Providing Context for the Land-Sharing and Land-Sparing Debate

Elizabeth A. Law & Kerrie A. Wilson

The University of Queensland, School of Biological Sciences, Brisbane, QLD 4072, Australia

Abstract

Agricultural and environmental policies that encourage multifunctional landscapes are referred to as land sharing or land sparing. Numerous assessments of the preferences for either policy exist, but a systematic evaluation of the impact of landscape or decision contexts has not been performed. We assess the impact of multiple contextual parameters using a simple model of landscape change. Past emphasis on different types of species appears warranted, but also important in determining preferences are the proportion of the landscape initially allocated to agriculture, the management intentions for spared land, crop types, and level of policy effectiveness. Variations in objective and decision criteria were less important but still altered the policy preferences under some circumstances. We provide foundational evidence that context warrants explicit inclusion in assessments of agricultural and environmental policy. Our methodological approach is broadly applicable, but generalizations from this case to others should still be made with caution.

Introduction

The world’s population may exceed 9.6 billion by 2050 (Tilman et al. 2011). Feeding and providing livelihoods for this growing and increasingly affluent population represents a global challenge. Concurrently, biodiversity continues to decline (Thomas et al. 2004), with species extinctions occurring in both recently disturbed areas and areas with a long history of anthropogenic disturbance (Butler et al. 2007). Multiple objectives are increasingly sought from agricultural landscapes, but land scarcity is forcing trade-offs between provision of food and conserving biodiversity.

How can agricultural landscapes be most effectively managed to capitalize on and enhance this multifunctional capacity? Contrasting land-use policies to achieve this are referred to as land sharing (e.g., agri-environmental schemes, organic agriculture, environmental certification) or land sparing (e.g., biodiversity offsets, or agricultural subsidies in combination with protected areas; Figure 1; Phalan et al. 2011a; Baudron & Giller 2014). While there are parallel debates in forestry and urban settings, and for diverse ecosystem services, much of the focus has been on biodiversity in agricultural landscapes as presented in the seminal article by Green et al. (2005).

Green et al. (2005) present a simple model with two baseline land-uses (natural area and agricultural production), two hypothetical species types (displaying a concave and convex relationship between abundance and yield), and the objective of maximizing biodiversity for a given level of production. They conclude that, ceteris paribus, when the biodiversity-yield trade-off is concave a sharing strategy is optimal, and conversely, when convex, a sparing strategy gains preference. The models of Green et al. (2005) have catalyzed a body of scientific discourse and evaluations of the preference and validity of land-sharing and land-sparing policies (e.g., Matson & Vitousek 2006; Fischer et al. 2008; Chappell et al. 2009; Fischer et al. 2011; Fischer et al. 2014; von Wehrden et al. 2014).

Evaluations of “context” have predominately focused on variation of species types (Appendix A) and describing biodiversity patterns across gradients of agricultural intensity (Hulme et al. 2013; Macchi et al. 2013). These studies have identified species types that are either...
Influence of context on land-use preferences

Figure 1 Comparison of land-sharing and land-sparing strategies in a biodiversity–agricultural production trade-off. Starting from baseline land-uses, land sharing involves actions that increase the biodiversity value of agricultural land, but can also reduce the agricultural yield. To maintain a constant amount of total production, area under agriculture would likely need to increase. Land sparing involves increasing yield in agricultural lands. In some cases this can reduce the biodiversity value of agricultural land, but the area of land required under agricultural production to produce a certain level of total production can decrease, allowing land to be “spared” (Appendix B). The two baseline land-uses and three future land-use options for spared land combine to form five “land-use scenarios.”

Methods

We conducted a targeted literature review to determine the extent to which different contexts have been incorporated into land-sharing and land-sparing research. We reviewed all articles recorded on the Web of Science citing the seminal article of Green et al. (2005). We caution this review is not systematic or comprehensive, but rather provides a brief overview of relevant literature. Full details are provided in Appendix A.

