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Over a century of data reveal more than 80% decline in butterflies in the Netherlands

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ABSTRACT

Opportunistic butterfly records from 1890 to 2017 were analysed to quantitatively estimate the overall long-term change in occurrence of butterfly species in the Netherlands. For 71 species, we assessed trends in the number of occupied 5 km × 5 km sites by applying a modified List Length method, which takes into account changes in observation effort. We summarised the species trends in a Multi-Species Indicator (MSI) by taking the geometric mean of the species indices. Between 1890–1930 and 1981–1990, the MSI decreased by 67%; downward trends were detected for 42 species, many of which have disappeared completely from the Netherlands. Monitoring count data available from 1992 showed a further 50% decline in MSI. Combined, this yields an estimated decline of 84% in 1890–2017. We argue that in reality the loss is likely even higher. We also assessed separate MSIs for three major butterfly habitat types in the Netherlands: grassland, woodland and heathland. Butterflies strongly declined in all three habitats alike. The trend has stabilised over recent decades in grassland and woodland, but the decline continues in heathland.

1. Introduction

The worldwide loss of biodiversity is widely acknowledged and intentions have been announced to restore biodiversity as far as possible, as formulated in the Aichi Biodiversity Targets (SCBD, 2014), which have been adopted by the European Union including the Netherlands. In order to assess restoration success adequately, we need to know how the state of biodiversity has changed, not only over recent years, but also over a long period. This requires data on historic species occurrences, but such knowledge is fragmentary (Bonebrake et al., 2010). As a result, we are suffering from the shifting baseline syndrome: while evaluating recent trends in biodiversity, we are unaware of historic losses (Pauly, 1995).

Detailed quantitative information on changes in biodiversity is available for the last 30–40 years (e.g., Butchart et al., 2010), as around 1980 large-scale monitoring schemes have started for several species groups (e.g., butterflies in the UK 1976: Pollard and Yates, 1993; breeding birds in the Netherlands 1984: Van Turnhout et al., 2008). A wealth of data from earlier periods also exists, but these data were not collected using standardised field protocols; they are 'opportunistic'. In the Netherlands, for instance, many opportunistic data are available on the occurrence of butterfly species from the end of the 19th century, while a standardised monitoring scheme was launched only in 1990. While changes in occurrence of species and in species diversity have been deduced from these historic data before (Van Swaay, 1990; Carvalheiro et al., 2013), an overall quantitative picture, summarising long-term trends in distribution or abundance of butterfly species, has been missing until now.

Here, we re-analyse historic opportunistic data to produce a quantitative estimate of the overall long-term change in butterfly occurrence in the Netherlands: the Multi-Species Indicator (MSI). We followed the calculation method of the Living Planet Index (LPI), a well-known indicator to assess biodiversity change (e.g. global LPI: Collen et al., 2009; national LPI: Van Strien et al., 2016).

In recent years, insight has grown into which statistical methods are available for and applicable to opportunistic data, what the strengths and weaknesses of these methods are, and how to avoid bias due to changes in observer effort during the period under study (Van Strien et al., 2013; Isaac et al., 2014). We applied a modified List Length method (hereafter called LL; Szabo et al., 2010; Barnes et al., 2015) to

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estimate changes in distribution (number of occupied 5 km × 5 km sites) in nearly all Dutch butterfly species for 1890–2017, which we then summarised to calculate the MSI. To examine the sensitivity of the method, we compared the results with (i) the changes in distribution on a 1 km × 1 km scale obtained by the LL method from 1990 on, (ii) the changes in abundance derived from monitoring data from 1992 on. We recalculated the MSI for 1890–2017, using the more detailed abundance data from 1992 onwards.

