

Biodiversity conservation in agriculture requires a multi-scale approach

David J. Gonthier, Katherine K. Ennis, Serge Farinas, Hsun-Yi Hsieh, Aaron L. Iverson, Péter Batáry, Jörgen Rudolphi, Teja Tschardt, Bradley J. Cardinale and Ivette Perfecto

Proc. R. Soc. B 2014 **281**, 20141358, published 6 August 2014

Supplementary data

["Data Supplement"](#)

<http://rspb.royalsocietypublishing.org/content/suppl/2014/08/05/rspb.2014.1358.DC1.html>

References

[This article cites 64 articles, 9 of which can be accessed free](#)

<http://rspb.royalsocietypublishing.org/content/281/1791/20141358.full.html#ref-list-1>

Subject collections

Articles on similar topics can be found in the following collections

[ecology](#) (1782 articles)

[environmental science](#) (298 articles)

Email alerting service

Receive free email alerts when new articles cite this article - sign up in the box at the top right-hand corner of the article or click [here](#)



Research

Cite this article: Gonthier DJ *et al.* 2014
Biodiversity conservation in agriculture requires
a multi-scale approach. *Proc. R. Soc. B* **281**:
20141358.
<http://dx.doi.org/10.1098/rspb.2014.1358>

Received: 3 June 2014
Accepted: 16 July 2014

Subject Areas:
ecology, environmental science

Keywords:
agri-environmental schemes, agroecology,
sustainable agriculture, intensification,
landscape complexity, organic farming

Author for correspondence:
David J. Gonthier
e-mail: gonthier.david@gmail.com

[†]Present address: Department of Environmental
Science, Policy, and Management, University of
California, 130 Mulford Hall, Berkeley,
CA 94720, USA.

Electronic supplementary material is available
at <http://dx.doi.org/10.1098/rspb.2014.1358> or
via <http://rspb.royalsocietypublishing.org>.

Biodiversity conservation in agriculture requires a multi-scale approach

David J. Gonthier^{1,†}, Katherine K. Ennis², Serge Farinas³, Hsun-Yi Hsieh¹,
Aaron L. Iverson³, Péter Batáry⁴, Jörgen Rudolphi⁵, Teja Tschamtké⁴,
Bradley J. Cardinale¹ and Ivette Perfecto¹

¹School of Natural Resources and Environment, University of Michigan, 440 Church Street, Ann Arbor,
MI 48109, USA

²Department of Environmental Studies, University of California, 156 High Street, Santa Cruz, CA 95064, USA

³Department of Ecology and Evolutionary Biology, University of Michigan, 830 N University Avenue, Ann Arbor,
MI 48109, USA

⁴Agroecology, Department of Crop Sciences, University of Göttingen, Grisebachstrasse 6, 37077 Göttingen,
Germany

⁵Department of Ecology, Swedish University of Agricultural Sciences, PO Box 7044, 75007 Uppsala, Sweden

Biodiversity loss—one of the most prominent forms of modern environmental change—has been heavily driven by terrestrial habitat loss and, in particular, the spread and intensification of agriculture. Expanding agricultural land-use has led to the search for strong conservation strategies, with some suggesting that biodiversity conservation in agriculture is best maximized by reducing local management intensity, such as fertilizer and pesticide application. Others highlight the importance of landscape-level approaches that incorporate natural or semi-natural areas in landscapes surrounding farms. Here, we show that both of these practices are valuable to the conservation of biodiversity, and that either local or landscape factors can be most crucial to conservation planning depending on which types of organisms one wishes to save. We performed a quantitative review of 266 observations taken from 31 studies that compared the impacts of localized (within farm) management strategies and landscape complexity (around farms) on the richness and abundance of plant, invertebrate and vertebrate species in agro-ecosystems. While both factors significantly impacted species richness, the richness of sessile plants increased with less-intensive local management, but did not significantly respond to landscape complexity. By contrast, the richness of mobile vertebrates increased with landscape complexity, but did not significantly increase with less-intensive local management. Invertebrate richness and abundance responded to both factors. Our analyses point to clear differences in how various groups of organisms respond to differing scales of management, and suggest that preservation of multiple taxonomic groups will require multiple scales of conservation.

1. Introduction

One of society's most pressing challenges is to slow the rate of global biodiversity loss and extinction [1–5]. There is now overwhelming evidence that the loss of species impacts the functioning of ecosystems [6] and that many services provided by species have important economic value [7,8]. Much conservation research has therefore focused on where biodiversity is being lost most rapidly and where the loss of biodiversity will have the most immediate consequences. Of the drivers of global biodiversity loss, the widespread conversion of land to monoculture crop production and the intensification of local agricultural practices, such as fertilizer and pesticide use, are considered to be among the most damaging to biodiversity [1,9–14]. In turn, the services that species provide related to pest control, pollination and nutrient cycling that benefit agricultural production and sustainability could be compromised [10]. With ever-increasing global demands for agricultural production of food and fuel, additional stresses

on species in and surrounding agricultural land are inevitable [1,15]. As such, conserving the biodiversity that lives in agricultural lands has become a major focus of much conservation policy.

But how to best conserve biodiversity in agricultural fields remains a major challenge. Programmes in numerous countries have attempted to reduce the severity of agriculture's negative influence on biodiversity by paying farmers to reduce management intensity through reduced pesticide inputs, synthetic fertilizer inputs or by converting farms to organic practices [12,16–19]. Several syntheses suggest that reduction in local management intensity does conserve biodiversity [12,16], but other empirical research has failed to support these claims [17,20,21]. Proponents of meta-population and meta-community theory are quick to point out that reduced intensity of one small farm may do little to conserve species with large range sizes or species that require adjoining subpopulations in the surrounding landscape [10,22]. Furthermore, reduced intensity may provide little benefit if a farm is surrounded by a landscape of high-input intensive farming, because poor species pools in the desolate landscapes may limit the colonization of the wildlife friendly farm [10,23]. Some propose that agriculture's larger scale effects, the homogenization of entire landscapes [24], may be the primary factor driving biodiversity loss [10,23,25,26]. In order to conserve biodiversity in agriculture, we may need to incorporate more natural and semi-natural habitats in areas surrounding farms and/or maintain high habitat diversity in agricultural landscapes [19,27–29].

