The Cost-Effectiveness of Biodiversity Conservation in Agricultural Landscapes

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Declaration

This thesis is my own work, except where otherwise acknowledged (see Preface and Acknowledgments).

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Preface

This thesis is comprised of several peer-reviewed published journal articles and book chapters written as stand-alone pieces of work. Its structure follows that of the Australian National University College of Medicine, Biology and Environment's Thesis by Publication guidelines. As a result, there is some inevitable overlap in the content of the chapters, and inconsistencies in style and formatting. For example, American English is used in some papers, following the requirements of the specific journals, whereas the chapters that are excerpts from my co-edited book, and the introductions to each section of the thesis, use British English.

The work presented in this thesis largely represents my own efforts, though with significant contributions from several colleagues as indicated in the authorship of each paper. This includes the major contribution of the members of my supervisory panel (Phil Gibbons, Nicki Munro and David Freudenberger) who provided advice throughout each stage of my candidature. I led the conceptualization, analysis and writing of each of the papers presented in this thesis (with specific assistance as acknowledged below), with the exception of Papers I and III which were equally co-written, Paper VIII for which I provided data and contributed to the writing, and Paper IX which was co-written. The specific contributions of co-authors on each paper are provided below, and have been agreed in writing.

> **Paper I**: Ansell, D., Gibson, F., Salt, D., 2016. Introduction: Framing the agri-environment, in: Ansell, D., Gibson, F., Salt, D. (Eds.), Learning from Agri-Environment Schemes in Australia: Investing in Biodiversity and Other Ecosystem Services on Farms. ANU Press, Canberra, Australia, pp. 1–16.

Conceptualisation and design: DA, FG, DS; Manuscript drafting: DA, FG, DS; Manuscript editing: DA, FG, DS.

> **Paper II.** Ansell, D., 2016. Defining and designing cost-effective agri-environment schemes, in: Ansell, D., Gibson, F., Salt, D. (Eds.), Learning from Agri-Environment Schemes in Australia: Investing in Biodiversity and Other Ecosystem Services on Farms. ANU Press, Canberra, Australia, pp. 193–206.

Conceptualisation and design: DA; Manuscript drafting: DA; Manuscript editing: DA.

> **Paper III.** Ansell, D., Gibson, F., Salt, D., 2016. Conclusion — Elements of good design, in: Ansell, D., Gibson, F., Salt, D. (Eds.), Learning from Agri-Environment Schemes in Australia: Investing in Biodiversity and Other Ecosystem Services on Farms. ANU Press, Canberra, Australia, pp. 293–311.

Conceptualisation and design: DA, FG, DS; Manuscript drafting: DA, FG, DS; Manuscript editing: DA, FG, DS.

Paper IV. Ansell, D.H., Fifield, G., Munro, N., Freudenberger, D. & Gibbons, P., 2016. Softening the agricultural matrix: a novel agri-environment scheme that balances habitat restoration and livestock grazing. *Restoration Ecology*, 24, 159–164.

Conceptualisation and design: DA, GF, DF, NM and PG; Data collection: DA, GF; Data analysis: DA; Manuscript drafting: DA; Manuscript editing: DA, GF, DF, NM and PG.

Paper V. Ansell, D.H., Freudenberger, D., Munro, N. & Gibbons, P., 2016. The cost-effectiveness of agri-environment schemes for biodiversity conservation: A quantitative review. *Agriculture, Ecosystems & Environment*, 225, 184-191.

Conceptualisation and design: DA, DF, NM and PG; Data collection: DA; Data analysis: DA; Manuscript drafting: DA; Manuscript editing: DA, DF, NM and PG.

Paper VI. Ansell, D.H., Munro, N., Freudenberger, D., Ikin, K., Yoon, H. & Gibbons, P., 2016. Comparing the effectiveness of alternative conservation strategies: an evaluation of woodland bird conservation actions in agricultural landscapes. Biological Conservation (In review).

Conceptualisation and design: DA, DF, NM and PG; Data collection: DA; Data analysis: DA, KI, HY; Manuscript drafting: DA; Manuscript editing: DA, KI, DF, NM and PG.

Paper VII. Ansell, D.H., Freudenberger, D., Blanchard, W., Munro, N. & Gibbons, P., 2016. Plant a tree or build a fence? Evaluating the cost-effectiveness of alternative bird conservation actions in an agricultural landscape. Journal of Environmental Management (Submitted)

Conceptualisation and design: DA, DF, NM and PG; Data collection: DA; Data analysis: DA, WB; Manuscript drafting: DA; Manuscript editing: DA, DF, PG, WB.

Paper VIII. Ikin, K., Tulloch, A., Gibbons, P., Ansell, D., Seddon, J. & Lindenmayer, D. 2016. Evaluating complementary networks of restoration plantings for landscape-scale occurrence of temporally dynamic species. Conservation Biology 30: 1027-1037.

Conceptualisation and design: KI, DL, PG, AT; Data collection: KI, DA, JS; Data analysis: KI; Manuscript drafting: KI; Manuscript editing: KI, AT, DA, PG, JS, DL.

Paper IX. Iftekhar, M. S., Polyakov, M., Ansell, D., Gibson, F. and Kay, G. M. (2016), How economics can further the success of ecological restoration. Conservation Biology. doi:10.1111/cobi.12778

Conceptualisation and design: SI, DA, MP, FG, DA, GK; Data collection: N/A; Data analysis: N/A; Manuscript drafting; SI, DA, MP, FG, DA, GK; Manuscript editing: SI, DA, MP, FG, DA, GK.

Acknowledgements

'It was the best of times, it was the worst of times'

Charles Dickens, 'A Tale of Two Cities'

There are many people to thank for what has been an incredibly rewarding and challenging few years. Firstly, I am immensely grateful to Phil Gibbons, my primary supervisor, for providing me the opportunity, the support and the encouragement to leave behind one part of my life and embark on a new, more fulfilling one. His attitude and pragmatic approach to both conservation and the arcane mysteries of academia has been the source of inspiration throughout my PhD.

I also want to thank my other supervisors, Nicki Munro and David Freudenberger, for their advice, encouragement and strong intellectual contribution to this research. I was blessed in having such a practical, supportive and responsive panel behind me. Special thanks to David Salt for prompting me to embark on several exercises which were central to the broad scope of experiences I obtained, not the least of which being the expert workshop that led to the book which forms part of this thesis. Thanks to Hwan-Jin Yoon and Wade Blanchard for their patience and advice with all matters statistical. I am indebted to several of my colleagues for everything ranging from stimulating discussion to epic whinging sessions and software trouble-shooting: David Johnson, Wendy Neilan, Fiona Gibson, Stephanie Pulsford, Geoff Kay, Karen Ikin, Shana Nerenberg and Darren Le Roux. To the fantastic folks at Greening Australia, for their advice, assistance and funding support, especially Graham Fifield, Ange Calliess, Bindi Vanzella, Nicki Taws and Jason Cummings. The constructive and insightful feedback provided by three anonymous examiners also improved this thesis.

Lastly, I cannot pretend to end this section without thanking a couple of very special people who have truly allowed me to undertake this whole endeavor. My family (both the in-laws and outlaws), whose constant enquiries as to my progress actually became a source of motivation. To Rosie, one little black dog that helped keep another at bay. To my wife Gemma, who has kept the lights on and been the smiling face at the end of each day, I cannot express enough how lucky I am to be supported by such a beautiful, smart woman.

Abstract

The worsening global biodiversity crisis combined with limited resources for conservation makes the prioritisation of cost-effective actions critical, particularly in agricultural landscapes where multiple conservation actions are available that vary widely in their effectiveness and cost. In such situations, understanding the complexities and drivers of the cost-effectiveness can improve the efficiency and effectiveness of future conservation investments.

In this thesis I present the results of a multidisciplinary investigation of cost-effectiveness in the conservation of biodiversity in agricultural landscapes, focussing on two key themes. The first involved the cost-effectiveness of agri-environment schemes, where farmers receive financial incentives for biodiversity outcomes, a major policy mechanism that accounts for billions of dollars in public conservation expenditure globally. I include three chapters (Papers I-III) from a book I co-wrote and edited on the lessons learned from agri-environment schemes in Australia, in which I discuss the various environmental and economic factors that influence cost-effectiveness. I applied the principles and practices identified through this research in an evaluation of the cost-effectiveness of an Australian agri-environment scheme (Paper IV). I found that the total cost per hectare of habitat restored through the scheme was less than half that achieved using conventional designs such as windbreak plantings. Despite such clear benefits of considering cost-effectiveness, through a review of the global agri-environmental literature I show that the integration of economic factors in evaluations of biodiversity outcomes is still lacking and shows little evidence of improving; fewer than 15% of the 239 studies reviewed include any measure of cost-effectiveness (Paper V).

The second key theme emphasized the equal importance of combining appropriate measures of effectiveness with detailed financial costs, and focused on two specific actions commonly employed in the conservation of birds in agricultural landscapes: revegetation of cleared land, and the passive restoration of remnant vegetation. Through field evaluations of 84 habitat restoration sites in southeastern New South Wales, I found significantly higher gains in bird species richness, including woodland birds, following revegetation than those from protection of remnant vegetation (Paper VI). Despite the higher cost of revegetation, I show the superior cost-effectiveness of this approach where remnant vegetation is unlikely to cleared under the counterfactual (Paper VII) and the strong influence of site design factors such as geometry in determining cost-effectiveness. In another study (Paper VIII), I demonstrate improved cost-effectiveness of habitat restoration through the integration of economic data in a systematic conservation planning approach that accounted for the temporal dynamics of threatened birds. The third theme explored the potential environmental and

social benefits of the adoption of broader economic principles and techniques in the planning of ecological restoration (Paper IX).

Combined, this research reveals the many factors that influence the financial costs of conservation on agricultural land and the complex interactions with ecological factors that influence the overall cost-effectiveness. It also highlights the relative simplicity of the economic evaluation techniques available, and showcases the conservation benefits that can be achieved through the improved collection and integration of financial costs and biodiversity benefits in the planning of conservation expenditure.

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Context statement

Introduction

Agriculture is now the dominant global land use, covering up to 40% of Earth's surface (Ramankutty et al., 2008). With much of the world's threatened species and ecosystems found outside of the formal protected area system (Tognelli et al., 2008; Watson et al., 2011), farming landscapes have become a key focus of global conservation efforts (Secretariat of the Convention on Biological Diversity, 2010).

Biodiversity conservation can be achieved through a range of policy approaches, from regulatory and legal mechanisms, to land acquisition, volunteer-based programs and incentive-based measures (Tanentzap et al., 2015). The latter includes large scale agri-environment schemes which provide financial incentives to farmers for environmental goods and services and account for billions of dollars in global public expenditure annually (European Commission, 2015; USDA, 2016). The specific conservation actions employed under these various policy approaches range from those that focus on protection and restoration of habitats embedded in the agricultural matrix, to those that aim to reduce the impacts of agricultural practices on biodiversity such as reduced fertilizer and pesticide use and organic farming (Rey Benayas and Bullock, 2015).

The effectiveness of biodiversity conservation in agricultural landscapes is often highly variable (Barral et al., 2015) in response to complex ecological, environmental and socio-economic factors. This creates challenges for decision makers looking to design programs that maximize conservation outcomes. This challenge is exacerbated by the substantial financial costs of conservation on agricultural land, which include costs of acquiring or renting land as associated transactions costs, costs of labour and materials of on-ground works, and the opportunity costs of lost agricultural production (Naidoo et al., 2006). These costs, which can range from hundreds to tens of thousands of dollars per hectare (Hunt, 2008; Preece et al., 2013), also vary widely in response to factors such as access and technical difficulty. This variation is compounded by the opportunity costs which themselves can fluctuate in response to land productivity and market forces (Adams et al., 2010).

The variation in effectiveness and costs of conservation actions creates potential for significant inefficiencies, fueling debate over the efficiency of conservation in agricultural landscapes (Batáry et al., 2015; Kleijn et al., 2001). It is widely acknowledged that the funding available for conservation falls short of that required to address the scale of the threats to global biodiversity (Balmford et al., 2003; Garnett et al., 2003; McCarthy et al., 2012). When funding is limited, prioritizing those actions that are the most cost-effective represents the most efficient conservation investment (Wilson et al., 2007). Cost-effective actions can be considered those that provide the greatest benefit per dollar spent, or the least cost to achieve a specific outcome (McDonald et al., 2015; Nunes et al., 2015).

Understanding the cost-effectiveness of conservation actions requires knowledge of both the effectiveness of the action (i.e. the conservation benefit) and the associated financial cost. The use of economic valuation methods such as cost-effectiveness analysis (Drummond et al., 1987) can then facilitate more efficient spending by comparing the relative cost-effectiveness of alternative actions. Such evaluations are routine and have a long history in many areas of public policy and research, particularly those where there is an ethical or practical barrier to the use of traditional economic evaluation techniques such as cost-benefit analysis that require monetization of the benefit measure (e.g. health care) (Weinstein and Stason, 1977). However, there remains a widespread lack of integration of ecological and economic data, concepts and methods within the conservation sciences (Kleijn and Sutherland, 2003; TEEB, 2009; Wortley et al., 2013), despite repeated demonstrations of the conservation benefits that can be achieved (e.g. Boyd et al., 2015; Joseph et al., 2009; Stoneham et al., 2003).

Overview of aims and approach

The overall aim of this thesis was to investigate the cost-effectiveness of biodiversity conservation in agricultural landscapes. By its nature, this field of research involves both economic and ecological theories and practices and, as such, this thesis was compiled using a multi-disciplinary approach that combined field-based research, collation and analysis of economic data, quantitative literature reviews and qualitative assessment of conservation policies and programs. It ranges in scope from a global review of agri-environmental policies and specific schemes in Australia, to field-based comparison of the cost-effectiveness of individual conservation actions. Though its scope thesis is intentionally broad, I centred the research on three key themes in particular (Fig. 1).

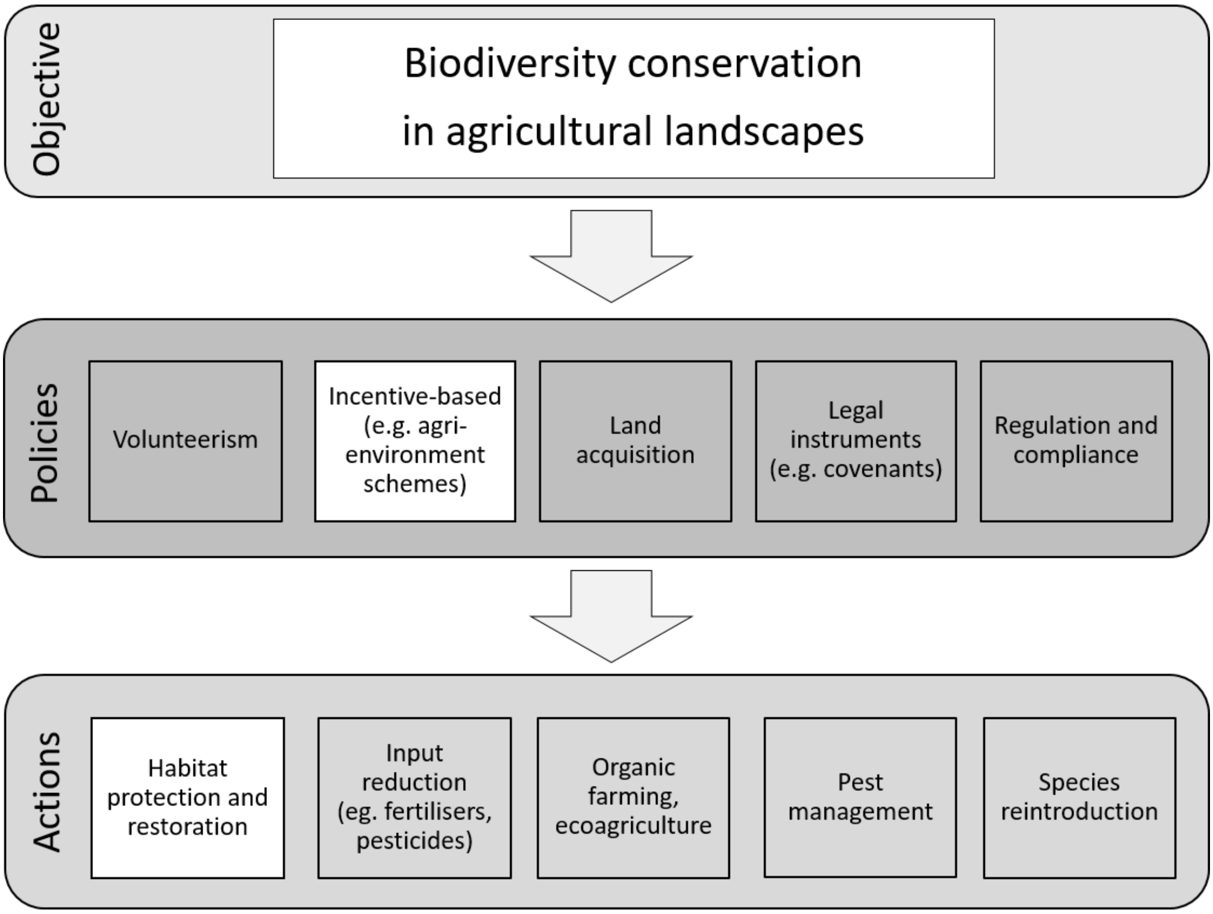

Figure 1. Overview of the key themes of this thesis (white boxes) in the broader context of biodiversity conservation in agricultural landscapes, showing the distinction between specific conservation actions and the policy options available to facilitate their delivery. Note: the policies and actions shown are not intended to be exhaustive.

Firstly, I focused on agri-environmental schemes as the policy instrument for achieving conservation outcomes in agricultural landscapes. Though there are alternative policy approaches (Fig. 1), agri-environment schemes arguably represent the largest investment in conservation on private land globally and have been suggested as the only feasible mechanisms for achieving broad-scale outcomes in such systems (Donald and Evans, 2006). The first half of this thesis (Papers I-V) is dedicated to an investigation of the cost-effectiveness of agri-environment schemes, drawing on evidence collected from multiple research techniques to explore the various factors that influence cost-effectiveness in agri-environmental policy and the trends in the economic evaluation techniques applied. It begins with three chapters (Papers I-III) from a book I co-wrote and edited with colleagues following an expert workshop on investment in the conservation of biodiversity in Australian farming landscapes. I also conducted a detailed evaluation of the cost-effectiveness of an Australian agri-environment scheme (Paper IV), and a quantitative review of the global evaluation literature (Paper IV) to determine the extent to which published evaluations of the biodiversity benefits of agri-environment schemes consider the economic costs, and integrate those costs in an assessment of cost-effectiveness.

The second theme of my thesis was the cost-effectiveness of two specific conservation actions: the active restoration of habitat through revegetation of heavily cleared sites ('restoration plantings') and the passive restoration of remnant habitat through fencing to exclude livestock ('remnant protection') which aims to promote recruitment of native vegetation. While various other actions are used to protect and restore biodiversity on farms around the world (Fig. 1), in Australian farming landscapes these two approaches represent the dominant conservation strategies. They have been the focus of major publicly funded programs such as the Environmental Stewardship Program (Burns et al., 2016), as well as environmental organisations such as Greening Australia, Landcare and state natural resource management bodies. For example, in New South Wales an average of 242,000 ha of remnant vegetation on private land was placed under some form of conservation management each year between 2005-2014, while an average of 171,300 ha was planted annually during the same period (OEH, 2016). For this part of the thesis, I conducted my own field evaluations of the cost-effectiveness of restoration plantings and remnant protection in southeastern Australia (Papers VI, VII). I also conducted economic analysis in an application of spatial optimization (using the software Marxan) in the selection of cost-effective complementary networks of actively restored (revegetated) sites that maximize the occurrence of threatened bird species (Paper VIII).

The final theme involves the benefits that can be achieved by the enhanced integration of economics in the conservation of biodiversity in agricultural landscapes. This is an issue that pervades much of this thesis, though I dedicate the final part of this thesis to a paper that explored the integration of economics and ecological restoration (Paper IX), a broad class of conservation actions applied extensively through the world's agricultural landscapes (Barral et al., 2015), and considered the broader economic principles and techniques that could be adopted by restoration scientists and practitioners to improve conservation, social and economic outcomes.

The relationship between each paper and the key themes of this thesis are shown in Figure 2, and the key findings of each are discussed in the following sections. Seven of the eight papers have been published, including the international journals *Agriculture, Ecosystems and Environment*, *Restoration Ecology* and *Conservation Biology*, as well as within the book I co-authored and edited *Learning from*

Agri-environment Schemes in Australia, published by ANU Press. At the time of submission of this thesis, a further manuscript was in review at *Biological Conservation* and the other submitted to *Journal of Environmental Management*.

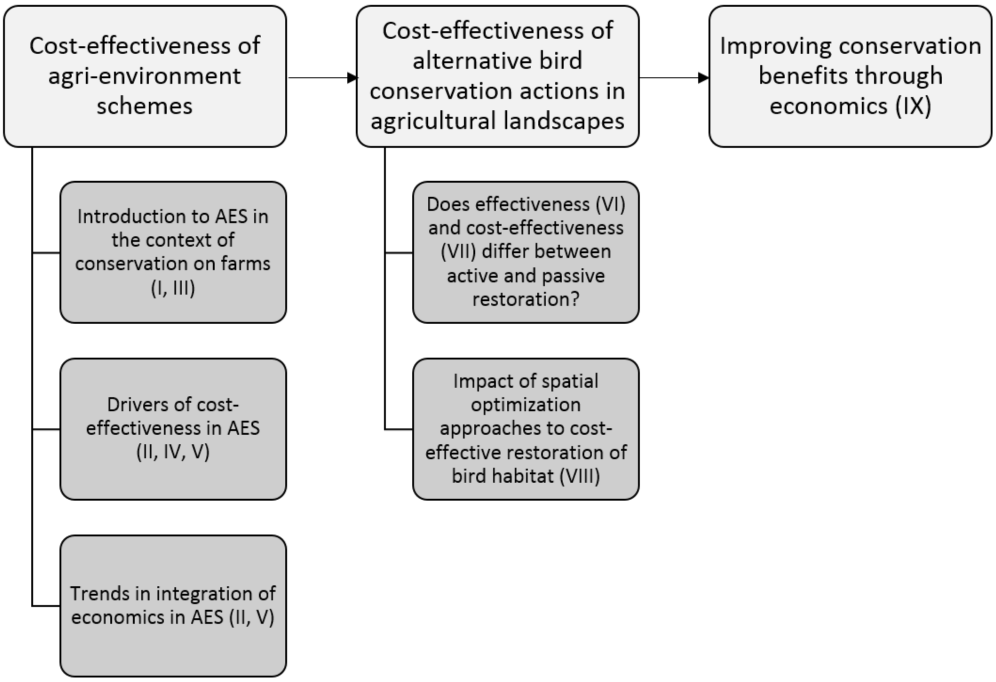

Figure 2. The relationship between the key themes of this thesis and the individual papers. Roman numerals refer to specific papers. Note 'AES' refers to agri-environment schemes.

The cost-effectiveness of agri-environment schemes

As a major policy mechanism for the delivery of conservation in agricultural landscapes, agri-environment schemes represent an important area for research into cost-effectiveness. I explored the application of these schemes in different agricultural and ecological contexts around the world and the many factors that influence their cost-effectiveness, through a combination of literature review and economic analysis combined with an expert workshop. This workshop, which I co-organised and facilitated in Canberra in September 2014, involved 23 ecologists, economists, social scientists, policy makers and conservation practitioners and led to the production of the book from which Papers I-III are drawn.

In Paper I, I discussed the key role that agricultural landscapes play in the conservation of global biodiversity, and the importance of agri-environment schemes in delivering those outcomes. I explained the economic principles underlying the concept of incentive-based measures, and introduce the history of their use in Australia. Much of the discussion in this paper is framed around a contrast between two Australian schemes (the Whole of Paddock Rehabilitation program and the Environmental Stewardship Program) that differ widely in their ecological basis, design and delivery, as well as in their costs and potential benefits. This presented a framework through which to raise issues of cost-effectiveness, public versus private benefits and additionality, and questions where public expenditure should be focused, particularly when budgets are constrained.

In Paper II, I more comprehensively explored the issue of cost-effectiveness in the planning and implementation of agri-environment schemes, particularly focusing on the diversity of economic evaluation techniques employed and the potential biodiversity benefits and efficiencies that can be achieved. I highlight variability in both the effectiveness and economic costs of biodiversity conservation actions on agricultural land, and the factors that influence that variation, drawing on examples from the global agri-environmental literature. I identify the potential for major inefficiencies in conservation expenditure through this variation, and provide an overview of the concept of cost-effectiveness in conservation and the methodologies that can be applied in its evaluation. I consider the timing of evaluations, the use of modelled versus actual benefits and costs, and recent developments in agri-environmental policy that have the potential to greatly improve cost-effectiveness. A key point made in this paper relates to the simplicity of the economic evaluation techniques available, and the diversity of agricultural land use contexts in which they can be applied.

In Paper III I distilled the reflections of the experts from various research disciplines that participated in the workshop and contributed to the resulting book, to identify the lessons learned from three decades of agri-environmental investment. I identify the key elements of importance to policy makers in the design and implementation of future agri-environment schemes, including: 1) maximizing the additionality of restoration and other conservation measures; 2) that the longevity of schemes considers ecological time lags and behavioural change; 3) that delivery mechanisms are 'fit for purpose'; 4) management of risk and uncertainty; 5) capacity-building needs and; 6) the importance of cost-effectiveness when prioritising agri-environmental expenditure.

In Paper IV I contrasted the cost-effectiveness of the Whole of Paddock Rehabilitation scheme—an agri-environment scheme focussed on habitat restoration in productive areas of the agricultural landscape and which combines agricultural production biodiversity benefits—with two alternative planting configurations: windbreak-style plantings, arguably one of the most dominant planting designs applied in agricultural landscapes, and block-shaped plantings, which represent a more idealistic configuration from an ecological perspective in terms of maximal habitat area (Fig. 3). I calculated the total costs (public and private) over ten years for a hypothetical 20ha paddock enrolled in the scheme with the two alternative planting approaches and derived a number of cost-effectiveness metrics to evaluate differences in efficiency and the major factors contributing to restoration costs. I demonstrated the superior cost-effectiveness of the Whole of Paddock Rehabilitation scheme over the traditional windbreak planting design and showed that the total cost per hectare of vegetation restored through the scheme is less than half that achieved using windbreak plantings, even when incentive payments to the farmer are considered. I identified several key features that contribute to this cost-effectiveness. This includes the use of existing farm infrastructure which removes the cost of fencing—a major cost component of restoration projects. Another feature identified is the integration of private benefits into the scheme design, which likely increases uptake by lowering the entry price for enrolling landholders. I conclude that innovative schemes such as the Whole of Paddock Rehabilitation program that incorporate opportunity costs to the farmer and seek to integrate, rather than displace, agricultural production allow access to parts of the agricultural landscape that have traditionally been 'off limits' to conservation and represent a high priority from a biodiversity perspective.

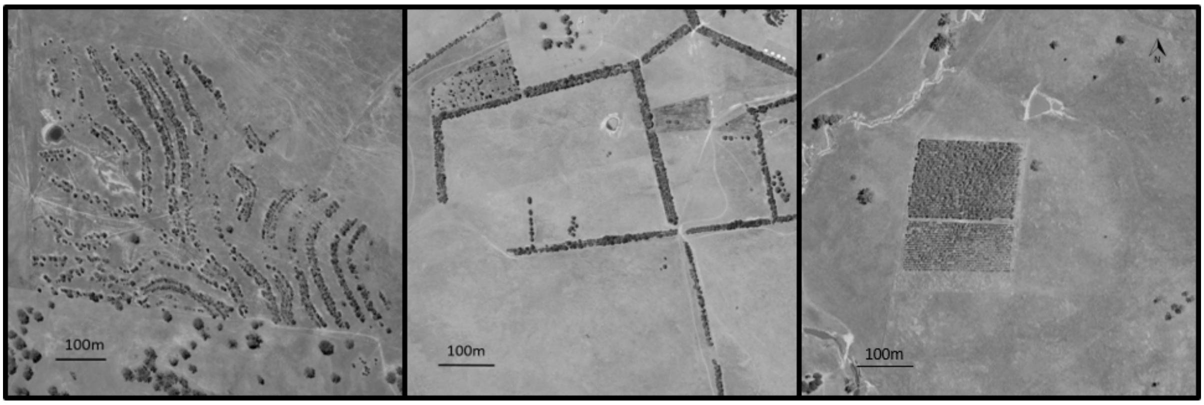

Figure 3. Examples of the three planting configurations evaluated in Paper V (left to right): a typical Whole of Paddock Rehabilitation site; a windbreak-style and; a planting block planting. All sites situated in the Boorowa study region in southeastern Australia (see Fig. 4). Image source: Google Earth V 7.1.5.1557

In Papers II-IV I highlight the potential improvements in the efficiency of public expenditure on agri-environment schemes that can be achieved through consideration of cost-effectiveness. I then sought to measure the extent to which published evaluations of the effectiveness of these schemes consider economic factors. In Paper V I present the results of a quantitative review of the global agri-environmental evaluation literature, in which I reviewed 239 articles, profiling the geographical, ecological and agricultural contexts of the schemes under evaluation. I quantified the coverage of economic factors, including assessment of temporal trends, and asked whether the costs of the schemes had been acknowledged and integrated into the evaluation. I found that less than half of the studies acknowledge financial costs and fewer than the 15% conducted any of economic evaluation (e.g. cost-effectiveness analysis). Furthermore, despite steady growth in the number of studies evaluating agri-environmental schemes over the past 15 years, the proportion published annually that integrate economic data remains largely unchanged. I discuss the potential reasons for this poor integration, including limited understanding of, and access to, economic evaluation tools, data and training, and a philosophical aversion to the mixing of economics and conservation. I argue, however, that these reasons are no longer justified, and provide several examples of the effective integration of economic and ecological data in evaluations to assist researchers and decision-makers address this deficiency.

Comparing the effectiveness and efficiency on alternative conservation actions

In the second part of this thesis I extend the principles and practice of cost-effectiveness to the evaluation of restoration plantings and remnant protection, two conservation actions commonly applied for conserving birds in agricultural landscapes within Australia and other parts of the world. This included restoration plantings (i.e. revegetation) in heavily cleared parts of the agricultural landscape, and the protection of remnant vegetation through the erection of fences to control livestock grazing and facilitate natural regeneration in landscapes without as much historic clearing. In Paper III I identify the concept of additionality as a key consideration in the design of cost-effective biodiversity conservation in agri-environmental landscapes. Demonstrating additionality requires measure of effectiveness that considers the counterfactual—the scenario that reflects the absence of the intervention. In Paper VI I present the results of a field evaluation of the effectiveness and cost-effectiveness of restoration plantings and remnant protection using an experimental design that facilitated the measurement of conservation gains relative to the counterfactual scenario. This represents a truer measure of conservation gain than analyses based on comparisons that do not reflect the counterfactual scenario. I conducted 336 bird surveys at 84 sites (32 active and 10 passive restoration each paired with a control) across the Boorowa region in south-eastern Australia (Fig. 4), an area that has seen significant investment in ecological restoration over the past 30 years. Each restoration site was matched with a control representing the counterfactual (Fig. 5), and the difference (i.e. gain) in species richness of three bird assemblages (all species, woodland species and woodland species of conservation concern) between the restoration sites and the matched counterfactual was used as the measure of conservation effectiveness.

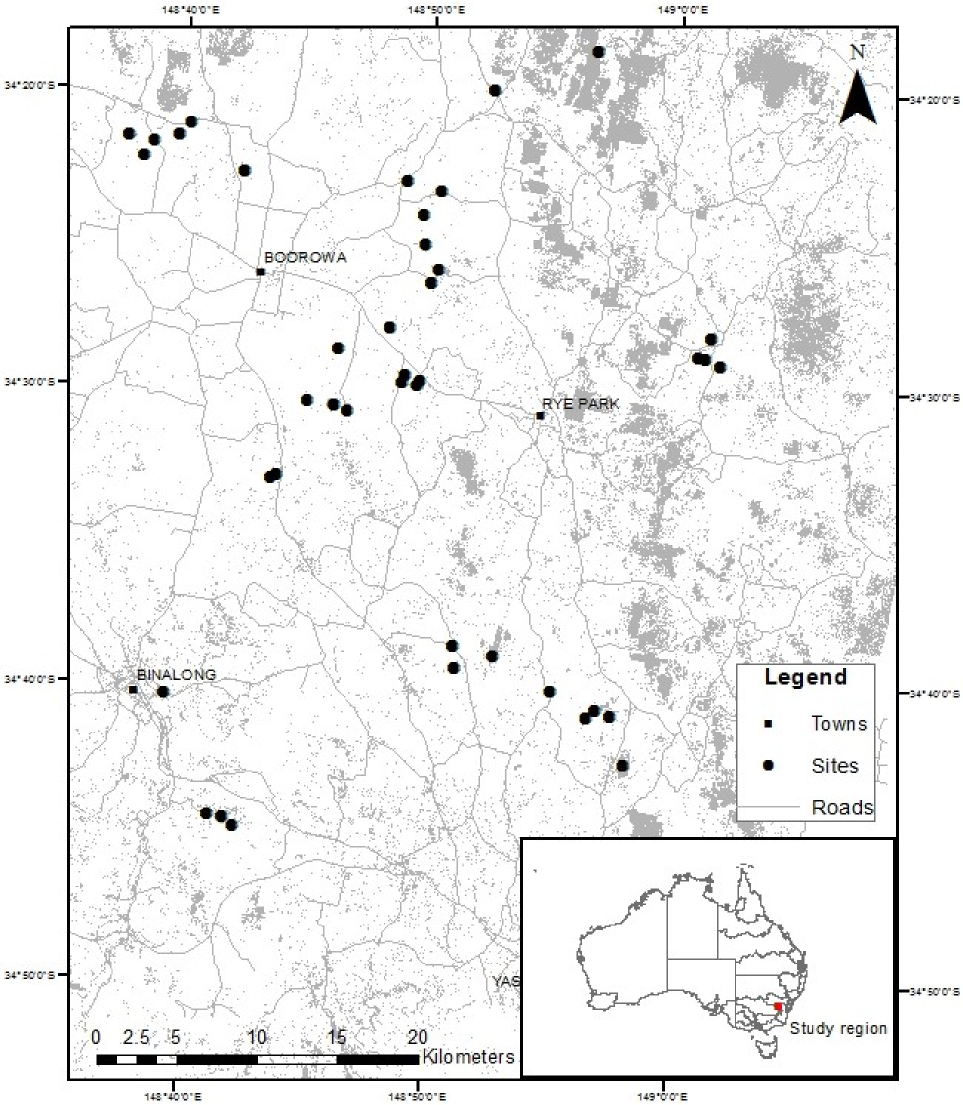

Figure 4. Study area for Papers VI and VII showing the location of bird study sites. The extent of remnant woody vegetation is shown in light grey.

I used Generalized Linear Mixed Models to compare the total richness between the four site types (both restoration treatments, and their respective controls), and used multiple regression models to compare gains in bird species following restoration. I also included various site design and landscape context factors measured through field surveys and spatial data analysis as covariates in my analyses. I found that gains in bird species richness following restoration planting on previously cleared land were more than 60 times greater than those following remnant protection where the counterfactual was continued livestock grazing. This trend was also found in woodland bird species (many of which are considered to be of conservation concern), with gains eight times higher following restoration planting than remnant protection. As a result, I question the emphasis of public investment on passive restoration of remnant habitats as a biodiversity conservation strategy in Australian agricultural landscapes. At the same time however, given observed differences in the bird assemblages found within the two restoration types, I advocate for a multi-faceted approach to bird conservation that combines strategic restoration planting with the continued protection of remnant vegetation, especially where the counterfactual scenario for remnant vegetation is severe degradation or loss. More broadly, my results highlight the importance of considering the counterfactual scenario when evaluating conservation actions and using conservation gains rather than absolute values to measure effectiveness.

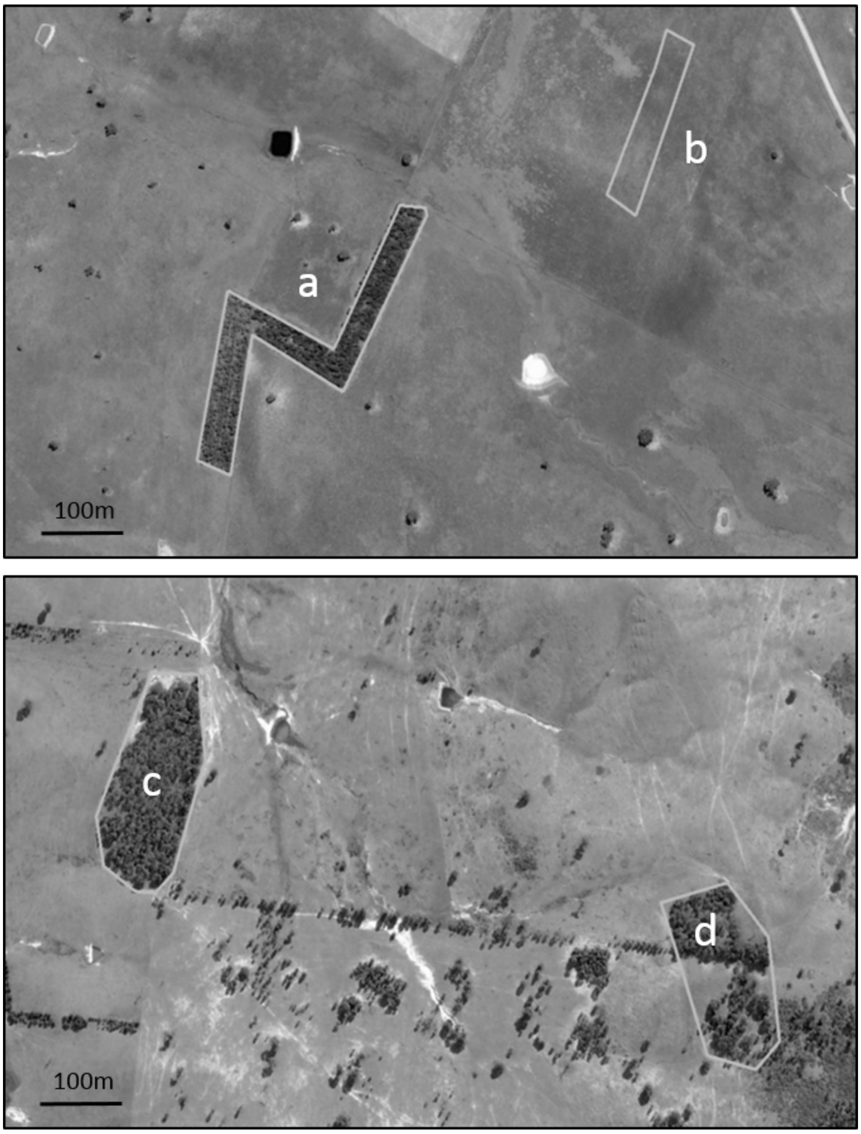

Figure 5. Examples of the two types of restoration approaches evaluated in this thesis: a) restoration plantings, or revegetation, in cleared agricultural land, and; c) remnant protection, or passive restoration of remnant woody vegetation through fencing to exclude livestock. Also shown are the controls sites for b) restoration planting and d) remnant protection sites, representing the counterfactual scenario. Image source: Google Earth V 7.1.5.1557 (13 November 2013 (top); 6 April 2016 (bottom)).

