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REVIEW

Assessing ecosystem service trade-offs and synergies: The need for a more mechanistic approach

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Abstract Positive (synergistic) and negative (trade-off) relationships among ecosystem services are influenced by drivers of change, such as policy interventions and environmental variability, and the mechanisms that link these drivers to ecosystem service outcomes. Failure to account for these drivers and mechanisms can result in poorly informed management decisions and reduced ecosystem service provision. Here, we review the literature to determine the extent to which drivers and mechanisms are considered in assessments of ecosystem service relationships. We show that only 19% of assessments explicitly identify the drivers and mechanisms that lead to ecosystem service relationships. While the proportion of assessments considering drivers has increased over time, most of these studies only implicitly consider the drivers of ecosystem service relationships. We recommend more assessments explicitly identify drivers of trade-offs and synergies, which can be achieved through a greater uptake of causal inference and process-based models, to ensure effective management of ecosystem services.

Keywords Ecosystem management · Ecosystem services · Social-ecological system · Synergy · Trade-off

INTRODUCTION

One of the goals of many environmental policy initiatives is improved human well-being through the provision of ecosystem services derived from natural and human modified ecosystems (MA 2005; Tallis et al. 2008; Raudsepp-Hearne et al. 2010; Kandziora et al. 2013; Guerry et al. 2015). Initiatives such as the United Nation's Sustainable Development Goals (SDGs) and the Reduced Emission from Deforestation and Environmental Degradation (REDD+) focus on managing multiple ecosystem services (Alexander et al. 2011; Griggs et al. 2013). Considering multiple services complicates the design of policies because complex relationships exist among ecosystem services that can lead to simultaneous positive and negative changes in the provision of different ecosystem services in response to a policy change (Bennett et al. 2009; Howe et al. 2014). Understanding these relationships among ecosystem services in order to inform policy is therefore important, but it requires a consideration of the specific drivers of change (such as policy interventions) and the mechanisms that link drivers to ecosystem service outcomes across multiple services. Theoretical and conceptual models have been developed to help us understand the mechanisms that determine the relationships between ecosystem services within different systems (Bennett et al. 2009; De Groot et al. 2010; Rounsevell et al. 2010; Villamagna et al. 2013). However, the extent to which this mechanistic understanding has been used in empirical assessments of ecosystem service relationships, where both the drivers and mechanisms linking these drivers to ecosystem service provision are identified, is unclear (Bennett et al. 2015).

Relationships between ecosystem services can occur as trade-offs, where the provisioning of one service increases as another decreases, or as synergies, where the provisioning of two services increase or decrease simultaneously (Rodríguez et al. 2006). These relationships arise in response to exogenous or endogenous changes to the system, referred to as drivers (Bennett et al. 2009). Drivers can

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be related to human interventions and natural variability, including policy instruments, climate change and technological advances. For example, Schröter et al. (2005) determined that climate change drives a trade-off between two ecosystem services, carbon storage and food production in Europe, as it increases the suitable area for forests while decreasing the area suitable for arable land. The biotic, abiotic, socio-economic and cultural processes that link drivers to the provision of ecosystem services, referred to as mechanisms, are also crucial for trade-offs or synergies between ecosystem services (De Groot et al. 2002; Potschin and Haines-Young 2011). For example, increasing temperatures (a driver) in boreal forests resulting from global climate change have been found to decrease the rate of soil nutrient cycling. Since this rate is a mechanism that affects two final ecosystem services, below ground carbon storage and maintenance of soil fertility, increasing temperatures create a negative synergy between these two services (Allison and Treseder 2008). Thus, identifying drivers and the mechanisms linking drivers to ecosystem services is key to understanding whether trade-offs or synergies between services are likely to occur.

There are multiple ways in which drivers can influence ecosystem service relationships. Bennett et al. (2009) developed a framework for understanding how drivers can influence the provisioning of ecosystem services via different mechanisms, referred to as different mechanistic pathways, and hence how these influence the relationships among ecosystem services. They outlined four main mechanistic pathways by which drivers can affect ecosystem service relationships (Fig. 1). First, a driver can directly affect the supply of one ecosystem service, with no effect on another ecosystem service. Second, a driver can affect a single ecosystem service that has a unidirectional (one way) or bidirectional (two way) interaction with another ecosystem service. Third, a driver can directly affect two ecosystem services that do not interact with each other. Fourth, a driver can directly affect two ecosystem services that also have either a unidirectional or bidirectional interaction between them.

An important insight from Bennett et al. (2009) is that trade-offs and synergies between ecosystem services can vary depending on the drivers and mechanistic pathways that link drivers to ecosystem services (Fig. 1). For example, a policy for reforesting abandoned cropland, where there is no competition between forest and cropland, will result in an increase in carbon sequestration, but with no direct effect on food production (Rey Benayas et al. 2007). This represents the first pathway of the Bennett et al. (2009) framework (Fig. 1a) with no trade-off or synergy occurring between these services. Contrastingly, a forest restoration policy such as the Grain to Green program in China (Liu et al. 2008) may incentivise reforestation and lead to increased carbon sequestration, but could also lead to decreased food production due to competition for land if cropland is replaced by forest. This interaction between the two services could result in a trade-off between carbon sequestration and food production (Fig. 1b). In comparison, a policy promoting the restoration of riparian vegetation within agricultural landscapes, where riparian zones are often unsuitable for agriculture and there is little competition between these two land uses, could lead to both increased carbon sequestration as tree cover increases, and increased crop production since riparian vegetation can improve soil retention and consequently crop production (Stutter et al. 2012). Therefore, a synergy results between the services despite no direct interaction between the two (Fig. 1c). Alternatively, a policy that incentivises urban expansion could negatively affect the area of both forests and croplands and result in a negative synergy (Fig. 1d) through the fourth mechanistic pathway (Lawler et al. 2014). However, if there is subsequent expansion of crops at the expense of forest to meet food demand, and a strong negative interaction between the two services is created, then a trade-off between the two services is also possible.

Given that different drivers and mechanistic pathways lead to very different synergistic or trade-off outcomes, failing to incorporate a mechanistic understanding into the assessment of ecosystem service trade-offs and synergies that explicitly identifies the drivers is likely to result in misidentified policy solutions. This is because without isolating the mechanistic pathways between policy instruments and ecosystem services it is likely that the effect of a policy will be wrongly estimated through the confounding with other variables (Ferraro 2009). For example, the introduction of policy instruments to achieve a policy objective, such as management actions or incentives for behavioural change, in the absence of careful thinking about these mechanisms can result in unexpected declines in ecosystem services. This is due to the presence of other confounding variables such natural variability, and therefore this policy instrument fails to achieve the original policy objective (Lindenmayer et al. 2012). There have been a number of reviews on the assessment of trade-offs and synergies (Howe et al. 2014; Mouchet et al. 2014; Lee and Lautenbach 2016; Spake et al. 2017), but none have quantified the extent to which drivers and mechanisms are considered in these assessments. This information could provide important information on which assessments of relationships between ecosystem services could potentially result in poorly informed management and policy decisions. Without this information, and the consequent lack of understanding on the drivers and mechanisms underpinning ecosystem service relationships, assessments could inform management actions that target the wrong mechanism and lead to negligible, or even declines, in the target ecosystem

