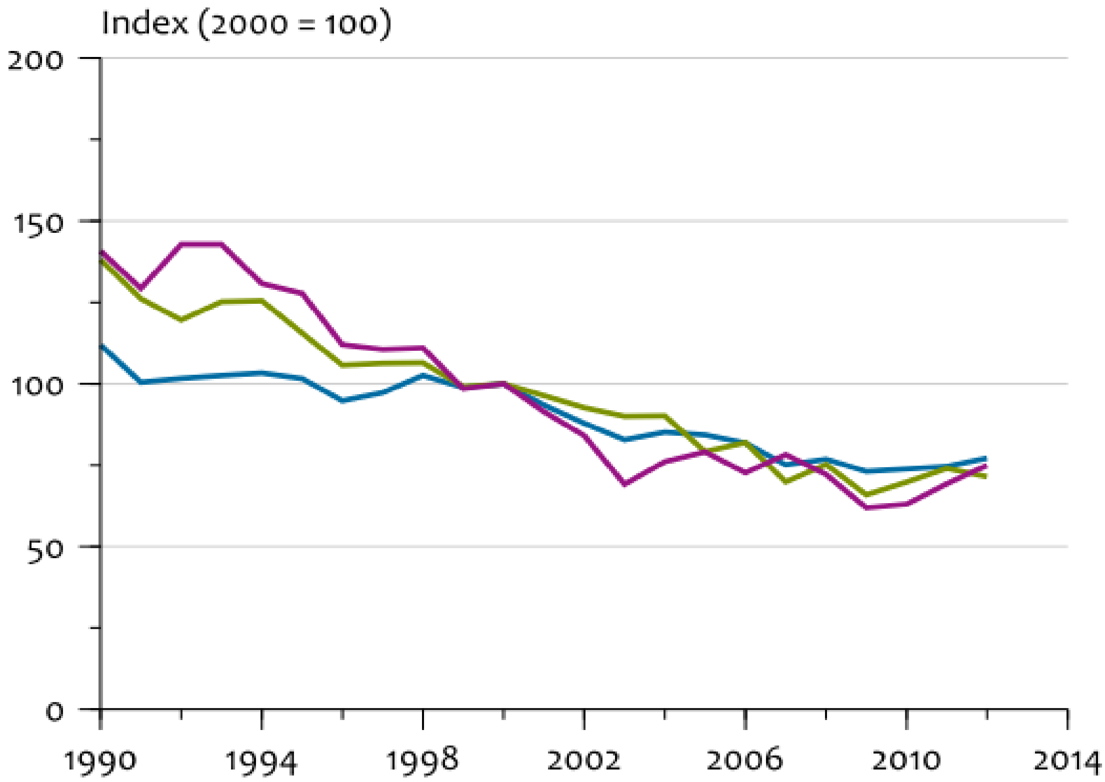
3.2.1.3 Data collection remote sensing

The remote sensing data have been retrieved through the API of Ellipsis Earth Intelligence (Ellipsis Earth 2019). The model on herb richness has currently not been finalized, so only a preliminary application has been produced for the study area. Because of the developmental stage, the method of Ellipsis Earth is not outlined in any formal publication. Therefore, much of the information was retrieved from the website and through a phone conversation with CEO Rosalie van der Maas. The remote sensing data come from the map called *LNV maai en oogst kaart*, a mowing and yielding map commissioned by the Dutch Ministry of Agriculture, Nature and Food Quality. This map will be one of the three models which will eventually be incorporated in the herb richness map (2019 May 27 phone conversation with R van der Maas). Measurements of this model have been collected only during the spring of 2019, namely from April 4 until June 9 (Ellipsis Earth 2019).

3.2.2 SINGLE SPECIES

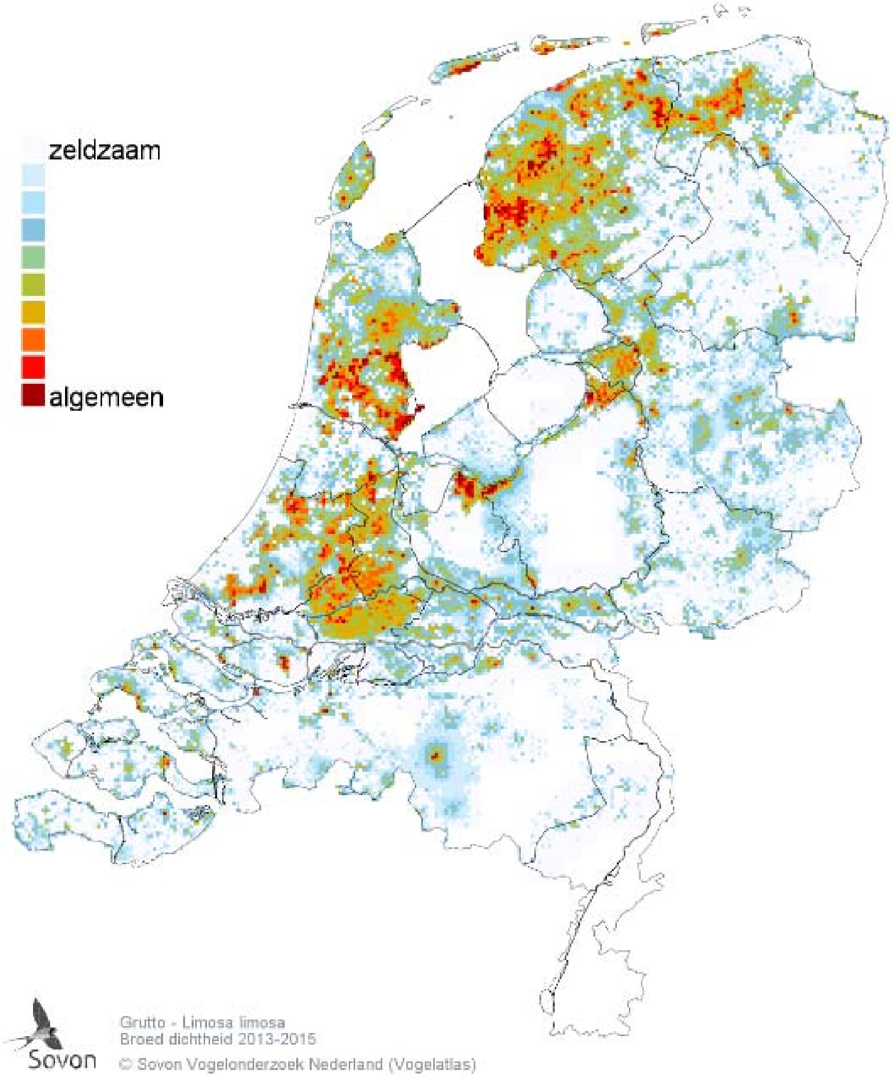
As indicator species for biodiversity in the western peat meadow area in the Netherlands, the population concentration of the godwit has been selected based on reviewing existing literature. In addition, as shown in Figure 7 below, the population trends of the godwit correspond well to the overall population of meadow birds.

Figure 7: Population trends of all meadow birds (blue), godwit (green), and skylark (purple) (Sanders and Westerink 2015)



Sovon, the Dutch Centre for Field Ornithology, collects data from many different bird counting projects to generate national overviews. Based on these data, a Bird Atlas of the Netherlands is published (Vogelatlas 2018), as well as different types of statistics and distributions available on the Sovon website. One of the maps that are created is the distribution of the population in the breeding season. An example of this map produced for the godwit in the years 2013 through 2015 is shown in Figure 8 on the next page. Although these types of maps have already been produced, the data behind it are not publicly available. Since the godwit is a widely accepted biodiversity indicator in agricultural areas, a similar population distribution map has been produced for the godwit in the *Ronde Hoep* as the Single species indicator of biodiversity.

Figure 8: Concentration of breeding godwit in 2013-2015 from high (red) to low (blue/white) (Vogelatlas 2018)



Observations of the godwit were included in the bird data set of the NDFF. These observations were delivered as polygon shapefiles of different shapes and sizes so that the uncertainty of the location was considered. After being transformed into point files, the observations have been divided based on the year of observation and subsequently converted to a raster with cells of one by one kilometer for which the value represents the sum of individuals. Next, the layers have been normalized based on the highest value found for that species throughout all years. As such, when no individuals were observed the cell received a value of 0 and the highest found population received a value of 1. All these steps have been performed in the ArcGIS software with the Model Builder tool (ArcGIS 2019). The model used for the Single species indicator can be found in Appendix 2.

In addition, the Living Planet Index has been calculated of this single species for a graphical representation of the time trends. This index is calculated with the chain method, which consists of two formulas, namely

$$d_t = \log(N_t/N_{t-1}) \text{ and } I_t = I_{t-1}10^{d_t} ,$$

where *N* stands for the total number of observed individuals and *I* for the plotted index value. These calculations are based on the paper on the Living Planet Index by Loh et al. (2005). In this study, however, the index is only calculated for standard years (every 5 years). Because of the short time span in available data for the current research, the index is presented with a three-year running average instead. This has been done by Latham et al. (2008) for calculating the LPI as well.

3.2.3 Multi-species index

Of the included species types – birds, mammals, butterflies, and dragonflies – all observations within the years 1999 – 2018 have been obtained in the same format as the data for the Single species indicator. The observations of the individual species have been transformed into concentration maps per year the same way as has been done for the godwit as Single species indicator. Subsequently, the concentrations have been linearly normalized to the same scale between 0 and 1, where 0 means there were no observations of the species and 1 stands for the highest concentration found in the study area. Of these individual normalizations, the average is taken to form the collective measure of biodiversity.

The spatial distribution maps have been produced with several consecutive models in ArcGIS (ArcGIS 2019). Multiple iterators within one model are not allowed, and therefore a sub-model was produced to separate the data both per species and per year. Moreover, a third model type was made to calculate the averages of all species per year, which was repeated for each year individually. These models can be found in Appendix 3.

The graphical visualization of the index has been calculated with the same formulas as for the single species index. Before the index value is calculated, the average is taken of d_t with the following formula (Loh et al. 2005):

$$\bar{d}_t = \frac{1}{n_t} \sum_{i=1}^{n_t} d_{it}$$

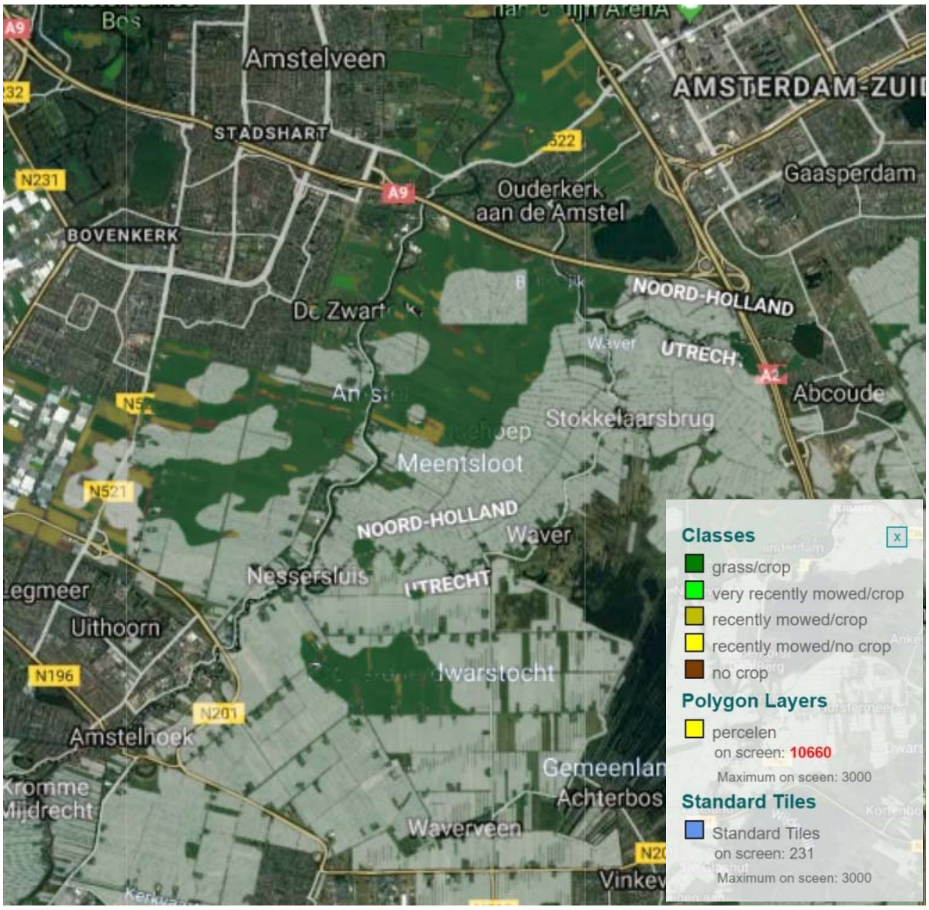
Again, a running average of three has been calculated of d_t. This process has been repeated for all observations remaining after preprocessing the dataset with the use of a script in the programming language R (R 2019).