Our simple model determines total biodiversity value and total production as an aggregate of the scores from different land-uses in each contextual scenario, and indicates a preference for land sharing, land sparing, the baseline condition, or a combination of these options (Appendix B). The model is initiated with two baseline land-uses (“agriculture” and “other”), and varies the proportion of the landscape initially allocated to agriculture. Under a land-sharing policy “agriculture” is converted to “low-impact agriculture.” Under land sparing some “agriculture” is converted to “high-yield agriculture” and some is “spared”. We assessed three management options for “spared” land, and two options for the baseline “other” land-uses, which combine to form five land-use scenarios (Figure 1). Land-use scenarios 1 and 2, in which “other” is pristine, and “spared” is either pristine or restored, are commonly evaluated in the literature (referred to herein as “classic”; Macchi et al. 2013). In contrast, scenarios involving degraded areas (scenarios 3 to 5) have rarely been addressed. Given these scenarios, we systematically varied assumptions regarding policy effectiveness, species and crop types, objectives, and decision criteria. All parameters are defined in Table 1 and further explained in Appendix B.
Table 1  Contextual parameters varied in the simple model of landscape change, and parameters derived from these for use in the generalized analysis

| Contextual parameter | Characteristics |
|----------------------|-----------------|
| “Other” land-use      | Baseline nonagricultural land-uses. “Pristine” (natural areas, for example, primary forest), or “degraded” (due to land abandonment). |
| Baseline P( agriculture) | Proportion of the landscape initially allocated to agriculture. Varied from 0% to 100%. The production expected from this initial proportion determines both the “production target” (cf. Green et al. 2005), as well as the target for biodiversity value. |
| Spared land-use       | Management intentions for spared land. Spared land is (1) initially “pristine,” and remains “pristine,” (2) initially cleared and is “restored,” or (3) is abandoned and becomes “degraded.” |
| Policy effectiveness  | Level of policy effectiveness. The expected impacts on biodiversity and yield within different land uses, which occur due to variations in policy implementation and the social and biophysical characteristics of the system. We represent impacts by a modifier of the yield and biodiversity outcomes per unit of land (Figure 2). |
| Land sharing          | Land sharing: The biodiversity value of agricultural land is increased although the level of increase will vary, and under some circumstances a yield penalty could be incurred (Figure 2). |
| Land sparing          | Land sparing: Higher yields are derived from agricultural land although the level of increase will vary, and under some circumstances the biodiversity values is reduced (Figure 2). |
| Crop types            | Smallholder agriculture and oil palm are included for the EMRP parameter set, which have a baseline value of one unit of yield per unit of land, and have different initial values for biodiversity, depending on species types. |
| Species types         | For the EMRP parameter set species types (modeled on primate species) were classified as “sensitive,” “moderate,” and “tolerant” of agricultural production. This category set the base level of abundance per unit area for each land-use type. |
| Objective             | Either maximize total production while maintaining total biodiversity value constant, or maximize total biodiversity value while maintaining constant total production. Total production is defined as the aggregate yield from agriculture. Total biodiversity value is defined as the aggregate value of the biodiversity metric (reflective of the abundance of the species type specified) across both agricultural and nonagricultural land-uses. Both yield and biodiversity value scale linearly with area. |
| Decision criteria     | Either a Pareto or Kaldor-Hicks criteria for aggregating biodiversity and production values. Pareto: Outcomes are improved for at least one objective, while not negatively impacting others (Farrow 1998). If the “maintain” constraint is not achieved under this criterion the outcome defaults to zero (reflecting an unacceptable solution). Otherwise, outcomes for agriculture and biodiversity are aggregated (as potentially both objectives are improved), here assuming parity of units of biodiversity value and agricultural production. Kaldor-Hicks: Potential gains can compensate for potential losses, yet overall performance improves (Stavins et al. 2003). Under this criteria outcomes for agriculture and biodiversity are aggregated, assuming parity of units of biodiversity value and agricultural production. |
| Convexity score       | Land sharing: The difference between the biodiversity value for low-impact agriculture, and that expected for the respective yield level given a linear biodiversity-yield relationship between baseline agriculture and pristine. |
| Biodiversity gain from sparing | Land sparing: Similarly, the difference between the biodiversity value obtained and expected for high-yield agriculture. Convexity scores greater than zero indicates a concave relationship and values less than zero indicates a convex relationship (Appendix D, Figure D1). |
| Biodiversity loss from other land-use | The difference in biodiversity value per unit of spared land and baseline agricultural land-use. |
| Biodiversity loss from other | The difference in biodiversity value per unit of the baseline nonagricultural land-use and baseline agricultural land-use. |
Influence of context on land-use preferences