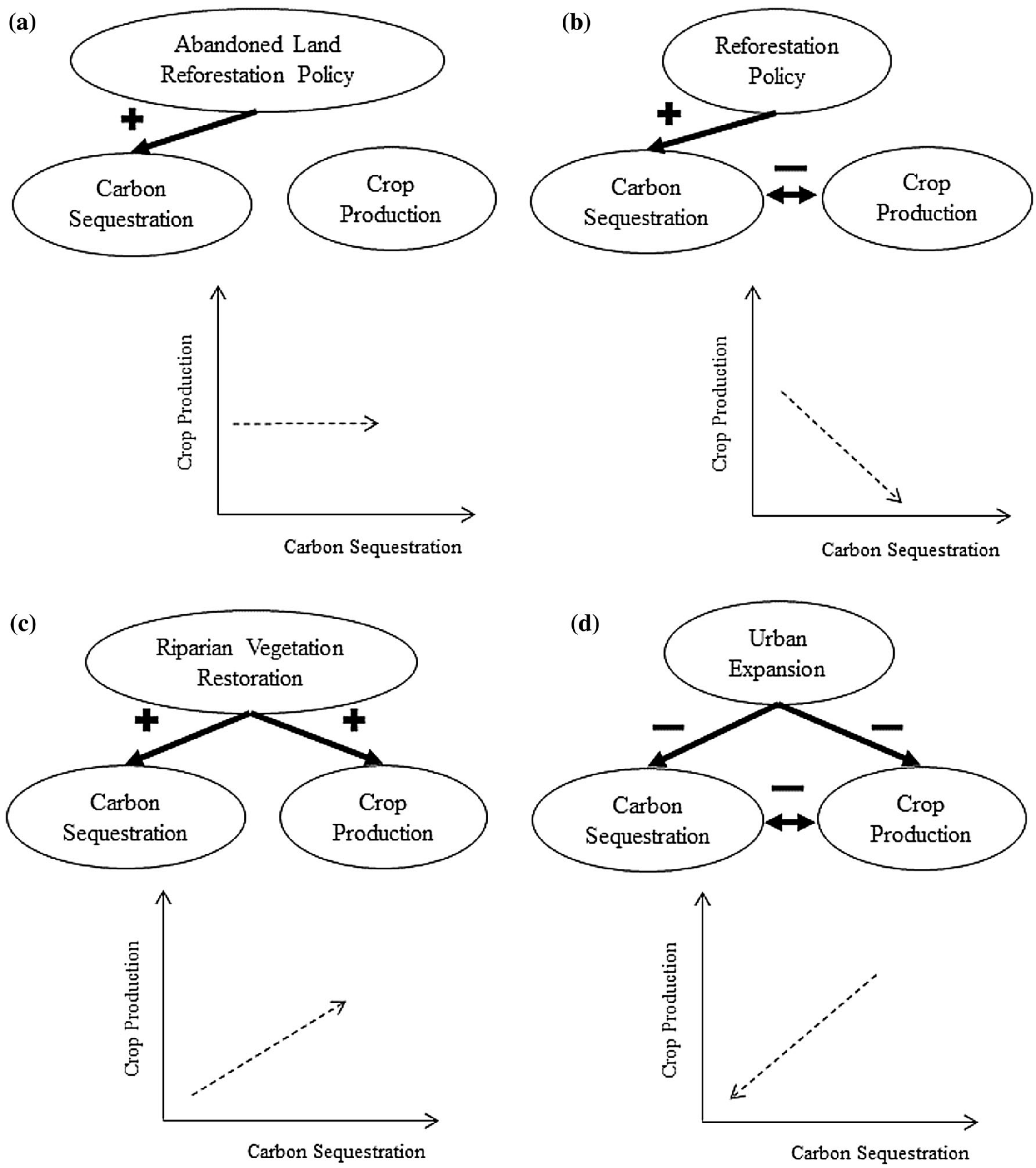

Fig. 1 Schematic demonstrating how the mechanistic pathways in which drivers affect ecosystem services can affect the relationships between ecosystem services. In **a**, the reforestation of abandoned agricultural land (the driver) increases forested area, and consequently carbon sequestration. However, since this land is no longer used for crop production, this reforestation has no effect on food production. Therefore, there is no trade-off or synergy between the two ecosystem services. In **b**, a restoration policy (the driver) positively affects the forested area. However, because cropland and forest compete for land, forest area increases at the expense of cropland. This leads to a trade-off between the two ecosystem services. In **c**, management actions to restore degraded riparian vegetation will increase carbon sequestration due to increased tree cover, and increase crop production as it increases soil fertility, creating a synergy. As riparian zones are often unsuitable for agriculture, there is no competition between the two ecosystem services under this management action. In **d**, urban expansion (the driver) negatively affects the area available for both cropland and forest. Cropland and forest also negatively interact with one another as they compete for land. However, since the driver simultaneously decreases the area available for cropland and forests, this leads to a negative synergy between carbon sequestration and food production Adapted from Bennett et al. (2009)

services, as well as unexpected changes in the provisioning of multiple other services (Turkelboom et al. 2018).

In this paper, we address this limitation by systematically reviewing the relevant literature and quantifying how often drivers and the mechanisms linking drivers to ecosystem services are accounted for when assessing synergies and trade-offs between pairs of ecosystem services. We also assess the types of drivers considered and the methods being used to identify trade-offs and synergies. We then discuss the implications for research into ecosystem service trade-offs and synergies so that policy and planning can be better informed.

MATERIALS AND METHODS

We conducted a systematic literature review of peer-reviewed articles using the ISI Web of Knowledge database and the search string: "ecosystem service*" AND ((synerg*) OR (trade-off* OR trade off* OR tradeoff*)). We limited the search to between 2005 and 2017 as previous ecosystem service reviews (Howe et al. 2014; Lee and Lautenbach 2016) found minimal literature before the publication of the Millennium Ecosystem Assessment (MA 2005). Our review consisted of a two-step screening process, where the abstracts of the articles were first screened for relevancy and, if deemed relevant, the entire paper was then read in the second stage of screening. Our search initially identified 1993 scientific articles meeting the search criteria. One of the authors (MD) then screened the abstracts of each article and removed any articles that were not written in English, that were a review or conceptual paper or that lacked consideration of trade-offs or synergies between pairs of ecosystem services clearly mentioned in the abstract. Any articles where MD was unable to precisely determine this information from the abstract were not removed and included in the next stage of screening. This process resulted in 315 articles. A second round of screening was then conducted by reading the full text of each article for relevancy, using the same criteria as in the first screening, which reduced the final number of articles to 158.

