Chapter 1

Prioritising Land-Use Decisions for the Optimal Delivery of Ecosystem Services and Biodiversity Protection in Productive Landscapes

Emily S. Weeks, Norm Mason,
Anne-Gaelle E. Ausseil and Alexander Herzig

Additional information is available at the end of the chapter

http://dx.doi.org/10.5772/58255

1. Introduction

Over the past 50 years, ecosystems have changed more rapidly than at any other period of human history [1]. Considerable portions of the world's thirteen terrestrial biomes are being converted to less ecologically diverse ecosystems [2]. Such a high degree of conversion is leading to extensive changes in biodiversity composition and ecological processes, which results in the diminishing of the ecosystem services that help sustain biological diversity and human populations [3].

Estimates of current extinction rates are several magnitudes above average extinction rates through geological time [4]. Some biologists suggest that a sixth mass extinction is underway, but there is large uncertainty in estimates of global extinction rates [5]. Recently, however, there has been considerable evidence for widespread loss of species at the local and regional level. Studies have shown that the loss of biodiversity at this level has led to the simplification of ecosystem function and resilience [6], and is altering key process important to productivity and sustainability of Earth's ecosystems.

Biodiversity is considered to provide a range of services of varying values to humanity [7] associated with the normal functioning of both their individual components and different combinations of these components in integrated functional ecological systems (Figure 1).The type and level of service inevitably varies among ecosystems but each one can contribute significantly depending on type and their degree of intactness. As the human population increases so do the demands on most ecosystem services provided by indigenous ecosystems, but their ability to provide these services generally decreases with increasing degradation of the system. This is generally associated with increased human population pressure. Understanding this complex relationship in a particular ecosystem or related ecosystems is most important and should be an integral component of the planning of ecosystem management.

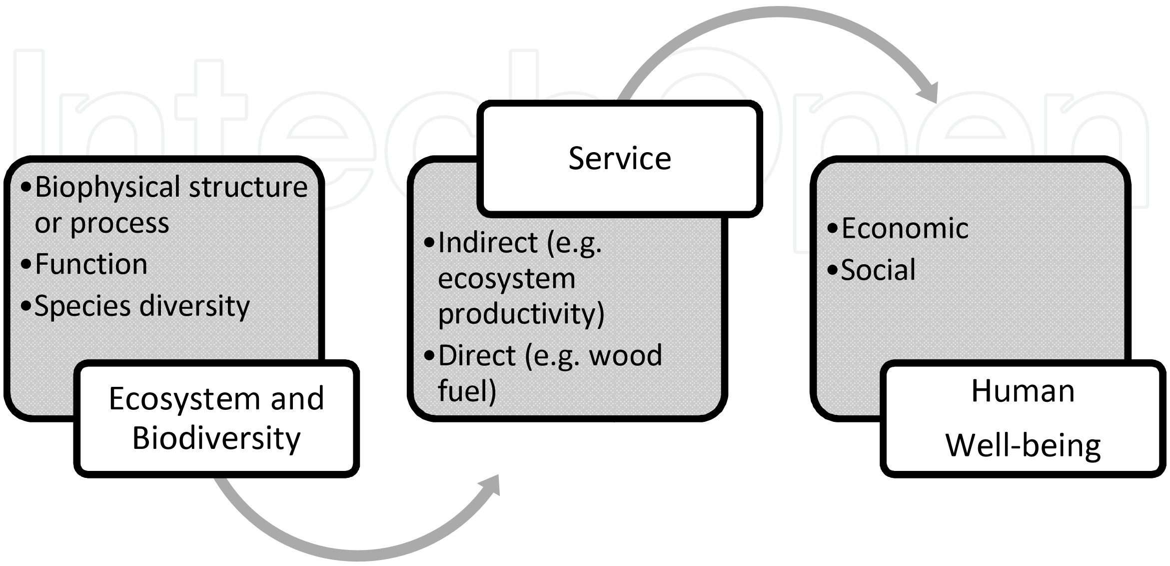

Figure 1. The links between natural ecosystems and human well-being adapted from Haines-Young et al. [7].

A global synthesis reveals that biodiversity loss is a major driver of ecosystem change [6, 8]. There are numbers of causes of biodiversity loss and differences across ecosystems, including: land-use change, climate change, nitrogen deposition, biotic exchanges, and atmospheric carbon dioxide changes [9]. Land-use change has been shown to be one of the leading causes of biodiversity loss in terrestrial ecosystems [3, 10, 11]. An increasing global population and greater demand for food, fodder, fibre and fuel has led to rapid changes in land-use patterns. Areas of low production value, once considered impervious to human activity, have increasingly become susceptible to intensive land-use changes [2, 12, 13].

There is growing evidence of disconnection or opposition between environmental conservation and socio-economic development. In 2005, The Millennium Assessment showed that changes to ecosystems have contributed to human well-being and economic development, but this has been achieved at the expense of many ecosystem services, and increased poverty for some groups of people [1]. Furthermore, the degradation of ecosystem services could escalate during the first half of this century.

A global study, The Economics of Ecosystems and Biodiversity (TEEB, more information at http://www.teebweb.org/), recently revealed the global economic benefit of biodiversity and made the case for better natural resources management [10]. As part of the process, an Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, www.ipbes.net) has been established as a follow-up initiative to the Millenium Ecosystem Assessment [1]. Since then, there has been significant interest in converting the concept of ecosystem services into practice, both as a rationale for conservation of biological diversity and as a method to design policies that maximize benefits from the sustainable management of ecosystems. Yet despite this level of attention, understanding of the relationship between biodiversity and ecosystem function is far from complete and is biased towards a small number of ecosystems [14, 15].

The following chapter provides an overview of some the research completed over the last decade for ecosystem services and biodiversity protection. It discusses the general question of how best to allocate land use (or prioritise decisions) to ensure optimal delivery of both ecosystem services and biodiversity protection in productive landscapes. The first section explores the links between biodiversity and ecosystem function and then discusses the role of biodiversity in productive landscapes. This is followed by a brief overview of the different tools currently available that aim to prioritise land-use decisions based on the optimal delivery of ecosystem services and/or biodiversity protection within productive landscapes. The last section highlights challenges and opportunities for multi-objective land-use planning and concludes with recommendations for future research.

2. Linking biodiversity to ecosystem services

Although there are good estimates of society's willingness to pay for various non-marketed ecosystem services, as yet there is no universally accepted framework for assigning values to biological diversity. One of the most useful frameworks to value biodiversity divides direct and indirect use values. Direct-use values can be readily calculated by observing the activities of representative groups of people, by monitoring collection points for natural production, and by examining import and export statistics. Indirect-use values are harder to measure. They are based on the indirect benefits people gain from biological diversity such as ecosystem productivity, water purification, climate regulation, eco-tourism, and recreation [16].

The role of biodiversity in ecosystem function and services is widely debated. Most theoretical and empirical work on measuring this relationship has focused on the congruence between species richness and ecological function. Some studies have found a strong link between the two [8, 17], while others have found little evidence supporting these findings [19]. A review of existing studies illustrates that only a few empirical studies demonstrate improved function with high level of species richness [19]. This suggests changes in species evenness (relative abundance) may deserve greater attention than species richness [20]. Species evenness has a more immediate impact on ecosystems and in most services few dominant species play a major role [6]. In most cases many species are critical for a range of ecosystem functions.

Recent studies have found that certain biodiversity facets co-occur with ecosystem services [21–25]. Large ecosystems, e.g. grasslands, show a high overlap along with ecosystems that provide a range of services that could be used to justify conservation action [21]. There is also a positive relationship between many ecosystem services and high species richness illustrated in several studies that reveal a positive relationship between species richness and productivity [17, 25–27]. However, with some services, such as soil services, the relationship with species diversity is not clear, indicating no single biodiversity measure could be used as a substitute for ecosystem services, and vice versa.

Global efforts to conserve biological diversity have the potential to deliver economic benefits to people [28]. Whether biodiversity conservation could be justified on economic grounds depends on the scale over which benefits are measured [29] or the policy context [30]. A recent study [29] found that at local and global scales conservation benefits outweigh the benefits offered by development, but the balance changes when assessing economic benefits at the national level. For example, the strong positive correlations between species richness and productivity might be interpreted as a win-win for economic growth and the protection of species; however, the economic benefit of conserving high value land at the national scale is limited, and conservation is more likely through smaller scale reserve selection, which would likely benefit only a few species.

Greater diversity in terms of numbers of species may be linked to greater system productivity [19]. For example, decreased local diversity can lead to lower ecosystem productivity, lower use of limiting resources, and lower temporal stability. This is of potential relevance to economists because it provides evidence of the direct value of biodiversity to ecosystem function and the services provided to society by ecosystems. However, there are a number of ecosystem services that are not adequately represented by productivity measures, such as pollination and control of pests. This presents an on-going joint challenge between economists and ecologists – how to quantify these services in a way that can be valued by tools of economists.

3. Scheduling conservation action into productive landscapes

3.1. Potential conservation value of productive landscapes

Production ecology and conservation biology have long been viewed as two opposing approaches to managing ecological systems. With increasing global food demand projected in the next 50 years and decreasing biological diversity that is vital to future productivity, separating the two fields seems counter-productive. A study by Broussard *et al.* [31] evaluated a range of approaches to agricultural management and found that by integrating an ecological approach into environmental management it is possible to feed growing populations without further encroaching into natural systems. To do so, however, may require adopting a landscape management approach to intensification, biodiversity, and ecosystem service protection [32].

There is growing evidence that productive land use can contribute to the conservation of biological diversity [33–36]. Some studies highlight the importance of population exchanges among areas of different disturbance regimes and among early and late successional habitats [37, 38]. Though intensified land use is undeniably the main cause of biodiversity loss, there are opportunities for low intensity land-use systems or those in the form of polycultures and/or agroforestry patterns to play important roles in large-scale biodiversity conservation. Moreover, pockets of native vegetation found in productive landscapes are refuges for native flora and fauna [39–41] and provide land corridors for a range of wildlife [42]. These fragmented landscapes are also of great importance for the establishment of studies related both to species preservation in the long term (including the re-introduction and translocation of species) and to the genetic health of isolated populations. Remaining riparian habitats can also play a major role for both humans and nature in productive landscapes [35] by providing habitat for wildlife and maintaining important ecosystem services (such as clean water) that are important for productivity.