In Paper VII, I extended the analysis undertaken in Paper VI by integrating the costs of conservation in an evaluation of the cost-effectiveness of both restoration plantings and remnant protection in the conservation of woodland birds in agricultural landscapes. I calculated the total estimated public cost of conservation based on management and transaction costs, and combined these data with the measures of conservation effectiveness to derive benefit-cost ratios for each site. I then used these benefit-cost ratios as response variables in Linear Mixed Models to compare cost–effectiveness between the two conservation actions. I demonstrate firstly that, despite having significantly greater costs per hectare, restoration planting represents a more cost-effective strategy in the conservation of woodland birds than protection of remnant protection. I also found that the cost-effectiveness of restoration was strongly influenced by the geometry and size of the site, the direction and magnitude of these effects varying between the conservation actions. In this paper I provided stark demonstration of the variation in the efficiency of alternative actions for achieving a conservation outcome and the strong influence that site design factors can have on effectiveness and efficiency of conservation expenditure. The results not only further question the cost-effectiveness of remnant protection for bird conservation, but also the efficiency of elongated (i.e. windbreak style) plantings, which represent the dominant planting configuration in many farm landscapes.

An important consideration in the interpretation of the results of these studies relates to the objective of the conservation action and the careful selection of an appropriate metric or effectiveness measure. In this particular research I focused on the effectiveness and efficiency of alternative conservation actions for increasing bird species richness. The results obtained will not necessarily translate directly to other measures (e.g. population density, other taxa). The implication is an obvious but nonetheless important one – selection of effectiveness measures should correspond to, or be strongly aligned to the conservation objective.

Whereas Papers VI and VII were derived from a study comparing the relative cost-effectiveness of two alternative conservation actions, in Paper VIII I focused on restoration plantings in an application of contemporary conservation planning. It complements the preceding studies by investigating, in greater detail, the specific design factors that maximize conservation benefit by considering the complementarity of networks of plantings at the landscape scale, and by accounting for temporal dynamism associated with highly mobile species such as birds. Using the software Marxan, we compared the efficiency of a dynamic complementarity approach that selected restoration plantings to maximize occurrences of threatened species across several years, with a static approach that maximized occurrence or richness at a point in time. I estimated the total public establishment cost of each restoration planting, which was then combined with bird survey data collected over five years as part of a long-term ecological research project. Occurrence targets for each species were set as objectives, and the optimization procedure then selected networks of sites that minimized cost while achieving the identified targets. We found that for an equivalent cost, the dynamic complementarity approach resulted in greater conservation benefits—greater minimum occurrence and number of species meeting occurrence targets—than did a richness-based ranking approach. This study also revealed the importance of a diversity of planting attributes in the conservation of woodland birds, with no single planting attribute (e.g. age, area, habitat complexity) influencing site selection in optimal networks. This paper demonstrates that the coupling of economic data with the principles of complementarity represent further potential to increase the cost-effectiveness of conservation actions.

Improving conservation benefits through economics

Many of the studies and discussion Papers I presented in this thesis serve to illustrate efficiencies that can be achieved through the integration of economic factors in the conservation of biodiversity in agricultural landscapes. In Papers IV, VI, VII and VIII I provide empirical evidence for the biodiversity gains that can be achieved with limited resources by considering financial costs in the planning and evaluation of conservation actions. Extensive review of the literature summarised in Papers II and V provide further evidence—by factoring costs into decision-making and designing future programs based on an assessment of cost-effectiveness, we can increase the efficiency of conservation expenditure. Economics, however, represents a much broader science than the calculation of costs with the potential to provide a greater contribution to improving both the biodiversity and social outcomes of conservation. In Paper IX I discuss, this issue in the context of ecological restoration, using five main challenges of restoration projects as a framework to identify how economic principles and practices can improve biodiversity outcomes, particularly in agricultural landscapes. In this paper, borne out of the observation of the trend towards the limited use of economics in the planning and evaluation of restoration, I aimed to broaden awareness of the broader contribution that economics can make to the effectiveness and efficiency of ecological restoration. This includes more comprehensive accounting of the full range of ecological and social benefits and costs of restoration projects through application of economic valuation techniques, and the potential improvements in project uptake and efficiency of conservation expenditure associated with such an approach. The importance of appropriate project prioritization metrics are highlighted through a hypothetical cost-benefit analysis. Developments in the application of economic instruments to the environmental sciences and their potential to contribute to the challenge of long-term restoration project financing are discussed including the use of alternative funding sources, such as private organizations, levies and crowdfunding, as well as better alignment of restoration projects with existing public initiatives. The key message in this paper is that there is much potential for improved outcomes of ecological restoration through broader consideration of economic principles and policies.

Concluding remarks

Biodiversity conservation increasingly needs to compete for limited resources with other key public policy issues such as health, infrastructure and economic development. This dictates the need for conservation actions to demonstrate a similar level of accountability and efficiency as those competing areas which, in turn, requires evidence-based measures of effectiveness and evaluation of cost-effectiveness. Despite recent high profile examples of the application of these principles in applied conservation (e.g. Addison and Walshe, 2015), there is much opportunity for improvement in the current conservation evaluation and planning paradigm.

Though broad in its examination of this issue, some common themes emerge from this examination of the cost-effectiveness of conservation in agricultural landscapes. Firstly, it highlights the benefits of multi-disciplinary approaches to the planning and evaluation of conservation actions, which can ultimately manifest in greater biodiversity outcomes with available resources. The evaluations presented and profiled throughout this thesis serve to demonstrate both the simplicity and versatility of the techniques available and the potential magnitude of the conservation benefits that can accrue through their use. One possible factor behind the slow uptake of economic factors in conservation evaluation observed here, and reported by others (Medvecky, 2015), is an ethically-driven opposition to the mingling of economics and nature. Such views, possibly fuelled by the debate over the monetary valuation of nature (e.g. species, ecosystems) for use in more traditional economic valuations such as cost-benefit analysis, are becoming increasingly unrealistic in modern conservation practice. Hopefully, this thesis illustrates that consideration of costs and cost-effectiveness can enhance, rather than threaten, the integrity of conservation research and practice.

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Paper I. Introduction: Framing the agri-environment.

Agri-environment schemes represent a major policy mechanism for biodiversity conservation in agricultural landscapes. In Paper I, the first chapter from a book I co-wrote and co-edited, *Learning from agri-environment schemes in Australia*, I discuss the ecological and economic basis idea the history of their application in the Australian context.

Photo: D. Ansell

Ansell, D., Gibson, F., Salt, D., 2016. Introduction: Framing the agri-environment, in: Ansell, D., Gibson, F., Salt, D. (Eds.), Learning from Agri-Environment Schemes in Australia: Investing in Biodiversity and Other Ecosystem Services on Farms. ANU Press, Canberra, Australia, pp. 1–16.

1

Introduction: Framing the agri-environment

Dean Ansell, Fiona Gibson, and David Salt

Conservation in an agricultural space

Do our agricultural landscapes hold the key to protecting our declining biodiversity? If they do, how would it be done? And who would pay? Would it be the landowner, or the general public (via the government)? These might sound like simple questions, but when you consider some of the environmental, social, and economic factors at play, it quickly becomes apparent that we are dealing with very complex issues.

To illustrate this, consider these two relatively simple situations, both examples of efforts to conserve biodiversity on farmland in Australia. The first involves a run-down paddock from which the landowner has removed his sheep and sown a mixture of native trees and shrubs in strips several metres apart. In exchange for a stewardship payment of $50 per hectare per year, the farmer agrees to keep his sheep out of the paddock for five years. He gets half the payment at the beginning and the rest at the end of the initial five-year period, at which time grazing stock are permitted back into the paddock under a regime where sheep are allowed into the site in short bursts (called 'pulse grazing') for the last five years of the agreement. By this time, the native vegetation should have developed enough to be able to cope with the reintroduction of grazing. Indeed, the presence of trees and shrubs will provide the grazing animals with valuable shelter.

Figure 1.1: Do our agricultural landscapes hold the key to protecting our declining biodiversity?
Source: Photo by Greening Australia.

The second situation involves a farmer agreeing to remove grazing sheep from a patch of box gum grassy woodland — an ecosystem now threatened in Australia. The farmer is allowed to let sheep into the woodland for pulse grazing, whereas previously the woodland experienced set stocking, meaning a certain number of animals were always there. The landowner also agreed not to use fertiliser in the woodland. For these actions, the government is prepared to pay the farmer over $200 per hectare per year, and the farmer has entered into a contract that will run for 15 years.

The first situation describes a process of restoration, with the aim of returning native vegetation to the landscape. It is about improving the natural value of degraded land, providing habitat for biodiversity and other environmental benefits. The second example is more about the preservation or conservation of an existing ecosystem. It is about sustaining the health and resilience of land with high natural values. Both schemes are undertaken in production landscapes, and the land under each scheme is expected to continue to provide agricultural outputs into the future.

Even with these simple descriptions, many questions immediately arise:

- Which approach is better for biodiversity, restoration, and/or conservation?
- Where do we get the best value for money? One farmer is paid four times the amount the other farmer receives; do we receive four times the return?
- Why should the government pay for a scheme which benefits the farmer (in the case of new trees providing shelter for stock)?
- Why does one scheme only run for 10 years when the other goes for 15?

Of course, there are many answers to each of these questions given by different groups. 'Which approach is better?', for example, would most likely be responded to differently by ecologists, economists, farmers, policymakers, and the public — and there would be considerable variation within each group. This variation simply underscores the complexity and uncertainty surrounding the operation of these schemes.

The two case studies described here are far from hypothetical exercises. They are based on real-life examples of publicly funded programs currently in operation on farmland in south eastern Australia. The first example (restoration) is called the Whole of Paddock Rehabilitation scheme (WOPR) being operated by Greening Australia (an environmental non-government organisation (eNGO)). The second case study (conservation) is part of an Australian Government program called the Environmental Stewardship Program. Both are described in more detail in this book (see Chapter 2 by Graham Field for background on WOPR, and Chapter 3 by Emma Burns and colleagues on the Environmental Stewardship Program).

Figure 1.2: Agri-environment researchers and practitioners in a five-year-old WOPR site.
Source: Photo by David Salt.

In addition to having differing aims, payments, and duration, the schemes are also quite different in how they were developed and managed. WOPR came out of a grass-roots engagement between farmers and Greening Australia. The Environmental Stewardship Program was developed as a top-down government program to protect natural values that are considered to have national significance — in this case, the conservation of a threatened ecosystem. WOPR involved many 'back paddock' experiments, custom-made equipment, discussion, reflection, and trial and error (Streatfield et al. 2010). The Environmental Stewardship Program involved ecological, economic, and social science inputs, the development of legal contracts and the setting aside of funds beyond the traditional three- to four-year budget cycle.

WOPR and the Environmental Stewardship Program are but two examples of what are commonly referred to as 'agri-environment schemes'. There are many other variations of such schemes in Australia and around the world. Some, like WOPR, aim at restoring lost natural values. Others, like the Environmental Stewardship Program, aim to modify existing practice to conserve natural values.

We are not holding up these two schemes as examples of good or bad schemes. Rather, the differences between them offer a valuable reference point to discuss the strengths and weaknesses of society's effort to achieve environmental outcomes, generally regarded as public goods and services, from working agricultural landscapes, generally operated in the private realm. The particular environmental outcome this book focuses on is the conservation of biodiversity.

Before we begin to explore the many issues surrounding the design and implementation of effective agri-environment schemes, it is worth reflecting on the relationship between agriculture and biodiversity.

Why our farms are part of the solution

What is the connection between biodiversity conservation and our agricultural landscapes? Doesn't government look after biodiversity on behalf of the public through the creation and operation of national parks and nature reserves? Biodiversity conservation is an important goal of the management of most national parks, but the sad truth is that the world's system of nature reserves is not protecting biodiversity. A mere 15 per cent of threatened species on land are adequately covered by the existing network of reserves (Venter et al. 2014). In Australia, 80 per cent of threatened species are inadequately protected by the reserve system, with 12 per cent receiving no protection at all (Watson et al. 2010).

This is important because the world is witnessing a crisis of declining biodiversity. Species are being lost at 100–1,000 times what is believed to be the natural background rate of extinction, which scientists believe may have profound consequences for the future of human civilisation (Rockström et al. 2009). Governments around the world have signed up to the Convention on Biological Diversity, pledging that they will take actions that will slow and hopefully reverse these declines (Watson et al. 2014). To date, despite this commitment, little has been achieved. The fourth Global Biodiversity Outlook released by the United Nations in 2014 revealed that the rate of species loss is increasing and that the five principal drivers of extinction — habitat change, overexploitation, pollution, invasive species, and climate change — are getting worse (Secretariat of the Convention on Biological Diversity 2014).

Figure 1.3: An Environmental Stewardship Program site — a box gum grassy woodland in which grazing has been modified to protect the woodland's natural values.

Source: Photo by David Salt.

So what is the connection with farming? There are several broad areas to consider. The first relates to the point made above: our public reserve system is simply not providing adequate protection to our threatened biodiversity, as most threatened species and ecosystems lie outside of reserves, much of it on and around agricultural land. At least 40 per cent of global land surface is used for agriculture (Foley et al. 2005). In Australia, agriculture accounts for more than half of the land surface, with the majority of that land (86 per cent) used for grazing (Australian Bureau of Statistics 2014). If we want to conserve our biodiversity, we need to focus our efforts on agricultural land.

The second area relates to the impact of agriculture on biodiversity. About 70 per cent of the projected global loss of terrestrial biodiversity is attributed to agricultural drivers (Secretariat of the Convention on Biological Diversity 2014). The conversion of land to agriculture results in the loss and degradation of habitats. This directly impacts on plant and animal populations and communities, and alters ecological and hydrologic processes that underpin key ecosystem functions (Millennium Ecosystem Assessment 2005). Australia's settlement by Europeans over two centuries ago was followed by rapid and extensive landscape modification, as the settlers sought to tame the bush and establish grazing and cropping land. The initial focus was on clearing temperate grasslands and grassy woodlands (Kirkpatrick 1999). Records suggest that approximately half of the woody vegetation in Australia has been cleared since European settlement (Barson et al. 2000).

In addition to habitat loss, farming practices such as tillage, burning, livestock introduction, and nutrient and chemical usage have had significant negative impacts on biodiversity as well as soil, water, and air quality (Stoate et al. 2001). The early to mid-1900s saw a shift from smaller, low-input, mixed-enterprise farms to more intensive, specialised systems focusing on increased yields from fewer commodities, bringing with it increased fertiliser and pesticide use, and further loss of natural and man-made habitat (Bignal and McCracken 2000; Young et al. 2007). It is undeniable: agriculture has contributed greatly to the global decline in biodiversity.

The third area concerns the importance of biodiversity to the sustainability of our agricultural enterprises. Some elements of biodiversity underpin the quality and quantity of our agricultural output, through the provision of 'ecosystem services' — the wide range of benefits that we receive from ecosystems, ranging from food and water to recreation and cultural use. For example, bee pollination contributes more than €1 billion every year to Europe's strawberry producers (Klatt et al. 2014). Biodiversity can also provide benefits in the control of agricultural insect pests, improved soil fertility, and agricultural productivity (Altieri 1999). (The perceived importance of ecosystem goods and services to farmers is discussed by Saul Cunningham in Chapter 8.)

Lastly, as well as improving the financial outcome from farming, biodiversity benefits farmers by improving the amenity value of some properties and satisfying some farmers' goals of stewardship. (See Chapter 14 by Maksym Polyakov and David Pannell on the private benefits of biodiversity, and Chapter 12 by Saan Ecker and Chapter 13 by Romy Greiner on non-financial drivers of biodiversity conservation.) Biodiversity is also known to influence peoples' health and well-being (Keniger et al. 2013).

The bottom line is that agriculture requires the support of a raft of ecosystem services. The problem is that some of these services are valued more highly than others by agricultural producers, whose values may not align with those of the broader public. Juggling these contrasting values is one of the major challenges of farmland environmental management, and is a key theme throughout this book.

Solutions in the agri-environment

To anyone with an interest in conservation and agriculture, these ideas are hardly revolutionary. The question is, what can we do to conserve biodiversity in productive landscapes? There are many answers here, ranging from individual farmers volunteering their time and effort to re-establish native plants and animals on their farms, through to governments proclaiming laws regulating what farmers can and cannot do. As a spectrum of activity, these approaches might represent end points, going from volunteer effort through to regulation.

Most landowners have a limited capacity to sacrifice the productive capacity of their land (or their time) for non-income earning activities, and volunteer efforts have real limitations on what can be achieved (Curtis 2000). Indeed, the early investment in agri-environmental policy in the 1980s and 1990s focused on stimulating volunteer effort through programs such as Landcare. While popular, this effort failed to address the growing problems of land and water degradation and declining biodiversity (see Chapter 7 by David Salt).

Regulatory approaches, on the other hand, usually entail high transaction costs — especially, for example, in terms of compliance and enforcement — and are widely considered less efficient and cost-effective than alternative strategies (Hahn and Stavins 1992). They are also often unpopular in the agricultural sector. Indeed, the prevailing belief in most western democracies is that farmers have the implicit right to carry out the most profit-maximising activity on their land, irrespective of the external costs (and benefits) of doing so (Hanley et al. 1999). Regulation is usually only introduced where the activity is seen as being clearly unacceptable by the broader population, such as controlling the use of dangerous chemicals or the unacceptable treatment of livestock.

Between volunteering and regulation, however, there are many options employed and implemented by governments and conservation groups around the world, chief among which is the agri-environment scheme. Agri-environment schemes, though highly variable in their structure and application, can be broadly defined as programs involving payments to farmers in exchange for the provision of environmental goods and services (Burrell 2012; European Commission 2014). Most involve an acknowledgement that the farmer is sacrificing some aspect of their productive potential by providing environmental goods and services for the public good. The two case studies discussed at the beginning of this introductory chapter are examples of agri-environment schemes.

Over time, agri-environment schemes have attracted a growing share of government investment in agriculture across Organisation for Economic Co-operation and Development (OECD) countries, and now represent a significant component of biodiversity conservation in agricultural landscapes, with billions of dollars spent on such schemes around the world each year (Hajkowicz 2009).

Along with Europe and the United States, Australia has been working in the agri-environmental space for some 30 years. Australia's investment in this area has been tiny compared to Europe or the United States, partly reflecting our smaller population and economy, although the size of our agricultural landscape is comparable (Hajkowicz 2009). Given the enormous scale of the environmental challenges being faced in Australia, it is important that our investments in the agri-environment area are cost-effective.

Learning from agri-environment schemes in Australia: About this book

This book is targeted primarily at anyone working in agri-environmental policy or looking at establishing an agri-environment scheme in Australia, including policymakers, project officers, and non-government organisations. It has a secondary aim of producing a short and readable text for anyone interested in the topic of biodiversity conservation on agricultural land.

Chapters are short, engaging, and seek to educate rather than exhaustively prove finer points of analysis. Where possible, we have kept the use of jargon and acronyms to a minimum. Each chapter is a stand-alone story, and we have organised the book into the following three themed sections.

Part I — The agri-environment in the real world sets the scene by describing the challenges and tensions that go hand in hand with running agri-environment schemes. The chapters in this section present a variety of discussions of the complexities surrounding how agri-environment schemes function in real life, and discuss the case studies of the WOPR scheme and the Environmental Stewardship Program, which were presented at the beginning of this chapter. Part I also provides some contextual history of agri-environment schemes in Australia and Europe, and discusses dealing with different types of farmers and the importance of non-government organisations.

Part II — The birds and the beef explores the many natural, social, and economic values involved in agri-environment schemes, and the ways these are framed or marketed. In this section, we discuss the concept of ecosystem services, consider the debate over different conservation strategies, and are presented with an economics perspective on restoration. We also explore the issue of scale in designing agri-environment schemes, the importance of accounting for private benefits in project selection, and the social and psychological dimensions of agri-environment schemes.

Part III — Planning, doing and learning examines many of the issues surrounding the design, implementation, and evaluation of agri-environment schemes. It examines the many challenges of ranking different projects, given that most schemes are oversubscribed — there is never enough money to go around, so how do you get the best outcomes? We discuss approaches to measuring and maximising the conservation benefits, the importance of counterfactual thinking, and the choice of different policy tools. We conclude with the reflections of David Pannell, one of Australia's most experienced agricultural economists, on the performance of agri-environmental policies. He provides a checklist of factors that experience has shown are important to the success of any agri-environment scheme. For anyone with an interest or responsibility in agri-environment policy, this is one list you cannot afford to ignore.

So, what does it all this add up to? We attempt to make sense of the many perspectives in this book in the concluding chapter. We begin our conclusion with a simple hypothetical: if circumstances were to suddenly create a funding opportunity for a new agri-environment scheme, how should the nation respond? This is not idle speculation, because in many ways the Decade of Landcare was not an opportunity that was widely anticipated. It arose from a historic agreement between the National Farmers Federation and the Australian Conservation Foundation in 1989, coupled with a receptive prime minister. And while that scheme was enthusiastically embraced, it did not generate the enduring environmental outcomes that many hoped for.

A quarter of a century later, and the threat of environmental decline is as great, if not greater, with a rising expectation that our agricultural landscapes will dramatically increase their productive output in order to feed a growing population (see Box 1.1). Furthermore, biodiversity decline is just one of several issues facing society that must compete for limited funds.

Box 1.1: Farming, biodiversity and the future

The world's population is changing rapidly. In the next three decades there will be up to 10 billion people on the planet; Australia's population alone is expected to double by 2075 (Australian Bureau of Statistics 2013). Not only can we expect a lot more mouths to feed, but improvements to the socio-economic status of people across many regions, including Asia and Africa, will lead to changes in diet. This will result in a large increase in food demand, which will in turn require increased food production through the expansion and intensification of agriculture (Phalan, Green, and Balmford 2014). We will need to produce more with less.

The Australian Government's agricultural policy is heavily focused on capitalising on this growth by increasing productivity. The National Food Plan seeks to increase agricultural productivity by 30 per cent by 2025, aiming to increase the value of agricultural exports by 45 per cent (DAFF 2013). The Agricultural Competitiveness Green Paper sets out a plan to increase farm-gate profits by reducing costs and 'unnecessary barriers to productivity and profitability' (Commonwealth of Australia 2014). At the same time, this policy is aiming to 'streamline' the environmental approvals established through key legislation such as the *Environmental Protection and Biodiversity Conservation Act 1999*.

These major changes to agriculture present a significant threat to our biodiversity. Agricultural intensification carries greater biodiversity impacts than extensive farming practices (Reidsma et al. 2006). The amount of remnant vegetation expected to be cleared globally for agricultural use in the next 35 years is in the order of 0.2–1 billion hectares (Tilman et al. 2011). Facilitating the conservation of biodiversity in agricultural landscapes in the face of growing agricultural production represents a key conservation challenge at a global scale (Green et al. 2005). Policies that use incentives to balance conservation and agricultural production will play an increasingly vital role in safeguarding biodiversity in agricultural landscapes.

Should another major opportunity present itself — the announcement of a substantial government investment in agri-environment schemes, for example — will we be able to say we are ready? We should be, after 25 years of experience and research in these programs. Many of the perspectives in this book question our efforts in agri-environment investment and ask exactly what we have learnt. In many places it is suggested we can do a lot better than we currently do with the available resources, in areas including planning, prioritisation, monitoring, evaluation, and learning. In that light, it is our hope that this book will prove an invaluable resource and reference.

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Paper II. Defining and designing cost-effective agri-environment schemes.

In Paper I, I discuss the theory and practice of agri-environment schemes as a primary mechanism for conserving biodiversity in agricultural landscapes. In Paper II, also taken from our book, I extend this discussion by highlighting the economic costs of such schemes, the variability in costs and the impact of this variability on cost-effectiveness. I draw on examples from the agri-environmental literature to demonstrate the broad potential of economic valuation methodologies in the evaluation of biodiversity conservation, and the potential economic and biodiversity benefits of considering cost-effectiveness in program design.

Photo: D. Salt

Ansell, D., 2016. Defining and designing cost-effective agri-environment schemes, in: Ansell, D., Gibson, F., Salt, D. (Eds.), Learning from Agri-Environment Schemes in Australia: Investing in Biodiversity and Other Ecosystem Services on Farms. ANU Press, Canberra, Australia, pp. 193–206.

15

Defining and designing cost-effective agri-environment schemes

Dean Ansell

Key lessons

- Agri-environment schemes are often highly variable in both their economic cost and biodiversity benefit, creating the potential for significant inefficiencies in conservation expenditure.
- Evaluation of the cost-effectiveness of agri-environment schemes can identify opportunities to significantly improve the conservation gains with the available resources, however, such evaluations are uncommon.
- Simple economic evaluation tools can be applied by researchers or policymakers, using minimal economic data, to compare the cost-effectiveness of agri-environment schemes at different scales and at stages through the implementation process.

Introduction

Over the past decade, concerns have been raised regarding the effectiveness of agri-environment schemes in conserving biodiversity. Studies have shown that the success of these schemes is highly variable, ranging from strong positive biodiversity benefits to neutral and even negative consequences. With global biodiversity declining dramatically and further threatened by agricultural intensification, a focus on the most effective strategies for conservation is critical. This issue is even more pertinent given that the funds available for biodiversity conservation are not sufficient to address the scale of the problem, and so agri-environment schemes are in competition with other conservation activities for limited resources. It is critical, therefore, that the cost-effectiveness of agri-environment schemes are maximised to increase the biodiversity benefits obtained with available resources.

Figure 15.1: The cost-effectiveness of agri-environment schemes can be influenced by many factors, from the location of sites to the specific conservation techniques used.

Source: Photo by Brisbane City Council available at www.flickr.com/photos/brisbanecitycouncil/7926277216 under a Creative Commons Attribution 2.0.

Typically, evaluation of agri-environment schemes has been dominated by ecological or economic perspectives (Uthes and Matzdorf 2013). There has been comparatively little attention given to the cost-effectiveness of these schemes. A recent review of 239 studies on the effectiveness of agri-environment schemes around the world found that fewer than 15 per cent considered economic costs in the evaluation

(Ansell et al. in preparation). This is surprising, given the scale of the public expenditure in agri-environment schemes and increasing recognition of the biodiversity benefits that can be achieved through consideration of economic costs in the conservation planning process (see Chapter 17 by Fiona Gibson and colleagues).

This chapter explores issues of the effectiveness and efficiency of agri-environment schemes, first defining cost-effectiveness in the context of such schemes, and providing an overview of common evaluation approaches. I conclude with a discussion on the outcomes of previous evaluations of the cost-effectiveness of agri-environment schemes, highlighting key aspects relevant to the design and implementation of agri-environment schemes in Australia.

What is cost-effectiveness?

Cost-effectiveness refers to the relative efficiency of an action in achieving an outcome. It can be expressed as the total cost of producing a single unit of benefit (i.e. cost/benefit), or alternatively as the total benefit produced per unit of cost (i.e. benefit/cost) (Wätzold and Schwerdtner 2005). Both approaches generate a ratio, referred to as the cost-effectiveness ratio or benefit–cost ratio, which forms the basis of a cost-effectiveness analysis. The ratio allows one to compare the efficiency of alternative actions. Note that while I focus here on evaluation approaches involving non-monetary measures of conservation benefit, as opposed to methods that assign a monetary value to the effectiveness measure, the concepts discussed apply generally across both approaches.

When applied to the evaluation of biodiversity benefits of agri-environment schemes, we can take cost-effectiveness as the biodiversity benefit produced per unit of cost (or, alternatively, cost per biodiversity unit). Comparison of the cost-effectiveness of different agri-environment schemes allows identification of those that provide the greatest biodiversity benefit per dollar spent.

Variation in both the economic costs and effectiveness of conservation activities give rise to differences in the cost-effectiveness of agri-environment schemes (Wätzold and Schwerdtner 2005). Agricultural ventures are rarely static in time and space, with farming practices, production intensity, and commodity choice varying according to various external market factors (Barraquand

and Martinet 2011). This gives rise to significant variation in the opportunity costs of conservation on farmland at multiple spatial and temporal scales. Similarly, the effectiveness of agri-environment schemes in conserving biodiversity is highly variable, both temporally and spatially, and is influenced by factors at the field or farm scale (e.g. site size, management history) as well as at the landscape or regional scales (e.g. surrounding land use, connectivity, climate) (e.g. Concepción and Díaz 2011). Effectiveness also varies between taxa, with some schemes providing positive conservation outcomes for some taxa while providing no benefit or even negative outcomes for other taxa (Kleijn et al. 2006).

Complex interdependencies also exist between the effectiveness and cost of conservation activities in agri-environment schemes. For example, the overall cost of an agri-environment scheme is strongly influenced by the configuration (i.e. size and shape) of the particular field, with larger sites incurring a higher opportunity cost to the landholder, in turn requiring an increased payment rate, and often requiring increased materials. This can also influence biodiversity outcomes, with factors such as field size and shape shown to be important determinants of conservation effectiveness (Conover et al. 2011). This variation in costs and effectiveness, and the complex interactions between the two, create the potential for significant inefficiencies in conservation expenditure. Simple evaluation of the cost-effectiveness of agri-environment schemes can reveal factors driving conservation efficiency and identify opportunities to maximise the conservation benefits from investments.

Cost-effectiveness in practice

A critical step in the evaluation of the cost-effectiveness of agri-environment expenditure is the assessment of the costs of the scheme itself. The costs associated with agri-environment schemes can be categorised as acquisition (e.g. land rent), management (e.g. site establishment, maintenance), transaction (e.g. negotiation, legal), and opportunity costs (e.g. forgone agricultural production) (Naidoo and Ricketts 2006). Consideration of the latter is particularly critical, as it can influence the design, uptake, and ultimately the effectiveness of biodiversity conservation in farmland, but is often omitted from project evaluations.

Evaluations can use realised or actual costs (Klimek et al. 2008), or estimated costs based on market rates, averages, or surrogates (Bamière et al. 2013). While evaluation based on actual costs provides improved accuracy, such information is not always readily available. Naidoo and Ricketts (2006) provide an overview of approaches for the estimation of common cost components of biodiversity conservation. Evaluations should attempt to take account of the full costs (and benefits) of agri-environment schemes (Bamière et al. 2013).

The other key component in the evaluation of cost-effectiveness is obviously the measure of the benefit or the effectiveness of conservation. As the numerator in most cost-effectiveness equations, it can strongly influence the outcomes of the evaluation and therefore careful selection is critical. Evaluation can use direct or field-based measures of effectiveness such as changes in single species abundance or density, or, alternatively, look at measures of community diversity (e.g. Ulber et al. 2011). Measures of habitat area or quality have also been used, either as direct measures of agri-environment schemes effectiveness (e.g. Wynn 2002), or as surrogate measures of broader biodiversity benefits (e.g. Hansen 2007). Thompson et al. (1999) use area of land enrolled in the particular agri-environment schemes under review as a proxy for effectiveness.

Many evaluations, particularly those carried out at the planning stages of agri-environment schemes (i.e. ex ante, see below) are based on modelled or predicted outcomes as measures of effectiveness. For example, Bamière et al. (2013) use spatial modelling to assess the efficiency of agri-environment policies by focusing on the spatial configuration of farm land for habitat conservation, specifically aiming for a random mosaic of sites, noting that such a configuration is more effective in the conservation of certain species, such as their model species, the little bustard, which depends on a mosaic of agricultural land use (i.e. crops, grassland).

Except where the particular objectives of the agri-environment schemes or research question dictates the use of a specific measure of effectiveness — for example, changes in the abundance of a species or in the area of a certain habitat — the researcher will be faced with the difficult task of selecting a suitable measure to capture, to the extent possible, the extent of the biodiversity benefits resulting from the scheme. In such cases, multiple benefits can be captured in a metric that can then be used in economic analysis. Multi-criteria analysis approaches have also been used to combine multiple disparate environmental values into single measures (e.g. Hajkowicz and Collins 2009). (In Chapter 17, Fiona Gibson and David Pannell discuss the consequences of using the wrong metric, while in Chapter 20 Phil Gibbons provides an overview of the development of metrics.)

Irrespective of the particular effectiveness measure selected, careful consideration should be given to the means of collecting that information and its expression within the cost-effectiveness evaluation. Experimental design is critical in the assessment of cost-effectiveness. Failure to account for conservation status in the absence of the treatment (i.e. control or counterfactual) can lead to inflated measures of benefit and cost-effectiveness (a topic discussed by Duncan and Reich in Chapter 19). Kleijn and Sutherland (2003) found significant shortcomings in the design of studies evaluating the effectiveness of European agri-environment schemes, stemming largely from either poor, or absent, controls. The authors propose a number of remedies, including the use of baseline data, comparison of trends in treatments and controls, and use of carefully selected treatment and control site pairs. Of particular importance is the use of conservation gain (i.e. the difference in the biodiversity value between the treatment and control) as the measure of conservation benefit, rather than absolute values. This provides a more accurate measure of the benefit that has been purchased with the investment and controls for differences in the baseline condition or value (Maron et al. 2013).

Cost-effectiveness can be considered at a variety of scales in the agri-environment schemes process. For example, we can consider the efficiency of different agri-environment policies in achieving environmental outcomes at a broad scale. Bamière et al. (2013) use a modelling approach to compare the cost-efficiency of three different agri-environment schemes, each using a different incentive mechanism, in achieving a specific objective for the conservation of little bustard habitat in French farmland. In contrast, we can compare the cost-effectiveness of specific measures in achieving their biodiversity objectives. Wilson et al. (2007) evaluate the cost-effectiveness of two different conservation activities (a low cost habitat preservation option and a high cost habitat restoration option) aimed at improving wading bird populations in southern England under the Environmentally Sensitive Areas scheme. They find that, despite the habitat restoration measure costing 50 per cent more per hectare than the habitat preservation measure, the return on investment from the higher cost option, measured as cost per breeding pair of waders, was more than 90 per cent higher than the low-cost option. This provides a strong example of the power of a simple cost-effectiveness analysis in comparing the efficiency of different conservation activities. The choice of scale for evaluation should be appropriate for the research or policy question, and will influence the detail or resolution of the ecological and economic information required in the evaluation.

Box 15.1: Before, during and after — timing of AES evaluation.

We can also consider the cost-effectiveness of agri-environment schemes at different time stages throughout the process. Evaluations carried out prior to the implementation are referred to as ex ante evaluations and can provide important input into scheme design and implementation. Such evaluations provide the opportunity to optimise the efficiency and biodiversity benefits of agri-environment schemes investments. Van der Horst (2007) assessed the efficiency gains from spatial targeting of a woodland agri-environment schemes and found biodiversity gains of 1.6–2.1 times greater than that achieved through the untargeted scheme. White and Sadler (2011) achieve a 17 per cent budget saving through the use of conservation contracts based on variable payments tailored to outcomes achieved on individual enrolled farms compared to traditional fixed-price contracts.

Evaluations can also be carried out during (in media res) or upon completion (ex post) of a scheme. In contrast to ex ante evaluations, which typically involve modelling of predicted biodiversity benefits and costs, such evaluations can use realised benefits and actual costs as inputs, provide a retrospective assessment of the efficiency of expenditure, and identify improvements for future programs. Both ex ante and ex post evaluations provide useful information about the biodiversity benefits of agri-environment schemes. While the use of predicted biodiversity benefit and cost information in ex ante evaluations may provide less accurate information than approaches using realised benefits and actual costs (i.e. ex ante evaluations) (Boardman et al. 2010), conducting evaluations at this stage may improve the efficiency of an agri-environment scheme before funding is expensed. In contrast, ex post evaluations, while providing information to improve the efficiency of future expenditure, can be hampered by limited availability of financial data and methodological issues around the measurement of biodiversity gains.

Despite these shortcomings, both approaches can contribute to the refinement of agri-environment schemes and increase the biodiversity gains and efficiency of agri-environment expenditure. Ultimately, the choice of evaluation approach may be determined by financial and logistical constraints.

Lessons learnt

While we may be tempted to think that the more we spend on agri-environment schemes, the better the biodiversity outcomes, evaluations reveal the relationship between the two is anything but straightforward. While some studies support this concept by demonstrating higher levels of conservation benefit with increasing expenditure (Barraquand and Martinet 2011; Wilson et al. 2007), others reveal more complex relationships. For example, in an ex post evaluation of the Scottish Woodland Grants Scheme, which aimed to improve priority habitats in farmland, Wynn (2002) found wide variation in cost, biodiversity benefit, and cost-effectiveness across different farm types.

Shining a light on the economics of biodiversity conservation in farming landscapes can reveal some ugly truths that would otherwise not be uncovered by traditional ecological evaluations. Examples include the prevalence of significant windfall effects in agri-environment schemes, where farmers receive payments for environmental services or biodiversity outcomes that would have occurred regardless of whether the scheme was implemented (Chabé-Ferret and Subervie 2013; Sierra and Russman 2006; Ulber et al. 2011), and a reliance on agri-environment schemes payments for farm income (Pietzsch et al. 2013). Recent modelling of the cost-effectiveness of habitat restoration on Australian farmland suggests that our current focus on restoring remnant habitats, as is the focus of the Australian Government's largest agri-environmental scheme, the Environmental Stewardship Scheme (see Chapter 3 by Burns and colleagues), is suboptimal, with revegetation of cleared areas demonstrating higher biodiversity gains per dollar spent (Jellinek et al. 2014).

While it is important that seemingly negative research outcomes such as these be evaluated and communicated, there is a potential risk of perverse conservation outcomes where seemingly adverse economic results drive policy decisions (i.e. cancellation of programs) at the expense of important biodiversity values or priorities. The challenge is in maintaining perspective in evaluating the cost-effectiveness of agri-environment schemes and assessing the outcomes of such evaluations in the context of the scheme's overall biodiversity objectives.

Evaluation can also provide important lessons for the design of future conservation programs. Many agri-environment schemes use a simple incentive system, where payments to farmers are based on fixed rates per hectare. Such approaches are relatively easy to administer, but risk significant inefficiency though overcompensation of farmers otherwise willing to accept a lower price for conservation (Klimek et al. 2008). This can also exacerbate the problem of marginal, low productivity areas dominating the enrolled land as farmers seek to minimise opportunity cost and maximise returns from enrolment (Bamière et al. 2013).

Several studies demonstrate the efficiency gains that can be achieved through more complex delivery mechanisms, such as auction-based and payment-by-results type systems (e.g. Barraquand and Martinet 2011; Thompson et al. 1999; Klimek et al. 2008). Stoneham et al. (2003) compared the outcomes of a pilot auction for the Victorian BushTender scheme and found such an approach would achieve the same biodiversity outcomes at a cost seven times less than those achieved using a fixed-rate incentive payment. The increased efficiency of these approaches, however, must be balanced against the higher administrative or transaction costs associated with their implementation (Klimek et al. 2008; White and Sadler 2011).