Papers were analysed using predefined questions and criteria (Table 1) drawing on previous ecosystem service reviews (Haase et al. 2014; Howe et al. 2014; Mouchet et al. 2014; Runting et al. 2017). Geographical data (study location) were extracted to identify any geographical patterns among the assessments of ecosystem service trade-offs and synergies, and the mechanisms and drivers identified. Data on the ecosystem services assessed in each study, and the relationships identified between services (synergy, trade-off or no relationship) were then recorded. Because the names of specific ecosystem services were not consistent among the articles, we categorised each ecosystem service studied into "groups" using the Common International Classification of Ecosystem Services (CICES) V4.3 (see Table 1). This allowed for consistency in the identification of ecosystem service types among articles.

The articles were categorised into three groups based on the extent to which they considered the drivers and mechanisms leading to trade-offs and synergies. These groups were: no mention, implicit and explicit. *No mention* was defined as identifying an ecosystem service relationship, but not mentioning the driver or mechanisms leading to the synergy or trade-off. For example, Baral et al. (2013) identified a trade-off between forage production and water regulation, but did not mention what processes were driving the trade-off. *Implicit* was defined as identifying an ecosystem service relationship, and identifying or discussing potential drivers associated with the trade-off or synergy, and the pathways by which these drivers influence the relationship, but not explicitly quantifying the drivers or integrating them into the assessment. For example, Su et al. (2012) identified a trade-off between carbon storage and food production, and identified that human activity was associated with the presence of this trade-off, and therefore concluded it was a driver. However, that study did not explicitly identify the mechanistic links that explain how this driver influences this trade-off. *Explicit* consideration of drivers was defined as explicitly identifying the mechanistic pathways through which drivers influence ecosystem service relationship, and integrating this into the assessment. For example, Classen et al. (2014) used a controlled experimental design, by excluding and controlling variables in field plots, to explicitly identify pathways by which the presence of vertebrates drive a synergy between pollination, and therefore food production and pest control. From the articles that either implicitly or explicitly identified the driver, we then extracted information on the type of driver being considered. Drivers were categorised into 12 groups that were adapted from the drivers of ecosystem service change outlined in the Millennium Ecosystem Assessment (MA 2005), and included both direct and indirect drivers (Table 1). A detailed definition for each driver can be found in the Millennium Ecosystem Assessment (MA 2005). The methodological approaches used by each article to identify trade-offs and synergies were also recorded to determine if there was a link between the methods applied and how drivers are considered. Methods were categorised into six classes: correlation, ordination, overlap analysis, ANOVA, regression and scenario analysis.

We calculated the different spatial and temporal extents to which drivers were considered in the analyses (in total and per year), the frequency of the different methods used

Table 1 Details of variables extracted from each article during the literature review

Variables extracted	Categories
Study area	Country(ies) where the study was located
Consideration of the drivers of ecosystem service relationships	Explicit: defined as identifying an ecosystem service relationship with potential drivers explicitly integrated into the assessment
	Implicit: defined as identifying an ecosystem service relationship, with potential drivers identified or discussed, but not explicitly incorporating it into the assessment
	No mention: defined as identifying an ecosystem service relationship, but not mentioning the driver or mechanisms leading to the synergy or trade-off
Driver identified	Categories adapted from drivers of change for ecosystem services identified in MA (2005): demographic (i.e. population size); socio-economic (i.e. average income); socio-political (i.e. type of governance); scientific and technological advances (i.e. advances in harvesting machinery); cultural and religious (i.e. religious values); policy instruments (i.e. incentives for behavioural change); land use/land cover change (i.e. decreased tree cover); species introductions/removals (i.e. the introduction of pest control species); natural resource management (i.e. fertiliser use); harvest and resource demand (i.e. meat consumption); climate change (i.e. increasing atmospheric temperatures); natural, physical and biological drivers (i.e. soil type)
Method used to calculate ecosystem service trade-offs and synergies	Correlation: measures the association between the supply of ecosystem services using correlation coefficients
	Overlap analysis: quantifies percentage of locations where two ecosystem services are provided at the same time. Trade-offs occur where one service is in high supply, and another is in low supply at different locations. Synergies occur where both service are simultaneously in high or low supply at different locations
	Ordination: multivariate analyses that order ecosystem service supply by values on multiple variables so that similar objects are near each other and dissimilar objects are farther from each other in ordination space
	ANOVA: tests whether there are statistical differences between the means of different ecosystem services
	Regression: quantifies how the supply of an ecosystem service changes when the supply of one or more other ecosystem services change. Regression methods include general linear models, logistic models, structural equation models and path analysis
	Scenario analysis: a systematic method for developing alternative futures about the supply of ecosystem services
Ecosystem services that were assessed	Categorised based on CICES V4.3 Ecosystem Service Classifications, group level (http://cices.eu/): biomass—nutrition (i.e. food production); water (for human consumption); biomass—materials (i.e. timber); water—materials (i.e. water used for industrial manufacturing); biomass-based energy sources (i.e. biofuel); mechanical energy (i.e. hydropower); mediation by biota (i.e. carbon storage and sequestration); mediation by ecosystems (i.e. mediation of noise or smells); mass flows (i.e. erosion control); liquid flows (i.e. flood mitigation); Gaseous/airflows (i.e. air ventilation); lifecycle maintenance, habitat and gene pool protection (i.e. pollination); Pest and disease control (i.e. pest regulation); soil formation and composition (i.e. soil fertility); water conditions (i.e. regulation of water quality); atmospheric composition and climate regulation (i.e. regulation of greenhouse gases); physical and experiential interactions (i.e. hiking); Intellectual and representative interactions (i.e. education); spiritual and/or emblematic (i.e. spiritual identity); other cultural outputs (such as enjoyment provided by existence of wild species) (Haines-Young and Potschin 2013)
Relationships identified between the ecosystem services	Trade-offs: one service increases, while the other decreases
	Synergy: two ecosystem services increase or decrease simultaneously

(in total and per year), the ecosystem services assessed, the trade-offs identified, the synergies identified and the number of studies conducted in each country. A Pearson's Chi squared test was used to assess whether the consideration of drivers and mechanisms was associated with the assessment method used.

RESULTS

Temporal and geographic patterns of ecosystem services assessed

Of the 158 articles examined, a total of 830 pairs of ecosystem service relationships were assessed (see the

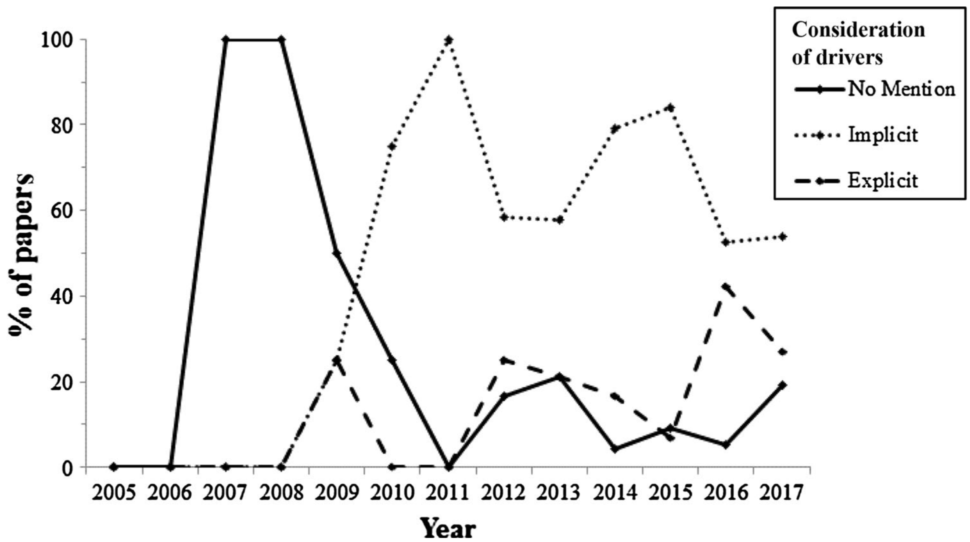

Fig. 2 The percentage of papers considering the different categories of the drivers of ecosystem service trade-offs and synergies over time