Understanding how and which species respond to different scales of intensification will aid conservation strategies. Previous research suggests interactions between local and landscape strategies are important [10], but that not all taxonomic groups and not all types of agriculture consistently exhibit these patterns [29]. This implies that other mechanisms may also be at work. One hypothesis suggests that because species vary in many functional traits, such as mobility, range size, dispersal capability and sensitivity to agricultural activities, biodiversity in different taxonomic groups may respond to different scales of agricultural intensification [10,22]. For example, while some plants may have high seed-dispersal capability, they are all non-mobile organisms. Therefore, the application of herbicide within farms may largely eliminate many species from farmlands. Other organisms, such as mammals and birds, are capable of foraging across many habitats and over a large spatial area. These species may require landscape-level features to persist in farmlands. Although some empirical studies have supported this hypothesis [22,27], there is need for a quantitative synthesis of the published literature that simultaneously evaluates local- and landscape-level strategies for a broad range of taxonomic groups.

We performed a quantitative review to investigate the influence of local management intensification and landscape complexity on biodiversity in agriculture. We reviewed 31 field studies that provided 266 observations of species richness and abundance. Observations were defined as the within study measurement of species richness or abundance of plants, invertebrate or vertebrate animals taken across agro-ecosystems that varied simultaneously by low- versus high-local management intensities, and that varied in the surrounding landscape complexity. Low-intensity agro-ecosystems consisted of farms that were certified organic, had reduced chemical inputs or reduced planting and/or

grazer densities when compared with high-intensity conventional agro-ecosystems. Landscape complexity was defined as the proportion of natural and semi-natural areas (non-crop lands) or the variety of different habitat types (measured as Shannon's Diversity Index) in landscapes surrounding farms. We predicted that sessile organisms (e.g. plants) would be influenced more by local factors than by landscape complexity due to their low mobility that makes them susceptible to disturbances at small spatial scales. We predicted that more mobile organisms would be buffered against local management intensification and respond more to landscape complexity. As we show next, both local and landscape strategies are needed to promote plant, invertebrate and vertebrate biodiversity because different groups respond more consistently to different scales.

2. Material and methods

(a) Selection of studies

We conducted an ISI Web of Science literature search of studies that compared species richness and abundance in low- and high-intensity agricultural fields that were nested within a gradient of landscape complexity (last search 13 January 2012; electronic supplementary material, S1 text). In addition to our primary search, we also reviewed the reference sections of several recent reviews and meta-analyses [10,26,29] and we also encountered and obtained data from two studies [30,31] via data requests for other studies. In total, we reviewed 822 published studies. Inclusion of a study within our quantitative review was contingent on the following criteria: (i) the study must have collected empirical data on species richness within agriculture, (ii) the study must have compared categorically defined local-scale factors related to agricultural management intensity, and (iii) the study must have included variation in landscape-scale factors related to landscape complexity in each sampling site. These measurements must have been taken at a minimum scale of 1.96 ha (approx. 250 m radius surrounding the sampling site).

Of the 822 published studies reviewed, 44 fitted our criteria (33 from search, 11 from references). We were unable to obtain the data from nine studies. Some studies reported results from the same datasets; see references [32–36]. In the end, we obtained data from 31 studies [30–62]. We were unable to recover all data from some highly collaborative studies that spanned multiple countries and research groups because of data-sharing issues across the large-scale projects [34,35,61]. For richness, we obtained a total of 71 observations for local factors and 71 observations for landscape factors (from 31 studies). We obtained observations of abundance, activity abundance or per cent cover for a total of 63 observations for local factors and 63 observations for landscape factors (28 studies; see figure 2 for taxonomic group sample sizes). Two outliers were removed from the dataset for analysis of local management statistical models to improve the model fit and the normality of the data; for plant richness [35] and plant abundance [41]. The results of our analysis were not influenced by the inclusion or exclusion of these outliers.

(b) Local management factors

All local-scale management factors fell under a comparison of a low-intensive form of agricultural practice versus a high-intensive form of agricultural practice. Low-intensity agriculture consisted of certified organic practices, practices in compliance with an agri-environment scheme aimed to benefit the environment or biodiversity (i.e. the planting of flower strips in field margins [17]),

and intensified agriculture in which chemical inputs, plant densities or grazing densities were low. High-intensity farms had conventional management levels of chemical inputs and planting or grazing densities that were always greater than the low-intensity farms that they were compared against in each study. For each measurement of species richness or abundance (y) within a study, we calculated the local management effect size on biodiversity as a log response ratio $LR_M = \ln(y_L/y_H)$, where y_L is the mean of biodiversity in low-intensity farms and y_H is the mean of biodiversity in high-intensity farms. Log response ratios are unitless metrics that allow us to determine whether there is a proportional difference between mean levels of species richness in low- and high-intensity farms [63,64]. In studies from Sweden [32,46,56,57,59], study designs were such that low- and high-intensity farms were paired to control for variation in management type and location. Log response ratios compare unpaired means between low- and high-intensity farms, therefore for these Swedish studies, the effect sizes calculated had less power than if we were able to maintain a paired design within our analysis.

(c) Landscape factors

Within each study, all sites also fell along a landscape complexity gradient. Both the diversity of habitat types and the per cent non-crop area are considered important components of landscape complexity across the literature sampled [10]. Thus, we defined the landscape factors per cent natural area, semi-natural area and woodlands as per cent non-crop area. We also defined the inverted percentage of arable land, croplands, managed lands and agriculture as the per cent of non-crop area and assumed all measures of non-crop area correlated with landscape complexity. We also included the diversity of habitat types (measured as the Shannon's Index) in the analysis even though it is measured on a different range of values (0 to ∞) than the percentage non-crop area. We excluded landscape measurements of mean field size, length of habitat boundaries, per cent grasslands and per cent intensive agricultural area because they do not intuitively correlate with landscape complexity. In one case, we included a measure of per cent grassland because authors stated clearly that it was strongly positively correlated with the diversity of habitat types [55]. If multiple landscape factors met our criteria within a study, we included the factor that best explained variation in richness or abundance. To calculate effect sizes, we calculated correlation coefficients (R) that related y (richness or abundance) to the measure of landscape complexity, and then standardized the coefficients to Fisher's Z as: $Z_L = 0.5 \times \ln(1 + R_L)/(1 - R_L)$ [63], where Z_L is Fisher's Z and R_L is the correlation coefficient of y versus landscape complexity. Quantitative reviews comparing continuous variables often use R or Fisher's Z as an effect size, because they are intuitively interpreted and they are standardized to take into account the original scales of different metrics [63].