Figure 2 Three possibilities surrounding the expected impacts of policy (A, B, and C) on biodiversity and yield employed in the simple model of landscape change. A has improvement of one objective, and no impact on the other; B has improvement of one objective, and a negative impact on the other; and C has a large improvement of one objective, and a negative impact on the other. Values were derived from an expansive literature review (Appendix C), and “biodiversity value” is a metric that reflects the abundance of different species types.

Table 2 Modifiers for the biodiversity value (as represented by a metric of species abundance for each species type) and yield for different baseline land-uses and crop types included in the simple model of landscape change, derived from Law et al. (2015)

| Land-use and crop types | Species types | Agricultural production |
|-------------------------|---------------|-------------------------|
|                         | Sensitive     | Moderate                | Tolerant | Yield |
| Pristine (other/spared) | 1             | 1                       | 1        | 0     |
| Degraded (other/spared) | 0.1           | 0.3                     | 0.4      | 0     |
| Restored (spared)       | 0.6           | 0.7                     | 0.9      | 0     |
| Smallholder agriculture | 0.2           | 0.4                     | 0.6      | 1     |
| Oil-palm (agricultural) | 0.1           | 0.2                     | 0.3      | 1     |

We parameterized our model based on the Ex-Mega Rice Project (EMRP) area of Central Kalimantan, Indonesia. This region is a high priority for both economic development and environmental restoration, and questions of whether land should be shared or spared are pertinent (Law et al. 2015). We defined six landscape contextual parameters using values specific to the EMRP landscape and two parameters related to the decision context (i.e., the objective and decision criteria; Table 1). Values for biodiversity and yield for different baseline land-uses, crop types, and species types were taken from Law et al. (2015; Table 2). Estimates for policy effectiveness (i.e., the expected impacts on biodiversity and yield within different land-uses occurring due to variations in policy implementation and the social and biophysical characteristics of the system) were derived from an expansive literature review (i.e., not limited to the targeted review based on Green et al. 2005; Appendix C), and captured three possibilities for the relative impacts on biodiversity and yield (Table 1; Figure 2).

To test the potential generalizability of our analysis method (necessary for comparison across case studies), we assessed how well a generalized parameter set could reflect the outcomes derived from the EMRP specific parameters. This second parameter set generalized the EMRP specific categories into numeric variables: three variables in common with the EMRP categories (proportion of the landscape initially allocated to agriculture, objective, and decision criteria), two metrics of species response to changes in agricultural intensity, and two indicators of landscape context (Table 1).

We examined the importance of contextual variation by applying random forests of conditional inference trees. This ensemble recursive partitioning method outperforms simpler partitioning methods when discerning the contribution of explanatory variables that exhibit correlations, interactions, or are a mixture of categorical and continuous variables (Strobl et al. 2007; Strobl et al. 2008). We included 500 trees in each forest, evaluating three parameters at each split, and bootstrapped using 10 random subsets of the data (n = 3,000 for both the EMRP and generalized parameter sets). We compared the results with those derived from subsets of the data including only the “classic” scenarios (i.e., scenarios 1 and 2, random subset n = 1,200). Further details are available in Appendix D.

We conducted all model analyses in R (R Core Team 2012) including the contributed packages “party” (Strobl et al. 2007; Strobl et al. 2008) and “plotrix” (Lemon 2006). The programming code for the land-use model is provided in the Supplementary Materials.