supplementary material for a full list of articles reviewed (S1) and a spreadsheet of all data extracted from these articles). Of these pairs, 470 were synergies and 360 were trade-offs. The most common trade-off was between mediation by biota (ecosystem services provided by individual plants and animals, such as carbon storage) and biomass (food production) ($n = 23$). The most common synergy was between one type of biomass (e.g. crops) and another type of biomass (e.g. meat production) ($n = 22$), such as would occur between two different types of food production systems (see Fig. S1 and Fig. S2 for further details).

There was an increase in the number of ecosystem service relationship articles published from 2005 to 2017. Our literature review failed to find any articles published from 2005 and 2006 that focused on ecosystem service trade-offs and synergies. However, from 2007 onwards, the number of articles recorded increased with each consecutive year, with one article published in 2007 and 26 articles in 2017. This indicates less of an increase than seen in the ecosystem service literature within the ISI Web of Knowledge database, with 85 articles published on the topic of ecosystem services in 2005 and 3143 published in 2017. Within the articles we reviewed, there was a geographically wide distribution of case studies, with assessments of ecosystem service trade-offs and synergies in every continent, other than Antarctica. The country with the highest number of assessments was China ($n = 32$), followed by the United States of America ($n = 18$) and Spain ($n = 16$).

Drivers and mechanisms assessed

Only 19% ($n = 30$) of the articles that we examined explicitly incorporated drivers of ecosystem service trade-offs or synergies into their assessment. However, a large proportion of articles (68%, $n = 107$) implicitly considered drivers. In general, articles that implicitly considered drivers either suggested possible drivers by identifying variables correlated with trade-offs and synergies using statistical analyses (e.g. Ai et al. (2015)), or hypothesised potential drivers based on a review of the literature or field observations without the use of statistical analyses (e.g. Cohen-Shacham et al. (2011)). A small proportion of articles (14%, $n = 22$) made no mention of any drivers of the trade-offs or synergies. There was also a distinct trend over time in the way drivers and mechanisms have been considered in the assessment of ecosystem service synergies and trade-offs (Fig. 2). Prior to 2008, while only two relevant articles were published, neither mentioned drivers in their assessments of ecosystem service synergies and trade-offs. However, from 2008 onwards there was an increase in the proportion of articles that implicitly considered drivers, with 25% ($n = 1$) of papers implicitly considering drivers in 2009 to 53% ($n = 14$) in 2017 (Fig. 2). Also from 2008, a small number of articles began to explicitly incorporate drivers into their assessments. However, while the number of papers implicitly considering drivers continued to grow, the number explicitly considering drivers has remained low, with 25% ($n = 1$) of papers explicitly considering drivers in 2008 and 27% ($n = 6$) in 2017 (Fig. 2).

The majority (75%, $n = 50$) of papers that identified trade-offs and synergies involving cultural ecosystem services only implicitly identified the drivers underpinning these relationships (Fig. 3). Only 4% ($n = 3$) of these articles explicitly identified the drivers underpinning these relationships. However, the papers that assessed trade-offs and synergies involving atmospheric composition and

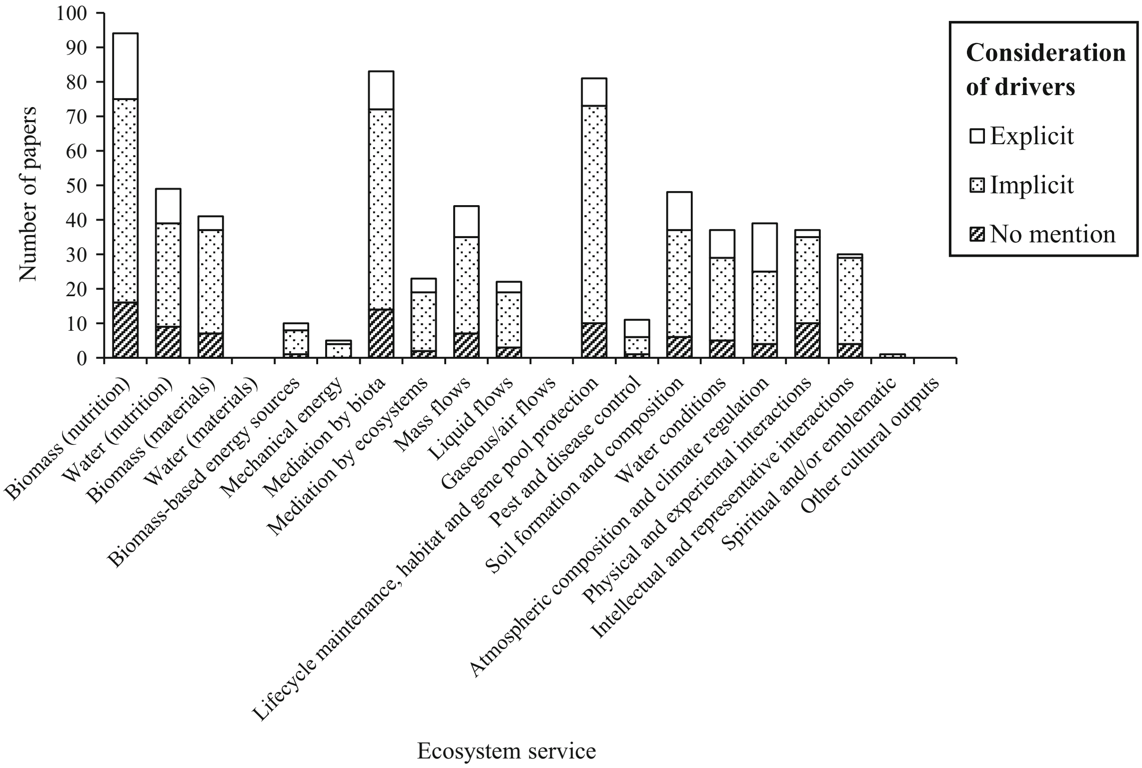

Fig. 3 The number of papers reviewed that assessed trade-offs and synergies involving each ecosystem service group, and whether these assessments explicitly, implicitly or did not mention the drivers of the relationships between the services