(d) Analysis

We adjusted the analysis to consider non-independence within and between studies. To account for the fact that some studies had multiple observations, we created a block by study (random effect of study). For two studies that reported two sampling events per site [37,52], we averaged means before calculating local management LR_M and averaged landscape factor Z_L across the two time points. To account for the different cropping types reported, we considered cropping type within each study (cereal, mixed, vegetable/fruit and pasture/meadow). Observations also varied by geographical location and by research group (affiliation); therefore, we created a random effect of country to largely account for these effects. However, the country-random effect generally had a very small estimated variance, suggesting that it did not explain significant variance in effect sizes. The random effect of

country also often resulted in problems related to over-fitting the model; therefore, we proceeded without this random factor within models. To account for the differences between landscape factors, we considered landscape factor type (percentage non-crop area or habitat diversity) and landscape factor scale that ranged from 250 to 5000 m radii surrounding site locations. Observations of richness and abundance within each taxonomic group did not always contain variation in the other fixed effects measured, therefore we were unable to include interactions between fixed effects in statistical models.

We performed general linear mixed models (GLMM) to determine whether mean local management (LR_M) and landscape (Z_L) effect sizes differed from 0 and to compare the differences between taxonomic groups [65]. For all GLMMs, we performed type III F -tests of significance for main effects with maximum likelihood to estimate the fixed effect parameters and variance of the random effects. All models included the random effect of study. For the local management models, we used the response variable LR_M with taxonomic group and crop type as fixed effects. For landscape models, we used the response variable Z_L with taxonomic group, crop type, landscape factor type and landscape factor scale (covariate) as fixed effects. We performed model selection using likelihood ratio tests to exclude fixed effects that did not improve model fit [65]. We used the final GLMM models to estimate mean and 95% confidence intervals (CIs) of each effect size with the function `EMMEANS` in SPSS (20.0). Mean effect sizes that were significantly more positive or more negative than 0 were interpreted as significant at $\alpha = 0.05$. In addition to unweighted effect sizes, we also ran analyses with effect sizes weighted by the inverse of the variance [64]. To determine whether publication bias had the potential to influence our results, we performed correlations between sample size and effect size and calculated fail safe values (electronic supplementary material, S1 text and table S1). We present unweighted models because discrepancies between unweighted and weighted models were small (for description of weighted models, see the electronic supplementary material, table S2 and figure S1), and unweighted models allow observations with few large plots to have the same effect as observations with many small plots. We conducted all statistical analysis in SPSS (20.0).

3. Results

Our results consistently show that both local management and landscape complexity impact species richness (figure 1a and table 1). However, the importance of each factor differs among the three taxonomic groups examined (figure 2a). Overall, 52 out of 70 (74%) observations showed that low-intensity farms had more species than high-intensity farms (figure 1a). Mean overall richness, estimated across all organisms, was 40% higher in low-intensity relative to high-intensity farms (figure 1a). While plant and invertebrate richness was 92% and 21% higher in low-intensity relative to high-intensity farms, respectively, vertebrate richness did not differ significantly among local management types (figure 2a). Plant richness responded more strongly to local management intensification than did invertebrate (*post hoc test*, mean difference (\pm s.e.) = 0.46 ± 0.12 , $p = 0.001$) and vertebrate richness (0.48 ± 0.16 , $p = 0.012$). There was no difference in effect size between invertebrate and vertebrate richness (0.02 ± 0.14 , $p > 0.999$).

Forty-seven out of 71 (66%) observations showed a positive relationship between landscape complexity and species richness within farms (figure 1a and table 1). The mean correlation between plant richness and landscape complexity was not significantly positive (figure 2c). By contrast, both

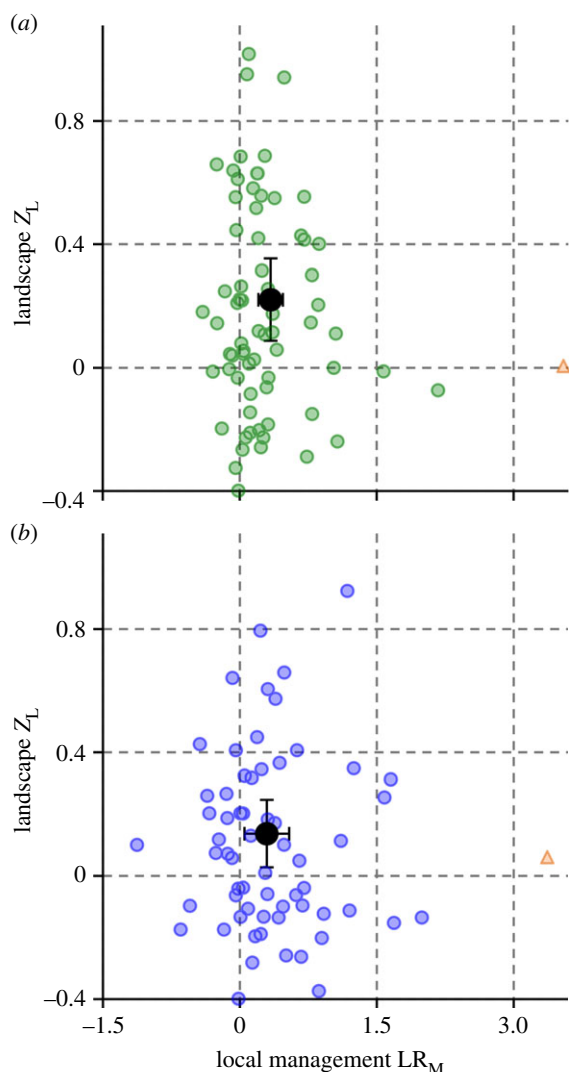


Figure 1. Scatterplots of estimated marginal means and 95% CI (black points) of local management (LR_M) and landscape complexity (Z_L) effect sizes for richness ($n = 70$ observations, 31 studies) (a) and abundance (b) ($n = 62$ observations, 28 studies). Outliers were removed from local management analysis, but remained in the landscape complexity analysis (triangular points). Summary statistics of the GLMM used to estimate marginal means and 95% CIs are available in table 1 for richness and table 2 for abundance. (Online version in colour.)