Results

Of the 362 articles that cite Green et al. (2005), 26 considered the relationship between biodiversity and agricultural yield or profit (Appendix A). These explicitly compared land-sharing and land-sparing policies using modeling (n = 23), or theoretical analysis (n = 3). A further 122 articles analyzed biodiversity metrics over multiple land-use scenarios (mostly gradients of agricultural intensity), but did not simultaneously consider the impacts on agricultural yield or profit.
Influence of context on land-use preferences

E.A. Law & K.A. Wilson

Figure 3  Importance ranking of context variables from conditional inference random forest models for the (a) EMRP and (b) generalized parameter sets. Light gray indicates the models specified using the full data, whereas dark gray show models for land-use scenarios 1 and 2 only. Variable importance is derived from 10 bootstrap replicates, and is scaled relative to the maximum importance within each model and parameter set (a score of zero being of no importance, 1 as most important, with boxplots showing mean, interquartile ranges, and outliers). The mean accuracy of the random forest models (for predicting results of the simplified model of landscape change) were 92% and 91% for the full and subset EMRP specific categories, and 97% and 98% for the full and subset generalized parameter sets, respectively.

Our random forest model results demonstrate high prediction accuracy when using EMRP specific context variables (92%; Figure 3). The proportion of the landscape initially allocated to agriculture was the most important parameter, followed by the management intentions for spared land, and variation in policy effectiveness, crop
Figure 4 Proportion of outcomes of the simplified model of landscape change, by policy preference for (a) EMRP parameter set, (b) variables shared between EMRP and generalized parameter sets, and (c) the generalized parameter set. Gray bars represent all land-use scenarios and the black bars indicate the subset composed of land-use scenarios 1 and 2. The width of the bar (for categorical variables) or height of the bar (for continuous variables), indicate the relative proportion of the outcomes that were allocated to each policy preference within the respective contextual category. “Even” includes cases where both land sharing and land sparing are equal, and improve on the baseline (“base”); “None” includes where all policy options, including the baseline, give equivalent results.

Type, baseline nonagricultural (“other”) land-uses, and species types (Figure 3; Appendix E). Increased preference toward land sparing was associated with higher proportions of the landscape initially allocated to agriculture, crop types with a greater impact on biodiversity (oil-palm in this case), when baseline land-uses were “pristine” and remained in this state when spared, and species sensitive to agriculture (Figure 4, Appendix E). While both objectives resulted overall in preferences for land sparing, land sharing was associated slightly more with an objective to maximize biodiversity (Figure 4). A Pareto decision criterion increased the incidence of both land-sparing and land-sharing policies performing worse than baseline conditions (Figure 4). Altering either the decision criteria or objective alone led to a change of preference in 8% and 16% of the landscape contexts, respectively (Appendix E).

When context parameters were generalized the prediction accuracy remained high (97%; Figure 3) indicating that the complex interactions captured by the EMRP specific categories were also captured by the generalized variables. The proportion of the landscape initially
allocated to agriculture was again the most important parameter, followed by the landscape context indicators (biodiversity gain from sparing and loss from nonagricultural land-uses; Figure 3). A greater preference toward land sharing was observed for the most tolerant species types (with the lowest convexity index scores), and landscapes with the least biodiversity to lose from the baseline nonagricultural land-use or the least to gain from sparing (Figure 4).

Results from classic scenarios (1 and 2) were not reflective of the wider variety of possible contexts. Species type and convexity score parameters increased in importance, while the importance of variables describing land-use contexts diminished (Figure 3). In initially degraded landscapes (scenarios 3 and 5), sparing was preferred less often than in the classic land-use scenarios (scenarios 1 and 2; Appendix E). This was particularly the case when the objective was to maximize biodiversity in scenario 3, or when a Pareto decision criterion was implemented in scenario 5 (Figure 4; Appendix E). Consideration of a wider array of contexts highlighted a number of cases where no policy was superior or the baseline condition was preferred, particularly for scenarios 4 and 5 with oil-palm cropping (Figure 4; Appendix E).