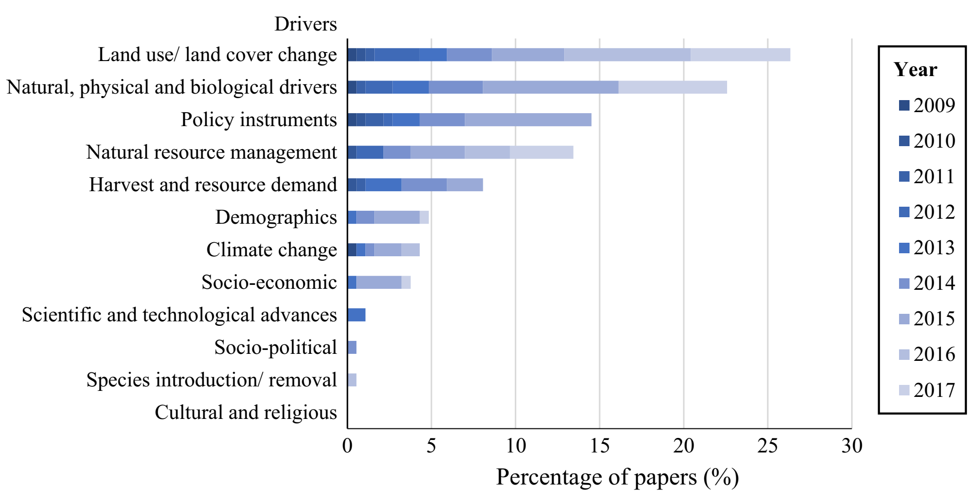

Fig. 4 The percentage of articles examined that focused on each driver of ecosystem service trade-offs and synergies, and their distribution across each year. No drivers were mentioned in the reviewed articles from 2005 to 2008

climate regulation services had the highest number of papers (36%, $n = 14$) explicitly identifying the drivers underpinning the relationships. Of the articles that did at least implicitly consider drivers of ecosystem service trade-offs and synergies ($n = 137$), there was a wide variety in the type of drivers identified (Fig. 4). The most commonly identified driver was land management actions related to land use/land cover change (26%, $n = 49$). This included drivers such as changes in vegetation cover and changing land use. Other commonly considered drivers of ecosystem service relationships were related to physical and biological changes to ecosystems (22%, $n = 42$) and policy instruments (14%, $n = 27$). The effect of cultural and religious practices as drivers of ecosystem service trade-offs and synergies were not considered by any of the articles examined.

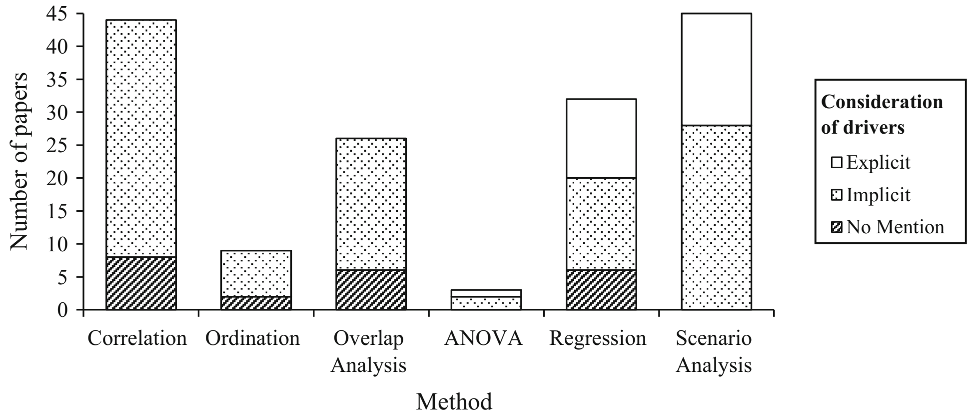

Fig. 5 Frequency of the different types of methods used to identify ecosystem service trade-offs and synergies, and the number of articles within each method that implicitly, explicitly or did not mention the potential drivers of the relationships. *ANOVA* analysis of variance

Methods used to assess ecosystem service trade-offs and synergies

A variety of methods were used to assess trade-offs and synergies among ecosystem services (Fig. 5). The most utilised method was scenario analysis (28%, $n = 45$). This approach estimates ecosystem service trade-offs and synergies by projecting the provision of ecosystem services under different scenarios, and then using these values to identify services that change in the same direction (suggesting a synergy) or in opposite directions (suggesting a trade-off). The second most commonly utilised method was correlation (28%, $n = 44$). This method identifies ecosystem service trade-offs and synergies by simply quantifying the statistical association between pairs of ecosystem services. Regression analysis was also a common method (20%, $n = 31$), but consisted of a wide variety of different techniques, including generalised linear models, logistic regression models and structural equation models. Overlap analysis was used less commonly (16%, $n = 26$). This approach employs a similar approach to correlation where trade-offs and synergies are identified based on spatial association or overlap. The least common methods utilised were ordination (6%, $n = 9$) and ANOVA (2%, $n = 3$).

There was a significant association between the methods used to identify ecosystem service relationships and whether the drivers and mechanisms were considered explicitly, implicitly or not at all (Chi square test: $p \leq 0.001$, $\chi^2 = 38.59$, df = 12). Articles that explicitly incorporated drivers into the assessments tended to use scenario analysis, regression or ANOVA to identify trade-offs and synergies (Fig. 5). On the other hand, articles where there was no mention of drivers tended to use correlation, ordination, overlap analysis or regression methods.

DISCUSSION

Since the Millennium Ecosystem Assessment was published in 2005 (MA 2005), there has been a rapid increase in the research into the relationships among ecosystem services driven by the need to manage for multiple services (Rodríguez et al. 2006; Bennett et al. 2009; Lee and Lautenbach 2016). Despite recognition of the importance of drivers and mechanisms in determining synergies and trade-offs among ecosystem services (Bennett et al. 2009), our results show that few assessments are explicit about these processes. There is particularly a lack of focus on human drivers of ecosystem service relationships, such as cultural values, species management and socio-political drivers. Nonetheless, there is evidence of improving recognition of the overall role of drivers and underlying mechanisms for assessing ecosystem service relationships. Future challenges lie in developing methods and data capable of a more explicit consideration of the drivers and mechanisms within socio-ecological systems relevant to ecosystem service provision. This will provide much greater confidence in predicting the consequences of policy interventions and other impacts on multifunctional outcomes.