3.2.4 Remote sensing

Several studies have explored the use of remote sensing for the classification of grassland use (Franke et al. 2012; Dusseux et al. 2014; Asam et al. 2015; Sibanda et al. 2017; Bekkema and Eleveld 2018). Most satellite images are freely available for viewing and downloading through the internet. However, not many tools exist in which processed and classified images of grassland use can be found. Ellipsis Earth is currently collaborating with the Biodiversity Monitor and with farmers in the peat meadow area to improve and validate their herb richness map and has already released several other layers of processed remote sensing data. This close connection to the current study makes their data and methodology very valuable and appropriate for this research (Ellipsis Earth 2019). Through the API, accessible with a Python script (Python 2019), metadata, tabular data, geometries, and visualizations can be viewed. An elaborate tutorial in addition to some practical examples on the website explains potential uses of these data. The map visualizing herb richness is expected to be released towards the end of 2019 so it can fully be utilized in the spring of 2020 (2019 June 18 e-mail correspondence with D van der Maas (CTO)). As explained in the research context, herb richness is considered to be a suitable indicator for biodiversity in agricultural landscapes, so this data set would be an appropriate indicator for biodiversity. Therefore, the potential use of this layer is explained with examples from another layer, namely *LNV maai en oogst kaart*, a map of mowing and harvesting commissioned by the Dutch Ministry of Agriculture, Nature, and Food (Ellipsis Earth 2019). When the herb layer is released, these data will be accessible through very similar steps.

The following figure shows the map viewer for the *Ronde Hoep* on June 18, 2019 (Ellipsis Earth 2019). Because of clouds (white) not all parcels have been classified, but different classifications are still visible.

Figure 9: Web viewer on June 18, 2019, layer LNV maai en oogst kaart (Ellipsis Earth 2019)



The first step of accessing data through the API is requesting metadata. For data to be accurately requested, it is necessary to be aware of what data exist. First, the available layers need to be known. All observations are connected to a time span of usually three days in which the measurements have taken place, referred to as timestamps. Moreover, the available classes, such as 'Grass/crop' in Figure 9 and spectral indices used to produce these classifications can be obtained as metadata. Lastly, an area of which the data are to be retrieved needs to be defined. Ellipsis Earth makes use of predefined polygons, the parcels (*percelen*) of which some can be seen in Figure 10. In addition, their entire study area is divided into standard tiles of a little bit over two square kilometers in surface. Lastly, it is possible to create a custom polygon as a user. Through the metadata, it is also possible to retrieve all IDs of the predefined polygons or standard tiles available for a layer in its entirety or within an area bound by coordinates.

Figure 10: Geometries of parcels in study area from API in QGIS (Ellipsis Earth 2019; QGIS 2019)

The data on the classes and spectral indices are retrievable in tabular form for various combinations of the metadata. For example, all data on classes are provided in terms of square kilometers of surface for each class within the given geometries. For all available timestamps, the surface area of each class can be requested for a custom polygon. Furthermore, the surface areas per class for each standard tile intersecting with the custom polygon or predefined polygon can be acquired. Data of predefined polygons and standard tiles can also be requested for one timestamp for several intended IDs, or for a particular polygon or tile all different timestamps can be requested. An example of this last request is shown in Table 6.

Spectral data are always provided as a mean of the index values of all pixels within a given geometry. When a custom polygon is used, the mean of all spectral indices for all intersecting standard tiles can be obtained, or the individual indices for intersecting standard tiles can be viewed for one timestamp. Moreover, for predefined polygons or standard tiles, all indices at a certain timestamp can be requested for geometries defined by their IDs, of which an example is shown in Table 7. In addition, all timestamps can be acquired for one polygon on standard tile. Lastly, it is possible to obtain the spectral data for all standard tiles intersecting with a certain predefined polygon.

Table 6: Five timestamps for the classes of one standard tile ({'tileX': 8411, 'tileY': 5389, 'zoom': 14}) (Ellipsis Earth 2019)

Time stamp	blanc	grass/ crop	mask	no class	no crop	recently mowed/crop	recently mowed/ no crop	very recently mowed/ crop	area	date_ to	date_ from
0	2.145	0.047	0.008	0	0.002	0.036	0.002	0.003	2.241	18/04/ 2018	15/04/ 2018
1	2.241	0	0	0	0	0	0	0	2.241	21/04/ 2018	18/04/ 2018
2	2.145	0.082	0	0	0.001	0.002	0.007	0.005	2.241	24/04/ 2018	21/04/ 2018
3	2.145	0	0.096	0	0	0	0	0	2.241	27/04/ 2018	24/04/ 2018
4	2.241	0	0	0	0	0	0	0	2.241	30/04/ 2018	27/04/ 2018

Table 7: Spectral indices of five predefined polygons for timestamp 16 (2019 Apr 1 to 2019 Apr 4) (Ellipsis Earth 2019)

id	NDII	NDVI	RE6NDVI	cloud_cover	area
231883	0.129	0.748	0.33	0.18	0.112
232392	0.097	0.754	0.231	0.12	0.105
232663	0.146	0.834	0.399	0.15	0.336
232791	0.094	0.702	0.185	0.11	0.128
233587	0.143	0.839	0.383	0.14	0.104

In addition to tabular data, it is also possible to obtain the geometries of predefined polygons and standard tiles as geoJSON files, which can be loaded into GIS software and thus placed over base map layers (see Figure 10 on the previous page). However, these shapes do not contain any additional data. Moreover, visualizations can also be retrieved as pictures in PNG format. These have four channels: red, green, blue, and transparent. The pictures portray averages per predefined polygon of a spectral layer value between two given timestamps. The bounds of the area for these pictures are given through minimum and maximum X- and Y- coordinates. An example of such an image is shown in Figure 11. The transparency of the layer is caused by cloud disturbances.

Figure 11: Example mage of study area from API: Timestamps: 25-30 (2019 Apr 28 – 2019 May 16), layer: 'Label'; Brown = 'no crop', bright yellow = 'recently mowed/no crop', dark yellow = 'recently mowed' bright green = 'very recently mowed', dark green = 'grass/crop'

Static images of the current mowing and harvesting layer give only a vague indication of potential biodiversity. Monitoring over time can allow for frequencies being established on parcel level, which demonstrate land-use intensity and thus biodiversity. This, however, is expected to be integrated into the herb richness layer. This new layer will distinguish the following three classes: 'very rich in herbs', 'moderately rich in herbs', and 'low in herbs' (2019 June 18 e-mail correspondence with D van der Maas). These three layers portray a better distribution of biodiversity through herb richness within a single timestamp. However, monitoring a longer time span is still valuable for tracking changes and to compensate for cloud disturbances within single images.

Several steps would still be required to combine these data with the other two biodiversity indicators. It must be decided at what stage the data are aggregated to the resolution used for the species indicators. The coordinates of the raster cells could be extracted and used to make predefined polygons of the same size and position. Alternatively, the data can be obtained in the form of the predefined polygons (although the large amount within the study area would require processing the data in at least two separate chunks), which are later aggregated in GIS. Secondly, a Python script is required to combine the geometric (geoJSON) data with the tabular (CSV) data regarding the surfaces per class within the polygons (Python 2019). Lastly, it must be decided how the class data are used to determine the level of biodiversity. Only the surface area of the 'very rich in herbs' class could be used to differentiate the cells or geometries, or the 'moderately rich' layer can be included but with a lower weight. For this, a concrete formula for normalizing the area should be defined. Next, the distribution of the area should be transformed into a raster with the same exact extent and cell size, which subsequently can be overlaid and combined with the other two indicators, for comparison and to produce a collective distribution consisting of the average of all three indicators.

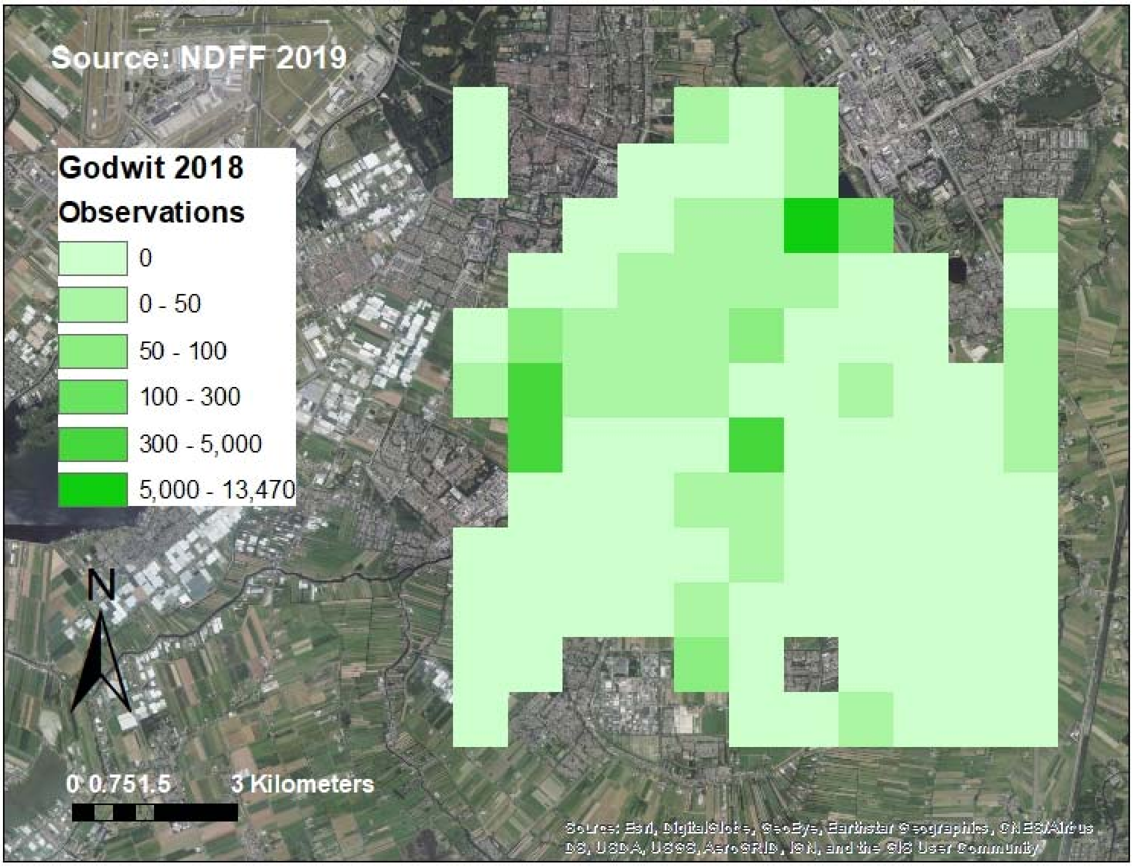
4. Results

4.1 Single species indicator

The spatial distribution of the godwit is composed of 6510 observations in total after the preparatory steps listed in section 3.2.1.2. The average count of individuals per observation is 29 individuals with a standard deviation of 73. The average location uncertainty, i.e. the indicated surface area, is 4.7 hectares with a standard deviation of 17 hectares. Three different maps have been created. Firstly, the distribution in 2018 shows the most recent distribution map of the godwit in the area. Secondly, the average of taken of the years 2016 – 2018 as a measure of 2017 in order to reduce outliers and inconsistencies, similarly as done by Sovon (Figure 8). Lastly, the difference between the latter and the average of the years 1999 – 2001 is measured to observe the trend in local biodiversity over those years.

