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The Effects of Common Gardening Practices on Biodiversity

Bachelor Thesis

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Bremen, 10.04.2021

J. Böhme

Jana Böhme (5016625)

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List of Abbreviations

et al.	et aliae (and others)
CBD	United Nations Convention on Biological Diversity
COP	Conference of the Parties
EU	European Union
URL	Uniform Resource Locator (web address)
NGO	Non-Governmental Organisation
WWF	World Wildlife Fund
UN	United Nations
NABU	Naturschutzbund Deutschland e. V
e.g.	exempli gratia (for example)
i.e.	id est (that means)
No.	Number
CI	Confidence Interval

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1 Abstract

The globally increasing urbanisation is a threat to the biodiversity of insects and birds, resulting in severe consequences for humans such as decreasing pollination services and, thus, the loss of food security. In order to counteract this developments, alternative biodiversity-friendly habitats must be established within urban areas. Private gardens as intensively managed, major components of cities are a decisive factor within urban biodiversity conservation. As private gardens underly the individual management decisions of their owners, adequate information on biodiversity-friendly gardening practices is necessary. Therefore, the present study examined the effects of common gardening practices like the planting of native and non-native plants and flowers, the use of chemical pesticides, mowing and bird feeding on the biodiversity of insects and birds throughout a global meta-analysis. Out of 178 identified publications, assessing the effects of common gardening practices, 21 publications were considered within the present meta-analysis. In order to measure the effects of gardening practices on the species biodiversity of insects and birds, abundance, richness, diversity, breeding measures and mortality, directly correlated to the implementation of certain garden management actions, were used as indicators. It was emphasised that heterogeneous managed gardens consisting of plenty of plant types and species offer various habitats to insects and birds. Thereby, native insect and bird species were best promoted by native planting patterns. Exceptionally, butterflies relied on specific butterfly-friendly, also non-native, plant species. Adverse actions aiming at homogeneous plant compositions throughout gardens such as mowing and the use of chemical pesticides destructed important species habitats. The effects of bird feeding were inconsistent as positive impacts on the current generation, but negative impacts on their breeding measures, thus the following generation, were revealed, whereby the timespan of feeding and the type of supplemented food were important influence factors. These results of the meta-analysis were compared with popular scientific recommendations on common gardening practices on the internet to identify knowledge gaps and potential misinformation. Similar to the foregoing meta-analysis, a heterogeneous garden design was the major recommendation on internet platforms in order to promote insects and birds. However, a lack of information on the importance of native vegetation and the appropriate timespan of bird feeding was prevalent. Generally, articles on sustainable gardening were primarily available for people that are explicitly interested in such information. These findings imply a broader need of public scientifically profound education on the value of biodiversity, the role of private gardens in conservation strategies and applicable ecological gardening practices as well as appropriate tools like native wildflower mixtures in order to implement these. Generally, facing the current challenge of biodiversity decline worldwide, heterogeneous and wild gardens should be commonly perceived as aesthetically pleasing instead of homogeneous and neat properties.

Die weltweit zunehmende Urbanisierung bedroht die Biodiversität von Insekten und Vögeln und hat für den Menschen schwerwiegende Folgen wie zum Beispiel abnehmende Bestäubungsleistungen und damit den Verlust der Ernährungssicherheit. Um dieser Entwicklung entgegenzuwirken, sollten alternative, biodiversitätsfreundliche Lebensräume innerhalb städtischer Gebiete entstehen. Privatgärten, als intensiv bewirtschaftete, wesentliche Bestandteile von Städten sind ein entscheidender Faktor innerhalb des städtischen Biodiversitätsschutzes. Da private Gärten den individuellen Entscheidungen ihrer Besitzer unterliegen, müssen ausreichend und angemessene Informationen über biodiversitätsfreundliche Gartenpraktiken verfügbar sein. Die vorliegende Studie hat daher die Auswirkungen gängiger Gartenpraktiken wie die Bepflanzung mit einheimischen und nicht-einheimischen Pflanzen und Blumen, den Einsatz chemischer Pestizide, Rasenmähen und das Füttern von Vögeln auf die Biodiversität von Insekten und Vögeln im Rahmen einer globalen Meta-Analyse untersucht. Von 178 identifizierten Publikationen, die sich mit den Auswirkungen gängiger Gartenpraktiken auf Biodiversität beschäftigten, wurden 21 Publikationen in der vorliegenden Meta-Analyse berücksichtigt. Um die Auswirkungen von Gartenpraktiken auf die Artenvielfalt von Insekten und Vögeln zu messen, wurden Abundanz ('abundance'), Reichtum ('richness'), Diversität ('diversity'), Brutmaße ('breeding measures') und Mortalität ('mortality'), die direkt mit der Umsetzung bestimmter Gartenpraktiken korreliert waren, als Indikatoren verwendet. Es wurde festgestellt, dass heterogen bewirtschaftete Gärten mit einer Vielzahl von Pflanzenarten und -sorten, Insekten und Vögeln viele verschiedene Lebensräume bieten. Dabei wurden einheimische Insekten- und Vogelarten am besten durch einheimische Pflanzen gefördert. Schmetterlinge bildeten dabei eine Ausnahme, da sie auf bestimmte schmetterlingsfreundliche, auch nicht einheimische, Pflanzenarten angewiesen sind. Maßnahmen, die auf eine homogene Pflanzenzusammensetzung in den Gärten abzielen, wie beispielsweise Rasenmähen und der Einsatz von chemischen Pflanzenschutzmitteln, zerstörten wichtige Lebensräume für Arten. Die Auswirkungen der Vogelfütterung waren inkonsistent, da positive Auswirkungen auf die aktuelle Generation, aber negative Auswirkungen auf deren Brutmaße, also die nachfolgende Generation, festgestellt wurden, wobei der Zeitraum der Fütterung und die Art der zugeführten Nahrung wichtige Einflussfaktoren waren. Diese Ergebnisse der Meta-Analyse wurden mit populärwissenschaftlichen Empfehlungen bezüglich gängiger Gartenpraktiken im Internet verglichen, um Wissenslücken und mögliche Fehlinformationen zu identifizieren. Ähnlich wie in der vorangegangenen Meta-Analyse war eine heterogene Gartengestaltung die wichtigste Empfehlung zur Förderung von Insekten und Vögeln auf Internetplattformen. Allerdings war ein Mangel an Informationen über die Bedeutung von einheimischer Vegetation und die angemessene Zeitspanne der Vogelfütterung festzustellen. Generell waren Artikel über nachhaltiges Gärtnern vor allem für Personen verfügbar, die explizit an diesen Informationen interessiert sind. Diese Ergebnisse implizieren einen großen Bedarf an öffentlicher, wissenschaftlich fundierter Aufklärung über den Wert der

Biodiversität, die Rolle des Privatgartens im Naturschutz und anwendbare ökologische Gartenpraktiken sowie geeignete Mittel wie einheimische Wildblumenmischungen, um diese umzusetzen. Anstelle von homogenen und gepflegten Gärten sollten, angesichts der aktuellen Herausforderung des weltweiten Rückgangs der Biodiversität, heterogen gestaltete und wilde Gärten gesellschaftlich als ästhetisch angesehen werden.

2 Introduction

The world is rapidly urbanising. In 2015, more than a half million km^2 of the global land area were covered by cities (Melchiorri et al., 2018), and an additional coverage of 1.2 million km^2 is expected until 2030, which would constitute an increase of urban area of 185 % within 30 years (Seto et al., 2012). Few years ago, in 2007, more people were recorded living in urban than in rural areas for the first time (*World Urbanization Prospects*, 2019). In 2018, 55 % of the world's population, in European and North American countries even a median of 74 % of a country's population, settled in cities (*World Urbanization Prospects*, 2019). An ongoing growth of the urban population is predicted (*World Urbanization Prospects*, 2019). Urbanisation is regarded as a major reason for habitat fragmentation (Butchart et al., 2010) and habitat loss (Plascencia & Philpott, 2017; Forister et al., 2019) and could therefore be a decisive driver of declines in species biodiversity worldwide. Throughout the recent years, particular concern has been raised on the decreasing species diversity of insects (Hallmann et al., 2017). Currently, there is an ongoing contentious debate on insect declines due to divergent findings of scientific studies (van Klink et al., 2020) as well as a lack of scientific proof regarding the majority of insect species and regions of the world (Forister et al., 2019; van Klink et al., 2020).

However, the loss of insect, species and overall biodiversity are already politically and societally recognised topics on international, national and regional scales. The worldwide 196 parties of the United Nations Convention on Biological Diversity (CBD) aim for the definition and implementation of international and national goals on biodiversity conservation throughout the regularly occurring Conference of the Parties (COP) (*Convention on Biological Diversity*, 1992). Until 2020, their framework in order to meet ambitious conservation goals was the 'strategic plan for biodiversity 2011-2020', including the 'Aichi-goals' (*Strategic Plan for Biodiversity 2011-2020 and the Aichi-Targets*, 2010). The next COP will take place in Kunming, China in October 2021 (URL 1). In the European Union (EU), conservation actions are determined by the 'EU biodiversity strategy for 2030', released in May 2020, as part of the Green Deal (*EU Biodiversity Strategy for 2030*, 2020). Besides political institutions, various organisations and NGOs such the World Wildlife Fund (WWF) (URL 2) or Greenpeace (URL 3) are addressing and fighting, amongst other environmental issues, the decline of biodiversity worldwide.

In many ways, floral and faunal biodiversity and its conservation are of tremendous value for humans. A vast range of advantages from economic (Guo et al., 2010) up to mental health benefits (Sandifer et al., 2015) have been reported in various publications throughout the recent years. Especially the species diversity of insects is regarded as a substantial part of human life on earth. Besides predation, dung burial and wildlife nutrition, a common example of a life-sustaining ecosystem service provided by insects is the pollination of plants (Forister et al., 2019). 70 % of the crops directly used for human consumption rely on the pollination by insects (Klein et al., 2007). The gross value of the ecosystem service of pollination was estimated to account 153 billion Euros in 2009 (Gallai et al., 2009). Insects are an essential component of the human food production, and potentially, their decline could therefore cause food shortages (Forister et al., 2019).

Although cities are a threat to species biodiversity, it is repeatedly indicated that they could potentially impact wildlife conservation. Ecologically managed, the complex mosaic of various habitat types such as built space, water, parks and private gardens could promote a variety of species (Baldock, 2020; Braschler et al., 2020). Thereby, the value of urban private gardens, being major components of urban areas, was a frequently addressed study focus throughout the recent years (Gaston et al., 2005; Goddard et al., 2010; Baldock, 2020). In six different cities throughout the UK, 21 % up to 26.8 % of city spaces were found to consist of private gardens (Gaston et al., 2005; Loram et al., 2007), and in Dunedin, New Zealand, a proportion of even 36 % was determined (Mathieu et al., 2007). This implies a coverage of 35 % up to 47 % of the total urban greenspace by private gardens (Loram et al., 2007). The market research institute 'statista' (URL 4) determined that throughout Germany, 36.07 million people (≈ 43.40 % of entire population (URL 5)) held gardens in 2020 (URL 6) and 22.12 million implemented some type of garden management about once a week (URL 7) (≈ 61.33 % of people holding gardens (URL 5)). The individual garden characteristics, including the management decisions of private gardeners, vary a lot (Gaston et al., 2005) and are mostly not underlying any public policies (Mathieu et al., 2007). These small heterogeneous arranged, dynamic greenspaces in an artificial urban environment are regarded as a potential source of various resources and habitats as well as connective elements between urban greenspaces, directly impacting wildlife conservation (Mathieu et al., 2007; Shwartz et al., 2013; Braschler et al., 2020; Baldock, 2020). Although already one single garden might consist of various resources and habitats, especially the accumulation of many gardens in residential areas within urban regions, together with public managed urban greenspaces, are regarded as major drivers of urban biodiversity (Mathieu et al., 2007). Therefore, besides public urban conservation strategies, researchers advocate the need of biodiversity-friendly gardening practices within private urban gardens to enhance the conservation value of entire urban regions (Goddard et al., 2010).

Studies among private gardens throughout the UK and Switzerland determined that many gardening practices were not biodiversity-friendly (Lindemann-Matthies & Marty, 2013; Goddard et al., 2013).

Besides intensive and neatly managed gardens due to a conservative perception of garden aesthetics and social norms (Goddard et al., 2013; Lindemann-Matthies & Marty, 2013), a clear lack of know-how on ecological garden management was prevalent (Lindemann-Matthies & Marty, 2013). Thus, in order to utilise the conservation value of private urban gardens, there is a need of appropriate information on biodiversity-friendly gardening practices including applicable management implications. So far, various single publications have emphasised the consequences of particular common gardening practices such as the planting of native and non-native plants and flowers, the use of pesticides, lawn mowing and bird feeding on species biodiversity (Goddard et al., 2010; Cameron et al., 2012). The present study aims at examining and summarising the actual impacts of these gardening practices on species biodiversity on a broader scale by conducting a global meta-analysis underlying the following research question:

1) Do common gardening practices within private (urban) gardens have an impact on species biodiversity?

In general, due to a lack of appropriate scientific communication from scientists to citizens, e.g. through public media, a misinformation of citizens on scientific topics is prevalent within the society (Scheufele & Krause, 2019). Considering gardening practices, gardeners tend to get their information primarily on the internet (Clayton, 2007). Many websites and articles informing on environmental-friendly gardening practices and advertising particularly insect and bird conservation within private gardens (URL 8; URL 9; URL 10) as well as further advantages such as aesthetic quality (URL 8) and reduced work intensity (URL 10) are available online. However, it is common knowledge that the institutions behind such websites are mostly commercially motivated. As sustainable gardening is recognised as a current trend (URL 11), magazines and blogs such as 'Mein schöner Garten' (URL 12) and 'Utopia' (URL 13) but also conservation associations such as 'NABU' (URL 14) try to meet the interests of their readers by their published articles. Thus, the commonly available information in popular scientific literature online is probably not scientifically profound (Scheufele & Krause, 2019), encouraging gardeners to engage unintentionally in non-biodiversity gardening practices. The present study will examine knowledge gaps and potential misinformation about the conservation value of the common gardening practices considered within the foregoing meta-analysis throughout the public media, addressing the following research question:

2) Are common gardening practices as advertised in public media biodiversity-friendly?

Answering these research questions, the present study will give management implications on biodiversity-friendly gardening practices that private (urban) gardeners can easily realise on their own properties. Also, proposals will be made on possible tools informing gardeners properly on such practices, simplifying their implementation and initiating a mind shift towards biodiversity-friendly garden management within the society.

3 Methods

3.1 Systematic Literature Research

The underlying systematic literature research was conducted using the database 'Web of Science' (URL 15). In order to gain experimental publications dealing with the effects of common gardening practices performed in private urban gardens on biodiversity, the following search code was developed. Making use of synonyms and related terms, each separate element aimed at one focal part of the first research question. The terms 'species richness' and 'species abundance' were included because many, especially experimental studies referred to species richness and abundance as a measure of biodiversity. The term 'socio-economic' was excluded due to the high number of publications in social science focussing on human-nature interactions.

((urban garden*) AND (private OR non-public OR domestic OR home OR residential) AND (practice* OR management OR habit* OR implementation OR custom* OR characteristic* OR resource* provision) AND (biodiversity OR species richness OR species abundance) AND (experiment* OR effect*)) NOT socio-economic

A total of 88 results in the Web of Science core collection were obtained. Out of these, 25 publications were excluded throughout the screening of the titles because they did not refer to the first research question. An additional 27 additional publications were excluded throughout the screening of the abstracts. Exclusion of publications was frequently due to a focus on social science, biological corridors, soil parameters or cat predation on birds. The majority of the remaining 36 publications determined the effects of vegetation on insects and the effects of supplementary feeding on birds.

Aiming to identify the majority or even all experimental studies related to the first research question, the reference lists of the 36 remaining publications were examined. As a result, 123 additional publications were obtained. Another 19 publications were obtained during literature research without using the standardised search term.

3.1.1 Publications

A variety of gardening practices and their effects on species and species groups were identified in the subsequent 178 publications. Gardening practices were clustered in twelve groups (hereafter: codes) and target species were coded in five different taxonomic groups (hereafter: taxon codes), where both related gardening practices and taxonomic groups were divided into subgroups if the number of publications and related effect sizes of conservation measures allowed for detailed analyses. Within the publications, different measures of biodiversity were applied in order to quantify the impact of gardening practices on the species biodiversity of certain taxa, which were clustered into abundance (a), richness (r), diversity (d), breeding measures (bm) and mortality (m). Thereby, all biodiversity measures referred to the biodiversity on a population or community level. The studies were conducted at various study sites, which could be clustered into 1) urban gardens (ug), 2) public urban greenspaces (pug) and 3) rural greenspaces (rg). (Tab. 1; Appendix Tab. 1)

Tab. 1: Codes of gardening practices and taxa examined within the scientific literature on the effects of gardening practices on the biodiversity of certain taxonomic groups. A1 - A6 refer to practices related to planting. 3a – 3f refer to subgroups of arthropods. 3abcd refers to pollinators, consisting of species primarily belonging to the taxonomic groups hymenopterans (3a), butterflies (3b), bugs (3c) and true bugs (3d).

Code	Practice
A1	(Non-)native plant planting
A2	Flower planting
A3	Native tree planting
A4	Tree planting
A5	Shrub and forb planting
A6	Bed creation
B	Fertiliser use
C	Pesticide use
D	Mowing
E	Dead wood provision
F	Irrigation
G	Cleaning
H	Artificial nest site provision
I	Bird feeding
J	Water provision
K	Water body creation
L	Sealing

Code	Taxon
1	Mammalia (mammals)
2	Aves (birds)
3	General Arthropoda (arthropods)
3a	Hymenoptera (hymenopterans)
3b	Lepidoptera (butterflies)
3c	Coleoptera (bugs)
3d	Hemiptera (true bugs)
3abcd	pollinators
3e	General Insecta (insects)
3f	Arachnida (arachnids)
4	Amphibia (amphibians)
5	Plantae (plants)

3.1.2 Explanations and Definitions

<u>Gardening Practices</u>

The publications referring to the planting of certain vegetation types as a gardening practice were assigned to the subgroups of (non-)native plant planting (A1), flower planting (A2), native tree planting (A3), tree planting (A4), shrub and forb planting (A5) and bed creation (A6) because separate analyses were possible due to the individual study designs (Tab. 1). It is important to note that publications dealing with native plant planting (A1) mainly examined the effects of native plant planting in comparison to non-native plant planting, whereas publications dealing with flower planting (A2) regarded the correlations between the presence, abundances or richness of flowers to insects or compared the effects of floral presence, abundances or richness on insects to a control group comprised of few or zero flowers. As the publications dealing with (non-)native plants primarily referred to (non-)native flowers, a massive overlap between the subgroups was prevalent, and one publication was both included into the subgroups of (non-)native plant planting and flower planting (Matteson & Langelotto, 2011), but the differences in study focusses allowed individual analyses. Also, due to the data provided within the publications considered within the meta-analysis, pesticide use (C) was examined regarding pesticide use in general as well as insecticide, herbicide, fungicide and snail pellet use. The effects of lawn mowing (D) were measured by comparing sample groups having obtained different mowing frequencies and mowing heights. One publication already considered within the meta-analysis on native plant planting additionally dealt with the effects of lawn mowing and was therefore included into both gardening practice groups (Smith et al., 2015). Bird feeding practices (I) were divided into winter- and spring-bird-feeding because clear differences in the effects of food supplementation according to the time of the year were prevalent. Depending on the study location, winter referred to the months of November or December until March (Robb et al., 2008b; Plummer et al., 2013) and October until April (Brittingham & Temple, 1988), while spring referred to the months of March until July (Harrison et al., 2010) and May until July (Fuller et al., 2008). Effects both on the current bird generation and their breeding measures, thus the following bird generation, were determined. All scientific names of taxonomic plant groups or individual species named in the present study were either derived from the original publications or the database 'FloraWeb' (URL 16).