In addition to contributing to biodiversity protection, productive landscapes are interrelated to a range of ecosystems services that are associated with biological conservation. Such landscapes receive services such as pollination, soil fertility, and water retention from surrounding natural systems but also contribute to services such as soil retentions and food production. The approach adopted for managing productive landscapes can have significant impact on the services on which it depends or provides. Water quality, pollination and nutrient cycling, soil retention, carbon sequestration, and biodiversity conservation are all highly vulnerable to changes in management practices. Some relationships are easy to identify, while others are much more difficult to measure [43]. For example, the relationship between the number of pollinators, crop yields, and the use of pesticides is easy to identity (pollinators will directly increase with crop yield and decrease with increased use of insecticides), while the benefits of wetlands are much more indirect (wetlands reduce the load of nitrogen in surface water resulting from agricultural fields).

Another important ecosystem service that has been associated with biodiversity is natural pest control [32, 44, 45]. Natural pest control provides environmental and economic benefits. Although productive landscapes with networks of natural habitat can provide refuge for a range of pests [46], there is evidence that multiple non-productive habitat types may also favour natural pest control (e.g. grassland, herbaceous wooden habitats and wetlands) [44, 47, 48]. Spatial scale and the distribution of natural habitat may influence the natural pest control function. For example, diverse small-scale landscapes provide better conditions for natural pest control than do large-scale landscapes [49]. Overall, there is a need for more studies to quantify the effects of landscape composition on natural pest control, and further investigation into the benefits biodiversity restoration programmes may offer to productive landscapes.

3.2. Integrated conservation planning in productive landscapes

There is an increasing expectation that productive (i.e. agricultural) landscapes should be managed to preserve or enhance biodiversity (e.g. [50]). Often, the impacts of pressures associated with productive landscapes (and management interventions aimed at mitigating them) are assessed using local measures, such as native species richness or dominance. However, it is questionable how relevant such measures are for national-scale conservation priorities, since they may merely reflect changes in the occurrence and abundance of common, unthreatened indigenous species [50]. Ideally, any attempts to enhance biodiversity in productive landscapes should contribute to national conservation objectives. Integrated conservation planning [51] provides an obvious means for achieving this.

Generally, two independent strands contribute to integrated conservation planning – ecosystem-centred and species-centred prioritisation [52]. An ecosystem-centred approach prioritises efforts that increase the representation of indigenous biodiversity across the full range of environment, ecosystem, and habitat types by enhancing or protection highly modified ecosystem types (thus enhancing or protecting Environmental Representation). A species-centred approach prioritises species based on their conservation status, or some measure of current vs potential distribution – with conservation efforts benefiting the most severely threatened species receiving the highest priority. Some frameworks also consider existing conservation efforts in prioritising new efforts. For instance, threatened species or environment types that already receive a high degree of protection may be assigned lower priority than those that receive little or no protection.

Clearance of indigenous vegetation for agriculture and land-use intensification has severely reduced indigenous biodiversity representation within productive lowland ecosystems (i.e. has reduced environmental representation), so that there is often little or no remnant habitat available for conservation (e.g. [53, 54]. Consequently, ecological restoration is necessary to ensure representation of these ecosystems in conservation networks [53, 55, 56].

In many countries, clearance of indigenous vegetation has been especially severe in environments of limited geographic extent, such as coastal and riparian habitats (e.g. [57]), or ecosystems on unusual substrates (e.g. [58, 59]). Thus areas providing very high gains in environmental representation through restoration or protection will often occupy very small sites. This leads to a right-skewed distribution where most sites provide low environmental representation gains while a few sites provide very large gains. Because environmental representation gain will often be strongly right-skewed, it may be especially vulnerable to trade-offs in multi-objective optimisations of restoration effort. This arises because high values for environmental representation gain are unlikely to co-occur with high values for ecosystem service gains [60]. This means that when ecosystem service benefits are included as criteria for deciding where to apply restoration effort, environmental representation gains will often be much lower than if it were the only criterion. The environmental representation strand of integrated conservation planning thus reveals that a focus on non-biodiversity objectives in designing restoration programmes may result in drastically lower rates of biodiversity gain per unit of restoration effort.

Perhaps the most important implication of integrated conservation planning for biodiversity enhancement schemes is that programmes focussed on the farm scale will likely be very inefficient at contributing to national biodiversity objectives. Not all farms will contain significant areas of highly modified environment types. Hence the potential gain in environmental representation for many farms will often be quite low. Similarly, few farms are likely to contain any threatened species, or have the potential to provide suitable habitat for threatened species. Therefore, any scheme that operates primarily by incentivising individual landowners to manage for biodiversity will result in relatively low gains in national-level conservation priorities per unit effort. By contrast, schemes focussing on the landscape scale will be able to target resources to areas where the potential gains are highest.

This has obvious implications for designing funding models and legislative frameworks to enhance biodiversity in productive landscapes. For instance, if society decides that biodiversity enhancement in a requirement for agricultural industries to obtain a licence to operate, it may be inefficient to demand that every landowner embarks on a significant biodiversity enhancement programme. Rather, it will be more efficient for biodiversity enhancement to operate at the industry level, where industry bodies collect fees from landowners which are then used to fund management in areas of high potential biodiversity gain. Similarly, it would be inefficient to offer every landowner a subsidy in return for carrying out biodiversity enhancements. Rather, it would be better to target subsidies at landowners whose farms contain large areas of high potential biodiversity gain.

4. Land-use planning for ecosystem services and biodiversity protection in productive landscapes

4.1. Spatial optimisation of ecosystem services

Spatial optimisation is a powerful method to explore the potentials of a given area to improve the spatial coherence of land-use functions. It is suitable for identifying land-use configurations which optimally match with spatially varying ecosystem characteristics as well as stake-holder expectations.

Spatial optimisation models have been successfully used to address complex spatial planning problems [61–65] including forest management and timber harvest [66], agricultural issues [61, 65, 67], general issues of land-use change [68], and habitat suitability [69]. Modelling methodologies range from dynamic models based on difference equations of exponential growth [66, 69] to complex models based on systems of non-linear differential equations [70].

The complexity of an optimisation model depends on the complexity of the ecosystem (number of variables, degree of non-linearity, etc.) and the spatial complexity (size of the study area, grid cell size, number of spatially interacting processes). Within land-use planning linear optimisation methods are often not applicable because of the qualitative character of the relations and the large number of variables and/or relations to be optimised. In this case, heuristic methods such as Genetic Algorithms are applied, given that there are few restrictions regarding the formulation of the variables and their relations [61].

Using spatial optimisation tools that systematically consider a range of scenarios, objectives, constraints, and stakeholder or societal preferences helps decision-makers gain insight into the full spectrum of feasible solutions. The tools also allow them to explore opportunities creatively in relation to the imposed limits. However, such use also can result in a simplified representation of options and trade-offs. The accuracy of the result of a spatial optimisation exercise depends on the quality of the input data and the complexity of the model. The more complex the model and the more spatial relationships considered, the greater the uncertainty in the optimisation. Furthermore, a relatively stable land-use pattern indicates a larger degree of freedom in terms of planning alternatives, whereas a relatively unstable land-use pattern indicates there is little room for trade-offs without significantly changing the expectations (i.e. constraints) [71].

In recent years spatial optimisation tools have also been used to link supply of ecosystem services to land use, climate and soil information [71]. Spatial land-use optimisation techniques can help to raise awareness on trade-offs and understand how a landscape configuration could be optimally manage for ecosystem services. LUMASS, for example, is a freeware that has been specifically developed to address this situation [71]. It is a multi-objective decision making tool that can be used for spatial optimisation of ecosystem services. It uses linear programming to optimise the spatial allocation of resources to satisfy an objective (single or multiple), subject to some constraints. Objectives and constraints are specified with regard to a set of criteria representing indicators (such as sediment loss, nitrate leaching and carbon sequestration) of ecosystem services (Figure 2). The optimal allocation of (area of) land uses across the land scape is expressed by the decision variables.

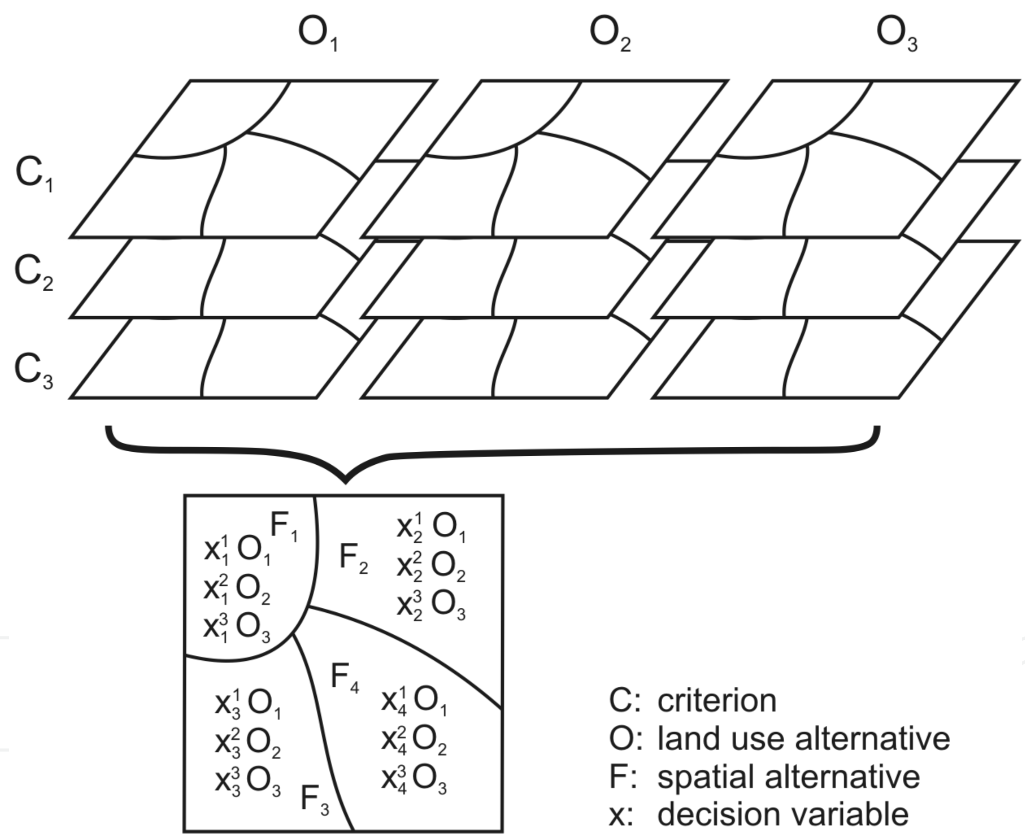

Figure 2. Diagram of the layers incorporated into a multi-objective decision making tool (adapted from71).