The following map shows the spatial distribution of the godwit in 2018.

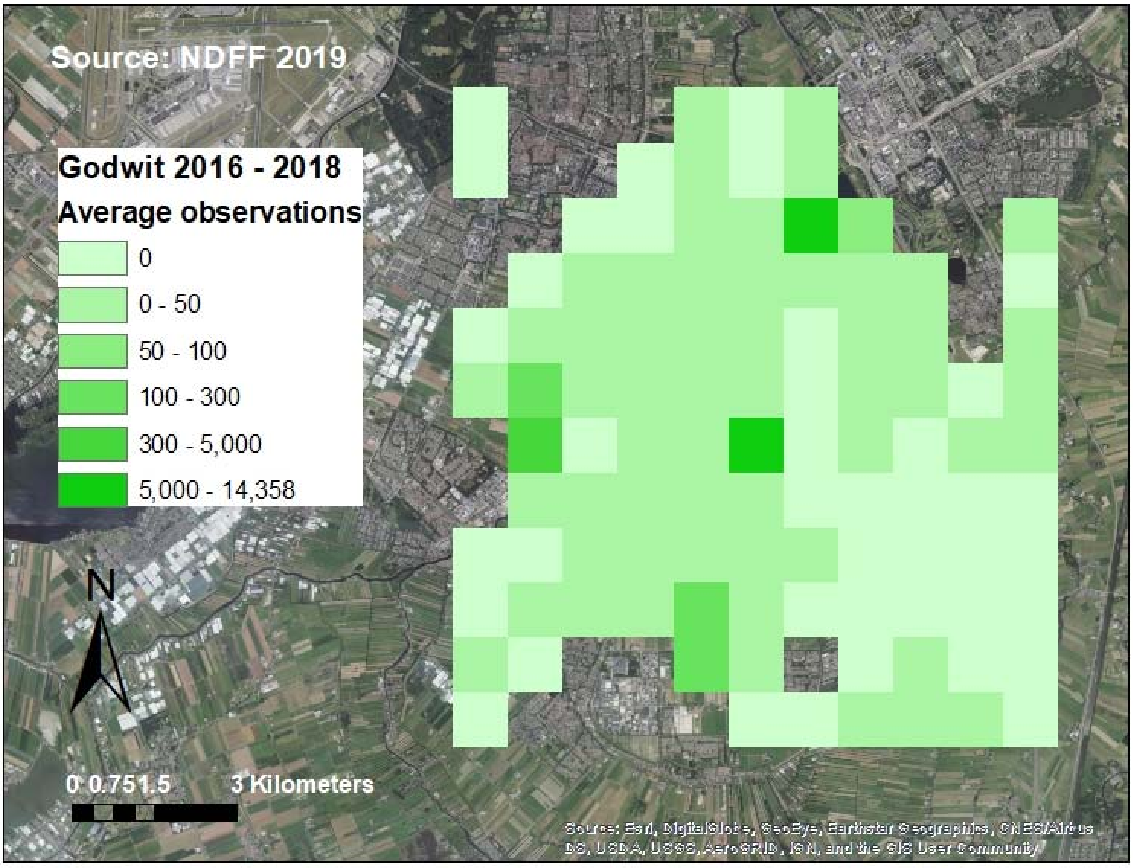
Figure 12: Total observed individuals of the godwit per square kilometer in 2018



There are large differences between raster cells, which can be explained by the various observations with very high counts. Therefore, the classes have been divided irregularly to ensure the visibility of spatial differences. The classification is based on rounded off values of natural breaks automatically generated in ArcGIS (ArcGIS 2019). The highest distribution is concentrated in four raster cells and furthermore the godwit is mostly observed in the northern part of the area. Near the edges of the bottom half, for many cells no individuals of the godwit have been observed in 2018.

The following map shows the average spatial distribution of the godwit in the years 2016-2018.

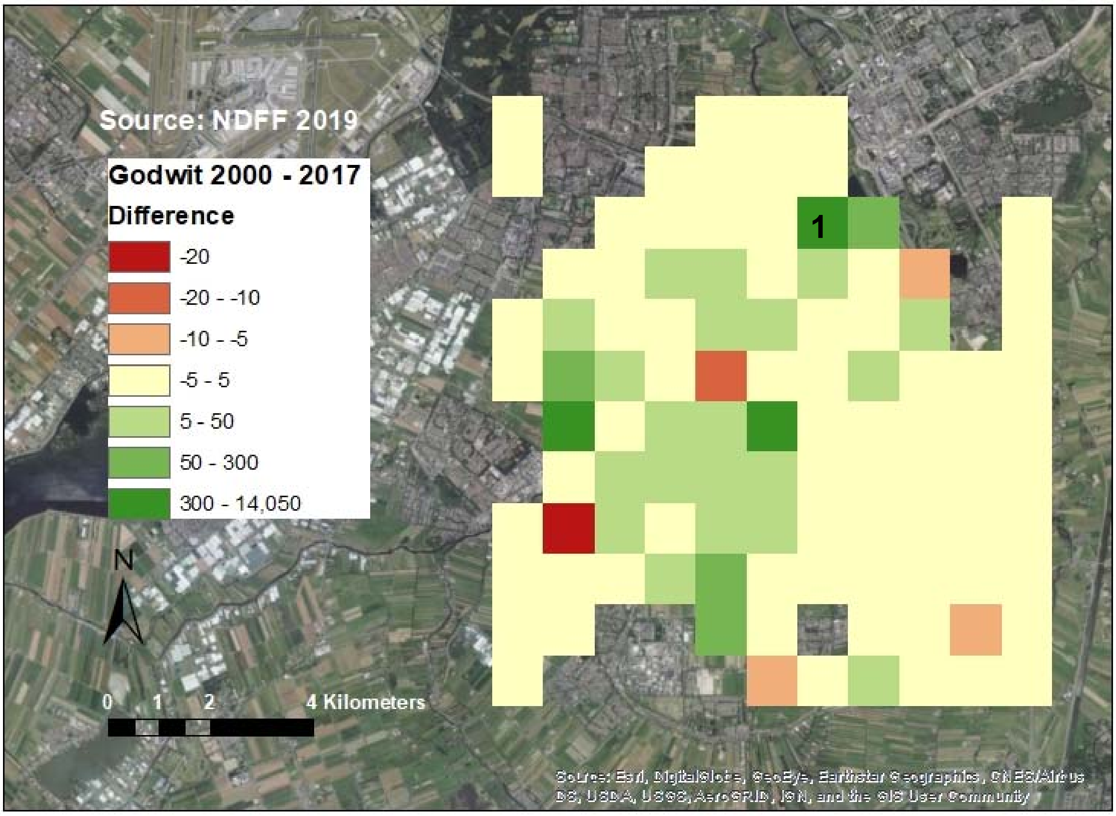
Figure 13: Average of total observed individuals of the godwit per square kilometer in 2016 – 2018



In this map, the extreme values are located in approximately the same cells as in the 2018 map. However, there are less cells without any observations than in the 2018 map. Furthermore, the edges of the area generally show lower counts than the center and the concentration is generally higher towards the northern part of the area.

The following map shows the difference between 2000 (1999 – 2001) and 2017 (2016 – 2018).

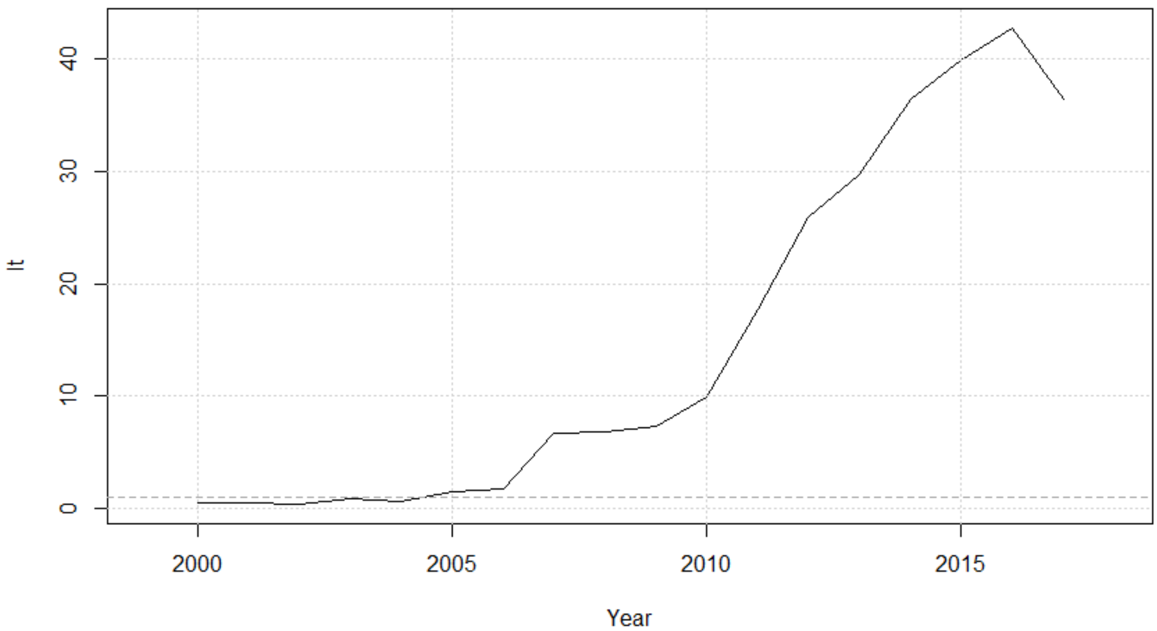
Figure 14: Difference between concentrations 2000 and 2017 as average of three years



The concentration of the godwit has remained the same in a large part of the area. Only five cells have decreased slightly in observed individuals. Still, towards the center of the *Ronde Hoep*, several cells show an increase. Moreover, the largest increases are located in the cells with extreme values in the map of 2016 – 2018. The maximum value of the 1999 – 2001 map is 308 in the same cell (marked with a 1), so logically the extreme values in the more recent map largely influence the difference.