In addition, we composed MSIs for the three major butterfly habitat types that occur in the Netherlands and that host different butterfly assemblages: grassland, woodland and heathland. One century ago, almost all grasslands in the Netherlands were rich in herb species, although they were used as farmland. Since then, farmland practice has intensified strongly as indicated by the nowadays 30–40 fold higher application of artificial nitrogen fertiliser (Table S1), eliminating grassland herb species (Bobbink et al., 2010; Soons et al., 2017). The area of semi-natural grassland, still harbouring herb species and nowadays managed as nature reserves, shrunk from 40% of land cover to a mere 3.0% nowadays (Table S1). Woodland cover has expanded over the 20th century and the proportion of mature woodland increased (Table S1). Woodland structure and composition diversified with woodland age. Around 1900, 13.4% of the Netherlands consisted of heathland, but a large part was lost due to reclamation, mostly before 1940 (Table S1). Since then, both heathland and woodland have undergone soil acidification and eutrophication due to atmospheric deposition of SO_2, NO_x and NH_y (Roem et al., 2002; Jones et al., 2014). On the basis of these changes, we expect to find declining long-term trends in butterfly species in all three habitat types. Over recent decades, we hypothesise declines in heathland and grassland species to continue, but trends in woodland species to stabilise (Van Strien et al., 2016).

2. Material and methods

2.1. Study species

We included all sedentary butterfly species of the Netherlands, except the White-letter hairstreak (*Satyrium w-album*), for which data were sparse ($n = 71$; Table S2). Nomenclature follows Fauna Europaea (www.faunaeur.org) as of July 2018.

2.2. Data

Two datasets were used.

(i) *National Database Flora and Fauna* (*NDFF*; *opportunistic data*) – This database comprises records of butterfly specimens in all Dutch natural history museums and private collections. In addition, all records found in scientific journals including ‘grey’ literature were included in this database. Until several decades ago, butterflies in the Netherlands were mainly caught by a small group of entomologists to build collections. For each of the four periods up to 1980, 13,000–29,000 records were collected (Table 1); observations came from one-third to half of the terrestrial area of the Netherlands, mainly from coastal dunes and higher sandy soils, i.e., from the most important butterflies habitats.
From 1980 onwards, copious field data have been collected by volunteer field workers covering almost the entire country. In recent years, many volunteers report sightings on on-line observation platforms, such as landkaartje.vlinderstichting.nl/index.php, www.telmee.nl and www.waarneming.nl. The resulting database currently contains over 5 million observations of 113 butterfly species, including many vagrants. In recent periods, almost all sites were surveyed (Table 1). All database records have been validated by butterfly experts.

(ii) *Dutch Butterfly Monitoring Scheme* (*standardised data*) – This scheme started in 1990 and applies the method developed for the British Butterfly Monitoring Scheme (Pollard and Yates, 1993). Counts were conducted along fixed transects, which were typically up to 1 km long. Observers recorded all butterflies within 2.5 m on either side and within 5 m ahead and above them. Weekly surveys were conducted between 1 April and 30 September when weather conditions met specified criteria (Van Swaay et al., 2018). Sufficient data for trend estimation were available from 1992 onwards.

Table 1
Characteristics of opportunistic presence records per period. The maximum number of 5 km × 5 km terrestrial sites in the Netherlands amounts 1696.

	Total number of records collected	Mean number of records per species	Number of sites surveyed	List length
1890–1939	16,301	229	598	14.2
1940–1960	28,538	402	861	16.1
1961–1970	13,668	192	725	11.3
1971–1980	24,900	351	892	12.2
1981–1990	202,416	2850	1476	20.1
1991–2000	913,686	12,868	1647	24.3
2001–2010	1,614,616	22,741	1675	25.2
2011–2017	2,219,394	31,259	1672	25.9