The accuracy of the result of a spatial optimisation model depends critically on the quality of the input data. Performance scores used as input data for an optimisation module can be derived from quantitative process-based landscape models or from expert empirical knowledge. Little is known about the impact of uncertain input data in terms of performance scores and constraints on the produced land-use pattern. However, Herzig et al. [73] found that uncertainty associated with input performance scores may lead to an overestimation of the optimisation benefits.

4.2. Modelling patterns of land-use change

Modelling patterns of land-use change is a fundamental component of conservation planning in productive landscapes [74–78]. Land-use change models can provide a tool for capturing the essence of where land-use change is likely to take place and what is driving these patterns [75, 79] This information can then be used to assess the vulnerability of remaining indigenous habitats and help identify their relative urgency for protection.

Assessment of the vulnerability of species and habitats to imminent proximate threats such as land-use change is a fundamental component of conservation management and planning [76, 77, 80]. Spatially explicit models can provide a tool for capturing the essence of where land-use change is likely to take place and what is driving these patterns [79]. This offers the opportunity to develop an understanding of how land-use change responds to different changes in policy or land-management plans. They can also be used to assess the vulnerability of remaining indigenous habitat and help identify the relative urgency of their protection.

Over the past two decades a range of models of land-use change have been developed, including process-based and statistical models [81]. Statistical models typically rely on the implicit assumption that land-use change processes are stationary. Process-based models, on the other hand, are able to deal with temporal changes in driving forces or processes. While process-based models have sound theoretical basis, statistical models can be easier to implement [82]. As a result, most land-use change models have relied more on a statistical approach to land-use change modelling. These models include Markov, logistic function, and regression models [83–88].

Each land-use change model adopts different statistical techniques to capture present and future land-use patterns. These techniques are either regression or transition based. The regression-based approach is used to understand historical, current, and future land-use patterns by establishing a relationship between a wide range of environmental or socio-economic variables [82]. The influence of these locational factors on land-use change is modelled with distance decay functions, where influence decreases with increasing distance from some feature [89]. In comparison, the transition-based models are rooted in a stochastic Markov-chain technique, where the transition probability is calculated by determining how often the system moved from one state to another [83].

Most land-use change models are spatial transition models and are designed to be part of a decision support system aimed to capture present and future land-use patterns [81, 84, 85, 90]. These models have relied on user-supplied assumptions about how people actually used the land [91]. These assumptions are based on the 'maximum power principle', i.e. people will use the most economically productive land first [92]. However, recent observations of land-use conversion indicate these patterns are changing [93]. Economically viable land is becoming less available, although demand is rapidly increasing [94, 95]. This suggests current and future land-use trends are no longer following traditional decision-making processes supported by the maximum power principle.

Spatial transition models are useful for land-use change modelling because of their robustness but they do not account for temporal heterogeneity. Land-use decisions are often triggered by single events such as economic crises and/or are often remote in space and time and operate at a higher hierarchical level [93]. In these cases, process-based models or models using an economic framework provide better representations of the decision making process. However, for systematic conservation planning these different models may complement each other – logistic regression models effectively identifying areas vulnerable to change, while process-based models can be used to better understand the drivers of these changes.

Most models attempt to illustrate the temporal patterns of land-use change [12, 79, 88, 96]. Quantifying the extent of future land-use change is difficult in itself, given the numerous social, political, and economic factors that drive such change [84, 97–99]. As a result, decision makers often prefer to use a Boolean map (change/no change) to illustrate the temporal and spatial patterns of land-use change. Boolean maps are easier to interpret and decipher and allow decision makers to compare scenarios of land-use change and make management decisions based on the interpolations of the various scenarios.

Because land-use systems respond to a combination of proximate (biophysical) and ultimate (socio-economic) drivers, modelling land-use change ideally requires a multidisciplinary approach [81, 100, 101]. It is often useful to incorporate a wide range of socio-economic and environmental predictors of change; particularly because the individual importance of factors in explaining patterns of land use for a past period will not necessarily reflect their ability to predict a future landscape. The predictive strength of empirical (observed) patterns of land-use change can be enhanced or diminished based on the combination of different. This also underlies the importance of validation in the modelling process [75].

Because land-use change is a dynamic threat, it is important that practitioners keep abreast of change and regularly validate the utility of their vulnerability assumptions and models [80, 102]. A common weakness of land-use modelling is the use of the same data for both calibration (making the model as consistent as possible with the data from which the parameters were estimated) and validation (assessment of the predictive power of the model) [87]. Lack of consideration of model uncertainty through rigorous validation has been shown to result in inaccurate and over-confident predictions. Validation therefore requires testing the predictions of independent data (i.e. not those used in model parameterisation) to ensure the relationships inferred by a model are robust and the predictions are reliable [75].

Spatial statistical models provide a tool for predicting where land-use change is most likely to take place [75, 79, 91]. However, models based on patterns of past change will not necessarily provide reliable predictions of future change because over time exhaustion of formerly suitable areas and changes in global markets, technology, and crops can alter both the distribution and rate of habitat conversion [77, 103]. The reliability of land-use change predictions is therefore likely to decrease as they are projected further into the future, risking misallocation of scarce conservation resources [75]. However, though recent land-use change data will provide the best estimates of future land-use change, vulnerability estimates based on too narrow a time range may also provide less accurate forecasts, because they are based on a small sample of conversion events.

4.3. The application of conservation planning tools for ecosystem services

Systematic conservation planning is the process of identifying and configuring complementary actions required to achieve conservation goals [73, 76]. Since the 1980s, numerous spatial approaches have been developed for identifying priority areas for conservation [103]. These approaches rely on information on the distribution of biodiversity (e.g. [104]); the distribution and effects of threatening processes or 'pressures' on biodiversity (such as pest, weeds, pollution and habitat conversion) and consequent vulnerability (the likelihood or imminence of biodiversity loss to current or impending threatening processes, in the sense of Pressey et al [51] and Wilson et al. [77]; and the effects, and costs, of potential management of pressures [105, 106].

A number of software packages have been developed for conservation planning and resource allocation. Among these are Marxan [107], C-Plan [108], ResNet [109] and Zonation [112, 113]. The two most widely used tools for conservation planning that have integrated ecosystem series into their configuration are Marxan and Zonation.

Marxan is popular with conservation practitioners worldwide and has been applied to both terrestrial and marine ecosystems [107]. It is designed to identify a set of planning units that meets a number of targets for a minimum cost. It can be applied to a variety of conservation features considered in conservation planning and can incorporate ecological processes, site condition, or socio-political influences [110]. It addresses most objectives typically considered in conservation planning and also uses a flexible algorithm that has a variety of applications.

Chan et al. [111] were the first to publish the integration of ecosystem services into Marxan. They compared differences in service provisions between conservation and development in the central coast of California. Marxan has also been used to examine the spatial congruence between biodiversity and ecosystem services in South Africa [21]. In both cases, Marxan produces a map of reserve networks that captured all biodiversity and economic targets at an optimal cost. However, the integration of ecosystem services into the Marxan framework still requires further development and the application should be used with caution. The tool lacks several features that are required for ecosystem service planning [16, 111].

Zonation [112, 113] differs from Marxan and other conservation planning approaches in that it primarily produces problem ranking rather than meeting targets at a minimum cost. Zonation can produce a priority ranking across an entire landscape using large data sets, while also identifying fine-scale prioritisation of biodiversity. Zonation provides priority ranking that balances the needs of biodiversity and competing land uses [114]. This capability, along with its ecologically based model of conservation value, makes it possible for Zonation to incorporate the ecosystem services provision into the prioritisation process

Both Marxan and Zonation address the basic principles of conservation planning. However, neither account for the dynamic processes such as on-going habitat loss, site availability rates, changing or unknown acquisition costs, species-specific connectivity requirements, or temporally varying distributions of features [115]. Nor do they formally incorporate multiple conservation actions such as land acquisition, restoration or easements. Furthermore, both software packages allow for detailed non-linear process descriptions and/or account for sophisticated spatial neighbourhood relationships. However, they are predominantly focused on conservation biology and hence offer only limited flexibility to configure the number and type (i.e. minimisation or maximisation) of objective functions as well as the specification of constraints. It would therefore appear they are less applicable to general land-use pattern optimisation for maximising ecosystem services.

5. Challenges and opportunities for multi-objective land-use planning

5.1. Funding multiple objectives

Despite the mounting interest in focusing conservation efforts on ecosystem services, there is still much debate over the implications for the protection of biological diversity. There is growing evidence of disconnection or opposition between environmental conservation and socio-economic development. Although the maintenance of ecosystem services is often used to justify biodiversity conservation actions, it is still unclear how ecosystem services relate to different aspects of biodiversity and to what extent the conservation of biodiversity will ensure the provision of services.

Part of the difficulty of using biodiversity as a measure of success is that its link to value is unclear. Value of ecosystem services, on the other hand, is easier to define and provides a useful common metric and measure of success [116–118]. Budgets for biodiversity conservation are thinly stretched and thus measuring success is essential to ensuring that scarce funds are optimally used. This recent switch to ecosystem services rather than biodiversity as an organizing principal has led to a more trans-disciplinary approach to biodiversity planning. Though the concept of ecosystem services is anthropogenic (measurements of success are particularly focused on monetary gains or losses), tools such as Ecosystem Services Valuation aim to build and link economic and ecological/biological metrics and work out solutions that correspond to optimal social and ecological decisions. Providing a monetary value to ecosystem services also provides a mechanism to uncover which economic decisions affect biodiversity of an ecosystem and consequently resilience, productivity and value of services.