The following graph shows the Living Planet Index of the godwit population in the *Ronde Hoep*, presented as a three-year running average to decrease uncertainties and outliers. The dotted line represents the number of observations in the year 1999, which has the index value I_t *of 1.*

Figure 15: Living Planet Index of godwit observations for the years 1999-2018 (three-year running average) (NDFF 2019)

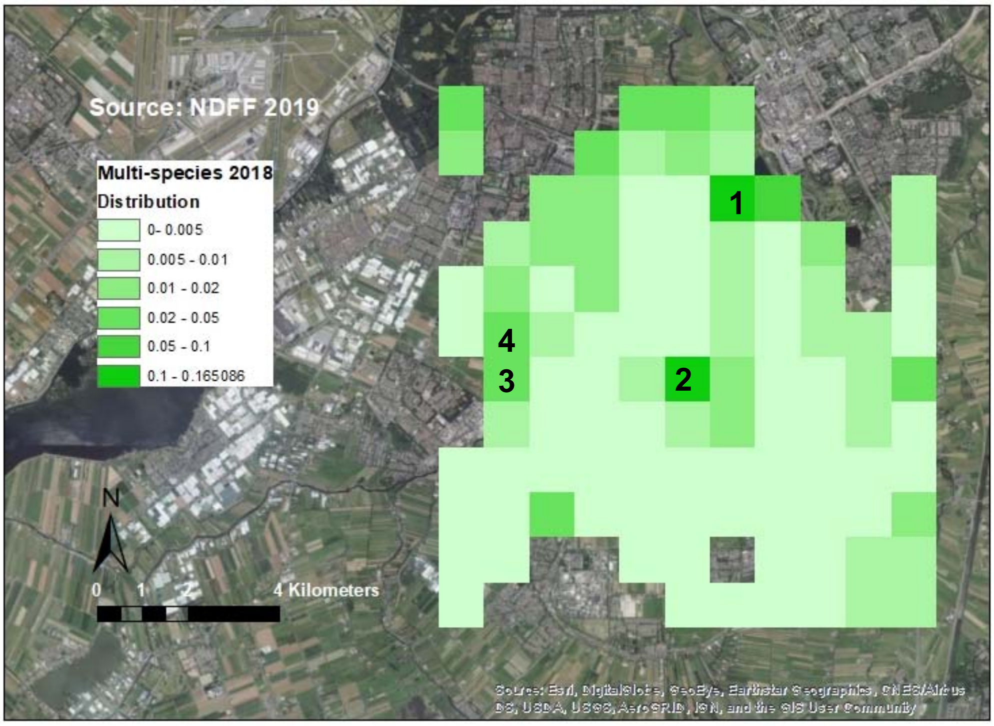


This graph shows that in the first years the population remains somewhat stable, since the growth rate stays close to 1. However, between 2005 and 2010, the growth rate dramatically increases to above 40 in 2017. Only in the last few years, the growth index decreases slightly. This suggests that the population of the godwit has increased by a lot in the last decade. However, other factors considering the observations could have influenced these results as well, for example when the measuring network has increased, such that there are more volunteers available for making observations.

4.2 MULTI-SPECIES INDEX

For the multi-species index, the same three distribution maps and Living Planet Index graph as for the godwit have been produced. These are found below. The data set consisted of 132690 observations of 408 species after the preprocessing steps. The list of species can be found in Appendix 4. The average number of individuals per observation was 8.

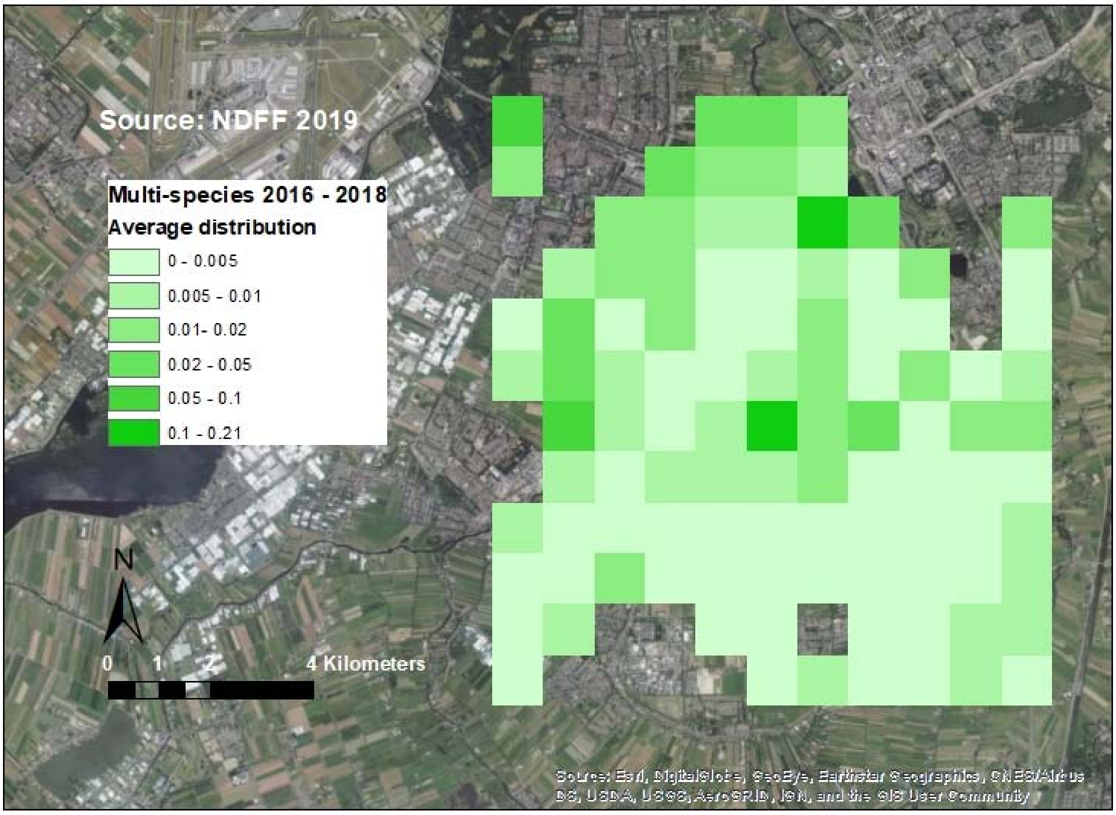
Figure 16: Multi-species distribution in 2018



From this map, it becomes clear that the species are generally concentrated towards the northern edges of the study area. The cells with high values in the distribution of the godwit, marked with numbers 1-4, again have some of the highest concentrations in this map, even though the surrounding cells have much lower values. Moreover, the edge on the right-side of the study area shows higher concentrations of biodiversity as well, whereas no individuals of the godwit had been observed there. The same goes for the two separated grid cells on the upper left of the study area.

The following map shows the average per cell of the normalized distribution values taken for all species.

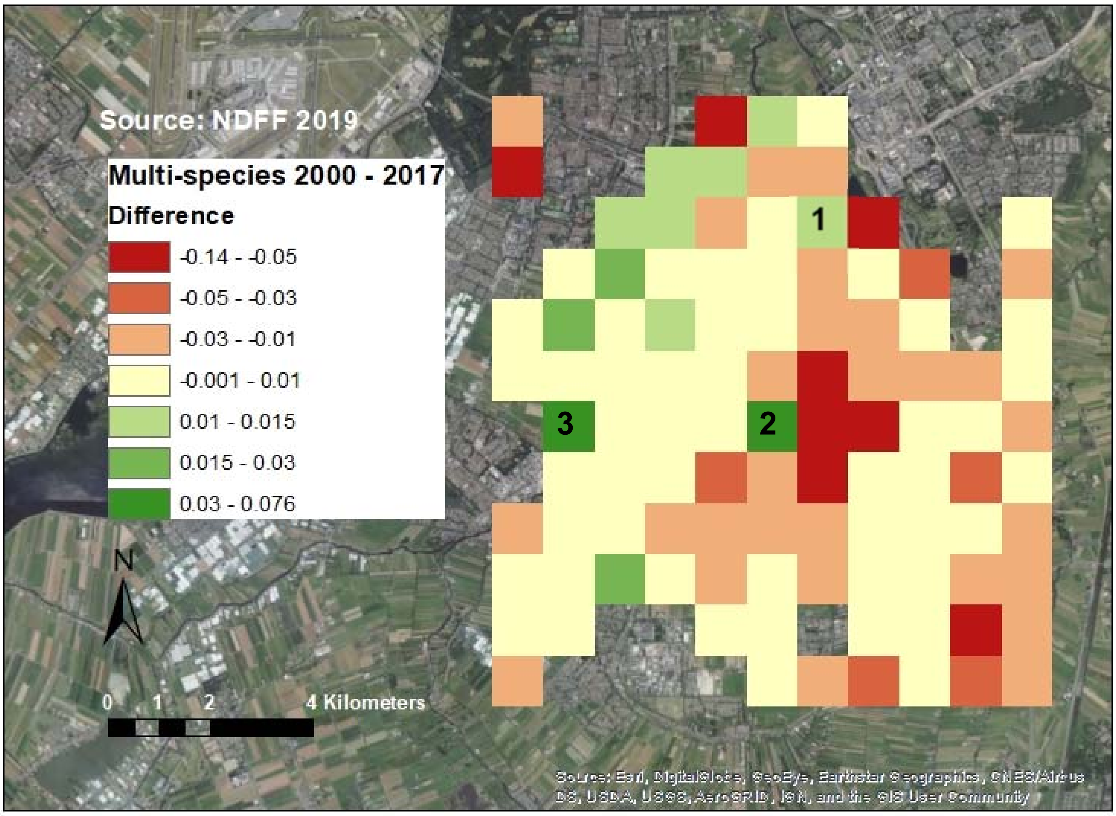
Figure 17: Average distribution of the multi-species indicator for the years 2016 – 2018



The average distribution of the multi-species indicator for the years 2016 – 2018 shows largely similar patterns as the map of 2018. However, in the center of the area, the values are more mixed, whereas they were predominantly low in the previous map.

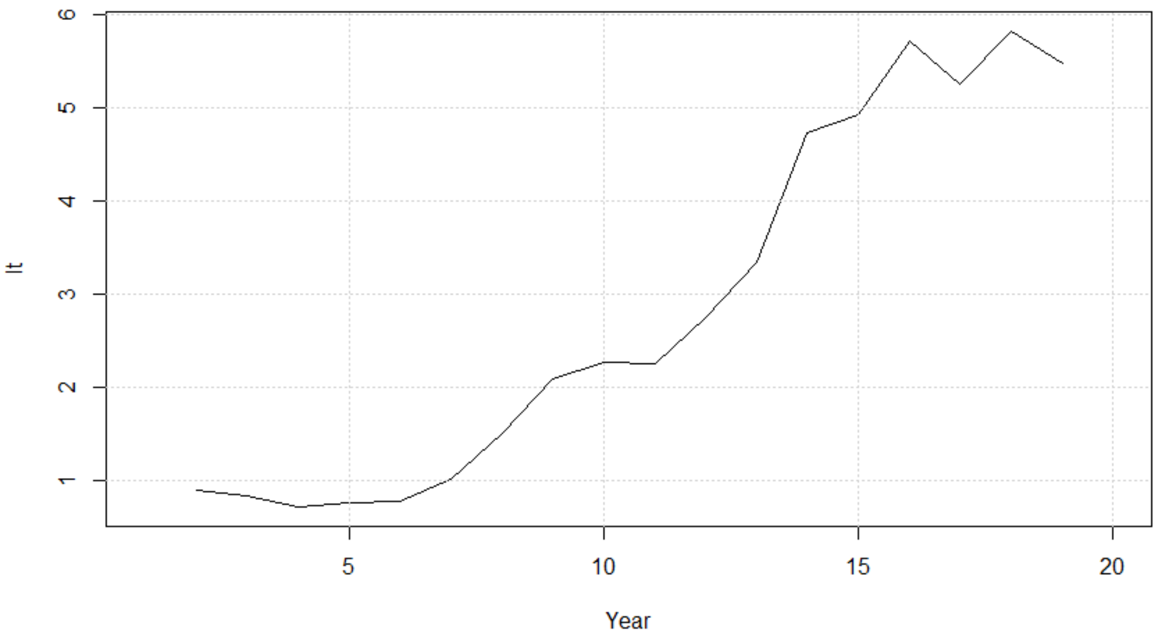
In the following map, the difference between the previous map and the average of the years 1999 – 2001 is showed.