Conclusion

It is unfortunate that better use is not made of simple tools of economic evaluation in the planning and assessment of conservation expenditure in agricultural land. By focusing only on biological or ecological aspects in our evaluations, we miss opportunities to significantly increase the biodiversity benefits that can be achieved with the limited funding available. As demands to feed a growing population place even greater pressure on biodiversity in agricultural landscapes and the conservation purse strings tighten, maximising the efficiency of our conservation dollar becomes even more critical. Understanding the cost-effectiveness of our agri-environment investments is a critical step towards meeting this aim.

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Paper III. Conclusion — Elements of good design.

In Paper III, the final chapter in our book, we distill the key points taken from the contributions of more than 22 authors representing the fields of ecology, economics and social science, as well as restoration practitioners and policy-makers, in an exploration of what drives effective, and cost-effective biodiversity conservation on farms. This paper identifies the key factors that require consideration in the design of cost-effective agri-environment schemes in the future.

Photo: D. Ansell

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Conclusion — Elements of good design

Dean Ansell, Fiona Gibson and David Salt

> Breaking News — Australia's national agricultural lobby, Farmers for Farmers, have just signed a historic accord with the conservation lobby, Conservationists at Large, pledging a redoubled effort to renew the natural values of our national farming estate. What makes the accord particularly noteworthy is that the federal government has acknowledged the importance of this new consensus and has pledged $3 billion over five years to reverse the rising rate of extinctions, and declining quality of our land and water resources. The investment will be made primarily through a ramp up of the country's agri-environment schemes. 'This is a once in a lifetime opportunity', says the prime minister.

Of course, this is a hypothetical news story, but you never know what lies around the political corner. The Decade of Landcare announced in 1989 was not anticipated by many in the years preceding it. While it was well received by all and sundry, it did not produce the level of enduring environmental outcomes that was expected (see Chapter 7 by David Salt).

Perhaps that is not surprising. Back then, our understanding of community-based natural resource management (NRM), robust environmental frameworks, market-based instruments, and environmental accounting was basic at best. A quarter of a century later, these fields have developed enormously, and we now have innumerable case studies to reflect on and learn from.

Figure 23.1: A native tree planted in a farm paddock in south east Australia.

Source: Photo by Dean Ansell.

So, when the next big opportunity to conserve biodiversity on farms comes around, will we be able to show that we have learnt from our experiences in agri-environmental policy? What are the key factors a policymaker needs to consider when designing and delivering an agri-environment scheme? Each chapter in this book provides valuable lessons and insights that policymakers should keep in mind when developing agri-environment schemes. We discuss here six central themes that have emerged from the discussions contained in the previous 22 chapters.

Additionality

Agri-environment schemes arguably have two main goals: (1) to shift certain agricultural practices and behaviours towards more environmentally sustainable alternatives; and (2) in doing so to protect or enhance environmental values. Consideration of both is critical. Faced with the decision of where to invest our scarce conservation funds, the decision maker should follow the mantra of any savvy investor and ask the question: 'where can I maximise my returns while minimising the risk?'

Additionality is a term used to define the size of the effect, or the amount of benefit, resulting from an action. In the context of agri-environment schemes, we can think about additionality from two different but equally important perspectives. The first concerns farmers' adoption of on-farm activities.

Consider a farmer who adopts an environmentally desirable agricultural practice and receives payment from an agri-environment scheme as a result. If the farmer had not received a financial incentive, would she have undertaken that specific practice anyway? If the answer is no, we would say that the benefits of the scheme are additional. If the answer is yes, however, we would say that the farmer has received a windfall — a payment for something she was going to do regardless of the scheme.

The extent of additionality achieved by agri-environment schemes varies widely. The USDA Agricultural Resources Management Survey for 2009–2011 showed that additionality in conservation payments ranges between 56 and 88 per cent, depending on the type of scheme (Claassen and Duquette 2014). In other words, for some schemes, close to half of the farmers receiving payments would have undertaken the particular action without the payment. An evaluation of several different agri-environment schemes in France showed that the complexity or scale of change required in farming practice influences additionality. The additionality of more complex measures such as a shift from conventional to organic farming was typically much higher than that of more simple measures (e.g. changing crop diversity) (Chabé-Ferret and Subervie 2013).

We can also think of additionality in terms of environmental outcomes, and ask whether the scheme has led to any change in conservation value or ecological condition. In Chapter 20, Phil Gibbons stresses the importance of focusing on those conservation actions that provide the highest additional benefits, specifically the greatest biodiversity gains relative to the status quo. In doing so, he challenges the traditional focus on investing in the conservation of high-quality habitats on farm land and instead advocates emphasis on 'smaller, more modified remnants that are more vulnerable to loss' and which provide the greatest biodiversity gains as they are starting from a lower ecological condition.

A key challenge lies in identifying and measuring additionality, be it during the planning of an agri-environment scheme and selection of sites, or in retrospect during evaluation of a scheme's effectiveness. Both are important and contribute to improving the efficiency of conservation expenditure. However, as David Duncan and Paul Reich point out in Chapter 19, the consideration of additionality (through a comparison of results with and without the investment) is lacking in the evaluation of Australian agri-environment schemes. They also note that some decision makers hold the false perception that the use of counterfactuals in the evaluation of agri-environment schemes adds considerably to the cost of evaluation. They argue that simplified designs that ignore the counterfactual represent a waste of resources, as their results are unreliable. For an example of cost-effective monitoring and evaluation of agri-environment schemes, the reader is encouraged to review the work of David Lindenmayer and colleagues on the Environmental Stewardship Program (Lindenmayer et al. 2012).

Longevity

Program and project longevity is an important ingredient in designing effective agri-environment schemes. In Australia, agri-environmental schemes tend to be temporary and short-term — typically five years or less. Longevity refers to how long a particular agri-environment scheme (or program) needs to run to be successful. It refers to two different things: (1) whether a scheme is run for long enough to induce a change in landholder behaviour; and (2) whether it is long enough to achieve environmental objectives.

The first four chapters in Part 1 of this book (the agri-environment in the real world) all commented on the long-term nature of environmental action on private land.

'Achievement of these outcomes requires significant, long-term changes in land use and land management, which come at considerable financial and social cost to farmers', observes Geoff Park in Chapter 4.

In Chapter 2, Graham Fifield further supports this, noting that ongoing commitment to a site is important if the landholder is to achieve a good environmental return on the initial investment.

Emma Burns and colleagues (Chapter 3) describe the Environmental Stewardship Program as a policy innovation that delivered this long-term support, providing payments over a 15-year period, but note the challenge of operating such schemes over multiple political and accounting cycles. Indeed, the designed longevity of this program was possibly both its greatest strength and weakness (and a major reason it was discontinued).

But longevity is not just about the completion of on-ground works, numbers of hectares enrolled into a program, or even ecological benefit. It is just as much about changing the behaviour, attitudes, and values of landholders — a program needs to run long enough for this to occur. As David Pannell points out in his chapter on improving the performance of agri-environment schemes (Chapter 22), Australian programs provide only temporary support to farmers. Pannell notes that, where support is temporary, 'it is important to ensure that the actions being supported are attractive enough that farmers will continue to undertake them once funding ends. Otherwise the investment has no enduring benefit.'

When it comes to program longevity, enduring benefit is an important goal against which to judge policy proposals. And if the provision of long-term funding is not possible, then, as Pannell suggests in Chapter 22, a hard truth should be acknowledged about what should be funded: 'projects that would require significant funding in the long term to maintain the benefits generated by an initial project should not be supported.'

Long-term funding is important to creating enduring ecological and social outcomes, but it also contributes to the generation of human capital (skills and knowledge) and social capital (networks, trust and information sharing). Burns and colleagues concluded their review of the Environmental Stewardship Program in Chapter 3 with the observation that 'a valuable outcome that the Commonwealth secured through this program (in addition to the hectares being managed) was the relationships forged with the contracted land managers and developed with the CSIRO and ANU. These relationships should be nurtured to foster further learning and trust'.

Longevity is also an important characteristic influencing the desirability of a scheme to land owners. When it comes to schemes based on tenders, Graeme Doole and Louise Blackmore found in Chapter 21 that 10-year contracts seemed the most desirable. They noted that shorter contracts (five years or less) often fail to achieve desired outcomes, whereas longer contracts (15 years or more) represent a longer commitment than farmers are willing to accept.

Further, farmers may require a higher price to enrol in programs that run for longer time frames and potentially impact on their agribusiness flexibility (Ruto and Garrod 2009).

In Chapter 13, Romy Greiner also commented on how the duration of a scheme influenced the willingness of land owners to participate. She noted that, for the land owners she surveyed, 'graziers were asking for a \$0.40 increase in annual per hectare payment … for an additional year of contract duration'. (This might sound a paltry sum per hectare but keep in mind the properties she surveyed ranged in size from 2,500–10,000 km^2.) This underlines a tension between land owners wanting to participate but not wanting to commit to anything for too long.

Given the short time frames of most programs, the role of environmental non-government organisations (eNGOs) as brokers is critical. As David Freudenberger states in Chapter 5:

> The advantage of engaging a broker is the ability to build lasting relationships to help navigate the complexities and risks of entering and persisting in any market. Many eNGOs have persisted through decades of agri-environment schemes that often don't last for more than one election cycle. Continuity and organisational identity is a strength of many eNGOs.

Policy mechanisms for changing behaviour

The primary aim of an agri-environment scheme is to get landholders to adopt farming practices that deliver improved environmental outcomes. Over the years, a range of mechanisms have been used to try to achieve behavioural change. There have been payments, government regulation (to prevent damaging farming practices), tax advantages, extension (technology transfer, education, communication, demonstrations, support for community network), and development of improved land management options, such as through strategic R&D, participatory R&D with landholders, and provision of infrastructure to support a new management practice. Academic research and on-ground experience shows that the success of these mechanisms in changing behaviour is varied (Pannell et al. 2006). Several chapters in this book point to some of the reasons why.

First, the incentive for participating in a scheme is not always predominately financial. For example, the results of surveys of landholders presented in chapters 12, 13, and 21 reveal:

> [Commercial] farmers rated environmental factors as most frequently influencing their adoption of native vegetation management practices (Chapter 12).

> [F]armers in northern Australia have a high intrinsic stewardship motivation for safeguarding their cattle, land, and biodiversity assets, and that this is fundamentally linked to the pursuit of pastoralism as a chosen lifestyle (Chapter 13).

> Landholders with an altruistic attitude and strong conservation focus, with a relatively low focus on monetary outcomes, are more likely to participate in future [conservation tender] programs (Chapter 21).

It is clear that at least some landholders adopt pro-conservation practices voluntarily, without requiring payments. For example, Saan Ecker in Chapter 12 described a survey of landholder motivations to participate in the Environmental Stewardship Program. She noted: 'Most respondents "strongly agreed" that conservation and enhancement of native vegetation contributed to improved property or landscape health, aesthetics, soil stabilisation, and controlling rising water tables.'

Another example is Chapter 14, in which Maksym Polyakov and David Pannell estimate the extent to which private benefits from native vegetation on farms are built into the price of land, and how those price premiums vary in different circumstances. One potential problem occurring when there are private benefits from conservation is that these benefits are not additional (as discussed earlier).

Another is the problem of 'crowding out', where government funding for a practice reduces the level of unfunded voluntary adoption of that practice by people who are not supported by the program. This can occur if landholders feel that it is unfair for them to receive no recognition for their voluntary efforts while other landholders are receiving payments for the same actions. Graeme Doole and Louise Blackmore in Chapter 21 note that 'tender programs must employ options to counteract crowding out if they are to achieve additional environmental outcomes'. It's not obvious what these other options may be — aside from not to provide incentive payments at all — and is therefore an issue worthy of further research.

We have also seen that flexibility in delivery mechanisms is important. For example, the features of a land management contract may encourage or inhibit landholder participation if certain conditions aren't available. As Greiner states in Chapter 13: 'in general, farmers prefer higher payments, shorter contracts, more flexibility, less accountability and less paperwork.' This is a point supported by Doole and Blackmore in Chapter 21. The importance of each of these features is likely to vary depending on the location, farming system, and characteristics of the landholder. We don't suggest that policymakers pander to these desires — there are public benefits from opposite contract features — rather that they weigh up the public benefits and private costs in delivery mechanisms.

The message here is that agri-environment scheme designers should carefully consider the range of policy mechanisms they use, as some will be more suitable for some groups of farmers than others. Several evaluations of the effectiveness of schemes, both in Australia (e.g. Michael et al. 2014) and around the world (e.g. Gabriel et al. 2010), have found that a one-size-fits-all approach often fails to deliver the best biodiversity outcomes.

Prioritisation

We need to prioritise because there is never enough money available to fund all the available projects. To maximise the environmental benefits delivered by the budget of a program, governments should seek to deliver the best possible value for money. This is done by comparing the benefits and costs of proposed projects and funding those that provide the best return on investment — that is, the highest ratio of benefits to costs (Joseph et al. 2009).

In his chapter on improving the performance of agri-environment programs (Chapter 22), David Pannell provides a checklist of the key aspects of (cost-effective) prioritisation including a focus on projects (actions); ranking according to value for money; using counterfactuals to calculate benefits (as the difference in outcomes with versus without the investment); incorporating all the benefits and risks; and using a sound metric to rank investments. These elements were highlighted separately in several chapters.

Central to the prioritisation process is the explicit consideration of the costs of each project. Failing to acknowledge cost, or failing to appropriately compare costs between projects, has been a major weakness of project prioritisation in the past (Pannell 2013) and is poorly done across environmental evaluation in general (Wortley et al. 2013; Armsworth 2014). This is one of the key reasons that Pannell recommends that prioritisation should be applied to projects or actions, not to different regions, problems, and issues. Only by defining projects is it possible to meaningfully estimate investment costs.

Projects should be ranked according to value for money — a measure of their benefit divided by their cost. In Chapter 15, Dean Ansell points out that the application of this simple principle could result in significant improvements in efficiency in conservation expenditure. He also notes that there is a variety of simple economic tools available to perform such evaluations that remain relatively under-used.

Decision makers should make sure all the benefits and risks are being incorporated. If the level of adoption or likelihood of success is not factored in when projects are being ranked, inferior projects may be selected. Saan Ecker (Chapter 12) and Romy Greiner (Chapter 13) both discuss the importance of understanding the willingness of land managers to participate in agri-environment schemes as being central to the success of the projects included in the schemes.

As Fiona Gibson and David Pannell explain in Chapter 17, the way the metric used to rank projects is calculated and the choice of variables included are important. Errors here can lead to significant losses of environmental benefits. Interestingly, they also show that investing in the collection of accurate information for ranking projects may not be as critical as is often assumed. In many cases, improving the quality of the metric used to rank projects makes a larger impact on the overall level of benefits generated by a program.

Figure 23.2: A failed effort at native revegetation on a farm in NSW.

Source: Photo by David Salt.

Managing risk and uncertainty

As with many types of investment, agri-environment schemes carry significant risks. Chief among these is the risk of failure — primarily the failure to achieve the intended conservation outcomes, which in essence translates to the failure of the scheme. While there are many examples of successful agri-environment schemes, there are many that have failed to achieve their objectives or even led to negative consequences. For example, an evaluation of agri-environment schemes in Victoria found little evidence for benefits to the conservation of reptiles and amphibians (Michael et al. 2014), while a large scheme in Ireland led to an increase in agricultural pests, at the same time failing to achieve its goal of increasing the abundance of the threatened Irish hare (Reid et al. 2007). In Italy, declines in the population of the corn crake, a threatened grassland bird found in farmland, coincided with the introduction of government subsidies for grassland conservation management (Brambilla and Pedrini 2013).

Outcomes such as these may be partly or fully attributed to poor planning and implementation, but are often the result of unexpected ecological response to management. The process of ecological restoration, a primary aim of many agri-environment schemes, is complex and remains poorly understood, particularly in agricultural landscapes where the legacies of past land use and current management and climatic factors create much uncertainty in the response of biodiversity to conservation. This uncertainty not only has the potential to impact on the environmental values delivered from agri-environment schemes, but, as Sayed Iftekhar and colleagues remind us in Chapter 10, also impacts on the adoption of scheme practices. Repeated failures run the risk of alienating farmers and undermining their participation in future schemes.

This underscores the importance of a number of key factors in the design and implementation of agri-environment schemes. In particular, it highlights that identifying specific objectives for conservation is critical in defining and demonstrating success, yet the omission of such objectives is a perennial issue (Hobbs 2007). As Geoff Park outlines in Chapter 4, the use of SMART (Specific, Measurable, Achievable, Relevant, and Time-Bound) targets is crucial in the design of the scheme. Not only do SMART targets play a key role in establishing project budgets and time frames, but they also provide a centrepiece for negotiations with farmers around the aims, feasibility, and risks of proposed interventions.

Several chapters in this book contain ideas and strategies for managing risk in agri-environment schemes. The benefits of starting small as a risk mitigation strategy is highlighted by several authors. We learnt in Chapter 2 that Greening Australia's successful Whole of Paddock Rehabilitation (WOPR) scheme started with a single pilot site, which served not only as a way to assess the feasibility of the approach, but also as a demonstration to farmers interested in the program. As the saying goes, the proof is in the pudding. The Environmental Stewardship Program scheme also started small, focusing on a single target ecosystem and using the outcomes of that initial stage to broaden the coverage of the scheme as it evolved (see Chapter 3).

As David Freudenberger points out in Chapter 5, the level of acceptable risk differs between types of organisations. Non-government organisations, being largely free of the political constraints of government agencies, typically display a higher willingness to fail, and can therefore play a key role in innovation and trialling new approaches.

It is worth remembering, however, that many risks associated with agri-environment schemes, such as the uncertainty in ecological response, cannot be entirely removed. The efforts of the decision maker will be better spent factoring risk into the planning and prioritisation of agri-environment schemes, with various tools available to assist (see the simple metric provided by Fiona Gibson and David Pannell in Chapter 17, which includes the probability of success). The use of an adaptive management framework to identify, respond to, and learn from this uncertainty and unpredictability is strongly advocated by researchers (Lindenmayer et al 2008; Sayer et al. 2013). It should be noted that such an approach brings additional challenges (e.g. funding, expertise), albeit surmountable, for the policymaker.

Above all, understanding, acknowledging, and communicating these risks, particularly the risk of failure, was identified by many of our contributing authors as a critical factor in agri-environmental policy.

Capacity

In Chapter 22, David Pannell provides a list of 22 elements of good agri-environment scheme design. There was a 23rd element put forward by Pannell: 'success requires recognition that there is a body of expertise that needs to be mastered … Agencies with responsibility for agri-environment programs should foster the development of this expertise amongst their staff.' Which leads us to our final theme — capacity. Capacity is not just about the skills and knowledge contained in the organisations running these schemes; it also relates to the human and social capital found in the regions where agri-environment schemes are being implemented (Curtis and Lefroy 2010).

This book makes it clear that designing, implementing, and managing robust and effective agri-environmental programs requires a range of knowledge and technical skills. For agri-environment schemes to be effective, these skills and knowledge need to be available to policymakers, NRM managers, and the landholders participating in the schemes.

There are a number of ways to develop expertise amongst agency staff. The three approaches recommended by Cook et al. (2013) are: scientists embedded within agencies (internships), formal links between researchers and decision makers, and staff training. Formal links between researchers and scientists are facilitated through various government programs, such as cooperative research centres, Australian Research Council Linkage Projects and programs such as the National Environmental Research Program run by the Australian Department of the Environment. However, as pointed out by Emma Burns and colleagues in Chapter 3, the issue of dealing with scientific knowledge and its application in agri-environment policy within a government department is challenging and will require cultural reform for a more effective integration in future. Attwood and Burns (2012) discuss the disjunct between the spheres of science and NRM policy, suggesting it is systematic in nature. They recommend that scientists need to spend more time understanding the policymakers' bureaucratic and hierarchical system, while the public service structure needs to better reward scientific literacy.

Returning to the issue of landholder capacity, Graham Fifield and David Freudenberger both pointed out in their chapters (chapters 2 and 5 respectively) that landholders and agencies working in the agri-environment need somewhere to turn when things go wrong. Often they seek advice from trusted sources — other landholders, locals, and environmental NGOs they have worked with over time. In recent decades, there have been cutbacks to the level of extension services offered by government (Pannell et al. 2006), and staffing levels of many NRM organisations (Curtis et al. 2014), all of which erodes the capacity of agencies and communities to participate in agri-environment schemes.

In his brief history of agri-environment programs (Chapter 7), David Salt noted that earlier investments in agri-environment programs focused more on building social capital (networks and community groups) and human capital (knowledge and awareness) than targeting specific environmental outcomes. Over time, we have improved our knowledge of what is required to develop programs that will generate these outcomes. It is clear that enhancement of landholder capacity remains an important element of programs, although it is not the only element. There are likely to be benefits from targeting efforts to build capacity to situations where it can make the greatest difference to environmental outcomes.

Box 23.1: The question of value.

The question of value arises throughout this book. It also underpinned much of the discussion at the workshop that gave rise to the book. Whose values are we talking about? Which values do we mean? How do we ensure value for money? Will there ever be enough political pressure for society to adequately protect the multiple values provided by our agricultural landscapes?

During workshop discussions, Rob Fraser, an economist based in the United Kingdom, pointed out that the broader UK society placed a high value on the country's agricultural landscapes. They wanted this landscape to be available for the public to access for recreation, but they also wanted it to be there because it was part of their shared cultural history — even if they never visit it. This led him to raise the issue of different types of value: use values and non-use values.

The use values of an agricultural landscape are the benefits it generates through people making direct use of it, such as for agricultural production (e.g. cropping and livestock activities), or recreation.

Non-use values arise when an agricultural landscape generates benefits even without people making direct use of it. Examples include existence value (the benefit of knowing that the landscape still exists in good environmental condition) or option value (the benefit of retaining the landscape in a condition that does not rule out various options for its future use).

In the UK, much of the agricultural landscape provides a combination of these use and non-use values, with the social-use value of recreation particularly recognised by policymakers. This feature is set to continue into the future, with recent agri-environmental policy changes identifying the need to target areas of land for the provision of recreation values near major urban sites (European Commission 2013).

In Australia, Rob suggested the balance of social values in relation to the agricultural landscape is more towards the non-use value of nature conservation, and less towards the use value of recreation of the UK. It seems likely that non-use values would be considered by many people to be less significant than use values, reducing the prospect of major increases in public funding in the Australian context.

Figure 23.3: Rob Fraser (on the right) at the agri-environment scheme workshop discussing how the UK society values the country's agricultural landscapes. Australia's agricultural landscapes can be glimpsed in the windows in the background.
Source: Photo by David Salt.

Making 'the next big thing' a success story?

Despite the many challenges and criticisms of agri-environment schemes, the fact remains that they represent one of the strongest tools available in the quest to conserve biodiversity in farming landscapes. In our opening chapter, we discussed two contrasting schemes: one focused on restoration (WOPR, see Chapter 2), and the other on conservation (the Environmental Stewardship Program, see Chapter 3), and asked a set of questions about which was better and where the community is most likely to get value for money? The answers to these questions, of course, are 'it depends'.

We would now qualify this context-dependent answer by stating that we believe that the key criteria for successful agri-environmental policy making revolve around our six central themes of additionality, longevity, the application of appropriate policy mechanisms, robust prioritisation, effective risk management, and sufficient levels of capacity.

There are no simple black-and-white answers in addressing these themes, but it is important that the policy designer, implementer, and manager can, at the very least, frame more specific questions against each of them. Our aim in this book has been to help with that framing (and we would emphasise the more detailed list of questions posed by David Pannell in Chapter 22 — see Box 22.1).

If the public mood or political pendulum were to suddenly give rise to a large amount of money being put up for an agri-environment program across Australia, would we as a nation be ready to make the most of it? It is our opinion that we have both the experience and expertise on hand to improve substantially upon past performance.

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Paper IV. Softening the agricultural matrix: a novel agri-environment scheme that balances habitat restoration and livestock grazing.

In Paper IV, I apply many of the principles and techniques identified in the preceding papers in an evaluation of the cost-effectiveness of a novel agri-environment scheme in mixed-use agricultural landscapes in south-eastern Australia. The potential biodiversity benefits, costs and the cost-effectiveness of the scheme is compared with alternative habitat restoration strategies, and the success factors behind its implementation discussed in the broader context of achieving conservation outcomes in farm land.

Photo: G. Fifield

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TECHNICAL ARTICLE

Softening the agricultural matrix: a novel agri-environment scheme that balances habitat restoration and livestock grazing

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The loss and degradation of woody vegetation in the agricultural matrix represents a key threat to biodiversity. Strategies for habitat restoration in these landscapes should maximize the biodiversity benefit for each dollar spent in order to achieve the greatest conservation outcomes with scarce funding. To be effective at scale, such strategies also need to account for the opportunity cost of restoration to the farmer. Here, we critique the Whole-of-Paddock Rehabilitation program, a novel agri-environment scheme which seeks to provide a cost-effective strategy for balancing habitat restoration and livestock grazing. The scheme involves the revegetation of large (minimum 10 ha) fields, designed to maximize biodiversity benefits and minimize costs while allowing for continued agricultural production. The objectives and design of the scheme are outlined, biodiversity and production benefits are discussed, and we contrast its cost-effectiveness with alternative habitat restoration strategies. Our analysis indicates that this scheme achieves greater restoration outcomes at approximately half the cost of windbreak-style plantings, the prevailing planting configuration in southeastern Australia, largely due to a focus on larger fields, and the avoidance of fencing costs through the use of existing farm configuration and infrastructure. This emphasis on cost-effectiveness, the offsetting of opportunity costs through incentive payments, and the use of a planting design that seeks to maximize biodiversity benefits while achieving production benefits to the farmer, has the potential to achieve conservation in productive parts of the agricultural landscape that have traditionally been "off limits" to conservation.

Key words: agricultural landscapes, cost-effective conservation, ecological restoration, farmland biodiversity

Implications for Practice

- The use of existing farm infrastructure and configuration, efficient restoration technologies, and a focus on large fields can achieve woody vegetation restoration within the agricultural matrix at lower cost than prevailing approaches.
- An emphasis on co-benefits of restoration to the farmer and offsetting of opportunity costs through incentive payments, coupled with minimal disruption to farming systems, creates a restoration approach with the potential to achieve large-scale adoption in grazing-dominated landscapes.
- Careful consideration of economic costs, including private opportunity costs, in restoration scheme design and implementation can greatly increase the conservation outcomes per dollar spent.

Introduction

Conserving biodiversity in agricultural landscapes in the face of growing agricultural production represents a key challenge at a global scale (Green et al. 2005). Conservation efforts typically focus on the protection and restoration of remnant habitats, which provide important refuges in an environment that is otherwise largely uninhabitable for many species (Fischer & Lindenmayer 2002*a*). Recent research, however, has highlighted the importance of the agricultural matrix itself in the ecology and conservation of species in fragmented landscapes (e.g. Driscoll et al. 2013).

Within the agricultural matrix, woody vegetation represents important habitat, with much of the remaining cover restricted to narrow, linear features such as fence boundaries, riparian strips, roadside remnants, and scattered trees (Manning et al. 2006; Welsch et al. 2014). These habitat features serve to "soften" the agricultural matrix (Franklin 1993) and facilitate important ecological functions at a variety of scales (Fischer & Lindenmayer 2002*a*).

However, continued clearing of remnant woody vegetation (Hansen et al. 2013) and the loss of scattered trees (Gibbons et al. 2008) will lead to an increasingly homogeneous and impermeable agricultural matrix suitable for a much narrower spectrum of species (Duncan & Dorrough 2009), prompting

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increasing calls for restoration of habitat in the agricultural matrix as part of a comprehensive landscape-scale response to biodiversity conservation (Vandermeer & Perfecto 2007; Fischer et al. 2010).

Restoring Habitat in the Matrix

Conservation efforts in the agricultural matrix need to account for opportunity cost; that is the loss of revenue from agricultural production that results from an alternative use of that land (i.e. conservation; House et al. 2008). Agri-environment schemes (AES), which broadly involve the payment of incentives to farmers in return for the provision of ecological goods (Burrell 2012), provide such a mechanism. Billions of dollars are spent annually on AES around the world (European Commission 2014; USDA 2014). These schemes typically focus on the maintenance of traditional farming practices (e.g. organic farming) and grassland-focused restoration (e.g. sown wildflower strips). There have been relatively few attempts to restore native woody vegetation at scale in the agricultural matrix (e.g. Benayas et al. 2008; Zahawi et al. 2013). These approaches typically establish small or linear habitat patches, focus on expansion of existing woodlands, or target abandoned agricultural land. There is a clear need for large-scale, cost-effective strategies to restore woody vegetation in the agricultural matrix using measures that integrate production and biodiversity conservation.

The Whole-of-Paddock Rehabilitation Scheme

Up to 95% of some woodland communities have been cleared across Australia's southeast, with remaining patches of vegetation typically small and isolated or restricted to reserves (Yates & Hobbs 1997). Opportunities for biodiversity conservation on private land are therefore limited. Whole-of-Paddock Rehabilitation (WOPR) is a voluntary, self-nominating AES developed as a strategy for achieving large-scale, cost-effective landscape restoration in agricultural landscapes through the integration of production and biodiversity benefits.

WOPR was established in 2008 by Greening Australia (GA), a national environmental nonprofit organization, with the primary objective of integrating biodiversity conservation and agricultural production through the generation of multiple environmental and production benefits (Greening Australia 2014). The scheme targets livestock grazing or mixed enterprise farms and involves active restoration, using direct seeding, to restore native woody plants across large (>10 ha) fields through a 10-year management agreement that provides farmers with a fixed, area-based incentive payment to offset production losses (Streatfield et al. 2010) (see Box 1 for overview). Existing fields in active production areas are targeted, focusing on sites with the greatest potential for environmental benefits and issues that impact on farm productivity (e.g. exposed, eroding, saline, or weed-infested fields).

Enrolment in WOPR has increased from 10 agreements (covering a total of 198 ha) in 2008 to 80 agreements in 2014 which now cover a total of 2,012 ha (Table 1). The average size of enrolled fields has doubled from 19.8 to 38.6 ha (Table 1). Two fields greater than 90 ha have been enrolled, and the scheme now has farms enrolled across approximately 24,000 km^2. The enrolled fields are within southeastern New South Wales where the dominant agricultural land use is sheep and cattle grazing on modified pastures, with increasing dryland crop production to the west.

Here, we profile and critique this novel AES. We aim to explore the strengths and weaknesses of the design and implementation of the scheme with a view to identifying factors that may contribute to the development of similar programs elsewhere.

Box 1: Typical WOPR Establishment Process and Timeline

1. Contract developed between farmer and GA outlining project design, management obligations, and payment schedule.
2. Field prepared by farmer, including herbicide application, supplementary fencing, and pest control.
3. Field sown by GA with 10–20 native tree and shrub species in 15 m wide bands consisting of four lines of direct seeding 5 m apart, with each band 40 m apart.
4. Farmer excludes livestock for 5 years in return for \$50/ha year stewardship payment (50% in year 1 and 50% in year 5).
5. Livestock reintroduced under rotational grazing for the remaining 5 years.

Methods

We focused our critique on the potential biodiversity benefits of WOPR as well as other environmental and production benefits. The relative infancy of the program and limited number of established fields prevents a detailed evaluation of on-ground outcomes. Therefore, we draw here on the restoration literature to consider potential outcomes, focusing where possible on studies of revegetation in similar landscapes across the region.

We also evaluated the scheme's cost-effectiveness by comparing the costs of establishing a typical WOPR field with two

Table 1. Enrolment statistics for the WOPR scheme for the period 2008–2014.

Year of Establishment	*2008*	*2009*	*2010*	*2011*	*2012*	*2013*	*2014*	*Overall*
No. of fields enrolled	10	5	3	10	23	17	12	80
Total area enrolled (ha)	198	154	58	299	382	454	467	2,012
Cumulative total area (ha)	198	352	410	709	1,091	1,545	2,012	2,012
Average field size (ha)	19.8	30.8	19.3	29.9	16.6	26.7	38.9	25.2

Table 2. Comparison of the characteristics, cost, and cost-effectiveness of three alternative scenarios for the restoration of woody vegetation in a hypothetical 20 ha field (see Methods). Costs provided in AUD.

	WOPR	*Block Planting*	*Windbreak Planting*
Field characteristics			
Vegetation cover (%)	27	100	13
Vegetation cover (ha)	5.4	20	2.6
No. of stems/ha[a]	7,200	20,000	3,600
Private opportunity cost			
Loss of agricultural revenue[b]	$13,654	$24,578	$3,195
Public costs			
Site design ($50/ha), overheads ($200/ha), and contingency ($100/ha)	$7,000	$7,000	$7,000
Direct-seeding ($200/km)	$2,880	$8,000	$1,440
Fencing ($3,000/km)[c]	$0	$0	$5,040
Stewardship payment ($50/ha/year)	$4,051	$0	$0
Total public cost	$13,931	$15,000	$13,480
Total public cost per hectare of vegetation	$2,580	$750	$5,185
Total public cost per hectare of land	$696	$750	$674
Total public cost per stem	$1.93	$0.75	$3.74

[a]Based on 0.5 stems/m. [b]After Stewardship payment. [c]Assumes existing perimeter fencing used for WOPR and block scenarios; 50% of existing perimeter fencing used for windbreak scenario.

alternative scenarios selected to represent the spectrum of active restoration approaches in agricultural landscapes, including (1) a "windbreak" planting involving revegetation of a field perimeter with 15 m wide bands, arguably the most prevalent planting configuration and (2) a "block planting" involving revegetation and livestock exclusion of an entire field, a much less common planting design but beneficial for biodiversity, given minimization of edge effects (Helzer & Jelinski 1999).

We calculated the total public cost of each scenario in a restoring field of 20 ha (0.4×0.5 km). Costs were calculated over a 10-year time frame with a 10% discount rate, using 2015 prices ($AUD) provided by GA. We also estimated private opportunity costs for each scenario based on typical gross margins ($200/ha/year) for a sheep grazing enterprise on native pasture in the study region (NSW Department of Primary Industries 2014). We assumed a reduction in gross margin equal to the area of land under revegetation (i.e. approximate vegetation cover; Table 2). In other words, for the windbreak and block planting scenarios, the reduction in gross margin was 13 and 100%, respectively, whereas for the WOPR scenario, the reduction was 100% for 1–5 years (reflecting the livestock exclusion period) and 27% for the remaining 5 years.

Ideally, such an evaluation would use realized biodiversity benefits as the measure of effectiveness. In the absence of these data, we used two measures of native vegetation structure as a measure of the scheme effectiveness: native vegetation cover (% cover/ha) and number of stems per hectare. As the establishment of habitat for biodiversity is a key objective for WOPR, this provides an adequate measure of benefit for comparative purposes. Lastly, we consider the transferability of the scheme to other agricultural landscapes and explore the potential for its long-term viability.

Results

Biodiversity Benefits

WOPR aims to maximize biodiversity value by integrating a number of design aspects known to influence biodiversity response to restoration. Firstly, by establishing a minimum patch size of 10 ha, the scheme aims to avoid the proliferation of small plantings (e.g. 1–5 ha), which can support fewer species and abundances (Munro et al. 2011) and often dominate restoration on farms (Smith 2008). The design aligns with minimum patch size recommendations for woodland bird conservation in southeastern Australia (Freudenberger 1999), and the amount of vegetation cover across the WOPR field (approximately 30%) is consistent with recommended landscape-scale cover for sustainable land management in Australian temperate woodlands (McIntyre et al. 2000).

Secondly, the design seeks to avoid edge effects associated with narrow linear habitat patches (Helzer & Jelinski 1999), with the gaps between direct-seeded vegetation (40 m) sufficient to allow future agricultural production but less than movement thresholds identified for some woodland fauna (Robertson & Radford 2009; Doerr et al. 2011). Lastly, the sowing of several tree and shrub species in WOPR fields increases habitat complexity and provides habitat structures that are often missing from grazed woodland remnants in farmland but are important for many species (Kavanagh et al. 2007; Munro et al. 2011). In addition to planting design, of similar importance is the location in the landscape, such as proximity to remnant vegetation or other restoration sites (Lindenmayer et al. 2010). Currently, there are no criteria in the WOPR program relating to landscape context that guide site selection. As the scheme expands, consideration of landscape context will be an important factor in prioritizing sites.

Although WOPR fields provide habitat features often missing from agricultural landscapes within a much shorter time frame than natural regeneration often permits (Dorrough & Moxham 2005), the long-term biodiversity benefits are difficult to forecast. Direct-seeded habitat patches in the same region became more simplified over time as stem density and species richness decreased (Schneemann & McElhinny 2012). Similarly, the impact of resumed grazing in WOPR fields on biodiversity values is unknown at this stage. Rotational grazing, required under the WOPR agreement, can improve natural tree regeneration (Fischer et al. 2009) and the habitat value of direct-seeded vegetation in the long term compared to traditional grazing regimes (Sherren et al. 2011). Management choices beyond the WOPR agreement, however, rest with the farmer and will be influenced by factors beyond that of environmental gains. Maximizing biodiversity value in the long term is likely to be dependent on the continued use of rotational grazing, potentially coupled with activities aimed at maintaining vegetation structural complexity and diversity (e.g. use of native pasture species, soil disturbance, and prescribed-burning) (Schneemann & McElhinny 2012).

Agricultural Production and Other Benefits

The spatial configuration of vegetation planted within a WOPR field allows for the use of the space between vegetation bands for continued livestock grazing. This focus on co-benefits is the key to facilitating ecological restoration measures in productive parts of the agricultural landscape that have traditionally been "off limits" to conservation.

Anticipated production benefits to the farmer of enrolment in WOPR include the provision of shelter, shade, and additional forage, with potential improvements in the health, survival, and productivity of livestock and pasture (Lynch & Donnelly 1980; Monjardino et al. 2010; Greening Australia 2014). The approximate vegetation cover in an established WOPR field is consistent with identified vegetation cover thresholds for maximal pasture output (Walpole 1999). Additional benefits provided by the use of deep-rooted perennial plants include improved soil stability and erosion control as well as salinity mitigation (Bird et al. 1992; Schofield 1992; Lovell & Sullivan 2006). WOPR fields could also provide economic and environmental benefits through enrolment in carbon accreditation schemes (Department of Agriculture 2013).

Cost-Effectiveness

Comparison of the costs of restoration reveals the greater cost-effectiveness of WOPR relative to the prevailing windbreak planting configuration, with WOPR yielding more than twice the amount of vegetation for the same cost (Table 2). A key factor in this is WOPR's use of existing fields within the farm layout, thus avoiding additional fencing costs, a major cost component of restoration on farm land (Freudenberger et al. 2004; Table 2).