2.3. Estimating trends per species

The data analysis is challenging as the distributional data were not collected using standardised methods. The huge rise in opportunistically collected records over the study period (Table 1) brings a risk of producing biased positive trend estimates if not taken into account. To prevent this, occupancy modelling is currently viewed as the best statistical correction method available. It takes into account the detection probability of species and through that it enables to adjust for observer effort (MacKenzie et al., 2006; Van Strien et al., 2013; Isaac et al., 2014). Occupancy modelling method requires replicated visits to the same site within the same season and, unfortunately, such replicates are scarce before 1990, ruling out the use of this method. As an alternative, we applied the LL method developed by Szabo et al. (2010), which uses the number of species recorded to correct for variation in observer effort. This method performs reasonably well in studies with simulated data, although it cannot cope with all inherent shortcomings of opportunistic data (Isaac et al., 2014). It takes into account the number of species in a logistic regression model (Szabo et al., 2010):

$$\mathrm{logit}(P_{it}) = \mathrm{alpha} + \mathrm{year}_t + \log(\mathrm{no.\ species}_{it}) \tag{1}$$

where P_{it} is the probability of a species to be observed in site i in year t, alpha is the intercept, year_t is estimated as fixed effect and no. $\mathrm{species}_{it}$ is the number of species observed in a site in a year. The use of the log expresses that P increases with the number of species observed, but with a reduced magnitude. However, it is plausible that this relation is asymptotic rather than logarithmic, i.e., after a particular list length is obtained, a further increase in the number of species on the list will not any longer affect P. Therefore, we modified the LL method to a form that is similar to the Michaelis-Menten equation (e.g. Raaijmakers, 1987). Furthermore, we used periods spanning a number of years instead of single calendar years, because historic data were too sparse to produce annual estimates. Data were aggregated into 1890–1939, 1940–1960, 1961–1970, 1971–1980, 1981–1990, 1991–2000, 2001–2010 and 2011–2017.

We used squares of 5 km × 5 km as sites as many observations were not precisely georeferenced before 1990. For each site we compiled a list per period of all species observed and we deduced non-detection records for each study species by assuming all cases in which a species was not on the list as non-detected in that particular site and period. Finally, we added site_i as random effect, to account for spatial differences in surveys over time. This leads to the following modified LL model:

$$\text{logit}(P_{it}) = \text{period}_t + (b_1 * \text{no. species}_{it})/(b_2 + \text{no. species}_{it}) + \text{site}_i \quad (2)$$

where b_1 and b_2 are the two parameters describing the Michaelis-Menten equation. Hence, a longer list length increased the probability of a species to be observed. Generally, this probability hardly increased if a list contains > 40 species.

For each species, only those sites were included in which the species had been recorded at least once. We used R 3.5.1 (R Core Team, 2018) and fitted models in a Bayesian mode of inference using JAGS with vague priors for all parameters (Plummer, 2009). We standardised the number of species in the model to enhance model convergence (Kéry, 2010). Posterior means are reported as point estimators of occurrence probability and associated Bayesian standard deviations as standard errors (SE) and 1.96 ∗ SE as 95% confidence intervals (CI). Occurrence probability was converted into period indices with the first period set to 100. We assessed the trend per species as a derived parameter in the model by assessing the slope of the linear regression line through the estimates of the period effects.

In a separate analysis, we ran the LL model with data from 1990 to 2017 using years instead of longer periods, both with 5 km × 5 km sites and 1 km × 1 km sites. The finer spatial scale was possible because observations after 1990 were precisely georeferenced. We also analysed the monitoring data to assess trends in population abundance in 1992–2017, using the software TRIM (Pannekoek and van Strien, 2005), a Poisson GLM programme to produce annual indices and linear trends. These analyses were not feasible for all species though, because a number of species had gone extinct before 1990 or are too scarce to produce reliable trend estimates (Table S3).

2.4. Composing indicator values

We summarised trends of groups of species by calculating the geometric mean per period (or year) of the annual species indices, thereby composing the MSI. This procedure is widely adopted to create indicators for biodiversity change (Collen et al., 2009; Van Strien et al., 2016). The geometric mean is stable when positive and negative trends, as well as their magnitude, are in balance. The geometric mean goes down when the number of declining species is higher than the number of species that are increasing at the same rate, and vice versa. The geometric mean values were converted into annual indices with first period (or year) set as 100. We included the uncertainty due to sampling error of species indices into the confidence limits of the MSI, thereby ensuring error propagation. Finally, we applied LOESS (Locally weighted polynomial regression) to produce smoothed indices and confidence intervals (Soldaat et al., 2017).