**Discussion**

The land-sharing and land-sparing debate requires an understanding of the trade-off between agricultural production and biodiversity (Green et al. 2005), and the biodiversity value of spared land. However, after almost a decade of research since the seminal work of Green et al. (2005), only 26 of the 362 articles (2.5%) we reviewed explicitly considered these relationships. Of these, only nine included comparisons of sharing and sparing that formally included spared land (e.g., set-asides). Further, while differences in context have been identified to influence preference toward land-sharing or land-sparing policies (Grau et al. 2013; Baudron & Giller 2014), the majority (81%) of articles we reviewed varied at most two contextual parameters (with a focus on species type; Appendix A). We provide the first systematic assessment of the influence of multiple parameters in a model landscape. Further, while detailed studies such as Brady et al. (2012) are useful within their respective contexts, developing recommendations from specific cases will be hampered by confounding variables. We developed a generalized parameter set that performed as well as a set specific to our case study region, suggesting that application of our analysis method to other landscapes is possible, allowing generalizations to be made.

Our results suggest contextual parameters other than species type are potentially more important determinants of whether land sharing or land sparing is preferred. The development history (e.g., the proportion of the landscape initially allocated to agriculture) determines the baseline level of biodiversity and agricultural production, yet few studies address this important source of variation. Our results show an increased preference for sparing when the proportion of agricultural land is high, likely reflecting the disproportionate amount of biodiversity often found within remnant native fragments in such landscapes (Egan & Mortensen 2012). While land sharing has a large area of opportunity to benefit biodiversity in these same landscapes, when land availability is finite there are limited opportunities for agriculture to expand to compensate for yield foregone. The effectiveness of land-sharing and land-sparing policies will vary (Baudron & Giller 2014), and policy effectiveness was an important parameter in our study, but over half of the studies reviewed have not accounted for this. Crop type was considered in few studies reviewed (<20%), but was one of the most important parameters in our analysis. A preference for land sparing was associated with higher impact crops, aligning with results of earlier studies (Anderson-Teixeira et al. 2012; Macchi et al. 2013). A broader spectrum of land-use scenarios also warrant further investigation, as we found the classically invoked scenarios are not representative of contexts including abandoned and degraded land management.

Our targeted literature review revealed that policy objectives were typically not made explicitly, though often were equivalent to maximizing biodiversity (Appendix A). Similarly, decision criteria were often not employed and never explicit, though a Pareto criterion was implied by several studies analyzing threshold targets or production-possibility frontiers. Both the maximize biodiversity objective and Pareto decision criteria slightly increased the preference for land-sharing or baseline conditions in our study. The Kaldor-Hicks criterion is common in policy evaluation, but assumes that one objective could be completely sacrificed if there were commensurate gains in the other (Stavins et al. 2003). A Pareto decision criterion is perhaps more appropriate in the case of conservation as it preserves the independent value of each objective, which may accrue to different stakeholders (Farrow 1998).

Overwhelmingly we found a preference for land sparing across most contexts for our case study system, in particular if yield improvements could be gained without reducing biodiversity, or if management actions improve the biodiversity value of spared land to above that of other nonagricultural land-uses. This is likely in this case due to the relatively low divergence in yields,
but high divergences in biodiversity between land-uses, particularly given many species in this region are relatively sensitive to forest loss (Edwards et al. 2010; Foster et al. 2011). However, we also reiterate the emphasis made by other authors that demand thresholds of yield and biodiversity need not be immutable, and problems of food scarcity, biodiversity decline, and economic development can be approached through a diversity of policy instruments that may not be readily aligned with a simple dichotomous land-sharing–land-sparing framework (Balmford et al. 2005; Fischer et al. 2014; Loos et al. 2014). Further, the assumption that yield increases will be coupled with sparing strategies may not hold in reality (Matson & Vitousek 2006; Fischer et al. 2008; Chappell et al. 2009; Fischer et al. 2011; Fischer et al. 2014). Land sparing in particular may be associated with displacement of agricultural activity and increases in land rents associated with the “Jevons Paradox” (Angelsen & Kaimowitz 2001; Ceddia et al. 2013; Hertel et al. 2014).