This comparison also shows the superior cost-effectiveness of the block planting scenario, which achieves restoration at costs approximately 71 and 85% less per vegetation hectare than WOPR and the windbreak planting, respectively. This is partly due to the absence of a stewardship payment under this hypothetical scenario. Our estimation of private opportunity costs (Table 2) demonstrates the substantial increase in cost to the farmer associated with this design, which would in turn necessitate an increase in public cost in the form of stewardship payments to offset the private costs. The efficiency of this configuration would be further reduced by the addition of fencing costs, which are assumed to be nonexistent in this scenario but which, unlike WOPR, would likely be required in reality. These factors coupled with the incompatibility of such an approach with future agricultural production make it non-viable for large-scale application. In contrast, the low opportunity costs of the windbreak configuration explain its prevalence in farming landscapes. However, this configuration is the least cost-effective in terms of public expenditure and likely to have lower biodiversity benefit as outlined above. This underscores the critical influence of private opportunity cost on the total public cost and potential benefits of restoration, and the necessity of offsetting these costs through incentive-based schemes and/or the incorporation of private benefits into scheme design.

Opportunities exist to further improve the cost-effectiveness of the scheme. Preliminary monitoring reveals an average eucalypt density (667 stems/ha; Greening Australia unpublished data), which greatly exceeds that found in natural Yellow box *Eucalyptus melliodora* woodlands (212–343 stems/ha; Gibbons et al. 2010). Although the higher density provides insurance against unpredictable losses due to grazing or adverse environmental conditions, the cost of *Eucalyptus* seed represents approximately 40% of the cost of direct seeding and could potentially be reduced without compromising biodiversity benefits.

Scheme Transferability and Longevity

To date, WOPR has largely targeted livestock grazing and mixed enterprise systems. Although such systems currently remain dominant in Australia, the extent of cropland is increasing (Mewett et al. 2013) and represents the most common agricultural land use type at a global scale (Wood et al. 2000). Though there are predicted production benefits of WOPR in grazing-dominated systems, the potential benefits to crop production have not been established. Revegetation using deep-rooted perennial species can provide benefits for crop production through salinity mitigation (Knight et al. 2002) and encourage important crop pollinating insects (Arthur et al. 2010). However, farmer concerns over potential competition between crops and planted vegetation for water presents a possible barrier (Woodall & Ward 2002). Landholders who focus on cropping are less likely to restore tree cover than those based on grazing (Schirmer et al. 2012). Cropping systems typically maximize the use of available land, and thus opportunities for biodiversity restoration are limited to marginal and highly fragmented areas of the landscape such as field edges, corners, and gullies (G. Fifield 2014, personal observation). Although more complex agri-environment delivery mechanisms (e.g. auctions and results-based payments) could be used to increase uptake in other agricultural systems, their inherent complexity increases transaction costs (Klimek et al. 2008) and places greater demands on already scarce funding. Current efforts to establish a modified WOPR design in a cropping landscape in Western Australia will provide a valuable test of the scheme's transferability.

Long-term funding stability is a priority issue in biodiversity conservation in agricultural landscapes (Lindenmayer et al. 2013). AES by their nature are not financially self-sustaining, requiring long-term funding to achieve restoration across large scales. The low rates of agricultural subsidy in Australia mean that additional funds must be found for conservation (Hajkowicz 2009). The WOPR scheme has been funded primarily through government, however, funding is secure for at most 3–5 years as a result of government expenditure cycles. Diversifying the program's funding sources to spread risk will be equally as important as looking to secure funds from existing government sources in the future.

Discussion

The WOPR scheme provides an example of a simple, innovative, and cost-effective solution to large-scale habitat restoration integrated with agricultural production. It uses an incentive-based mechanism to target biodiversity outcomes and production benefits and fills an important gap in the conservation of biodiversity within productive areas of agricultural landscapes. The scheme has been designed to balance factors known to be important in determining biodiversity benefits, such as size, shape, and habitat complexity of restoration sites, while seeking to minimize costs by using existing farm infrastructure, focusing on large fields, using efficient revegetation technologies, and permitting a rapid return to agricultural production.

Innovative approaches to ecological restoration in agricultural landscapes such as WOPR should be seen as complementary, providing additional tools in the restoration practitioner's toolbox, rather than replacing traditional measures (Benayas et al. 2008). The conservation of remnant habitat patches remains a key biodiversity conservation priority in farming landscapes (Cunningham et al. 2008). Likewise, windbreak plantings can provide important ecological functions such as improved connectivity and to some farmers may provide the only acceptable planting configuration and therefore will continue to be important habitat restoration strategies.

Despite the success of the WOPR scheme to date, there are several issues that present barriers to continued growth. These include continuity of funding, limited transferability to other agricultural systems, and uncertainty regarding the long-term biodiversity benefits. Continued ecological, economic, and social research will be required to further demonstrate the potential of the scheme to achieve biodiversity and production benefits.

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Paper V. The cost-effectiveness of agri-environment schemes for biodiversity conservation: A quantitative review.

In the preceding papers, I highlight the potential biodiversity benefits that can be achieved through the integration of economic costs in the planning and evaluation of conservation in agricultural land. In Paper V, I reviewed the global agri-environment literature to explore the extent to which studies consider the costs of conservation, and the integration of those costs in evaluation through the use of techniques such as cost-effectiveness analysis. I also asked whether there has been an increase in the evaluation of cost-effectiveness commensurate with the growth in investment in agri-environment schemes in recent decades.

Photo: D. Salt

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Review

The cost-effectiveness of agri-environment schemes for biodiversity conservation: A quantitative review

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ABSTRACT

Agri-environment schemes (AES), where farmers receive payments in exchange for providing public goods and services such as biodiversity, account for a major proportion of conservation expenditure in agricultural landscapes around the world. The variable effectiveness of such schemes and increasing recognition of the importance of cost-effective conservation – maximizing conservation benefit for a fixed cost or minimizing cost of achieving a specific conservation outcome – has prompted calls over the past decade for integration of economic costs into evaluation. We reviewed the global agri-environmental evaluation literature to determine what proportion of studies evaluating biodiversity conservation effectiveness consider costs and cost-effectiveness and whether there has been an increase in this integration over time. Less than half of the studies reviewed made any reference to the costs of AES, and fewer than 15% included any measure of cost-effectiveness. Despite steady growth in the number of published AES evaluations over the past 15 years, and a gradual increase in the number of studies that acknowledge costs, the proportion of studies published annually that integrate economic data into evaluation remains largely unchanged. Various reasons have been identified for this poor integration, including limited understanding of, and access to, economic evaluation tools, data and training, and a philosophical aversion to the mixing of economics and conservation. We argue however that these reasons are no longer justified, and highlight several examples of the effective integration of economic and ecological data in evaluations to assist researchers and decision-makers in addressing this deficiency.

Contents

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1. Introduction

Balancing the agricultural development required to feed a growing global human population with the conservation of biodiversity is a key challenge for society (Green et al., 2005; Tilman et al., 2011). Agricultural development and intensification has been linked to biodiversity declines and other ecosystem impacts around the world (Donald et al., 2001; Stoate et al., 2009; Venter et al., 2006) and represents the largest single threat to biodiversity conservation globally (Secretariat of the Convention on Biological Diversity, 2014). Over the past three decades, governments have increasingly used incentive-based mechanisms to protect and restore biodiversity on farmland. Agri-environment schemes (AES), which broadly involve payments to farmers in exchange for environmental goods and services such as biodiversity conservation (Burrell, 2012), provide one such approach. Schemes range widely in scale, complexity and focus, from those that promote input reduction (e.g. organic farming), to land retirement and active habitat restoration, though they have the common broad objective of maintaining or improving specific environmental values such as biodiversity as well as water, soil and air quality (Barral et al., 2015; Rey Benayas and Bullock, 2012).

AES are now the focus of significant investment around the world, with agri-environmental investment in many countries often equal to, or surpassing that of other conservation expenditure (Batáry et al., 2015). In the past decade, the European Union and the US combined have spent more than USD$35 billion on AES (European Commission, 2014; USDA Farm Service Agency, 2015a). European Union member states are required under the Common Agricultural Policy (CAP) to establish AES. The CAP committed EUR95.58 billion to rural development over the next five years, the majority of which is dedicated to AES (European Commission, 2013). The United States Conservation Reserve Program (CRP), a long running land retirement initiative with an annual budget of approximately USD $2 billion (Stubbs, 2013), has more than 24 million acres (9.7 million hectares) enrolled (USDA Farm Service Agency, 2015b). In Australia, the Environmental Stewardship Program committed approximately AUD $152 million in payments to farmers for restoration and protection of priority ecosystems (Burns et al., in press). Significant schemes have also been implemented elsewhere in North America (McMaster and Davis, 2001) as well as within Latin America (Sierra and Russman, 2006), Africa (Kehinde and Samways, 2014) and Asia (Li et al., 2013).

The growth in AES investment has fueled ongoing debate over the effectiveness and efficiency of these schemes as strategies for biodiversity conservation in agricultural landscapes. While several studies have found biodiversity improvements in response to changed agricultural practices under AES programs (e.g. Knop and Kleijn, 2006; MacDonald et al., 2012), others have shown mixed or limited benefits (e.g. Feehan et al., 2005; Kleijn et al., 2004; Verhulst et al., 2007), and even negative biodiversity outcomes (e.g. Besnard and Secondi, 2014; Fuentes-Montemayor et al., 2011). Despite their mixed success, AES now represent the dominant policy instrument for conserving biodiversity in agricultural landscapes. Indeed, some have suggested AES provide the only realistic tool to address biodiversity declines in farmland (Donald and Evans, 2006). The continued political and public support for these initiatives requires increased confidence that they represent the best use of public funds. This requires consideration of cost-effectiveness, being a comparison between alternatives of the benefits per dollar spent or identification of the lowest cost alternative to achieve a specific outcome (Wätzold and Schwerdtner, 2005).

Evaluating the cost-effectiveness of AES requires an understanding of not only the ecological effectiveness of schemes, but also understanding of the economic costs (hereafter referred to generally as costs). However, there remains a lack of integration between economic and ecological perspectives and techniques across conservation science in general, with crucial economic information (e.g. program costs) often ignored in program evaluation (Naidoo et al., 2006; Wortley et al., 2013). A review of 2000 restoration studies found that none performed any analysis of cost-effectiveness, and fewer than 5% provided 'meaningful' cost data (TEEB, 2009). Kleijn and Sutherland (2003) found that none of 62 European AES evaluation studies surveyed addressed issues of cost-effectiveness. These issues have prompted repeated calls over the past 15 years for the integration of economic and ecological factors in the evaluation of AES (Balana et al., 2011; Kleijn and Sutherland, 2003; Uthes and Matzdorf, 2013; Whitby, 2000). But have these calls been answered?

This paper aims to address these questions by reviewing, at a global scale, the extent to which studies evaluating the biodiversity benefits of agri-environment Schemes 1) acknowledge economic costs, and 2) provide any measure of cost-effectiveness. While there may be other public or private benefits of AES, we consider only evaluation of biodiversity-related benefits. We consider the nature of the AES employed, the type of evaluation tools used and the agricultural context in which they are applied to investigate whether there are biases in coverage of different AES. We also explore possible reasons behind observed trends in the integration of costs in AES evaluation and identify solutions to assist evaluators and program managers to improve future evaluations. To our knowledge, this is the first global scale, quantitative review of agri-environment schemes, and one of few studies to focus on the cost-effectiveness of agri-environmental policy (Balana et al., 2011; Claassen et al., 2008; Uthes and Matzdorf, 2013). By exploring the coverage of cost-effectiveness in the evaluation literature, we hope to draw further attention to an increasingly important issue which can ultimately improve the efficiency of conservation expenditure.

2. Methods

2.1. Literature search

We performed a quantitative review of the literature published up to, and including, 2014 using ISI Web of Science and Scopus databases. We aimed to identify studies focusing on the evaluation of the effectiveness, from a biodiversity conservation perspective, of conservation activities—for example planting for habitat, organic farming and sustainable grazing (hereafter referred to as 'interventions')—delivered through AES exclusively on agricultural land. We considered as AES any voluntary scheme that involved any payments (one-off or ongoing) made to landholders by any public or private funding body for any type of intervention. We did not consider schemes implemented under regulatory mechanisms (e.g., EU Nitrate Directive) that mandate or encourage adoption of conservation measures. We only included studies where the protection or restoration of populations, species, communities or ecosystems represented at least one objective of management.

Initial review of the literature revealed geographic bias in the use of the term 'agri-environment scheme', which is used extensively in Europe but less so elsewhere, particularly in the Americas. Our search terms therefore were broad in order to capture schemes labeled under different terms. The following search terms were used: (habitat$ OR bird$ OR amphibian$ OR mammal$ OR reptile$ OR plant$ OR invertebrate$ OR threaten* OR threatened$species) AND (farm* OR agricultur*) AND (agri-environment OR ecological$restoration OR restoration OR biodiversity$conservation OR biodiversity$protection OR conserv*) AND (cost* OR cost$effective* OR effective* OR evaluat* OR outcome$ OR monitor* OR success* OR assess* OR cost$benefit OR benefit$cost). To minimize the number of non-target articles, we excluded database categories that were of no relevance to the subject (e.g. engineering, medical, health, legal, political).

This search strategy identified 16,574 references (Scopus: 9529; Web of Science: 7045; searches performed on 4 February 2015) which were initially screened by journal title, article title and topic to remove those clearly not relevant to the study. This process identified 931 references which were then further screened through review of abstracts, excluding those that were: published in languages other than English; not considered AES by our definition (see above); focused solely on economic, social or public policy aspects; concerned with schemes targeting resource-extraction (i.e. agro-forestry, mining) and urban environments; or published as book chapters, conference proceedings, or in non-peer reviewed publications. We also excluded discussion-type studies and literature reviews from analysis but cross-referenced studies cited therein. This process reduced the list from 931 to 239 references which formed the basis of our analysis.

2.2. Literature analysis

Our approach scaled the level of analysis to the relevance of the paper using a three-tiered system. Group 1 included all (239) studies that provided some evaluation of conservation effectiveness. A subset of these (Group 2) comprised studies that made any reference to the cost of interventions and/or the cost of the AES policy as a whole. This included any use of the cost-related terms and symbols identified through full text searches (e.g. expenditure, budget, cost, economic, investment, dollars, $) and did not require identification of actual expenditure. Lastly, Group 3 was a subset of Group 2 that included studies that explicitly considered cost-effectiveness. To be included within this group, studies needed to use any cost data in any form in their evaluation of the AES. We included studies that used any economic evaluation technique, regardless of complexity, and including techniques using both monetized (i.e. cost-benefit analysis) and non-monetized benefits (i.e. cost-effectiveness analysis or multi-criteria analysis). This approach allowed us to address our key research questions by identifying the proportion of studies that consider cost and cost-effectiveness across the AES evaluation literature. We then explored the types of techniques used and the context in which they were implemented.

Details of the information extracted from studies under each group are provided in Supplementary information (Table S1). In summary, for Group 1, we extracted general information such as publication details, as well as details of the study, the scheme and its objectives. We also identified the effectiveness measure/s used and whether or not costs were considered. For Group 2 we further described the cost data used, and for Group 3 we extracted information relating to the type of economic evaluation used to assess cost-effectiveness.

3. Results

3.1. General information (Group 1)

The 239 Group 1 studies were published between 1992 and 2014, with 53% published since 2010. The studies were from 67 journals, though four journals (*Agriculture, Ecosystems and Environment*,*Biological Conservation*, *Journal of Applied Ecology* and *Journal of Wildlife Management*) accounted for 41% of total publications (Supplementary information, Table S2).

Studies were conducted on AES across 25 countries covering each of the major geographic regions. The majority of the studies were concerned with AES in Europe (160 studies; 67%), of which most studies were in England (40 studies), France (19), Netherlands (16), Switzerland (15) or Germany (10). North America was the second most studied region (67 studies; 28%), of which most were undertaken in the United States. While studies were also conducted on AES within Asia, Africa, Oceania and Latin America, combined they only represent 5% of studies reviewed. Consistent with the geographic focus of the studies reviewed, more schemes were aligned to the EU Common Agricultural Policy than with any other initiative (69 studies; 29%), followed by the US CRP (50 studies; 20.9%) and Switzerland's Ecological Compensation Areas scheme (13 studies; 5%). Interestingly, 47 studies (19.7%) did not identify the particular AES on which the study was undertaken.

Cropping-dominated landscapes (104 studies; 43.5%) were represented more strongly than those dominated by

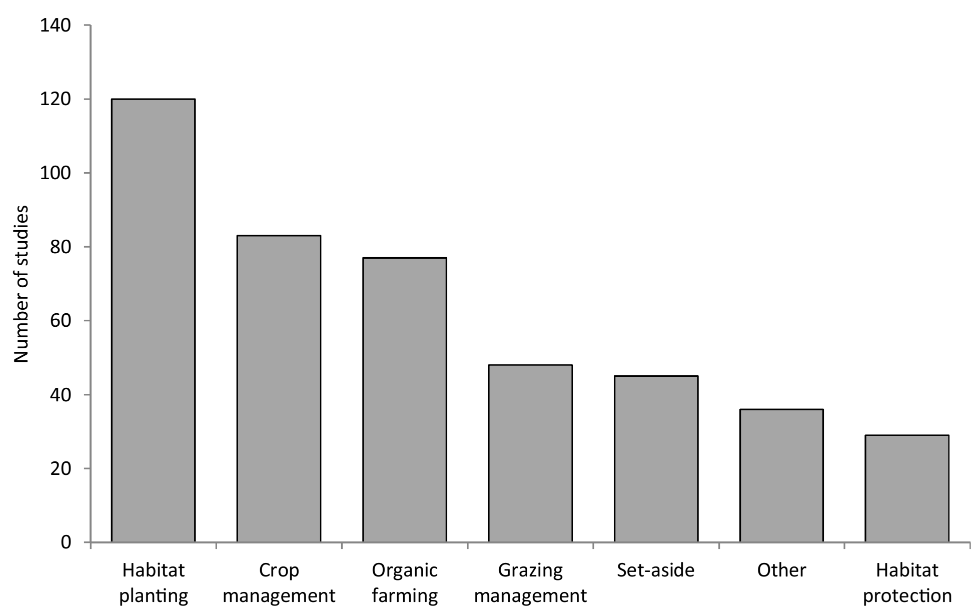

Fig. 1. Types of interventions applied in the agri-environment schemes under evaluation. Note: many studies covered schemes involving multiple interventions. See Supplementary information (Table S1) for further information on categories.

pasture-grazing systems (37 studies; 15.4%), though 29.7% of studies were conducted in mixed (grazing and cropping) landscapes. The most dominant intervention type was habitat plantings (120 studies; 50%), predominantly involving wildflower or grass buffers or strips around crop margins, followed by crop management interventions (83 studies; 35%), including measures such as retention of crop stubble (e.g. Suárez et al., 2004) and altering timing of agricultural practices (Adams et al., 2013) (Fig. 1).

The most common reported objective of the AES under evaluation was biodiversity in general (118 studies; 49%). The conservation of a single species was the focus of AES in 16 studies (7%), whereas the schemes evaluated in 39 studies (16%) were targeted at multiple species, particularly groups of similar species (e.g. waterbirds; Wilson et al., 2007). Accordingly, the objective of most evaluations (rather than the objective of the scheme itself) was the effectiveness of the AES on multiple species (190 studies; 79%), varying from whole taxonomic groups (e.g. butterflies; Aviron et al., 2011) to as few as two species (Conover et al., 2011). Forty-five studies (18%) focused on the benefits for single species. Similar numbers of studies were concerned with broader biodiversity benefits (32 studies) or habitat and/or ecosystem-related objectives (45 studies).

Birds were the most commonly studied species (123 studies; 51%), followed by plants, (101 studies; 42%) and invertebrates (62 studies; 26%). Only 13 (5%) studies were concerned with mammals, 5 (2%) focused on herpetofauna and 1 (<1%) on fish. Evaluations mostly used multiple measures of effectiveness (189 studies; 79%), combining measures such as abundance, breeding success and habitat quality (e.g. Blank et al., 2011), compared to those using only a single measure (50 studies; 21%). Direct measures of effectiveness were dominant (201 studies; 84% of total), with variables such as abundance, richness and vegetation cover most commonly used. The 34 studies that used proxies or indirect measures of effectiveness predominantly focused on spatial area (e.g., amount of land enrolled) (15 studies). Benefit indices were also used as surrogates (6 studies), predominantly in model-based evaluations. For example, Uthes et al. (2010) used an aggregate index combing multiple environmental values (biodiversity, soil, water and landscape) to compare the cost-effectiveness of alternative interventions and spatial targeting approaches.

Most studies were conducted at a single scale, with the majority focused on the 'landscape or regional' scale (187 studies; 78%), followed by 'farm' scale (48 studies; 20%) and 'field' scale (4 studies; 2%). The remaining studies were conducted across multiple scales, most commonly at the farm and landscape or regional scales (15 studies; 6%). More than 200 (87%) of the studies were undertaken during or after the implementation of the scheme(s) (*ex post*), whereas only 36 studies (15%) included an *ex ante* component, typically involving modelling of likely biodiversity outcomes (e.g. Chiron et al., 2013).

3.2. Consideration of cost (Group 2)

Of the 239 studies reviewed, only 115 (48%) made some reference to the cost associated with the AES (see Methods). These articles were spread across 50 journals, though two journals, *Agriculture, Ecosystems and Environment* (16 articles) and *Biological Conservation* (11), represented almost a quarter of studies. The average annual percentage of Group 1 studies that referred to costs (i.e. Group 2) was 46.5% ± 27% (mean ± S.D, n = 19), and did not significantly increase over time (linear regression, r = 0.05, p = 0.8, n = 19) (Fig. 2).

Fifty-six of the 115 Group 2 studies (49%) reported specific costs, of which 42 provided the total cost of the scheme in question, the remainder providing only costs of components of the scheme (e.g. incentive payment rates; Elts and Lohmus, 2012). Thirty-three studies gave actual costs, the remainder used estimated costs or did not specify. There was a strong focus on public expenditure, with 21 of the 56 studies that reported cost information providing public costs exclusively, or in combination with private costs (14 studies), both in terms of privately funded (e.g. nongovernment organizations) and costs incurred by the farmer. The remaining 21 studies did not specify the source of the cost data provided. Twenty-two of 56 studies (39%) measured the opportunity cost to the farmer of enrollment in AES. For example, Wynn (2002)

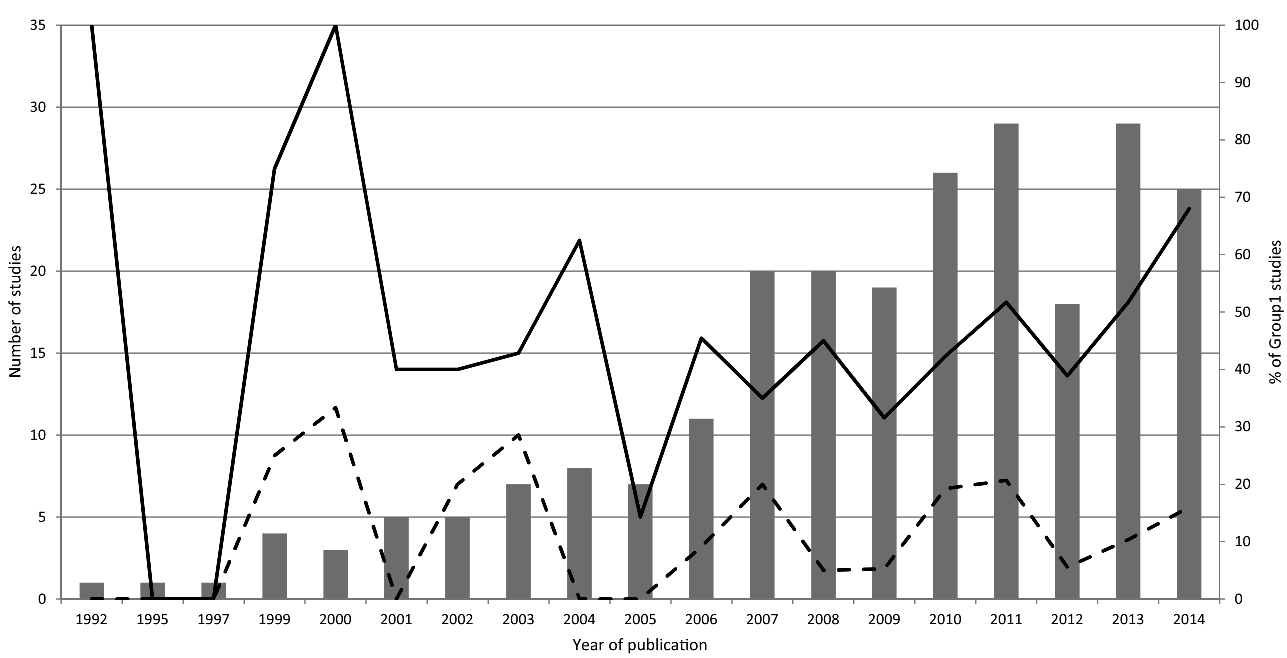

Fig. 2. 'Temporal trends in publication of agri-environment scheme evaluations. The year of publication for all 239 studies included in the review (categorized as Group 1 in text) is shown in columns. Of the Group 1 studies, the annual percentage that acknowledge economic costs (Group 2) is displayed in the solid line, whereas the percentage that include some measure of cost-effectiveness (Group 3) appears as the dashed line.

calculated per hectare opportunity costs of enrollment in the UK Environmentally Sensitive Areas scheme to the farmer through a regression of reduced gross margin with area enrolled.

3.3. Evaluation of cost-effectiveness (Group 3)

Of the 239 Group 1 studies, only 31 (13%) involved some form of evaluation of cost-effectiveness (Fig. 2; see Methods for criteria). These studies were published from 1999 to 2014, 61% of them since 2010. The studies were published in 21 journals, though more than a quarter were published in *Ecological Economics* (5 studies) and *Biological Conservation* (3 studies). The average annual proportion of Group 1 studies that integrate cost data in analysis (i.e. Group 3) was 11.5% ± 10.7% (mean ± S.D, n = 19), and did not significantly increase over time (linear regression, r = 0.18, p = 0.47, n = 19) (Fig. 2).

The majority of the Group 3 studies (24 studies; 77%) used cost-effectiveness analysis or variants thereof, where the measure of effectiveness was not monetized (e.g., species richness, area). These varied from a simple comparison of biodiversity response with estimated costs, to more sophisticated model-based approaches. Wilson et al. (2007), for example, simply compared the average cost to produce one additional breeding pair of waders between three different subsidy levels under the UK's Environmentally Sensitive Areas scheme. In contrast, Barraquand and Martinet (2011) used a dynamic ecological-economic model to test the cost-effectiveness of a grassland conservation subsidy, comparing it to a compliance-based (i.e. taxation) measure, revealing the complex relationship between costs and benefits and highlighting the importance of accounting for spatio-temporal variability in evaluation.

Five studies (16%) used cost-benefit analysis –type methods, where costs of the scheme were compared to a measure of benefit assigned a monetary value. For example, Chabé-Ferret and Subervie (2013) conducted separate cost-benefit analyses on each of five AESs, deriving estimates of social value for each scheme from the literature and comparing this to costs of implementation. Hansen (2007) combined estimates of the social value of habitat for wildlife viewing and hunting to generate a monetized measure of benefit in an analysis of the CRP.

Most (18 of 31) of the Group 3 studies were *ex ante* evaluations, using estimated costs, whereas the majority of the 13 *ex post* studies used actual costs. Most (20) studies used proxies for measures of ecological effectiveness, with only 11 involving direct measurement. Area-based measurements (e.g. amount of land enrolled; Thompson et al., 1999) were most common among those using proxies, followed by the use of benefit indices (e.g. Stoneham et al., 2003). Studies using direct measures of effectiveness tended to use actual costs in the analysis (6 of 11), whereas those involving proxies focused on estimated costs (16 of 20). However, authors of some of the modelling-based evaluations urged field-based research to validate conclusions (e.g. Bamière et al., 2013; Barraquand and Martinet, 2011).

4. Discussion

The benefits of considering cost in the planning and evaluation of conservation programs have been well demonstrated by several key studies (Boyd et al., 2015; Joseph et al., 2009; Stoneham et al., 2003). And yet this review shows that the integration of economic and ecological data in evaluations is significantly lacking and shows no indication of improving. Less than half of the studies reviewed here included any reference to costs of agri-environment schemes, and only 13% considered issues of cost-effectiveness. Below we consider the potential reasons behind this lack of integration and highlight several studies that illustrate the benefits of considering cost-effectiveness.

4.1. The AES evaluation literature

The AES evaluation literature in general reflects the focus of agri-environmental investment and research around the world. While there were studies from each major geographic region, there was a strong bias towards European and North American studies, explained by those regions committing billions of dollars annually to AES (European Commission, 2013; USDA Farm Service Agency, 2015a). The emphasis on arable landscapes, and on measures involving restoration of vegetative buffers around crop margins, further reflects the focus of conservation investment within those regions. Unfortunately this translates to limited measures of biodiversity effectiveness, with a focus on a small number of taxa, particularly grassland or open field birds and plants. This taxonomic bias, evident across the broader conservation literature (Fazey et al., 2005), comes at the expense of knowledge of the benefits of AES to other taxonomic groups, such as mammals and reptiles, that could potentially benefit through restoration measures on farmland (MacDonald et al., 2007). This may reflect difficulties in obtaining sufficient sample sizes of these taxa in farmland, or alternatively could be indicative of a focus of AES towards certain taxonomic groups, possibly due to concerns over potential impacts of certain species, particularly mammals, on agricultural production (Reid et al., 2007).

There was a strong focus on *ex post* evaluations which are considered important because they allow assessment of whether anticipated benefits materialized, and can be used to inform the design of future programs to improve effectiveness and efficiency (OECD, 2012). Such evaluations, however, may underestimate benefits if carried out too soon after scheme completion owing to the long time lags that can occur before ecological outcomes are achieved (Burrell, 2012). *Ex ante* evaluations can address this by using expected costs and benefits to model cost-effectiveness in advance of the scheme and can improve the effectiveness and efficiency of AES expenditure through spatial targeting of conservation measures (e.g. Reynolds et al., 2006), selecting between policies or delivery mechanisms (e.g. Bamière et al., 2013), or maximizing the biodiversity benefits of individual measures (e.g. Delattre et al., 2010). Such evaluations have the added advantage of being less resource intensive than field-based *ex post* approaches, but are subject to different challenges such as uncertainty in biodiversity outcomes and accounting for future costs (OECD, 2012; Robbins and Daniels, 2012).

AES evaluation studies have increased over the past two decades, particularly from 2000 onwards. Uthes and Matzdorf (2013) found a similar trend in the publication of AES-related studies in Europe. This is most likely a reflection of the increased investment in the CAP (European Commission, 2013) and the entrenchment of AES in EU policy in 2000, making them mandatory for EU member states (Uthes and Matzdorf, 2013). Annual funding for Rural Development under the CAP, for which AES are the dominant mechanism, has increased from approximately EUR 2 billion in 1990 to closer to EUR15 billion in 2014 (European Commission, 2013). Similarly, total annual rental payments under CRP increased from USD$82.9 million in 1987 to $1.63 billion in 2014 (USDA Farm Service Agency, 2015a). This growth in agri-environmental policy does not appear, however, to have been matched with a commensurate increase in economic evaluations, or at least integration of economic data into evaluation. As a proportion of total studies published annually, the number looking at issues of cost-effectiveness has remained low since calls were made to consider economic issues in AES evaluation (Kleijn and Sutherland, 2003; Whitby, 2000).

4.2. AES cost-effectiveness studies

The few studies reviewed that integrated cost information demonstrate the versatility in approaches and agricultural land use contexts in which evaluations can be undertaken, including cropping-dominated systems (e.g. Santangeli et al., 2014), as well as grazing (e.g. Boitani et al., 2010; Wynn, 2002) and mixed-enterprise landscapes (e.g. Bamière et al., 2011).

As also noted by Wätzold and Schwerdtner (2005), we observed a focus on spatiotemporal allocation of conservation measures in the AES literature, possibly in recognition of the high variability in cost and benefits in space and time. Spatial variation in effectiveness can be a major factor influencing the variable cost-effectiveness of conservation measures (Kimball et al., 2015). The studies reviewed here show this variation operates at all scales, from within individual farms and even fields (e.g. Pietzsch et al., 2013) to landscapes and across states (e.g. Hansen, 2007). This is further complicated by variability in cost, largely as a result of variation in productivity and therefore opportunity costs, which can be substantial. For example, Klimek et al. (2008) reported 600% variation in the conservation costs identified by farmers in a scheme targeting protection of plant diversity.

Many studies focussed on the efficiency of scheme delivery mechanisms, often contrasting fixed rate, area-based payments with alternatives such as auctions (e.g. Bamière et al., 2013; Stoneham et al., 2003) or spatial targeting approaches (e.g. Lewis et al., 2009; Thompson et al., 1999). Stoneham et al. (2003) found that a fixed-price AES delivered 25% less biodiversity benefit than the same budget administered using an auction mechanism. Bamière et al. (2013) reported a cost saving of 50% using an auction-based approach in the conservation of avian habitat, potentially doubling the amount of conservation that could be achieved with the same budget using a simple area-based subsidy. A 'payment by results' approach achieved a 17% saving compared to fixed payments in the conservation of remnant habitats on agricultural land (White and Sadler, 2012). While more sophisticated delivery mechanisms such as these can be more cost-effective (Thompson et al., 1999), the increased transaction costs associated may decrease overall program efficiency (Klimek et al., 2008; Lewis et al., 2009). Uthes et al. (2010) also suggested that such approaches are less cost-effective than more general ('horizontal') approaches when multiple environmental objectives are involved.

The complex relationship between cost and benefit is also further illuminated by these studies. While some show an increase in benefit with increasing cost (e.g. Wilson et al., 2007), others show benefits varying independent of cost (e.g., Wynn, 2002) and provide further evidence that greater investment does not equate to greater biodiversity outcomes. Benefit-cost relationships may even differ significantly between co-occurring species within the same taxonomic group (e.g. Holzkämper and Seppelt, 2007), further stressing the importance of considering costs and benefits specific to the particular scheme and its objectives.

The inclusion of economics can reveal some 'ugly truths' of AES investment, such as significant windfall effects for farmers (Bamière et al., 2013; Chabé-Ferret and Subervie, 2013; Sierra and Russman, 2006), ineffective schemes (Boitani et al., 2010) and inefficiencies in expenditure where more cost-effective options are available to that commonly employed (Santangeli et al., 2014). While this contributes to criticisms of AES, such learnings are critical to enable future improvements.

4.3. The poor integration of economics and ecology

This review provides further evidence of limited integration of economics into biodiversity conservation. While indicative of a wider trend in the conservation sciences (TEEB, 2009; Wortley et al., 2013), it is particularly troubling in the evaluation of agri-environmental policy given the magnitude of investment allocated globally each year and the high variability in effectiveness (Batáry et al., 2015). There are several potential reasons for this limited integration.

Firstly, a lack of integration of the disciplines of economics and conservation may be a key factor (Aronson et al., 2010). Holl and Howarth (2000) identified perceived differences in the beliefs, techniques and language of economic and conservation disciplines as possible barriers. They suggested a philosophical aversion of some conservationists to the integration of economics with the conservation of nature, led by a belief that biodiversity shouldn't be valued in monetary terms (see Parks and Gowdy, 2013). This may stem in part from the misguided belief that the integration of economics with conservation necessitates the assignment of monetary value to natural assets (e.g. biodiversity), and that the primary goal is to 'weigh up' conservation over other outcomes. The challenges of assigning monetary value to outcomes or benefits for which there is no ready market value are not unique to conservation. The health care field has overcome these challenges through the use of non-monetary evaluation techniques such as cost-effectiveness analysis, thereby avoiding the technical and ethical challenges of monetizing the quality or quantity of human life (Medvecky, 2015). Several studies in this review demonstrate that conservation benefits can be obtained through the use of non-monetary techniques such as cost-effectiveness analysis. Increased promotion and education on economic principles and techniques may further improve uptake. However, as noted by Medvecky (2015), there is a significant lack of training within tertiary institutions in conservation economics, observing that none of the 21 top universities surveyed offered a dedicated conservation economics course, whereas 17 offered health economics.

Another potential factor is the shortcomings typical in the design of conservation programs. AES are often characterized by poorly defined objectives (Kleijn et al., 2006), which makes the design and implementation of monitoring and evaluation studies difficult. Uthes and Matzdorf (2013) suggested that the absence of clear objectives of AES explains the absence of cost-effectiveness analyses which, by their nature, require objectives against which to measure the efficiency of interventions.

A third key factor includes the limited availability of cost data (Holl and Howarth, 2000; Robbins and Daniels, 2012), particularly spatially explicit costs (Naidoo and Ricketts, 2006). As noted by Kimball et al. (2015) in the field of ecological restoration, the practitioner and researcher are seldom the same individual or organization. The former may be aware of costs but not undertake the research. The latter's expertise lies in evaluation, but not costs. Funding institutions may also fail to collect, or disseminate cost information (Boitani et al., 2010). Where accurate cost data are not available and is critical for the particular analysis, such as cost-benefit analysis (Boardman et al., 2010), costs can be estimated using surrogates such as agricultural production value (i.e. opportunity cost) (e.g. Bamière et al., 2011; Lewis et al., 2009), and area-based approaches (e.g. Chabé-Ferret and Subervie, 2013). Where the objective of evaluation is to identify the most cost-effective intervention from a range of potential options, the use of actual data is less critical than the use of standardized costs across interventions, enabling comparison of the *relative* cost-effectiveness.

5. Conclusions

If AES investment is to be more effective, conservation actions and conservation research need to shift its focus to align with global priorities (Lawler et al., 2006). Current global economic realities dictate cost-effective conservation as one of those priorities. Despite repeated calls for a shift towards more integrated evaluation of AES, to date only a small proportion of studies consider economics when measuring the overall effectiveness of these major investments. Whatever the reasons for this lack of integration in the past, it is clear that many are no longer valid. There is a growing awareness of the benefits of multidisciplinary evaluation of conservation programs (Cullen and White, 2013), and a wealth of practical guidance intended to bridge the divide between the economics and conservation disciplines (see Duke et al., 2013; Naidoo and Ricketts, 2006; Naidoo et al., 2006; Robbins and Daniels, 2012). With careful, but minor, modification to the experimental design of scheme evaluations, the collation or estimation of costs, and simple analytical approaches, the potential for substantial biodiversity gains from future schemes become possible.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2016.04.008.

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Supplementary information

This document contains supplementary information for the article by Ansell et al. titled 'The cost-effectiveness of agri-environment schemes for biodiversity conservation: A quantitative review'. It provides details of the information extracted from each study reviewed (Table S1), and also shows the top 10 journals (by number of publications reviewed) against each of the three groups (see Methods) (Table S2).

Table S1. Detailed explanation of categories for data collected.