MSIs were compiled for all butterfly species combined and per habitat type. Trends in the indicator were statistically tested by estimating the percentage change between the smoothed index values of the last period and that of the first period. The percentage change is significant if its CI does not include 0. Differences in trends between periods were tested similarly (Soldaat et al., 2017).

2.5. Trends per habitat type

For grassland, woodland and heathland MSIs we used a number of selected species only (see Table S2). For grassland (farmland) species, we followed Van Swaay and van Strien (2005) by adopting the 17 species selected for the European grassland butterfly indicator. Two species from this set, which never inhabited the Netherlands, were replaced by Small Pearl-bordered Fritillary (*Boloria selene*) and Glanville Fritillary (*Melitaea cinxia*). Ten species are considered characteristic grassland species (see Table S2); the remaining seven species are eurytopic species, i.e., they also occur in other habitat types, especially in heathland. In order to produce trends in grassland, we selected 5 km × 5 km sites that contained over 80% agricultural area in both 1900 and 2000 and < 10% heathland in 1900, obtained through GIS-analysis of digitized maps of historic land use (Knol et al., 2004).

For woodland and heathland, we included only species with a close association with these habitat types. Following Van Strien et al. (2016), we selected species as specialists of woodland or heathland if their current density in the respective habitat types was at least twice their density in any other habitat types. Trends in woodland were assessed using sites with woodland present in 2000. Trends in heathland were derived from 5 km × 5 km sites that had heathland present in 1900. For heathland, we also compared trends in sites where heathland persisted, i.e., at least some heathland was still present in 2000, with those in sites where heathland had disappeared completely before 2000.

Finally, we produced three habitat-specific MSIs for 1992–2017 derived from monitoring data using transects in grassland, woodland and heathland.

3. Results

3.1. Trends of butterfly species

In 2011–2017, 29 out of 71 species were reported from fewer 5 km × 5 km sites than in 1890–1939, although observer effort has increased strongly (Table 1). After adjusting for observer effort, downward trends were assessed for 42 species, many of which have disappeared completely from the Netherlands (Table S2); 21 species increased in distribution. From 1890 to 2017, the MSI based on 5 km × 5 km sites steeply declined ($P < 0.05$; Fig. 1). The MSI dropped from 100 at the start of the study period to 33 in 1990, i.e., the distribution of butterfly species overall decreased by 67%. From 1990 onwards, the MSI based on 5 km × 5 km sites further declined with 18% (Fig. 2), resulting in an estimated index value of 33 ∗ 0.82 = 27. Thus, in 1890–2017 butterflies overall decreased by 73% in the mean number of occupied 5 km × 5 km sites. This decline cannot entirely be attributed to extinctions, because even if extinct species are excluded from the analysis, a significant decline is found ($P < 0.05$). Overall, many habitat-specific species have decreased strongly or have become extinct, e.g. Lulworth Skipper (*Thymelicus acteon*), while many eurytopic species have increased, e.g. Holly Blue (*Celastrina argiolus*) (Table S2).

While the decline in MSI based on 5 km × km sites occurred mainly between the first (1890–1939) and the fifth (1981–1990) period (Fig. 1), a more detailed look at what happened after 1990 reveals a considerable further decline from 1990 to 2017 ($P < 0.05$; Fig. 2). In this period, the distribution of 22 of 46 species for which we had sufficient data decreased, while that of 16 species increased (Table S3). When the calculations for these years are based on the number of occupied 1 km × 1 km sites, a slightly stronger decline is found ($P < 0.05$; Fig. 2). This is not unexpected as species trends assessed at a more fine-grained spatial scale are more sensitive to changes in distribution (Thomas and Abery, 1995; Gaston et al., 2000). Species trends based on abundance data are expected to be even more sensitive than trends in distribution. Indeed, the MSI based on abundance data is more negative and drops to a value of 50 in 2017 ($P < 0.05$; Fig. 2).