Taxonomic Groups

In the present study, the taxonomic group of general insects (3e) includes all publications dealing with insects, hymenopterans (3a), butterflies (3b), true bugs (3d) and pollinators (3abcd). Pollinators (3abcd) mainly consisted of the subgroups hymenopterans (3a), butterflies (3b), bugs (3c) and true bugs (3d), but only publications purposely examining pollinators were included into this group. Within the group of hymenopterans (3a), publications referred to the subgroups of bees (Apidae) and wasps (Vespidae) (Tab. 1). According to the taxonomic groups examined within the publications considered within the present meta-analysis, the groups of insects (3e) and hymenopterans (3a) were mostly comprised of different taxa for each management practice and weighed mean effect size (*M*) calculation (Appendix Tab. 2). Regarding the impact of bird feeding (I) on birds (2), most of the determined weighed mean effect sizes (*M*) are directly referring to the family of tits (Paridae). All scientific names of taxonomic groups or individual species used in the present study were either derived from the original publications or the database 'ITIS' (URL 17).

Biodiversity Measures

Biodiversity is defined by the United Nations (UN) as ' the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems' (*Convention on Biological Diversity*, 1992). The complexity of biodiversity and its definition can cause difficulties in the formulation of conservation strategies (Swingland, 2013). Therefore, in the present study, biodiversity refers solely to the diversity within and among species. Measures of species diversity were all based on methods that yielded concrete numbers, e.g. number of species. However, it must be kept in mind that these measures only represent small components of the highly complex term biodiversity and should be regarded as an approach to make biodiversity tangible (Swingland, 2013).

Species abundance and richness are common measures of species biodiversity. Species abundance expresses the number of individuals per species (Baumgärtner, 2006). Species richness describes 'the total number […] of different species found' (Baumgärtner, 2006). Also, abundance and richness can refer to other taxonomic levels beyond species such as genera or families (Swingland, 2013). Thus, the term abundance refers to the diversity within one taxonomic group and the term richness to the diversity among different taxonomic groups.

Mortality is not commonly used as a biodiversity measure. However, throughout the present study, the term refers to the parasitism rates on hymenopterans correlated to the planting of flowers and the survival rate of birds in the context of artificial bird feeding. Therefore, mortality rates and related effect sizes were used as negative measures of biodiversity.

Regarding the effects of the gardening practices of flower planting (A2) on insects and bird feeding (I) on birds within the present study, breeding measures were used as indicators. Considering flower planting, the applied measure was brood cell density, thus the number of brood cells of bees and wasps per study plot (Ebeling et al., 2012). Considering bird feeding, applied terms were nest box occupancy, lay date, clutch size, incubation period, brood size and fledging success. Nest box occupancy was expressed as the percentage of nest boxes that were occupied by birds out of four next boxes that were offered per hectare (Plummer et al., 2013). Lay date refers to the date when adult birds initiate egg laying and clutch size to the number of eggs laid (Dijkstra et al., 1990). According to Harrison et al. (2010), the incubation period is defined as the 'number of days between clutch completion date (day 0) and hatching day of the first egg', and the brood size is calculated as 'clutch size minus any unhatched eggs (i.e. the maximum possible brood size)'. Subsequently, fledging success is defined as 'the proportion of hatchlings that fledged' (Plummer et al., 2013).

3.2 Meta-Analysis

3.2.1 Publications

After first sightings and coding of the available literature, 157 publications were excluded from the calculation of effect sizes. Major exclusion criteria were a lack of useable data for meta-analyses and an inexpedient study focus. Due to a lack of appropriate publications, initially planned analyses of the gardening practices related to trees (A3 & A4), shrubs and forbs (A5), beds (A6), fertilisers (B), dead wood (E), irrigation (F), cleaning (G), artificial nest sites (H), water provision (J), water bodies (K) and sealing (L) and the taxonomic groups of mammals (1), bugs (3c), arachnids (3f), amphibians (4) and plants (5) were not considered within the meta-analysis. Thus, the final meta-analysis focussed on the effects of (non-)native plant planting (A1), flower planting (A2), pesticide use (C), mowing (D) and bird feeding (I) on birds (2), insects in general (3e), pollinators (3abcd), hymenopterans (3a), butterflies (3b) and true bugs (3d) on the basis of 21 publications (Fig. 1; Fig. 2; Fig. 3; Appendix Tab. 1).

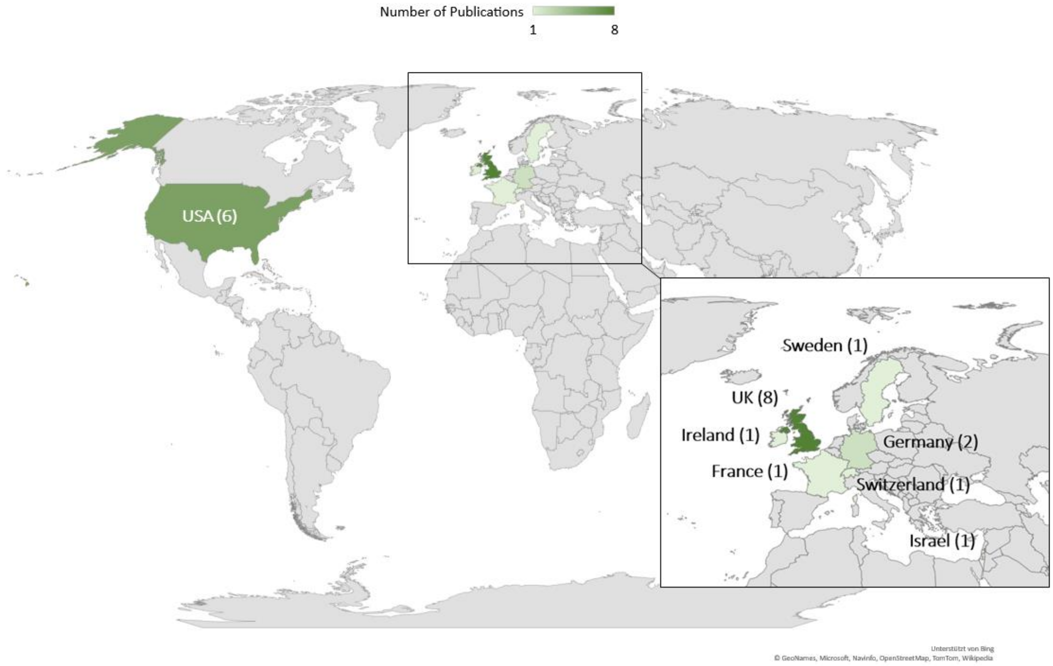


Fig. 1: Origins (numbers) of publications considered within the meta-analysis.

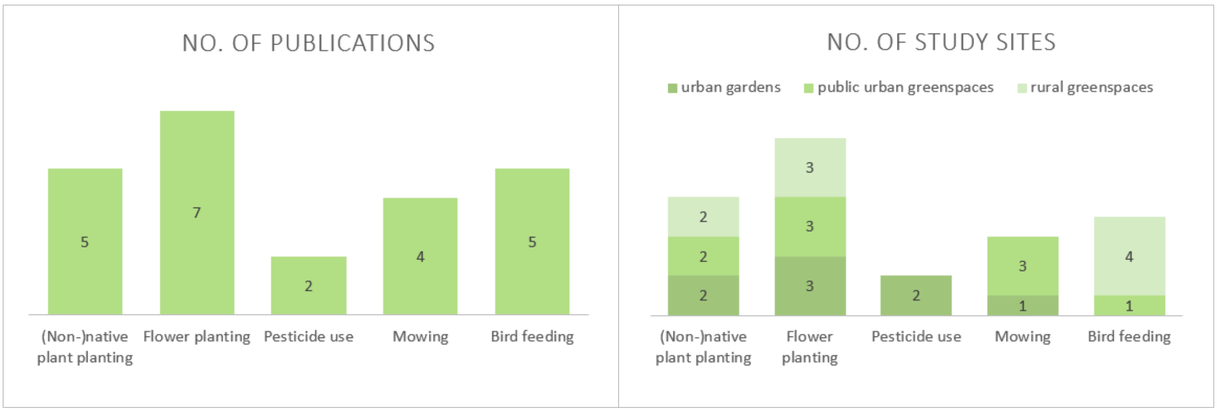


Fig. 2: No. of publications and study sites according to each gardening practice ((non-)native plant planting, flower planting, pesticide use, mowing, bird feeding) considered within the meta-analysis.

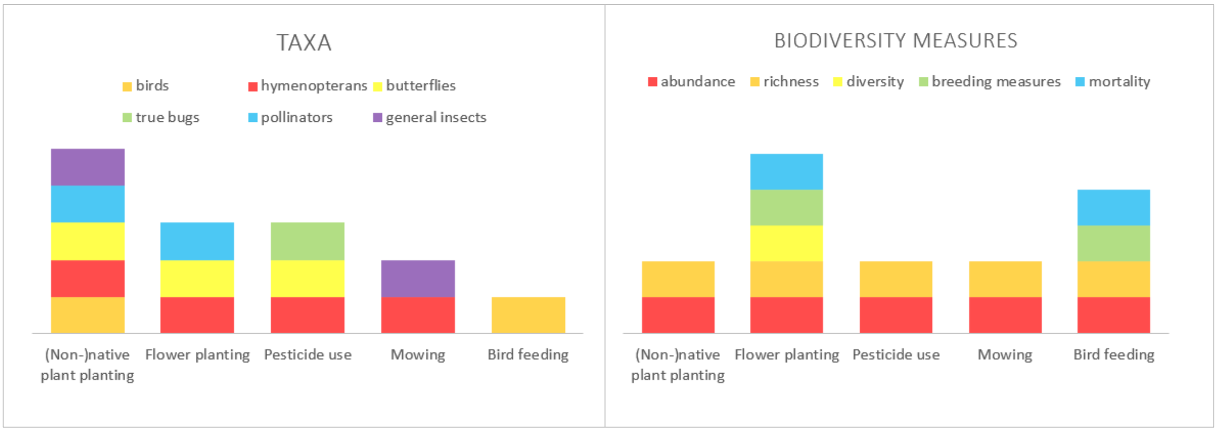


Fig. 3: Taxa and biodiversity measures examined in the publications considered within the meta-analysis according to each gardening practice ((non-)native plant planting, flower planting, pesticide use, mowing, bird feeding).

The publications considered within the meta-analysis were based on different experimental study designs and therefore differed in their quality and significance. In order to consider these differences, the publications obtained study weights (*w*) according to Norris et al. (2012), regarding study designs and the number of sample units, i.e. urban gardens, public urban greenspaces or rural greenspaces, and replications (Tab. 2; Tab. 3). Only after (A)-, control vs. impact (CI)- and gradient-response (G)-study designs were prevalent in the publications considered within the meta-analysis (Fig. 4). Thereby, study qualities, measured as study weights (*w*), ranged from low ($w \leq 2$) to high ($w \geq 6$), and the majority of publications were of medium quality ($w = 3 - 5$) (Fig. 5).

Tab. 2: Ranking of study designs according to Norris et al. (2012).

Study design	Code	Weight (*w*)
After impact only	A	1
Control vs. impact	CI	2
Before vs. after	BA	2
Gradient-response	G	3
Before vs. after & control vs. impact	BACI	4
Before vs. after & reference vs. impact	BARI	4

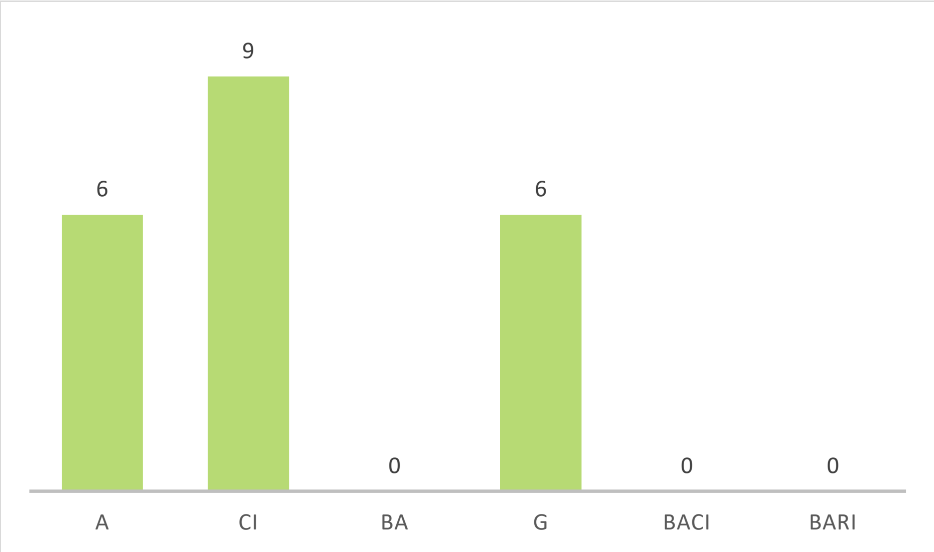


Fig. 4: Study designs of publications considered within the meta-analysis.

Tab. 3: Ranking of number of replications in study designs according to Norris et al. (2012). In factorial study designs (1), replication of sampling units, either designated as reference / control or as impact / treatment units, are considered. In gradient-response-models (2), total replication is considered.

1 Replication of factorial designs	
1.1 Number of reference / control sampling units	Weight (*w*)
0	0
1	2
> 1	3
1.2 Number of impact / treatment sampling units	
1	0
2	2
> 2	3
2 Replication of gradient-response-models	
< 4	0
4	2
5	4
> 5	6

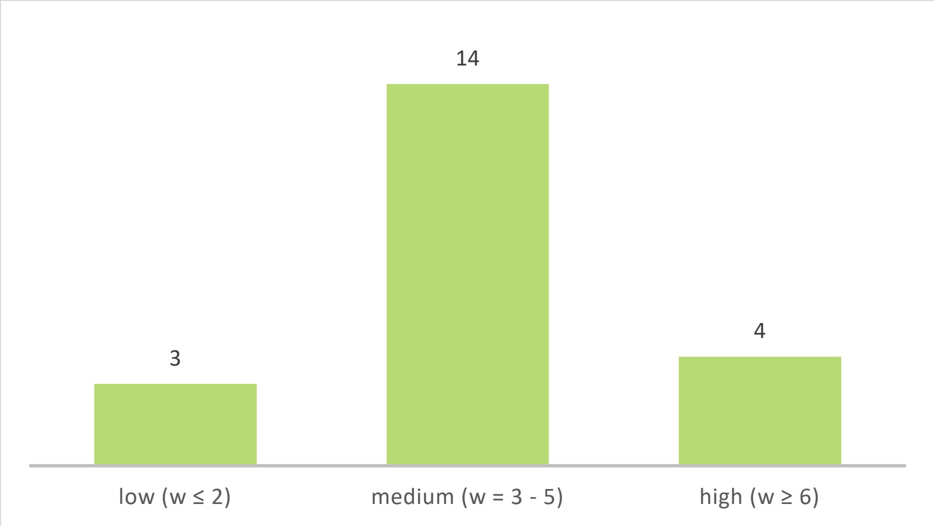


Fig. 5: Study quality of publications considered within the meta-analysis. Study quality was measured as study weights (w) calculated according to Norris et al. (2012).

3.2.2 Effect Sizes

In order to quantify, compare and summarise the effects of common gardening practices on biodiversity, the standardised mean difference between treatment groups (i.e. effect sizes of urban gardening practices on measures of biodiversity) was calculated as Cohen's *d*. Cohen's *d* is obtained by dividing the difference between the means $\bar{x}_1$ and $\bar{x}_2$ of two independent sample groups 1 and 2 by the pooled standard deviation S_{pooled} (Formula 1 adapted according to Borenstein et al. (2010)):

Formula 1

$$d = \frac{\bar{x}_2 - \bar{x}_1}{S_{pooled}}$$

The pooled standard deviation S_{pooled} is an estimate of the common standard deviations S_1 and S_2 of the two independent groups, which are likely to differ in their standard deviations (Formula 2 adapted according to Borenstein et al. (2010)):

Formula 2

$$S_{pooled} = \sqrt{\frac{(n_1 - 1)S_1^2 + (n_2 - 1)S_2^2}{(n_1 + n_2 - 2)}}$$

,where n_1: sample size group 1 and n_2: sample size group 2.

Thus, in order to calculate the standardised mean difference as Cohen's *d*, the means $\bar{x}$, standard deviations *S* or standard errors *SE* (as $S = SE * \sqrt{n}$) and sample sizes *n* are needed for sample groups 1 and 2. Publications presented their data either in terms of text, tables or figures within the main publication or in the appendix. In case of data illustrated in figures, the data were extracted by making use of the online tool 'WebPlotDigitizer' (URL 18).

In many publications, not the means and standard deviations of sample groups were given but the results of further statistical tests. Under certain circumstances, the conversion of these measures to Cohen's *d* was possible following the formulas summarised in Tab. 4.

Tab. 4: Conversion formulas from statistical measures to Cohen's *d*. Pearson's coefficient of determination r^2 and Spearman's rank correlation coefficient *p* are primarily converted to Pearson's correlation coefficient *r*, which can be converted to Cohen's *d*.