We have not accounted for the full scope of variation in the parameters employed or all potentially important variables. For example, land-use patterns (Henderson et al. 2012) and interactions between land-uses and species are likely to be important (Shapira et al. 2008; Butsic et al. 2012; Mendenhall et al. 2014), as may spatial and temporal heterogeneity (Piha et al. 2007; Mahood et al. 2012; Macchi et al. 2013; Maskell et al. 2013; von Wehrden et al. 2014), different decision makers (Barraquand & Martinet 2011; Brady et al. 2012), inclusion of uncertainty (Johnson et al. 2012), and the possibility of leakage (Fischer et al. 2014). Land sparing and sharing is also perceived differently depending on the scale (for instance set-asides on private farmland may be considered sparing at local scales, and sharing at regional scales), and land management applied at different scales may have nonlinear costs and benefits (Fischer et al. 2014). Furthermore, policy preference will be highly dependent on the governance systems, institutional capacity to support implemented policies, and the uptake and social preferences for these (Angelsen & Kaimowitz 2001; Ceddia et al. 2013; Meijaard et al. 2013).

Analysis of the impact of contextual parameters, when undertaken in conjunction with an analysis of landscape heterogeneity, will provide greater clarity on the extent to which preferences for land sharing or sparing are due to contextual details or heterogeneity in and of itself. This is particularly important when considering alternative definitions of “biodiversity value.” In this study we used a simplistic, aspatial metric for biodiversity (reflecting the abundance of a particular species type that responds linearly with increasing area). In reality the concept of biodiversity is much more complex, and the relationship with area might be nonlinear and saturate (e.g., species richness) or be dependent on site characteristics (e.g., composition). Use of different definitions (or “metrics”) of biodiversity is likely to result in different outcomes (von Wehrden et al. 2014), and results of simple metrics may not necessarily be aggregated to capture outcomes for biodiversity that are a function of spatial or temporal heterogeneity. For instance, complementarity-based metrics have generally favored sparing strategies (Appendix A), but this result could be driven by certain species requiring the benefits of spared land, rather than sparing performing better for all species in complex landscapes. This emphasizes the importance of constructing and evaluating contextual variables (for biodiversity, but also other features) that are most relevant to the case study system before drawing inferences of preference.

Conclusions

Whether land sharing or land sparing is preferred in different locations is determined by numerous contextual features beyond the sensitivity of species to changes in land-use intensity. The model of landscape change that we have developed and applied is a simple tool to determine the impact and importance of different contextual features on the preference for land sharing and land sparing. This first systematic analysis of the role of context in determining preferences improves the basic knowledge of contextually dependent policy impacts that underpin land-use planning and management.

Acknowledgments

We thank Erik Meijaard for providing expert opinion for the impacts of land-use on biodiversity, as well as Brett Bryan, Thilak Mallawarachchi, Erik Meijaard, and two anonymous reviewers for providing feedback on the manuscript. EL is supported by the ARC Centre of Excellence for Environmental Decisions, a University of Queensland-CSIRO Integrated Natural Resource Management fellowship, and an Australian Postgraduate Award. KAW is supported by the ARC Centre of Excellence and Future Fellowship programs.

Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher’s web site:

Appendix A. Targeted literature review methods and references.
Appendix B. Overview of the model.
Appendix C. Model parameters.
Appendix D. Conditional inference trees and random forest parameterization methods.

Appendix E. Full model results.

R code for the simple model of landscape change is provided in SimpleModel.LSLS.LawWilson.r

References

Anderson-Teixeira K.J., Duval B.D., Long S.P. & DeLucia E.H. (2012). Biofuels on the landscape: is "land sharing" preferable to "land sparing"? *Ecol. Appl.* 22, 2035-2048.

Angelsen A. & Kaimowitz D. (2001). *Agricultural technologies and tropical deforestation*. CABI Publishing, Wallingford, UK, New York, NY.

Balmford A., Green R.E. & Scharlemann J.P.W. (2005). Sparing land for nature: exploring the potential impact of changes in agricultural yield on the area needed for crop production. *Glob. Chang. Biol.* 11, 1594-1605.

Barraquand F. & Martinet V. (2011). Biological conservation in dynamic agricultural landscapes: effectiveness of public policies and trade-offs with agricultural production. *Ecol. Econ.* 70, 910-920.