If, using the most sensitive trend estimates, we combine the decline based on 5 km × 5 km sites in 1890–1990 with the decline in abundance in 1990–2017, the index value in 2017 would be 33 ∗ 0.50 = 16.5, so the estimated total loss over the entire study period would be over 80%.

3.2. Trends per habitat type

Butterflies strongly decreased in grassland, woodland and heathland alike (Fig. 3). For grassland species, the MSI based on 5 km × 5 km sites declined to about 20 in 1990. All characteristic species decreased, while no eurytopic species increased or decreased significantly (Fig. 3; Table

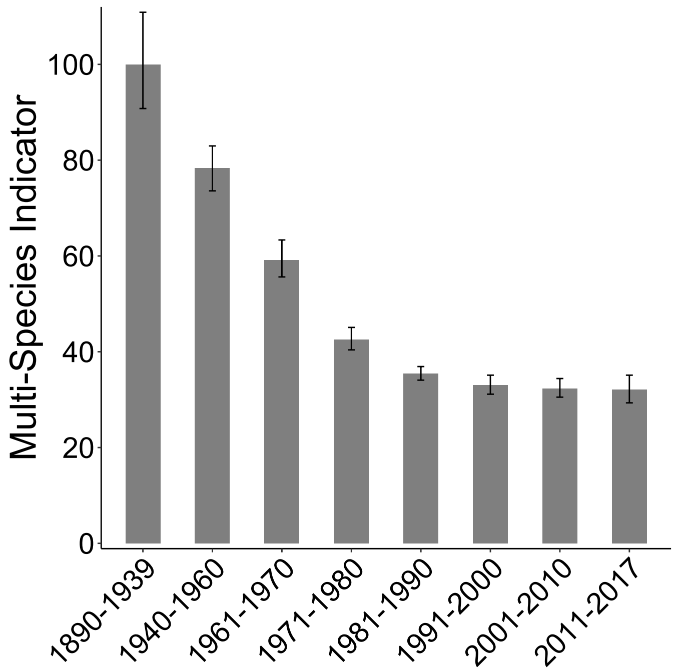

Fig. 1. Multi-Species Indicator (± 95% confidence intervals) for butterfly species of the Netherlands (n = 71 species), based on trends per species derived from List Length analysis using presence/absence data from 5 km × 5 km sites.

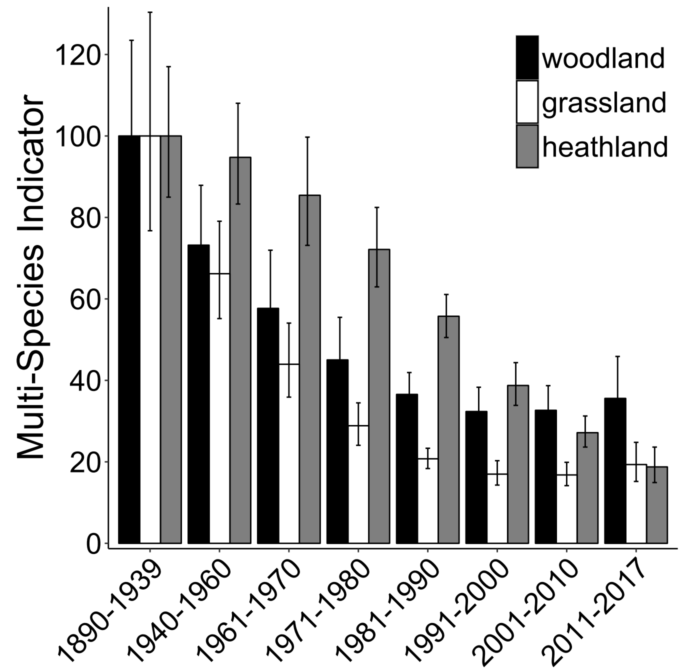

Fig. 3. Multi-Species Indicators (± 95% confidence intervals), based on trends per species derived from List Length analysis using 5 km × 5 km sites, for woodland (n = 8 species), grassland (n = 17 species) and heathland (n = 15 species).