Statistical test (measure) *Requirements*	Conversion formulas	Source
Pearson's (correlation coefficient *r*) Pearson's (coefficient of determination r^2)	$d = \frac{2r}{\sqrt{-1-r^2}}$ $r = \sqrt{r^2}$	Borenstein et al. (2010)
Spearman's (rank correlation coefficient *p*)	$r = 2\,sin * \left(\rho \frac{\pi}{6}\right)$	Ivarsson et al. (2013)
t-Tests (*t*) *1) two groups with equal sample sizes n* *2) two groups with unequal sample sizes n_1 and n_2*	$d \approx 2 * \frac{t}{\sqrt{n}}$ $d = t * \sqrt{\frac{1}{n_1} + \frac{1}{n_2}}$	Polanin and Snilstveit (2016)
Chi-square-test (x^2) *1 df**	$r = \sqrt{\frac{\chi^2}{n}}$	Rosenberg (2010)
ANOVA (*F*) *1) two groups with equal sample size n (1 df*)* *2) two groups with unequal sample sizes n_1 and n_2 (1df*)*	$d = 2 * \sqrt{\frac{F}{n}}$ $d = \sqrt{\frac{\mathrm{F} * (n_1 + n_2)}{n_1 * n_2}}$	Polanin and Snilstveit (2016)

* df = degree(s) of freedom

Cohen's *d* is known to overestimate the magnitude of the effect at small sample sizes. Therefore, Hedge's *g* as a correction for small sample sizes was applied (Borenstein et al., 2010). Hedge's *g* is obtained by multiplying Cohen's *d* with the correction factor *J* (Formulas 3 adapted to Borenstein et al. (2010)):

Formulas 3

$g = d * J$

,where $J = 1 - \frac{3}{4 * (n_1 + n_2 - 2) - 1}$

To allow coherent comparisons and interpretations of the data, signs of effect sizes were modified so that positive effects appeared positive and negative effects appeared negative. The effects sizes concerning the effects of flowers on bees and wasps measured in terms of parasitism rate (Ebeling et al., 2012) were changed in order to be negative, as parasitism diminishes biodiversity. Vice versa, the effect sizes concerning the effects of supplemental feeding on birds measured in terms of lay dates (Harrison et al., 2010; Plummer et al., 2013) were changed in order to be positive because advanced lay dates promote bird biodiversity (Robb et al., 2008a).

In order to include the quality of the studies into the meta-analysis, each effect size *g* was multiplied with the weight (*w*) of the original publication as summarised in Appendix Tab. 1. For each gardening practice, weighed mean effect sizes *M* were calculated by dividing the sum of the weighed effect sizes (*w* * *g*) by the sum of the weights (*w*) (Formula 4 adapted according to Borenstein et al. (2010)). In addition, the variances V_M and standard errors SE_M were determined in order to subsequently calculate the lower limits LL_M and upper limits UL_M of the 95 % confidence intervals (CI) (Formulas 5 according to Borenstein et al. (2010)). Weighed mean effect sizes (*M*) and lower and upper limits (LL_M & UL_M) of the CI for each gardening practice are presented in Appendix Tab. 2.

Formula 4

$$M = \frac{\sum_{i=1}^{k} w_i * g_i}{\sum_{i=1}^{k} w_i}$$

Formulas 5

$LL_M = M - 1.96^* SE_M$

$UL_M = M + 1.96^* SE_M$

,where $SE_M = \sqrt{V_M}$ and $V_M = \frac{1}{\sum_{i=1}^{k} w_i}$

A weighed mean effect size (*M*) of zero indicates no effect of the target garden practice on biodiversity. Accordingly, larger effect sizes indicate larger effects, where a positive weighed mean effect size (*M*) indicates positive impacts of the gardening practice on biodiversity measures of certain taxa and, thus, negative weighed mean effect sizes (*M*) indicate negative effects. The effects can be considered statistically significant if the lower (LL_M) or upper limits (UL_M) of the 95 % CI are both below or above zero.

3.3 Popular Scientific Literature

In order to examine knowledge gaps and potential misinformation about the conservation value of the common gardening practices considered within the meta-analysis throughout the public media in Germany, an unsystematic online research using the search engine 'Google' (URL 19) was conducted. Thereby, three websites were prevalent among the search results:

1) 'Mein schöner Garten' (URL 12)

'Mein schöner Garten' is an online and print gardening magazine that is claimed to be the most popular gardening magazine throughout Europe with ca. 6 million unique users monthly online and ca. 1.9 million readers per print issue (*Mein Schöner Garten*, 2021).

2) 'Utopia' (URL 13)

'Utopia' is described as Germany's leading website on information and inspiration related to a sustainable lifestyle with ca. 7.8 million unique users per month (*Deutschlands Website Nr. 1 Für Nachhaltigen Konsum*, 2020).

3) 'NABU' (URL 14)

The 'NABU' ('Naturschutzbund Deutschland e. V.') is the oldest and most popular conservation association in Germany comprised of ca. 820,000 members and promoters (URL 20).

Subsequently, articles on ecological gardening practices in general as well as on planting patterns, pesticide use, lawn mowing and bird feeding were identified in order to be analysed within the present study.

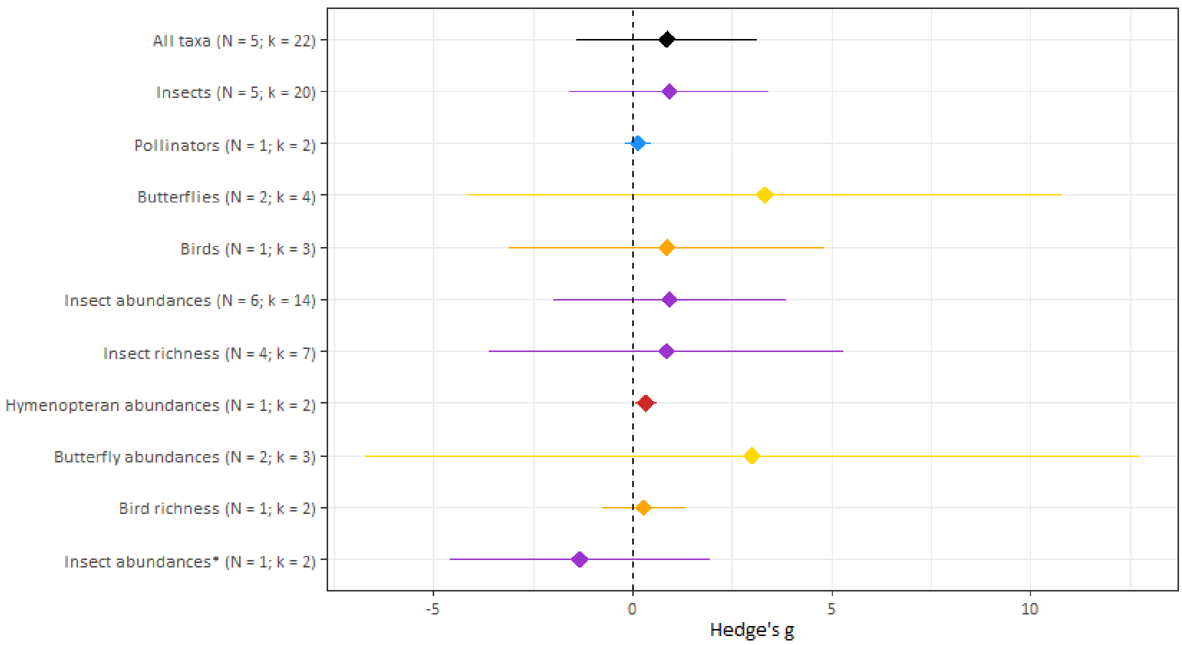
4 Results

4.1 Meta-Analysis

4.1.1 Planting

4.1.1.1 Native and Non-Native Plants

The overall mean effects of native plant planting on all examined taxa were non-significantly positive (*M* = 0.86 [95 % CI: -1.42; 3.15]), which also applied to the individual taxonomic groups of insects (*M* = 0.92 [95 % CI: -1.58; 3.42]), pollinators (*M* = 0.14 [95 % CI: -0.19; 0.46]), butterflies (*M* = 3.33 [95 % CI: -4.11; 10.77]) and birds (*M* = 0.86 [95 % CI: -3.11; 4.82]). Likewise, non-significant positive mean effects were determined on insect abundances and richness (abundances: *M* = 0.93 [95 % CI: -2.00; 3.58]; richness: *M* = 0.85 [95 % CI: -3.61; 5.31]), butterfly abundances (*M* = 3.01 [95 % CI: -6.71; 12.74]) and bird richness (*M* = 0.29 [95 % CI: -0.77; 1.34]). The effects on hymenopteran abundances were significantly positive (*M* = 0.34 [95 % CI: 0.07; 0.62]). Also, no significant but negative mean effects of non-native plant planting on insect abundances were ascertained (*M* = -1.32 [95 % CI: -4.57; 1.93]). (Fig. 6)

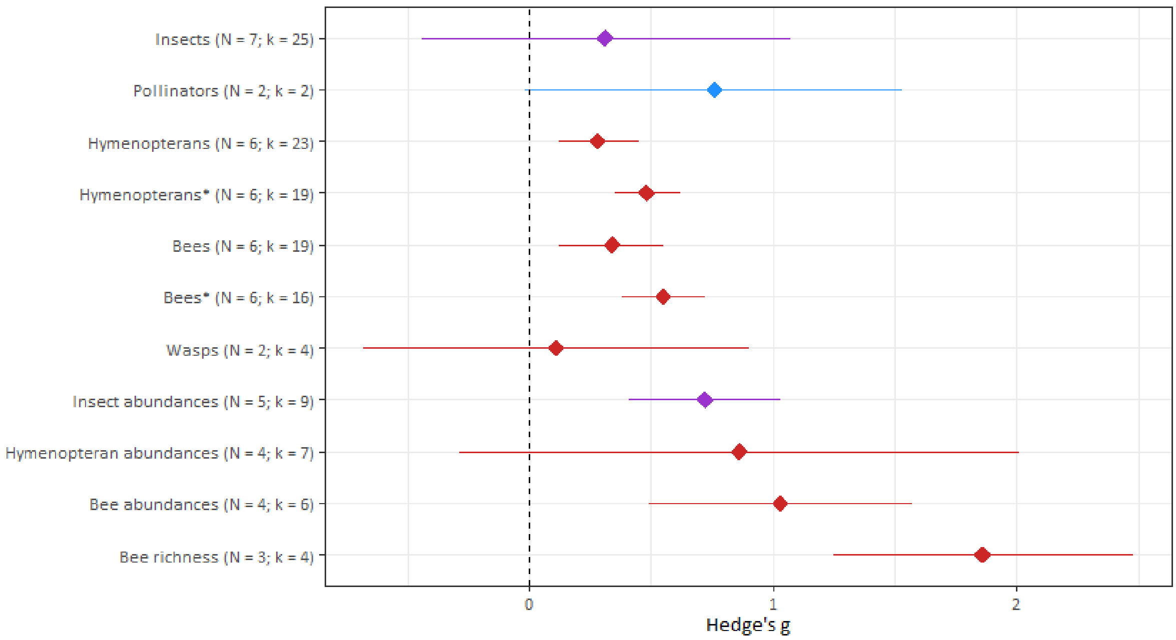


* Hedge's *g* and 95 % CI refer to the effects of non-native plant planting on insect abundances.

Fig. 6: Effects of native and non-native plant planting on insects (purple), pollinators (blue), butterflies (yellow) and birds (orange). Shown are weighed mean effect sizes (*M)* (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

4.1.1.2 Flowers

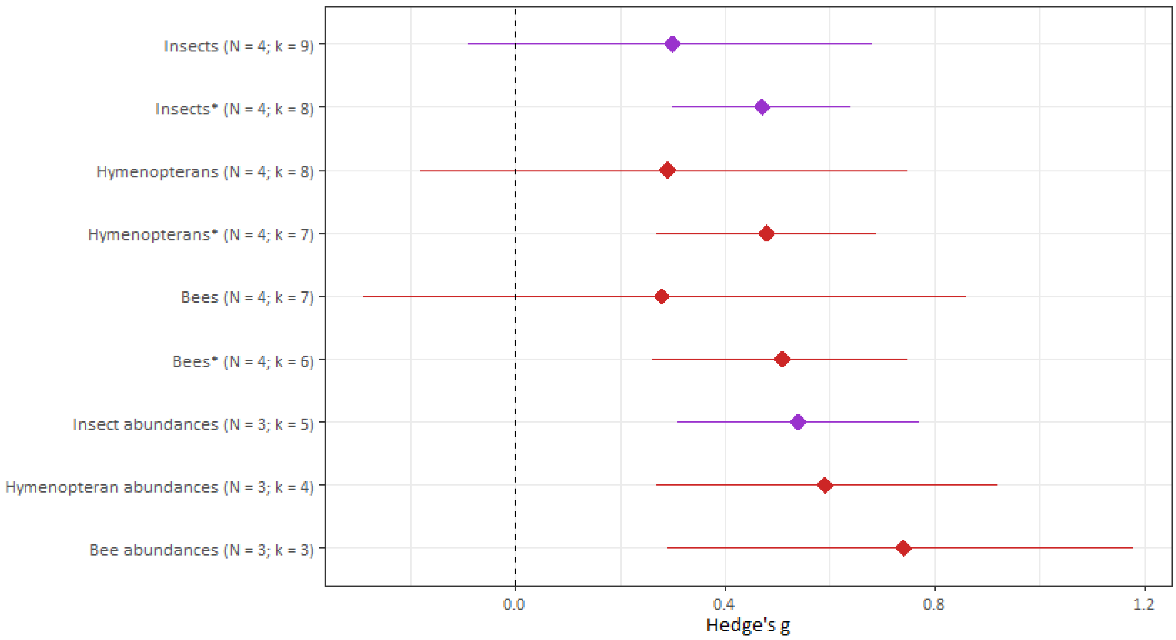
Overall, the presence of flowers showed non-significant positive mean effects on insects (*M* = 0.31 [95 % CI: -0.44; 1.07]), pollinators (*M* = 0.76 [95 % CI: -0.02; 1.53]) and wasps (*M* = 0.11 [95 % CI: -0.68; 0.90]) as well as significant positive effects on hymenopterans (*M* = 0.28 [95 % CI: 0.12; 0.45]) and bees (Apidae) (*M* = 0.34 [95 % CI: 0.12; 0.55]). Also, insect abundances were significantly positively affected (*M* = 0.72 [95 % CI: 0.41; 1.03]). The mean effects on hymenopteran abundances were positive but non-significant (*M* = 0.86 [95 % CI: -0.29; 2.01]). Particularly high significant positive effects were determined on bee abundances (*M* = 1.03 [95 % CI: 0.49; 1.57]) and richness (*M* = 1.86 [95 % CI: 1.25; 2.48]). (Fig. 7)



* Hedge's *g* and 95 % CI are calculated excluding effect sizes referring to parasitism rates on bees and wasps.

Fig. 7: Effects of flower planting on insects (purple), pollinators (blue) and hymenopterans (red). Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

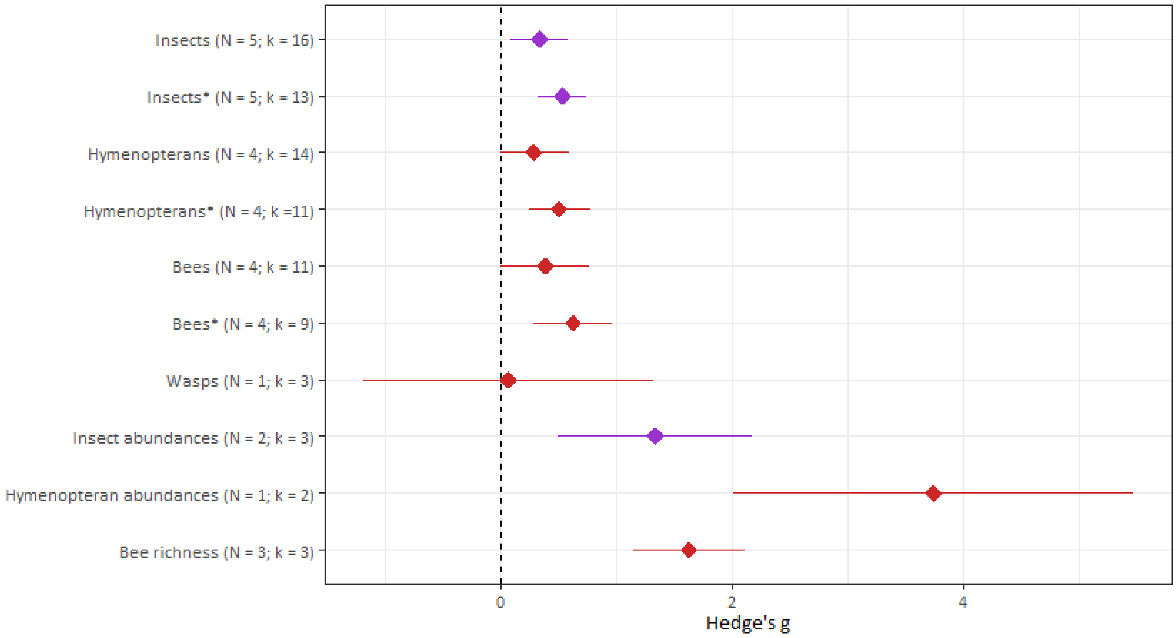
Floral abundances showed non-significant mean effects on insects (*M* = 0.30 [95 % CI: -0.09; 0.68]), hymenopterans (*M* = 0.29 [95 % CI: -0.18; 0.75]) and bees (*M* = 0.28 [95 % CI: -0.29; 0.86]). Significant positive effects of floral abundances on the abundances of insects (*M* = 0.54 [95 % CI: 0.31; 0.77]), hymenopterans (*M* = 0.59 [95 % CI: 0.27; 0.29]) and bees (*M* = 0.74 [95 % CI: 0.29; 1.18]) were ascertained. (Fig. 8)



* Hedge's *g* and 95 % CI are calculated excluding effect sizes referring to parasitism rates on bees and wasps.