Our review found that non-mechanistic methods are used more often than mechanistic methods to assess ecosystem service relationships. These non-mechanistic approaches, such as correlation and overlap analysis, have provided an important foundation for our understanding of ecosystem service relationships (Maes et al. 2012; Mouchet et al. 2014). However, they are unable to identify the causal drivers and mechanistic pathways that explain the relationships among services, limiting their ability to explicitly identify drivers (Iriondo et al. 2003; Sugihara et al. 2012). For example, one of the reviewed papers, Baral et al. (2013), determined that replacing agricultural land with intensely managed forest plantation was spatially correlated with an increase in carbon storage and the regulation of water quality in the Lower Glenelg Basin, Australia, which suggests using land for forest plantations is driving a synergy between carbon storage and water regulation. However, there may be other confounding variables, such technological advances and plant species diversity, which are affecting the provisioning of these services instead of land use intensification (George et al. 2012). Without explicit consideration of the mechanistic links between potential drivers and ecosystem service relationships it is still unclear what is actually driving the relationships between these services. As such, it is uncertain which variables new policies should target to best increase the provisioning of these services and avoid trade-offs. Instead, typical mechanistic approaches that were used in the papers we reviewed, including experiments, scenario analyses and process-based models, were more capable of identifying and characterising the effect of causal drivers. For example, one paper (Lauf et al. 2014) used a process-based simulation model, capable of modelling the underlying mechanisms and processes influencing ecosystem service provision, to identify the mechanistic pathway by which urbanisation drives a trade-off between energy production and food production in metropolitan Berlin, Germany. Furthermore, scenario analyses were often conducted using process-based models that allow the simulation of the consequences of alternative drivers on multiple ecosystem services (Bagstad et al. 2013). These mechanistic approaches are able to quantify the strength of the mechanistic links between ecosystem services and drivers in order to provide explicit information about trade-offs and synergies under different scenarios. This means they are much more likely to be able to effectively inform policy choices and avoid perverse outcomes (Lindenmayer et al. 2012).

Controlled experiments are the gold standard for identifying causal links between ecosystem variables (Schindler 1998). This approach was used in a small number of the reviewed articles, such as Classen et al. (2014), to identify how biodiversity drives a synergy between pest control and coffee production. However, the use of controlled experiments may be limited by the difficulty of controlling for multiple variables in the complex systems and at the broad spatial scales that are relevant for ecosystem services (Sutherland 2006; Martinez-Harms et al. 2015). For example, relationships among ecosystem services rarely involve only two services, but rather multiple services influencing each other that create a complex web of relationships. Due to the large number of variables present (Dee et al. 2017), it can be difficult to quantify these networks using controlled experiments. An alternative when experiments are not possible is causal inference that involves developing hypotheses about the causal links between variables, while controlling for confounding factors in the sampling and statistical design to test these hypotheses (; Rubin 2005; Pearl 2009; Law et al. 2017). This approach can be used to explicitly characterise the causal pathways by which drivers influence trade-offs and synergies between services as it controls for confounding variables (Law et al. 2017). For example, causal inference could be used to identify whether a driver influences one ecosystem service, which then leads to a change in another service, or whether it directly affects both services simultaneously. However, no causal inference approaches were used by any of the reviewed articles so this is a key area of development for future research. Process-based ecosystem service models were used in a number of the reviewed articles, including the ARIES and InVEST models (Nelson et al. 2009; Balbi et al. 2015). These types of models allow for the evaluation of the consequences of alternative scenarios (e.g. policy scenarios) for the relationships between ecosystem services, something that is not possible with purely correlative approaches. Therefore, there is great potential for applying these methods to identify general patterns in trade-offs and synergies under different drivers.

There was considerable variation in the number of papers focussing on each driver, but policy instruments were one of the most commonly studied drivers. This suggests that there is a strong recognition of the importance of policy decisions for influencing trade-offs and synergies among ecosystem services. Properly dealing with the mechanisms underlying relationships that emerge from policy interventions would appear to be a particularly high priority as it can lead to more effective policy interventions being implemented that avoid unexpected decreases in ecosystem service provisioning (Lindenmayer et al. 2012; Miteva et al. 2012; Ferraro and Hanauer 2014). However, achieving this can be difficult due to the complexity of interactions between the mechanisms that emerge from different policy actions and the relationships that occur among multiple ecosystem services (Dee et al. 2017). Thus, although there is recognition of the importance of policies as drivers, it is important that their influence is assessed using methods that can account for these complex relationships. Human drivers, such as cultural and religious, socio-political and scientific and technological drivers were the least considered drivers in our review. This may be due to a separation of the ecological and social sciences, which has led to a primary focus on the ecological processes underlying ecosystem service provision (Liu et al. 2007; Chan et al. 2012; Daniel et al. 2012). It could also be due to the difficulty in quantifying cultural factors, as our review found that few studies assessing trade-offs and synergies concerning cultural services explicitly identified the drivers underpinning these relationships. In recent years, there has been increased interest in assessing ecosystem services using a socio-ecological approach (Meacham et al. 2016). Applying more mechanistic approaches to identify ecosystem service relationships can aid in this due to their capacity to quantify the strength of linkages between social and environmental processes for ecosystem service provision (Spake et al. 2017). This could help identify trade-offs and synergies that are often ignored or misunderstood due to their social complexity (Daw et al. 2015), or because the social and environmental interactions are occurring in different locations, referred to as telecoupling (Liu et al. 2016).

The limited use of mechanistic approaches may simply reflect the often slow uptake of new methods, as they are often perceived as being risky, too difficult to implement or because awareness of them is limited (Marra et al. 2003). Data availability may also play an important role in determining whether drivers and mechanisms can be incorporated into assessments of ecosystem service relationships. In many cases, the necessary data may not be available, and this will likely depend on the drivers or ecosystem services being assessed, the type of data required, the spatial scale and available research budget (Spake et al. 2017; Bagstad et al. 2018). For example, drivers of cultural ecosystem service trade-offs and synergies may often require surveys and stakeholder interviews (Crouzat et al. 2016), which can be difficult to collect and incorporate into quantitative simulation models (Daniel et al. 2012). In this case, the integration of qualitative and quantitative data, through the use of mechanistic models can be a way forward to better reveal the relationships between ecosystem services (Martín-López et al. 2014). Therefore, when assessing ecosystem service trade-offs and synergies, we recommend that appropriate data collection to accommodate a mechanistic approach is identified early in the design phase to evaluate the data requirements and appropriate methodologies. This includes ensuring data are collected at an appropriate scale for analysing the mechanisms hypothesised as underpinning the ecosystem service relationships, and considering both social and ecological data requirements necessary to understand ecosystem service provisioning.

CONCLUSION

An incomplete understanding of ecosystem service trade-offs and synergies increases the likelihood of policy and management being ineffective, or being environmentally or financially costly (Kremen 2005; Degnbol and McCay 2007; Spake et al. 2017). From a review of the literature, it is evident that the majority of assessments of ecosystem service trade-offs and synergies are not explicitly identifying the drivers underpinning these relationships. If used to inform policy and management these assessments could lead to perverse outcomes, where management actions are introduced that that lead to unexpected changes in the provisioning of multiple ecosystem services. To prevent this, a challenge for the assessment of ecosystem service trade-offs and synergies lies in developing research with a greater emphasis on drivers and the mechanisms that link drivers to ecosystem services. This requires assessments of ecosystem service relationships considering drivers early in the design phase of research projects and encouraging greater uptake of methods, and collection of data, capable of identifying the mechanisms. A shift towards a more mechanistic understanding of the relationships between ecosystem services will result in better informed decisions for achieving sustainable and landscapes and maintaining human well-being.

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REFERENCES

Ai, J.Y., X. Sun, L. Feng, Y.F. Li, and X.D. Zhu. 2015. Analyzing the spatial patterns and drivers of ecosystem services in rapidly urbanizing Taihu Lake Basin of China. *Frontiers of Earth Science* 9: 531–545.