Baudron F. & Giller K.E. (2014). Agriculture and nature: trouble and strife? *Biol. Conserv.* 170, 232-245.

Brady M., Sahrbacher C., Kellermann K. & Happe K. (2012). An agent-based approach to modeling impacts of agricultural policy on land use, biodiversity and ecosystem services. *Landsc. Ecol.* 27, 1363-1381.

Butler S.J., Vickery J.A. & Norris K. (2007). Farmland biodiversity and the footprint of agriculture. *Science* 315, 381-384.

Butsic V., Radeloff V.C., Kuenmerle T. & Pidgeon A.M. (2012). Analytical solutions to trade-offs between size of protected areas and land-use intensity. *Conserv. Biol.* 26, 883-893.

Ceddia M.G., Sedlacek S., Bardens N. & Gomez-y-Paloma S. (2013). Sustainable agricultural intensification or Jevons paradox? The role of public governance in tropical South America. *Glob. Environ. Change* 23, 1052-1063.

Chappell M.J., Vandermeer J., Badgley C. & Perfecto I. (2009). Wildlife-friendly farming vs land sparing. *Front. Ecol. Environ.* 7, 83-84.

Edwards D.P., Hodgson J.A., Hamer K.C. et al. (2010). Wildlife-friendly oil palm plantations fail to protect biodiversity effectively. *Conserv. Lett.* 3, 236-242.

Egan J.F. & Mortensen D.A. (2012). A comparison of land-sharing and land-sparing strategies for plant richness conservation in agricultural landscapes. *Ecol. Appl.* 22, 459-471.

Farrow S. (1998). Environmental equity and sustainability: rejecting the Kaldor-Hicks criteria. *Ecol. Econ.* 27, 183-188.

Ferraro P.J. (2009). Counterfactual thinking and impact evaluation in environmental policy. *New Direct. Eval.* 2009, 75-84.

Fischer J., Abson D.J., Butsic V. et al. (2014). Land sparing versus land sharing: moving forward. *Conserv. Lett.* 7, 149-157.

Fischer J., Batary P., Bawa K.S. et al. (2011). Conservation: limits of land sparing. *Science* 334, 593-593.

Fischer J., Brosi B., Daily G.C. et al. (2008). Should agricultural policies encourage land sparing or wildlife-friendly farming? *Front. Ecol. Environ.* 6, 382-387.

Foster W.A., Snaddon J.L., Turner E.C. et al. (2011). Establishing the evidence base for maintaining biodiversity and ecosystem function in the oil palm landscapes of South East Asia. *Philos. Trans. R. Soc. B-Biol. Sci.* 366, 3277-3291.

Grau R., Kuehmerle T. & Macchi L. (2013). Beyond ‘land sparing versus land sharing’: environmental heterogeneity, globalization and the balance between agricultural production and nature conservation. *Carr. Opin. Environ. Sustain.* 5, 477-483.

Green R.E., Cornell S.J., Scharlemann J.P.W. & Balmford A. (2005). Farming and the fate of wild nature. *Science* 307, 550-555.

Henderson I.G., Holland J.M., Storkey J., Lutman P., Orson J. & Simper J. (2012). Effects of the proportion and spatial arrangement of un-cropped land on breeding bird abundance in arable rotations. *J. Appl. Ecol.* 49, 883-891.

Hertel T.W., Ramankutty N. & Baldos U.L.C. (2014). Global market integration increases likelihood that a future African Green Revolution could increase crop land use and CO2 emissions. *Proc. Natl. Acad. Sci.* 111, 13799-13804.

Hulme M.F., Vickery J.A., Green R.E. et al. (2013). Conserving the birds of Uganda’s banana-coffee arc: land sparing and land sharing compared. *PLoS One* 8, e54597.

Johnson K.A., Polasky S., Nelson E. & Pennington D. (2012). Uncertainty in ecosystem services valuation and implications for assessing land use tradeoffs: an agricultural case study in the Minnesota River Basin. *Ecol. Econ.* 79, 71-79.