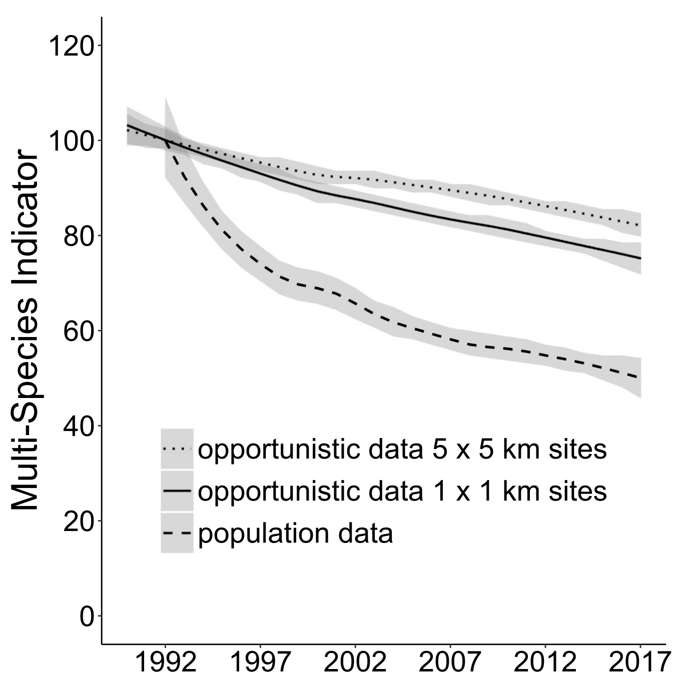

Fig. 2. Multi-Species Indicators (± 95% confidence intervals) for butterfly species of the Netherlands, based on trends per species derived from List Length analysis using presence/absence data from 5 km × 5 km sites and from 1 km × 1 km sites as well as trends derived from population abundance data using Poisson regression. The same 46 species were included in all three MSIs.

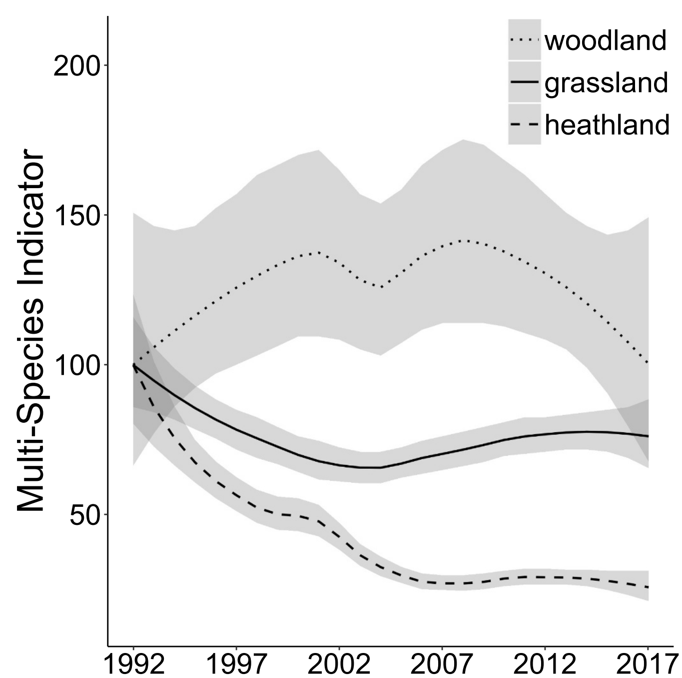

Fig. 4. Multi-Species Indicators (± 95% confidence intervals) for woodland (n = 5 species), grassland (n = 10 species) and heathland (n = 13 species), all derived from population abundance data using Poisson regression.

S2). In 1990, mainly eurytopic species remained. After 1990, the MSI based on abundance data did not change significantly ($P > 0.05$; Fig. 4; Table S4). Some species decreased, such as Wall Brown (*Lasiommata megera*), but others increased, e.g., Orange Tip (*Anthocharis cardamines*).