Category	Explanation	Sub-categories
Group 1 – all studies		
Year	Year of publication in peer-reviewed journal	N/A
Journal	Journal title	N/A
Location	Country or countries, and geographic region where study undertaken	Europe (EU-member)
		Europe (non-EU member)
		North America
		Latin America
		Oceania
		Asia
		Africa
Scheme	Name of agri-environment scheme (sub-programs rolled into overarching scheme)	N/A
Intervention type	Intervention/s under evaluation	Crop management - Increase crop diversity, harvest timing, mowing, ploughing, disking, tilling
		Grazing management
		Habitat planting -Wildflower/grass buffers and strips, trees
		Habitat protection/land sparing
		Organic farming, input reduction, maintenance of traditional farming practice

Category	Explanation	Sub-categories
		Set-aside/passive restoration
		Other
Objective of intervention	Main objective of intervention as identified by author/s	Single species
		Multiple species
		Biodiversity
		Habitat/ecosystem
Agricultural land use	Dominant agricultural land use of study location as identified by authors	Grazing
		Cropping
		Mixed
		Unknown
Scale	Scale of evaluation undertaken	Field - 1 field within a farm
		Farm – >1 fields within a farm
		Landscape/Regional/National - >1 farm
Single or multiple intervention comparison?	Number of separate intervention types evaluated	Single
		Multiple
Objective	Focus of evaluation	Single species
		Multiple species
		Biodiversity
		Habitat/ecosystem
Number of effectiveness measures	Single or multiple effectiveness measures in evaluation	Single
		Multiple
Type of effectiveness measure	Effectiveness measured directly or indirectly	Direct –parameters directly associated with objective of study
		Indirect- parameters
Taxonomic focus	Specific biotic and/or abiotic value/s measured	Plant
		Bird

Category	Explanation	Sub-categories
		Herpetofauna (reptile/amphibian)
		Mammal
		Invertebrate
		Fish
Group 2 - Studies including any reference to economic costs in the publication, including use of cost-related terms and symbols (i.e. financial, expenditure, budget, cost, economic, investment, dollars, $)		
Reference to economic cost		Yes
		No
Costs provided	Whether any costs were provided	Yes
		No
Total cost reported	Whether total costs were provided	Yes
		No
		Unknown
Estimated or actual costs	Whether costs reported are actual or estimated	Actual
		Estimated
		Unknown
Cost components	Whether costs provided were identified as public and/or private costs	Public costs
		Private costs
		Not specified
Opportunity costs	Whether opportunity costs (e.g. foregone income from agricultural production) were quantified	Yes
		No
Group 3 – Studies using cost data in analysis of AES effectiveness		
Cost-effectiveness approach	Approach used to evaluate cost-effectiveness	Cost-effectiveness - effectiveness divided by cost (or vice-versa)
		Cost-benefit - monetized effectiveness measure divided by cost (or vice-versa)

Category	Explanation	Sub-categories
		Multi-criteria - Use of multiple effectiveness and cost measures to generate index
		Other
Effectiveness measure valued?	Whether effectiveness measure was assigned monetary value (e.g. using non-market valuation techniques)	Yes
		No

Table S.2. Top 10 journals represented in each Group in this study. Numbers following journal title indicate total number of publications/percentage of total publications.

Group 1	Group 2	Group 3
Agriculture, Ecosystems and Environment 38/16	Agriculture, Ecosystems and Environment 15/14	Ecological Economics 5/16
Biological Conservation 27/11	Biological Conservation 11/10	Biological Conservation 3/10
Journal of Applied Ecology 20/8	Journal of Wildlife Management 7/6	Australian Journal of Agricultural and Resource Economics 2/6
Journal of Wildlife Management 14/6	Ibis 6/5	Conservation Biology 2/6
Wildlife Society Bulletin 9/4	Journal of Applied Ecology 6/5	Journal of Environmental Management 2/6
Ibis 8/3	Ecological Economics 5/4	Journal of Soil and Water Conservation 2/6
American Midland Naturalist 7/3	Wildlife Society Bulletin 5/4	Agricultural and Resources Economic Review 1/3
Bird Study 7/3	Conservation Biology 4/4	Agriculture, Ecosystems and Environment 1/3
Restoration Ecology 6/3	Journal of Environmental Management 4/4	American Journal of Agricultural Economics 1/3
Conservation Biology 5/2	Land Use Policy 3/3	American Midland Naturalist 1/3

S3. Full list of references reviewed, categorized by Group. Group 1 represents all studies evaluating conservation effectiveness of agri-environment identified by search strategy included in this review. For presentation purposes, Group 2 and Group 3 studies are listed separately below. Group 2 included those studies that made any reference to the cost of interventions and/or the cost of the AES policy as a whole. This included any use of the cost-related terms and symbols identified through full text searches. Group 3 included studies that explicitly considered cost-effectiveness through any use of cost data in their evaluation of the AES.

Group 1 studies (excluding Groups 2 and 3 studies)

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Paper VI. Comparing the effectiveness of alternative conservation strategies: an evaluation of woodland bird conservation actions in agricultural landscapes.

A key component in the evaluation of cost-effectiveness is the measurement of the effectiveness, or the benefit, of the action/s undertaken. This is best measured as the difference in biodiversity values at a treatment site with that of the counterfactual – the scenario that reflects the absence of the treatment. The difference, or gain, in biodiversity values provides a more appropriate measure of the conservation benefit that can be directly attributed to the action, and when paired with economic costs, provides a sound basis for comparison of cost-effectiveness. Surprisingly few evaluations, however, explicitly consider the counterfactual and use gains as measure of conservation benefit, with potential implications for the prioritization of future investments. In Paper VI, I compared the gains in richness in response to two alternative conservation actions commonly applied in agricultural landscapes around the world: the active restoration of habitat through revegetation of heavily cleared sites ('restoration plantings') and the passive restoration of remnant habitat through fencing to exclude livestock ('remnant protection') which aims to promote recruitment of native vegetation.

Photo: G. Dabb

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Comparing the effectiveness of alternative conservation strategies: an evaluation of woodland bird conservation actions in agricultural landscapes

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Abstract

The effectiveness of conservation actions is best measured as the difference between the biodiversity values following the treatment with that of the counterfactual—the scenario that reflects the absence of the conservation action. However, evaluation is often based on total biodiversity values rather than measure of the gains in biodiversity values that can be attributed to a conservation action. This can lead to potential misrepresentations of conservation effectiveness. In this study we compared the change in bird communities between 32 restoration plantings and 10 fenced woodland remnants, each matched with a control site representing the most likely counterfactual for each site. Gains in native bird species richness in restoration plantings were more than 60 times those in woodland protection sites, while gains in woodland-dependent bird species richness was eight times greater in restoration plantings than protected woodlands. Neither strategy led to significant increases in the richness of birds of conservation concern. Our results suggest that restoration plantings in heavily cleared farmland yield greater gains in bird species richness than protecting remnants that are unlikely to be cleared under the counterfactual. However, differences in bird community composition between recently planted and remnant habitats suggest that an effective conservation strategy for birds should combine restoration plantings in cleared landscapes and ongoing conservation of remnants patches, particularly where the counterfactual for that remnant habitat is loss.

Keywords: Farmland biodiversity, revegetation, conservation benefit, counterfactual, woodland birds

1. Introduction

As the global biodiversity crisis deepens (Newbold et al., 2016), the prioritization of actions that maximize conservation benefits becomes even more critical (Brooks et al., 2006). This is especially important when decision-makers are faced with multiple alternative actions to achieve a particular conservation outcome. In such situations, conservation investments should be directed towards those actions that provide the greatest benefits. In agricultural landscapes, where farming practices represent the second largest threat to global terrestrial biodiversity (Maxwell et al., 2016), conservation practices fall broadly into two categories: the management of existing habitat features (e.g. protection or restoration of remnant vegetation) and the establishment of new habitats on modified sites (e.g. tree planting). These broad strategies form the basis of most of the current efforts to conserve biodiversity in agricultural landscapes around the world, including the billions of dollars spent annually on agri-environment schemes across Europe and North America (Ansell et al., 2016).

While there are many studies into the effectiveness of these actions, a number of issues prevent direct comparison of their conservation benefits. Firstly, many studies focus on the effectiveness of single actions, typically comparing biodiversity values at treatment sites with those of control or reference sites (e.g. Gardali et al., 2006; Twedt et al., 2002). Studies that compare the effectiveness of alternative conservation actions are relatively rare. Secondly, the majority of studies use absolute biodiversity values as measures of effectiveness (e.g. species richness, diversity, abundance) (Ruiz-Jaen and Aide, 2005; Wortley et al., 2013). In such cases, comparisons are often made between the biodiversity values at the treatment site with that of a reference (i.e. undisturbed site), which typically contain higher biodiversity values than restored sites (Benayas et al., 2009; Curran et al., 2014). Such approaches assume that the counterfactual scenario—defined as the outcome that would have occurred in the absence of the intervention (Ferraro, 2009)—is total habitat loss, which may not be a correct assumption.

An alternative approach involves the evaluation of gains (or losses) for a given conservation action, rather than absolute values, as a measure of conservation benefit (Maron et al., 2013). This requires an understanding of the counterfactual scenario. The difference in biodiversity values between a realistic counterfactual and that of the management treatment represents the gain that can be directly attributed to a conservation action and therefore provides a more accurate measure of effectiveness (Ferraro and Pattanayak, 2006). A common issue across much of the conservation literature, however, is the lack of explicit consideration of the counterfactual (Ferraro and Pattanayak, 2006; Maron et al., 2013) (but see Cullen et al., 2005 for example). This can be due to difficulty in obtaining information regarding an appropriate counterfactual, or because the study was designed for a purpose other than calculating gains in biodiversity from management interventions. A failure to explicitly consider the counterfactual can yield biased estimates of the effectiveness of conservation actions and suboptimal conservation investments (Ferraro and Pattanayak, 2006).

We explored this issue in an evaluation of the effectiveness of two alternative actions commonly employed to conserve avian biodiversity in agricultural landscapes. The first, remnant protection, involves mitigation of the impacts of livestock grazing through the installation of fences surrounding remnant woodland patches, commonly without further intervention (Maron and Lill, 2005). Livestock grazing alters the structure and composition of vegetation (Dorrough et al., 2011) and can facilitate increases in predatory and hyper-aggressive species (Maron et al., 2011), negatively impacting woodland birds (Martin and McIntyre, 2007) and other faunal groups (Kay et al., 2016). Reduction in grazing pressure can also assist natural regeneration (Weinberg et al., 2011) and positively impact bird communities (Earnst et al., 2012). The alternative action, restoration planting (i.e. revegetation), involves the planting of perennial woody vegetation (primarily native trees and shrubs) in previously cleared agricultural land to restore habitat for farmland biodiversity (Rey Benayas and Bullock, 2015).

Both management actions are widely practiced to conserve birds in Australian farming landscapes, for which loss and degradation of native vegetation provide the greatest threat (Ford, 2011). Remnant protection forms the basis of the Environmental Stewardship Program, a $152m agri-environment scheme aimed at conserving biodiversity in agricultural landscapes (Burns et al., 2016), while restoration plantings are undertaken across the country by government and non-government organizations, community groups and private individuals (Booth et al., 2012). Together, they form the dominant strategies available to managers seeking to conserve declining woodland fauna in heavily cleared agricultural landscapes, and therefore it is important for decision makers to understand differences in their effectiveness for conserving woodland birds.

We directly compared the effectiveness of remnant protection and restoration plantings for birds using an experimental design that explicitly considers the counterfactual in agricultural landscapes. To our knowledge, there have not been any evaluations that directly compare the conservation benefits of these alternative actions using measures of gain derived from counterfactual contrasts. Specifically, we asked whether gains in bird species richness varies between remnant protection and restoration plantings on agricultural land, and if evaluation of the effectiveness of these conservation actions based on measurement of gains in richness differs from that based on total richness.

2. Methods

2.1. Study area

Our study was conducted across a 2500 km^2 area in the Boorowa region (34°17'-34°45' S, 148°30'-149°02'E; 420–736 m a.s.l.; annual rainfall 613mm) in southeastern New South Wales, Australia. Agriculture is the major land use across the region, covering more than 90% of the area, with livestock (sheep and cattle) grazing most dominant (Australian Bureau of Statistics, 2014). More than 85% of the native vegetation in the region has been cleared, with the remaining vegetation highly fragmented, consisting of small (typically <2 ha) patches found mostly on slopes and ridgelines less suitable for grazing and cropping (Benson, 2008; NSW National Parks and Wildlife Service, 2002). Over the past 30 years, the region has been the focus of biodiversity conservation efforts through various state and federal initiatives, as well as the activities of non-government organizations, volunteer-based groups, and individual landholders using both restoration planting and remnant protection approaches (Freudenberger et al., 2004).

2.2. Site selection

Candidate sites were identified through discussion with key organizations and individual landholders. Restoration planting sites were revegetated patches established through direct-seeding, planting of tubestock (i.e., individual plants) or a combination of both, using predominantly native woody tree and shrub species. Plantings were established in continuously grazed fields cleared of woody vegetation approximately 100 years ago, with some sites integrating scattered remnant trees. Remnant protection sites were remnant patches of *Eucalyptus*-dominated woodland and dry forest around which fences had been erected to exclude livestock grazing, primarily to protect and enhance native vegetation and woodland bird habitat (Driver and Davidson, 2002), with sites of relatively high ecological quality prioritized for inclusion (Driver et al., 2000). Evidence of historic selective logging and removal of firewood was visible at several remnant protection sites. Remnant protection was undertaken primarily for biodiversity conservation objectives (Driver et al., 2000), while restoration plantings were also undertaken for socio-economic objectives (e.g. wind breaks, shelter for livestock, aesthetics) as well as biodiversity outcomes.

Sites were selected from this candidate list where: i) the area was at least 0.5 ha; ii) the date of restoration (i.e. planting or fencing) was known and occurred at least 7 years prior; iii) the minimum distance to other candidate sites was at least 500 m. We also selected sites to represent the typical diversity of sizes and shapes of such projects. We excluded restoration plantings in riparian areas or erosion gullies. Livestock grazing pressure in both plantings and remnant protection sites was either completely removed or limited to infrequent unintentional grazing episodes. Grazing by non-livestock species was not restricted. This process identified a total of 32 restoration planting sites and 10 remnant protection sites. The lower number of remnant protection sites in our study is a result of the widespread extent of historical land clearing in the region which limits conservation opportunities.

Selection of matched control sites to represent the counterfactual scenario, and against which conservation gains could be measured, was a vital component of the study. Counterfactual controls for remnant protection sites (hereafter 'unfenced remnants') were patches of remnant vegetation of the same dominant vegetation type, and size where feasible, that remained unfenced and subject to continuous livestock grazing. This best represents the counterfactual scenario for remnant protection in our study region and in many other extensive agricultural landscapes where remnant vegetation is restricted largely to lands less suitable for intensive agriculture, such as steep slopes and hilltops (Benson, 2008; Gibbons and Boak, 2002). Such vegetation is unlikely to be cleared through agricultural conversion, the dominant cause of vegetation clearing on private land in NSW (OEH, 2016). Furthermore, remnant native vegetation is afforded varying degrees of protection through government regulation (Bradshaw, 2012).

Restoration planting controls (counterfactual scenario) were adjacent fields cleared of native woody vegetation approximately 100 years ago, and now dominated by native and introduced pasture grasses and forbs and subject to continuous grazing. These sites ('paddocks') were typically situated in fields from which the restoration planting had been established through subdivision. Half (16) of our restoration plantings incorporated isolated remnant trees (live or dead), and so we situated matched controls for these sites in fields with a similar density of remnant trees as their presence can strongly influence bird assemblages (Hanspach et al., 2011). All control sites were 200-950 m from treatments and were selected to match, to the extent possible, the elevation, slope, aspect and agricultural land-use of the treatment site prior to conservation intervention. With the exception of one site pair, all sites were located on the same farm and were under the same management as they were at the time of the conservation intervention. Discussion with the previous and current owners of the remaining site revealed little change in management of the control or treatment since the change in ownership. This provided a total of 84 sites (64 restoration planting and 20 remnant protection treatment-control site pairs) across 22 farms.

2.3. Bird surveys

We conducted a total of 336 bird surveys across the 84 sites during spring (September-November) 2013. Each site was surveyed four times, using 5-minute point counts (Bibby et al., 2000) at the 0 m, 75 m and 150 m points along a 150 m fixed transect randomly selected. At each site, two surveys were conducted in the morning (from 15 mins prior to sunrise to 3 hours after sunrise) and two in the evening (less than two hours before sunset). All birds seen and heard within 0-50m were recorded, excluding those flying overhead. Treatment and control site pairs were surveyed in random order and within 15 minutes of each other. We allowed a minimum of two days between surveys at a site to maximize independence between surveys (Field et al., 2002). All surveys were conducted by DA to avoid the effect of observer heterogeneity on species detection probability (Cunningham et al., 1999). Surveys were not undertaken during extreme temperatures, high winds, rainfall or fog.

All species with fewer than three individuals recorded during the entire surveys were excluded from analysis (sensu Munro et al., 2011). Species were organized into groups for subsequent analysis, including woodland species, which we defined as those species primarily associated with remnant woodlands. Classification of woodland birds is often inconsistent across the literature (Fraser et al., 2015), therefore a consensus-based approach was used, where we compared classifications used in five key publications (Barrett et al., 2008; Bennett and Ford, 1997; Radford and Bennett, 2005; Reid, 1999; Silcocks et al., 2005) and categorized species as woodland species where the majority of these studies had done similarly. We also identified woodland species of conservation concern which included those listed under state or federal threatened species legislation as well as those species for which there was a 20% or greater decline in reporting rate either nationally or across South Eastern Highlands and South Western Slopes bioregions between the 1981 and 2002 Atlas of Australian Birds survey periods (Barrett et al., 2003, 2007).

2.4. Covariates

We measured several site and landscape-scale variables known to influence woodland birds to control for any potential confounding effects (Table 1). The area and perimeter of treatment sites was measured using satellite imagery and ArcGIS software (ESRI, 2015). The shape of each site was measured using an index of compactness, where $shape = (Area/perimeter^2) \times 4 \times \pi$ (sensu Mac Nally, 2007). We calculated the per cent of woody vegetation within 500m of the central point of the transect at each site, following similar studies (e.g. Montague-Drake et al., 2009), using a binary classification derived from 5 m resolution SPOT-5 satellite imagery (NSW Office of Environment and Heritage, 2014). The ownership of each site ('farm') was also recorded to account for potential influences of farm management on restoration response.

Table 1. Co-variates measured at each site that were included in statistical models. Figures provided are means, with the range in parentheses.

Variable	Definition	Remnant protection	Restoration planting
Age (years)	Age of the site as of 2014	11 (7-14)	14.4 (9-23)
Area (ha)	Area of the restoration	30.2 (3.1-94)	5.33 (0.7-21.7)
Shape	Index of the compactness of the site	0.59 (0.35-0.74)	0.32 (0.03-0.74)
Surrounding woody vegetation (% cover)	Cover of woody vegetation within 500m of restoration site	26.2 (1-64.2)	4.4 (1-19.8)
Grazing index	Index of livestock grazing pressure	0.5 (0-3)	0.38 (0-3)
Farm	Defined by ownership of the land on which the site was located.	10 sites situated on 8 different farms	32 sites situated on 23 different farms

2.5. Analysis

To investigate the effect of conservation treatment on the composition of bird communities, we firstly conducted correspondence analysis on a matrix of presences/absences for each species at each site, removing eight species observed at only a single site, as well as two sites at which no species were recorded. This analysis characterized each of the remaining 82 sites by the composition of the species present. We then used canonical correspondence analysis (CCA) to relate the composition at each site to the four site types (plantings, paddocks, remnant protection sites and unfenced remnants) and used Monte Carlo simulations (1000 steps) to test the significance of the first two axes of the ordination ($P < 0.05$).

We modelled the effect of site type on total bird species richness across all surveys at each site using generalized linear mixed models (Schall, 1991). It should be noted that while we refer to term total bird species richness for simplicity, our measure is more accurately described as richness per sampling unit. This is an important distinction as we used a fixed-area sampling approach across all sites rather than scaling for patch size. We compensate for any potential bias introduced by the variation in the size of treatments by including the size of sites as a covariate in our models. This approach is widely used and accepted in ecological comparison of ecological data across sites of varying size (Cunningham et al., 2007; Lindenmayer et al., 2010).

Separate models were fitted for the total richness of each of the three species groups (all species, woodland species and woodland species of conservation concern) across the four site types using a quasi-Poisson distribution (to allow for over- dispersion) with a logarithmic link function, and the variable 'farm' fitted as a random effect to account for potential dependence from the nestedness of greater than one site within individual farms. In addition to conservation treatment, we initially included all site and landscape variables, then sequentially removed non-significant ($P \geq 0.05$) variables until only those with a significant effect remained.

We then modelled the effect of conservation treatment on gains in bird species richness for the three species groups. Gains were calculated as the difference in richness between the treatment and control at each site pair. Note that, for simplicity, we use the term 'gain' here to denote the difference in richness between the treatment and control sites, acknowledging that in some cases this difference can be negative (i.e., a loss). Our approach to analyzing these data was different to the total richness models as the data contained negative values. After confirming the absence of a random effect of 'farm' using linear mixed modelling, we used multiple linear regression, again starting with a full model, then sequentially removed non-significant variables.

All analyses were conducted using GenStat (VSN, 2002), with the exception of the correspondence analysis and canonical correspondence analysis, which were conducted using the "MASS" and "vegan" packages in R (R Core Development Team, 2015).

3. Results

We recorded 97 bird species across all surveys, 13 of which were omitted from further analyses as fewer than three individuals were observed (Appendix A). Of the remaining 84 species, 56 were classified as woodland species, 11 of which were also classified as species of conservation concern. Seventy-five species (89%) were observed in restoration plantings, 59 species (70%) in remnant protection sites, 55 (65%) in unfenced remnants and 51 (61%) in paddocks. Thirty-five species (42%) were observed across all four site types. Seven species were only found in plantings. Two species were only observed in remnant protection sites. There were no species that were restricted to either paddocks or unfenced remnants.

3.1. Bird community composition

Bird community composition differed between conservation treatments (first canonical correlation = 0.60). We found that the first two CCA axes explained a significant amount of variation in species composition (*P* = 0.001). The first axis (CCA1), which contrasted paddocks with other site types, accounted for 45% of this variation (DF = 1, $\chi^2 = 0.16$, F = 3.09, P = 0.001). The second axis (CCA2), which contrasted plantings with the other sites, accounted for 42% of the variation (DF = 1, $\chi^2 = 0.15$, F = 2.95, P = 0.001) (Fig. 1). Bird communities occurring in restoration planting sites differed from the paddock sites, and both differed from remnant protection and matched control sites. There was no difference however in bird community composition between the remnant protection sites and unfenced remnants (DF = 1, $\chi^2 = 0.04$, F = 0.85, P = 0.76).

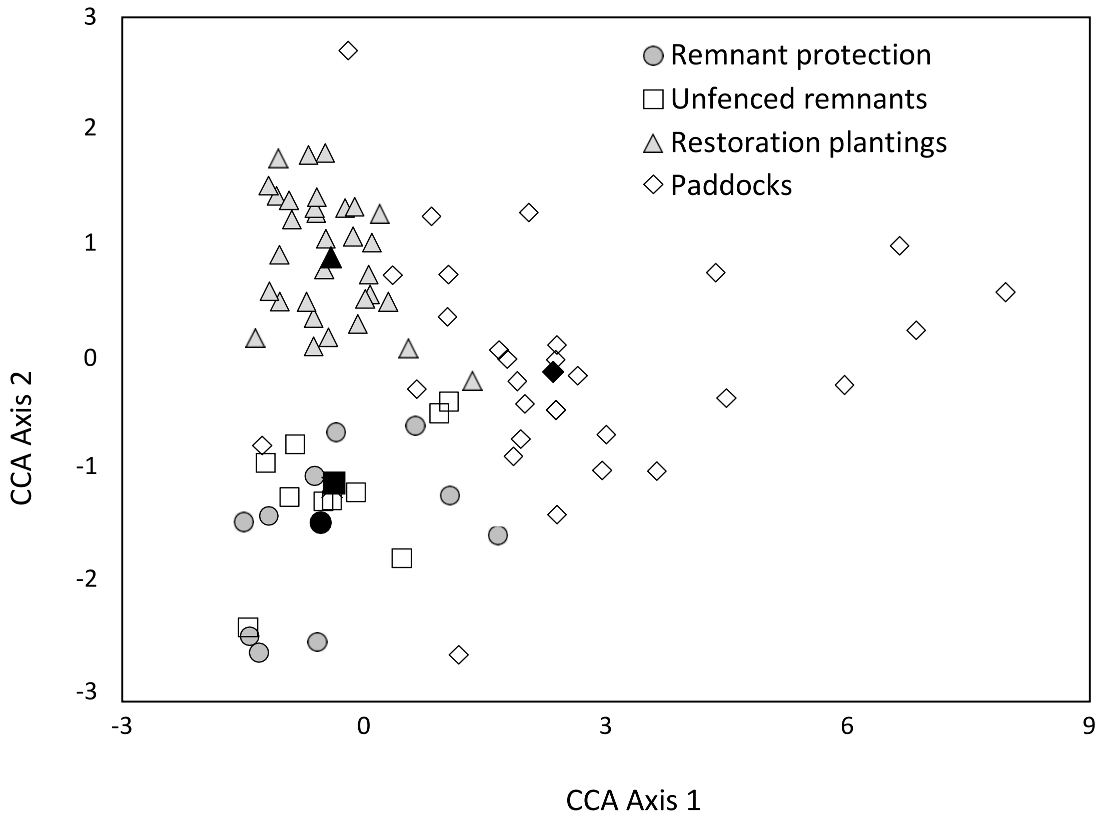

Figure 1. Ordination of the canonical correspondence analysis showing relationship between bird community and site type. Each white or grey point represents a site. Solid black shapes represent centroids for each site type. The first axis (CCA1) contrasts paddocks with other site types. The second axis (CCA2) contrasts restoration plantings with the other sites.

3.2. Total richness

The mean total richness of all bird species did not differ between remnant protection, plantings and unfenced remnants, though all were significantly higher than paddock sites (Wald statistic = 199, DF = 3, *P* < 0.001) (Fig. 2). There was no significant effect of any of the measured covariates on total bird species richness (Table 2). Woodland bird species richness followed the same pattern, with the number of species occupying plantings, remnant protection sites and unfenced remnants being significantly greater than paddock sites (Wald statistic = 200.75, DF = 3, $P < 0.001$) (Fig. 2). Woodland bird species richness was also positively associated with the amount of woody vegetation within 500m of a site (Wald statistic = 8.77, DF = 1, $P = 0.003$). The number of woodland species of conservation concern was significantly higher in plantings, remnant protection sites and unfenced remnants, than in paddock sites (Wald statistic = 16.99, DF = 3, $P = 0.001$) (Fig. 2). We found no effect of any covariates on the number of woodland species of conservation concern.

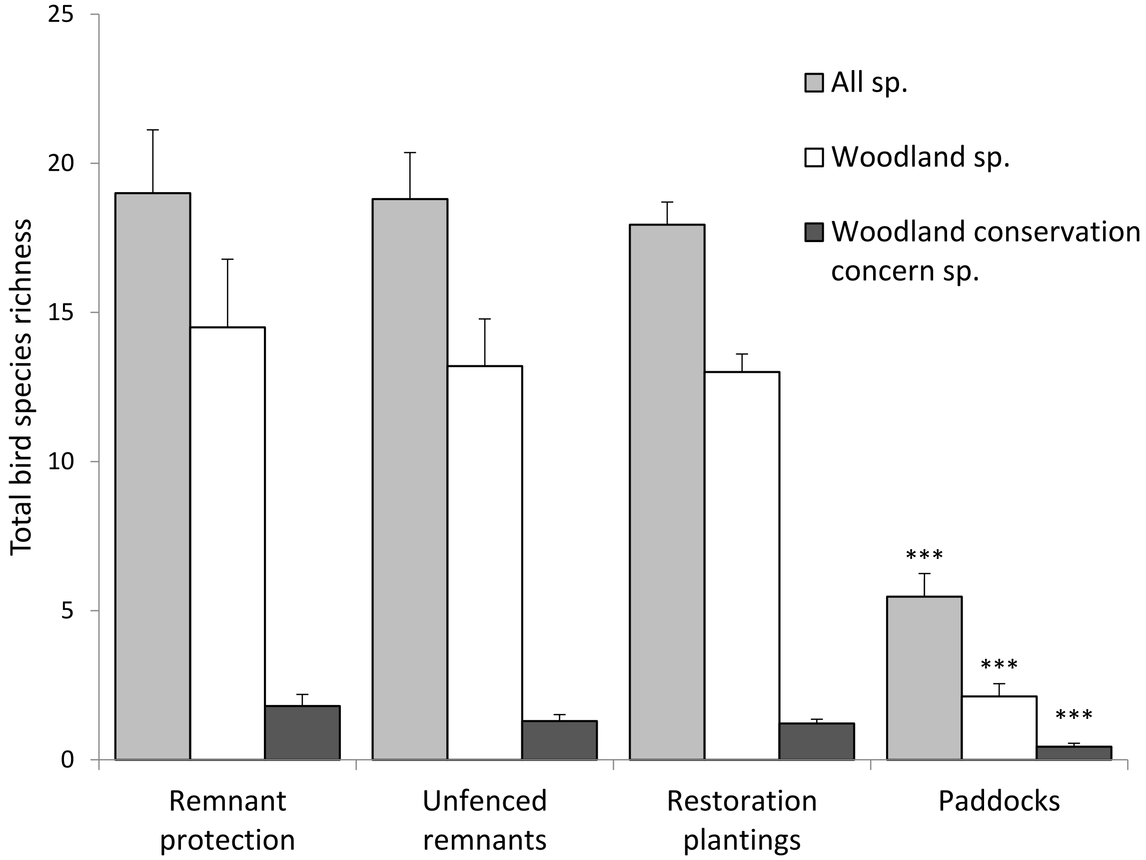

Figure 2. Mean richness of all bird species, woodland species and woodland species of conservation concern across site types. Values provided are predicted means and 95% confidence intervals from generalized linear mixed models. Asterisk indicates a significant difference between in richness of bird species groups between site types (*** P < 0.001).

Table 2. Effect of treatment and several environment variables on total richness for all bird species, woodland species and woodland species of conservation concern. Models are based on GLMM.

Response	Terms	Wald statistic	d.f.	P-value
All species	Treatment type	199.00	3	<0.001
Woodland species	Treatment type	200.75	3	<0.001
	Surrounding woody vegetation	8.77	1	0.003
Woodland species of conservation concern	Treatment type	16.99	3	0.001

3.3. Gains in richness

We defined gains as the difference in richness of the three bird groups between each conservation treatment site and its paired counterfactual. For restoration plantings, the counterfactual was paddock sites; and for remnant protection, the counterfactual was unfenced remnants. When compared with the relevant counterfactual, restoration planting led to significantly higher gains across all bird species than remnant protection (Wald statistic =43.38, DF = 1, P <0.001) (Fig. 3). Average gains from restoration planting were 62 times greater than gains from remnant protection. Restoration planting also led to significantly higher gains in woodland species (Wald statistic = 39.05, DF = 1, P <0.001), being eight times greater from restoration planting compared with remnant protection (Fig. 3). There was no difference in gains in woodland species of conservation concern between the two conservation treatments (Wald statistic = 0.7097, DF = 1, P = 0.405) (Fig. 3). There were several instances where the gains measured in the remnant protection sites were actually negative (i.e., richness was greater in the matched control). Gains for all bird species and for woodland species were negative in four out of 10 and five of 10 remnant protection sites respectively. We found no effect of site or landscape context on gains in species richness across the three species groups (Table 3).

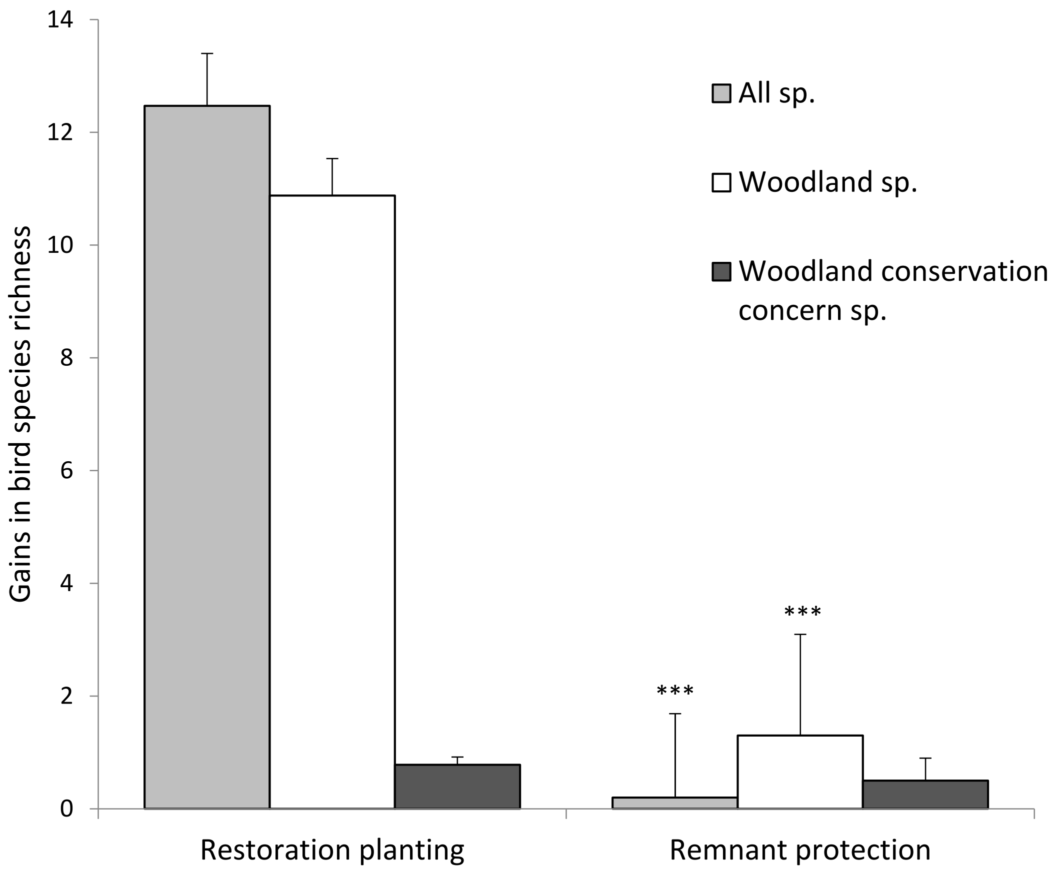

Figure 3. Mean (+ SEM) gains in bird species richness (treatment minus control; see Methods) of all bird species, woodland species and woodland species of conservation concern following restoration planting and remnant protection. Asterisk indicates a significant difference between in richness of bird species groups between the two conservation strategies (*** P < 0.001).

Table 3. Effect of conservation treatment on gains in richness of all bird species, woodland species and woodland species of conservation concern. Models are based on general linear regression.

Response	**Terms**	**Wald statistic**	**d.f.**	**P-value**
All species	Treatment type	43.38	1	<0.001
Woodland species	Treatment type	39.05	1	<0.001
Woodland species of conservation concern	Treatment type	0.71	1	0.405

4. Discussion

4.1. Planting versus protection of bird habitat

Given the limited funding available for conservation (McCarthy et al., 2012), it is important that we prioritize actions that provide the greatest biodiversity gain for least cost. However, most studies evaluate effectiveness based on total biodiversity values using designs that do not explicitly consider the counterfactual (Maron et al., 2013; Miteva et al., 2012). In this study, we used counterfactual data to demonstrate that restoration plantings in heavily cleared agricultural land delivers much greater biodiversity gains compared with protection of remnant habitats from livestock grazing.

We found that gains in bird species from restoration planting were on average 62 times greater than those following remnant protection. These benefits extended to woodland birds, which represent an at-risk group in many agricultural regions (Rayner et al., 2014), with gains following restoration planting eight times higher than remnant protection. We were surprised by the strong response of woodland birds in particular to restoration planting. Given the specific resource requirements of many species in this assemblage, we should expect fewer woodland species in plantings (Munro et al., 2011). Species such as the superb parrot (*Polytelis swainsonii)*, for example, which is normally strongly associated with woodland habitat (Manning et al., 2006), were found more often in restoration plantings.

Our results highlight the potential benefits of active habitat restoration in cleared agricultural land, and question the effectiveness of conservation of remnant habitats through livestock exclusion as a strategy for conserving birds, especially where the counterfactual scenario involves the retention of that habitat and thus averted loss is negligible (Maron et al., 2012b). In contrast to restoration plantings, gains in bird species richness following remnant protection were instead low (average gain of less than one species compared to unfenced remnants) and highly variable. Indeed, several remnant protection sites displayed lower species richness than the paired unfenced site. While livestock grazing has been implicated in the decline of bird species in agricultural landscapes across the world (Fleischner, 1994; Ford, 2011; Newton, 2004), the effectiveness of conservation of woodland remnants through fencing to reduce grazing pressure is highly variable (Spooner and Briggs, 2008). Management plans are often not developed for these sites, or where they are, contracts are rarely established that obligate the landholder to follow certain management prescriptions (but see Burns et al., 2016). As such, fenced woodland remnants are often still grazed to varying degrees (Spooner and Briggs, 2008) with potential for ongoing biodiversity impacts. Even when grazing pressure is removed or reduced as in our study, passive restoration may be impacted by climatic factors and the legacy of past land use practices such as introduction of weeds (Kay et al., 2016). This suggests that, in many landscapes, conservation of remnant woodland patches, particularly high quality patches, through fencing may represent a suboptimal strategy in the conservation of birds, including woodland birds (Ford, 2011). Redirecting some resources towards restoration planting, and prioritizing fencing of lower quality remnant patches that are likely to degrade further in the absence of conservation may produce greater conservation benefits (Gibbons, 2016).

There are some important caveats to this however. Firstly, our measure of biodiversity benefit was limited to birds in this study. While our results could transfer to other taxa that have been shown to respond rapidly to restoration planting (e.g., beetles; Gibb and Cunningham, 2010), other taxa may respond differently. Second, differences in the underlying productivity of the parts of the landscape in which restoration plantings and remnants habitats are situated may confound observed biodiversity responses (Maron et al., 2012a). Soil productivity can play an important role in structuring bird communities (Montague-Drake et al., 2011; Watson, 2011), and the restriction of remnant habitats to less productive areas may contribute to the disparity in conservation gains observed in this study. Third, we acknowledge the difference in cost between these two conservation approaches and that cost-effectiveness is an increasingly important factor in conservation planning (McDonald et al., 2015). Further research should integrate economic costs to compare the efficiency of these techniques.