Fig. 8: Effects of floral abundances on insects (purple) and hymenopterans (red). Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

Floral richness had significant positive effects on insects (*M* = 0.34 [95 % CI: 0.09; 0.58]), hymenopterans (*M* = 0.29 [95 % CI: 0.00; 0.59]) and bees (M = 0.39 [95 % CI: 0.00; 0.77]). Particularly high significant effects were found on insect abundance (*M* =1.34 [95 % CI: 0.50; 2.17]), hymenopteran abundances (*M* = 3.74 [95 % CI: 2.01; 5.47]) and bee richness (*M* =1.63 [95 % CI: 1.15; 2.11]). The mean effect of floral richness on wasps (*M* = 0.07 [95 % CI: -1.18; 1.32]) was positive, but non-significant. (Fig. 9)



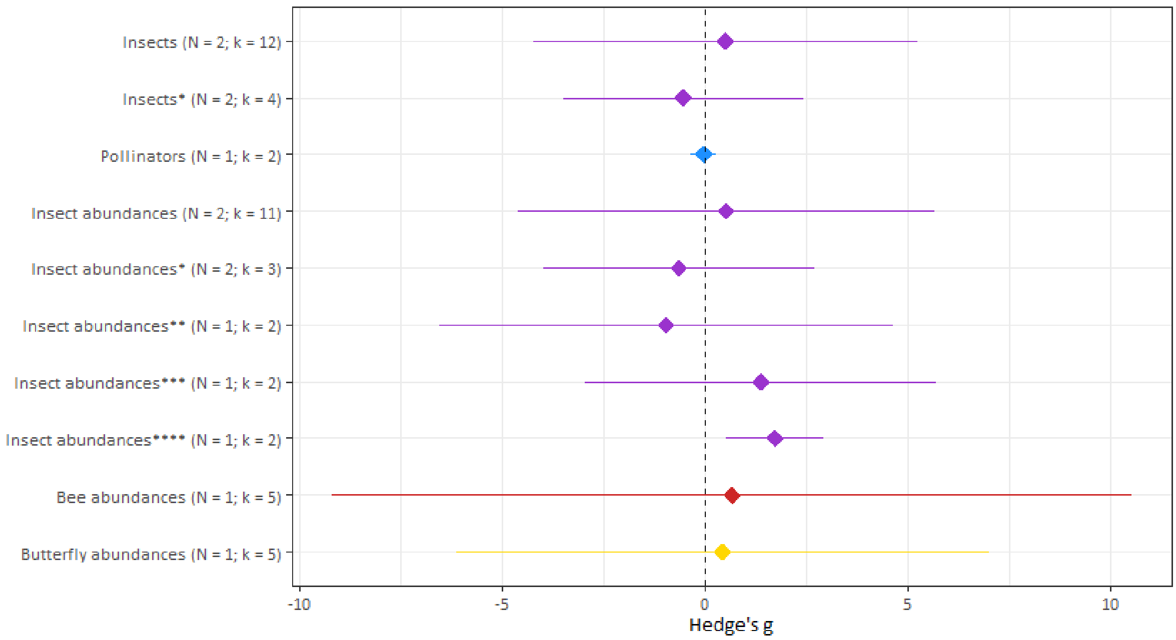
* Hedge's *g* and 95 % CI are calculated excluding effect sizes referring to parasitism rates on bees and wasps.

Fig. 9: Effects of floral richness on insects (purple) and hymenopterans (red). Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

The parasitism rates on bees and wasps were the only negative effects of flower planting that were included in the weighed mean effect sizes. An exclusion of these numbers altered the overall results of flower presence, abundances and richness positively. The significant positive effects of flower presence on hymenopterans and bees became stronger (hymenopterans: *M* = 0.48 [95 % CI: 0.35; 0.62]; bees: *M* = 0.55 [95 % CI: 0.38; 0.72]), similar to floral richness on insects (*M* = 0.54 [95 % CI: 0.33; 0.75]), hymenopterans (*M* = 0.51 [95 % CI: 0.25; 0.78]) and bees (*M* = 0.63 [95 % CI: 0.29; 0.97]). Furthermore, the effects of floral abundances on insects, hymenopterans and bees were changed in order to be significantly positive after parasitism rates were excluded (insects: *M* = 0.47 [95 % CI: 0.30; 0.64]; hymenopterans: *M* = 0.48 [95 % CI: 0.27; 0.69]; bees: *M* = 0.51 [95 % CI: 0.26; 0.75]). (Fig. 7; Fig. 8; Fig. 9)

4.1.2 Pesticide Use

Most mean effects of pesticide use on biodiversity were statistically non-significant. Nonetheless, pesticides in general affected insects positively (*M* = 0.51 [95 % CI: -4.21; 5.24]). In contrast, the overall mean effects on pollinators only were slightly negative (*M* = -0.02 [95 % CI: -0.34; 0.29]). Regarding the abundances of insects, bees and butterflies, positive mean effects were determined (insects: *M* = 0.53 [95 % CI: -4.60; 5.67]; bees: *M* = 0.68 [95 % CI: -9.19; 10.55]; butterflies: *M* = 0.44 [95 % CI: -6.14; 7.01]). Taking insecticides into account, the overall mean effects on insects and insect abundances were negative (insects: *M* = -0.53 [95 % CI: -3.50; 2.44]; insect abundances: *M* = -0.63 [95 % CI: -3.97; 2.71]). Also, herbicides led to negative mean effects on insect abundances (*M* = -0.96 [95 % CI: -6.56; 4.65]). On the contrary, fungicides and snail pellets impacted positively insect abundances, whereby the effects of snail pellets were statistically significant (fungicides: *M* = 1.39 [95 % CI: -2.94; 5.72]; snail pellets: *M* = 1.73 [95 % CI: 0.52; 2.94]). (Fig. 10)

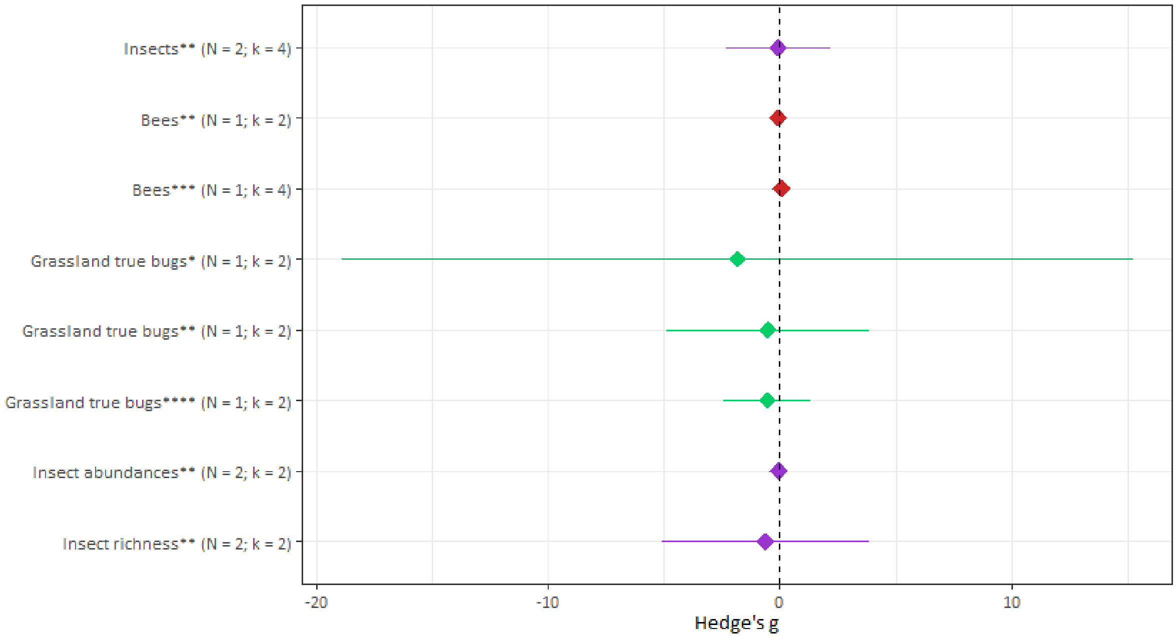


* Insecticide use ** Herbicide use *** Fungicide use **** Snail pellet use

Fig. 10: Effects of pesticide use on insects (purple), pollinators (blue), hymenopterans (red) and butterflies (yellow) and effects of insecticide, herbicide, fungicide and snail pellet use on insects. Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

4.1.3 Mowing

No significant mean effects of mowing frequencies on biodiversity measures were determined. Nonetheless, weekly mowing in comparison to a mowing frequency of every two weeks revealed slightly negative mean effects on insects (*M* = -0.06 [95 % CI: -2.30; 2.18]), bees (*M* = -0.06 [95 % CI: -0.14; 0.03]) and grassland true bugs (*M* = -0.51 [95 % CI: -4.89; 3.87]). This also applied to insect abundances (*M* = -0.05 [95 % CI: -0.24; 0.14]) and richness (*M* = -0.62 [95 % CI: -5.10; 3.86]). Compared to zero mowing and every-six-weeks mowing, grassland true bugs were negatively affected by weekly mowing (comparison to zero mowing: *M* = -1.84; [95 % CI: -18.92; 15.24]; comparison to every-six-weeks mowing: *M* = -0.55 [95 % CI: -2.41; 1.31]). Compared to a mowing frequency of every three weeks, the mean effects of weekly mowing on bees were slightly positive (*M* = 0.07 [95 % CI: -0.23; 0.37]). (Fig. 11)



* weekly vs. zero mowing ** weekly vs. every-two-week mowing *** weekly vs. every-three-weeks mowing **** weekly vs. every-six-weeks mowing

Fig. 11: Effects of weekly mowing vs. zero / every-two-week / every-three-week / every-six-week mowing on insects (purple), hymenopterans (red) and grassland true bugs (green). Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

A mowing height of two cm versus four cm revealed non-significant negative mean effects on insects (M = -0.38 [95 % CI: -1.23; 0.48]), insect richness (M = -0.71 [95 % CI: -1.67; 0.25]) and, slightly significant effects on insect abundances (M = -0.04 [95 % CI: -0.09; 0.00]). (Fig. 12)

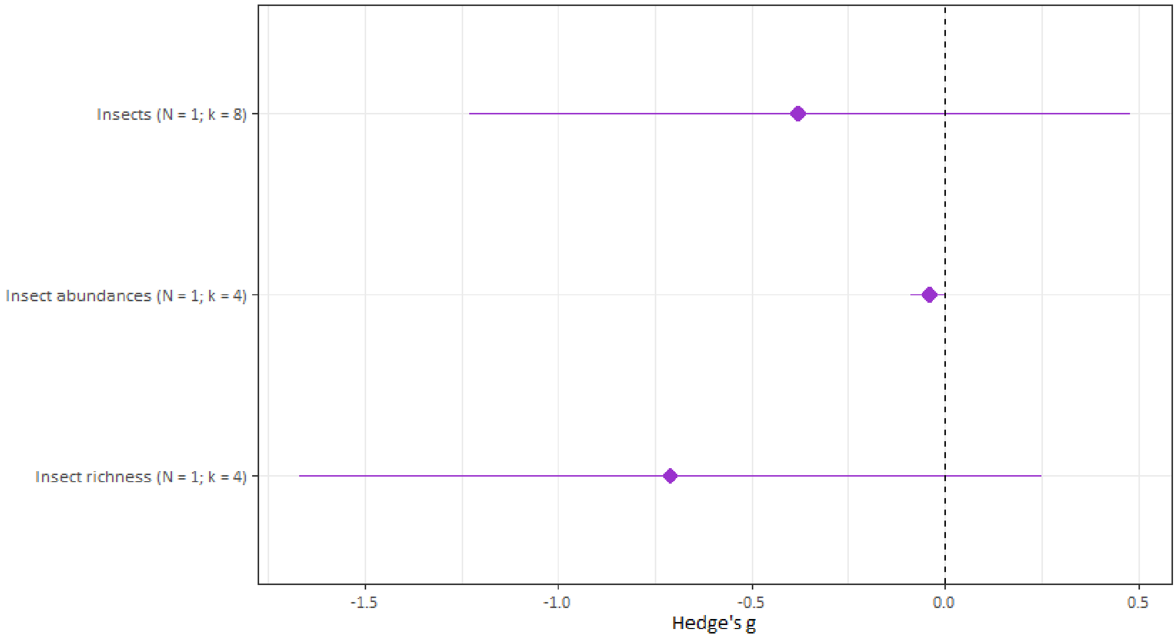
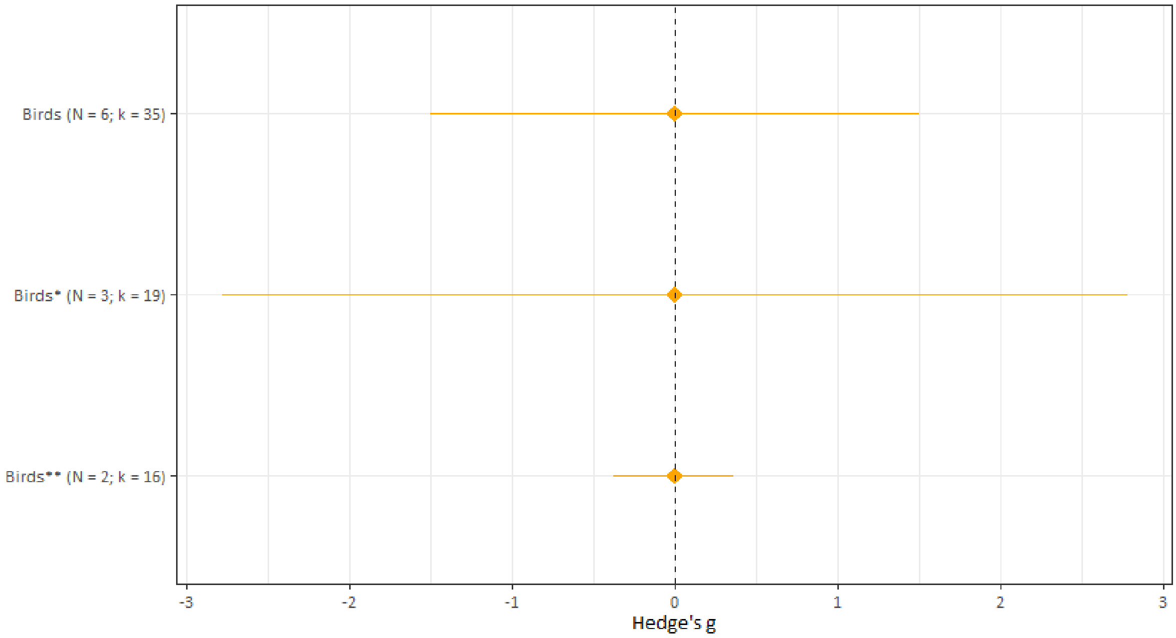


Fig. 12: Effects of mowing heights on insects. Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

4.1.4 Bird Feeding

Most mean effects of winter- and spring-feeding on birds were non-significant. The overall mean effects of bird feeding in general as well as of winter- and spring-feeding amounted zero (general: *M* = 0.00 [95 % CI: -1.50; 1.50]; winter-feeding: *M* = 0.00 [95 % CI: -2.78; 2.78]; spring-feeding: *M* = 0.00 [95 % CI: -0.37; 0.36]). (Fig. 13)



* winter-bird-feeding **spring-bird-feeding

Fig. 13: Effects of bird feeding in general and winter- and spring-feeding on birds. Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

Bird abundances and winter survival rates of black-capped chickadees (*Poecile atricapillus)* were positively affected by food supplementation (abundances: *M* = 0.30 [95 % CI: -1.31; 1.92]; winter survival rates: *M* = 1.28 [95 % CI: -3.29; 5.84]). (Fig. 14)

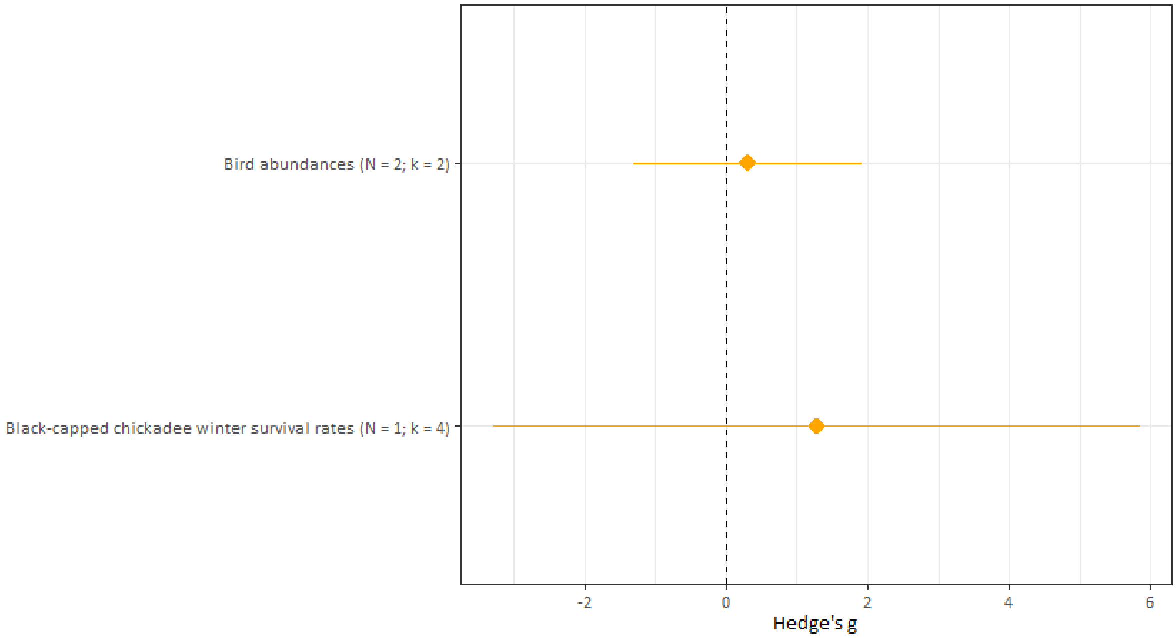


Fig. 14: Effects of bird feeding on bird abundances and winter survival rates. Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

The overall mean effects of winter- and spring-feeding on the breeding measures of tits were negative (winter-feeding: *M* = -0.31 [95 % CI: -3.53; 2.91]; spring-feeding: *M*= -0.05 [95 % CI: -0.28; 0.19]). Taking only spring-feeding into account, the mean effects were positive for blue tits (*Cyanistes caeruleus*) (*M* = 0.98 [95 % CI: -3.74; 5.71]) but slightly negative for great tits (*Parus major*) (*M* = -0.07 [95 % CI: -0.51; 0.37]). The mean effects on tit's lay dates of both winter- and spring-feeding accounted almost zero (winter-feeding: *M* = 0.00 [95 % CI: -2.73; 2.73]; spring-feeding: *M* = 0.01 [95 % CI: -0.05; 0.07]). In contrast, clutch sizes were negatively affected, whereby the effects of spring-feeding were statistically significant (winter-feeding: *M* = -1.61 [95 % CI: -16.18; 12.96]; spring-feeding: *M* = -0.02 [95 % CI: -0.03; -0.01]). While winter-feeding had a positive impact on tit's brood sizes (*M* = 0.20 [95 % CI: -1.18; 1.59]), spring-feeding had a slightly negative one (*M* = -0.01 [95 % CI: -0.09; 0.06]). Further breeding measures could only be determined for either winter- or spring-feeding. Winter-feeding impacted slightly statistically significant positively tit's nest box occupancies (*M* = 0.06 [95 % CI: 0.03; 0.09]) and statistically significant negatively tit's fledging success (*M* = -0.18 [95 % CI: -0.28; -0.07]). Spring-feeding had slightly negative effects on tit's egg incubation periods (*M* = -0.10 [95 % CI: -0.58; 0.38]). (Fig. 15; Fig. 16)