An ecosystem service approach to managing productive landscapes offers a way to align multiple objectives such as the protection of biodiversity and increased land productivity. Where traditional approaches focus on setting aside land for protection the ecosystem service approach aims to engage a wider range of stakeholders and integrate economic incentives into the planning framework. This is particularly important given that the majority of Earth's natural systems, (containing important biodiversity and provide key ecological services) rest outside protected areas and it is projected that human impacts on these systems are to continue to intensify. In recent years ecosystem services projects have attracted on average more than four times as much funding and are more likely to expand opportunities for conservation [119, 120]. Given that ecosystems services projects are engaging a wider set of funders and becoming increasingly popular around the globe, there is a great need to continue to build alignment between biodiversity protection, human well-being and the delivery of ecosystem services.

There is growing support for understanding the economic costs and benefits of conserving ecosystems, particularly if it will help allocate scarce dollars more efficiently. Investments in biodiversity conservation may be strategically aligned to ecological services of high economic value, and vice versa. By explicitly valuing the costs and benefits associated with services, it may be possible to achieve meaningful biodiversity conservation at lower costs with greater co-benefits [111]. Cost-benefit analyses are widely used in other fields to inform policy decision making (e.g. health, safety, and transport), however conservation planning tools have been slow to integrate this into their framework [121–123]. Spatial cost-benefit analysis could prove invaluable for informing conservation planning, even when relevant data may be limited [122]. There is increased awareness of the economic value of ecosystem services (including biodiversity) and quantifying these values can help decision makers best allocate scarce resources to various policy objectives.

5.2. Thinking beyond habitat provision

Provision of habitat is a necessary but not sufficient condition for threatened species population increases in productive landscapes. Threatened species may be completely absent from the landscape so that translocations will be required for them to occupy habitat made available through restoration or preserved through protection. Further management actions may be required, such as exclusion of domestic livestock, control of invasive predators, herbivores, and weeds (e.g. 124, 125). Consequently, ecological restoration activities required to improve ecosystem services such as water quality or carbon storage may often be insufficient to enhance biodiversity (Figure 3;60).

Obviously, any attempt to enhance threatened species requires an understanding of the primary factors limiting threatened species populations in productive landscapes. We also need to know whether or not threatened species populations are likely to increase in response to management interventions before they are applied on large scales. Obtaining this information for all groups of the indigenous biota is challenging. It may be impracticable to document responses of all high priority species to the pressures imposed by productive landscapes. Similarly, it may not possible to document the response of all high priority species to management interventions aimed at mitigating pressures. Indeed, many studies focus on demonstrating the effect of pressures and management interventions on community-level changes in species composition, without considering implications for high-priority species. This is understandable since rare species are often poorly captured by objective sampling designs. A shift towards studies focussed on capturing variation in high-priority species might help improve our understanding of how pressures and management interventions harm and benefit national conservation goals. However, it might be more efficient to find a way of using existing studies on changes in species composition to predict responses of high-priority species to pressures and management interventions. Functional traits provide such a means of

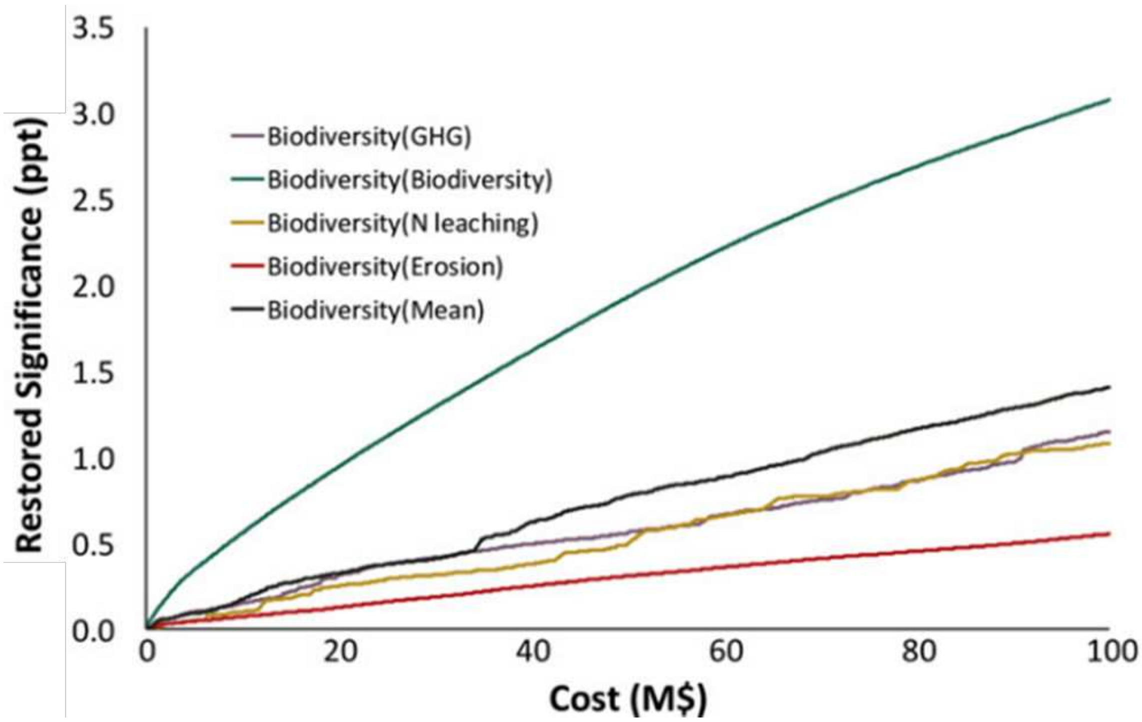

Figure 3. Example of the cumulative averted impacts against restoration effort (millions of New Zealand dollars biodiversity (i.e. restored significance) gain, the cost–benefit optimisations. The variable being optimised is given in brackets. For example, Erosion (Biodiversity) indicates the cumulative erosion reduction achieved when allocating restoration effort to maximise biodiversity gains. Mean indicates that the mean cost–benefit ratio across all variables was optimised. For each optimisation, land was selected in steps of NZ$100, 000 in order of decreasing benefit:cost ratio. In each panel, the vertical distances between the uppermost curve and the other curves indicate the size of trade-offs [60].

generalising results obtained from detailed studies on small subsets of the total indigenous biota to predict responses for threatened species.

Advanced statistical methods using traits to predict species distributions along environmental and disturbance gradients [126–128] could also be used to develop trait-based responses to pressures and management interventions. There is a large literature demonstrating relationships between functional traits to pressures such as anthropogenic disturbance or herbivory from domestic livestock (e.g. 129–133). These relationships could be used to predict threatened species vulnerabilities to such pressures. A smaller number of studies have demonstrated the influence of traits on species responses to the removal or reduction of pressures, such as exclusion of invasive herbivores [59], and post-agricultural forest spread in heavily fragmented landscapes [134]. Thus there appears to be potential for traits to fill in the knowledge gaps that might hinder integrated conservation planning from being applied to productive landscapes, especially in terms of responses to pressure and management. However, we are unaware of any studies testing the ability of trait-based response models to predict accurately responses on independent data or for a different set of species. There is a real need to test whether trait relationships demonstrated in a small-scale study on a subset of species can really be used to extrapolate outcomes for a larger set of species.

5.3. Land-use planning when data are scarce

In most practical land-use and biodiversity planning situations, the information available falls short of that required for informed prioritisation of resource allocation or conservation actions. Many organisations therefore invest resources into gathering and developing data instead of other environmental management activities. There is a growing literature on the cost-effectiveness and optimality of data gathering for guiding land-use planning [103, 131, 135, 136]. A key message from this work is that diminishing returns are inherent in data gathering for conservation planning; at some point, it becomes more effective for conservation organisations to stop data gathering and instead implement protection, albeit with imperfect information.

When data or data-gathering resources are scarce, surrogates are often used. There has been considerable assessment of, and debate on, the effectiveness of surrogates for the distribution of species and taxa [137–141]. Most land-use planning approaches use surrogates for mapping pressures on biodiversity and vulnerability [77], yet relatively little attention has been paid to their relative effectiveness. Surrogates for vulnerability in planning land protection included tenure and land use, environmental or spatial variables correlated with past conversion, threatened species distributions, and maps compiled from expert judgement [77]. Many assumptions are inherent in the application of these surrogates. For example, use of land tenure as a surrogate assumes vulnerability can be estimated from associated permitted land uses; surrogates based on past conversion and threatened species patterns assume that future distributions and impacts of threatening processes are indicated by those in the past [78].

Recently, attention in land-use planning research has shifted from techniques to static reserve blueprints, such as those produced by optimisation, to solving the challenge of conservation planning in the context of dynamic threats [142, 143]. Practitioners preferred not to use static optimised maps, and required tools that helped them make quick decisions for estimating marginal benefits [104]. In fact simple decision rules may have greater practical utility than detailed optimised plans when degradation rates and uncertainty are high, and implementation is carried out over a number of years [144]. These approaches acknowledge that threats are dynamic in most conservation planning situations, that prioritisation that ignores dynamism can be ineffective, and that the need for dynamic updating of conservation priorities is based on updated vulnerability data. However, a major implication of dynamic prioritisation is that solutions may be more demanding of data, and more complex to produce, than those that assume stasis [143].

This situation brings trade-offs between data gathering and conservation effectiveness into stark relief. Clearly, as with data on biodiversity distribution, diminishing returns are inherent in the gathering and validation of accurate data on expanding threats. Land-use planning tools based on less accurate data and simpler solutions may be more effective for conservation, once the cost and flexibility limitations of incorporating more accurate data are accounted for. For example, comprehensive reserve network design may be counterproductive in situations where site availability is uncertain, reserve acquisition is protracted, and rates of biodiversity loss are high [44]. In these situations, simple decision rules, such as protecting the available site with the highest irreplaceability or with the highest species richness, may be more effective for protecting biodiversity.

6. Conclusion

Most of the world's biological diversity currently exists outside protected areas and this is likely to remain true for the foreseeable future. Maintaining the ecological integrity of this matrix is essential to support biological diversity, maintain the embedded protected areas and support changing land use needs. However, achieving both conservation and resource extraction across the landscape will require careful consideration of the different trade-offs between biodiversity protection, ecosystem function, and socio-economic well-being. Future research needs to bring a quantitative approach to the general question: How best to allocate land use (or prioritise decisions) to ensure optimal delivery of both ecosystem services and biodiversity protection – more specifically, which land-use strategy delivers the greatest return on investment.