We do not propose a diminished role of remnant habitats in the conservation of farmland biodiversity or advocate a total shift in resources towards restoration planting of cleared landscapes. Our study adds to the growing evidence that effective biodiversity conservation in agricultural landscapes requires a diversity of habitat types and conservation approaches (Ikin et al., 2016; Lindenmayer et al., 2012). While restoration planting often leads to increases in richness and diversity, community composition rarely resembles that of intact habitats (Curran et al., 2014). Our data reveal differences in the composition of bird communities in planted sites compared with remnant habitats (both fenced and unfenced sites), which typically provide greater structural complexity and key resources (e.g. hollows, fallen branches) that are often absent from restoration plantings and may take many decades to develop (Munro et al., 2009). The restriction of species such as the brown treecreeper *(Climacteris picumnus),* a species of conservation concern that require certain bark types on mature trees for foraging (Bennett et al., 2013), to remnant habitats in our study provides an example. Plantings may provide important transitional habitat (Barrett et al., 2008), which when combined with remnant woodland patches and scattered trees, provide a key resource in the conservation of fauna in production landscapes. The absence of any appreciable gains in woodland birds of conservation concern also questions the effectiveness of either strategy in the protection of threatened bird species in farmland. Management actions targeting the specific resource requirements of individual species and the threatening processes affecting them may be required.

4.2. Evaluating conservation effectiveness

Our study raises two key interrelated issues regarding the evaluation of conservation actions: the importance of explicitly considering the counterfactual scenario (Ferraro and Pattanayak, 2006; Miteva et al., 2012) and of using biodiversity gains, rather than total biodiversity value, as a measure of effectiveness (Maron et al., 2013). Evaluations that fail to account for the counterfactual, by default, make a series of assumptions that can significantly bias evaluations and overestimate effectiveness of conservation actions (Joppa and Pfaff, 2010). In our study, the magnitude of the difference in the conservation gains between restoration planting and remnant protection reflects the differences in their respective counterfactual scenarios. In the absence of the treatment, restoration planting sites remain paddocks with limited habitat value for many bird species, and remnant protection sites remain remnant vegetation with much greater habitat values. If instead, our evaluation was based on comparison of the total biodiversity values of restoration plantings and remnant protection sites with undisturbed habitats, as is often the case (Ruiz-Jaen and Aide, 2005; Wortley et al., 2013), we make the assumption that the counterfactual scenario is the total absence or loss of that habitat. While we acknowledge that the total loss of habitat is a realistic counterfactual in some regions and circumstances (Seto et al., 2012), this is not true across much of our study area and other landscapes where remnant native vegetation is afforded protection through government regulation or poor suitability of the land for alternative uses.

In our study bird species richness in remnant habitats, both remnant protection and unfenced sites, was similar to that of restoration planting sites. In many studies, however, the biodiversity values in the latter, especially younger plantings, are often significantly lower (Munro et al., 2007). Traditional evaluation approaches using total biodiversity values often conclude the limited effectiveness of, for example, restoration planting as a result of the lower total biodiversity values relative to reference sites (e.g., Wilkins et al., 2003). This is a valid comparison where restoration plantings are used as a conservation action to offset the loss of reference sites or where the intention is to highlight the value of relatively unmodified habitat. However, when contrasts such as this are applied more broadly, we believe this contributes to an inherent bias in conservation towards management of higher quality sites, despite the potential for greater marginal benefits to be achieved through a focus on lower quality sites (Huth and Possingham, 2011).

We acknowledge that the integration of counterfactual information can be difficult (see Ferraro and Pattanayak, 2006; Miteva et al., 2012). However, we argue that explicitly defining the counterfactual scenario represents a more robust conceptual approach for evaluating the true gain from conservation actions (Ferraro, 2009) and will stimulate greater research on what is a realistic counterfactual for intact, or partially degraded, habitats.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found online.

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Appendix A. List of all bird species observed, also showing those classified as (a) woodland species and (b) woodland species of conservation concern. * Species omitted from analyses as less than 3 individuals were observed during study.

Australasian pipit	*Anthus novaeseelandiae*	Galah	*Eolophus roseicapillus*
Australian magpie	*Cracticus tibicen*	Golden whistler [a]	*Pachycephala pectoralis*
Australian raven	*Corvus coronoides*	Grey butcherbird [a]	*Cracticus torquatus*
Black duck*	*Anas superciliosa*	Grey fantail [a]	*Rhipidura albiscapa*
Blackbird	*Turdus merula*	Grey shrike-thrush [a]	*Colluricincla harmonica*
Black-chinned honeyeater*	*Melithreptus gularis*	Horsfield's bronze cuckoo [a]	*Chalcites baslis*
Black-eared cuckoo*	*Chalcites osculans*	Jacky winter [a,b]	*Microeca fascinans*
Black-faced cuckoo-shrike [a]	*Coracina novaehollandiae*	Laughing kookaburra [a]	*Dacelo novaeguineae*
Brown falcon*	*Falco berigora*	Leaden flycatcher [a]	*Myiagra rubecula*
Brown goshawk*	*Accipiter fasciatus*	Little button quail*	*Turnix velox*
Brown songlark	*Cincloramphus cruralis*	Little corella	*Cacatua sanguinea*
Brown thornbill [a]	*Acanthiza pusilla*	Little friarbird [a]	*Philemon citreogularis*
Brown treecreeper [a,b]	*Climacteris picumnus*	Magpie lark	*Grallina cyanoleuca*
Brown-headed honeyeater [a]	*Melithreptus brevirostris*	Masked lapwing	*Vanellus miles*
Buff-rumped thornbill [a]	*Acanthiza reguloides*	Masked woodswallow	*Artamus personatus*
Collared sparrow hawk [a]	*Accipiter cirrocephalus*	Mistletoe bird [a]	*Dicaeum hirundinaceum*
Common bronzewing pigeon [a]	*Phaps chalcoptera*	Nankeen kestrel	*Falco cenchroides*
Common starling	*Sturnus vulgaris*	Noisy friarbird [a]	*Philemon corniculatus*
Crested pigeon	*Ocyphaps lophotes*	Noisy miner [a]	*Manorina melanocephala*
Crested shriketit*	*Falcunculus frontatus*	Olive-backed oriole [a]	*Oriolus sagittatus*
Crimson rosella [a]	*Platycercus elegans*	Pallid cuckoo [a,b]	*Cacomantis pallidus*
Diamond firetail*	*Stagonopleura guttata*	Pied butcherbird	*Cracticus nigrogularis*
Dollarbird*	*Eurystomus orientalis*	Pied currawong [a]	*Strepera graculina*
Double-barred finch	*Taeniopygia bichenovii*	Rainbow bee-eater	*Merops ornatus*
Dusky woodswallow [a,b]	*Artamus cyanopterus*	Rainbow lorikeet	*Trichoglossus haematodus*
Eastern rosella	*Platycercus eximius*	Red wattlebird [a]	*Anthochaera carunculata*
Eastern spinebill [a]	*Acanthorhynchus tenuirostris*	Red-browed finch [a]	*Neochmia temporalis*
Eastern yellow robin [a]	*Eopsaltria australis*	Red-capped robin [a]	*Petroica goodenovii*
European goldfinch	*Carduelis carduelis*	Red-rumped parrot	*Psephotus haematonotus*
Fairy martin	*Petrochelidon ariel*	Restless flycatcher*	*Myiagra inquieta*
Fan-tailed cuckoo*	*Cacomantis flabelliformis*	Rufous songlark	*Cincloramphus mathewsi*

Appendix S1. (cont'd)

Rufous whistler [a]	*Pachycephala rufiventris*	White-winged Triller [a,b]	*Lalage sueurii*
Sacred kingfisher [a]	*Todiramphus sanctus*	Willie wagtail	*Rhipidura leucophrys*
Satin flycatcher [a]	*Myiagra cyanoleuca*	Yellow thornbill [a]	*Acanthiza nana*
Scarlet robin [a,b]	*Petroica boodang*	Yellow-faced honeyeater [a]	*Lichenostomus chrysops*
Silvereye [a]	*Zosterops lateralis*	Yellow-rumped thornbill [a]	*Acanthiza chrysorrhoa*
Skylark*	*Alauda arvensis*		
Southern whiteface [a,b]	*Aphelocephala leucopsis*		
Speckled warbler [a,b]	*Chthonicola sagittata*		
Spotted pardalote [a]	*Pardalotus punctatus*		
Striated pardalote	*Pardalotus striatus*		
Striated thornbill [a]	*Acanthiza lineata*		
Stubble quail	*Coturnix pectoralis*		
Sulpher-crested cockatoo	*Cacatua galerita*		
Superb fairy-wren [a]	*Malurus cyaneus*		
Superb parrot [a,b]	*Polytelis swainsonii*		
Tree martin	*Petrochelidon nigricans*		
Varied sitella [a,b]	*Daphoenositta chrysoptera*		
Wedge-tailed eagle*	*Aquila audax*		
Weebill [a]	*Smicrornis brevirostris*		
Welcome swallow	*Hirundo neoxena*		
Western gerygone [a]	*Gerygone fusca*		
White-browed babbler [a]	*Pomatostomus superciliosus*		
White-browed scrubwren [s]	*Sericornis frontalis*		
White-browed woodswallow [a,b]	*Artamus superciliosus*		
White-eared honeyeater [a]	*Lichenostomus leucotis*		
White-faced heron	*Egretta novaehollandiae*		
White-plumed honeyeater [a]	*Lichenostomus penicillatus*		
White-throated gerygone [a]	*Gerygone albogularis*		
White-throated treecreeper [a]	*Cormobates leucophaea*		
White-winged chough [a]	*Corcorax melanorhamphos*		

Paper VII. Plant a tree or build a fence? Evaluating the cost-effectiveness of alternative bird conservation actions in an agricultural landscape.

In Paper VI, I show substantial differences in the effectiveness of restoration plantings and remnant protection, with the gains in bird species richness more than 60 times greater following restoration planting. The financial costs of these alternative actions however also differ substantially and therefore have the potential to influence their cost-effectiveness. In Paper VII, I calculated the costs of these alternative actions and combined this data with the measures of conservation benefit in a use cost-effectiveness analysis that directly compared their efficiency in the conservation of birds in agricultural landscapes.

Photo: D. Ansell

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Plant a tree or build a fence? Evaluating the cost-effectiveness of alternative bird conservation actions in an agricultural landscape.

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Abstract

When faced with alternative conservation actions, decision-makers should be guided by those that are the most cost-effective, that is, deliver the greatest benefit per dollar invested. Despite substantial investment of public funds in conservation, the use of economics to evaluate conservation actions has been limited. We compared the cost-effectiveness of two restoration strategies used to conserve birds in agricultural landscapes: restoration plantings in cleared farmland, and remnant protection (passive restoration of remnant vegetation on farmland through removal of livestock grazing pressure). We measured conservation benefit by observing bird communities at 84 sites comprised of 42 restoration sites each matched with a control site reflecting the counterfactual scenario—the most likely scenario were the investment not to occur. We calculated conservation gains based on the difference in richness of assemblages of birds between each treatment-control site pair and estimated total public costs of restoration. We used cost-utility analysis to compare the efficiency of the two actions. Gains in bird species richness were, on average, greater and more cost-effective following restoration planting than remnant protection. Investment in restoration planting yielded an average gain of 7 ± 0.7 species per $10,000 compared to investment in remnant protection that yielded an average loss of 0.2 ± 0.7 species per $10,000. Shape and size contributed significantly to variation in efficiency between the two restoration approaches. Compact, less elongated restoration plantings generated the most cost-effective conservation outcomes. From a conservation investment perspective, the variability in the gains and gains per unit cost—both of which included negative values (i.e. losses)—following remnant protection suggest it is a riskier strategy than restoration planting.

Keywords: farmland biodiversity; impact evaluation; active restoration; cost-utility analysis

1. Introduction

It is broadly acknowledged that the funds available for conservation are insufficient to address the scale of the threats to biodiversity (Balmford et al., 2003; McCarthy et al., 2012). Biodiversity conservation strategies therefore need to consider those actions that are the most cost-effective, that is, deliver the greatest benefit per dollar spent or achieve a particular objective at lowest cost (McDonald et al., 2015; Nunes et al., 2015). This is especially critical where multiple alternative conservation actions are available that vary in effectiveness and financial cost, creating the potential for significant inefficiencies in conservation expenditure. Failure to select the most cost-effective alternative can lower the biodiversity benefits that can be achieved with a fixed budget, or increase the cost of achieving a particular conservation outcome.

Cost-effectiveness studies can increase the efficiency of conservation expenditure by identifying the most cost-effective management intervention (Cullen et al., 2005) or parcels of land (Carwardine et al., 2008) for conservation. The benefits for biodiversity and cost-savings can be substantial, with some studies demonstrating several orders of magnitude difference in cost-effectiveness between alternative conservation strategies (e.g. Kimball et al., 2015; Stoneham et al., 2003). Despite the benefits of evaluating cost-effectiveness, the conservation evaluation literature suffers from a lack of integration of economic information (Wortley et al., 2013) and such analyses remain relatively uncommon, particularly in agricultural landscapes, where billions of conservation dollars are spent annually (Ansell et al., 2016b).

Ecological restoration is now a dominant strategy in the conservation of biodiversity in agricultural landscapes around the world (Rey Benayas and Bullock, 2012; Secretariat of the Convention on Biologival Diversity, 2010). Two of the primary restoration options available are broadly categorised as active and passive restoration (Suding, 2011). Active restoration aims to re-establish habitat primarily through planting (by seed or individual plants), but also other resource intensive actions such as weed and pest control and burning (Benayas et al., 2008). Passive restoration, in contrast, relies on natural regeneration and successional processes, often facilitated by the removal of a stressor such as livestock grazing. These approaches vary widely not only in conservation effectiveness (Curran et al., 2014; Rey Benayas et al., 2009), but also in financial cost (Kimball et al., 2015; Morrison and Lindell, 2011).

Direct comparison of the cost-effectiveness of alternative conservation actions through field evaluation are rare (Ansell et al., 2016b). To date, studies of the cost-effectiveness of these restoration approaches have largely been limited to ex ante evaluations using modelled benefits and costs (e.g., Birch et al., 2010). There have been very few ex post field-based evaluations of cost-effectiveness of ecological restoration, especially those that directly compare active and passive restoration (Ikin et al., 2016). In the absence of this information, managers are more likely to select the least cost option which does not necessarily equate to the most cost-effective. In doing so, managers and funders forgo potential biodiversity gains from the limited funds available.

In this study, we conducted an evaluation of the cost-effectiveness of active restoration ('restoration planting' or revegetation of heavily cleared sites) and passive restoration ('remnant protection' or fencing of remnant vegetation to exclude livestock grazing and facilitate natural regeneration) in the conservation of birds in an agricultural landscape in south-eastern Australia.

2. Materials and methods

2.1. Study area and site selection

Our study was focussed on a 2500km^2 landscape in southeast New South Wales (34°17'-34°45' S, 148°30′-149°02'E; 420–736 m a.s.l.; annual rainfall 613mm). This area has been the focus of extensive ecological restoration efforts over the past 30 years (Freudenberger et al., 2004), largely in response to heavy land clearing undertaken in the early 1900s (NSW National Parks and Wildlife Service, 2002). Agriculture represents the dominant land use across the region, primarily involving livestock grazing on rainfed pastures (Australian Bureau of Statistics, 2014).

We identified candidiate restoration plantings and remnant protection sites across the region through discussion with landholders and natural resource management organsiations. Remnant protection sites were remnant woodland and dry forest patches, dominated by *Eucalytpus* species, that had been fenced to reduce livestock grazing pressure and to assist natural regeneration. These patches of remnant vegetation typically occur in parts of the landscape less suitable for intensive agriculture (Gibbons and Boak, 2002). Restoration planting sites comprised predominantly native woody tree and shrub species planted (nursery tubestock and/or direct-seeding) into cleared agricultural land, commonly in more agriculturally productive parts of the landscape where most native vegetation has been previously cleared. Though landholder motivations often vary for undertaking restoration on agricultural land (Smith, 2008), we selected only those sites for which biodiversity conservation was a primary objective.

To enable measurement of conservation gains for use in subsequent analyses, each restoration site was paired with a control site to represent the counterfactual scenario, that is, the site in the absence of restoration. Measures of effectiveness derived from contrasts of the biodiversity values following a conservation action such as restoration with those from carefully selected counterfactuals affords a more rigourous evaluation of the impact of conservation interventions (Ferraro and Pattanayak, 2006; Ferraro, 2009). These conterfactual (control) sites were selected to match the aspect, slope, elevation and pre-restoration management regime of the treatment sites, and were located between 200m and 950m from treatment sites. For restoration planting sites, the controls were continuously grazed fields cleared of woody vegetation (with the exception of scattered remnant trees in some sites), whereas controls for remnant protection sites were continuously grazed patches of the same vegetation type. By matching the remnant protection sites with similar (albeit continuosly grazed) patches of the same vegetation, we make the assumption that the counterfactual does not involve clearing of that vegetation. This is a realistic scenario in our study region, where the remnant vegetation is found mostly on slopes and ridgelines less suitable for grazing and cropping (Benson, 2008; NSW National Parks and Wildlife Service, 2002) and is therefore unlikely to be cleared through agricultural conversion, the dominant cause of vegetation clearing on private land in NSW (OEH, 2016). Further, the clearing of remnant vegetation is regulated by legislation in our region and in other parts of Australia (Bradshaw, 2012).

In total, we selected 84 sites for evaluation, including 32 restoration planting-control site pairs, and 10 remnant protection site-pairs. The smaller number of remnant protection sites reflects their limited availability in our study region, largely a consequence of extensive historic land clearing.

2.2. Restoration effectiveness

Our measure of the effectiveness of the different restoration techniques was based on comparison of bird communities at restoration sites and their respective controls. We conducted four bird surveys at each treatment and each control site. Each site was surveyed four times between September-November 2013, with a minimum of two days between each surveys to maximize survey independence (Field et al., 2002). At each site we conducted two morning surveys (15 mins before sunrise to 3 hours after sunrise) and two evening surveys (< 2hrs before sunset). Each survey involved three 5 minute point counts at equally spaced points along a 150m transect (Bibby et al., 2000), with all birds seen and heard within 50m at each point recorded by a single observer to remove any observer effect on species detection (Cunningham et al., 1999). We pooled the survey data at each site to generate a species list, and from this list also identified a subset of species which we classified as woodland species, an at-risk group in many agricultural landscapes (Rayner et al., 2014). There is no formal classification of woodland birds in Australia, and classifications used are often inconsistent across the literature (Fraser et al., 2015). Therefore, we used a consensus-based approach where we reviewed five key studies of woodlands birds in our study area (Barrett et al., 2008; Bennett and Ford, 1997; Radford and Bennett, 2005; Reid, 1999; Silcocks et al., 2005), and classified species as woodland-dependant where the majority of these studies had done similarly. Non-native species and native species with fewer than three individuals recorded during the entire surveys were excluded from analysis (*sensu* Munro et al., 2011).

Our design not only allowed us to derive a measure of effectiveness that can be attributed to the restoration action, but it also allowed comparison of the benefits from actions that have different ecological baselines and endpoints. By measuring biodiversity gain (increase in bird species richness), rather than total biodiversity values, we could compare the effectiveness of restoration plantings and remnant protection on an equaivalent basis that ignores the fact that the respective habitats structures differ vastly (both in starting condition and endpoint); and their biodiversity values often differ greatly, with remant habitats typically displaying greater richness than cleared landscapes and actively restored sites (Curran et al., 2014).

Gains were calculated using the following equation:

$$gain_i = x_i - y_i \,, \tag{1}$$

where x_i represents bird species richness at restoration (treatment) site i and y_i represents bird species richness at the paired control site.

2.3. Restoration costs

We calculated the total public cost of restoration for each site as. These costs were estimated as actual costs were not known for all sites. Estimated costs comprised management costs, which included cost of labour and materials (e.g. plants, seed, machinery) required to undertake the restoration, as well as transaction costs, which reflect the administrative costs of identifying and establishing the site (Naidoo et al., 2006) (Table 1).

Table 1. Cost components and pricing structure used to estimate restoration costs. All prices are in 2015 $AUD.

Cost type	Item	Description	Cost/unit
Management costs	Fencing	Materials and labour	$10,000 km^{-1}
	Planting - tubestock [a]	Materials – plants and guards	$1,465 ha^{-1}
		Site preparation	$228 ha^{-1}
		Labour	$1508.60 ha^{-1}
	Planting – direct-seeding [a]	Site preparation	$228 ha^{-1}
		Materials – seed, equipment	< 2 ha = $750 ha^{-1} 2-4 ha = $625 ha^{-1} > 4 ha = $550 ha^{-1}
		Labour	$77.68 ha^{-1} [b]
Transaction costs		Labour	$1200 per site

[a] Restoration planting sites only; [b] Minimum cost $155.36

We calculated costs of materials based on the pricing structure used by Greening Australia, the largest restoration practitioner in Australia. The average number of field labour hours were obtained through discussion with Greening Australia, then multiplied by appropriate national hourly rates (ABS, 2015) to give a labour cost for each cost component. Remnant protection costs were restricted to transaction costs as well as fence installation, whereas restoration planting also included costs of site preparation (e.g. weed control), direct-seeding and/or tubestock planting of nursery grown plants. Direct-seeding costs were scaled to the size of the restoration site reflecting economies of scale (Table 1). Where restoration planting sites comprised both tubestock and direct-seeding, we used visual estimation of the proportion of each restoration type to calculate the cost. We did not include costs of ongoing maintenance as this is relatively uncommon in such restoration projects in this region. The analysis also did not include private costs (i.e. opportunity costs) as our analysis is intended to evaluate cost-effectiveness from a public expenditure perspective. Also, there was not economic data available at a sufficient scale to enable this cost component to be measured. All costs provided are 2015 Australian dollars unless otherwise specified.

2.4. Cost-effectiveness

We used cost-utility analysis (CUA) to compare the efficiency of the two restoration approaches, a variant of cost-effectiveness analysis developed for evaluation of health care programs (Drummond et al., 1987) but with strong application to conservation (Hughey et al., 2003), including the evaluation of threatened species programs (Cullen et al., 2001; Fairburn et al., 2004; Laycock et al., 2011) and broader environmental management contexts (Haddock et al., 2007; Hajkowicz et al., 2008). CUA allows comparison of the efficiency of widely varying alternative actions by using a measure of gain or utility resulting from the actions as a measure of effectiveness, and comparing the costs of producing that utility. We calculated the cost-utility ratio (*CUR*) for both species groups at each restoration site using the equation:

$$CUR_i = \frac{gain_i}{z_i} \times 10000, \tag{2}$$

where $gain_i$represents the gain in bird species richness at site i (Equation 1) and z_i represents estimated total cost of restoration at site i. This was multiplied by 10,000 to give a measure of the gain (or loss) in number of species per $10,000 invested.

2.5. Covariates

At each site we measured a limited number of site and landscape-scale variables known to influence the response of birds to restoration to control for any confounding effects in our models (Table 2). The perimeter and area of each site was measured using ArcGIS 10.1 (Environmental Systems Research Institute, Inc., Redlands, CA) and from this we calculated an index of the shape for each treatment site as $(Area/perimeter^2) \times 4 \times \pi$ (*sensu* Mac Nally, 2007). The age (years since restoration treatment) of each site was measured, and the percentage cover of woody vegetation within a 500m radius of the central point of the study transect was measured at each site using a binary classification derived from 5 m resolution SPOT-5 satellite imagery (NSW Office of Environment and Heritage, 2014). Finally, we recorded ownership of each site (‘farm’) to account for potential dependence where greater than one site is nested within individual farms.

Table 2. Co-variates included in statistical models. Figures provided are means, with the range in parentheses. Area and vegetation cover were log transformed in analyses.

Variable	Definition	Remnant protection	Restoration planting
Age (years)	Age of the site as of 2015	12 (9-14)	14.4 (9-23)
Shape	Index of the compactness of the site (larger number indicates decreasing elongation) *	0.59 (0.35-0.74)	0.32 (0.03-0.74)
Vegetation cover	Percentage cover of woody vegetation within 500m of restoration site	26 (1-64)	4.4 (1-20)
Area	Area in hectares of restoration site	30.16 (3-94)	5.33 (0.7-22)
Farm	Defined by ownership of the land on which the site was located.	8 sites situated on 6 different farms	32 sites situated on 23 different farms

* *sensu* Mac Nally (2007)

2.6. Analysis

We used linear mixed effects models to compare the effectiveness and the cost-effectiveness of the two restoration approaches. Models were fitted using the "lmer" procedure in the *lme4* package (Bates et al., 2014) in the statistical program R (R Core Development Team, 2015), with *farm* included as the random effect to allow for potential dependence among the sites located on the same farm. For the effectiveness models, gains (Eq. 1) were used as response variables, whereas CURs (Eq. 2) were used as response variables in the cost-effectiveness models. This was undertaken for both species groups, giving a total of four models. Our initial models included all explanatory variables and interactions with the main treatment effect (restoration type). Model selection was then performed using the "step" function in the *lmerTest* package which involves automatic backward elimination of all non-significant (P >0.05) fixed effects. Residual analysis was performed on all candidate final models to confirm the underlying assumptions of normality and homoscedasticity. T-statistics and P values were calculated based on degrees of freedom approximated using the Satterthwaite approach in *lme4*. We also compared average cost per hectare using Welch's two sample t-test to measure differences in the costs of the two restoration types.

3. Results

3.1. Bird communities

We recorded a total of 81 bird species that were used in subsequent analyses. An additional 16 species were detected but were not used in any analysis because there were less than three individuals observed across all surveys (13 species) or were non-native (3 species). Seventy-two of the 81 species were observed in restoration planting sites, 48 in restoration planting controls, 58 species in remnant protection sites and 49 species in remnant protection controls. Thirty-three species (42%) were observed across all four site types (treatments/controls). Seven species were only found in plantings, whereas two species were only observed in remnant protection sites. There were no species that were restricted to either control site types. Fifty-six species were classified as woodland species (see Supplementary Material), of which 22 species were observed across all site types. Of the remaining 34 woodland species, nine were observed exclusively in restoration planting sites, and three only in remnant protection sites.

3.2. Restoration effectiveness

Gains in richness of all native bird species were greater following restoration planting (Table 3). On average there was a gain of 12.5 ± 5 species (SD) resulting from restoration plantings, whereas there was only an average gain of 0.2 ± 4.7 species from remnant protection (Fig. 1). Gains in all species were not affected by the shape, age or percentage cover of surrounding woody vegetation (Table 3).

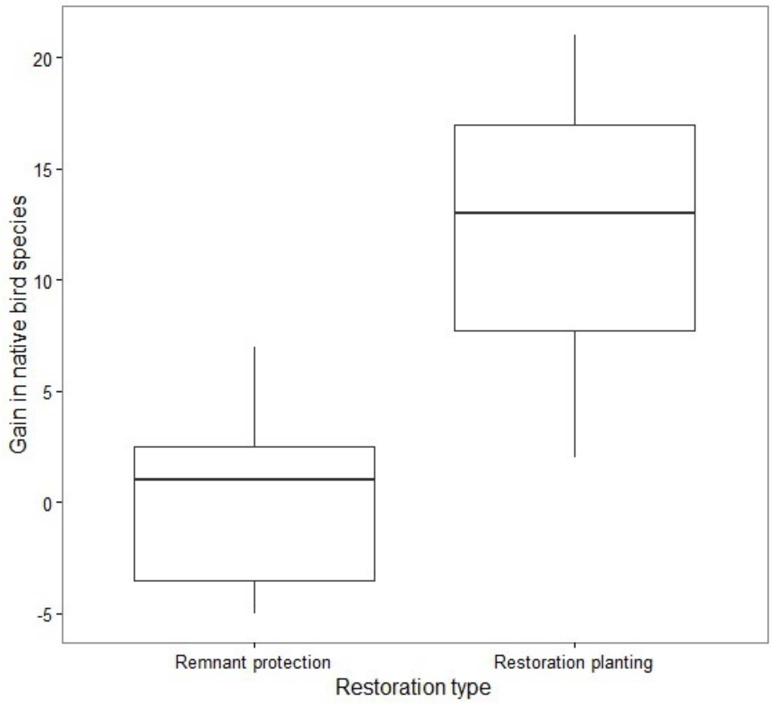

Fig. 1. Gains in the number of bird species following restoration planting and remnant protection (boxplots show the maximum, upper quartile, median, lower quartile and minimum).

Table 3. Effects of restoration type and site design on gains in a) all bird species and b) woodland bird species. Results of linear mixed effects models. Reference level for restoration type is revegetation.

Parameter	Estimate (SE)	t	P
a) All species			
Intercept	0.4 (1.53)	0.26	0.79
Restoration type	-11.94 (1.75)	6.81	<0.001
b) Woodland species			
Intercept	14.313 (5.39)	2.67	<0.05
Restoration type	-5.17 (5.50)	-0.94	0.35
Site shape	-21.99 (8.87)	-2.48	<0.05
Restoration type *x* site shape	27.43 (9.28)	2.96	<0.01

Gains in the number of woodland bird species also differed between the two restoration types, with an average of 10.9 ± 3.7 woodland species following revegetation compared to gains of 1.3 ± 5.7 species following remnant protection. The effect of restoration type on gains in woodland species was strongly determined by the shape of the restoration site (Table 3). Gains in woodland species increased in restoration plantings as the shape of the sites became less elongated (Fig. 2). Gains following remnant protection seemed to follow the opposite relationship, decreasing as sites became more elongated.

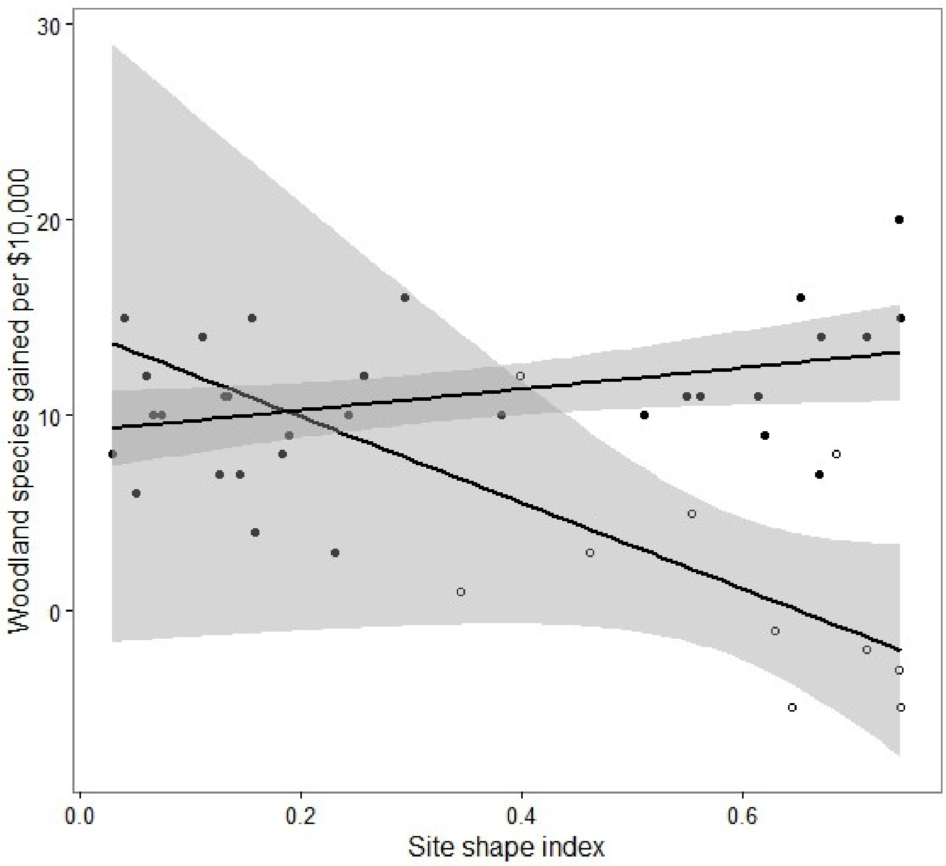

Fig. 2. Relationships between site geometry and mean gains in woodland species following restoration planting (filled circles) and remnant protection (open circles). Increasing site shape index values indicate decreasing elongation of the site. Shaded region represents 95% confidence intervals.

3.3. Restoration cost and cost-effectiveness

The average cost of restoration planting was significantly higher than that of remnant protection (Welch's T-test -6.98, d.f. = 37.92, $P < 0.001$) (Table 4). Comparison of cost-utility ratios revealed differences in the cost-effectiveness between the restoration types and significant influence of site design factors (Table 5). Restoration planting led to average gains of 7.1 ± 4.4 bird species per $10,000, compared to average gains of -0.2 ± 2.8 bird species per $10,000 following remnant protection. Average gains in woodland bird species following restoration plantings were 6.2 ± 3.6 species per $10,000, compared to 0.2 ± 2.9 species per $10,000 following remnant protection.

Table 4. The mean, standard deviation (SD), minimum (min) and maximum (max) costs per hectare of restoration planting and remnant protection across the 42 restoration sites sampled in this study. Amounts are \$AUD ha^{-1}.

Restoration type	Mean cost	SD	Min-Max
Restoration planting	7109	4200	1985-16720
Remnant protection	1564	896	467-3012

The cost-effectiveness of restoration for all bird species was influenced by restoration type and by the size and shape of sites. The efficiency of restoration plantings increased with decreasing elongation and decreased with increasing size of sites (Fig. 3) for both the all species and woodland species groups. Gains per unit cost following remnant protection followed the opposite pattern, with efficiency increasing with increasing size and elongation.

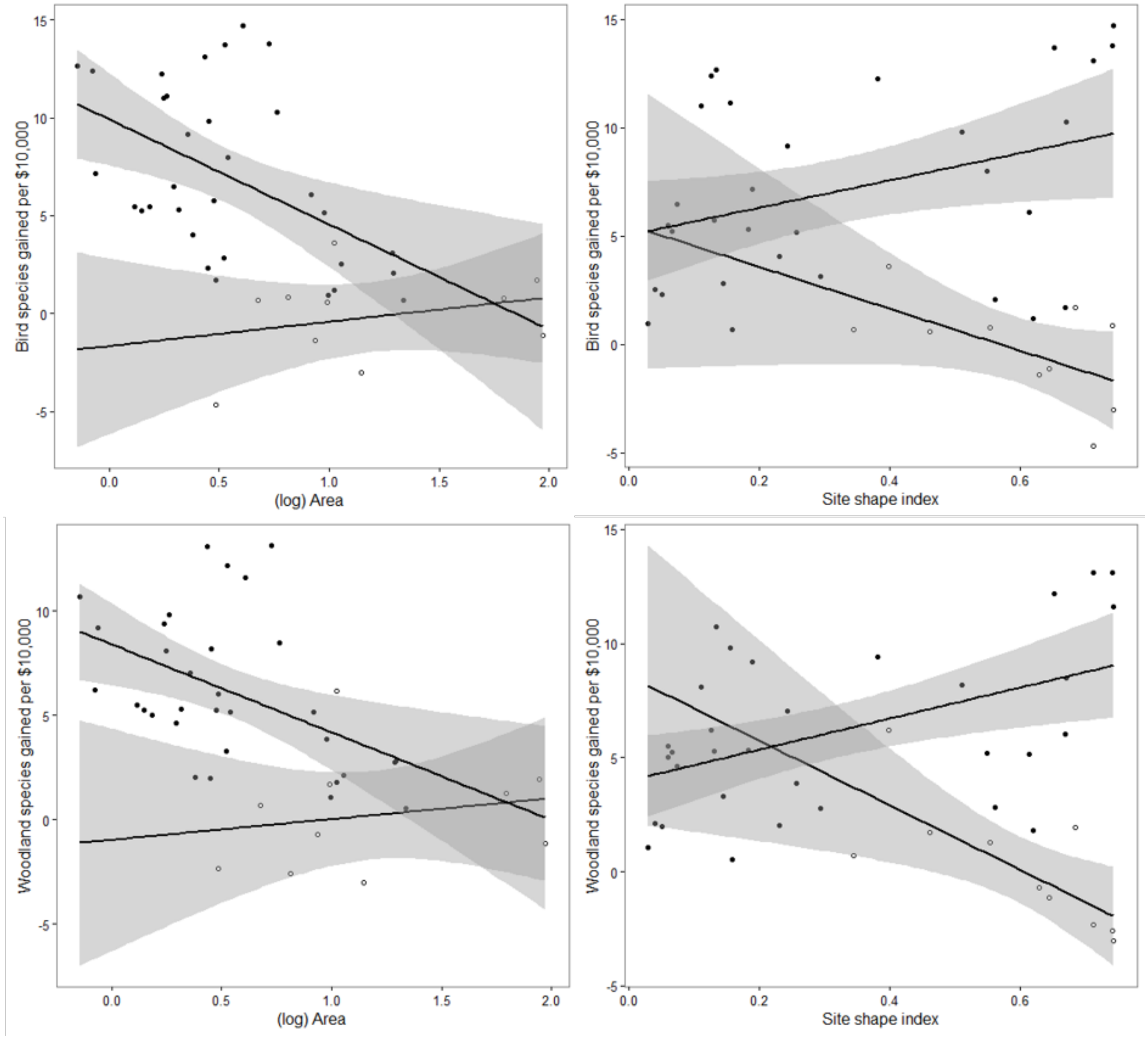

Fig. 3. Relationship between cost-effectiveness of restoration planting (filled circles) and remnant protection (open circles), area (left) and shape (right) for all bird species (top row) and woodland species of restoration sites. Increasing site shape index values indicate decreasing elongation of the site. Shaded region represents 95% confidence intervals.

Table 5. The effect of restoration and site design factors on the efficiency of restoration for a) all bird species and b) woodland bird species conservation. Results of linear mixed effects models. Reference level for Restoration type is revegetation.

	Estimate (SE)	t	P
a) All bird species			
Intercept	4.10 (4.32)	0.95	0.35
Restoration type	3.71 (4.43)	0.84	0.41
Site shape	-10.70 (6.77)	-1.58	0.12
Restoration type *x* site shape	20.55 (7.11)	2.89	0.006
(log) Area	1.72 (1.83)	0.94	0.35
Restoration type x (log) Area	-8.98 (2.28)	-3.94	<0.001
b) Woodland bird species			
Intercept	7.20 (3.21)	2.25	0.031
Restoration type	-0.89 (3.29)	-0.27	0.787
Site shape	-15.14 (-5.02)	-3.02	0.005
Restoration type *x* site shape	24.91 (5.27)	4.72	<0.001
(log) Area	1.69 (1.36)	1.23	0.228
Restoration type x (log) Area	-7.76 (1.69)	-4.58	<0.001

4. Discussion

The use of conservation actions that are cost-effective has become increasingly important as global biodiversity continues to decline and financial resources become further strained. Studies that compare the cost and realized benefits of past projects are comparatively rare (Ansell et al., 2016b), especially in the field of ecological restoration, despite the potential to improve understanding of the relative efficiency of alternative approaches and design more cost-effective conservation programs in future. In many mixed-use agricultural landscapes, the dominant conservation actions include the protection and restoration of remnant habitats and the revegetation of cleared agricultural land (Rey Benayas and Bullock, 2015). We conducted an ex-post evaluation of the cost-effectiveness of these alternatives for conserving woodland birds in an agricultural landscape, using a design that combined multiple measures of conservation gain derived from counterfactual contrasts with the financial costs. We found that the effectiveness, the costs and the cost-effectiveness of these conservation actions varied substantially, and that these differences were strongly influenced by the spatial design of sites. Despite the much higher per hectare cost of restoration planting compared to woodland protection, the number of native bird species (when considering all species and woodland species) gained per unit cost was substantially greater in restoration planting sites.