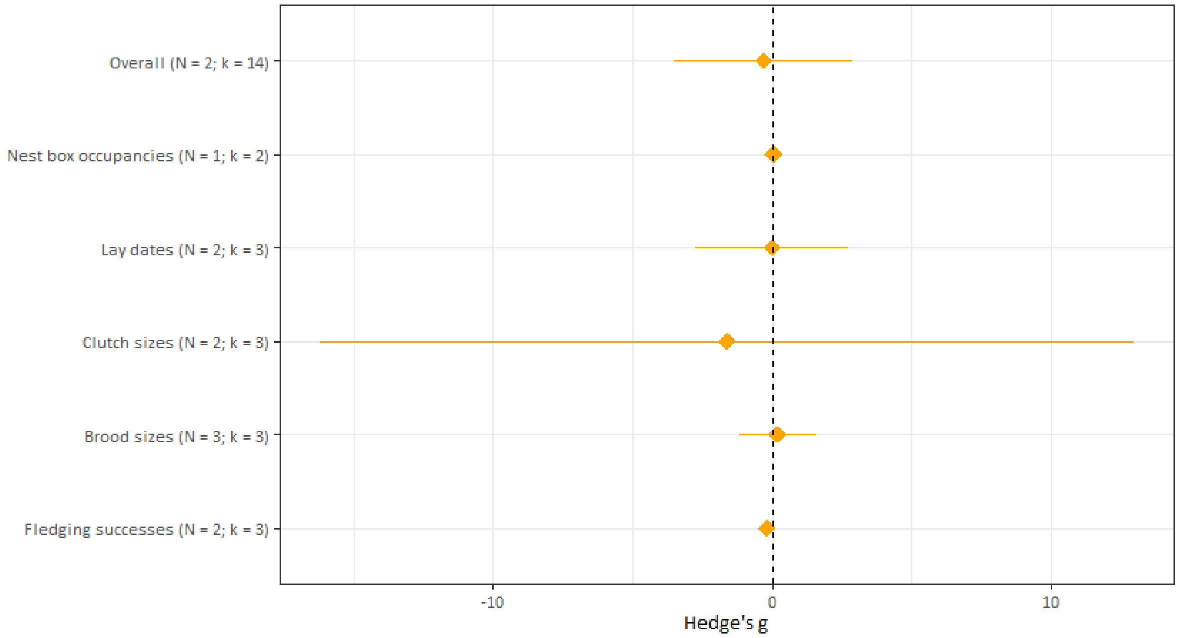


Fig. 15: Effects of winter-bird-feeding on tit's breeding measures. Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

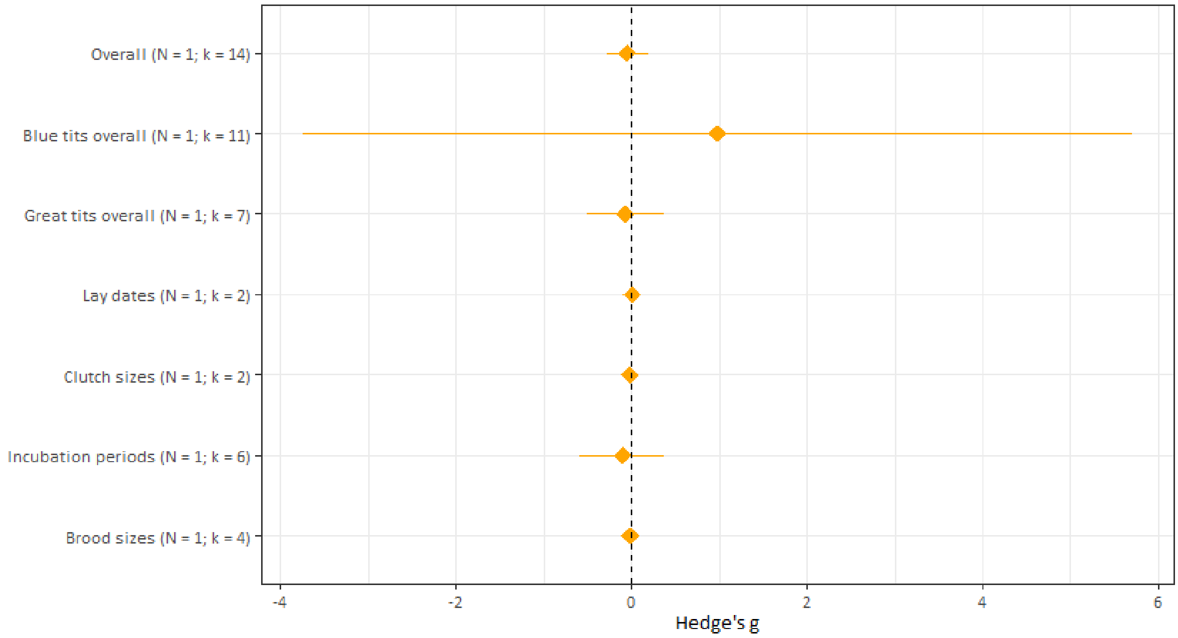


Fig. 16: Effects of spring-bird-feeding on tit's breeding measures. Shown are weighed mean effect sizes (*M*) (diamonds), calculated as Hedge's *g*, 95 % CI (horizontal lines), number of studies (*N*) and number of individual effect sizes (*k*) considered within the weighed mean effect size calculation. Effects can be considered statistically significant if the 95 % CI does not overlap zero (vertical dashed line).

4.2 Popular Scientific Literature

4.2.1 Planting

As plants are the major component of gardens, various hints were given on plant types and species and planting patterns throughout private gardens on the internet. In many cases, on the websites of 'Mein schöner Garten' and 'Utopia', the decisive factor behind these hints was the aesthetic quality rather than the conservation value of gardens. For instance, the planting of exotic plants due to sun resistance (URL 21), tropic flair (URL 22), scent (URL 23) or the design of stone gardens (URL 24) were advertised. However, as sustainable gardening is recognised as a current trend (URL 11), many articles explicitly dealing with biodiversity-friendly gardening practices and planting patterns, particularly advertising the conservation of insects, bees, butterflies and birds, could be found on all three examined websites (URL 8; URL 9; URL 10). The general recommendation were diverse planting patterns of predominantly wild sorts of both annual and perennial non-double flowers, herbs, forbs, shrubs and trees with different blooming and fruiting periods throughout the whole garden area, enabling the availability of pollen, nectar and fruits as many months as possible (URL 8; URL 25; URL 10). Considering flowers, sun-exposed locations were suggested in order to attract especially insects (URL 29). Also, lists on specific beneficial plant species for insects (URL 26; URL 27; URL 28), bees (URL 29; URL 25; URL 30), butterflies (URL 31; URL 32; URL 33) and birds (URL 8; URL 34; URL 35) were commonly available. It was mostly indicated that insects in general, bees and birds benefit the most from native plants (URL 27; URL 28). However, the conservation value of native plants was not mentioned in some articles published by 'Mein schöner Garten' (URL 26; URL 29) and the planting of exotic species was even encouraged (URL 8). Considering butterflies, it was commonly predicted that, besides natives, specific non-native plant species might be highly beneficial (URL 36; URL 37).

4.2.2 Pesticide Use

Throughout articles explicitly dealing with pesticides (URL 38; URL 39) and articles on ecological gardening practices in general (URL 8; URL 9; URL 10), the use of chemical pesticides in private gardens was collectively labelled as a threat for human health and biodiversity, particularly regarding insects and birds. Therefore, natural pest control alternatives such as weeding, mulching (URL 39), useful plants and insects or natural pesticides (URL 9; URL 10) were recommended.

4.2.3 Mowing

Articles explicitly dealing with lawn mowing were only available on the websites of 'Mein schöner Garten' and 'Utopia'. Mowing frequencies of approximately once a week and mowing heights of at least two thirds of the original height were regarded as the optimal lawn care in order to obtain aesthetically pleasing results (URL 40; URL 41). Especially throughout the months of May and June, frequent mowing was indicated to be essential (URL 40). In contrast, articles on ecological gardening practices highlighted the conservation benefits of extensive mown lawns (URL 9) and the replacement of parts of lawns with flowering meadows (URL 8; URL 10) being mown twice per year (URL 34) or wild, unmanaged garden parts (URL 8; URL 10).

4.2.4 Bird Feeding

Bird feeding was a frequently addressed but controverse topic online. While 'Mein schöner Garten' recommended the implementation of bird feeding practices all-year-round (URL 42), the 'NABU' advertised it from November until the end of February (URL 43) and 'Utopia' for severe winter months only (URL 44). Thereby, 'Mein schöner Garten' stated that winter- and, even better, all-year-round-feeding impacts positively bird species conservation. In contrast, the 'NABU' and 'Utopia' were convinced that only few non-threatened bird species would benefit from human food supplementation in terms of survival rate, which could not be labelled as an effective species conservation strategy. 'Utopia' proposed to rather support environmental conservation by donations than by bird feeding, but the 'NABU' considered it as an important educational practice raising awareness on the natural environment. However, besides or instead of bird feeding, it was highly recommended to provide natural food resources such as flowering and fruiting plants or withered flowers in winter (URL 43; URL 44). Although bird feeding practices were partly regarded sceptically, hints were given on its appropriate implementation. It was generally predicted that various natural food items such as seeds, peanuts, oats, fruits and raisins (URL 42; URL 43) should be used in order meet the dietary preferences of as many bird species as possible. A mixture of fat and seeds known as 'Meisenknödel', commonly available in German supermarkets during winter, were emphasised to be particularly beneficial for tits, in case of not being wrapped in plastic nets (URL 43). Human-induced food items such as bread or old cooking fat were claimed to be harmful (URL 42). The food should be primarily offered in bird-feed dispensers, especially during the summer months, in order to prevent the development and spread of diseases (URL 42; URL 43; URL 44). Other types of bird feeders must be cleaned regularly with hot water (URL 42; URL 43; URL 44). Also, bird feeders should be located at sheltered garden sites, and only small amounts of food should be exposed, protecting the food items to become rotten quickly (URL 42; URL 43; URL 44). More than one feeding site per garden should be available, located at different heights, preventing food competition and promoting various bird species (URL 42; URL 43).

5 Discussion

5.1 Meta-Analysis

5.1.1 Planting

5.1.1.1 Native and Non-Native Plants

Non-significant positive mean effects of native vs. non-native plants on insects in general, pollinators, butterflies and birds as well as significant positive effects on hymenopteran abundances were determined. These positive effects have been shown by a large number of publications before (Tallamy & Shropshire, 2009; Paker et al., 2014; Fukase, 2016). Only one publication, which was also considered within the present meta-analysis, did not find any significant positive effects of native plant addition to urban gardens on bees and butterflies, but the authors claimed their own results to be unexpected and likely inaccurate due to their experimental study design (Matteson & Langelotto, 2011). Generally, it should be considered that native plants always refer to the flora of the study origin so that specific recommendations for plant species cannot be made on the basis of this global meta-analysis.

In general, it is assumed that native vegetation supports the diversity and occurrence of the corresponding native insects (Tallamy, 2004; Zuefle et al., 2008; Tallamy & Shropshire, 2009). This mechanism is explained by the lack of a common evolutionary history of native insects with introduced plants (Tallamy, 2004; Tallamy & Shropshire, 2009) and the long time period that is needed in order to adapt to these (Zuefle et al., 2008). Thereby, the magnitude and direction of the effects of native plant planting on biodiversity depend on the individual interactions between flora and fauna. For instance, native plants of the family Asteraceae received more pollinator visits than non-native congeners, but the opposite was found regarding plants of the family Balsaminaceae (Chrobock et al., 2013). Additionally, native plants are known to be more attractive to bee species others than the genera *Apis* and *Bombus* (Garbuzov & Ratnieks, 2014).

Besides the ascertainment of positive effects of native plants on butterfly biodiversity (Burghardt et al., 2009; Tallamy & Shropshire, 2009), as it was also determined within the present meta-analysis, missing effects have been reported throughout various publications (Matteson & Langelotto, 2011; Fukase, 2016). Butterflies are known to be dependent on host and food plants (Olivier et al., 2016), whereby each butterfly species, both generalists and specialists, prefer certain plant species (Shackleton & Ratnieks, 2016). These preferences are correlated to the butterfly's tongue length, the plant's corella length and nectar availability (Shackleton & Ratnieks, 2016). Garbuzov and Ratnieks (2014) therefore concluded that butterflies could not be easily promoted by garden plants due to their high specialisation, which could explain the non-significance of the positive effects on butterfly biodiversity within the present meta-analysis. Butterflies are probably best supported through garden management actions directly targeting specific butterfly species. Alternatively, very high degrees of heterogeneity among garden flowers will likely also support butterfly specialists as comprehensive heterogeneity increases the probability of rare matches between highly specialised butterflies and plants.

Similar to insects, the mechanism of native species being attracted by native plants is assumed in birds (French et al., 2005; Lerman & Warren, 2011; Paker et al., 2014). Native vegetation is used by native birds as foraging and nesting resource. Natural food sources for small garden birds are mostly insects, nectar and fruits and berries (Paker et al., 2014). Therefore, native plants that host such food items are directly correlated to native bird biodiversity, explaining the positive effect size of native plants on bird richness determined within the present meta-analysis. Moreover, sufficient trees as nesting and resting sites are determinative for particularly tree-nesting birds (French et al., 2005; Shwartz et al., 2013; Paker et al., 2014). As tree-nesting birds are known to be more abundant than hole- or ground-nesting birds in urban areas, trees are the primary beneficial nesting resource (Sandström et al., 2006). However, an equal balance between tree coverage and open space in private gardens is crucial for bird species richness (Paker et al., 2014). Because single small scale gardens are unsuitable to provide the needed habitat heterogeneity, mostly generalist bird species with omnivorous diets such as house sparrows (*Passer domesticus*), black-billed magpies (*Pica pica*) and Eurasian jackdaws (*Corvus monedula*) are prevalent in urban areas (Sandström et al., 2006). More specialised species usually show higher habitat requirements and are equally affected by the surrounding landscape (Belaire et al., 2014). In order to replace the missing natural habitats in larger cities, the distribution of various greenspaces within and throughout urban areas with different habitat characteristics consisting of diverse native plants is essential to promote particularly native bird biodiversity (Sandström et al., 2006; Belaire et al., 2014).

Regarding the effects of non-native plants on biodiversity, negative impacts were determined on insect abundances within the present meta-analysis. This finding is due to Burghardt et al. (2009), who have shown that non-native plants were negatively correlated to avian and lepidopteran abundances. Throughout further publications beyond the present meta-analysis, it was emphasised that, in comparison to native plants, non-native plants support less faunal biodiversity (Tallamy & Shropshire, 2009; Smith et al., 2015). However, Matteson and Langelotto (2011) claimed that urban bees and butterflies benefitted likewise from native and non-native ornamentals, which was explained by the broad feeding habits of urban-adapted taxa. Nevertheless, it is assumed that the spread of non-native plants could be a threat to entire ecosystems, which would have particular negative consequences on native insects and birds (Tallamy, 2004). Exceptions could be the plantings of sterile non-native plants in order to prevent their further expansion (Shwartz et al., 2013).

Many private gardens are predominated by non-native plants due to their visual attractiveness (Thompson et al., 2003; Matteson & Langelotto, 2011; Bates et al., 2011). Apparently, this gardening habit is likely disadvantageous particular for native insects and birds, and, in reverse, non-native insects and birds may benefit from high amounts of non-native plants. Further research is required on the adjustment of native urban bee and butterfly species to non-native flowers. However, private gardens dominated by heterogeneous planting patterns of native flowers, shrubs and trees potentially impact best the conservation of primordial native ecosystems.

5.1.1.2 Flowers

Significant positive effects of flowers on the biodiversity of insects were revealed throughout the present meta-analysis, which referred primarily to bees (Apidae). On the contrary, no significant mean effects could be determined on wasps (Vespidae). Similar findings, particularly the individual effects of floral richness and floral abundances on bees, have been proposed by various publications throughout the recent years (Lowenstein et al., 2015; Shackleton & Ratnieks, 2016; Foster et al., 2017). It is known that floral abundances are positively correlated with bee abundances and, vice versa, floral richness with bee richness (Potts et al., 2003; Fründ et al., 2010). Moreover, floral richness and heterogeneity are major garden characteristics in order to promote bee biodiversity (Ghazoul, 2006; Lowenstein et al., 2014).

Regarding wasps, Ebeling et al. (2012) determined slightly positive effects of flowering plants but concluded that the surrounding landscape affected wasp biodiversity more clearly. Thereby, wasps were observed to prefer woody over open grassland habitats (Ebeling et al., 2012). Similar patterns have been stated in further publications beyond the present meta-analysis (Mello et al., 2011; Clemente et al., 2012). Wasps are known to forage on carbohydrates such as nectar, fruit and honeydew as well as on proteins derived from insect prey in order to feed their larvae (Prezoto et al., 2019). Potential natural nesting sites are primarily arboreal structures (Clemente et al., 2012). Therefore, habitats consisting of complex vegetation such as forests are most beneficial for wasps in terms of food and nesting resources (Clemente et al., 2012). Additionally, the foraging behaviour could explain the weak and non-significant correlation between wasps and flowers within the present meta-analysis. As wasps not only rely on nectar as food source and flowers not only on wasps as pollinators, the mutual dependency is rather weak (Mello et al., 2011).

Beyond bee biodiversity, positive effects of flowers on pollinators in general (Bates et al., 2011; Lowenstein et al., 2015) and bumblebees (Ahrné et al., 2009; Garbuzov & Ratnieks, 2014; Hanley et al., 2014) are well known. Various floral characteristics may impact the attractiveness of flowers to pollinators. The availability, quantity and quality of nectar and pollen (Garbuzov & Ratnieks, 2014), flower morphology (Fukase, 2016), flower colour (Clemente et al., 2012), flowering period (Garbuzov & Ratnieks, 2014) and floral distribution (Plascencia & Philpott, 2017) are major reasons for pollinators to make use of flowering plants. Further, preferences of taxonomic groups and species differ according to their body structure, especially tongue length (Garbuzov & Ratnieks, 2014; Shackleton & Ratnieks, 2016; Foster et al., 2017), seasonal cycle (Foster et al., 2017) and foraging and nesting requirements (Clemente et al., 2012; Lowenstein et al., 2015; Olivier et al., 2016), directly influencing the biodiversity of pollinators in dependence of flower species established within gardens. These findings suggest that heterogeneous planting of many flowering plants of different species throughout the whole garden as well as in adjacent gardens promotes pollinator biodiversity likely stronger than homogeneous management decisions. However, parasitism rate might also increase with increasing floral diversity, but in accordance with Ebeling et al. (2012), the benefits of floral diversity should superimpose potential risks of parasitism on bees and wasps.