Law E.A., Bryan B.A., Meijjaard E., Mallawaarachchi T., Struiebig M.J. & Wilson K.A. (2015). Ecosystem services from a degraded peatland of Central Kalimantan: implications for policy, planning, and management. *Ecol. Appl.* 25, 70-87.

Lemon J. (2006). Plotrix: a package in the red light district of R. *R-News* 6, 8-12.

Loos J., Abson D.J., Chappell M.J. et al. (2014). Putting meaning back into “sustainable intensification”. *Front. Ecol. Environ.* 12, 356-361.

Macchi L., Grau H.R., Zelaya P.V. & Marinaro S. (2013). Trade-offs between land use intensity and avian biodiversity in the dry Chaco of Argentina: a tale of two gradients. *Agric. Ecosyst. Environ.* 174, 11-20.
Mahood S.P., Lees A.C. & Peres C.A. (2012). Amazonian countryside habitats provide limited avian conservation value. *Biodivers. Conserv.* **21**, 385-405.

Maskell L.C., Crowe A. & Dunbar M.J. *et al.* (2013). Exploring the ecological constraints to multiple ecosystem service delivery and biodiversity. *J. Appl. Ecol.* **50**, 561-571.

Matson P.A. & Vitousek P.M. (2006). Agricultural intensification: will land spared from farming be land spared for nature? *Conserv. Biol.* **20**, 709-710.

Meijaard E., Abram N.K., Wells J.A. *et al.* (2013). People’s perceptions about the importance of forests on Borneo. *PLoS One* **8**, e73008.

Mendenhall C.D., Karp D.S., Meyer C.F., Hadly E.A. & Daily G.C. (2014). Predicting biodiversity change and averting collapse in agricultural landscapes. *Nature* **509**, 213-217.

Phalan B., Balmford A., Green R.E. & Scharlemann J.P.W. (2011a). Minimising the harm to biodiversity of producing more food globally. *Food Policy* **36**, S62-S71.

Phalan B., Onial M., Balmford A. & Green R.E. (2011b). Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science* **333**, 1289-1291.

Piha H., Luoto M. & Merila J. (2007). Amphibian occurrence is influenced by current and historic landscape characteristics. *Ecol. Appl.* **17**, 2298-2309.

R Core Team. (2012). R: A language and environment for statistical computing. R Foundation for Statistical Computing. ISBN 3-900051-07-0, Vienna, Austria. URL http://www.R-project.org/.

Shapira I., Sultan H. & Shanas U. (2008). Agricultural farming alters predator-prey interactions in nearby natural habitats. *Anim. Conserv.* **11**, 1-8.

Stavins R.N., Wagner A.F. & Wagner G. (2003). Interpreting sustainability in economic terms: dynamic efficiency plus intergenerational equity. *Econ. Lett.* **79**, 339-343.

Steffan-Dewenter I., Kessler M., Barkmann J. *et al.* (2007). Tradeoffs between income, biodiversity, and ecosystem functioning during tropical rainforest conversion and agroforestry intensification. *Proc. Natl. Acad. Sci. U.S.A.* **104**, 4973-4978.

Strobl C., Boulesteix A.-L., Kneib T., Augustin T. & Zeileis A. (2008). Conditional variable importance for random forests. *BMC Bioinformatics* **9**, 307.

Strobl C., Boulesteix A.-L., Zeileis A. & Hothorn T. (2007). Bias in random forest variable importance measures: illustrations, sources and a solution. *BMC Bioinformatics* **8**, 25.

Thomas C.D., Cameron A., Green R.E. *et al.* (2004). Extinction risk from climate change. *Nature* **427**, 145-148.

Tilman D., Balzer C., Hill J. & Befort B.L. (2011). Global food demand and the sustainable intensification of agriculture. *Proc. Natl. Acad. Sci.* **108**, 20260-20264.

von Wehrden H., Abson D.J., Beckmann M., Cord A.F., Klotz S. & Seppelt R. (2014). Realigning the land-sharing/land-sparing debate to match conservation needs: Considering diversity scales and land-use history. *Landsc. Ecol.* **29**, 941-948.