Biologists and economists have recognized that they need to work more closely to develop a framework to estimate the marginal value of biological diversity. Many countries that appear to have annual increases in gross domestic products may have stagnant or even declining economies when it comes to the costs of development on the loss of species and the depletion of ecosystems. Ecosystems deliver multiple services and many involve trade-offs. The value of biodiversity change to society depends on the net marginal effect of the change on all ecosystem services. Future work needs to quantify the marginal benefits of biodiversity (in terms of services gained) relative to marginal costs (in terms of lost).

Even with the growing literature on ecosystem services, there are still many challenges to integrate ecosystem services into decision making. Although a basic framework showing how to integrate ecosystem services may exist, the science needed to inform the link that connects decisions and ecosystems still remains unclear [3, 145]. The challenge is how to make the ecosystem services framework operational across a range of landscapes, changing conditions and management settings.

The increasing use of conservation planning is accompanied by an increasing number of software tools [146], reflecting diverse ideas about conservation objectives. As conservation planning continues to incorporate more complex and often competing objectives into the prioritisation framework, models should continue to be flexible and adaptable to these changes. The capability of most current conservation tools is limited by using a target-based approach to modelling conservation priorities. Recent research illustrates that a priority ranking rather than "satisfying targets with minimum costs" is more applicable to multi-objective conservation prioritisation, such as ecosystem services [114]. A more comprehensive treatment of the types of social and values of ecosystem services that might affect the implementation of conservation plans is a key area for improvement.

Ecosystem services prioritisation share some of the same elements as any conservation prioritisation problem – assessing the capacity for an ecosystem to meet demand, identifying ecosystem features that could supply services, determining threats to service provision, creating the potential for the future supply of services and the costs of these actions. The conceptual framework for integrating each of these components into spatial prioritization for ecosystem services is illustrated in Figure 4. A fuller characterisation of the biophysical and social context for ecosystem services should improve future prioritisation and the identification of locations where ecosystem-service management is especially important or cost effective.

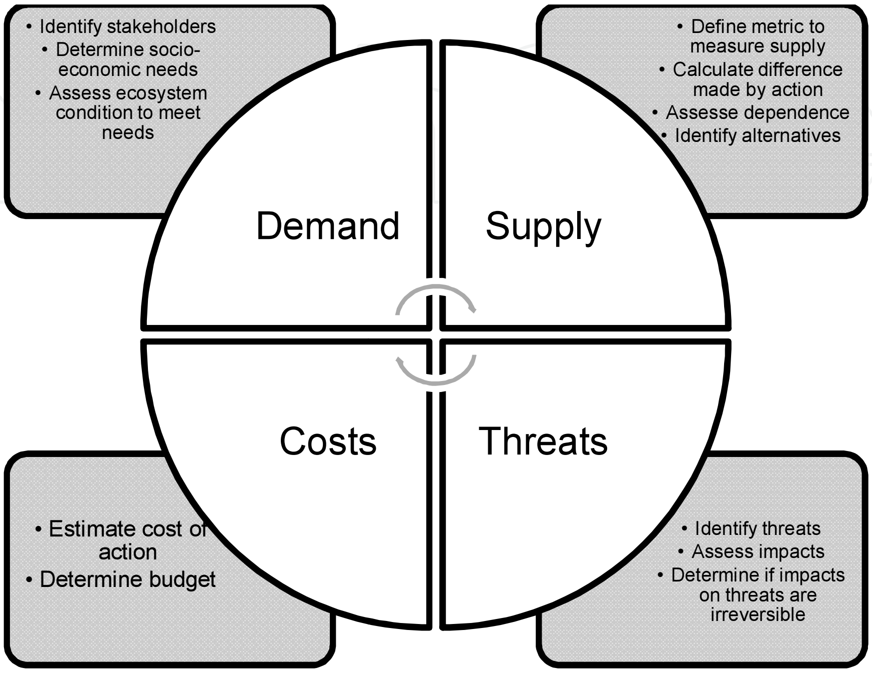

Figure 4. Key aspects to the prioritisation process considered for ecosystem services [147].

The extent to which ecosystem services can be integrated into conservation planning remains largely unknown. Although ecosystem services are increasingly acknowledged in conservation planning, the formal integration of ecosystem services into conservation assessments remains low [14, 21]. There is little research into developing methods to include ecosystem services in conservation assessments or into the extent of concordance between prioritising biodiversity features and the spatial features required for ecosystem services. As a result there is an urgent need to develop an appropriate conceptual framework, an operational model [149], and software tools for planning for ecosystem services. The first step, however, is to map the extent of ecosystem services and identify synergies and trade-offs in conserving areas of high biodiversity value and areas important for service delivery [28, 147, 148].The second step is to use this information to build a dynamic land-use model that can be integrated into a unified framework. This model should build on existing models but should also integrate habitat restoration and consider both biodiversity protection and ecosystem services. From this, further research is required to contrast the performance of habitat protection, restoration or resource management. This could be applied to divergent examples – one where the metric of performance is the persistence of threatened species, and the other where the metric of performance is an ecosystem service. Aside from finding optimal solutions to the problem the method should include rules of thumb for the general conditions and timing at which natural resource managers should shift their emphasis from management to protection or restoration.

Ideally, six key features should be incorporated into conservation planning tools in order to plan effectively for ecosystems services [110, 147, 148]. First, a new tool should incorporate different features within a single network. Second, it should include the option that targets may not be met with available resources or within the planned region. Third, an ideal tool would account, spatially and temporally, for potential impacts of management and threats on species and services. Fourth, the tool would also account for the likeliness of conflicting management practices. Fifth, a tool must consider the flow from ecosystems to user. Last, a tool should allow for "flexibility between the ends of benefit maximisation and suitability-maximizing target achievement, which will each be appropriate for individual ecosystem services in different circumstances".

In reality, coming up with a model that balances all needs is complicated by uncertain data, conflicting needs at different spatial resolutions, and the need to consider costs. There are currently two approaches to ecosystem service assessments. The first uses a broad-scale assessment of multiple services to extrapolate a few estimates of values, based on habitat types, at large spatial scales (including global assessments) [1, 150–152]. In contrast, in the second approach ecosystems services are assessed at the production level of a single service within a small area. Although this approach is more reliable than the first, it lacks both scope (number of services) and scale (geographic and temporal) to be relevant for most policy decisions. What is needed is an approach that is ecologically relevant and appropriate to the scale of land-management decisions.

Some studies have found a decline in the congruence between species richness of different higher taxa at higher spatial resolution [153, 154]. Such findings lead to the question – what is the 'ideal' spatial resolution to outweigh the benefits and costs of loss of biodiversity? It is obvious that management agreements should reflect the spatial scale of the biological processes that are important, but the challenge remains how best to fit this into the various legislative processes. Conservation planning for biodiversity has traditionally tended to adopt a vertical integration of governance to national scales [155, 156]. The emphasis on vertical integration stems from the nature of the spatial turnover in biodiversity, which is not necessarily the case in all ecosystem services (e.g. carbon storage). For ecosystems services, different relationships exist at different scales [19]. Though in general, investing in conservation that increases the value of ecosystems services is also beneficial for biodiversity [15, 64], policy should be underpinned by science that highlights the many roles biodiversity has in ecological processes at different scales

Currently, too little is known about the ecological interactions (including role of biodiversity) in major productive landscapes and about the economic value of the ecosystem services on which they rely or which they potentially provide. To address this lack of knowledge there is a need to adopt an ecological approach to current management. Crop and livestock production systems must be managed as ecosystems, with management decisions fully aware of environmental costs and benefits. Actively managing productive systems for both biodiversity protection and productivity could result in the delivery of multiple services. Many biodiversity management actions can result in multiple benefits. For example, maintaining invertebrate diversity in soils promotes fertility, plant water use efficiency, and increased carbon storage [157].Creative science should provide multiple options and a sound basis for decision.

Research must provide sufficient ecological understanding of productive systems and identify the value of important ecosystems services. Agriculture science, for example, must move beyond understanding ecological constraints to productivity and must focus on identifying the biodiversity and associated ecological processes that underpin the delivery of ecosystem services essential for maintaining a highly productive system [17, 18].

Ideally, conservation efforts in productive landscapes should contribute to national and regional conservation objectives. This means that the aim should be to go beyond enhancing "native dominance" in productive landscapes, which might simply reflect increasing the "bulk supply" of indigenous species and ecosystem types that are already common. Rather, the aim should be to increase the populations and distribution of rare and threatened species (i.e. increase "species occupancy") and the distribution of threatened ecosystems (i.e. increase "environmental representation"). The first step in achieving this is to determine which threatened species and ecosystems occur, or could potentially occur, in productive (unprotected) landscapes.

Ultimately an effective conservation plan is one that translates science into action, and it is increasingly recognised that systematic conservation planning must be complemented by an implementation strategy [149], or at least consider implementation issues in its design. Complex optimisation models may not only have greater information needs, but could also be more difficult to communicate to practitioners, and to embed and implement within the operational framework of an organisation. While there is a need for science to solve the problems of dynamic planning, there is also a pressing need for policy and practice to catch up with science [139, 148, 157]. Practitioners prefer not to use static optimised maps [104], and therefore require tools that help them make quick decisions for estimating marginal benefits. Simple decision rules may have greater practical utility than detailed optimised plans, particularly when land-use change is rapid, uncertainty is high, and implementation is carried out over a number of years. Fitting the solution to the practical situation is a challenge that still requires greater attention.

Acknowledgements

This work was supported by the National Land Resource Centre and Landcare Research. We particularly thank Chris Muckersie for useful comments on an earlier draft of the chapter and Anne Austin for editing.

Author details

Emily S. Weeks[1], Norm Mason[2], Anne-Gaelle E. Ausseil[2] and Alexander Herzig[2]

1 National Land Resource Centre, New Zealand

2 Landcare Research, New Zealand

References

[1] Millennium Ecosystem Assessment. *Millenium ecosystem assessment synthesis report*. Washington D.C., USA: Island Press. 2005.

[2] Hoekstra JM, Boucher TM, Ricketts TH, Roberts C. Confronting a biome crisis: global disparities of habitat loss and protection. Ecol Lett. 2005;8(1):23–9.