5.1.2 Pesticide Use

Mean effects of pesticide use on insect biodiversity in gardens were mostly non-significant and inconsistent. While positive impacts were determined on insects and bee and butterfly abundances, the group of pollinators only was negatively affected. In addition, the results differed according to the type of pesticide. Insect abundances were diminished due to insecticides and herbicides, whereas fungicides and snail pellets led to positive mean effects. The positive mean effects of pesticides, especially fungicides and snail pellets, on insects, bees and butterflies are rather surprising because a variety of publications throughout the recent years reported consistent negative effects of pesticides on insects (de Snoo, 1999; Frampton & Dorne, 2007; Forister et al., 2019), bees (Brittain et al., 2010), bumblebees (*Bombus*) (Baron et al., 2017) and butterflies (Longley & Sotherton, 1997; de Snoo, 1999; Feber et al., 2007). The authors of the most determinative publication on the topic of pesticides considered within the present meta-analysis (Muratet & Fontaine, 2015) hypothesised that these positive effects could be indirect consequences of the abundances of pest-free plants through the application of pesticides. For instance, these plants might be beneficial for insects in terms of nectar production. Also, it is known that the effects of pesticides on non-target species such as bees and butterflies are highly dependent on the characteristics of the floral target species (Longley & Sotherton, 1997; Carrié et al., 2018) and faunal non-target species (Brittain et al., 2010; Arena & Sgolastra, 2014), the affected habitat (Longley & Sotherton, 1997; Brittain et al., 2010), the frequency and manner of pesticide application (Hole et al., 2005; Goulson et al., 2015) and the type of pesticide (Arena & Sgolastra, 2014; Muratet & Fontaine, 2015). As a consequence, some pesticides might promote specific insect abundances if their target feeding plants are supported, but negative effects can be expected for insect richness and habitat heterogeneity.

In accordance with the present study, particularly the treatment of vegetation with insecticides and herbicides are known to be harmful practices to insects (Frampton & Dorne, 2007; Feber et al., 2007; Muratet & Fontaine, 2015). The exposure of bees (Brittain et al., 2010; Arena & Sgolastra, 2014;

Goulson et al., 2015), bumblebees (Brittain et al., 2010; Baron et al., 2017) and butterflies (Longley & Sotherton, 1997; Brittain et al., 2010) to insecticides revealed toxic effects on the faunal species. Thereby, neonicotinoids, a common group of insecticides, were found to cause particularly severe effects (Arena & Sgolastra, 2014; Wood & Goulson, 2017). Mostly applied to seeds of crops, neonicotinoids become abundant in all plant tissues (Wood & Goulson, 2017). Besides, non-target floral species in proximity are affected by neonicotinoid applications by the chemical's rapid dispersal due to water solubility (Wood & Goulson, 2017). As nectar and pollen of these flowers exhibit measurable neonicotinoid levels, especially pollinators have a high probability of exposure to the chemical (Wood & Goulson, 2017). Regarding bees, confrontations with the neurotoxic neonicotinoids result in lethal and sublethal conditions diminishing bee's foraging behaviours and memory and learning capabilities (Goulson et al., 2015; Wood & Goulson, 2017). In addition, degraded immune systems (Wood & Goulson, 2017) and reproductive outputs of affected bees were determined (Brittain et al., 2010; Baron et al., 2017; Wood & Goulson, 2017), which could be due to toxin-related reduced physiological conditions (Wood & Goulson, 2017; Baron et al., 2017). Regarding social bees, entire populations could suffer from the consequences of an increase of diseases, pathogens and reduced fecundity (Wood & Goulson, 2017). However, especially bumblebees are known to be less likely affected by neonicotinoid and in general insecticide applications compared to other pollinators (Brittain et al., 2010; Arena & Sgolastra, 2014; Wood & Goulson, 2017). Their long flying distances make them less vulnerable to local environmental conditions, and their rather large body size enables a smaller surface to volume ratio in case of direct insecticide applications (Brittain et al., 2010) as well as a higher resilience to indirect exposure through affected flowers (Wood & Goulson, 2017). This implies that wild bees with mostly small body sizes are particularly threatened by insecticides (Wood & Goulson, 2017). Butterflies are less severely affected by neonicotinoids (Wood & Goulson, 2017). However, sublethal conditions due to insecticide applications have been observed (Longley & Sotherton, 1997). Herbicide applications cause declines in plant abundances and, most importantly, richness (de Snoo, 1999; Goulson et al., 2015; Muratet & Fontaine, 2015). Because bees are dependent on sufficient floral resources in order to obtain pollen and nectar, herbicides indirectly impact bee biodiversity negatively (Goulson et al., 2015). Regarding butterflies, particularly the reduced availability of perennials, the primary target of herbicides, consequences a lack of host and food plant species (Longley & Sotherton, 1997; Feber et al., 2007). Additionally, both chemical pesticide and herbicide applications potentially kill butterfly larvae on target plants (Feber et al., 2007; Brittain et al., 2010).

The use of pesticides is mainly raised within the current ongoing discussion on biodiversity-friendly agricultural practices. It is repeatedly proposed that a conversion from conventional to organic agricultural systems is a key conservation strategy of various taxonomic groups (Longley & Sotherton, 1997; Carrié et al., 2018; Forister et al., 2019). In order to eliminate pesticide use, especially regarding neonicotinoids, alternative ecologically tolerable pest control opportunities such as crop rotation (Goulson et al., 2015), wasps or biopesticides (Samada & Tambunan, 2020) must become more popular. However, pesticide use seems to be a common practice amongst private gardeners as well, as it was revealed by a survey in peri-urban areas in Sweden that reported on a regular pesticide application on 80 % of the included private properties (Ahmed et al., 2011). While the transformation from conventional to organic agriculture is a complex challenge with several stakeholders included, ceasing the use of pesticides in private gardens would be rather simple. The collective elimination of pesticides, particularly insecticides and herbicides, from small patches such as private gardens could impact clearly positively insect conservation worldwide (Goulson et al., 2015; Muratet & Fontaine, 2015; Forister et al., 2019).

5.1.3 Mowing

Frequent mowing revealed non-significant slightly negative mean effects on insect biodiversity throughout the present meta-analysis. The most striking negative mean effect was found on grassland true bugs by comparing weekly mowing to a zero mowing regime. Low mowing heights led to non-significant negative mean effects on insects, insect abundances and insect richness. In the publications considered within the present meta-analysis, compared to a mowing regime of every two weeks, significant negative effects of a weekly mowing on grassland true bug richness (Helden & Leather, 2004) and on bee (Apidae) abundances (Lerman et al., 2018) were stated. Surprisingly, Lerman et al. (2018) detected lower bee richness on yards being mown every two weeks compared to those being mown once a week or every three weeks. Despite these results, the authors concluded that less frequent mowing would be most beneficial to bee biodiversity. Additionally, recent published publications that were not considered within the present meta-analysis reported on lawns obtaining an extensive mowing regime of twice per year holding significantly high wild bee and true bug abundances and richness in comparison to control sites (Unterweger et al.; Wastian et al., 2016). Therefore, in general, low mowing frequencies and zero mowing can considered to be beneficial for true bug and bee biodiversity (Unterweger et al.; Helden & Leather, 2004; Wastian et al., 2016; Del Toro & Ribbons, 2020). Also, because extensive mown lawns provide habitat for endangered bee and plant species (Wastian et al., 2016; Sehrt et al., 2020), and frequent mown lawns only promote highly resilient and adapted species (Shwartz et al., 2013; Garbuzov & Ratnieks, 2014; Sehrt et al., 2020), the mowing event itself can lead to immediate negative consequences for true bug abundances and richness, which could even result in the loss of entire populations (Unterweger et al.; Helden & Leather, 2004).

The higher faunal species abundances and richness on extensive mown study sites are clearly correlated to high floral abundances and richness. The herbaceous plants and flowers emerging on these lawns offer shelter as well as nesting and feeding resources to insects (Smith et al., 2015; Lerman et al., 2018; Del Toro & Ribbons, 2020). Furthermore, it was emphasised that the time of the year has an influence on the effects of mowing on bee biodiversity. For instance, the initiative 'No Mow May' in Appleton, Wisconsin, USA required gardeners to apply a zero mowing regime during the month of May, which is of utmost importance for early emerging bee species (Wastian et al., 2016; Del Toro & Ribbons, 2020). Because further resources are limited, these species rely on resources provided by unmown lawns (Del Toro & Ribbons, 2020). In addition, bees are known to benefit from unmown summer months (Wastian et al., 2016).

Smith et al. (2015) emphasised that insect abundances and richness were significantly lower in yards obtaining a two-cm mowing regime than yards obtaining a four-cm mowing regime, which applied to the insect orders dipterans (Diptera), true bugs (Hemiptera), beetles (Coleoptera) and hymenopterans (Hymenoptera) in particular. Because insect biodiversity is known to be directly related to floral biodiversity (Garbuzov & Ratnieks, 2014; Smith et al., 2015; Del Toro & Ribbons, 2020), this pattern could be explained by the diminished plant abundances and richness on short cut lawns. However, it is proposed that exclusively high vegetation on lawns supports only few plant species that are resistant to shade (Shwartz et al., 2013), which would directly correlate to a decline in insect biodiversity.

Although no statistical significance was revealed within the present meta-analysis, mowing frequencies and heights clearly affect insect biodiversity. Especially the application of extensive mowing regimes of every-four-weeks up to every-six-months as well as moderate mowing heights impact positively true bug and bee abundances, richness and the conservation of endangered species. Therefore, individual management decisions in private gardens will impact positively certain species, and thus, heterogeneous managed adjacent gardens altogether can provide habitats to a variety of species. Especially urban neighbourhoods could hence be of high conservation value. Because many gardeners perceive neatly mown gardens as appealing, it is worth knowing that already small changes such as the reduction of an every-week to an every-two-week mowing regime as well as a zero mowing regime during the month of May might promote insect biodiversity. These results suggest that a mind shift from the common towards rather insect-friendly mowing practices would potentially be of tremendous conservation value. Although more and more gardeners perceive wildlife-friendly gardening practices as positive, they do not want their gardens to be labelled as chaotic (Lindemann-Matthies & Marty, 2013). Most probable, initiatives such as the 'No Mow May' will raise the awareness on wildlife-friendly gardening practices and lead to more acceptance of less mown gardens.

5.1.4 Bird Feeding

Neither bird feeding in general nor winter- and spring-feeding impacted overall bird biodiversity. However, especially during winter, bird feeding led to mostly statistically non-significant positive mean effects on the current bird generation in terms of abundances and winter survival rates. More birds were attracted in comparison to control sites without food supplementation, which becomes confirmed throughout various publications (Brittingham & Temple, 1988; Wilson, 2001; Galbraith et al., 2015). Especially under severe winter temperatures of less than -18 °C in highly natural surroundings, black-capped chickadees (*Poecile atricapillus*) abundances and winter survival rates were observed to increase due to supplemented food (Brittingham & Temple, 1988; Wilson, 2001). It has been suggested that only such severe environmental conditions that lead to diminished food availability and a high dependency of birds on sufficient energy might have significant effects on individuals fitness and populations (Brittingham & Temple, 1988; Robb et al., 2008a). However, in urban habitats, increased bird abundances were detected around bird feeders under moderate temperatures as well (Fuller et al., 2008; Tryjanowski et al., 2015; Galbraith et al., 2015).

Bird feeding is a typical human-wildlife interaction of high social and educational value (Robb et al., 2008a). Thereby, increased winter survival rates and increased bird abundances around feedings sites are primary motivations for gardeners to engage in winter bird feeding (Jones & Reynolds, 2008). Probably, especially people maintaining a private garden often enjoy to promote wildlife on their property in order to protect but also observe the attracted species. Thus, the findings of the present study propose that the major expectations of bird feeding practices are met because survival rates and abundances in particularly urban private gardens can be increased by food supplementation. Thereby, it is proposed that food supply started in early winter is most beneficial in order to acquire maximal bird abundances and winter survival rates (Wilson, 2001).

Besides the findings of the present meta-analysis, various publications highlighted that bird richness might be negatively affected by bird feeders in urban areas because only a few certain species make use of the additional feeding opportunities (Fuller et al., 2008; Tryjanowski et al., 2015; Galbraith et al., 2015). During feeding experiments in New Zealand, house sparrows (*Passer domesticus*) were frequent visitors of bird feeders, whereas common mynas (*Acridotheres tristis*) did not show any interest in such food sources (Galbraith et al., 2015). Additionally, introduced bird species were found to dominate bird communities around feeding sites more likely than native bird species (Fuller et al., 2008; Galbraith et al., 2015). Possible explanations regarding these results could be the adjustment of certain bird species to urban environments and feeding sites (Fuller et al., 2008), the variety of bird's foraging and dietary preferences and increased competition amongst bird species around feeding sites (Jansson et al., 1981; Wilson, 2001; Galbraith et al., 2015). For example, as humans tend to provide seeds within their gardens, specifically granivores and omnivores are attracted by supplemented food (Galbraith et al., 2015). However, a long term data collection on the consequences of bird feeding within the UK showed an increase in the number of bird species making use of supplemental food over time (Plummer et al., 2019).

It is generally predicted that bird feeding might unintentionally result in additional ecological consequences. Frequently fed birds, particularly during winter, could develop a dependency on human-induced food resources (Robb et al., 2008a), changing their foraging (Wilson, 2001), breeding (Robb et al., 2008a) and even migration behaviour (Jansson et al., 1981; Orell, 1989; Jokimäki et al., 1996). Moreover, repeatedly addressed issues of bird feeding are the potential increase of predation rate by predatory birds and domestic cats (Robb et al., 2008a) and the transmission of diseases amongst birds and to humans (Robb et al., 2008a; Galbraith et al., 2015). In their review on the effects of bird feeding, Robb et al. (2008a) highlighted that increased predatory pressure on birds should not be of major concern. However, Brittingham and Temple (1986) and Fischer et al. (1997) reported an increased risk of disease transmission as a consequence of artificial feeding, whereby Robb et al. (2008a) emphasised that the type of feeder, the habitat it is located in and the respective bird abundances are determinative for the actual consequences.

These findings indicate that the ecological impacts of bird feeding on the current bird generation should be considered rather heterogeneous. Although particularly bird abundances and winter survival rates of birds are promoted by supplemental food, bird feeding might be a massive encroachment in species richness, behavioural ecology and disease transmission, which could even result in an increased risk of zoonosis for humans. Due to the popularity of bird feeding, consequences will appear on a broad scale (Galbraith et al., 2015). Because of the enjoyment of promoting wildlife on their own property, gardeners will likely not cease bird feeding practices entirely, highlighting the need to obtain more information on the potential ecological threats of bird feeding and on the provision of opportunities to prevent such negative consequences.

Breeding Measures

The overall primarily non-significant mean effects of winter-bird-feeding on the breeding measures of tits (Paridae) were negative, while only brood sizes were positively affected. Likewise, the overall mean effects of spring-feeding were slightly negative, with effects differing along individual tit species. The mean effects of winter-feeding were particularly greater than those in spring. Throughout the publications considered within the present meta-analysis (Brittingham & Temple, 1988; Robb et al., 2008b; Harrison et al., 2010; Plummer et al., 2013) and beyond (Jansson et al., 1981; Reynolds et al., 2003; Robb et al., 2008a) these effects were depicted rather contentious. Plummer et al. (2013) observed decreases in clutch sizes, chick masses and fledging successes but no effects on the lay dates in association with winter food supplementation of tits. By contrast, Robb et al. (2008b) reported on increases in clutch sizes, fledging successes and earlier lay dates but no effects on chick masses. Winter-feeding of tits has be assumed both to be a potential threat to the following bird generation (Plummer et al., 2013) and the exact opposite (Jansson et al., 1981; Robb et al., 2008b). Regarding black-capped chickadees, no effects on breeding measures were detected (Brittingham & Temple, 1988). Considering spring-bird-feeding, a heterogeneous picture on the subsequent breeding measures is revealed as well. While positive consequences such as advanced lay dates, clutch sizes, egg masses, incubation periods, hatching successes and chick's body conditions were reported (Reynolds et al., 2003; Harrison et al., 2010), negative consequences such as decreased clutch and brood sizes were equally stated (Reynolds et al., 2003; Harrison et al., 2010). However, spring-feeding is known to impact breeding measures less than winter-feeding (Brittingham & Temple, 1988; Wilson, 2001; Harrison et al., 2010). These heterogeneous findings throughout a variety of publications could explain the prevalent non-significance of the effects of bird feeding on breeding measures throughout the present meta-analysis. The most likely explanation regarding these discrepancies are the types of foods that were supplemented during individual feeding experiments. Popular food supplements in private gardens are solid fat and seeds, which are both known to hold energy-rich macronutrients. However, especially peanuts hold protein and additional valuable micronutrients such as Vitamin E, an antioxidant, whereas solid vegetable balls are poor in other nutrients beyond lipids (Robb et al., 2008b; Harrison et al., 2010; Plummer et al., 2013). It is known that protein- and nutrient-rich-diets of birds are the major drivers of advanced breeding measures (Reynolds et al., 2003; Robb et al., 2008a; Harrison et al., 2010). Therefore, the rather negative results regarding the breeding measures of fed tits might be primarily caused by a nutrient-poor diet of parental birds (Plummer et al., 2013). This assumption aligns with the additional finding that the provision of fat-balls supplemented with Vitamin E (α-tocopherol) resulted in a higher hatching success of tit chicks than the provision of sheer fat-balls (Plummer et al., 2013). It is likely that the consequences of bird feeding on the following bird generations remain underestimated among gardeners, which could lead to unintentionally caused decreases in breeding measures. Gardeners should therefore provide protein- and micronutrient-rich high quality food such as peanuts (Robb et al., 2008b), peanut-cakes (Harrison et al., 2010) or sunflower seeds (Jansson et al., 1981; Wilson, 2001) instead of lipid-rich low quality food within their properties.