Alexander, S., C.R. Nelson, J. Aronson, D. Lamb, A. Cliquet, K.L. Erwin, C.M. Finlayson, R.S. De Groot, et al. 2011. Opportunities and challenges for ecological restoration within REDD+. *Restoration Ecology* 19: 683–689.

Allison, S.D., and K.K. Treseder. 2008. Warming and drying suppress microbial activity and carbon cycling in boreal forest soils. *Global Change Biology* 14: 2898–2909.

Bagstad, K.J., D.J. Semmens, S. Waage, and R. Winthrop. 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. *Ecosystem Services* 5: 27–39.

Bagstad, K.J., E. Cohen, Z.H. Ancona, S.G. McNulty, and G. Sun. 2018. The sensitivity of ecosystem service models to choices of input data and spatial resolution. *Applied Geography* 93: 25–36.

Balbi, S., A. del Prado, P. Gallejones, C.P. Geevan, G. Pardo, E. Pérez-Miñana, R. Manrique, C. Hernandez-Santiago, et al. 2015. Modeling trade-offs among ecosystem services in agricultural production systems. *Environmental Modelling & Software* 72: 314–326.

Baral, H., R.J. Keenan, J.C. Fox, N.E. Stork, and S. Kasel. 2013. Spatial assessment of ecosystem goods and services in complex production landscapes: A case study from south-eastern Australia. *Ecological Complexity* 13: 35–45.

Bennett, E.M., G.D. Peterson, and L.J. Gordon. 2009. Understanding relationships among multiple ecosystem services. *Ecology Letters* 12: 1394–1404.

Bennett, E.M., W. Cramer, A. Begossi, G. Cundill, S. Díaz, B.N. Egoh, I.R. Geijzendorffer, C.B. Krug, et al. 2015. Linking biodiversity, ecosystem services, and human well-being: Three

challenges for designing research for sustainability. *Current Opinion in Environmental Sustainability* 14: 76–85.
Chan, K.M.A., A.D. Guerry, P. Balvanera, S. Klain, T. Satterfield, X. Basurto, A. Bostrom, R. Chuenpagdee, et al. 2012. Where are cultural and social in ecosystem services? A framework for constructive engagement. *BioScience* 62: 744–756.
Classen, A., M.K. Peters, S.W. Ferger, M. Helbig-Bonitz, J.M. Schmack, G. Maassen, M. Schleuning, E.K.V. Kalko, et al. 2014. Complementary ecosystem services provided by pest predators and pollinators increase quantity and quality of coffee yields. *Proceedings of the Royal Society B-Biological Sciences*. https://doi.org/10.1098/rspb.2013.3148.
Cohen-Shacham, E., T. Dayan, E. Feitelson, and R.S. de Groot. 2011. Ecosystem service trade-offs in wetland management: Drainage and rehabilitation of the Hula, Israel. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques* 56: 1582–1601.
Crouzat, E., B. Martín-López, F. Turkelboom, and S. Lavorel. 2016. Disentangling trade-offs and synergies around ecosystem services with the influence network framework: Illustration from a consultative process over the French Alps. *Ecology and Society.*. https://doi.org/10.5751/ES-08494-210232.
Daniel, T.C., A. Muhar, O. Arnberger, J.W. Aznar, K.M. Boyd, R. Chan, T.Elmqvist Costanza, et al. 2012. Contributions of cultural services to the ecosystem services agenda. *Proceedings of the National Academy of Sciences* 109: 8812–8819.
Daw, T.M., S. Coulthard, W.W.L. Cheung, K. Brown, C. Abunge, D. Galafassi, G.D. Peterson, T.R. McClanahan, et al. 2015. Evaluating taboo trade-offs in ecosystems services and human well-being. *Proceedings of the National Academy of Sciences* 112: 6949–6954.
De Groot, R.S., R. Alkemade, L. Braat, L. Hein, and L. Willemen. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity* 7: 260–272.
De Groot, R.S., M.A. Wilson, and R.M. Boumans. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* 41: 393–408.
Dee, L.E., S. Allesina, A. Bonn, A. Eklöf, S.D. Gaines, J. Hines, U. Jacob, E. McDonald- Madden, et al. 2017. Operationalizing network theory for ecosystem service assessments. *Trends in Ecology & Evolution* 32: 118–130.
Degnbol, P., and B.J. McCay. 2007. Unintended and perverse consequences of ignoring linkages in fisheries systems. *ICES Journal of Marine Science: Journal du Conseil* 64: 793–797.
Ferraro, P.J., and M.M. Hanauer. 2014. Advances in measuring the environmental and social impacts of environmental programs. *Annual Review of Environment and Resources* 39: 495–517.
Ferraro, P.J. 2009. Counterfactual thinking and impact evaluation in environmental policy. *New Directions for Evaluation* 2009: 75–84.
George, S.J., R.J. Harper, R.J. Hobbs, and M. Tibbett. 2012. A sustainable agricultural landscape for Australia: A review of interlacing carbon sequestration, biodiversity and salinity management in agroforestry systems. *Agriculture, Ecosystems & Environment* 163: 28–36.
Griggs, D., M. Stafford-Smith, O. Gaffney, J. Rockström, M.C. Öhman, P. Shyamsundar, W. Steffen, G. Glaser, et al. 2013. Policy: Sustainable development goals for people and planet. *Nature* 495: 305–307.
Guerry, A.D., S. Polasky, J. Lubchenco, R. Chaplin-Kramer, G.C. Daily, R. Griffin, M. Ruckelshaus, I.J. Bateman, et al. 2015. Natural capital and ecosystem services informing decisions: From promise to practice. *Proceedings of the National Academy of Sciences* 112: 7348–7355.
Haase, D., N. Larondelle, E. Andersson, M. Artmann, S. Borgström, J. Breuste, E. Gomez-Baggethun, Å. Gren, et al. 2014. A Quantitative review of urban ecosystem service assessments: Concepts, models, and implementation. *Ambio* 43: 413–433.
Haines-Young, R.H., and M.B. Potschin. 2013. *Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August–December 2012*. EEA Framework Contract No EEA/IEA/09/003.
Howe, C., H. Suich, B. Vira, and G.M. Mace. 2014. Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change* 28: 263–275.
Iriondo, J.M., M.J. Albert, and A. Escudero. 2003. Structural equation modelling: An alternative for assessing causal relationships in threatened plant populations. *Biological Conservation* 113: 367–377.
Kandziora, M., B. Burkhard, and F. Müller. 2013. Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators—A theoretical matrix exercise. *Ecological Indicators* 28: 54–78.
Kremen, C. 2005. Managing ecosystem services: What do we need to know about their ecology? *Ecology Letters* 8: 468–479.
Law, E.A., P.J. Ferraro, P. Arcese, B.A. Bryan, K. Davis, A. Gordon, M.H. Holden, G. Iacona, et al. 2017. Projecting the performance of conservation interventions. *Biological Conservation* 215: 142–151.
Lawler, J.J., D.J. Lewis, E. Nelson, A.J. Plantinga, S. Polasky, J.C. Withey, D.P. Helmers, S. Martinuzzi, et al. 2014. Projected land-use change impacts on ecosystem services in the United States. *Proceedings of the National Academy of Sciences* 111: 7492–7497.
Lauf, S., D. Haase, and B. Kleinschmit. 2014. Linakges between ecosystem services provisioning, urban growth and shrinkage—A modeling approach assessing ecosystem service trade-offs. *Ecological Indicators* 42: 73–94.
Lee, H., and S. Lautenbach. 2016. A quantitative review of relationships between ecosystem services. *Ecological Indicators* 66: 340–351.
Lindenmayer, D.B., K.B. Hulvey, R.J. Hobbs, M. Colyvan, A. Felton, H. Possingham, W. Steffen, K.A. Wilson, et al. 2012. Avoiding bio-perversity from carbon sequestration solutions. *Conservation Letters* 5: 28–36.
Liu, J., T. Dietz, S.R. Carpenter, M. Alberti, C. Folke, E. Moran, A.N. Pell, P. Deadman, et al. 2007. Complexity of coupled human and natural systems. *Science* 317: 1513–1516.
Liu, J., S. Li, Z. Ouyang, C. Tam, and X. Chen. 2008. Ecological and socioeconomic effects of China's policies for ecosystem services. *Proceedings of the National Academy of Sciences* 105: 9477–9482.
Liu, J., W. Yang, and S. Li. 2016. Framing ecosystem service in the telecoupled anthropocene. *Frontiers in Ecology and the Environment* 14: 27–36.
MA (Millennium Ecosystem Assessment). 2005. *Ecosystems and human well-being*. Washington, DC: Island Press.
Maes, J., M.L. Paracchini, G. Zulian, M.B. Dunbar, and R. Alkemade. 2012. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. *Biological Conservation* 155: 1–12.
Marra, M., D.J. Pannell, and A.A. Ghadim. 2003. The economics of risk, uncertainty and learning in the adoption of new agricultural technologies: Where are we on the learning curve? *Agricultural Systems* 75: 215–234.
Martín-López, B., E. Gómez-Baggethun, M. García-Llorente, and C. Montes. 2014. Trade- offs across value-domains in ecosystem services assessment. *Ecological Indicators* 37: 220–228.