[3] Daily GC, Polasky S, Goldstein J, Kareiva PM, Mooney HA, Pejchar L, et al. Ecosystem services in decision making: time to deliver. Front Ecol Environ. 2009;7(1):21–8.

[4] Barnosky AD, Matzke N, Tomiya S, Wogan GO, Swartz B, Quental TB, et al. Has the Earth/'s sixth mass extinction already arrived? Nature. 2011;471(7336):51–7.

[5] Stork NE, Coddington JA, Colwell RK, Chazdon RL, Dick CW, Peres CA, et al. Vulnerability and Resilience of Tropical Forest Species to Land-Use Change. Conserv Biol. 2009;23(6):1438–47.

[6] Hooper DU, Adair EC, Cardinale BJ, Byrnes JE, Hungate BA, Matulich KL, et al. A global synthesis reveals biodiversity loss as a major driver of ecosystem change. Nature. 2012;486(7401):105–8.

[7] Haines-Young R. Land use and biodiversity relationships. Land Use Policy. 2009;26:S178–S186.

[8] Cardinale BJ, Duffy JE, Gonzalez A, Hooper DU, Perrings C, Venail P, et al. Biodiversity loss and its impact on humanity. Nature. 2012;486(7401):59–67.

[9] Rockström J, Steffen W, Noone K, Persson \AAsa, Chapin FS, Lambin EF, et al. A safe operating space for humanity. Nature. 2009;461(7263):472–5.

[10] Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, et al. Global consequences of land use. science. 2005;309(5734):570–4.

[11] Reidsma P, Tekelenburg T, Van den Berg M, Alkemade R. Impacts of land-use change on biodiversity: An assessment of agricultural biodiversity in the European Union. Agric Ecosyst Environ. 2006;114(1):86–102.

[12] Sala OE, Chapin FS, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, et al. Global biodiversity scenarios for the year 2100. science. 2000;287(5459):1770–4.

[13] Imhoff ML, Bounoua L, Ricketts T, Loucks C, Harriss R, Lawrence WT. Global patterns in human consumption of net primary production. Nature. 2004;429(6994):870–3.

[14] Egoh B, Rouget M, Reyers B, Knight AT, Cowling RM, van Jaarsveld AS, et al. Integrating ecosystem services into conservation assessments: a review. Ecol Econ. 2007;63(4):714–21.

[15] Maestre FT, Quero JL, Gotelli NJ, Escudero A, Ochoa V, Delgado-Baquerizo M, et al. Plant species richness and ecosystem multifunctionality in global drylands. Science. 2012;335(6065):214–8.

[16] Dutton A, Edwards-Jones G, Macdonald DW. Estimating the Value of Non-Use Benefits from Small Changes in the Provision of Ecosystem Services. Conserv Biol. 2010;24(6):1479–87.

[17] Tilman D, Polasky S, Lehman C. Diversity, productivity and temporal stability in the economies of humans and nature. J Environ Econ Manag. 2005;49(3):405–26.

[18] Tilman D, Balzer C, Hill J, Befort BL. Global food demand and the sustainable intensification of agriculture. Proc Natl Acad Sci. 2011;108(50):20260–4.

[19] Schwartz MW, Brigham CA, Hoeksema JD, Lyons KG, Mills MH, Van Mantgem PJ. Linking biodiversity to ecosystem function: implications for conservation ecology. Oecologia. 2000;122(3):297–305.

[20] Chapin FS, Sala OE, Burke IC, Grime JP, Hooper DU, Lauenroth WK, et al. Ecosystem consequences of changing biodiversity. Bioscience. 1998;48(1):45–52.

[21] Egoh B, Reyers B, Rouget M, Bode M, Richardson DM. Spatial congruence between biodiversity and ecosystem services in South Africa. Biol Conserv. 2009;142(3):553–62.

[22] Fisher B, Turner RK, Morling P. Defining and classifying ecosystem services for decision making. Ecol Econ. 2009;68(3):643–53.

[23] Tallis H, Kareiva P, Marvier M, Chang A. An ecosystem services framework to support both practical conservation and economic development. Proc Natl Acad Sci. 2008;105(28):9457–64.

[24] Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron Dr, et al. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ. 2009;7(1):4–11.

[25] Chown SL, van Rensburg BJ, Gaston KJ, Rodrigues AS, van Jaarsveld AS. Energy, species richness, and human population size: conservation implications at a national scale. Ecol Appl. 2003;13(5):1233–41.

[26] Litt AR, Gordon DR. Predictors of species richness in northwest Florida longleaf pine sandhills. Conserv Biol. 2003;17(6):1660–71.

[27] Van Rensburg BJ, Chown SL, Gaston KJ. Species richness, environmental correlates, and spatial scale: a test using South African birds. Am Nat. 2002;159(5):566–77.

[28] Naidoo R, Balmford A, Costanza R, Fisher B, Green RE, Lehner B, et al. Global mapping of ecosystem services and conservation priorities. Proc Natl Acad Sci. 2008;105(28):9495–500.

[29] Kremen C, Williams NM, Bugg RL, Fay JP, Thorp RW. The area requirements of an ecosystem service: crop pollination by native bee communities in California. Ecol Lett. 2004;7(11):1109–19.

[30] Anderson BJ, Armsworth PR, Eigenbrod F, Thomas CD, Gillings S, Heinemeyer A, et al. Spatial covariance between biodiversity and other ecosystem service priorities. J Appl Ecol. 2009;46(4):888–96.

[31] Broussard W, Turner RE. A century of changing land-use and water-quality relationships in the continental US. Front Ecol Environ. 2009;7(6):302–7.

[32] Tscharntke T, Klein AM, Kruess A, Steffan-Dewenter I, Thies C. Landscape perspectives on agricultural intensification and biodiversity–ecosystem service management. Ecol Lett. 2005;8(8):857–74.

[33] Turner IM, T Corlett R. The conservation value of small, isolated fragments of lowland tropical rain forest. Trends Ecol Evol. 1996;11(8):330–3.

[34] Chiarello AG. Density and population size of mammals in remnants of Brazilian Atlantic forest. Conserv Biol. 2000;14(6):1649–57.

[35] Jobin B, Bélanger L, Boutin C, Maisonneuve C. Conservation value of agricultural riparian strips in the Boyer River watershed, Quebec (Canada). Agric Ecosyst Environ. 2004;103(3):413–23.

[36] Hector A, Bagchi R. Biodiversity and ecosystem multifunctionality. Nature. 2007;448(7150):188–90.

[37] Bengtsson J, Angelstam P, Elmqvist T, Emanuelsson U, Folke C, Ihse M, et al. Reserves, resilience and dynamic landscapes. AMBIO J Hum Environ. 2003;32(6):389–96.

[38] Elmqvist T, Folke C, Nyström M, Peterson G, Bengtsson J, Walker B, et al. Response diversity, ecosystem change, and resilience. Front Ecol Environ. 2003;1(9):488–94.

[39] Perfecto I, Vandermeer J. Biodiversity conservation in tropical agroecosystems. Ann N Y Acad Sci. 2008;1134(1):173–200.

[40] Schroth G, Harvey CA. Biodiversity conservation in cocoa production landscapes: an overview. Biodivers Conserv. 2007;16(8):2237–44.

[41] Schroth G. Agroforestry and biodiversity conservation in tropical landscapes [Internet]. Island Press; 2004 [cited 2013 Oct 23]. Available from: http://books.google.com/books?hl=en&lr=&id=etuh8kXYMDQC&oi=fnd&pg=PR11&dq=schroth+et+al+2004+biodiversity&ots=9-5MWJFL4G&sig=8bf5xHA-2K8luhM80bsT5Uwpe58

[42] Hoctor TS, Allen WL, Carr MH. Land corridors in the southeast: connectivity to protect biodiversity and ecosystem services. J Conserv Plan. 2007;4:90–122.

[43] Dale VH, Polasky S. Measures of the effects of agricultural practices on ecosystem services. Ecol Econ. 2007;64(2):286–96.

[44] Cardinale BJ, Harvey CT, Gross K, Ives AR. Biodiversity and biocontrol: emergent impacts of a multi-enemy assemblage on pest suppression and crop yield in an agroecosystem. Ecol Lett. 2003;6(9):857–65.

[45] Wilby A, Thomas MB. Natural enemy diversity and pest control: patterns of pest emergence with agricultural intensification. Ecol Lett. 2002;5(3):353–60.

[46] Finke DL, Denno RF. Predator diversity and the functioning of ecosystems: the role of intraguild predation in dampening trophic cascades. Ecol Lett. 2005;8(12):1299–306.

[47] Schmidt MH, Lauer A, Purtauf T, Thies C, Schaefer M, Tscharntke T. Relative importance of predators and parasitoids for cereal aphid control. Proc R Soc Lond B Biol Sci. 2003;270(1527):1905–9.

[48] Snyder WE, Ives AR. Interactions between specialist and generalist natural enemies: parasitoids, predators, and pea aphid biocontrol. Ecology. 2003;84(1):91–107.

[49] Bianchi F, Booij CJH, Tscharntke T. Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. Proc R Soc B Biol Sci. 2006;273(1595):1715–27.

[50] Kleijn D, Rundlöf M, Scheper J, Smith HG, Tscharntke T. Does conservation on farmland contribute to halting the biodiversity decline? Trends Ecol Evol. 2011;26(9):474–81.

[51] Pressey RL, Humphries CJ, Margules CR, Vane-Wright RI, Williams PH. Beyond opportunism: key principles for systematic reserve selection. Trends Ecol Evol. 1993;8(4):124–8.

[52] Overton JM, Stephens RT, Cook S, Wright E. Unpublished Landcare Research Contract Research Report. Wellington, New Zealand: Landcare Research; 2010. Report No.: LC0910/064.

[53] Awimbo JA, Norton DA, Overmars FB. An evaluation of representativeness for nature conservation, Hokitika Ecological District, New Zealand. Biol Conserv. 1996;75(2):177–86.

[54] Walker S, Price R, Rutledge D, Stephens RT, Lee WG. Recent loss of indigenous cover in New Zealand. New Zealand J Ecol. 2006;30(2):169–77.

[55] Crossman ND, Bryan BA. Systematic landscape restoration using integer programming. Biol Conserv. 2006;128(3):369–83.