invertebrate and vertebrate animals had significantly positive mean correlations, indicating that species richness of these groups increased as a function of increasing landscape complexity (figure 2c). Although vertebrate and invertebrate richness significantly responded to landscape complexity and plants did not, there was no significant difference between the effect sizes of plant, invertebrate and vertebrate richness.

The analysis of the abundance of the taxonomic groups also revealed important patterns. For the local management scale, overall 44 of 63 (69.8%) observations found higher abundance in low-intensity relative to high-intensity farms (table 2 and figure 1b). Although the mean overall abundance, estimated across all organisms, was 27% higher in low-intensity relative to high-intensity farms, within taxonomic groups, vertebrate and plant abundance did not differ between the two local management types (figure 2b). Only invertebrate abundance was significantly greater in low- relative to high-intensity farming. The overall mean correlation between abundance and landscape complexity was significantly positive, with

36 of 64 (56.2%) observations positively correlated (table 2 and figure 1b). However, within taxonomic group, only invertebrate abundance was significantly and positively correlated with landscape complexity (figure 2d).

The crop types examined within studies also partially explained the differences in richness and abundance between low- and high-intensity farms (tables 1 and 2; electronic supplementary material, figure S2a,c). However, only one significant pairwise comparison was found between farming types; there was greater total abundance (all taxonomic groups pooled) in less-intensive relative to high-intensive farms in cereal cropping systems compared to pasture/meadow systems (*post hoc test*, mean difference (\pm s.e.) = 0.5 ± 0.2 , $p = 0.024$; electronic supplementary material, figure S2b). All other pairwise comparisons of cropping types for abundance and richness were not statistically significant (electronic supplementary material, figure S2a–d). Landscape factor type (per cent non-crop habitat and habitat diversity) and landscape factor scale (250–5000 m radii) never significantly explained variation in local or landscape effect sizes for richness and abundance (tables 1 and 2).

4. Discussion

This is, to our knowledge, the first broad-scale meta-analysis to show that local and landscape conservation strategies in agriculture affect biodiversity of various taxonomic groups in different ways. Recent syntheses either concentrate on narrow taxonomic focus (e.g. arthropods; [28,66]) or solely evaluate the interactive effect of landscape complexity on local management and do not assess the actual landscape effects on biodiversity [29,67]. Our study advances these works by reviewing only those studies that simultaneously pair landscape effects with local management effects and also by evaluating broad taxonomic groups. Given this novel approach, we reveal that plant species more consistently respond to local factors than landscape factors, and vertebrate species more consistently respond to landscape factors than local factors.

Plant richness was more affected by local management intensification than was invertebrate and vertebrate richness (figure 2a). The strong decline in plant richness in intensively managed farms is probably the by-product of agricultural practices designed to eliminate the abundance of weedy crop competitors. Herbicide application, synthetic fertilization of crop plants and tilling can have direct or indirect negative effects on plant diversity within agricultural systems [10,62]. Though the reductions in arable ‘weedy’ plant diversity may seem insignificant to conservation agendas, many of these ‘weedy’ species are categorized as threatened on the International Union for Conservation of Nature red list (e.g. 38% of arable plants in Germany are red listed species [10]). Thus, the importance of this finding should not be negated. What is, perhaps, more surprising is that there was no significant, positive correlation between plant biodiversity and landscape complexity. Increased landscape complexity may promote plant diversity within farms through seed rain from neighbouring non-crop habitats [62], but if local management within farms is frequent and intensive, those seeds may never reach vegetative states. It is also important to consider that the value of the mean correlation between plant richness and landscape complexity was not significantly different from the richness of invertebrates and vertebrates; however, invertebrates had three times more observations

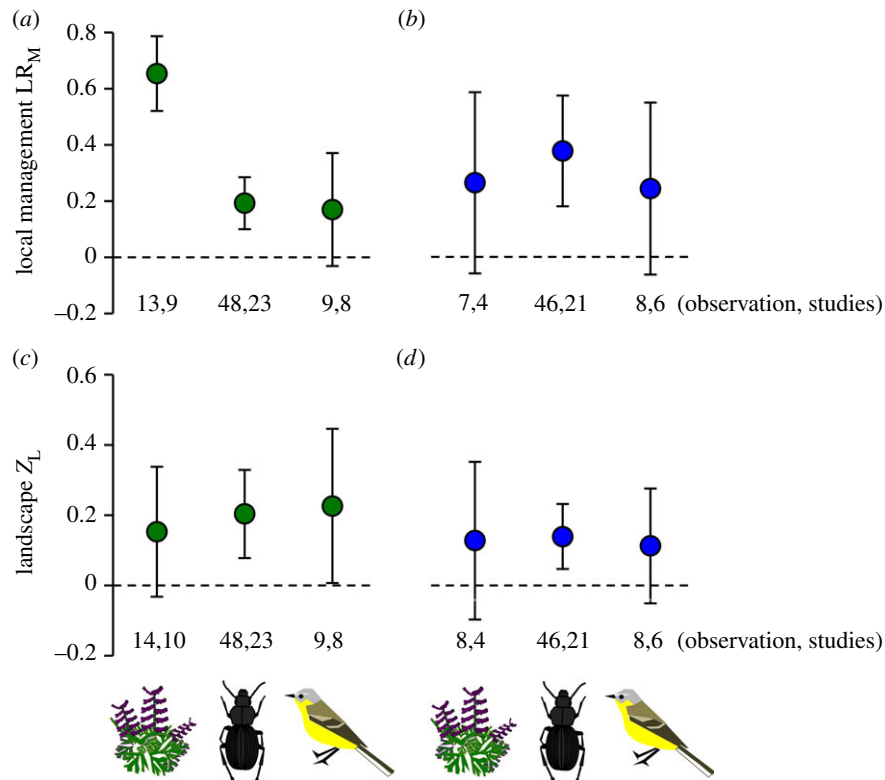


Figure 2. Estimated marginal means and 95% CI of plant, invertebrate and vertebrate (from left to right) effect sizes for local management (LR_M) for richness (a) and abundance (b), and landscape complexity (Z_L) effect sizes for richness (c) and abundance (d). Summary statistics of the GLMM used to estimate marginal means and 95% CIs are available in table 1 for richness and table 2 for abundance. (Online version in colour.)