The magnitude and direction of the effects of bird feeding on breeding measures is furthermore dependent on the location of feeding site (Reynolds et al., 2003; Robb et al., 2008a; Plummer et al., 2013), individual bird species and environmental conditions. For instance, as revealed in the present meta-analysis, a feeding experiment emphasised positive effects on the hatching successes of bird chicks in great tits (*Parus major*) but negative effects in blue tits (*Cyanistes caeruleus*) (Harrison et al., 2010). Potential environmental conditions impacting the breeding success of birds are precipitation, the availability of natural food (Reynolds et al., 2003) and temperature (Robb et al., 2008a). Moreover, the time period of food supplementation can have severe impacts. Bird feeding over several months (Reynolds et al., 2003) or even throughout the whole year (Plummer et al., 2013) might be more beneficial regarding breeding measures than short-term food provision. The benefits of long-term food supplementation are based on a potential ecological trap that is created by bird feeding. Adult birds might rely on the consistent food availability, which provides them with extra energy and saves time, and therefore encourages them to invest increasingly into their reproductivity (Robb et al., 2008a). As soon as the feeding practice stops, negative consequences are occurring. For instance, adult birds might not be able to feed the high number of chicks that could have emerged due to supplemented food (Robb et al., 2008a). Bird feeding is a particularly popular practice during winter. Considering the potential threats of the time limited food supplementation only during the winter months, it can be suggested that gardeners should provide bird food all-year-round or dispense such practices.

Due to increased urbanisation, the promotion of wildlife in private gardens particularly in cities might be of high importance. However, it is questionable if the massive interference into bird ecology should be additionally encouraged. The popularity and enjoyment of bird feeding worldwide imply the need of further research on the advantages and disadvantages of such practices as well as adequate information on its wildlife-friendly implementation.

5.2 Popular Scientific Literature

5.2.1 Planting

According to the scientific findings stated within the present study, gardeners looking for inspiration on their floral garden design online will most likely find management recommendations on planting patterns that are not necessarily biodiversity-friendly such as the use of non-native plant species. However, gardeners enjoying the sight of insects and birds or explicitly aiming at enhancing their garden's conservation value will find many articles dealing with the importance of a diverse plant composition and native plants, as it was already ascertained within the present meta-analysis. Also, butterflies were emphasised to be an exemption, as they would benefit most from specific, also non-native, butterfly-friendly garden plants. The online lists on insect-, bee-, butterfly- and bird-friendly plant species will probably facilitate the implementation of appropriate planting patterns. Because various of such lists were available, and many plant species were part of them, it is probable that individual gardeners will not make similar planting decisions, assuring heterogeneous management actions amongst private gardens in residential neighbourhoods. However, a lack of advertisement of native plants by the gardening magazine 'Mein schöner Garten' was recognised, which indicates that even environmentally interested gardeners could become disinformed, resulting in unintentional non-biodiversity-friendly management decisions.

5.2.2 Pesticide Use

On the internet, even gardeners that are not interested in the conservation value of their gardens are widely encouraged to cease their chemical pesticide use entirely, which goes in line with the findings of the present study. Also, information on ecological pest control opportunities can be easily found. Therefore, it is likely that many gardeners become more conscious of the severe ecological consequences of chemical pesticides and at least try to implement the biodiversity-friendly alternatives that are being recommended.

5.2.3 Mowing

Gardeners which are about to put effort into intensive lawn management, as neatly maintained lawns are commonly perceived as aesthetically appealing, will most likely find subsequent management recommendations on the internet without being informed on the negative impacts on biodiversity. In contrast, unecological mowing practices such as weekly mowing or more intensive mowing during the month of May were recommended online. Only gardeners that are interested in biodiversity-friendly gardening practices in general will obtain scientifically correct information on the benefits of extensive lawn management regarding insect and bird conservation, as it was also emphasised throughout the present meta-analysis.

5.2.4 Bird Feeding

Gardeners being interested in the environmental impact of their bird feeding practices will find contentious information on the internet, which is most probable due to the different scientific opinions on the topic, also reflected by the present study. Similar to the findings of some publications, it was partly highlighted that bird feeding does not impact species conservation because only abundances and winter survival rates of few species will probably be increased through food supplementation. Nonetheless, although proposals on ceasing such practices were made, many hints were given on its species-appropriate implementation. Information was provided on the types of bird feeders, their distribution throughout the garden and necessary hygiene practices. Throughout the present study, it was determined that more scientific research on these topics is needed in order to provide gardeners with helpful hints. Comparing the contents online to the findings of the present meta-analysis, scientifically profound information was given on the type of bird food in terms of being natural and protein- and micronutrient-rich because seeds, peanuts, oats, fruits and raisins were recommended. However, fat-balls, known as 'Meisenknödel', were also depicted as generally beneficial, whereas the present study ascertained that the lipid-rich fat-balls could be harmful for bird's breeding measures. On the internet, the optimal timespan of bird feeding remained unclear. Therefore, it is likely that gardeners will decide differently on times of food supplementation, potentially resulting unintentionally in ecological consequences for birds in terms of diminished breeding success in case of food availability during winter months only.

6 Method Discussion

Although the present meta-analysis aimed to gain global profound evidence on the effects of common gardening practices on biodiversity, the results are limited due to methodological reasons. Many publications derived from the systematic literature research could not be included into the meta-analysis due to a lack of adequate data in order to calculate effect sizes. The systematic literature research yielded particularly publications dealing with gardening practices related to planting of native plants (A1) and flowers (A2) and supplemental feeding (I) on different taxonomic groups of insects (3a, 3b, 3d, 3abcd, 3e) and birds (2). Apparently, these gardening practices and taxonomic groups are common and popular study focusses, directly influencing the major contents of the present study. The availability of publications on other gardening practices and taxonomic groups was limited, and the meta-analyses on the effects of mowing and pesticide use might therefore be biased due to a lack of sufficient publications. Also, within the publications, species biodiversity on population and community levels, measured particularly by abundance and richness, was examined. It is known that abundance and richness, especially regarding small sample sizes, are inappropriate measures in order to reflect the true abundance and richness of communities (Roswell et al., 2021). In addition, due to the prevalence of these biodiversity measures, the effects of common gardening practices on species biodiversity regarding individuals as well as genetic and ecosystem biodiversity remains unclear. Most of the publications finally considered within the meta-analysis presented statistically significant positive or negative effects rather than zero-effects, which is a common phenomenon known as 'publication bias' in scientific studies and implies that the mean effect sizes calculated within the present meta-analysis should not be overinterpreted (Borenstein et al., 2010). Moreover, apart from urban gardens, studies were conducted in public urban and rural greenspaces. As it is known that a large greenspace, thus study area, correlates positively to insect and bird biodiversity (Daniels & Kirkpatrick, 2006; Lowenstein et al., 2014; Olivier et al., 2016), the effect sizes derived from the corresponding publications might be slightly biased. Also, as the majority of publications were originated in Western Europe and the USA, the results do not apply globally but only to the Northern hemisphere.

Within the statistical analyses, it is common practice to conduct additional sensitivity analyses with the own data set (Borenstein et al., 2010). Due to time constraints, sensitivity analyses were not implemented in the present study, which might have caused bias within the presented data.

7 Management Implications

Although ecological gardening practices are recommendable to anyone, they should be of particular interest for private gardeners living in urbanised areas as their gardens were found to be of high conservation value within an otherwise degraded environment. Thereby, residential areas with many adjacent, individually managed gardens are a potential habitat for many insect and bird species, as long as a consensus on the need of biodiversity-friendly garden management among the individual gardeners is prevalent. For instance, neighbourhood associations could commonly decide on certain guidelines within their living areas. A common sense would probably also diminish the fear of acting against social norms where individual gardeners could be perceived as chaotic due to the implementation of ecological gardening practices such as reduced mowing regimes or renouncement of pesticides (Goddard et al., 2010). In general, the present study asks private gardeners to reconsider their perception of aesthetically managed gardens, as decisions aiming at neatly ordered gardens are opposed to biodiversity-friendly practices.

7.1 Planting

7.1.1 Pollinators

Instead of providing gardeners with lists of pollinator-friendly garden plants or certain flower-mixtures, it should rather be suggested to implement and maintain heterogeneous, native-dominated planting patterns in private gardens. Species abundances and richness of native pollinators are probably best promoted by many native flowering plants comprising various plant heights, floral morphologies, floral colours, blooming periods, annuals and perennials appearing both intentionally and spontaneously in patchy as well as clustered distributions all around the property (Fig. 17). Such a floral garden composition is probably best met by making use various native seeds and plants. Native flower-mixtures could be advantageous resources in order to simplify this management implication as long as different flower-mixtures are used, ensuring that adjacent gardens do not show similar floral compositions. As, currently, native seeds, flower-mixtures and plants are hardly obtainable and identifiable at conventional gardening shops, the 'NABU' provides a list of possible sources of ecological and regional seeds and plants online, for instance (URL 45).

Fig. 17: Pollinator biodiversity is promoted by heterogeneous native-dominated floral planting patterns, offering a large variety of food resources.

Source: Left: Ulrich Meyer-Spethmann (2013); right: www.pixabay.com [08.04.2021]

Bees

Beyond feeding resources in terms of flowers, especially bees are highly dependent on nesting sites, preferably within the same garden (Fründ et al., 2010; Bates et al., 2011; Egerer et al., 2020). As cavity-nesting bees are most prevalent in urban areas (Lowenstein et al., 2014; Egerer et al., 2020), gardeners should provide potential nesting opportunities such as dead wood, hollow, myelinated plant stems and old brick walls (Amiet & Krebs, 2012; Goulson et al., 2015) (Fig. 18) or adequately designed artificial nest sites commonly known as 'bee hotels' (MacIvor & Packer, 2015; Fortel et al., 2016). Although urban gardens are likely to already host at least a few human-made nesting and overwintering sites such as brick walls in close proximity, mostly generalist bee species will benefit from these (Bates et al., 2011; Lowenstein et al., 2014; Foster et al., 2017). Thus, a lack of appropriate habitat for nesting and overwintering sites rather than flowers and feeding opportunities could unintentionally diminish the biodiversity of bees in gardens.

Bees are known to be temperature dependent as they prefer sun exposed and warm habitats (Fründ et al., 2010; Fukase, 2016; Foster et al., 2017). Although gardeners are not capable of altering weather conditions, it can be proposed that intentionally created bee habitats, both feeding and nesting sites, should obtain a maximum of sun exposure. Furthermore, as urban areas are known to be warmer than rural areas due to anthropogenic structures and heat generation (Memon et al., 2008), this finding indicates the potential of cities to serve as alternative bee habitats from the perspective of temperature-related environmental conditions.

Fig. 18: Cavity-nesting bees need adequate nesting sites such as sun-exposed old brick walls.
Source: www.pixabay.com [16.03.2021]

Butterflies

Butterflies are generally known to prefer plants with long blooming periods (Shackleton & Ratnieks, 2016), perennials (Feber et al., 2007), ornamentals (Tallamy & Shropshire, 2009) and woody plants (Tallamy & Shropshire, 2009). However, due to the high specification of butterflies on certain plant species, clear recommendations on beneficial garden plants can be given. In Europe and North-America, common butterfly-friendly plant genera are known to be oregano *(Origanum*), buddleia (*Buddleia*) and boneset (*Eupatorium*) (Shackleton & Ratnieks, 2016) (Fig. 19). Beyond flowers, woody plants genera such as oak (*Quercus*), cherry and plum (*Prunus*) and willow (*Salix*) are proven to promote butterfly biodiversity in gardens (Tallamy & Shropshire, 2009). To support native butterflies, it is not as decisive as in native pollinators to focus entirely on native plant origins. Lists of scientifically examined recommendable butterfly-plants are given by Tallamy and Shropshire (2009) or the 'Bundesamt für Naturschutz' (BfN) (URL 46), for instance. Alternatively, highly heterogeneous planting patterns within private gardens could unintentionally promote butterfly biodiversity as well due to rare matches between certain plants with certain butterfly species.

Fig. 19: Common butterfly-friendly flowers. From left to right: Oregano (*Origanum*), buddleia (*Buddleia*) and boneset (*Eupatorium*).
Source: www.pixabay.com [16.03.2021]

Similar to bees, also butterflies benefit from combinations of habitats that can be provided within private local gardens. Beneficial traits that have been reported are dead wood (Muratet & Fontaine, 2015), nettles (Muratet & Fontaine, 2015), lawn cover (Shwartz et al., 2013), tree cover (Shwartz et al.,

2013) and large garden areas (Muratet & Fontaine, 2015). Thereby, different butterfly species show habitat-specific preferences according to their origin. For instance, woodland species would benefit from sufficient tree cover and field margin species from sufficient lawn cover (Shwartz et al., 2013). The landscape characteristics of garden surroundings are determinative on butterfly biodiversity as well. Highly urbanised areas consisting of high proportions of impervious surface but low proportions and small sizes of greenspaces, which are unappealing to butterflies (Muratet & Fontaine, 2015; Olivier et al., 2016), should be avoided. Instead, gardeners, especially in residential areas with a high proportion of greenspace, should prefer a heterogeneous garden management aiming at garden compositions of both open and closed space, various (butterfly-friendly) flowers, shrubs and trees and natural garden areas allowing rather disliked characteristics such as nettles and dead wood.

7.1.2 Birds

To promote native bird biodiversity within private gardens, heterogeneous planting patterns of native shrubs, forbs and trees preferably providing fruit and berries are recommendable in order to meet the dietary and nesting preferences of native bird species. Because most gardeners prefer to make use of fruit and berry yields themselves, only few individual plants producing fruit or berries could be left unprotected from foraging birds (Fig. 20). Generally, bird biodiversity is probably best promoted by heterogeneous vegetation compositions within and amongst private gardens and further urban greenspaces. Private gardeners are not capable of impacting public urban greenspace management. Therefore it is likely that the effects of heterogeneous shaped gardens on birds are most beneficial in residential areas consisting of individually managed small greenspaces providing habitat to various bird species in close proximity (Sandström et al., 2006).

Fig. 20: Bird biodiversity is best promoted by heterogeneous native vegetational patterns offering fruits and berries.

Source: www.pixabay.com [16.03.2021]

7.2 Pesticide Use

Chemical pesticide use, irrespective of the type of pesticide, should be entirely ceased within private gardens but could be replaced by ecological alternatives in some cases. Currently, pesticide use might be important for gardeners engaging in crop growing, having time constraints or placing value on neatly managed plants. The risk of crops and plants suffering from pests can be diminished by alternative pest control strategies such as crop rotation (Frampton & Dorne, 2007), the introduction of social wasps (Prezoto et al., 2019), (essential) vegetable and plant oils (Samada & Tambunan, 2020) or biopesticides (Samada & Tambunan, 2020), revoking the need of chemical insecticides and fungicides. Although biopesticides, consisting of substances derived from living organisms could also reduce the probability of weed growth (Samada & Tambunan, 2020), herbicide application (both chemically and ecologically) diminishes the vegetational diversity within gardens, and thus, usage of any kind of herbicide is not recommendable for the sake of insect conservation.

7.3 Mowing

Private gardeners should apply extensive mowing regimes and moderate mowing heights to their lawns, especially during May and the summer months (Fig. 21). So far, no exact numbers of optimal mowing frequencies and heights can be determined, but the results of this meta-analysis imply that the majority of private gardeners should reduce their current mowing patterns. In order to compromise with the common aspiration of neatly managed lawns, gardeners could intentionally apply extensive mowing regimes to certain small parts of their properties as a first approach.

Fig. 21: Extensive mown lawns provide insects with food and shelter.
Source: Left: www.pixabay.com [16.03.2021]; right: Ulrich Meyer-Spethmann (2020)

7.4 Bird Feeding

The massive interference of humans into bird ecology through bird feeding practices potentially affects bird biodiversity negatively and raises the question of dispensing such habits. However, the potential positive impacts into bird biodiversity in urbanised areas, its worldwide popularity and the enjoyment of bird feeding and its educational value imply a need of applicable bird feeding guidelines.

Firstly, as birds might rely on the food availability, it could be more beneficial to supply bird food in private gardens all year-round rather than only during winter months (Plummer et al., 2013) (Fig. 22). Secondly, birds might benefit most from supplemented food being protein- and micronutrient-rich. For instance, peanuts (Robb et al., 2008b) and sunflower seeds (Jansson et al., 1981; Wilson, 2001) were found to supply adequate nutrients in order to influence breeding measures of fed birds positively. In contrast, lipid-rich bird food such as cheap fat-balls, in Germany known as 'Meisenknödel', containing few seeds, impact breeding measures potentially negatively (Plummer et al., 2013) and should therefore be abandoned from private gardens. In order to meet the dietary needs of various bird species, other food sources such as fruits and berries should be considered as well (Fig. 22).

Fig. 22: Bird feeding with nutrient-rich food such as seeds and berries should probably be practiced all-year-round. Source: www.pixabay.com [16.03.2021]

Apart from food quality, the type of bird feeders and food distribution might be decisive factors for the ecological consequences of bird feeding. For example, it is predicted that bird feeders in trees (Jansson et al., 1981), other bird feeder types than platform feeders (Brittingham & Temple, 1986), supplemental food distributed on the bare ground (Galbraith et al., 2015) and rather large amounts of available bird food (Robb et al., 2008a) could counteract competitive bird behaviours as well as disease transmissions. Further scientific research is required on types of bird feeders, food distribution throughout the garden and also cleaning patterns in bird feeders.

7.5 Scientific Communication

More appropriate and scientifically profound, but easily understandable, information on concrete applicable environmental gardening practices must be commonly communicated and advertised. Particularly scientists should communicate their findings in an appealing way, e.g. through mass media, to the general public (Scheufele & Krause, 2019). A positive example of such an initiative is the online database 'Conservation Evidence'(URL 47), aiming on evaluating and presenting all available scientific findings on certain conservation actions. However, all stakeholders providing information on gardening practices such as conservation associations, gardening magazines or gardening shops should be responsible of publishing scientifically proven information only.

8 Conclusion

Despite prevalent statistically non-significant effects within the meta-analysis, the present study found clear impacts of common gardening practices on the biodiversity of insects and birds on the population level. In particular the planting of native plants and flowers affected native insect biodiversity positively, whereas the planting of non-native plants, mowing and pesticide use revealed opposite effects. These findings complement with each other because it is generally assumed that diverse native vegetation patterns, which become diminished by non-native plant planting, intensive mowing and pesticide use, promote native insect biodiversity best. The effects of bird feeding remain inconsistent because positive effects were revealed on the current bird generation whereas negative effects have been reported on fitness related breeding measures. In addition, general concerns have been raised about the massive interference of artificial feeding on bird ecology.

The magnitude and direction of garden management effects highly depend on the individual faunal and floral species, the manner of gardening practice implementation, local garden characteristics, and the surrounding landscape. Thereby, clear trends have been emphasised. First, heterogeneity on local and landscape levels is extremely important to promote biodiversity. Diversely designed and managed gardens, neighbourhoods, cities and rural areas in terms of vegetation and structures provide various habitats potentially utilised by various taxa. Secondly, the effects of common gardening practices within urban areas seem to be more striking than in rural areas because the availability of greenspace in cities is limited. Due to rapid urbanisation worldwide, cities must act as replacement habitats to slow down global biodiversity declines, whereby private, naturally managed gardens promoting heterogeneity can be considered as decisive components.