Figure 18: Difference between distribution in of years 2000 and 2017 as average of three years



Except for the north-eastern part of the study area, most cells show a decline or have remained mostly the same. Again, the same cells (1-3) as outlined in several previous maps show very different values than their surroundings. These can indicate popular habitats.

Figure 19: Living Planet Index of multi-species indicator for the years 1999-2018 (three-year running average) (NDFF 2019) (Year 1 = 1999)



In the first five years, biodiversity is declining according to this graph. However, after approximately the year 2005 (Year = 7) a strong increase is visible again. However, the index for the multi-species indicator approaches a growth rate of 6, whereas the index value of the godwit surpassed the value of 40. Moreover, the growth rate stabilizes, but does not seem to decrease in the more recent years such as visible for the godwit. This still indicates that biodiversity is increasing, yet with stable rate.

5. Discussion

5.1 Methodology

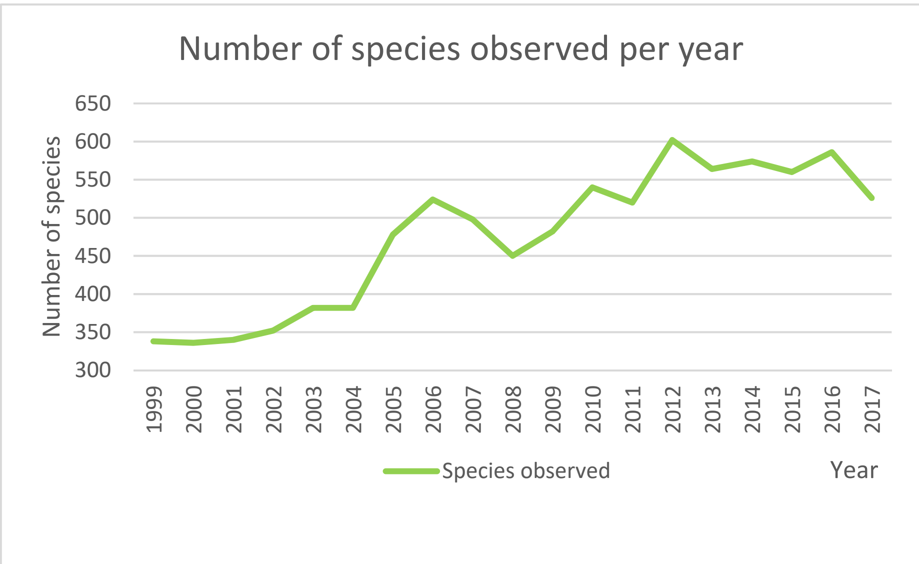
Several points with regards to the methodology of this study are important to consider. Firstly, the multi-criteria analysis is based on ordinal values and hypothetical weights. These encompass only rough estimations of reality, and considering the weighted scores did not differ much, there is considerable uncertainty considering the rank order of the alternatives. Moreover, the criteria listed by Bibby (1999) were not specifically intended for small scale biodiversity indicators, but also for those related to public policy. These differences were considered in choosing the weight distributions, but since the categories did not differ much in their weights, the influence of less relevant criteria could have been overestimated.

Furthermore, many of the considered indicators for biodiversity are spatially applicable but are often only described in numeric of qualitative terms for a predefined area, such as the Living Planet Index, Taxa, and the management index. Therefore, the methodology of applying the species indicators has been based on existing studies and examples, such as Living Planet Report (Wereld Natuur Fonds 2015; WWF 2018) and distribution maps produced by Sovon for the Bird Atlas (Vogelatlas 2018). These were further adjusted to incorporate a detailed spatial resolution and as such fit the objectives of this study.

As seen in the results of this study, the extreme values in the observation data have a large influence on the spatial distribution. Most of the high counts cannot be explained by other factors in which they differ from observations of fewer individuals. This is a disadvantage of the use of both species indicators, especially considering the NDFF database is the most elaborate source of species data available in the Netherlands.

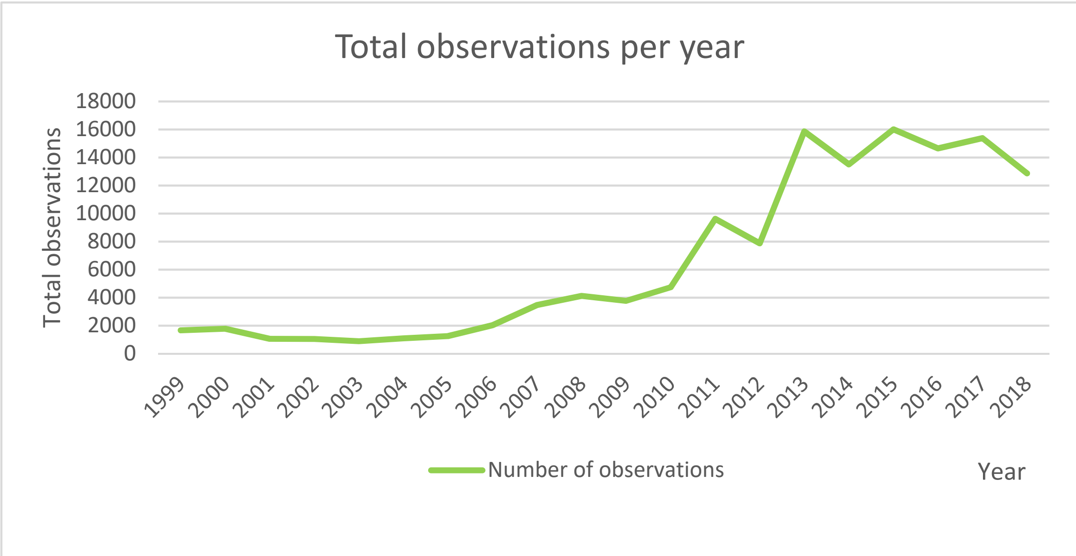
Lastly, one processing step has not successfully been incorporated into the models of the single and multi-species indicators. Since the dataset consist of individual observations, there are no null values for year in which no individuals have been observed of a species. Therefore, while these scenarios should be included into the yearly averages as zeroes for all raster cells, these species are currently not included for these years. To estimate how severely this could have influenced the results, the following graph has been produced, showing the number of species included per year in the normalized multi-species raster layers.

Figure 20: Graph of number of species incorporated



There is a clearly increasing trend visible regarding the amount of species included per year. This most likely has caused an underestimation of both the graphical and spatial Living Planet Index for the multi-species indicator in the earlier years, as zero values would have inevitable decreased the average values. However, the total amount of observations has increased even more than the number of observed species, as can be seen in Figure 21 below. This trend implies that this increase can also (partially) be explained by increased measuring networks and higher amounts of volunteers making these observations. Therefore, it is possible that this caused an underestimation of total individuals in the first half of the study period, which might compensate for the overestimation due to the missing empty raster files.

Figure 21: Graph of total amount of observations per year in the dataset



Since the Living Planet Index is based on the same dataset, it experiences the same data deficits. In this case, however, the impact is likely to be bigger, because of the logarithmic nature of the LPI formulas, as opposed to an average of values that only vary between 0 and 1.

5.2 Results

The results of this study show that the distribution of the godwit in the *Ronde Hoep* is concentrated around a vertical line through the center, with several locations with a very high concentration in the northern half. The higher concentrations in the center can be explained by the larger distance to urban areas and thus lower levels of human disturbances. Moreover, two of the high concentration areas are located next to a lake, namely the *Ouderkerkerplas* and the *Baambrugse Zuwe*. Although these areas are not classified as natural parks, it can be expected that biodiversity is higher around lakes as there is no possibility for agriculture. For example, some observations with high counts are classified as “Sleeping area measurements”, indicating that many birds collectively sleep in those areas. However, another reason for this concentration could be the fact that these lakes function as recreational areas as well. This can result in more volunteers going to these areas and thus more observations being documented. Moreover, the high concentration areas, marked with numbers, are located near highways and/or urban areas, which could indicate that accessibility for people could play a role as well. Moreover, because the observations were converted into centroid points, it could appear as though they were concentrated in accessible areas while in reality this is not the case.

The multi-species indicator shows somewhat different patterns than the godwit. Again, the concentrations in the northern area are higher, but they are more randomly spread out. Usually, taking an average of a large dataset causes results to become smoother, so this is an interesting and unexpected observation. However, the highest found concentrations are found the same few raster cells as for the godwit. This could indicate very nature friendly areas, sleeping areas, or popular areas amongst volunteers. Moreover, it is somewhat surprising that the concentrations are higher towards the north, as this borders on urban areas of the city of Amsterdam. Again, this can be related to the fact that volunteers might come from urban areas before entering the *Ronde Hoep*. Thus, these areas could be overrepresented.

Furthermore, the Living Planet Indices for the godwit and all collective species indicate a strong increase of biodiversity in the *Ronde Hoep*. Although the Living Planet Index of agricultural areas in the Netherlands has been steadily declining in the past few decades (Wereld Natuur Fonds 2015), a yearly report on meadow birds in the *Ronde Hoep* shows signs of a recovery of biodiversity in this area (Van Paassen 2016). However, the strong increase of the populations can also be caused by improved measuring networks, as mentioned earlier. For the godwit, this increase is also visible in the difference map between 2000 and 2017. However, for the multi-species index this is not the case, as the difference maps of the multi-species indicator show a decline in biodiversity. Although no concrete reason for this has been found, this is likely related to the aforementioned limitation regarding years in which no individuals of a species have been observed. Moreover, because of the normalization of the populations for the multi-species map, small populations have a relatively bigger influence on biodiversity. This could potentially have caused these seemingly contradictory results.

Moreover, the distributions of the two indicators which have been applied to the *Ronde Hoep* show quite different results. This implies that the use of a single species indicator is indeed not very scientifically credible, as several existing studies have also argued (e.g. Noss 1990; Lindenmayer 1999; Bonn et al. 2002). Lastly, based on the prospective of the Ellipsis Earth herb richness map, it is likely that this is a more appropriate indicator for biodiversity in this particular study area. These observations are more detailed, freely accessible, and updated more frequently.

6. Conclusion

In the introduction section, the following research questions has been formulated: ‘What is the spatial distribution of biodiversity in the western Dutch peat meadow area?’ This question was divided into five sub-questions, which are answered in the following section.

Biodiversity indicators are required because comprehensively measuring biodiversity is not efficient for large scale applications. Three indicators of biodiversity are based on species assemblages. Single-species indicators use one species to represent overall biodiversity, taxa are groups of closely related species, and thirdly, multi-species indices use large sets of species to estimate biodiversity. Moreover, in agricultural areas biodiversity can be derived from soil characteristics or landscape structures such as nature friendly areas between parcels, ecological connections between natural areas, and percentage land use cover. Moreover, indices about farm management can be used to predict on-site biodiversity. Lastly, spectral data from remote sensing images can often accurately predict grassland-use intensity which is related to biodiversity. In peat meadow areas, meadow birds are frequently used as indicators of biodiversity. These species are specific to this landscape type and therefore large parts of the European populations of these birds reside in these areas. Therefore, their conservation is considered important and their diversity is frequently monitored. Especially the godwit is widely accepted as an indicator of biodiversity in peatlands, because of its sensitivity to disturbance and criticalness of its habitat, among other things. Moreover, since grassland is the most appropriate type of land use for peatlands, grassland diversity is important for the overall biodiversity in these areas. Therefore, the herb richness in grasslands functions as an important indicator of biodiversity on peat soils as well.