[56] Wilson KA, Lulow M, Burger J, Fang Y-C, Andersen C, Olson D, et al. Optimal restoration: accounting for space, time and uncertainty. J Appl Ecol. 2011;48(3):715–25.

[57] Duarte CM, Dennison WC, Orth RJ, Carruthers TJ. The charisma of coastal ecosystems: addressing the imbalance. Estuaries Coasts. 2008;31(2):233–8.

[58] Niemelä J, Baur B. Threatened species in a vanishing habitat: plants and invertebrates in calcareous grasslands in the Swiss Jura mountains. Biodivers Conserv. 1998;7(11):1407–16.

[59] Ausseil A-GE, LIindsay Chadderton W, Gerbeaux P, Theo Stephens RT, Leathwick JR. Applying systematic conservation planning principles to palustrine and inland saline wetlands of New Zealand. Freshw Biol. 2011;56(1):142–61.

[60] Mason NW, Ausseil A-GE, Dymond JR, Overton JM, Price R, Carswell FE. Will use of non-biodiversity objectives to select areas for ecological restoration always compromise biodiversity gains? Biol Conserv. 2012;155:157–68.

[61] Seppelt R, Voinov A. Optimization methodology for land use patterns using spatially explicit landscape models. Ecol Model. 2002;151(2):125–42.

[62] Groot JC, Rossing WA, Jellema A, Stobbelaar DJ, Renting H, Van Ittersum MK. Exploring multi-scale trade-offs between nature conservation, agricultural profits and landscape quality—a methodology to support discussions on land-use perspectives. Agric Ecosyst Environ. 2007;120(1):58–69.

[63] Meyer BC, Lescot J-M, Laplana R. Comparison of two spatial optimization techniques: a framework to solve multiobjective land use distribution problems. Environ Manage. 2009;43(2):264–81.

[64] Polasky S, Johnson K, Keeler B, Kovacs K, Nelson E, Pennington D, et al. Are investments to promote biodiversity conservation and ecosystem services aligned? Oxf Rev Econ Policy. 2012;28(1):139–63.

[65] Ausseil A-GE, Herzig A, Dymond J.R. Optimising land use for multiple ecosystem services objectives: A case study in the Waitaki catchment, New Zealand. In Seppelt R, Voinov AA, Lange S, Bankamp D eds, International Congress on Environmental Modelling and Software, Leipzig, Germany, July 2012. Available from: http://www.iemss.org/sites/iemss2012//proceedings/F5_0473_Ausseil_et_al.pdf

[66] Loehle C. Optimal control of spatially distributed process models. Ecol Model. 2000;131(2):79–95.

[67] Wierzbicki A, Makowski M, Wessels J. Model-based decision support methodology with environmental applications [Internet]. Kluwer Academic Publishers Dordrecht; 2000 [cited 2013 Oct 23]. Available from: http://user.iiasa.ac.at/~marek/pubs/book_s.pdf

[68] Liu X, Li X, Shi X, Huang K, Liu Y. A multi-type ant colony optimization (MACO) method for optimal land use allocation in large areas. Int J Geogr Inf Sci. 2012;26(7):1325–43.

[69] Bevers M, Hof J, Uresk DW, Schenbeck GL. Spatial optimization of prairie dog colonies for black-footed ferret recovery. Oper Res. 1997;45(4):495–507.

[70] Randhir TO, Lee JG, Engel B. Multiple criteria dynamic spatial optimization to manage water quality on a watershed scale. Trans ASAE. 2000;43(2):291–300.

[71] Herzig A. A GIS-based Module for the Multi-Objective Optimization of Areal Resource Allocation. 11th AGILE Int Conf Geogr Inf Sci [Internet]. 2008 [cited 2013 Oct 24]. Available from: http://agile.gis.geo.tu-dresden.de/web/Conference_Paper/CDs/AGILE%202008/PDF/105_DOC.pdf

[72] Herzig, A., Ausseil, A.G., & Dymond, J.R. (2014). Spatial optimisation of ecosystem services. In J.R. Dymond (Ed.), Ecosystem services in New Zealand - conditions and trends. Lincoln, New Zealand: Manaaki Whenua Press

[73] Herzig A, Ausseil A-GE, Dymond JR 2013. Sensitivity of land-use pattern optimisation to variation in input data and constraints. In: In Piantadosi, J., Anderssen, R.S. and Boland J. (eds) MODSIM2013, 20th International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2013, pp. 1840–1846.

[74] Weeks ES, Walker SF, Dymond JR, Shepherd JD, Clarkson BD. Patterns of past and recent conversion of indigenous grasslands in the South Island, New Zealand. New Zeal J Ecol. 2013;37(1): 127–138

[75] Weeks ES, Overton JM, Walker S. Estimating patterns of vulnerability in a changing landscape: a case study of New Zealand's indigenous grasslands. Environ Conserv. 2013;1(1):1–12.

[76] Margules CR, Pressey RL. Systematic conservation planning. Nature. 2000;405(6783): 243–53.

[77] Wilson K, Pressey RL, Newton A, Burgman M, Possingham H, Weston C. Measuring and incorporating vulnerability into conservation planning. Environ Manage. 2005;35(5):527–43.

[78] Wilson KA, Westphal MI, Possingham HP, Elith J. Sensitivity of conservation planning to different approaches to using predicted species distribution data. Biol Conserv. 2005;122(1):99–112.

[79] Pontius RG, Cornell JD, Hall CA. Modeling the spatial pattern of land-use change with GEOMOD2: application and validation for Costa Rica. Agric Ecosyst Environ. 2001;85(1):191–203.

[80] Weeks ES, Walker S, Overton JM, Clarkson B. The Value of Validated Vulnerability Data for Conservation Planning in Rapidly Changing Landscapes. Environ Manage. 2013;1–12.

[81] Veldkamp A, Lambin EF. Predicting land-use change. Agric Ecosyst Environ. 2001;85(1):1–6.

[82] Theobald DM, Hobbs NT, Bearly T, Zack JA, Shenk T, Riebsame WE. Incorporating biological information in local land-use decision making: designing a system for conservation planning. Landsc Ecol. 2000;15(1):35–45.

[83] Muller MR, Middleton J. A Markov model of land-use change dynamics in the Niagara Region, Ontario, Canada. Landsc Ecol. 1994;9(2):151–7.

[84] Brown S, Hall M, Andrasko K, Ruiz F, Marzoli W, Guerrero G, et al. Baselines for land-use change in the tropics: application to avoided deforestation projects. Mitig Adapt Strat Glob Change. 2007;12(6):1001–26.

[85] Irwin EG, Geoghegan J. Theory, data, methods: developing spatially explicit economic models of land use change. Agric Ecosyst Environ. 2001;85(1):7–24.

[86] Barredo JI, Kasanko M, McCormick N, Lavalle C. Modelling dynamic spatial processes: simulation of urban future scenarios through cellular automata. Landsc Urban Plan. 2003;64(3):145–60.

[87] Pontius Jr RG, Huffaker D, Denman K. Useful techniques of validation for spatially explicit land-change models. Ecol Model. 2004;179(4):445–61.

[88] Echeverria C, Coomes DA, Hall M, Newton AC. Spatially explicit models to analyze forest loss and fragmentation between 1976 and 2020 in southern Chile. Ecol Model. 2008;212(3):439–49.

[89] Liebhold AM, Elkinton JS, Zhou C, Hohn ME, Rossi RE, Boettner GH, et al. Regional correlation of gypsy moth (Lepidoptera: Lymantriidae) defoliation with counts of egg masses, pupae, and male moths. Environ Entomol. 1995;24(2):193–203.

[90] Weng Q. Land use change analysis in the Zhujiang Delta of China using satellite remote sensing, GIS and stochastic modelling. J Environ Manage. 2002;64(3):273–84.

[91] Hall CAS, Tian H, Qi Y, Pontius G, Cornell J. Modelling spatial and temporal patterns of tropical land use change. J Biogeogr. 1995;753–7.

[92] Odum HT. Systems Ecology; an introduction. 1983 [cited 2013 Oct 24]; Available from: http://www.osti.gov/energycitations/product.biblio.jsp?osti_id=5545893

[93] Houghton RA. The worldwide extent of land-use change. BioScience. 1994;44(5):305–13.

[94] Parks PJ. Explaining" irrational" land use: risk aversion and marginal agricultural land. J Environ Econ Manag. 1995;28(1):34–47.

[95] Singh RB. Environmental consequences of agricultural development: a case study from the Green Revolution state of Haryana, India. Agric Ecosyst Environ. 2000;82(1):97–103.

[96] Verburg PH, Schot PP, Dijst MJ, Veldkamp A. Land use change modelling: current practice and research priorities. GeoJournal. 2004;61(4):309–24.

[97] Lambin EF, Rounsevell MDA, Geist HJ. Are agricultural land-use models able to predict changes in land-use intensity? Agric Ecosyst Environ. 2000;82(1):321–31.

[98] Mottet A, Ladet S, Coqué N, Gibon A. Agricultural land-use change and its drivers in mountain landscapes: A case study in the Pyrenees. Agric Ecosyst Environ. 2006;114(2):296–310.

[99] Wood EC, Tappan GG, Hadj A. Understanding the drivers of agricultural land use change in south-central Senegal. J Arid Environ. 2004;59(3):565–82.

[100] Veldkamp A, Fresco LO. CLUE: a conceptual model to study the conversion of land use and its effects. Ecol Model. 1996;85(2):253–70.

[101] Rounsevell MDA, Reginster I, Araújo MB, Carter TR, Dendoncker N, Ewert F, et al. A coherent set of future land use change scenarios for Europe. Agric Ecosyst Environ. 2006;114(1):57–68.

[102] Pressey RL, Hager TC, Ryan KM, Schwarz J, Wall S, Ferrier S, et al. Using abiotic data for conservation assessments over extensive regions: quantitative methods applied across New South Wales, Australia. Biol Conserv. 2000;96(1):55–82.

[103] Grantham HS, Moilanen A, Wilson KA, Pressey RL, Rebelo TG, Possingham HP. Diminishing return on investment for biodiversity data in conservation planning. Conserv Lett. 2008;1(4):190–8.

[104] Ferrier S, Drielsma M. Synthesis of pattern and process in biodiversity conservation assessment: a flexible whole-landscape modelling framework. Divers Distrib. 2010;16(3):386–402.