4.1. Drivers of variation in the cost-effectiveness of restoration

Differences in the cost-effectiveness of conservation actions can be attributed to variation in the effectiveness measure, in cost, or a combination of both (Laycock et al., 2012). In our study, the relative contribution of the effectiveness and the cost of the action varied depending on the bird assemblage under evaluation and was influenced by the geometry of sites. For the full assemblage of native bird species, we found no evidence of an influence of the shape of the site on gains in bird species richness (i.e. effectiveness) following restoration of either type. Therefore, the differences in cost-effectiveness of the two approaches can be attributed exclusively to the differences in cost associated with the design of the site. Fencing materials represent a major cost-component of conservation actions such as woodland protection and revegetation in agricultural landscapes, with costs exceeding AUD$10,000 km^{-1} (Ansell et al., 2016a). More elongated sites have a higher cost per hectare as a result of their greater perimeter to area ratios. However, for woodland bird species (a subset of bird species considered at-risk), we found gains following restoration planting increased with decreasing elongation of the site. Similarly, less elongated sites generated greater conservations gains per dollar than more elongated sites. Several studies show the importance of the geometry of actively restored sites on the response of woodland birds, with less elongated or wider plantings generally supporting greater species richness (e.g. Lindenmayer et al., 2016) (but see Selwood et al., 2009), possibly due to increased edge effects. Our study further supports this, but also reveals a compounding effect of costs. Regardless, the strong influence of shape on cost and cost-effectiveness indicates that restoration projects that require fencing should focus on more compact designs.

We did not observe any relationship between cost-effectiveness and age of conservation actions (9 – 23 years) across either approach. As the costs were one-off (i.e. no ongoing maintenance), any differences would be attributed to changes in the conservation benefits over time. We did not however find evidence of a change in effectiveness over time, which contrasts with the findings of several other studies where species richness increases with time as sites become more structurally complex and are colonized by additional species (Gardali et al., 2006; Kavanagh et al., 2007). These other studies however typically include sites much younger than those surveyed in our study. This suggests that, at least in our study area, the conservation benefits of woodland protection and revegetation plateau within the first decade, though there is potential for benefits to accrue at time scales beyond that surveyed here (i.e., decades).

4.2. Management implications and challenges

In our study, the lower cost of the protection of remnant vegetation relative to restoration planting was not sufficient to offset the lower effectiveness of this action for conserving birds. While the lower cost per hectare of larger remnant protection sites increased their efficiency, the greater gains of restoration planting still make for a more cost- effective proposition. The variability in the gains and gains per unit cost—both of which included negative values (i.e. losses)—following remnant protection also identify it as a much riskier strategy from a conservation investment perspective. A $10,000 investment in remnant protection may lead to a gain in woodland species, or it may be associated with a loss. Several studies demonstrate the variable response of native vegetation, particularly shrubs, grasses and ground layer plants—the common target of remnant protection actions and an important habitat resource for native birds, especially woodland species (Montague- Drake et al., 2009)—after the exclusion of livestock grazing. Even when there is an improvement in vegetation with decreased grazing pressure, there may not be a concomitant response among bird communities (Dorrough et al., 2012). In contrast, restoration planting represents safer investment, with gains and gains per unit cost always positive in our study. Such results question the effectiveness and efficiency of fencing remnant vegetation as a strategy for the conservation of birds in agricultural landscapes, particularly where more effective and cost-effective alternatives exist.

While it is tempting to advocate for a sole focus on less elongated (e.g. round or block-shaped) restoration planting sites over protection of remnant vegetation from grazing in the conservation of woodland birds in agricultural landscapes, several factors caution against such an approach. Firstly, such site configurations can have much lower appeal to farmers. Elongated plantings are normally located along fence lines and farm boundaries in areas perceived to have lower opportunity costs (Welsch et al., 2014), explaining the prevalence of this configuration in farm landscapes. Our study did not capture the private opportunity costs of lost agricultural production arising from the restoration approaches under evaluation. Instead, our results offer an evaluation of cost-effectiveness from a public expenditure perspective, as publicly-funded conservation actions should seek to maximize the public benefit per dollar spent (Hughey et al., 2003). Ultimately, however, conservation actions on private land require the consent of the landholder, for whom opportunity costs are typically a major factor driving decisions regarding uptake of environmental projects on their land (Conradie et al., 2013). Such costs can impact substantially on the cost-effectiveness of conservation (Carwardine et al., 2008). Shifting investment to less elongated planting designs will require recognition of associated opportunity costs, which may need to be offset through the use of financial incentives (e.g. agri-environment schemes) or the adoption of designs that integrate both agricultural production and conservation benefits (e.g. Ansell et al., 2016a)

Secondly, our design intentionally focused on the conservation gains (ie. changes in species richness) generated by the alternative actions as a measure of effectiveness, rather than the measurement of total conservation value (ie. total richness). Implicit in this approach is the assumption regarding the counterfactual scenario for each conservation action. Any changes to the counterfactual will dramatically alter the benefits of conservation actions and thus the level or priority. For example, remnant vegetation may be afforded protection to varying degrees by legislation (Bradshaw, 2012) or by way of its limited suitability for agricultural conversion as a result of its location in the landscape (typically hilltops and steep slopes; Gibbons and Boak, 2002). Our measure of conservation benefit is therefore derived through comparison of fenced and unfenced remnants. However, if this legislative protection did not exist, and the counterfactual was total loss of those habitats (as is the case in many landscapes at risk of agricultural intensification), the conservation benefits would be substantially higher which would likely change the cost-effectiveness. This further underscores the importance of considering the counterfactual in evaluating conservation actions.

While we consider these two conservation actions evaluated here as mutually exclusive (i.e. restoration planting or remnant protection), variations involving both approaches can be applied. For example, supplemental plantings within and surrounding remnant vegetation could increase effectiveness of remnant protection and potentially increase its cost-effectiveness. Also, the restriction of some species to specific restoration types suggests that a combination of approaches may be required to conserve a greater representation of the bird assemblage. This can be assessed using systematic conservation planning approaches to identify cost-effective complementary networks of restoration sites that maximise representation of the bird community (e.g. Ikin et al., 2016).

Finally, while our study included only the management and transaction costs of the restoration techniques; we did not include maintenance costs such as pest and weed control, re-planting, watering or fence repair costs. Our decision was based on the absence of these actions in our study region and many similar areas in which we have experience. In many areas, however, these actions, and therefore costs, will be required and may not be equal across conservation actions and through time. Therefore, we again reiterate the importance of accounting for the associated costs of conservation as comprehensively as possible.

5. Conclusions

Our case study demonstrates how the integration of economic and ecological factors can greatly inform and potentially improve conservation expenditure. Such analysis should become much more common—they are neither difficult nor costly to conduct. Studies such as this challenge traditional approaches to conservation planning that focus on just maximising benefits or minimizing costs. We demonstrate that simple economic evaluation tools coupled with ecological impact evaluation methods can identify opportunities to greatly enhance the conservation benefits that can be achieved with limited resources.

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Appendix A. Supplementary data

Supplementary data related to this article can be found online at [TBC].

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Appendix A. List of all bird species observed, also showing those classified as (a) woodland species. * Species omitted from analyses as less than 3 individuals were observed during study. ** Non-native species omitted.

Australasian pipit	*Anthus novaeseelandiae*	Galah	*Eolophus roseicapillus*
Australian magpie	*Cracticus tibicen*	Golden whistler [a]	*Pachycephala pectoralis*
Australian raven	*Corvus coronoides*	Grey butcherbird [a]	*Cracticus torquatus*
Black duck*	*Anas superciliosa*	Grey fantail [a]	*Rhipidura albiscapa*
Blackbird **	*Turdus merula*	Grey shrike-thrush [a]	*Colluricincla harmonica*
Black-chinned honeyeater*	*Melithreptus gularis*	Horsfield's bronze cuckoo [a]	*Chalcites baslis*
Black-eared cuckoo*	*Chalcites osculans*	Jacky winter [a]	*Microeca fascinans*
Black-faced cuckoo-shrike [a]	*Coracina novaehollandiae*	Laughing kookaburra [a]	*Dacelo novaeguineae*
Brown falcon*	*Falco berigora*	Leaden flycatcher [a]	*Myiagra rubecula*
Brown goshawk*	*Accipiter fasciatus*	Little button quail*	*Turnix velox*
Brown songlark	*Cincloramphus cruralis*	Little corella	*Cacatua sanguinea*
Brown thornbill [a]	*Acanthiza pusilla*	Little friarbird [a]	*Philemon citreogularis*
Brown treecreeper [a]	*Climacteris picumnus*	Magpie lark	*Grallina cyanoleuca*
Brown-headed honeyeater [a]	*Melithreptus brevirostris*	Masked lapwing	*Vanellus miles*
Buff-rumped thornbill [a]	*Acanthiza reguloides*	Masked woodswallow	*Artamus personatus*
Collared sparrow hawk [a]	*Accipiter cirrocephalus*	Mistletoe bird [a]	*Dicaeum hirundinaceum*
Common bronzewing pigeon [a]	*Phaps chalcoptera*	Nankeen kestrel	*Falco cenchroides*
Common starling **	*Sturnus vulgaris*	Noisy friarbird [a]	*Philemon corniculatus*
Crested pigeon	*Ocyphaps lophotes*	Noisy miner [a]	*Manorina melanocephala*
Crested shriketit *	*Falcunculus frontatus*	Olive-backed oriole [a]	*Oriolus sagittatus*
Crimson rosella [a]	*Platycercus elegans*	Pallid cuckoo [a]	*Cacomantis pallidus*
Diamond firetail *	*Stagonopleura guttata*	Pied butcherbird	*Cracticus nigrogularis*
Dollarbird *	*Eurystomus orientalis*	Pied currawong [a]	*Strepera graculina*
Double-barred finch	*Taeniopygia bichenovii*	Rainbow bee-eater	*Merops ornatus*
Dusky woodswallow [a]	*Artamus cyanopterus*	Rainbow lorikeet	*Trichoglossus haematodus*
Eastern rosella	*Platycercus eximius*	Red wattlebird [a]	*Anthochaera carunculata*
Eastern spinebill [a]	*Acanthorhynchus tenuirostris*	Red-browed finch [a]	*Neochmia temporalis*
Eastern yellow robin [a]	*Eopsaltria australis*	Red-capped robin [a]	*Petroica goodenovii*
European goldfinch **	*Carduelis carduelis*	Red-rumped parrot	*Psephotus haematonotus*
Fairy martin	*Petrochelidon ariel*	Restless flycatcher *	*Myiagra inquieta*
Fan-tailed cuckoo *	*Cacomantis flabelliformis*	Rufous songlark	*Cincloramphus mathewsi*

Appendix A. (cont’d)

Rufous whistler [a]	*Pachycephala rufiventris*	White-winged Triller [a]	*Lalage sueurii*
Sacred kingfisher [a]	*Todiramphus sanctus*	Willie wagtail	*Rhipidura leucophrys*
Satin flycatcher [a]	*Myiagra cyanoleuca*	Yellow thornbill [a]	*Acanthiza nana*
Scarlet robin [a]	*Petroica boodang*	Yellow-faced honeyeater [a]	*Lichenostomus chrysops*
Silvereye [a]	*Zosterops lateralis*	Yellow-rumped thornbill [a]	*Acanthiza chrysorrhoa*
Skylark*,**	*Alauda arvensis*		
Southern whiteface [a]	*Aphelocephala leucopsis*		
Speckled warbler [a]	*Chthonicola sagittata*		
Spotted pardalote [a]	*Pardalotus punctatus*		
Striated pardalote	*Pardalotus striatus*		
Striated thornbill [a]	*Acanthiza lineata*		
Stubble quail	*Coturnix pectoralis*		
Sulpher-crested cockatoo	*Cacatua galerita*		
Superb fairy-wren [a]	*Malurus cyaneus*		
Superb parrot [a]	*Polytelis swainsonii*		
Tree martin	*Petrochelidon nigricans*		
Varied sitella [a]	*Daphoenositta chrysoptera*		
Wedge-tailed eagle*	*Aquila audax*		
Weebill [a]	*Smicrornis brevirostris*		
Welcome swallow	*Hirundo neoxena*		
Western gerygone [a]	*Gerygone fusca*		
White-browed babbler [a]	*Pomatostomus superciliosus*		
White-browed scrubwren [a]	*Sericornis frontalis*		
White-browed woodswallow [a]	*Artamus superciliosus*		
White-eared honeyeater [a]	*Lichenostomus leucotis*		
White-faced heron	*Egretta novaehollandiae*		
White-plumed honeyeater [a]	*Lichenostomus penicillatus*		
White-throated gerygone [a]	*Gerygone albogularis*		
White-throated treecreeper [a]	*Cormobates leucophaea*		
White-winged chough [a]	*Corcorax melanorhamphos*		

Paper VIII. Evaluating complementary networks of restoration plantings for landscape-scale occurrence of temporally dynamic species.

While the two preceding papers compare the cost-effectiveness of remnant protection and restoration planting. In Paper VII I focused on the latter in a demonstration of the potential benefits of integrating economic data in a systematic conservation planning approach to ecological restoration in agricultural landscapes. This study asked whether a dynamic complementarity approach can achieve more cost-effective conservation outcomes, and what planting attributes are critical to achieving conservation objectives for threatened and declining woodland birds.

Photo: D. Ansell

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Conservation Biology

Contributed Paper

Evaluating complementary networks of restoration plantings for landscape-scale occurrence of temporally dynamic species

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Abstract: *Multibillion dollar investments in land restoration make it critical that conservation goals are achieved cost-effectively. Approaches developed for systematic conservation planning offer opportunities to evaluate landscape-scale, temporally dynamic biodiversity outcomes from restoration and improve on traditional approaches that focus on the most species-rich plantings. We investigated whether it is possible to apply a complementarity-based approach to evaluate the extent to which an existing network of restoration plantings meets representation targets. Using a case study of woodland birds of conservation concern in southeastern Australia, we compared complementarity-based selections of plantings based on temporally dynamic species occurrences with selections based on static species occurrences and selections based on ranking plantings by species richness. The dynamic complementarity approach, which incorporated species occurrences over 5 years, resulted in higher species occurrences and proportion of targets met compared with the static complementarity approach, in which species occurrences were taken at a single point in time. For equivalent cost, the dynamic complementarity approach also always resulted in higher average minimum percent occurrence of species maintained through time and a higher proportion of the bird community meeting representation targets compared with the species-richness approach. Plantings selected under the complementarity approaches represented the full range of planting attributes, whereas those selected under the species-richness approach were larger in size. Our results suggest that future restoration policy should not attempt to achieve all conservation goals within individual plantings, but should instead capitalize on restoration opportunities as they arise to achieve collective value of multiple plantings across the landscape. Networks of restoration plantings with complementary attributes of age, size, vegetation structure, and landscape context lead to considerably better outcomes than conventional restoration objectives of site-scale species richness and are crucial for allocating restoration investment wisely to reach desired conservation goals.*

Keywords: agrienvironmental schemes, complementarity, dynamic distributions, Marxan, spatial prioritization, systematic conservation planning, vegetation restoration, woodland birds

Evaluación de Redes Complementarias de Plantaciones de Restauración para la Ocurrencia a Escala de Paisaje de Especies Temporalmente Dinámicas

Resumen: *Las inversiones multimillonarias de dólares en la restauración de suelos hacen que los objetivos de conservación se obtengan de manera rentable. Las estrategias desarrolladas para la planeación estratégica de la conservación ofrecen oportunidades para evaluar a escala de paisaje los resultados de biodiversidad temporalmente dinámica obtenidos de la restauración y para mejorar las estrategias tradicionales que se*

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enfocan en las plantaciones más ricas en especies. Investigamos si es posible aplicar una estrategia basada en la complementariedad para evaluar la extensión con la que una red de plantaciones de restauración cumple con los objetivos de representación. Con el uso de un estudio de caso de aves de bosque de importancia para la conservación en el sureste de Australia, comparamos las selecciones basadas en la complementariedad de las plantaciones basadas en la ocurrencia de especies dinámicas temporalmente con las selecciones basadas en la ocurrencia de especies estáticas y con las selecciones basadas en la clasificación por riqueza de especies de las plantaciones. La estrategia de complementariedad dinámica, que incorporó la ocurrencia de especies durante cinco años, resultó en una mayor ocurrencia de especies y en una mayor proporción de objetivos alcanzados en comparación con la estrategia de complementariedad estática, en la que la ocurrencia de las especies fue tomada en un punto único en el tiempo. En la equivalencia de costos, la estrategia de complementariedad dinámica también resultó siempre en un mayor porcentaje promedio mínimo de ocurrencia de especies mantenido en el tiempo y en una proporción mayor de la comunidad de aves que cumplían con los objetivos de representación en comparación con la estrategia de riqueza de especies. Las plantaciones seleccionadas bajo las estrategias de complementariedad representaron la extensión completa de atributos de plantación, mientras que aquellas seleccionadas bajo la estrategia de riqueza de especies tuvieron un mayor tamaño. Nuestros resultados sugieren que las futuras políticas de restauración no deberían intentar alcanzar todos los objetivos de conservación dentro de plantaciones individuales, sino que en su lugar, deberían capitalizar con las oportunidades de restauración conforme surgen para obtener así el valor colectivo de las plantaciones múltiples a lo largo del paisaje. Las redes de plantaciones de restauración con los atributos complementarios de edad, tamaño, estructura de la vegetación y contexto del paisaje llevan a resultados considerablemente mejores que los objetivos convencionales de restauración de riqueza de especies a escala de sitio y son cruciales para asignar sabiamente las inversiones de restauración para alcanzar los objetivos deseados de conservación.

Palabras Clave: aves de bosque, complementariedad, distribuciones dinámicas, esquemas agroambientales, Marxan, planeación sistemática de la conservación, priorización espacial, restauración de la vegetación

Introduction

Restoration plantings are a widely implemented approach to biodiversity conservation in agricultural landscapes (Bullock et al. 2011). Worldwide, international biodiversity targets for 2020 include the restoration of at least 15% of degraded ecosystems (Aichi Biodiversity Target 15, CBD COP 2010). Similar international targets for the restoration of 150 million ha of degraded lands by 2020 (Rio+20, UNCSD 2012) are estimated to cost nations globally US$18 billion per year (Menz et al. 2013). Specific restoration targets have been set by the European Union (Bullock et al. 2011) and individual countries, including Australia, whose Biodiversity Fund aims to restore 18 million ha of native vegetation by 2020 with a budget of US$1 billion (Australian Government 2013). Given this substantial investment, to maximize biodiversity outcomes it is important that restoration initiatives be both efficient and cost-effective (Menz et al. 2013).

Most research on biodiversity outcomes relative to restoration plantings (a form of active restoration) has focused on whether and how individual plantings achieve high levels of species occurrence, richness, or abundance (Munro et al. 2007). In addition to comparing the value of restoration plantings to that of reference sites (e.g., Gould et al. 2013), these studies have identified attributes of plantings that contribute to increased biodiversity at the site scale, including planting age (Vesk et al. 2008) and area and shape (Lindenmayer et al. 2010; Jellinek et al. 2014) and vegetation structure (Munro et al. 2011). This earlier research recommends that future restoration investment be focused on maximizing site-scale attributes related to high individual planting biodiversity. Restoration plantings, however, also have value at the landscape scale (Thomson et al. 2009; Rappaport et al. 2015), and the collective features of different plantings across the landscape may be a better measure of biodiversity value than site-scale attributes. An alternative approach to restoration investment, therefore, is to maximize the number of species present across the entire landscape through a focus on networks of restoration plantings.

Systematic conservation planning, originally developed for locating and designing cost-effective protected areas (Margules & Pressey 2000), is increasingly being used for spatial prioritization of new restoration areas (Thomson et al. 2009; Lethbridge et al. 2010; McBride et al. 2010; Wilson et al. 2011; Yoshioka et al. 2014; Possingham et al. 2015). A key concept is complementarity, which ensures that each restoration planting contributes unrepresented features to the larger network of plantings (i.e., each planting complements the others in the network) (Margules & Pressey 2000). Complementarity approaches to the selection of restoration plantings thus differ from selection based on traditional measures of conservation value that focus on the most species-rich plantings. This is because plantings with high individual species richness may not necessarily contribute to overall conservation goals of maximizing diversity at a landscape or regional scale (Margules & Pressey 2000; see also Chadès et al. 2015). Systematic conservation planning has frequently been used to evaluate the performance of an existing set of protected areas (e.g., Stewart et al. 2003), and the same approach might be useful to evaluate the performance of an existing network of restoration plantings. Undertaking such an evaluation would identify the best complementary subset of plantings that contribute the most to the biodiversity benefits of the network and might be afforded protection in cases of impact assessment and future landscape clearing and elucidate the attributes of plantings important for landscape-scale biodiversity outcomes. In doing so, landscapes undergoing restoration may have more efficient investment and conservation outcomes.

Most systematic conservation planning considers species occurrence only at a single point in time, but plant and animal communities (particularly those in disturbed landscapes) are temporally dynamic (Grantham et al. 2011; Runge et al. 2014; Tulloch et al. 2016). Failure to incorporate dynamics into spatial prioritizations (e.g., basing them on static species distributions derived from a single year of data or pooled over years) can lead to insufficient representation of species over time (Runge et al. 2016; Tulloch et al. 2016). Although previous spatial prioritizations for restoration have considered dynamics in the age and structural complexity of restored vegetation (e.g., Thomson et al. 2009), we are not aware of any spatial prioritization study that has accounted for temporal dynamics in the distribution or occurrence of species colonizing restoration plantings, an oversight that could undermine the success of restoration schemes.

We investigated whether a temporally dynamic complementarity approach can be used to evaluate the contribution of existing restoration plantings to achieve landscape-scale species occurrence. We used, as a case study, a network of plantings in the South West Slopes bioregion of southeastern Australia. Only 15% of this once-extensive temperate eucalypt woodland remains within this agricultural region (Benson 2008); consequently, many woodland bird species are of conservation concern (Rayner et al. 2014). Since 1990, however, concerted investment has been made to establish restoration plantings for a range of conservation- and land-management objectives, including increasing woodland bird habitat. Through extensive programs managed by multiple stakeholders, thousands of hectares of vegetation have been planted, corresponding to increases of 3–4% of vegetation cover in the landscape (Lindenmayer et al. 2012; Cunningham et al. 2014). As part of the South West Slopes Restoration Study (Cunningham et al. 2007), 61 plantings have been surveyed for birds and vegetation in 5 spring seasons since 2006.

Our first aim was to find the best complementary network (i.e., subset) of established restoration plantings to support landscape-scale occurrence of species of conservation concern for minimal establishment cost. We accounted for temporally dynamic species occurrences by requiring representation targets for species occurrence to be met in every year (Runge et al. 2016). We compared the outcomes of taking a dynamic complementarity approach to find a network of plantings that met our desired representation target with networks selected using a static complementarity approach based on single years of data and with networks of plantings of an equivalent cost ranked by richness of species of conservation concern.

Our second aim was to identify the attributes of plantings that contributed most to the landscape-scale occurrence of species of conservation concern. The plantings in our study were established for a variety of reasons (e.g., wind breaks, soil erosion, and salinity); differed in age, area, shape, vegetation structure, and landscape context; and subsequently differed in their individual value for woodland birds (Lindenmayer et al. 2010). This opportunistically created a network of plantings that was ideal for exploring how subsets of plantings with different characteristics differed in terms of their ability to represent all bird species of conservation concern. Thus, we sought to quantify the value of evaluating biodiversity benefits of management at the landscape scale and incorporating temporally dynamic species distributions into restoration planning. The work seeks to inform future investment to ensure more efficient and cost-effective biodiversity outcomes across restoration landscapes.

Methods

Study Area, Experimental Design, and Data Collection

The South West Slopes Restoration Study is a 150 × 120 km area of the South West Slopes bioregion of New South Wales, Australia (Fig. 1). This region was once dominated by temperate box-gum *Eucalyptus* woodland but is now characterized by cropping and livestock grazing. Farms typically have 3–35% native vegetation cover, including old-growth woodland, regrowth, and plantings (Cunningham et al. 2014). We used data from 61 plantings distributed across 25 farms. Typically, plantings were a mix of local endemic and widely distributed Australian ground cover, understory, and overstory species and plants were spaced about 2 m apart. For each planting, we compiled data on variables important for bird species richness and occurrence in restoration plantings: years since establishment, area and width of plantings, vegetation structural complexity, surrounding woody vegetation cover (a proxy for connectivity), and landscape position (Supporting Information).

We used the area and shape of plantings to estimate the total establishment cost of each planting. Our estimates were based on 2015 pricing rates used by Greening

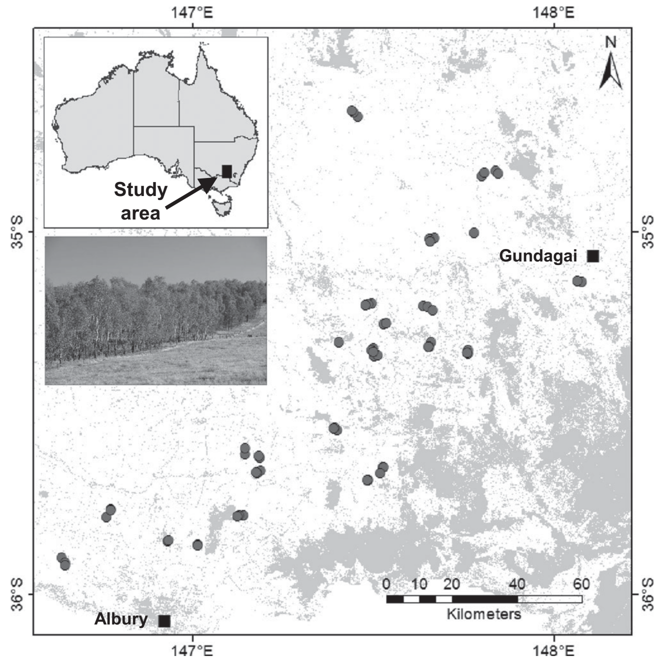

Figure 1. Map of study area showing restoration planting sites (points are not drawn to scale; gray shading, native woody vegetation cover). Insets show location of study area within Australia (top) and image of a typical planting site (bottom). Photo by D. Blair.

Australia, one of Australia's largest and longest running restoration practitioners. We calculated costs ($AU) of materials and labor for fencing and direct-seeding of sites (Supporting Information). Because our focus was on biodiversity as a public benefit, we considered only public costs of establishing restoration sites. We acknowledge the importance of considering private opportunity costs and ongoing management costs in conservation planning on public land, but the inclusion of such information was beyond the scope of this study.

We collected bird occurrence data in the spring seasons of 2006, 2008, 2009, 2011, and 2013. In each year, every planting was visited twice within 4 days in early November (2 days by different observers), and a 5-min point count was conducted at the 0-, 100-, and 200-m points of a permanent transect. All birds seen or heard within 50 m of the point, excluding those flying overhead, were recorded as present. Surveys were conducted between sunrise and midmorning on days when the weather was not inclement. This strict survey protocol was designed to address biases in observer heterogeneity (Cunningham et al. 1999) and false-negative errors (i.e., failure to detect species that are present [Banks-Leite et al. 2014]).

We defined woodland birds of conservation concern as those species dependent on woodland for foraging or nesting (Silcocks et al. 2005) and listed as threatened in New South Wales under the *Threatened Species Conservation Act 1995* (this also captured relevant nationally listed threatened species) or identified as having a >20% decrease in South West Slopes bioregion reporting rate between the first and second *Atlas of Australian Birds* (Barrett et al. 2003). Excluding very rare species (recorded only once during the 5 years), this definition resulted in 26 species of conservation concern for analysis (Supporting Information). We used permutational multivariate analysis of variance (PERMANOVA) to test for significant differences in species composition between years, based on a Bray–Curtis dissimilarity matrix adjusted for species presence–absence data, with the vegan package in R (R Development Core Team 2007).

Data Analyses

To identify targets of landscape-scale occurrence of species of conservation concern for the least investment, we compared the outcomes of using dynamic versus static complementarity approaches, and complementarity versus ranked approaches, to find the best subset network of restoration plantings. We set targets of 10–100% (in increments of 10) occurrence of each species per year in all years (equivalent to 10–100% of plantings where each species occurred in each year).

To find the best complementarity-based networks of plantings for each target, we used the decision-support software, Marxan, which uses a simulated annealing algorithm to solve the minimum-set problem (Ball et al. 2009). The objective was to minimize resources expended (i.e., cost of the planting network) while meeting prespecified representation targets (i.e., scenarios of 10–100% individual species occurrence per year in all years). To account for temporal variation in species occurrence between plantings (planning units), we created a conservation feature for each species for each survey year (5 conservation features per species of conservation concern for 130 conservation features in total), following Runge et al. (2016). Representation of conservation features in a given planting was based on presence–absence data (i.e., whether or not each species was recorded in each planting in each year). For each incremental increase in representation target, we compared the dynamic approach with 5 static approaches based on single years of data (i.e., 2006 only, 2008 only, 2009 only, 2011 only, and 2013 only). The objective of the static approaches was to meet representation targets only for that particular year. We parameterized Marxan to find the most cost-effective network irrespective of spatial configuration (by setting the boundary length modifier to zero) and performed 100 runs per scenario. We confirmed that the selected networks were not driven by planting cost by comparing the scenarios with baseline no-cost scenarios (Supporting Information). We considered 2 Marxan outputs for each scenario: the network of plantings that best met the representation target for the least cost and the selection frequency (i.e., irreplaceability) of each planting (the number of times each planting was selected across the 100 runs). For our scenarios, these 2 values were strongly positively correlated (≥ 0.9), and the average selection frequency for plantings selected in the best network was close to 100% (Supporting Information). Because this indicates high irreplaceability in selected plantings, we used the identified best networks of plantings for subsequent analyses.

We paired each dynamic complementarity scenario with a ranked scenario of equivalent cost, creating 10 matched pairs of networks (i.e., one for each species occurrence target [10–100%]). To do this, we calculated total richness of species of conservation concern across the 5 survey years and ranked individual plantings from high to low species richness. We calculated the cumulative cost of the plantings based on these rankings and included in the best network only those plantings that cost less than or the same as the cost of the dynamic complementarity scenario.

For each network selected by the dynamic, static, and ranked approaches, we calculated the cumulative establishment cost, number of plantings in the network, and summary statistics for the minimum percentage of the occurrence of each species that was met over the 5 years. We also calculated Bray–Curtis dissimilarity (adjusted for presence–absence data) between networks to assess spatial concordance between the selected plantings (e.g., low Bray–Curtis dissimilarity between a pair of networks indicates that the spatial locations of the plantings in the networks were similar). We confirmed that differences between the dynamic complementarity and ranked networks were not driven by cost-effectiveness by comparing our results with networks of equivalent cost that were based on ranking plantings by cost-effectiveness but ignoring complementarity (dividing species richness by cost) (Supporting Information).

To identify the attributes of plantings that contributed to landscape-scale occurrence of species of conservation concern, we modeled the relationship between planting attributes and the probability of the planting being selected in the dynamic and static complementarity and ranked scenarios for 2 representation targets (30% and 60% species occurrence in all years). We also modeled the number of times (frequency) each planting was selected in the static networks for these targets over the 5 years. The first target (30% occurrence) was chosen to reflect typical targets for conservation assessments (Svancara et al. 2005). The second target (60% occurrence) was chosen based on the results of the Marxan analyses because there was a threshold jump in planting benefits at this target level for the dynamic complementarity approach. Planting attributes included standardized site-level variables (Supporting Information). Planting width was strongly and positively correlated with planting area, so we excluded it from further analyses. We adopted an information-theoretic approach to model selection (Burnham & Anderson 2002) and compared a candidate set of 31 models that included single and additive combinations of all planting attributes (Supporting Information). We considered the univariate planting-area model the null model because previous research suggests that this attribute is of primary importance in restoration (e.g., Lindenmayer et al. 2010). We fitted generalized linear models with a binomial error distribution and log link (AICcmodavg package).We modeled each response variable against a distance-weighted spatial autocovariate (spdep package) to check for spatial autocorrelation between sites. For response variables that showed evidence of spatial autocorrelation, we included the distance-weighted spatial autocovariate in each alternative model. We ranked the candidate set of models with Akaike's information criterion corrected for small sample sizes (AICc). For top-ranked models (within 2 ΔAICc of the model with the lowest AICc), we assessed model support with Nagelkerke's coefficient of determination (R^2; fmsb package) and calculated model-averaged effect sizes for the model terms.

Results

Over the 5 survey years, we recorded 100 woodland bird species, including 26 of conservation concern (Supporting Information). Total richness of species of conservation concern ranged from 1 to 14 species per planting. Species composition differed significantly between years ($F = 2.723$, $p = 0.006$).

Dynamic Versus Static Complementary Restoration Planting Networks

The complementarity approach that incorporated dynamic species occurrences consistently resulted in higher mean minimum percent occurrence of species across the 5 survey years than the static complementarity approaches based on single years of data (Fig. 2a & Supporting Information). Although more expensive to achieve any given target than the static approaches, the dynamic approach always met the representation target for every species (Fig. 2b). In comparison, although plantings selected using a static single-year approach met the representation target for that year, they failed to meet the representation target over time (2006–2013) for more than one-third of species. This is because all 61 plantings were required to meet the dynamic representation target of 100% occurrence for each species over the period; for static targets, 42–54 plantings were required.

The spatial locations of the best network of selected plantings differed markedly between years. For example, for the 30% target, there was 44–78% Bray–Curtis

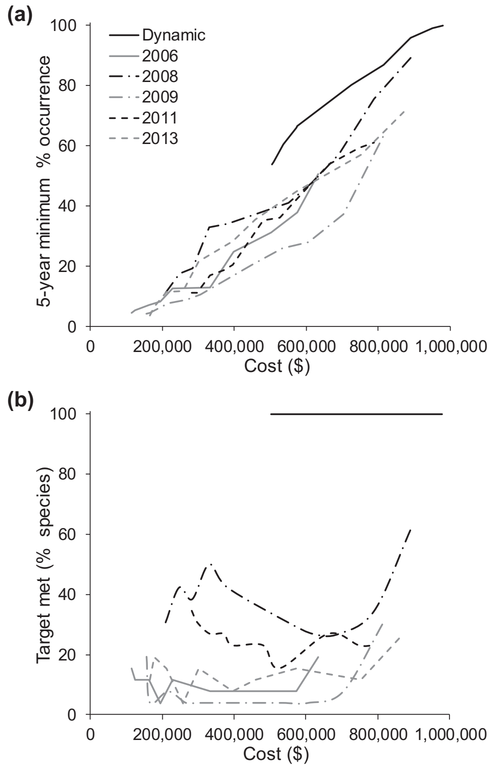

Figure 2. Comparison of 5-year outcomes under dynamic (incorporating species occurrences over 5 survey years) and static (based on single years of data, i.e., 2006, 2008, 2009, 2011, or 2013) complementarity approaches to maximize landscape-scale occurrence of species of conservation concern for (a) 5-year mean minimum percent species occurrence and (b) percentage of species meeting representation targets. Each line represents 10–100% representation targets.

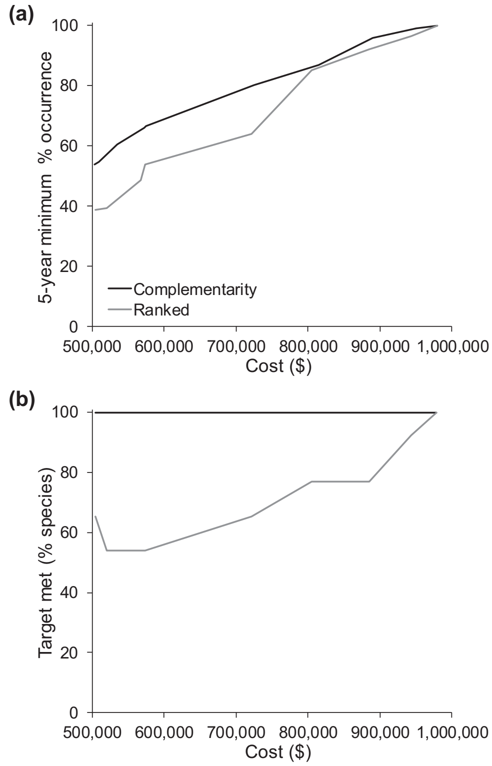

Figure 3. Comparison of dynamic complementarity (defined in the legend of Fig. 2) and species-richness ranked approaches to maximize landscape-scale occurrence of species of conservation concern for (a) 5-year mean minimum percent species occurrence and (b) percentage of species meeting representation targets. Each line represents 10–100% representation targets.

Table 1. Percent Bray–Curtis dissimilarity between the spatial locations of plantings selected in the static (2006 only, 2008 only, 2009 only, 2011 only, and 2013 only) and dynamic complementarity restoration planting networks for the representation target of 30% occurrence for each species of conservation concern.

Network	*2006*	*2008*	*2009*	*2011*	*2013*
Static					
2008	44.44				
2009	52.94	57.89			
2011	54.29	48.72	78.38		
2013	43.75	50.00	58.82	54.29	
Dynamic	50.00	38.46	36.00	45.10	41.67

dissimilarity in selected plantings between years (Table 1). To meet this target, each planting was selected an average of 1.46 times (out of 5 possible static networks); 20 plantings were never selected and only 1 planting was always selected. The spatial locations of the selected plantings also differed between the dynamic and static approaches (average Bray–Curtis dissimilarity for the 30% target = 49%). However, within each approach, plantings selected under low representation targets were usually also selected under higher targets (average Bray–Curtis dissimilarity between increments = 12%).

Dynamic Complementary Versus Ranked Restoration Planting Networks

The dynamic complementarity approach consistently resulted in higher mean percent species occurrence than the species-richness-ranked approach (Fig. 3a & Supporting Information). For equivalent cost, mean minimum percent occurrence of species was up to 30% higher in the complementarity scenarios. Further, although the representation target was achieved in every complementarity scenario (i.e., all species met the specified target), up to 46% of species did not meet the target in the equivalent-cost ranked scenarios (Fig. 3b). On average, there was 78% overlap in the spatial location of plantings

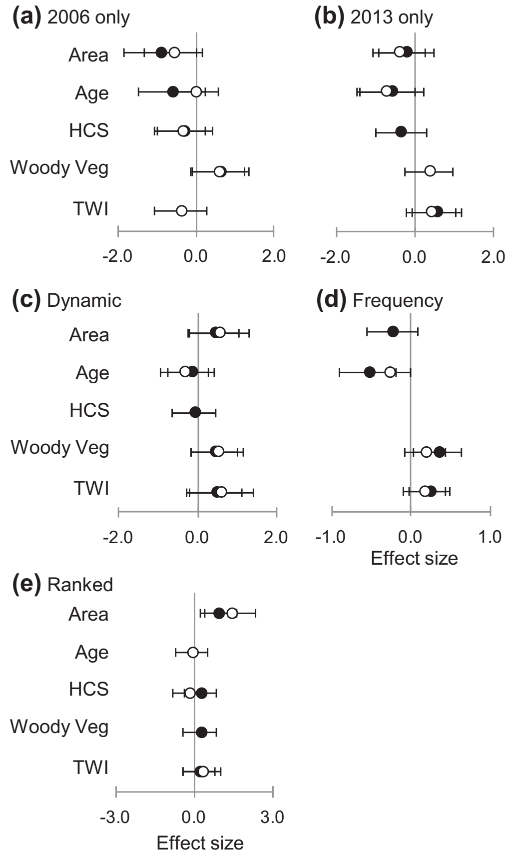

Figure 4. Summary of model-averaged effect sizes (and 95% CIs) for terms in the top-ranked models ($\Delta AIC_c \leq 2$) for 30% (closed circles) and 60% (open circles) representation targets. See Table 2 for a description of models and model terms. See Supporting Information for plots for 2008 only, 2009 only, and 2011 only models.

selected under the complementarity and ranked approaches (average Bray–Curtis dissimilarity = 22%).