With the use of a multi-criteria analysis, the single species indicator, multi-species index, and remote sensing images have been selected as the most suitable indicators for biodiversity for the western peat meadow area. Because of the prospect of the open access and preprocessed data from Ellipsis Earth, another data layer has been used to outline the steps required to obtain these data once published. With the help of a tutorial and some knowledge of the Python programming language, these data are easily retrieved through an API. The data from Ellipsis Earth are very spatially detailed and consists of both classifications and spectral index values, so the possible applications and uses of this data source are very promising.

The single species indicator and multi-species index have been used to spatially measure biodiversity for the *Ronde Hoep*. The godwit was used as the single species index and for the multi-species index, observations were compiled of birds, butterflies, dragonflies, and mammals. The two indicators do not show the exact same distributional patterns. However, throughout all years and across both indicators, the same three or four cells of a square kilometer appear to harbor a high level of biodiversity. Moreover, biodiversity generally is located more towards the north and center of the study area. The observations of the godwit are also spread out more evenly. However, it is possible that single observations consisting of a high number of individuals have distorted the results. Lastly, the Living Planet Index has been calculated for both indicators. Both show a strong increase in growth rates, which imply that biodiversity has increased in the area. For the godwit, this is supported by the map of the difference in populations. However, the multi-species indicator shows some contradictory results regarding the overall time trend between 2000 and 2017.

7. Relevance and further research

A multi-criteria analysis was performed to structure the process of selecting suitable biodiversity indicators. Using an MCA makes a decision-making process transparent and repeatable. Thus, the MCA performed for this study can serve as an example for the peat meadow area but also other agricultural areas for how to select appropriate indicators for biodiversity. Furthermore, the proposed weight distribution for studies focusing on the application of the results of the biodiversity distribution, referred to as AI in the methods section, could be used for such studies. Examples of such applications are communicating about the biodiversity levels to stakeholders of an area, implementing national or local policies on increasing biodiversity, and correlating biodiversity to management strategies.

Moreover, the decline of biodiversity is acknowledged as a major concern, as is represented in the Sustainable Development Goals (cite), the Habitats and Birds Directives of the European Commission (Habitats Directive 1992; Birds Directive 2009), and the National Ecological Network of the Netherlands (Dijksma 2014). In each of these policies, monitoring biodiversity plays an important role as it facilitates the determination of targets, developments, and policy assessments. The methodology of this study serves as an example of how biodiversity can be monitored in the peat meadow area without requiring field measurements, which can help improve existing schemes. Thus, the methodology of this study can assist in producing cost-effective research, which can assist in meeting the broader targets of these policies, such as mitigating climate change.

The portrayal of the spatial distribution of biodiversity in the *Ronde Hoep* is valuable for landowners and other parties and organizations affiliated with the area. Furthermore, the distribution maps resulting from this study can add to the results of existing monitoring schemes, such as those for butterflies, land mammals, water birds, and breeding birds (CBS 2019), for the *Ronde Hoep*. Lastly, these findings and the methodology can also be valuable for stakeholders in other peat areas, such as the Frisian peat meadow area in the Netherlands, and other agricultural areas to get a better understanding of agricultural biodiversity.

Several possibilities for further research are suggested based on the findings of this study. For example, more research could be done on appropriate weights of the ten criteria for biodiversity indicators listed by Bibby (1999) for the *Ronde Hoep* or any other area in which a similar MCA is performed. Moreover, it would be valuable to explore ways in which numeric scores could be given for the indicators as opposed to the ordinal scale used for the MCA in this research. Secondly, the effect of statistical methods such as bootstrapping on the results can be explored, which could eventually lead to the reduction of data distortions.

Moreover, a follow-up study could correlate current management strategies to the spatial and temporal differences in biodiversity in order to determine best management practices. These results could be further implemented in order to increase biodiversity, first locally and later regionally or even nationally.

Lastly, more methods should be developed on spatial applications of indicators for biodiversity. Many of the methods considered in this study, such as the Living Planet Index, are mostly calculated for a predefined area, instead of measuring distributional patterns. Especially remote sensing has lately been explored for this use and shows a lot of potential for measuring the spatial distribution of biodiversity in agricultural areas, as explained in the research and methodology sections. More research on its application would therefore be very valuable in order to improve the accuracy of the classifications based on spectral indices.

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APPENDIX 1: EFFECTS TABLES

Table 8: Weighted sperformance matrix IM

	Quantitative	Simplifying information	User driven	Policy relevant	Scientifically credible	Responsive to changes	Easily understood	Realistic to collect	Susceptible to analysis	Spatially specific	Total:
Assigned weights	**0.12**	**0.08**	**0.08**	**0.08**	**0.12**	**0.12**	**0.08**	**0.12**	**0.08**	**0.12**	**1.0**
Single species	0.36	0.16	0.24	0.24	0.12	0.36	0.24	0.36	0.08	0.36	**2.52**
Taxon	0.36	0.16	0.24	0.16	0.36	0.36	0.16	0.24	0.08	0.36	**2.48**
Multi-species	0.36	0.08	0.08	0.24	0.36	0.36	0.16	0.24	0.24	0.36	**2.48**
Landscape organization	0.12	0.24	0.16	0.24	0.24	0.12	0.08	0.24	0.08	0.24	**1.76**
Soil components	0.36	0.24	0.08	0.24	0.24	0.24	0.24	0.12	0.24	0.36	**2.36**
Management index	0.12	0.24	0.24	0.16	0.24	0.24	0.24	0.12	0.24	0.24	**2.08**
Remote sensing	0.36	0.24	0.24	0.24	0.36	0.24	0.16	0.24	0.16	0.24	**2.48**

Table 9: Equal-weights table

	Quantitative	Simplifying information	User driven	Policy relevant	Scientifically credible	Responsive to changes	Easily understood	Realistic to collect	Susceptible to analysis	Spatially specific	Total:
Assigned weights	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**0.1**	**1.0**
Single species	0.3	0.2	0.3	0.3	0.1	0.3	0.3	0.3	0.1	0.3	2.5
Taxon	0.3	0.2	0.3	0.2	0.3	0.3	0.2	0.2	0.1	0.3	2.4
Multi-species	0.3	0.1	0.1	0.3	0.3	0.3	0.2	0.2	0.3	0.3	2.4
Landscape organization	0.1	0.3	0.2	0.3	0.2	0.1	0.1	0.2	0.1	0.2	1.8
Soil components	0.3	0.3	0.1	0.3	0.2	0.2	0.3	0.1	0.3	0.3	2.4
Management index	0.1	0.3	0.3	0.2	0.2	0.2	0.3	0.1	0.3	0.2	2.2
Remote sensing	0.3	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	2.5

Table 10: Weighted performance matrix IA

	Quantitative	Simplifying information	User driven	Policy relevant	Scientifically credible	Responsive to changes	Easily understood	Realistic to collect	Susceptible to analysis	Spatially specific	Total:
Assigned weights	**0.08**	**0.12**	**0.12**	**0.12**	**0.08**	**0.08**	**0.12**	**0.08**	**0.12**	**0.08**	**1.0**
Single species	0.24	0.24	0.36	0.36	0.08	0.24	0.36	0.24	0.12	0.24	**2.48**
Taxon	0.24	0.24	0.36	0.24	0.24	0.24	0.24	0.16	0.12	0.24	**2.32**
Multi-species	0.24	0.12	0.12	0.36	0.24	0.24	0.24	0.16	0.36	0.24	**2.32**
Landscape organization	0.08	0.36	0.24	0.36	0.16	0.08	0.12	0.16	0.12	0.16	**1.84**
Soil components	0.24	0.36	0.12	0.36	0.16	0.16	0.36	0.08	0.36	0.24	**2.44**
Management index	0.08	0.36	0.36	0.24	0.16	0.16	0.36	0.08	0.36	0.16	**2.32**
Remote sensing	0.24	0.36	0.36	0.36	0.24	0.16	0.24	0.16	0.24	0.16	**2.52**

Table 11: Effects table with 3 : 1 ratio of weights between IM and IA, respectively

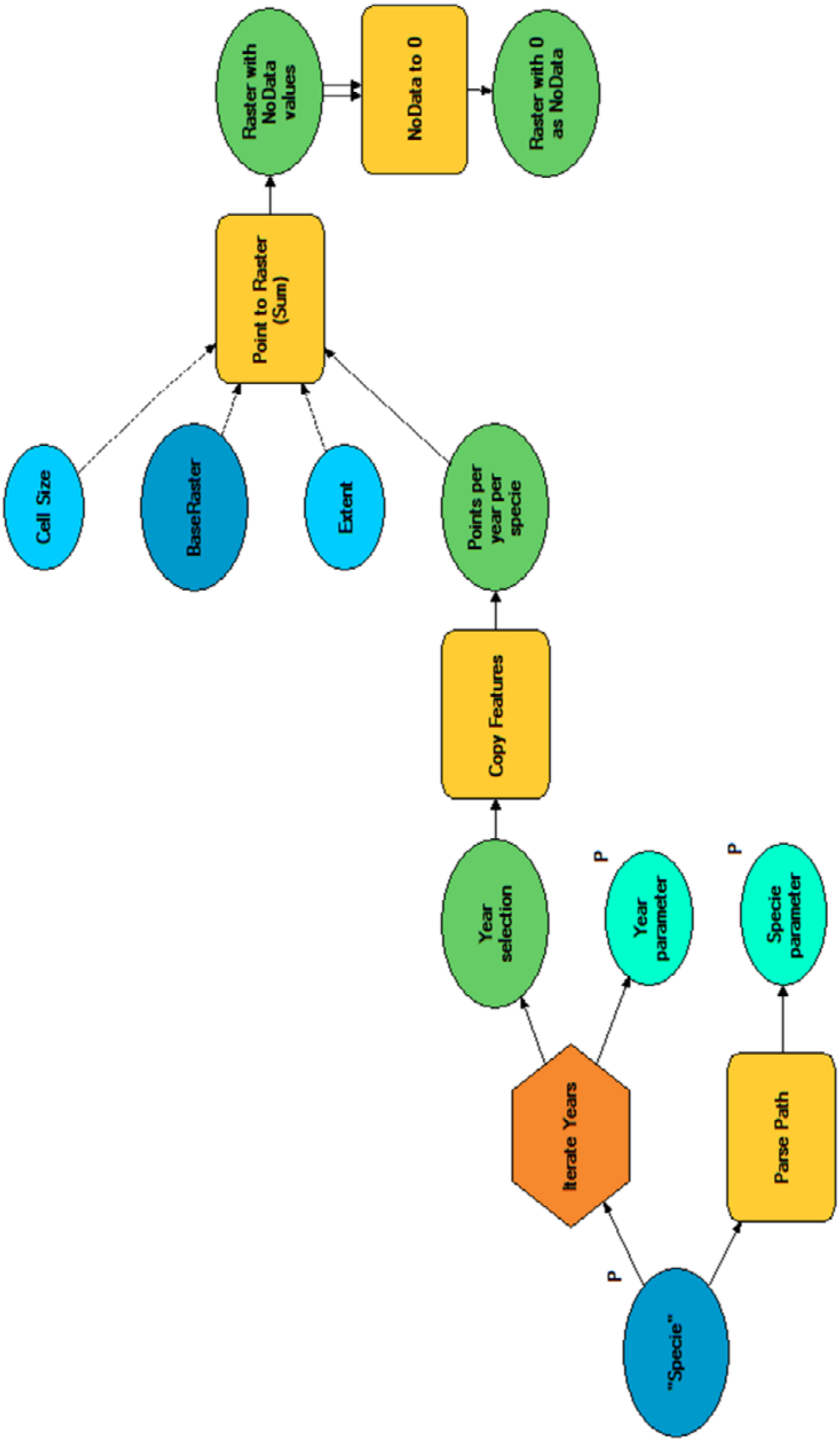
	Quantitative	Simplifying information	User driven	Policy relevant	Scientifically credible	Responsive to changes	Easily understood	Realistic to collect	Susceptible to analysis	Spatially specific	Total:
	0.15	**0.05**	**0.05**	**0.05**	**0.15**	**0.15**	**0.05**	**0.15**	**0.05**	**0.15**	**1.0**
Single species	0.45	0.1	0.15	0.15	0.15	0.45	0.15	0.45	0.05	0.45	**2.55**
Taxon	0.45	0.1	0.15	0.1	0.45	0.45	0.1	0.3	0.05	0.45	**2.6**
Multi species	0.45	0.05	0.05	0.15	0.45	0.45	0.1	0.3	0.15	0.45	**2.6**
Landscape organization	0.15	0.15	0.1	0.15	0.3	0.15	0.05	0.3	0.05	0.3	**1.7**
Soil components	0.45	0.15	0.05	0.15	0.3	0.3	0.15	0.15	0.15	0.45	**2.3**
Managemen t index	0.15	0.15	0.15	0.1	0.3	0.3	0.15	0.15	0.15	0.3	**1.9**
Satellite images	0.45	0.15	0.15	0.15	0.45	0.3	0.1	0.3	0.1	0.3	**2.45**