Table 1. Statistical models for local management and landscape complexity effect sizes on species richness. (For both models, a random effect of study was included. Statistical models were used to estimate mean and 95% CI of effect sizes for overall responses (figure 1a) and taxonomic groups (figure 2a,c).)

	d.f. ^a	F	p-value
local management LR_M richness			
intercept	1,49	25.8	<0.001
taxonomic group	2,65	8.0	0.001
crop type	3,39	3.2	0.035
landscape Z_L richness			
intercept	1,33	10.4	0.003
taxonomic group	2,56	0.1	0.874
crop type	3,34	2.3	0.098

^ad.f. = numerator, denominator.

than did plants. For plants, a larger sample size may reduce the variance around the mean of the plant effect size and could reveal significant response to landscape factors.

Another goal of intensification is to eliminate arthropod pests. Intentional spraying for arthropod pests might help explain lower species richness and abundance of invertebrates in more intensive farms, as pesticides have both intentional and non-intentional consequences on biological communities. Our results appear to reflect largely non-intentional negative consequences of intensification. Roughly, 77% (37 out of 44) of the observations of invertebrate biodiversity included in our analyses consisted of groups, such

Table 2. Statistical models for local management and landscape complexity effect sizes on total abundance. (For both models, a random effect of study was included. Statistical models were used to estimate mean and 95% CI of effect sizes for overall responses (figure 1b) and taxonomic groups (figure 2b,d).)

	d.f. ^a	F	p-value
local management LR_M abundance			
intercept	1,30	6.1	0.020
taxonomic group	2,55	0.3	0.756
crop type	3,24	3.2	0.043
landscape Z_L abundance			
intercept	1,63	5.1	0.027
taxonomic group	2,63	0.3	0.722
crop type	3,63	2.3	0.088

^ad.f. = numerator, denominator.

as bees, spiders and carabid beetles, all of which are not generally considered crop pests. In fact, these groups are often associated with important pollination and natural pest control services [10,26,28]. Thus, our results suggest that the losses of species in agricultural plots owing to local management and landscape-level intensification are very often associated with the loss of beneficial invertebrate diversity.

The pattern of vertebrate biodiversity is more difficult to explain. The lack of a response of vertebrate biodiversity to local management, coupled with the significant response to landscape complexity, is potentially explained by the high mobility of these taxa (e.g. mammals and birds).

High mobility allows them to experience the landscape at a larger scale and capture resources across larger areas in habitats outside of crop fields [10]. Greater mobility, in turn, has the potential to buffer these species from small-scale changes in local management [10]. However, a lack of statistical power might also explain the response of vertebrates. It should be noted that the magnitude of the response of vertebrate richness to local management intensification was no different than that of invertebrates, but the sample size of invertebrates was five times as large (figure 2a). A larger sample size may reduce the variance around the mean of the vertebrate effect size and reveal a significant response to local management factors.

The abundance analysis provides insights into the response of biodiversity to local and landscape conservation strategies. For invertebrates, higher levels of both richness and abundance in systems that are farmed less intensively, or that are surrounded by more complex habitats, suggests increases in richness could be driven by an increase in the overall abundance of invertebrates, as it is well known that the discovery of species is proportional to the number of individuals sampled [68]. By contrast, the abundance of plants and vertebrates did not vary with local or landscape factors, which suggests that higher levels of richness was independent of any impacts of factors on population sizes for these two groups. However, the non-significant responses of plants and vertebrates may also be owing to smaller sample sizes.

Although this study reveals clear patterns relating the local- and landscape-level effects on biodiversity, the literature included in our review does have several limitations that should be kept in mind. While many important metrics of biodiversity are well described, including functional diversity, phylogenetic diversity, evenness and other metrics of diversity, the body of literature describing local and landscape strategies for conservation in agriculture focuses on species richness and abundance. For that reason, our analysis was limited to richness and abundance. Vertebrates are poorly represented relative to other types of organisms in our dataset and most of the studies reviewed come from agro-ecosystems in Europe and the USA. We did find that our conclusions are relatively robust to select data deficiencies and the potential of publication bias, issues that are always a limitation of data

syntheses (electronic supplementary material, S1 text and table S1). Despite these limitations, the data presently available clearly show that both local management and landscape-scale strategies are important to conserving biodiversity in agriculture, as each scale influences a different set of species.

Our findings have major implications for conservation policies in agricultural landscapes. Policy strategies for conserving biodiversity in agriculture have historically focused on changing local management practices [17,18]. While these efforts are most certainly helpful for conserving certain groups of organisms, policy-makers and practitioners need to consider a broader agenda that focuses on both local and landscape strategies. Our results suggest continuing strategies that reduce the intensity of local farming practices will promote some taxonomic groups. At the same time, careful regional planning is also needed to conserve high-quality habitats that maintain heterogeneity in agricultural landscapes. As has been echoed by many other researchers, the best first steps may be to conserve existing complex agricultural landscapes and implementing changes to local management practices in regions with little remaining wild lands [10,18,25,27,28]. Developing conservation plans at large spatial areas may present major challenges for future policy-makers owing to the difficulty in coordinating multiple land ownerships within the same landscape [22]. Integrating multiple scales of conservation may also maximize the crop pollination, natural pest control and nutrient cycling services that are facilitated by biodiversity [10,11,26,28]. However, further analysis of multi-scale conservation is needed to evaluate the links between minimizing the multi-scale impacts of the agricultural industry on biodiversity and maximizing nature's services to that industry.