The MSI based on 5 km × 5 km sites for woodland species also strongly declined until 1890–1990; thereafter, it remained stable, both when calculated on the basis of distribution (Fig. 3; Table S2), as well as abundance (Fig. 4; Table S4). The heathland MSI based on distribution data showed a negative trend, which, in contrast to grassland and woodland MSIs, continued to decline after 1990, both in distribution as in abundance ($P < 0.05$; Fig. 3; Fig. 4; Table S2). Downward trends were steeper in areas where heathland disappeared completely than in areas where heathland is still present (Fig. 5).

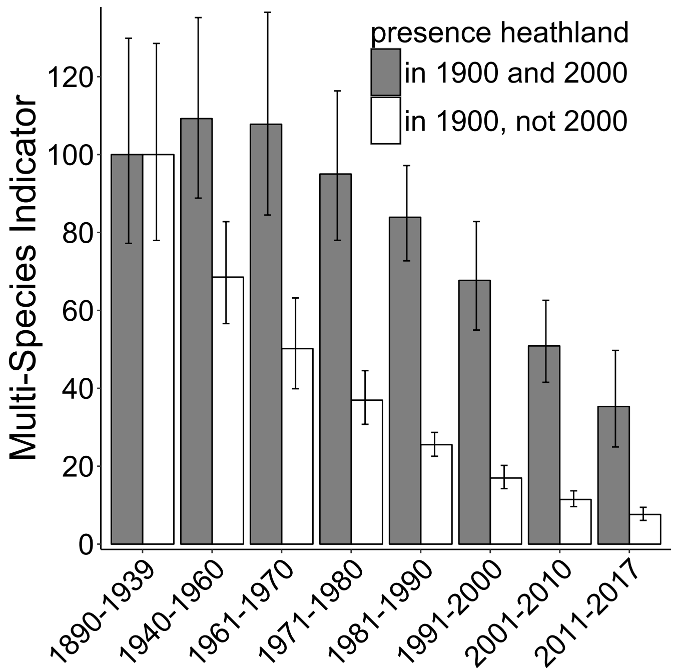

Fig. 5. Multi-Species Indicators (± 95% confidence intervals) for heathland butterflies in the Netherlands (n = 15 species), based on trends per species derived from List Length analysis using 5 km × 5 km sites which differ in the presence of heathland in 2000.

4. Discussion

4.1. Trends of butterfly species

The prevalence of declining butterfly trends is in line with other studies. Strong declines in butterflies have been reported from other European countries too, e.g., Belgium (Maes and van Dyck, 2001), Finland (Kuussaari et al., 2007) and the United Kingdom (Fox et al., 2011), though few studies look back as far as our paper does. On a timescale comparable to ours (1840–2013), Habel et al. (2016) recorded severe declines (46 out of 117 species lost) for a restricted area in southeastern Bavaria. Carvalheiro et al. (2013) reported a decline of species richness for butterflies in Belgium, the United Kingdom and the Netherlands in 1950–2009. On a continental scale, the European Red List of Butterflies (Van Swaay et al., 2010) shows many more declining species (31%) than increasing species (4%) for 1998–2008, and about 9% of the European species are currently considered threatened.

4.2. Biases in trend estimations

Our estimate of > 80% overall decline in butterflies is based on the use of the LL-method, which is known to have some shortcomings. Isaac et al. (2014) examined the LL method analysis in detail in a simulation study to assess how well this method deals with potential biases in opportunistic data. They tested several scenario's, such as conducting more visits and changes in detectability of species, several of which apply to our study.

First, our data show an increasing observation effort resulting in more sites surveyed and longer lists per site (Table 1). Isaac et al. (2014) demonstrated that the LL model most similar to ours ("RR + SF + LL + Site" in their paper) is not vulnerable to such changes (see scenario "more visits" and scenario "less effort per visit" in their paper).