Table 12: Effects table with 4 : 1 ratio of weights between IM and IA, respectively

	Quantitative	Simplifying information	User driven	Policy relevant	Scientifically credible	Responsive to changes	Easily understood	Realistic to collect	Susceptible to analysis	Spatially specific	Total:
	0.16	**0.04**	**0.04**	**0.04**	**0.16**	**0.16**	**0.04**	**0.16**	**0.04**	**0.16**	**1.0**
Single species	0.48	0.08	0.12	0.12	0.16	0.48	0.12	0.48	0.04	0.48	**2.56**
Taxon	0.48	0.08	0.12	0.08	0.48	0.48	0.08	0.32	0.04	0.48	**2.64**
Multi species	0.48	0.04	0.04	0.12	0.48	0.48	0.08	0.32	0.12	0.48	**2.64**
Landscape organization	0.16	0.12	0.08	0.12	0.32	0.16	0.04	0.32	0.04	0.32	**1.68**
Soil components	0.48	0.12	0.04	0.12	0.32	0.32	0.12	0.16	0.12	0.48	**2.28**
Managemen t index	0.16	0.12	0.12	0.08	0.32	0.32	0.12	0.16	0.12	0.32	**1.84**
Satellite images	0.48	0.12	0.12	0.12	0.48	0.32	0.12	0.32	0.08	0.32	**2.48**

APPENDIX 2: ARCGIS MODEL USED FOR SINGLE-SPECIES INDICATOR

Figure 22: ArcGIS model for producing godwit distribution maps



APPENDIX 4: ARCGIS MODELS USED FOR THE MULTI-SPECIES INDICATOR

Figure 23: Model 1: Separating observations by species

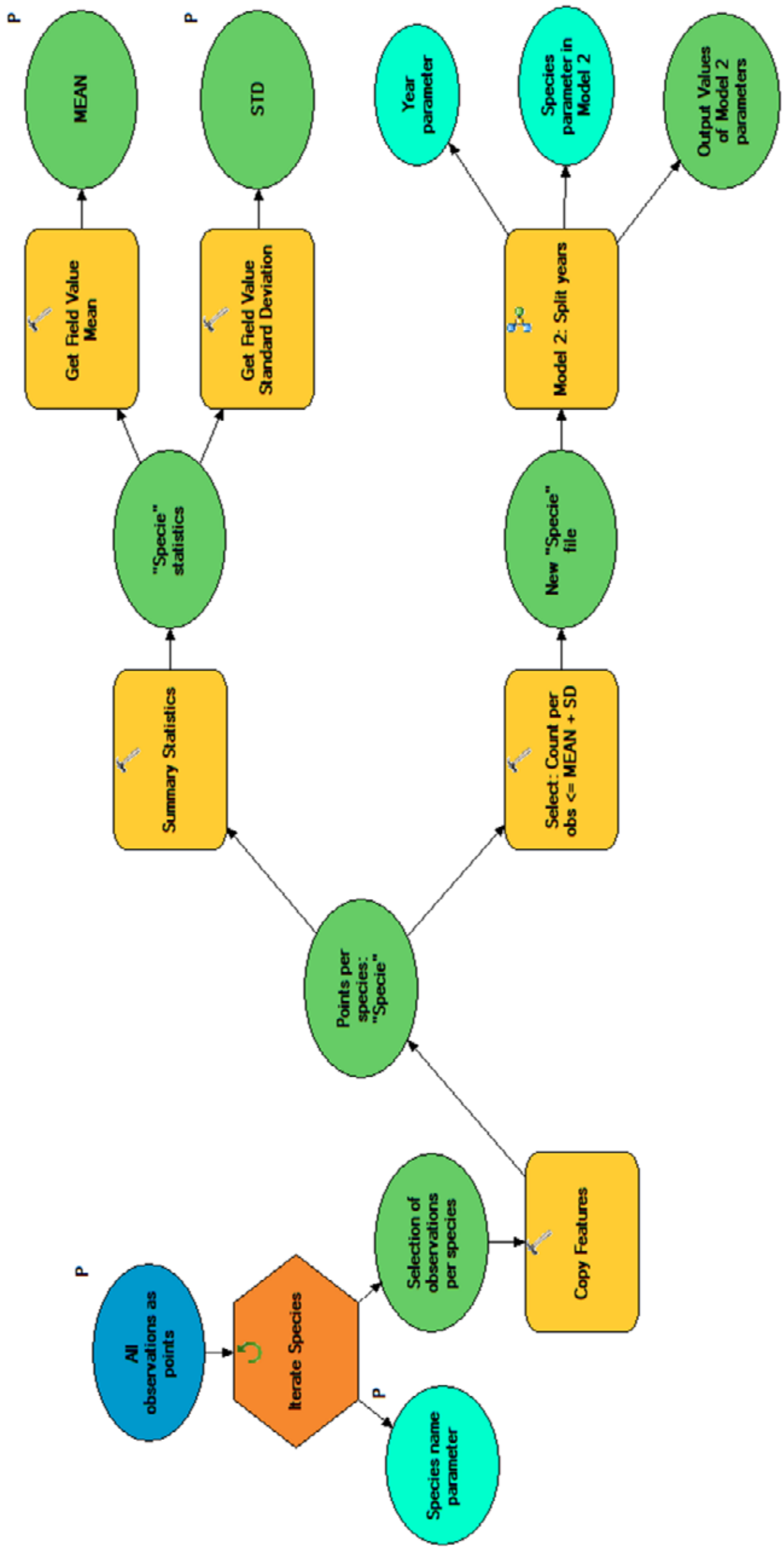


Figure 24: Model 2: Separating species observations by year, forming raster files, and normalizing the raster files

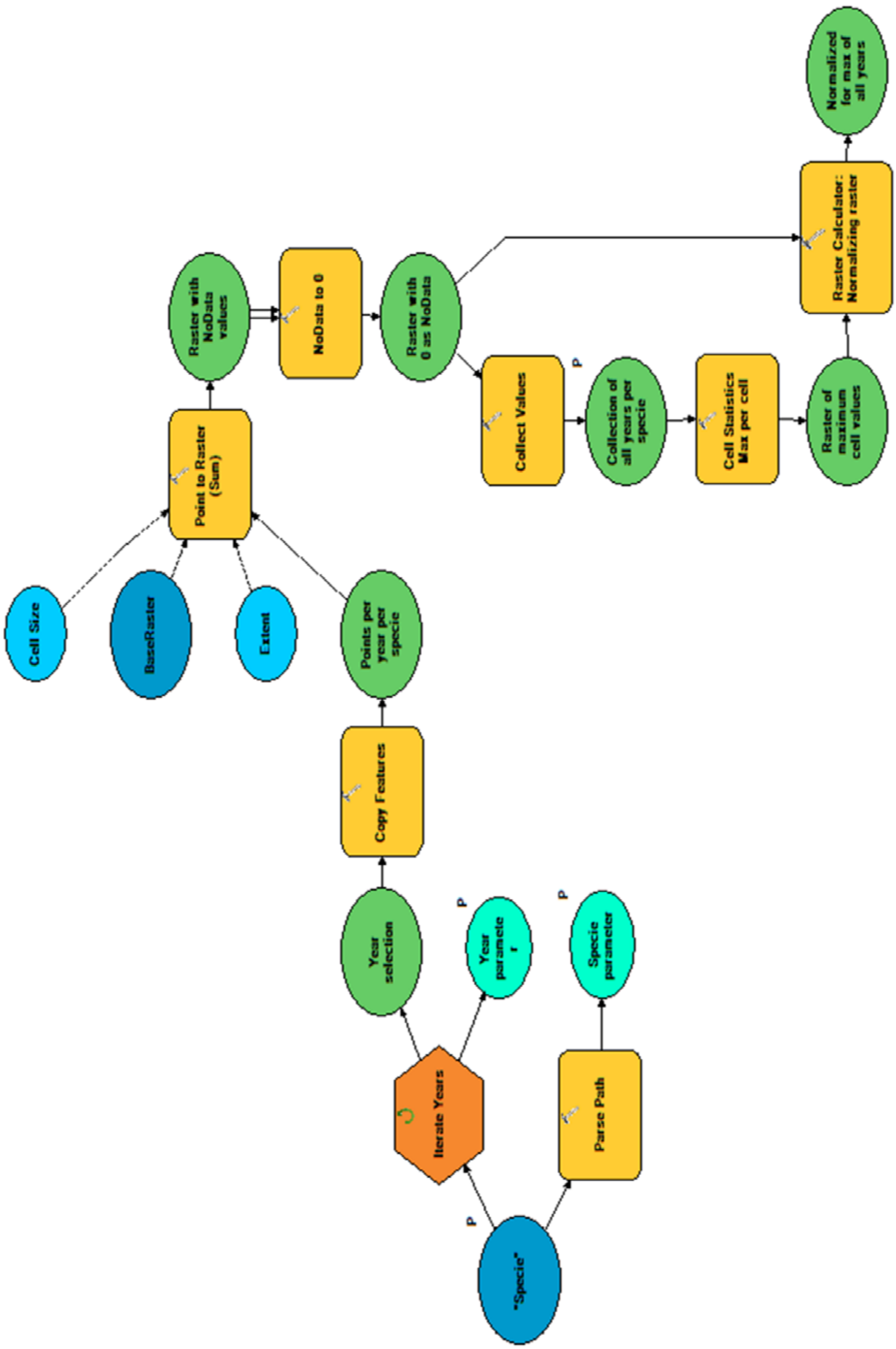
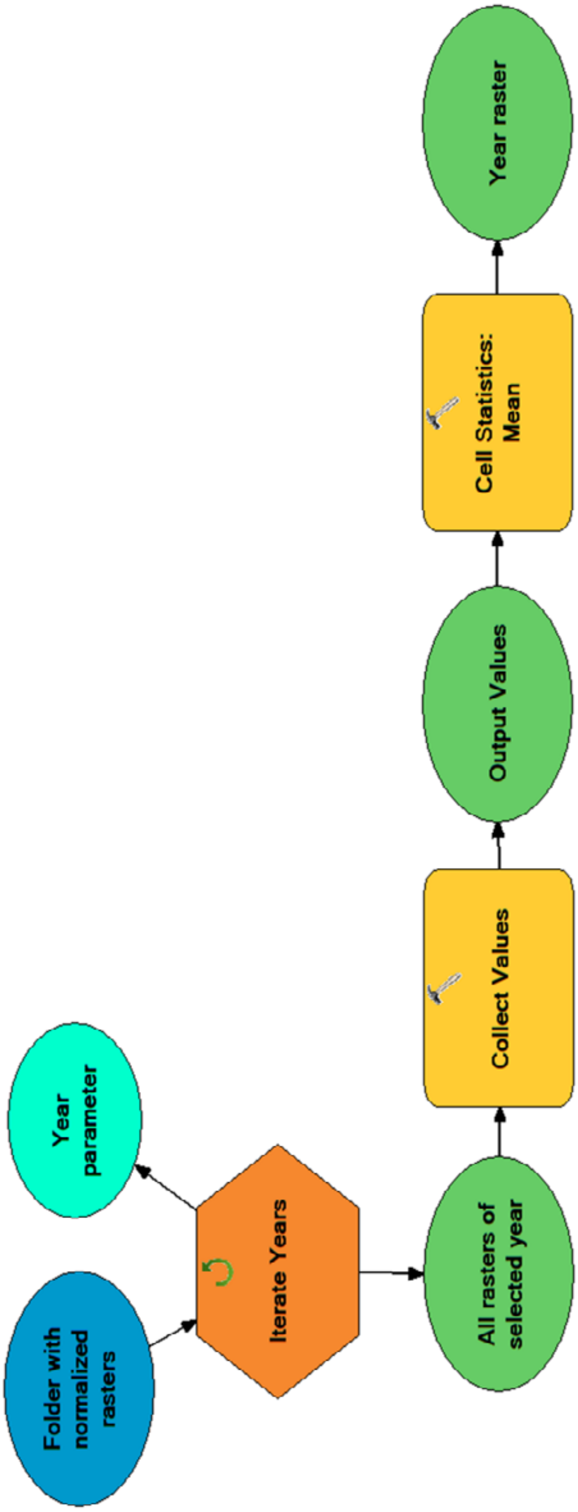