Acknowledgements. We thank all authors who shared data pertinent to this synthesis. We thank J. Vandermeer, D. Karp and anonymous reviewers for critiques of this work.

Data accessibility. Literature search and effect sizes (Dataset S1): dryad doi: 10.5061/dryad.mm6f4.

Funding statement. Funding was provided by an NSF grant (1050680) to S. Andelman and J. Parrish: Dimensions of Biodiversity Distributed Graduate Seminar, NSF-GRF (DGE-0718128) to D.J.G., NSF grant (DEB-1309786) to D.J.G. and I.P., and DFG (BA 4438/1-1) to P.B.

References

1. Tilman D *et al.* 2001 Forecasting agriculturally driven global environmental change. *Science* **292**, 281–284. (doi:10.1126/science.1057544)
2. Vitousek PM, Mooney HA, Lubchenco J, Melillo JM. 1997 Human domination of earth's ecosystems. *Science* **277**, 494–499. (doi:10.1126/science.277.5325.494)
3. Loreau M *et al.* 2001 Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science* **294**, 804–808. (doi:10.1126/science.1064088)
4. Foley JA *et al.* 2005 Global consequences of land use. *Science* **309**, 570–574. (doi:10.1126/science.1111772)
5. Sala OE *et al.* 2000 Biodiversity: global biodiversity scenarios for the year 2100. *Science* **287**, 1770–1774. (doi:10.1126/science.287.5459.1770)
6. Cardinale BJ *et al.* 2012 Biodiversity loss and its impact on humanity. *Nature* **486**, 49–67. (doi:10.1038/nature11148)
7. Daily GC. 1997 *Nature's services: societal dependence on natural ecosystems*. Washington, DC: Island Press.
8. Perrings C, Jackson L, Bawa K, Brussaard L, Brush S, Gavin T, Papa R, Pascual U, De Ruiter P. 2006 Biodiversity in agricultural landscapes: saving natural capital without losing interest. *Conserv. Biol.* **20**, 263–264. (doi:10.1111/j.1523-1739.2006.00390.x)
9. Green RE, Cornell SJ, Scharlemann JPW, Balmford A. 2005 Farming and the fate of wild nature. *Science* **307**, 550–555. (doi:10.1126/science.1106049)
10. Tschamtké T, Klein AM, Krüess A, Steffan-Dewenter I, Thies C. 2005 Landscape perspectives on agricultural intensification and biodiversity: ecosystem service management. *Ecol. Lett.* **8**, 857–874. (doi:10.1111/j.1461-0248.2005.00782.x)
11. Perfecto I, Vandermeer J, Wright A. 2009 *Nature's matrix: linking agriculture, conservation, and food sovereignty*. London, UK: Earthscan.
12. Attwood SJ, Maron M, House APN, Zammit C. 2008 Do arthropod assemblages display globally consistent responses to intensified agricultural land use and management? *Glob. Ecol. Biogeogr.* **17**, 585–599. (doi:10.1111/j.1466-8238.2008.00399.x)
13. Clough Y *et al.* 2011 Combining high biodiversity with high yields in tropical agroforests. *Proc. Natl Acad. Sci. USA* **108**, 8311–8316. (doi:10.1073/pnas.1016799108)
14. Chamberlain DE, Fuller RJ, Bunce RG, Duckworth JC, Shrubbs M. 2000 Changes in the abundance of farmland birds in relation to the timing of