On the internet, ecological gardening practices are commonly advertised by various institutions. However, most information on such practices will only be found by gardeners being interested in sustainable gardening, leaving conventional gardeners uninformed on the severe ecological consequences of their habits. Also, a lack of information on the importance of native plants and the optimal timespan of bird feeding was identified. Average environmental-conscious gardeners might therefore perform some unintentionally non-biodiversity-friendly gardening practices on their properties. There is a broader need of easily available, understandable and scientifically correct information on ecological gardening practices.

A clear mind shift within the entire society is necessary in order to commonly perceive rather diverse and wild biodiversity-friendly gardens as aesthetically appealing. There is a need of scientifically correct information on the value of biodiversity, the role of private (urban) gardens in conservation practice as well as primarily gardening advertisements emphasising the aesthetic and conservation value of such gardens. Also, sustainable gardening should not be an extra effort, asking especially gardening shops to make tools such as native wildflower-mixtures easily available, identifiable and appealing, addressing not only environmental-conscious consumers. Top-down initiatives such as financial incentives or regulations by governments could be opportunities to increase the popularity of biodiversity-friendly gardening practices (Goddard et al., 2010). Although only the individual gardeners themselves are responsible for their garden management practices, various stakeholders such as neighbours, members of neighbourhood associations, universities, municipalities, seed producers, owners of gardening shops, influencing personalities such as scientists and politicians and much more are directly and indirectly involved into the decision-making process of private gardeners (Goddard et al., 2010; Goddard et al., 2013; Lindemann-Matthies & Marty, 2013; Diduck et al., 2020). However, as a first step towards a biodiversity-friendly system, already environmental-conscious and scientifically well-informed gardeners must act as role models, initiating a bottom-up change within the society (Goddard et al., 2013; Lindemann-Matthies & Marty, 2013).

In order to provide broader scientific profound knowledge on ecological gardening practices, further research and meta-analyses should be conducted on the effects of additional popular actions such as the creation of vegetable beds, the use of fertilisers, irrigation, sealing, the creation of ponds and the provision of drinking water and artificial nest sites. Also, rather than solely focussing on species biodiversity of insects and birds, publications should regard the effects of such practices onto other organisms and, if viable, on genetic and ecosystem biodiversity. In order to include all relevant publications into meta-analyses, raw data, particularly sample sizes, means and standard errors or standard deviations, should be made easily available by the responsible authors.

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11 Appendix

Appendix Tab. 1: Countries, study site codes, taxon codes, biodiversity measure codes, study design codes, weights (*w*) and no. of effect sizes of publications considered within the meta-analysis according to each gardening practice. Study site codes: ug: urban gardens, pug: public urban greenspaces, rg: rural greenspaces. Taxon codes: 2: birds, 3a: hymenopterans, 3b: butterflies, 3d: true bugs, 3abcd: pollinators, 3e: general insects. Biodiversity measure codes: a: abundance, r: richness, d: diversity, bm: breeding measures, m: mortality. Study design codes: A: after impact only, CI: control vs. impact, G: gradient-response.

Practice (code)	Publication	Country	Study site codes	Taxon codes	Biodiversity measure codes	Study design codes	Weight (*w*)	No. of effect sizes
(Non-)native plant planting (A1)	Burghardt et al. (2009)	USA	ug	2; 3b	a; r	CI	4	8
(Non-)native plant planting (A1)	Chrobock et al. (2013)	Switzerland	pug; rg	3abcd	a; r	G	3	2
(Non-)native plant planting (A1)	Matteson and Langelotto (2011)	USA	ug	3a; 3b	a	CI	4	3
(Non-)native plant planting (A1)	Smith et al. (2015)	UK	pug	3e	a; r	G	5	6
(Non-)native plant planting (A1)	Zuefle et al. (2008)	UK	rg	3e	a	G	3	5
Flower planting (A2)	Ebeling et al. (2012)	Germany	rg	3a	d; m; bm	G	7	12
Flower planting (A2)	Fründ et al. (2010)	Germany	rg	3a; 3abcd	r; d	A	1	2
Flower planting (A2)	Ghazoul (2006)	UK	pug	3abcd	a	CI	7	1
Flower planting (A2)	Gunnarsson and Federsel (2014)	Sweden	ug; pug	3a	a	G	3	1
Flower planting (A2)	Lowenstein et al. (2014)	USA	ug; pug	3a	a; r	A	3	2
Flower planting (A2)	Matteson and Langelotto (2011)	USA	ug	3a; 3b	a	CI	4	3

Continuation Appendix Tab. 1:

Flower planting (A2)	Potts et al. (2003)	Israel	rg	3a	a; r	A	1	5
Pesticide use (C)	Lowenstein et al. (2015)	USA	ug	3a, 3d	a; r	A	3	2
Pesticide use (C)	Muratet and Fontaine (2015)	France	ug	3a; 3b	a	CI	7	10
Mowing (D)	Garbuzov and Ratnieks (2014)	UK	pug	3e	a	CI	7	1
Mowing (D)	Helden and Leather (2004)	UK	pug	3d	a; r	G	5	6
Mowing (D)	Lerman et al. (2018)	USA	ug	3a	a; r	G	3	4
Mowing (D)	Smith et al. (2015)	UK	pug	3e	a; r	G	5	8
Bird feeding (I)	Brittingham and Temple (1988)	USA	rg	2	a; m	CI	4	5
Bird feeding (I)	Fuller et al. (2008)	UK	pug	2	a; r	A	1	2
Bird feeding (I)	Harrison et al. (2010)	UK	rg	2	bm	CI	5	14
Bird feeding (I)	Plummer et al. (2013)	UK	rg	2	bm	CI	5	10
Bird feeding (I)	Robb et al. (2008b)	Ireland	rg	2	bm	CI	4	4

Appendix Tab. 2: Taxon (biodiversity measure), taxon codes (specific taxon), biodiversity measure codes (specific measure), number of studies (*N*), number of effect sizes (*k*), weighed mean effect sizes (*M*) and 95 % CI, (LL_M; UL_M) according to each forest plot (Fig.) and gardening practice (code) within the results of the present study. Considering flower planting (A2), parasitism rates (m) were excluded within the calculation of some weighed mean effect sizes. Considering mowing (D), different mowing frequencies and heights were compared in order to obtain effect sizes. Detailed information is given on the taxa included into insects (3e) and hymenopterans (3a) for affected calculations of weighed mean effect sizes. Taxon codes: 2: birds, 3a: hymenopterans, 3b: butterflies, 3d: true bugs, 3abcd: pollinators, 3e: general insects.

Forest plot (Fig.)	Gardening practice (code)	Taxon (biodiversity measure)	Taxon codes (specific taxon)	Biodiversity measure codes (specific measure)	Number of studies (*N*)	Number of effect sizes (*k*)	Weighed mean effect size (*M*)	Lower limit (95% CI) (LL_M)	Upper limit (95% CI) (UL_M)	Comments
5	Native plant planting (A1)	All Taxa	2; 3a; 3b; 3e; 3abcd	a; r	5	22	0.86	-1.42	3.15	
5	Native plant planting (A1)	Insects	3a; 3b; 3e; 3abcd	a; r	5	20	0.92	-1.58	3.42	
5	Native plant planting (A1)	Pollinators	3abcd	a; r	1	2	0.14	-0.19	0.46	
5	Native plant planting (A1)	Butterflies	3b	a; r	2	4	3.33	-4.11	10.77	
5	Native plant planting (A1)	Birds	2	a; r	1	3	0.86	-3.11	4.82	
5	Native plant planting (A1)	Insects (abundances)	3a; 3b; 3e	a	6	14	0.93	-2.00	3.85	
5	Native plant planting (A1)	Insects (richness)	3a; 3b; 3e	r	4	7	0.85	-3.61	5.31	
5	Native plant planting (A1)	Hymenopterans (abundances)	3a	a	1	2	0.34	0.07	0.62	Hymenopterans: Bees & wasps

Continuation Appendix Tab. 2:

5	Native plant planting (A1)	Butterflies (abundances)	3b	a	2	3	3.01	-6.71	12.74	
5	Native plant planting (A1)	Birds (richness)	2	r	1	2	0.29	-0.77	1.34	
5	Non-native plant planting (A1)	Insects (abundances)	3a; 3b	a	1	2	-1.32	-4.57	1.93	Insects: Bees & butterflies
6	Flower planting (A2)	Insects	3abcd; 3a; 3b	a; r; d; bm; m	7	25	0.31	-0.44	1.07	Insects: Pollinators, bees, wasps & butterflies
6	Flower planting (A2)	Pollinators	3abcd	a; d	2	2	0.76	-0.02	1.53	
6	Flower planting (A2)	Hymenopterans	1b	a; r; d; bm; m	6	23	0.28	0.12	0.45	Hymenopterans: Bees & wasps
6	Flower planting (A2)	Hymenopterans	3a	a; r; d; bm	6	19	0.48	0.35	0.62	Hymenopterans: Bees & wasps; without parasitism rates
6	Flower planting (A2)	Bees	3a (bees)	a; r; d; bm; m	6	19	0.34	0.12	0.55	
6	Flower planting (A2)	Bees	3a (bees)	a; r; d; bm	6	16	0.55	0.38	0.72	Without parasitism rates
6	Flower planting (A2)	Wasps	3a (wasps)	a; d; bm; m	2	4	0.11	-0.68	0.90	

Continuation Appendix Tab. 2:

6	Flower planting (A2)	Insects (abundances)	3abc, 3a, 3b	a	5	9	0.72	0.41	1.03	Insects: Pollinators, bees, wasps & butterflies
6	Flower planting (A2)	Hymenopterans (abundances)	3a	a	4	7	0.86	-0.29	2.01	Hymenopterans: Bees & wasps
6	Flower planting (A2)	Bees (abundances)	3a (bees)	a	4	6	1.03	0.49	1.57	
6	Flower planting (A2)	Bees (richness)	3a (bees)	r	3	4	1.86	1.25	2.48	
7	Floral abundances (A2)	Insects	3a, 3b	a; r; d; bm; m	4	9	0.30	-0.09	0.68	Insects: Bees, wasps & butterflies
7	Floral abundances (A2)	Insects	3a; 3b	a; r; d; bm	4	8	0.47	0.30	0.64	Insects: Bees, wasps & butterflies; without parasitism rates
7	Floral abundances (A2)	Hymenopterans	3a	a; r; d; bm; m	4	8	0.29	-0.18	0.75	Hymenopterans: Bees & wasps
7	Floral abundances (A2)	Hymenopterans	3a	a; r; d; bm	4	7	0.48	0.27	0.69	Hymenopterans: Bees & wasps; without parasitism rates
7	Floral abundances (A2)	Bees	3a (bees)	a; r; d; bm; m	4	7	0.28	-0.29	0.86	
7	Floral abundances (A2)	Bees	3a (bees)	a; r; d; bm	4	6	0.51	0.26	0.75	Without parasitism rates

Continuation Appendix Tab. 2:

7	Floral abundances (A2)	Insects (abundances)	3a, 3b	a	3	5	0.54	0.31	0.77	Insects: Bees, wasps & butterflies
7	Floral abundances (A2)	Hymenopterans (abundances)	3a	a	3	4	0.59	0.27	0.92	Hymenopterans: Bees & wasps
7	Floral abundances (A2)	Bees (abundances)	3a (bees)	a	3	3	0.74	0.29	1.18	
8	Floral richness (A2)	Insects	3a, 3abcd	a; r; d; bm; m	5	16	0.34	0.09	0.58	Insects: Pollinators, bees & wasps
8	Floral richness (A2)	Hymenopeterans	3a	a; r; d; bm; m	4	14	0.29	0.00	0.59	Hymenopterans: Bees & wasps
8	Floral richness (A2)	Hymenopterans	3a	a; r; d; bm	4	11	0.51	0.25	0.78	Hymenopterans: bees & wasps; without parasitism rates
8	Floral richness (A2)	Bees	3a (bees)	a; r; d; bm; m	4	11	0.39	0.00	0.77	
8	Floral richness (A2)	Bees	3a (bees)	a; r; d; bm	4	9	0.63	0.29	0.97	Without parasitism rates
8	Floral richness (A2)	Wasps	3a (wasps)	d; bm; m	1	3	0.07	-1.18	1.32	
8	Floral richness (A2)	Insects (abundances)	3a, 3abcd	a	2	3	1.34	0.50	2.17	Insects: Pollinators, bees
8	Floral richness (A2)	Hymenopterans (abundances)	3a	a	1	2	3.74	2.01	5.47	Hymenopterans: Bees & wasps

Continuation Appendix Tab. 2:

8	Floral richness (A2)	Bees (richness)	3a (bees)	r	3	3	1.63	1.15	2.11	
8	Floral richness (A2)	Insects	3a, 3abcd	a; r; d; bm	5	13	0.54	0.33	0.75	Insects: Pollinators, bees & wasps; without parasitism rates
9	Pesticide use (C)	Insects	3a; 3b; 3abcd	a; r	2	12	0.51	-4.21	5.24	Insects: Pollinators, bees & butterflies
9	Pesticide use (C)	Pollinators	3abcd	a; r	1	2	-0.02	-0.34	0.29	
9	Pesticide use (C)	Insects (abundances)	3a; 3b; 3abcd	a	2	11	0.53	-4.60	5.67	Insects: Pollinators, bees & butterflies
9	Pesticide use (C)	Bees (abundances)	3a	a	1	5	0.68	-9.19	10.55	
9	Pesticide use (C)	Butterflies (abundances)	3b	a	1	5	0.44	-6.14	7.01	
9	Insecticide use (C)	Insects	3a; 3b; 3abcd	a; r	2	4	-0.53	-3.50	2.44	Insects: Pollinators, bees & butterflies
9	Insecticide use (C)	Insects (abundances)	3a; 3b; 3abcd	a	2	3	-0.63	-3.97	2.71	Insects: Pollinators, bees & butterflies
9	Herbicide use (C)	Insect (abundances)	3a; 3b	a	1	2	-0.96	-6.56	4.65	Insects: Bees & butterflies

Continuation Appendix Tab. 2:

9	Fungicide use (C)	Insect (abundances)	3a; 3b	a	1	4	1.39	-2.94	5.72	Insects: Bees & butterflies
9	Snail pellet use (C)	Insect (abundances)	3a; 3b	a	1	2	1.73	0.52	2.94	Insects: Bees & butterflies
10	Mowing frequencies (D)	Insects	3a; 3d	a; r	2	4	-0.06	-2.30	2.18	Insects: Bees & grassland true bugs; weekly vs. every-two-weeks mowing
10	Mowing frequencies (D)	Bees	3a	a; r	1	2	-0.06	-0.14	0.03	Weekly vs. every-two-weeks mowing
10	Mowing frequencies (D)	Bees	3a	a; r	1	2	0.07	-0.23	0.37	Weekly vs. every-three-weeks mowing
10	Mowing frequencies (D)	Grassland true bugs	3d	a; r	1	2	-1.84	-18.92	15.24	Weekly vs. zero mowing
10	Mowing frequencies (D)	Grassland true bugs	3d	a; r	1	2	-0.51	-4.89	3.87	Weekly vs. every-two-weeks mowing
10	Mowing frequencies (D)	Grassland true bugs	3d	a; r	1	2	-0.55	-2.41	1.31	Weekly vs. every-six-weeks mowing
10	Mowing frequencies (D)	Insects (abundances)	3a	a	2	2	-0.05	-0.24	0.14	Insects: Bees & grassland true bugs; weekly vs. every-two-weeks mowing

Continuation Appendix Tab. 2:

10	Mowing frequencies (D)	Insects (richness)	3a	r	2	2	-0.62	-5.10	3.86	Insects: Bees & grassland true bugs; weekly vs. every-two-weeks mowing
11	Mowing heights (D)	Insects	3e	a; r	1	8	-0.38	-1.23	0.48	Two- vs. four-cm mowing
11	Mowing heights (D)	Insects (abundances)	3e	a	1	4	-0.04	-0.09	0.00	Two- vs. four-cm mowing
11	Mowing heights (D)	Insects (richness)	3e	r	1	4	-0.71	-1.67	0.25	Two- vs. four-cm mowing
12	Bird feeding (I)	Birds	2	a; r; bm; m	6	35	0.00	-1.50	1.50	
12	Winter bird feeding (I)	Birds	2	bm; m	3	19	0.00	-2.78	2.78	
12	Spring bird feeding (I)	Birds	2	bm; r; a	2	16	0.00	-0.37	0.36	
13	Bird feeding (I)	Birds (abundances)	2	a	2	2	0.30	-1.31	1.92	
13	Winter bird feeding (I)	Black-capped chickadees (winter survival rates)	2	m	1	4	1.28	-3.29	5.84	
14	Winter bird feeding (I)	Tits (breeding measures)	2	bm	2	14	-0.31	-3.53	2.91	

Continuation Appendix Tab. 2:

14	Winter bird feeding (I)	Tits (winter nest box occupancies)	2	bm	1	2	0.06	0.03	0.09	
14	Winter bird feeding (I)	Tits (lay dates)	2	bm (lay dates)	2	3	0.00	-2.73	2.73	
14	Winter bird feeding (I)	Tits (clutch sizes)	2	bm (clutch sizes)	2	3	-1.61	-16.18	12.96	
14	Winter bird feeding (I)	Tits (brood sizes)	2	bm (brood sizes)	2	3	0.20	-1.18	1.59	
14	Winter bird feeding (I)	Tits (fledging successes)	2	bm (fledging successes)	2	3	-0.18	-0.28	-0.07	
15	Spring bird feeding (I)	Tits (breeding measures)	2	bm	1	14	-0.05	-0.28	0.19	
15	Spring bird feeding (I)	Blue tits (breeding measures)	2	bm	1	11	0.98	-3.74	5.71	
15	Spring bird feeding (I)	Great tits (breeding measures)	2	bm	1	7	-0.07	-0.51	0.37	
15	Spring bird feeding (I)	Tits (lay dates)	2	bm (lay dates)	1	2	0.01	-0.05	0.07	
15	Spring bird feeding (I)	Tits (clutch sizes)	2	bm (clutch sizes)	1	2	-0.02	-0.03	-0.01	
15	Spring bird feeding (I)	Tits (incubation periods)	2	bm (incubation periods)	1	6	-0.10	-0.58	0.38	

Continuation Appendix Tab. 2:

15	Spring bird feeding (I)	Tits (brood sizes)	2	bm (brood sizes)	1	4	-0.01	-0.09	0.06	