Martinez-Harms, M.J., B.A. Bryan, P. Balvanera, E.A. Law, J.R. Rhodes, H.P. Possingham, and K.A. Wilson. 2015. Making decisions for managing ecosystem services. *Biological Conservation* 184: 229–238.

Meacham, M., C. Queiroz, A.V. Norström, and G.D. Peterson. 2016. Social-ecological drivers of multiple ecosystem services: What variables explain patterns of ecosystem services across the Norrström drainage basin? *Ecology and Society*. https://doi.org/10.5751/ES-08077-210114.

Miteva, D.A., S.K. Pattanayak, and P.J. Ferraro. 2012. Evaluation of biodiversity policy instruments: What works and what doesn't? *Oxford Review of Economic Policy* 28: 69–92.

Mouchet, M.A., P. Lamarque, B. Martín-López, E. Crouzat, P. Gos, C. Byczek, and S. Lavorel. 2014. An interdisciplinary methodological guide for quantifying associations between ecosystem services. *Global Environmental Change* 28: 298–308.

Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D. Cameron, K. Chan, G.C. Daily, et al. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment* 7: 4–11.

Pearl, J. 2009. Causal inference in statistics: An overview. *Statistics Surveys* 3: 96–146.

Potschin, M.B., and R.H. Haines-Young. 2011. Ecosystem services: Exploring a geographical perspective. *Progress in Physical Geography: Earth and Environment* 35: 575–594.

Raudsepp-Hearne, C., G.D. Peterson, and E. Bennett. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences of the United States of America* 107: 5242–5247.

Rey Benayas, J.M., A. Martins, J.M. Nicolau, and J.J. Schulz. 2007. Abandonment of agricultural land: An overview of drivers and consequences. *CAB Reviews: Perspectives in Agriculture, Vetrinary Science, Nutrition and Natural Resources* 2: 1–14.

Rodríguez, J.P., T.D. Beard, E.M. Bennett, G.S. Cumming, S.J. Cork, J. Agard, A.P. Dobson, and G.D. Peterson. 2006. Trade-offs across space, time, and ecosystem services. *Ecology and Society* 11: 28.

Rounsevell, M., T. Dawson, and P. Harrison. 2010. A conceptual framework to assess the effects of environmental change on ecosystem services. *Biodiversity and Conservation* 19: 2823–2842.

Rubin, D.B. 2005. Causal inference using potential outcomes: Design, modeling, decisions. *Journal of the American Statistical Association* 100: 322–331.

Runting, R.K., B.A. Bryan, L.E. Dee, F.J.F. Maseyk, L. Mandle, P. Hamel, K.A. Wilson, K. Yetka, et al. 2017. Incorporating climate change into ecosystem service assessments and decisions: A review. *Global Change Biology* 23: 28–41.

Schindler, D.W. 1998. Whole-ecosystem experiments: Replication versus realism: the need for ecosystem-scale experiments. *Ecosystems* 1: 323–334.

Schröter, D., W. Cramer, R. Leemans, I.C. Prentice, M.B. Araújo, N.W. Arnell, A. Bondeau, H. Bugmann, et al. 2005. Ecosystem service supply and vulnerability to global change in Europe. *Science* 310: 1333–1337.

Spake, R., R. Lasseur, E. Crouzat, J.M. Bullock, S. Lavorel, K.E. Parks, M. Schaafsma, E.M. Bennett, et al. 2017. Unpacking ecosystem service bundles: Towards predictive mapping of synergies and trade-offs between ecosystem services. *Global Environmental Change* 47: 37–50.

Stutter, M.I., W.J. Chardon, and B. Kronvang. 2012. Riparian buffer strips as a multifunctional management tool in agricultural landscapes: Introduction. *Journal of Environmental Quality* 41: 297–303.

Su, C.H., B.J. Fu, C.S. He, and Y.H. Lu. 2012. Variation of ecosystem services and human activities: A case study in the Yanhe Watershed of China. *Acta Oecologica- International Journal of Ecology* 44: 46–57.

Sugihara, G., R. May, H. Ye, C.H. Hsieh, E. Deyle, M. Fogarty, and S. Munch. 2012. Detecting causality in complex ecosystems. *Science* 338: 496–500.

Sutherland, W.J. 2006. Predicting the ecological consequences of environmental change: A review of the methods. *Journal of Applied Ecology* 43: 599–616.

Tallis, H., P. Kareiva, M. Marvier, and A. Chang. 2008. An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences* 105: 9457–9464.

Turkelboom, F., M. Leone, S. Jacobs, E. Kelemen, M. García-Llorente, F. Baró, M. Termansen, D.N. Barton, et al. 2018. When we cannot have it all: Ecosystem service trade-offs in the context of spatial planning. *Ecosystem Services* 29: 566–578.

Villamagna, A.M., P.L. Angermeier, and E.M. Bennett. 2013. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. *Ecological Complexity* 15: 114–121.

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