- agricultural intensification in England and Wales. *J. Appl. Ecol.* **37**, 771–788. (doi:10.1046/j.1365-2664.2000.00548.x)
15. Kremen C, Miles A. 2012 Ecosystem services in biologically diversified versus conventional farming systems: benefits, externalities, and trade-offs. *Ecol. Soc.* **17**, 40. (doi:10.5751/ES-05035-170440)
 16. Bengtsson J, Ahnström J, Weibull A-C. 2005 The effects of organic agriculture on biodiversity and abundance: a meta-analysis. *J. Appl. Ecol.* **42**, 261–269. (doi:10.1111/j.1365-2664.2005.01005.x)
 17. Kleijn D, Sutherland WJ. 2003 How effective are European agri-environment schemes in conserving and promoting biodiversity? *J. Appl. Ecol.* **40**, 947–969. (doi:10.1111/j.1365-2664.2003.00868.x)
 18. Kleijn D, Rundlof M, Scheper J, Smith HG, Tscharntke T. 2011 Does conservation on farmland contribute to halting the biodiversity decline? *Trends Ecol. Evol.* **26**, 474–481. (doi:10.1016/j.tree.2011.05.009)
 19. Whittingham MJ. 2011 The future of agri-environment schemes: biodiversity gains and ecosystem service delivery? *J. Appl. Ecol.* **48**, 509–513. (doi:10.1111/j.1365-2664.2011.01987.x)
 20. Kleijn D *et al.* 2006 Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecol. Lett.* **9**, 243–254. (doi:10.1111/j.1461-0248.2005.00869.x)
 21. Kleijn D, Berendse F, Smit R, Gilissen N. 2001 Agri-environment schemes do not effectively protect biodiversity in Dutch agricultural landscapes. *Nature* **413**, 723–725. (doi:10.1038/35099540)
 22. McKenzie AJ, Emery SB, Franks JR, Whittingham MJ. 2013 Landscape-scale conservation: collaborative agri-environment schemes could benefit both biodiversity and ecosystem services, but will farmers be willing to participate? *J. Appl. Ecol.* **50**, 1274–1280. (doi:10.1111/1365-2664.12122)
 23. Ricketts TH, Daily GC, Ehrlich PR, Fay JP. 2001 Countryside biogeography of moths in a fragmented landscape: biodiversity in native and agricultural habitats. *Conserv. Biol.* **15**, 378–388. (doi:10.1046/j.1523-1739.2001.015002378.x)
 24. Meehan TD, Werling BP, Landis DA, Gratton C. 2011 Agricultural landscape simplification and insecticide use in the Midwestern United States. *Proc. Natl Acad. Sci. USA* **108**, 11 500–11 505. (doi:10.1073/pnas.1100751108)
 25. Concepción ED *et al.* 2012 Interactive effects of landscape context constrain the effectiveness of local agri-environmental management. *J. Appl. Ecol.* **49**, 695–705. (doi:10.1111/j.1365-2664.2012.02131.x)
 26. Chaplin-Kramer R, O'Rourke ME, Blitzer EJ, Kremen C. 2011 A meta-analysis of crop pest and natural enemy response to landscape complexity. *Ecol. Lett.* **14**, 922–932. (doi:10.1111/j.1461-0248.2011.01642.x)
 27. Gabriel D, Sait SM, Hodgson JA, Schmutz U, Kunin WE, Benton TG. 2010 Scale matters: the impact of organic farming on biodiversity at different spatial scales. *Ecol. Lett.* **13**, 858–869. (doi:10.1111/j.1461-0248.2010.01481.x)
 28. Kennedy CM *et al.* 2013 A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. *Ecol. Lett.* **16**, 584–599. (doi:10.1111/ele.12082)
 29. Batáry P, Báldi A, Kleijn D, Tscharntke T. 2011 Landscape-moderated biodiversity effects of agri-environmental management: a meta-analysis. *Proc. R. Soc. B* **278**, 1894–1902. (doi:10.1098/rspb.2010.1923)
 30. Jonason D, Andersson GKS, Ockinger E, Rundlöf M, Smith HG, Bengtsson J. 2011 Assessing the effect of the time since transition to organic farming on plants and butterflies. *J. Appl. Ecol.* **48**, 543–550. (doi:10.1111/j.1365-2664.2011.01989.x)
 31. Jonason D, Smith HG, Bengtsson J, Birkhofer K. 2013 Landscape simplification promotes weed seed predation by carabid beetles (Coleoptera: Carabidae). *Landscape Ecol.* **28**, 487–494. (doi:10.1007/s10980-013-9848-2)
 32. Weibull A-C, Bengtsson J, Nohlgren E. 2000 Diversity of butterflies in the agricultural landscape: the role of farming system and landscape heterogeneity. *Ecography* **23**, 743–750. (doi:10.1034/j.1600-0587.2000.230611.x)
 33. Weibull A-C, Östman Ö, Granqvist Å. 2003 Species richness in agroecosystems: the effect of landscape, habitat and farm management. *Biodivers. Conserv.* **12**, 1335–1355. (doi:10.1023/A:1023617117780)
 34. Geiger F *et al.* 2010 Landscape composition influences farm management effects on farmland birds in winter: a pan-European approach. *Agric. Ecosyst. Environ.* **139**, 571–577. (doi:10.1016/j.agee.2010.09.018)
 35. Flohre A *et al.* 2011 Agricultural intensification and biodiversity partitioning in European landscapes comparing plants, carabids, and birds. *Ecol. Appl.* **21**, 1772–1781. (doi:10.1890/10-0645.1)
 36. Fischer C, Flohre A, Clement LW, Batary P, Weisser WW, Tscharntke T, Thies C. 2011 Mixed effects of landscape structure and farming practice on bird diversity. *Agric. Ecosyst. Environ.* **141**, 119–125. (doi:10.1016/j.agee.2011.02.021)
 37. Schmidt MH, Roschewitz I, Thies C, Tscharntke T. 2005 Differential effects of landscape and management on diversity and density of ground-dwelling farmland spiders. *J. Appl. Ecol.* **42**, 281–287. (doi:10.1111/j.1365-2664.2005.01014.x)
 38. Batáry P, Báldi A, Erdős S. 2006 Grassland versus non-grassland bird abundance and diversity in managed grasslands: local, landscape and regional scale effects. *Biodivers. Conserv.* **16**, 871–881. (doi:10.1007/s10531-006-9135-5)
 39. Batáry P, Báldi A, Samu F, Szűts T, Erdős S. 2008 Are spiders reacting to local or landscape scale effects in Hungarian pastures? *Biol. Conserv.* **141**, 2062–2070. (doi:10.1016/j.biocon.2008.06.002)
 40. Batáry P, Báldi A, Szél G, Podlussány A, Rozner I, Erdős S. 2007 Responses of grassland specialist and generalist beetles to management and landscape complexity. *Divers. Distrib.* **13**, 196–202. (doi:10.1111/j.1472-4642.2006.00309.x)
 41. Batáry P, Holzschuh A, Orci KM, Samu F, Tscharntke T. 2012 Responses of plant, insect and spider biodiversity to local and landscape scale management intensity in cereal crops and grasslands. *Agric. Ecosyst. Environ.* **146**, 130–136. (doi:10.1016/j.agee.2011.10.018)
 42. Batáry P, Matthiesen T, Tscharntke T. 2010 Landscape-moderated importance of hedges in conserving farmland bird diversity of organic vs. conventional croplands and grasslands. *Biol. Conserv.* **143**, 2020–2027. (doi:10.1016/j.biocon.2010.05.005)
 43. Batáry P, Orci KM, Báldi A, Kleijn D, Kisbenedek T, Erdős S. 2007 Effects of local and landscape scale and cattle grazing intensity on Orthoptera assemblages of the Hungarian Great Plain. *Basic Appl. Ecol.* **8**, 280–290. (doi:10.1016/j.baae.2006.03.012)
 44. Clough Y, Kruess A, Kleijn D, Tscharntke T. 2005 Spider diversity in cereal fields: comparing factors at local, landscape and regional scales. *J. Biogeogr.* **32**, 2007–2014. (doi:10.1111/j.1365-2699.2005.01367.x)
 45. Culman SW, Young-Mathews A, Hollander AD, Ferris H, Sánchez-Moreno S, O'Geen AT, Jackson LE. 2010 Biodiversity is associated with indicators of soil ecosystem functions over a landscape gradient of agricultural intensification. *Landscape Ecol.* **25**, 1333–1348. (doi:10.1007/s10980-010-9511-0)
 46. Dänhardt J, Green M, Lindström Å, Rundlöf M, Smith HG. 2010 Farmland as stopover habitat for migrating birds: effects of organic farming and landscape structure. *Oikos* **119**, 1114–1125. (doi:10.1111/j.1600-0706.2009.18106.x)
 47. Fischer C, Thies C, Tscharntke T. 2011 Small mammals in agricultural landscapes: opposing responses to farming practices and landscape complexity. *Biol. Conserv.* **144**, 1130–1136. (doi:10.1016/j.biocon.2010.12.032)
 48. Flohre A, Rudnick M, Traser G, Tscharntke T, Eggers T. 2011 Does soil biota benefit from organic farming in complex vs. simple landscapes? *Agric. Ecosyst. Environ.* **141**, 210–214. (doi:10.1016/j.agee.2011.02.032)
 49. Gabriel D, Roschewitz I, Tscharntke T, Thies C. 2006 Beta diversity at different spatial scales: plant communities in organic and conventional agriculture. *Ecol. Appl.* **16**, 2011–2021. (doi:10.1890/1051-0761(2006)016[2011:BDADSS]2.0.CO;2)
 50. Haenke S, Scheid B, Schaefer M, Tscharntke T, Thies C. 2009 Increasing syrphid fly diversity and density in sown flower strips within simple vs. complex landscapes. *J. Appl. Ecol.* **46**, 1106–1114. (doi:10.1111/j.1365-2664.2009.01685.x)
 51. Holzschuh A, Steffan-Dewenter I, Kleijn D, Tscharntke T. 2006 Diversity of flower-visiting bees in cereal fields: effects of farming system, landscape composition and regional context. *J. Appl. Ecol.* **44**, 41–49. (doi:10.1111/j.1365-2664.2006.01259.x)
 52. Holzschuh A, Steffan-Dewenter I, Tscharntke T. 2010 How do landscape composition and configuration, organic farming and fallow strips affect the diversity of bees, wasps and their parasitoids? *J. Anim. Ecol.*