Second, longer lists may also occur when many species increase. Statistical correction by LL then induces an overestimation of the magnitude of declines and underestimation of the increases (Szabo et al., 2010; Barnes et al., 2015). In our case most species decrease, which provokes an underestimation rather than an overestimation of the overall decline. Isaac et al. (2014) confirmed the latter in their scenario "non-focal species declines".

Third, historic data may suffer from selective recording against some common species. Most of the older records are from specimens in museum and private collections, for which collectors tried to balance the number of specimens across species. As a consequence, specimens of the most common species have often been ignored if enough specimens had already been collected. Nowadays, data are mainly collected for studying occurrence of species and population sizes, and there is no longer an obvious limit to recording common species. As a consequence, the reporting of common species in the first time periods may have been too low, implying a lower detection probability of these species in these periods, while the LL method cannot cope with changes in detection probability (see scenario "more detectable" in Isaac et al., 2014). So, the increase of some common species is probably overestimated, which may indirectly result in the exaggeration of the declines of less common species (Szabo et al., 2010). To test how these two effects affect the long-term MSI, we recalculated all species trends excluding the records of the most common species (three *Pieris* species, Small copper (*Lycaena phlaeas*), Common blue (*Polyommatus icarus*) and Small tortoiseshell (*Aglais urticae*)). The resulting MSI was very similar to the MSI based on all species (64% instead of 67% loss in 1890–1990 and 69% vs 68% in 1890–2017), suggesting that selective recording in the past has negligible effects on our MSI.

While the first scenario (more effort) and third scenario (selective recording) have no obvious consequences for the estimation of the MSI of all species together, the second scenario (non-focal species declines) may underestimate the overall decline in MSI. This adds to the underestimations due to deriving trends from distributional data on a coarse-grained spatial scale over a large part of the study period.

In conclusion, we found a decline of at least 80% in the Netherlands over 1890–2017, but in terms of abundance, butterfly species most probably have declined considerably more. This substantiates the common knowledge that the butterfly fauna of the Netherlands nowadays is a fraction of what it used to be in earlier days (Van Swaay, 1990).

4.3. Trends per habitat type

We expected strong declines of butterfly species over the entire study period in all habitat types and continuing declines in heathland and grassland species over recent decades, but stabilisation in woodland butterflies. These expectations were largely confirmed, with the exception that the trend of grassland species stabilised recently, after a strong decrease until 1980. In recent years, efforts of nature restoration have caused the area of semi-natural grasslands to slightly expand, which might explain the stable trend during the last decades. The trend of woodland butterflies, which decreased until 1980, stabilised afterwards. The significant expansion recently of several rare woodland butterflies: Wood White (*Leptidea sinapis*), White Admiral (*Limenitis camilla*), Purple Emperor (*Apatura iris*) and Silver-washed Fritillary (*Argynnis paphia*) (Table S3), indicates that habitat conditions are improving, possibly favoured by a modest revival of small-scale woodland management practices (WallisDeVries and Prick, 2015) in combination with climatic warming (Settele et al., 2008). Before 1980, heathland butterfly species decreased less strongly in occurrence than grassland and woodland species (Fig. 3). At sites where heathland persisted, heathland butterflies remained stable until 1980, even though the heathland area was reduced (Fig. 5). Their decline after 1980 is probably due to the combined effects of fragmentation of the remaining heathland area (Van Strien et al., 2011) and atmospheric nitrogen deposition, as characteristic heathland species typically depend on low-nitrogen environments (WallisDeVries and van Swaay, 2017).

5. Significance

It is typical to report changes in biodiversity for only the last few decades. The global Living Planet Index for instance is assessed from 1970 onwards (Collen et al., 2009) and other trends are assessed over an even shorter period (e.g. Butchart et al., 2010; Van Strien et al., 2016; Hallmann et al., 2017). However, the emerging pictures are distorted by the shifting baseline syndrome (Pauly, 1995), because, as we here showed for Dutch butterfly species, much more was already lost before 1970. Reporting over a few decades implies that this long-term loss remains hidden and that recorded recoveries cannot be compared against a proper historic baseline.

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