Figure 25: Model 3: Calculating the average of the normalized populations of all species for a particular year



APPENDIX 4: SPECIES LIST MULTI-SPECIES INDICATOR

The multi-species indicator consisted of the following 430 species (Dutch names):

Aalscholver, Aalscholver (Sinensis), Aardmuis, Aeshna Isosceles, Amerikaanse Smient, Anas Platyrhynchos Domesticus, Anser Anser Domesticus, Appeltak, Appelvink, Argusvlinder, Artogeia Napi, Artogeia Rapae, Atalanta, Athene Noctua Vidalii, Azuurwaterjuffer, Baardman, Baardvleermuis, Baardvleermuis / Brandts Vleermuis, Beflijster, Bergeend, Beukenvouwmijnmotje, Blauwborst, Blauwe Glazenmaker, Blauwe Kiekendief, Blauwe Reiger, Blinkend Langsprietje, Bloedrode Heidelibel, Boerenzwaluw, Bokje, Bont Zandoogje, Bontbekplevier, Bonte Bessenvlinder, Bonte Brandnetelroller, Bonte Brandnetelroller, Bonte Strandloper, Boomblauwtje, Boomklever, Boomkruiper, Boomleeuwerik, Boompieper, Boomvalk, Bosbesuil, Bosmuis, Bosrietzanger, Bosruiter, Bosuil, Braamsluiper, Brandgans, Breedbekstrandloper, Brilduiker, Bruin Blauwtje, Bruin Zandoogje, Bruine Glazenmaker, Bruine Kiekendief, Bruine Rat, Bruine Snuituil, Bruine Winterjuffer, Bruinrode Heidelibel, Buffelkopeend, Buizerd, Bunzing, Canadese Gans (Soort Onbekend), Canadese Gans X Grauwe Gans, Carduelis Cannabina, Carduelis Chloris, Carduelis Flammea/Cabaret, Carduelis Spinus, Carolina-Eend, Casarca, Casmerodius Albus, Chileense Smient, Citroenvlinder, Clethrionomys Glareolus, Columba Domestica, Cynthia Cardui, Dagpauwoog, Daurische Klauwier, Distelvlinder, Dodaars, Drieteenstrandloper, Dwerggans, Dwergmeeuw, Dwergmuis, Dwergspitsmuis, Echte Rat (Soort Onbekend), Eekhoorn, Egel, Eikenprocessierups, Ekster, Engelse Kwikstaart, Fazant, Fitis, Fuut, Gaai, Gamma-Uil, Gans (Soort Onbekend), Gehakkelde Aurelia, Gekraagde Roodstaart, Gele Kwikstaart, Gele Kwikstaart (Soort Onbepaald), Geoogde Bandspanner, Geoorde Fuut, Gerande Spanner, Gestreepte Strandloper, Gevlamde Bladroller, Gewone Dwergvleermuis, Gewone Grootoorvleermuis, Gewone Oeverlibel, Gewone Pantserjuffer, Gewone Worteluil, Gewone Zakdrager, Gewone/Tweekleurige Bosspitsmuis, Gewoon Berkenvouwmijnmotje, Gewoon Koolmotje, Gierzwaluw, Glanzend Meidoornvouwmijnmotje, Glassnijder, Glyphodes Perspectalis, Goudhaan, Goudplevier, Goudvenstertje, Goudvink, Grasmus, Graspieper, Grauwe-Elsvouwmijnmotje, Grauwe Franjepoot, Grauwe Gans, Grauwe Vliegenvanger, Grijze Stipspanner, Groene Eikenbladroller, Groene Specht, Groenling, Groenpootruiter, Groente-Uil, Groentje, Groot Avondrood, Groot Koolwitje, Groot Meidoornstippelmotje, Grote Barmsijs, Grote Bonte Specht, Grote Canadese Gans, Grote Gele Kwikstaart, Grote Karekiet, Grote Keizerlibel, Grote Lijster, Grote Mantelmeeuw, Grote Roodoogjuffer, Grote Zaagbek, Grote Zilverreiger, Grutto, Grutto (Limosa), Haarbos, Haas, Hagendoornvlinder, Halsbandparkiet, Havik, Hazelwisselmotje, Heggenmus, Heidelibel (Soort Onbekend), Heilige Ibis, Helmkruidvlinder, Hermelijn, Holenduif, Hooibeestje, Houtduif, Houtpantserjuffer, Houtsnip, Houtspaander, Huiskat, Huismoeder, Huismuis, Huismus, Huisspitsmuis, Huiszwaluw, Icarusblauwtje, Ijslandse Grutto, Ijsvogel, Inachis Io, Indische Gans, Itame Brunneata, Kaapse Casarca, Kanoet, Kastanjemineermotje, Kauw, Keep, Keizergans, Kemphaan, Kerkuil, Kievit, Klaverspanner, Klein Geaderd Witje, Klein Hazeldwergmotje, Klein Koolwitje, Kleine Barmsijs, Kleine Beer, Kleine Bruine Zwenkgrasmot, Kleine Karekiet, Kleine Mantelmeeuw, Kleine Mantelmeeuw (Graellsii), Kleine Plevier, Kleine Rietgans, Kleine Roodoogjuffer, Kleine Schapengrasmot, Kleine Strandloper, Kleine Vos, Kleine Vuurvlinder, Kleine Wintervlinder, Kleine Zilverreiger, Kleine Zomervlinder, Kleine Zwaan, Kleinst Waterhoen, Kluut, Kneu, Knobbelzwaan, Koekoek, Koereiger, Kokardezaagbek, Kokmeeuw, Kolgans, Kolibrievlinder, Konijn, Koolmees, Koperuil, Koperwiek, Kraanvogel, Krakeend, Kramsvogel, Krombekstrandloper, Kromzitter, Krooneend, Kroosvlindertje, Kruisbek, Kuifduiker, Kuifeend, Kwak, Kwartel, Kwartelkoning, Laatvlieger, Landkaartje, Lantaarntje, Larus Graellsii, Larus Minutus, Larus Ridibundus, Lepelaar, Lestes Viridis, Lieveling, Maisboorder, Mandarijneend, Matkop, Meerkoet, Meervleermuis, Meldevlinder, Merel, Metaalvlinder, Moeflon, Mol, Motacilla Alba Ssp, Motacilla Flava Ssp, Muntvlindertje, Mus Domesticus, Muskuseend, Muskusrat, Nachtegaal, Nijlgans, Nonnetje, Noordse Kwikstaart, Oeverloper, Oeverzwaluw, Ongebandeerd Elzenvouwmijnmotje, Ongebandeerd

Hazelvouwmijnmotje, Ooievaar, Oranje Luzernevlinder, Oranje O-Vlinder, Oranjetipje, Paapje, Paardenbijter, Paardenbloemspanner, Paarlemoerlichtmot, Parus Caeruleus, Parus Montanus, Patrijs, Peper-En-Zoutvlinder, Pestvogel, Phalacrocorax Carbo Ssp, Philomachus Pugnax, Pijlstaart, Pimpelmees, Plakker, Plataanvouwmijnmotje, Platbuik, Poelruiter, Pontische Meeuw, Porseleinhoen, Porzana Pusilla, Purperreiger, Putter, Raaf, Ransuil, Ratelaarspanner, Regenwulp, Rietgors, Rietvink, Rietzanger, Ringmus, Rode Wouw, Roek, Roerdomp, Roesje, Roodborst, Roodborsttapuit, Roodhalsfuut, Roodhalsgans, Rosse Grutto, Rosse Vleermuis, Rosse Woelmuis, Rotgans, Rouwkwikstaart, Ruige Dwergvleermuis, Ruigpootbuizerd, Scholekster, Seringensteltmot, Sijs, Sint-Jacobsvlinder, Sint-Jansvlinder, Slechtvalk, Slobeend, Smaragdlibel, Smelleken, Smient, Snor, Soepeend, Soepgans, Sperwer, Spotvogel, Spreeuw, Sprinkhaanzanger, Staartmees, Stadsduif, Steenloper, Steenrode Heidelibel, Steenuil, Steltkluut, Stormmeeuw, Strandplevier, Stro-Uiltje, Tafeleend, Taiga-/Toendrarietgans, Tapuit, Taxusspikkelspanner, Temmincks Strandloper, Tjiftjaf, Toendrarietgans, Topper, Torenvalk, Tuinfluiter, Tureluur, Turkse Tortel, Tweebandig Elzenvouwmijnmotje, Tweebandig Hazelvouwmijnmotje, Tweestreepvoorjaarsuil, Variabele Waterjuffer, Veldleeuwerik, Veldmuis, Velduil, Viervlek, Vink, Visarend, Visdief, Vleermuis (Onbekend), Vogelkersstippelmotje, Volgeling, Vos, Vroege Glazenmaker, Vuurgoudhaan, Vuurjuffer, Wachtervlinder, Wapendrager, Waterhoen, Waterpieper, Waterral, Watersnip, Watersnuffel, Waterspitsmuis, Watervleermuis, Wespendief, Wezel, Wilde Eend, Wilde Zwaan, Wilgenslakkenspoortje, Windevedermot, Winterkoning, Wintertaling, Wit Appelstippelmotje, Witgat, Witgestreepte Beemdgrasmot, Witoogeend, Witsterblauwborst, Witte / Rouwkwikstaart, Witte Grijsbandspanner, Witte Kwikstaart, Witte Tijger, Witvleugelstern, Woelrat, Woelrat/Molmuis, Wulp, Zanglijster, Zeearend, Zilvermeeuw, Zilverplevier, Zilverstreep, Zomertaling, Zomertortel, Zwaangans, Zwartbandspanner, Zwarte-C-Uil, Zwarte Heidelibel, Zwarte Ibis, Zwarte Kraai, Zwarte Mees, Zwarte Roodstaart, Zwarte Ruiter, Zwarte Stern, Zwarte Wouw, Zwarte Zwaan, Zwartkamdwergspanner, Zwartkop, Zwartkopmeeuw, Zwartkruin Bramendwergmotje, Zwartkruin Iepenvouwmijnmotje, Zwartsprietdikkopje