[105] Wilson KA, McBride MF, Bode M, Possingham HP. Prioritizing global conservation efforts. Nature. 2006;440(7082):337–40.

[106] Underwood EC, Shaw MR, Wilson KA, Kareiva P, Klausmeyer KR, McBride MF, et al. Protecting biodiversity when money matters: maximizing return on investment. PLoS One. 2008;3(1):e1515.

[107] Watts ME, Ball IR, Stewart RS, Klein CJ, Wilson K, Steinback C, et al. Marxan with Zones: software for optimal conservation based land-and sea-use zoning. Environ Model Softw. 2009;24(12):1513–21.

[108] Pressey RL, Watts ME, Barrett TW, Ridges MJ. The C-Plan conservation planning system: origins, applications, and possible futures. 2009 [cited 2013 Oct 24]; Available from: http://eprints.jcu.edu.au/10708/

[109] Kelley C, Garson J, Aggarwal A, Sarkar S. Place prioritization for biodiversity reserve network design: a comparison of the SITES and ResNet software packages for coverage and efficiency. Divers Distrib. 2002;8(5):297–306.

[110] Chan KM, Shaw MR, Cameron DR, Underwood EC, Daily GC. Conservation planning for ecosystem services. PLoS Biol. 2006;4(11):e379.

[111] Chan K, Klinkenberg B. Ecosystem services in conservation planning: Less costly as costs and side-benefits. J Ecosyst Manag [Internet]. 2011 [cited 2013 Oct 24];12(1). Available from: http://jem.forrex.org/index.php/jem/article/viewArticle/80

[112] Moilanen A, Franco AM, Early RI, Fox R, Wintle B, Thomas CD. Prioritizing multiple-use landscapes for conservation: methods for large multi-species planning problems. Proc R Soc B Biol Sci. 2005;272(1575):1885–91.

[113] Moilanen A, Kujala H, Leathwick J. The Zonation framework and software for conservation prioritization. Spat Conserv Prioritization. 2009;196–210.

[114] Moilanen A, Anderson BJ, Eigenbrod F, Heinemeyer A, Roy DB, Gillings S, et al. Balancing alternative land uses in conservation prioritization. Ecol Appl. 2011;21(5): 1419–26.

[115] Moilanen A, Wilson KA, Possingham HP. Spatial conservation prioritization: quantitative methods and computational tools [Internet]. Oxford University Press Oxford; 2009 [cited 2013 Oct 23]. Available from: http://library.wur.nl/WebQuery/clc/1909723

[116] Brock W, Xepapadeas A. Optimal ecosystem management when species compete for limiting resources. J Environ Econ Manag. 2002;44(2):189–220.

[117] Sodhi NS, Lee TM, Sekercioglu CH, Webb EL, Prawiradilaga DM, Lohman DJ, et al. Local people value environmental services provided by forested parks. Biodivers Conserv. 2010;19(4):1175–88.

[118] Wenny DG, Devault TL, Johnson MD, Kelly D, Sekercioglu CH, Tomback DF, et al. The need to quantify ecosystem services provided by birds. The Auk. 2011;128(1):1–14.

[119] Goldman RL, Tallis H, Kareiva P, Daily GC. Field evidence that ecosystem service projects support biodiversity and diversify options. Proc Natl Acad Sci. 2008;105(27): 9445–8.

[120] Tallis H, Goldman R, Uhl M, Brosi B. Integrating conservation and development in the field: implementing ecosystem service projects. Front Ecol Environ. 2009;7(1):12–20.

[121] Polasky S, Camm JD, Garber-Yonts B. Selecting biological reserves cost-effectively: an application to terrestrial vertebrate conservation in Oregon. Land Econ. 2001;77(1):68–78.

[122] Naidoo R, Ricketts TH. Mapping the economic costs and benefits of conservation. PLoS Biol. 2006;4(11):e360.

[123] Gutman P. Putting a price tag on conservation: cost benefit analysis of Venezuela's national parks. J Lat Am Stud. 2002;34(1):43–70.

[124] Burns BR, Floyd CG, Smale MC, Arnold GC. Effects of forest fragment management on vegetation condition and maintenance of canopy composition in a New Zealand pastoral landscape. Austral Ecol. 2011;36(2):153–66.

[125] Inness J, Overton JM, Gillies C. Predation and other factors currently limiting New Zealand forest birds. New Zealand J Ecol. 2010;34(1):86–114.

[126] Shipley B, Vile D, Garnier É. From plant traits to plant communities: a statistical mechanistic approach to biodiversity. science. 2006;314(5800):812–4.

[127] Laughlin DC, Joshi C, Bodegom PM, Bastow ZA, Fulé PZ. A predictive model of community assembly that incorporates intraspecific trait variation. Ecol Lett. 2012;15(11):1291–9.

[128] Pollock LJ, Morris WK, Vesk PA. The role of functional traits in species distributions revealed through a hierarchical model. Ecography. 2012;35(8):716–25.

[129] Lavorel S, Garnier E. Predicting changes in community composition and ecosystem functioning from plant traits: revisiting the Holy Grail. Funct Ecol. 2002;16(5):545–56.

[130] Cornelissen JHC, Lavorel S, Garnier E, Diaz S, Buchmann N, Gurvich DE, et al. A handbook of protocols for standardised and easy measurement of plant functional traits worldwide. Aust J Bot. 2003;51(4):335–80.

[131] Cleary DF, Boyle TJ, Setyawati T, Anggraeni CD, Loon EEV, Menken SB. Bird species and traits associated with logged and unlogged forest in Borneo. Ecol Appl. 2007;17(4):1184–97.

[132] Diaz S, Lavorel S, McIntyre SUE, Falczuk V, Casanoves F, Milchunas DG, et al. Plant trait responses to grazing–a global synthesis. Glob Change Biol. 2007;13(2):313–41.

[133] Barbaro L, Van Halder I. Linking bird, carabid beetle and butterfly life-history traits to habitat fragmentation in mosaic landscapes. Ecography. 2009;32(2):321–33.

[134] Brunet J, De Frenne P, Holmström E, Mayr ML. Life-history traits explain rapid colonization of young post-agricultural forests by understory herbs. For Ecol Manag. 2012;278:55–62.

[135] Bottrill MC, Joseph LN, Carwardine J, Bode M, Cook C, Game ET, et al. Is conservation triage just smart decision making? Trends Ecol Evol. 2008;23(12):649–54.

[136] McDonald-Madden E, Baxter PW, Possingham HP. Making robust decisions for conservation with restricted money and knowledge. J Appl Ecol. 2008;45(6):1630–8.

[137] Rodrigues AS, Brooks TM. Shortcuts for biodiversity conservation planning: the effectiveness of surrogates. Annu Rev Ecol Evol Syst. 2007;38:713–37.

[138] Rodrigues AS, Gaston KJ. Optimisation in reserve selection procedures—why not? Biol Conserv. 2002;107(1):123–9.

[139] Brooks T, Fonseca DA, Rodrigues AS. Species, data, and conservation planning. Conserv Biol. 2004;18(6):1682–8.

[140] Lombard AT, Cowling RM, Pressey RL, Rebelo AG. Effectiveness of land classes as surrogates for species in conservation planning for the Cape Floristic Region. Biol Conserv. 2003;112(1):45–62.

[141] Pressey RL, Watts ME, Barrett TW. Is maximizing protection the same as minimizing loss? Efficiency and retention as alternative measures of the effectiveness of proposed reserves. Ecol Lett. 2004;7(11):1035–46.

[142] Higgins SI, Richardson DM, Cowling RM. Using a dynamic landscape model for planning the management of alien plant invasions. Ecol Appl. 2000;10(6):1833–48.

[143] Pressey RL, Cabeza M, Watts ME, Cowling RM, Wilson KA. Conservation planning in a changing world. Trends Ecol Evol. 2007;22(11):583–92.

[144] Meir E, Andelman S, Possingham HP. Does conservation planning matter in a dynamic and uncertain world? Ecol Lett. 2004;7(8):615–22.

[145] De Groot RS, Alkemade R, Braat L, Hein L, Willemen L. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol Complex. 2010;7(3):260–72.

[146] Sarkar S, Pressey RL, Faith DP, Margules CR, Fuller T, Stoms DM, et al. Biodiversity conservation planning tools: present status and challenges for the future. Annu Rev Env Resour. 2006;31:123–59.

[147] Luck GW, Chan KM, Klien CJ. Identifying spatial priorities for protecting ecosystem services. F1000Research [Internet]. 2012 [cited 2013 Oct 25];1. Available from: http://f1000research.com/articles/1-17

[148] Chan KM, Hoshizaki L, Klinkenberg B. Ecosystem services in conservation planning: targeted benefits vs. co-benefits or costs? PloS One. 2011;6(9):e24378.

[149] Knight AT, Driver A, Cowling RM, Maze K, Desmet PG, Lombard AT, et al. Designing systematic conservation assessments that promote effective implementation: best practice from South Africa. Conserv Biol. 2006;20(3):739–50.

[150] Costanza R, d' Arge R, De Groot R, Farber S, Grasso M, Hannon B, et al. The value of the world's ecosystem services and natural capital. nature. 1997;387(6630):253–60.

[151] Troy A, Wilson MA. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. Ecol Econ. 2006;60(2):435–49.

[152] Turner WR, Brandon K, Brooks TM, Costanza R, Da Fonseca GA, Portela R. Global conservation of biodiversity and ecosystem services. BioScience. 2007;57(10):868–73.

[153] Pearson DL, Carroll SS. Global patterns of species richness: spatial models for conservation planning using bioindicator and precipitation data. Conserv Biol. 1998;12(4): 809–21.

[154] Grenyer R, Orme CDL, Jackson SF, Thomas GH, Davies RG, Davies TJ, et al. Global distribution and conservation of rare and threatened vertebrates. Nature. 2006;444(7115):93–6.

[155] Erasmus BF, Freitag S, Gaston KJ, Erasmus BH, van Jaarsveld AS. Scale and conservation planning in the real world. Proc R Soc Lond B Biol Sci. 1999;266(1417):315–9.

[156] Strange N, Thorsen BJ, Bladt J. Optimal reserve selection in a dynamic world. Biol Conserv. 2006;131(1):33–41.

[157] Lal R. Soil carbon sequestration to mitigate climate change. Geoderma. 2004;123(1):1–22.