- 79, 491–500. (doi:10.1111/j.1365-2656.2009.01642.x)
53. Kovács-Hostyánszki A, Batáry P, Báldi A. 2011 Local and landscape effects on bee communities of Hungarian winter cereal fields. *Agric. Forest Entomol.* **13**, 59–66. (doi:10.1111/j.1461-9563.2010.00498.x)
54. Letourneau DK, Goldstein B. 2001 Pest damage and arthropod community structure in organic vs. conventional tomato production in California. *J. Appl. Ecol.* **38**, 557–570. (doi:10.1046/j.1365-2664.2001.00611.x)
55. Purtauf T, Roschewitz I, Dauber J, Thies C, Tschamtko T, Wolters V. 2005 Landscape context of organic and conventional farms: influences on carabid beetle diversity. *Agric. Ecosyst. Environ.* **108**, 165–174. (doi:10.1016/j.agee.2005.01.005)
56. Rundlöf M, Bengtsson J, Smith HG. 2007 Local and landscape effects of organic farming on butterfly species richness and abundance. *J. Appl. Ecol.* **45**, 813–820. (doi:10.1111/j.1365-2664.2007.01448.x)
57. Rundlöf M, Smith HG. 2006 The effect of organic farming on butterfly diversity depends on landscape context. *J. Appl. Ecol.* **43**, 1121–1127. (doi:10.1111/j.1365-2664.2006.01233.x)
58. Sjödin NE, Bengtsson J, Ekblom B. 2007 The influence of grazing intensity and landscape composition on the diversity and abundance of flower-visiting insects. *J. Appl. Ecol.* **45**, 763–772. (doi:10.1111/j.1365-2664.2007.01443.x)
59. Smith HG, Dänhardt J, Lindström A, Rundlöf M. 2010 Consequences of organic farming and landscape heterogeneity for species richness and abundance of farmland birds. *Oecologia* **162**, 1071–1079. (doi:10.1007/s00442-010-1588-2)
60. Winfree R, Williams NM, Gaines H, Ascher JS, Kremen C. 2007 Wild bee pollinators provide the majority of crop visitation across land-use gradients in New Jersey and Pennsylvania, USA. *J. Appl. Ecol.* **45**, 793–802. (doi:10.1111/j.1365-2664.2007.01418.x)
61. Winqvist C *et al.* 2011 Mixed effects of organic farming and landscape complexity on farmland biodiversity and biological control potential across Europe. *J. Appl. Ecol.* **48**, 570–579. (doi:10.1111/j.1365-2664.2010.01950.x)
62. Roschewitz I, Gabriel D, Tschamtko T, Thies C. 2005 The effects of landscape complexity on arable weed species diversity in organic and conventional farming. *J. Appl. Ecol.* **42**, 873–882. (doi:10.1111/j.1365-2664.2005.01072.x)
63. Borenstein M, Hedges LV, Higgins JPT, Rothstein HR. 2009 *Introduction to meta-analysis*. Chichester, UK: John Wiley & Sons Ltd.
64. Hedges LV, Gurevitch J, Curtis PS. 1999 The meta-analysis of response ratios in experimental ecology. *Ecology* **80**, 1150–1156. (doi:10.1890/0012-9658(1999)080[1150:TMAORR]2.0.CO;2)
65. West BT, Welch KB, Galecki AT. 2007 *Linear mixed models: a practical guide using statistical software*. Boca Raton, FL: Chapman & Hall/CRC.
66. Shackelford G, Steward PR, Benton TG, Kunin WE, Potts SG, Biesmeijer JC, Sait SM. 2013 Comparison of pollinators and natural enemies: a meta-analysis of landscape and local effects on abundance and richness in crops. *Biol. Rev.* **88**, 1002–1021. (doi:10.1111/brv.12040)
67. Tuck SL, Winqvist C, Mota F, Ahnström J, Turnbull LA, Bengtsson J. 2014 Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta analysis. *J. Appl. Ecol.* **51**, 746–755. (doi:10.1111/1365-2664.12219)
68. Gotelli NJ, Colwell RK. 2001 Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. *Ecol. Lett.* **4**, 379–391. (doi:10.1046/j.1461-0248.2001.00230.x)