Relationships with Planting Attributes

Plantings selected under the static and dynamic complementarity approach for the 30% and 60% targets did not consistently differ in their attributes from the plantings that were not selected. Model uncertainty was high because top-ranked models had relatively low R^2 values (Table 2). The effect sizes of terms in the models were generally small and variable (confidence intervals crossed 0) (Figs. 4a–c & Supporting Information). Similarly, there was no consistent relationship between the number of times each planting was selected in the static networks over the 5 years and planting attributes (Fig. 4d). Plantings selected more frequently to meet the 30% target were younger and surrounded by more woody vegetation cover, but effect sizes were small, and these effects were variable for plantings selected to meet the 60% target. In comparison, plantings selected under the richness-ranked scenarios were larger than plantings that were not selected, and model certainty was relatively high (Table 2, Fig. 4e).

Discussion

The restoration of degraded lands is an international conservation goal with multibillion dollar annual investments that require wise allocation of resources (Bullock et al. 2011; Menz et al. 2013). We found that it is possible to apply the principles of systematic conservation planning to evaluate the extent to which an existing network of restoration plantings meets representation targets for woodland birds of conservation concern. Incorporating dynamics in species occurrences across a 5-year period resulted in higher species occurrences and proportion of targets met compared with using species occurrences that represented a single point in time. Importantly, we found that for equivalent cost, the dynamic complementarity approach always resulted in higher average minimum percent occurrence of species maintained through time and a higher proportion of the bird community meeting representation targets compared with ranking plantings by species richness (aim 1). We also found that plantings selected to achieve goals of both representation and complementarity represented the full range of planting attributes, whereas plantings selected under the richness approach were larger (aim 2).

Incorporating dynamic species occurrences led to more expensive networks of restoration plantings but considerably higher long-term species occurrences and achievement of representation targets compared with static approaches. This was because the bird community was highly spatially and temporally dynamic, and there was little overlap between networks selected based on single years of data. Compared with static-distribution approaches, incorporating temporally dynamic species' ranges in systematic conservation planning led to more expensive and less flexible networks but improved biodiversity outcomes (see also Grantham et al. 2011; Lourival et al. 2011; Van Teeffelen et al. 2012). For example, Runge et al. (2016) found that accounting for annual and seasonal range variation in nomadic bird species leads to greater areas of land needing to be conserved to achieve targets but greater levels of species protection. Similarly, in their case study of the South American Pantanal wetlands, Lourival et al. (2011) found that incorporating dynamic vegetation distributions, although it increases expense, improves the reliability and long-term adequacy of their reserve networks. A dynamic prioritization approach is thus crucial for allocating investment wisely to reach desired conservation goals (Tulloch et al. 2016).

Table 2. Nagelkerke's coefficient of determination (R^2) for top-ranked models[a] of the relationship between planting attributes and the probability of plantings being selected in the dynamic and static complementarity networks and ranked networks and the frequency that plantings were selected in the static networks for the representation targets of 30% and 60% species occurrence.

		Static networks										*Dynamic networks*		*Frequency in static networks*[c]		*Ranked networks*	
		2006		*2008*		*2009*		*2011*		*2013*							
Model[b]	*Representation target (%)*	*30*	*60*	*30*	*60*	*30*	*60*	*30*	*60*	*30*	*60*	*30*	*60*	*30*	*60*	*30*	*60*
Area[d]		0.08*	0.04	0.00*	0.01*	0.01	0.00	0.04*	0.02*	0.01*	0.01*	0.05	0.06	0.17*	0.07*	0.20	0.27
Age					0.20	0.02	0.02	0.18	0.07	0.08	0.10	0.01	0.04		0.29		
Age + woody veg				0.15	0.21	0.02		0.23	0.10		0.11	0.07	0.10	0.36	0.31		
Area + age		0.14				0.04		0.19	0.13	0.10	0.14		0.08				0.27
TWI			0.04			0.01	0.00		0.06	0.09		0.03	0.03				
Age + TWI					0.21	0.02		0.20	0.11	0.14	0.13				0.31		
Age + woody veg + TWI								0.26	0.16		0.16	0.11	0.15	0.40	0.35		
Woody veg			0.04	0.09		0.00	0.00					0.05	0.06				
Area + TWI			0.10						0.10			0.10	0.12			0.21	0.28
Area + woody veg		0.15	0.12									0.08	0.09			0.21	
Woody veg + TWI				0.11					0.11			0.10	0.12				
Area + age + woody veg		0.22		0.16							0.18			0.40			
Area + age + TWI									0.14	0.15	0.16		0.13				
Area + woody veg + TWI			0.16						0.18			0.14	0.17				
Area + HCS							0.03									0.21	0.28
Age + HCS							0.03	0.20	0.12								
Age + HCS + woody veg				0.15				0.23	0.10								
Area + age + woody veg + TWI											0.20		0.18	0.43			
HCS						0.00	0.02										
Area + HCS + woody veg		0.20	0.17														
Age + HCS + TWI									0.15	0.16							
HCS + woody veg				0.12													
HCS + TWI										0.13							
HCS + woody veg + TWI												0.11					
Area + age + HCS + woody veg		0.23															
Age + HCS + woody veg + TWI									0.14								
Area + age + HCS + woody veg + TWI														0.44			

[a] *Models within $\Delta AICc \leq 2$.*
[b] *Definitions: area, planting area; age, years since planting establishment; HCS, habitat complexity score, which represents vegetation structural complexity; woody veg, percentage of vegetation cover within 1 km; TWI, topographic wetness index, which represents position in landscape ranging from ridges to valley floors.*
[c] *Spatial autocovariate included in all alternative models.*
[d] *Null model. An asterisk indicates the model was not within the top-ranked model set.*

Using a landscape-scale complementarity approach was critical to achieve cost-effective subsets of restoration plantings across the existing network. For example, to achieve similar species representation (for targets ≤90% species occurrence), the complementarity approach required less investment, fewer plantings, and less combined restored area than the ranked approach based on site-scale species richness. Further, even with the substantial additional investment needed for the ranked approaches, many species still did not meet the representation target in every year (compared with all targets achieved under the complementarity approach). Complementarity approaches to reserve design have long been recognized as superior to ranked approaches (Chadès et al. 2015), and our study supports their utility in restoration programs (Yoshioka et al. 2014). However, by definition, the high efficiency that complementarity achieves may result in low redundancy across the network of restoration sites, with implications for network robustness to disturbance (O'Hanley et al. 2007). In our scenarios, we incorporated multiple years of data, including from severe drought (2006–2009) and postdrought recovery (2011–2013). Incorporating these dynamics within our system likely reduced the trade-off between complementarity and robustness by accounting for stochastic processes (Lourival et al. 2011; Van Teeffelen et al. 2012).

Our finding that no single attribute makes plantings best for bird occupancy over space and time challenges conventional thinking that there is a type of restoration planting best for woodland birds (Lindenmayer et al. 2010). Instead, our findings support previous research on the differing and complementary suitability of plantings for different functional groups (Loyn et al. 2007). By collectively considering occupancy of plantings by each species in our analyses, we specifically accounted for the variable habitat requirements of our bird community. However, it is difficult to evaluate to what extent the bird occurrence patterns within the best networks of plantings were influenced by bird occurrence in unselected plantings (to which highly mobile taxa like birds could disperse) or by other vegetation types (e.g., regrowth and remnant vegetation) in the study landscape (Lindenmayer et al. 2012). Future research should investigate complementarity and connectivity between restoration plantings, regrowth vegetation (i.e., passive restoration), and remnant vegetation for landscape-scale species persistence. Future research could also integrate dynamics in planting attributes with dynamics in species occurrences. For instance, we held planting attributes constant, yet some attributes such as age, structural complexity, and connectivity may change through time (Thomson et al. 2009). As such, the attributes of plantings that are likely to maximize complementarity may also change through time, as suggested by our findings from our static models. Ideally, any future research that uses cost-effectiveness analysis to prioritize restored habitat in agricultural landscapes should also incorporate costs associated with lost farming opportunities in restored areas (Naidoo et al. 2006).

Translating our findings into future restoration policy involves some challenges. Our results show that it is desirable from a complementarity perspective to encourage a mixed portfolio of restoration projects that differ in the attributes of plantings and landscape context. Given real-world social, economic, and political constraints on biologically-driven conservation planning, "informed opportunism" (sensu Noss et al. 2002) may be appropriate. That is, in addition to available biodiversity knowledge, future investment in restoration initiatives should also be guided by the capacity and willingness of land owners to participate (Knight et al. 2010). A key difficulty is developing policy that can capitalize on informed opportunism to achieve complementary planting networks. One approach may be to implement policies that support consistent, incremental funding of restoration plantings in a region, so that a breadth of planting ages and structural attributes is maintained. Another more resource-intensive approach could be to allocate funding for new plantings that would complement the attributes of existing restoration plantings.

We found that a complementarity approach can be used to find the best network of established restoration plantings and that this network is more cost-effective and represents more of species' landscape occupancy than a traditional species-richness approach. Further, incorporating temporally dynamic species occurrences leads to a more cost-effective and robust restoration-plantings network compared with using static single-year data (Grantham et al. 2011; Lourival et al. 2011; Van Teeffelen et al. 2012; Runge et al. 2014). Substantial resources will continue to be invested in restoration initiatives in response to international and national policy and as part of wider agrienvironmental schemes (Bullock et al. 2011; Menz et al. 2013). This investment should not attempt to achieve all conservation goals within individual plantings but could instead be implemented incrementally to capitalize on restoration opportunities as they arise (Noss et al. 2002) to achieve collective value of multiple plantings across the landscape. Adopting a landscape-scale temporally dynamic approach leads to considerably better outcomes for a faunal community of conservation concern than applying conventional site-scale metrics and is crucial for the wise allocation of restoration investment to reach desired conservation goals.

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Supporting Information

Summary and definition of planting attributes (Appendix S1), habitat-complexity score calculation (Appendix S2), establishment costs of planting (Appendix S3), list of woodland bird species of conservation concern (Appendix S4), comparison with no-cost scenarios (Appendix S5), mean selection frequencies of selected plantings (Appendix S6), candidate set of models considered in model selection (Appendix S7), summary of dynamic and static complementarity scenarios (Appendix S8), summary of dynamic complementarity and ranked scenarios (Appendix S9), and summary of the 2008, 2009, and 2011 static complementarity models (Appendix S10) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Conservation Biology

Evaluating complementary networks of restoration plantings for landscape-scale occurrence of temporally dynamic species

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Supporting Information

Appendix S1: Summary of planting attributes considered in the analyses, and example of previous studies that have found the attribute to be important in explaining bird diversity in restoration plantings.

Planting attributes	Definition	Mean (Range)	Example studies
Cost	Establishment cost	$16, 052 ($4, 948 – $75,869)	Polyakov et al. 2015
Age	Number of years since the establishment of the planting (since 2006).	11 (0 – 44)	Lindenmayer et al. 2010 Munro et al. 2011
Area	Size of planting (ha).	4.24 (0.3 – 60.3)	Kavanagh et al. 2007 Lindenmayer et al. 2010 Munro et al. 2011
Width	Width of planting (m).	65.16 (10 – 300)	Kinross 2004 Lindenmayer et al. 2007 Lindenmayer et al. 2010 Munro et al. 2011
Habitat complexity score (HCS)	Vegetation structural complexity was based on vegetation data collected in 2007/08 and 2013: (i) the percent cover of overstorey, midstorey and understorey vegetation, the number of logs per ha, and the presence of large trees (> 50 cm diameter at breast height) were recorded within three 20 x 20 m plots located at the 0 m, 100 m	18 (9 – 29)	Lindenmayer et al. 2010 Munro et al. 2011

Planting attributes	Definition	Mean (Range)	Example studies
	and 200 m transect points; and (ii) the percent cover of native grass, exotic grass, exotic perennials, broadleaf weeds, forbs, leaf litter, and moss and lichen were recorded within twelve 1 m x 1 m quadrats located at the corners of the plots. A combined site-level habitat complexity score was calculated from these data, following Munro et al. (2011) (Table S2).		
Woody vegetation (WoodyVeg)	Percentage of vegetation cover within a 1 km buffer from the 100 m transect point. Derived from Landsat satellite imagery (Danaher 2011).	5.45% (0.00% – 23.00%)	Kavanagh et al. 2007 Lindenmayer et al. 2010 Munro et al. 2011 Radford et al. 2005
Topographic wetness index (TWI)	Position in landscape, ranging from ridge tops to valley floors. Derived from a 20 m resolution Digital Elevation Model (DEM) (Montague-Drake et al. 2011), and calculated at the 100 m transect point	0.61 (-2.68 – 10.23)	Lindenmayer et al. 2010 Montague-Drake et al. 2011

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Appendix S2: Habitat complexity score (HCS). Planting HCS was the sum of the scores for each element.

Score	Strata % cover*	Logs/ha	Trees > 50 cm/ha
0	< 1%	< 1	< 1
1	1-5%	1-10	
2	6-30%	11-50	
3	31-70%	51-100	
4	> 70%	> 100	≥ 1

*Strata includes overstorey, midstorey, understorey and ground layer (native tussock, exotic tussock, exotic grass, broadleaf weeds, forbs, and leaf litter).

Appendix S3: Costs of materials and labour for fencing and direct-seeding of restoration sites, based on 2015 pricing rates used by Greening Australia

Item	Description	Rate ($AU)
Fencing	Fencing materials and labour	$10,000/km
Direct-seeding - materials	Seed, machinery	< 2 ha = $750/ha
		2-4 ha = $625/ha
		>4 ha = $550/ha
Direct-seeding - labour	Labour, site preparation	$77.68/ha

Appendix S4: Woodland bird species of conservation concern, justification for inclusion and number of observations between 2006 and 2013. 'Legislation': listed as threatened in NSW under the *Threatened Species Conservation Act 1995* (this also captures relevant nationally-listed threatened species) or 'Atlas': identified as having a >20% decrease in South West Slopes bioregion reporting rate between the first and second Atlas of Australian Birds.

Common name	Scientific name	Source	Records
Black-chinned Honeyeater	Melithreptus gularis	Legislation	6
Brown Songlark	*Cincloramphus cruralis*	Atlas	56
Brown Treecreeper	*Climacteris picumnus*	Legislation	8
Cockatiel	*Nymphicus hollandicus*	Atlas	15
Crested Shrike-tit	*Falcunculus frontatus*	Atlas	30
Diamond Firetail	*Stagonopleura guttata*	Legislation	21
Dollarbird	*Eurystomus orientalis*	Atlas	2
Dusky Woodswallow	*Artamus cyanopterus*	Atlas	10
Fairy Martin	*Petrochelidon ariel*	Atlas	5
Grey-crowned Babbler	*Pomatostomus temporalis*	Legislation	11
Jacky Winter	*Microeca fascinans*	Atlas	3
Little Lorikeet	*Glossopsitta pusilla*	Legislation	3
Masked Woodswallow	*Artamus personatus*	Atlas	7
Pied Butcherbird	*Cracticus nigrogularis*	Atlas	5
Rainbow Bee-eater	*Merops ornatus*	Atlas	13
Restless Flycatcher	*Myiagra inquieta*	Atlas	9
Scarlet Robin	*Petroica boodang*	Legislation	2
Southern Whiteface	*Aphelocephala leucopsis*	Atlas	10
Speckled Warbler	*Chthonicola sagittata*	Legislation	9
Superb Parrot	*Polytelis swainsonii*	Legislation	19
Weebill	*Smicrornis brevirostris*	Atlas	66
White-browed Woodswallow	*Artamus superciliosus*	Atlas	54
White-fronted Chat	*Epthianura albifrons*	Legislation	8
White-winged Triller	*Lalage sueurii*	Atlas	46
Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*	Atlas	119
Zebra Finch	*Taeniopygia guttata*	Atlas	2

Appendix S5 Comparison of dynamic complementarity scenarios with cost included and excluded, for the representation targets of 30% and 60% species occurrence. The locations of plantings selected under the two scenarios were similar (Bray-Curtis dissimilarity 13% and 23% for the 30% target and 60% target, respectively).

Scenario	Cost	Plantings	Area (ha)	% Occurrence	Target met
30%: cost included	$535,125.80	32	185.00	60.63	100
30%: cost excluded	$591,778.60	30	203.80	60.28	100
60%: cost included	$725,628.00	43	222.20	80.19	100
60%: cost excluded	$754,132.20	42	227.50	80.53	100

Appendix S6. Mean (SD) selection frequencies of plantings selected the in the best solutions for each representation target under dynamic complementarity scenarios and those not selected.

	Best solution	
Target	Selected	Not selected
10%	98.71 (6.42)	1.06 (5.92)
20%	96.34 (13.32)	3.22 (9.90)
30%	96.88 (10.93)	3.55 (10.92)
40%	98.00 (7.91)	3.07 (19.93)
50%	98.06 (7.80)	3.19 (9.03)
60%	98.21 (7.56)	4.28 (11.40)
70%	96.25 (12.42)	13.92 (17.29)
80%	97.22 (11.57)	24.00 (21.76)
90%	99.62 (2.89)	19.67 (15.31)
100%	100.00 (0.00)	-

Appendix S7. Candidate set of models including single and additive combinations of all planting attributes. See Appendix S1 for explanation of planting attributes.

Models
Area
Age
HCS
Woody Veg
TWI
Area + Age
Area + HCS
Area + Woody Veg
Area + TWI
Age + HCS
Age + Woody Veg
Age + TWI
HCS + Woody Veg
HCS + TWI
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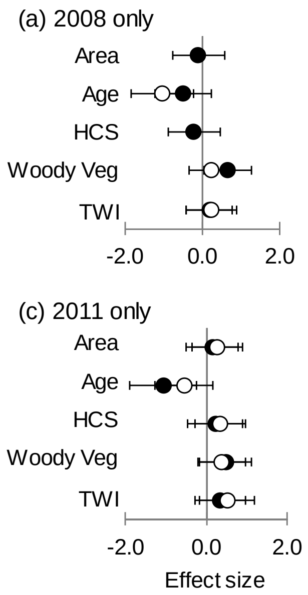
Appendix S8. Summary of dynamic and static complementarity scenarios

	Dynamic			2006			2008			2009			2011			2013		
Target (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)
10	$503,891	54	100	$114,068	5	15	$209,546	11	31	$156,488	4	19	$281,986	11	35	$166,364	4	12
20	$509,593	55	100	$123,890	5	12	$247,124	17	42	$166,002	5	4	$297,123	11	31	$177,061	6	19
30	$535,126	61	100	$163,907	7	12	$283,631	19	38	$216,123	8	8	$332,201	17	27	$212,421	11	15
40	$573,122	66	100	$195,243	8	4	$330,638	33	50	$266,335	9	4	$370,808	19	27	$257,570	12	4
50	$575,591	66	100	$229,085	13	12	$381,571	34	42	$305,878	10	4	$395,025	20	23	$300,573	22	15
60	$725,628	80	100	$332,518	13	8	$549,920	41	31	$448,961	21	4	$480,195	35	23	$395,403	28	8
70	$814,979	87	100	$397,461	25	8	$620,428	48	27	$534,510	26	4	$525,713	36	15	$459,828	36	12
80	$889,818	96	100	$503,252	31	8	$690,304	57	27	$606,834	28	4	$661,279	54	27	$572,794	45	15
90	$951,035	99	100	$574,287	38	8	$789,620	76	35	$704,481	37	8	$746,849	59	23	$751,666	57	12
100	$979,198	100	100	$633,418	51	19	$890,734	89	62	$816,398	64	31	$788,795	61	23	$870,880	71	27

Appendix S9. Summary of dynamic complementarity and ranked scenarios.

	Dynamic			Species-richness ranked			Species-richness / cost ranked		
Target (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)
10	$503,891	54	100	$504,144	39	65	$502,903	51	77
20	$509,593	55	100	$504,144	39	65	$502,903	51	77
30	$535,126	61	100	$520,595	39	54	$525,424	51	73
40	$573,122	66	100	$567,577	49	54	$565,378	55	81
50	$575,591	66	100	$574,061	54	54	$565,378	55	81
60	$725,628	80	100	$721,545	64	65	$701,886	73	69
70	$814,979	87	100	$805,461	85	77	$819,627	86	81
80	$889,818	96	100	$885,168	92	77	$865,478	91	88
90	$951,035	99	100	$943,044	96	92	$892,227	91	85
100	$979,198	100	100	$979,198	100	100	$979,198	100	100



Appendix S10. Summary of model-averaged effect sizes (and 95% CIs) for terms in the top-ranked models (ΔAICc≤ 2) for 30% (closed circles) and 60% (open circles) representation targets. See Appendix S1 for a description of model terms.

Paper IX How economics can further the success of ecological restoration.

As several papers in this thesis demonstrate, significant conservation benefits can be achieved through the integration of financial costs in the evaluation and planning of conservation investments. This, however, represents only a small part of the potential contribution that economics can make to improving conservation outcomes. In this final paper, I explored the broader economic principles and techniques that have the potential to improve the conservation outcomes and social benefits of ecological restoration, particularly within agricultural landscapes.

Photo: Greening Australia

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Conservation Biology

Review

How economics can further the success of ecological restoration

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Abstract: *Restoration scientists and practitioners have recently begun to include economic and social aspects in the design and investment decisions for restoration projects. With few exceptions, ecological restoration studies that include economics focus solely on evaluating costs of restoration projects. However, economic principles, tools, and instruments can be applied to a range of other factors that affect project success. We considered the relevance of applying economics to address 4 key challenges of ecological restoration: assessing social and economic benefits, estimating overall costs, project prioritization and selection, and long-term financing of restoration programs. We found it is uncommon to consider all types of benefits (such as nonmarket values) and costs (such as transaction costs) in restoration programs. Total benefit of a restoration project can be estimated using market prices and various nonmarket valuation techniques. Total cost of a project can be estimated using methods based on property or land-sale prices, such as hedonic pricing method and organizational surveys. Securing continuous (or long-term) funding is also vital to accomplishing restoration goals and can be achieved by establishing synergy with existing programs, public-private partnerships, and financing through taxation.*

Keywords: benefit transfer, environmental economics, hedonic pricing, nonmarket valuation, opportunity cost, project prioritization

Cómo la Economía puede Prolongar el Éxito de la Restauración Ecológica

Resumen: *Los científicos y quienes practican la restauración recientemente han comenzado a incluir aspectos sociales y económicos en el diseño y en las decisiones de inversión para los proyectos de restauración. Con pocas excepciones, los estudios de restauración ecológica que incluyen a la economía se enfocan solamente en la evaluación de los costos de los proyectos de restauración. Sin embargo, los principios, herramientas e instrumentos económicos pueden aplicarse a una gama de otros factores que afectan a la proyección del éxito. Consideramos la relevancia de aplicar la economía para señalar cuatros obstáculos clave que enfrenta la restauración ecológica: la valoración de los beneficios sociales y económicos, la estimación del costo total, la priorización y selección del proyecto y el financiamiento a largo plazo de los programas de restauración. Encontramos que no es común considerar todos los tipos de beneficios (como los valores intangibles) y costos (como el costo de transacción) en los programas de restauración. El beneficio total de un proyecto de restauración puede estimarse utilizando métodos basados en los precios de la propiedad o de venta de suelo, como el método de fijación hedónica de precios y las encuestas de organización. Asegurar el financiamiento continuo (o a largo-plazo) también es vital para cumplir los objetivos de restauración y puede alcanzarse al establecer una sinergia entre los programas existentes, las asociaciones público-privadas y el financiamiento por medio de impuestos.*

Palabras Clave: costo de oportunidad, economía ambiental, fijación hedónica de precios, priorización de proyectos, transferencia de beneficios, valoración de intangibles

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Introduction

Although the importance of ecosystem restoration as a conservation strategy is well established (Benayas et al. 2009), the design and implementation of restoration projects is complex and their effectiveness is highly variable. Jones and Schmitz (2009) reviewed 236 restoration case studies and found that two-thirds reported partial or no recovery. Benayas et al. (2009) conducted a meta-analysis of 89 studies of restoration projects and found that, even though restoration increased provision of biodiversity and ecosystem services by 44% and 25%, respectively, the values of biodiversity and ecosystem services of restored sites were below that of undisturbed reference sites. The variable effectiveness of restoration practices highlights the many challenges facing the successful implementation of ecological restoration (Maron et al. 2012). These challenges are ecological, technical, social, and economic in nature. We considered ways that economics can contribute to tackling these challenges.

Only recently have restoration scientists and practitioners begun to include economic aspects in the design of restoration projects (Blignaut et al. 2014). With few exceptions (e.g., Schultz et al. 2012; Groot et al. 2013), ecological restoration studies that include economics focus heavily on project cost (Adame et al. 2015). Although cost information is an important part of sound economic decisions in conservation, many other facets of economics can contribute to improving the effectiveness and efficiency of restoration programs (Yin et al. 2013).

We identified opportunities for the wider application of economic principles and tools in ecological restoration, from project planning to long-term financing. We considered how economic tools and principles can address key project challenges and improve effectiveness and efficiency of ecological restoration. Our aim is not to provide a comprehensive review of the literature; rather, we drew on previous research to identify ways in which economics may specifically address 4 key restoration issues: estimation of restoration benefits (Bullock et al. 2011), estimation of the costs of restoration (Armsworth 2014), selection and prioritization of projects (Miller & Hobbs 2007; Suding 2011), and securing long-term financial resources to support restoration (Holl & Howarth 2000; Bullock et al. 2011; Halme et al. 2013). We considered each challenge in detail (Table 1), how each could impede restoration success, and the opportunities for using economic principles and tools to address them.

Estimating Restoration Benefits

Restricting objectives of restoration projects to purely ecological benefits is justified in cases of a statutory requirement to conserve or restore an ecosystem or species. When public funding is involved, other benefits need to be considered; however, practitioners may fail to demonstrate the links between ecological restoration, society, and policy and may undersell the social benefits of restoration (Aronson et al. 2010; Wortley et al. 2013). Some studies show that private landholders are more likely to participate in restoration projects if they benefit financially or nonfinancially (Januchowski-Hartley et al. 2012). Therefore, consideration of broader social and economic benefits of restoration may help practitioners tailor their programs to promote better engagement (Aronson & Alexander 2013).

Knowing the social and economic benefits of restoration could be particularly useful when seeking cooperation from a private landholder. Landholder cooperation can be vital because conservation-agency budgets are constrained and the substantial opportunity cost of restoration can present a barrier to restoration (House et al. 2008). Program designs that reduce economic costs to landholders (e.g., by providing financial incentives) may facilitate restoration in areas that have been traditionally difficult to access by conservation managers (Ansell et al. 2016). Although such an approach may necessitate some compromise in restoration design (e.g., size or location of project), it could lead to higher social acceptability and higher overall environmental gains relative to no restoration (Petursdottir et al. 2013).

A key challenge to incorporating social benefits in planning and selection of restoration projects is how to assess them. Several economic methods are available for assessment of ecosystem services and other social benefits. The method applied depends on the type of value likely to be produced by the project. Market-based methods are generally not applicable because most of these values are not traded in formal markets (i.e., nonmarket values). These nonmarket values have either a use value (e.g., recreation) or a nonuse value (e.g., preserving a threatened species for future generations). Revealed-preference approaches are applied to measure use values, and stated-preference approaches are applied to nonuse values (Whitehead et al. 2008).

Revealed-preference approaches, such as hedonic-pricing and travel-cost methods, use observed behavior to estimate an individual's willingness to pay for goods or services (Whitehead et al. 2008). The hedonic-pricing method uses heterogeneous goods that are sold in a market, such as land, houses, or cars, to determine the values of key underlying characteristics of these goods, including values of environmental assets (Taylor 2003). The travel-cost method is used to estimate the benefits of outdoor recreation based on the assumption that costs of travel and time are the main costs of outdoor recreation (Whitehead et al. 2008).

The stated-preference approaches include contingent valuation and choice experiments and use hypothetical data, typically from community surveys, to estimate

Table 1. A summary of the common challenges to ecological restoration and the economic principals, tools, and instruments available to address them.

Challenges	*Potential reasons for not meeting the challenge*	*Consequence of not meeting the challenges*	*Economic principles, tools, and instruments*
Assessing benefits	narrow focus in program scope (i.e., some benefits or preferences excluded); lack of familiarity and skills with nonmarket valuation techniques in conservation agencies; limited funding	suboptimal project selection; lack of political or community support, resistance during and after implementation	nonmarket valuation techniques; benefit transfer technique when adequate resources to conduct primary nonmarket valuation studies are not available
Assessing costs	lack of understanding of types of costs and their importance; difficulty in estimating types of costs; limited funding available	suboptimal project selection; failure to complete project due to insufficient financing; discontinuation of funding as the project becomes financially unviable	capitalized gross revenue or gross margin of the productive use of land; methods based on property or land sales prices, such as hedonic pricing method; estimation of transaction costs based on organizational surveys; collecting and publishing establishment costs
Prioritization and targeting	inadequate information on benefits and costs; using incorrect metrics; failure to capture all the elements of the decision-making process during prioritization	wasting of valuable public resources; failure to meet environmental targets; negative or unintended environmental consequences	selection and use of appropriate metric; use of comprehensive prioritization protocol; real-option analysis
Long-term financing	inadequate information on net benefits; wrong project selection	project becomes financially unviable due to lack of funding	working with existing funding arrangement; developing synergy with existing programs; financing through taxation; public–private partnership; offsetting; volunteerism

individuals' willingness to pay for the gain or avoided loss in the value of a public good or service. In choice experiments, respondents are presented with options each of which specifies the attributes of a project and the amount of money one would pay to achieve that option. The choices made by the respondents are used to estimate an individual's willingness-to-pay and to aggregate value of the nonmarket good to society.

Another approach is to use benefit transfer, where the results from the existing primary valuation studies are used to predict the values of benefits or services in a new area (Rolfe et al. 2015). The decision to use benefit transfer instead of a primary study depends on availability of valuation data for the policy site or for similar policy sites and whether decision makers require exact valuation data for the policy site or can use approximations (Holland et al. 2010). If carefully conducted, benefit transfers may provide a reasonable approximation of the value of unstudied resources. (Johnston et al. [2015] provide a comprehensive guide to benefit transfer.) Many high-profile environmental policies in the United States (Loomis 2015) and Europe (Brouwer & Navrud 2015) and some in Australia and New Zealand (Rolfe et al. 2015) used benefit transfer to estimate nonmarket values of intangible benefits.

Estimating nonmarket values of intangible benefits is useful in planning a restoration project. First, it provides a broad understanding of the value of the monetary investment in the restoration. For example, the restoration of grassland bird populations through the Conservation Reserve Program in the United States is estimated to generate US$33 million/year in nonuse (or existence) value (Ahearn et al. 2006). Second, it allows direct comparison of expected benefits and expected costs (which are rarely expressed in nonmonetary terms). Finally, it helps demonstrate the distribution of benefits among types of stakeholders.

The application of nonmarket valuation techniques requires careful planning and judgment. They are often expensive and require specific skills. Each of the above-mentioned methods has strengths and weaknesses (Bateman et al. 2002; Kanninen 2007). Nonmarket valuation may not be suitable for all restoration projects, particularly small projects (Rogers et al. 2015) or projects designed only to provide ecological benefits. However, in many instances, it would be beneficial for agencies to consider nonmarket values (which includes ecological benefits) of restoration programs.

Estimating Cost of Ecological Restoration

Cost information is important for ecological restoration planning because it informs decisions on whether to conserve or to restore, which projects to pursue, and which methods to use. Inappropriate accounting for costs during planning could waste public funds and result in failure to select the best projects. Accounting for costs is especially important when multiple methods with different costs could be used to achieve the desired restoration outcome. However, restoration costs are rarely reported by ecological restoration studies, published cost data are often collected using different approaches, making them hard to compare (Bullock et al. 2011), and sometimes not all types of costs are considered during planning (Pastorok et al. 1997; Groot et al. 2013). Acquisition, establishment, maintenance, and transaction are the 4 main costs in restoration.

Acquisition costs are the costs of acquiring the property rights to the land to be restored. The acquired rights could be total (e.g., a parcel is purchased outright) or partial. Partial costs include purchasing some of the property rights, as for conservation easements, covenants, or restrictions (Kabii & Horwitz 2006), or purchasing rights for a specific period, such as for conservation contracts. When a formal acquisition is not required, for example to restore public lands, the allocation of land to ecological restoration or protection still incurs an acquisition cost to society in the form of an opportunity cost. Opportunity cost is a measure of what could have been gained via the next-best use of land. Opportunity costs are often used to estimate landholders' compensation when conservation is conducted on private lands, particularly agricultural land (Mewes et al. 2015).

An important feature of acquisition costs and opportunity costs is their variability among properties (Armsworth 2014), which is caused by heterogeneity or fragmentation of land quality, land use, and ownership. At a regional scale, as land is used up for restoration, the acquisition costs of the remaining land increase due to increasing demand (Jantke & Schneider 2011). At a local scale, ecological restoration on one property may change values of neighboring properties (Butsic et al. 2013). At a property scale, when a fraction of a property is being acquired for restoration, acquisition cost of each additional unit of land may be higher than the cost of the previous unit due to the diminishing marginal benefit of land (Polyakov et al. 2015). Heterogeneity of acquisition costs could influence outcomes of a restoration program. For example, if heterogeneity between different properties is not considered, the financial incentive rate paid to private landholders could be set too low or too high. The former will result in lower participation and the latter will result in cost-ineffective outcomes (Iftekhar et al. 2012).

Establishment costs are upfront capital investments in restoration and, depending on the project, could include engineering works (e.g., in mine site or wetlands restoration), site preparation, planting or seeding, and fencing. Such costs are often highly variable but have not received sufficient attention. In the literature, they are usually considered part of management costs (Naidoo et al. 2006). However, these should be treated separately in the evaluation of ecological restoration because they could constitute a substantial portion of total costs. The costs depend on the type of ecosystem being restored, level of modification of the site, selected methods, and biophysical conditions. Even aspects of the design of individual sites, such as shape, can have a substantial influence on cost and cost-effectiveness of a project (Ansell et al. 2016).

Maintenance costs include ongoing management, administration, and monitoring. Ongoing management, such as the upkeep of fencing to control of invasive and feral species, is crucial for the project to succeed and is often neglected when estimating costs of projects. Monitoring the outcomes of restoration provides the basis for assessing the performance of restoration interventions and informs funders and society of the results. Ironically, this element of the project cycle is often poorly funded and conducted (Nichols & Williams 2006). As a result, there is growing attention paid to improving the rigor of monitoring and its cost-effectiveness (e.g., Lindenmayer et al. 2012).

Transaction costs may include searching for suitable sites, organizing programs, and negotiating and signing contracts. Transaction costs are especially important for ecological restoration because they represent a large upfront investment and may provide a barrier to otherwise feasible restoration projects. However, these costs are often omitted in the evaluation of environmental programs (McCann et al. 2005). This is a critical omission because transaction costs could range from 20% to 50% of total program costs (Coggan et al. 2010).

Different economic tools are used to estimate different types of costs. Establishment and maintenance costs are often easiest to estimate because market prices are available for most items in these cost categories. Acquisition costs and opportunity costs are estimated using capitalized gross revenue or gross margin of the productive use of land or using methods based on property prices. When entire properties are acquired for restoration, methods based on property values are more appropriate because they capture all the values associated with the property beyond its productive (e.g., agricultural) value, such as amenity values. Where partial property rights are acquired, methods based on estimation of the value of foregone benefits are appropriate because the owner retains some of the rights to the property. Transaction costs can be estimated by conducting surveys among the participating landholders or agencies and reviewing documents (Falconer & Saunders 2002).

Prioritizing Restoration Projects

Project funding in many countries follows a democratic process; a project analyst provides information on each

Table 2. A hypothetical example of the impact of using information on the partial cost of restoration projects in the calculation of cost-benefit ratios of individual projects.

Project	*Benefit (\$)*[a]	*Cost (\$)*[a]	*Partial cost (\$)*[a]	*Partial cost-benefit ratio*	*Cost-benefit ratio*
1	40	75	60	1.50[b]	1.88[c]
2	10	25	20	2.00[b]	2.50[c]
3	35	90	72	2.06[b]	2.57[c]
4	20	60	48	2.40[b]	3.00
5	30	150	120	4.00	5.00
6	15	80	64	4.27	5.33

[a] *Benefits and costs are in tens of thousands.*
[b] *Projects selected under partial cost-benefit ratios.*
[c] *Projects selected under full cost-benefit ratios.*

project option so that the decision maker can make a well-supported judgment about a projects' social desirability (Nyborg 2012). Benefit-cost analysis provides information on the efficiency and social welfare of a project. It informs decision makers about the projects that will lead to the greatest net benefits to the community as a whole.

Efficient project prioritization relies on the accurate identification and estimation of benefits and costs. The potential loss from not considering all the benefits and costs is demonstrated in the following hypothetical example (Table 2). Assume a public agency has a fixed budget (\$200,000) and can select only a subset of restoration projects from a set of projects. Each project has an estimate of expected environmental benefits and costs. The agency uses a cost-benefit ratio criterion to select projects. The project with the lowest cost-benefit ratio is selected first followed by the project with second lowest cost-benefit ratio and so on. Assume the agency does not have full information about the cost and uses the partial-cost information to calculate cost-benefit ratio. If the partial cost-benefit ratio is used, then the first 4 projects would be selected within the \$200,000 budget. However, if the true cost is used to calculate the cost-benefit ratio, then only the 3 projects with the lowest cost-benefit ratio would be selected.

Therefore, by using the partial-cost information, the agency assumes the total expenditure may be higher than the allocated budget, and there are 2 possible outcomes: The agency finds additional money to implement the selected projects or some projects are not completed to reduce expenditure. The former makes a restoration program expensive and socially inefficient and the latter may result in not achieving the objectives, which is also cost-ineffective.

Projects with low cost may be selected even if partial-cost information is used, and high-cost projects have little chance of being selected. The medium-cost projects are the most sensitive to the cost estimates. Empirically, Carwardine et al. (2010) showed that the impact of cost-data uncertainty on the prioritization of sites for conservation largely depends on the importance of the projects to achieving conservation goals. The sites essential or unimportant for meeting conservation goals maintained high or low priorities, respectively, regardless of cost estimates. Sites of intermediate conservation priority were sensitive to cost-data uncertainty: These represented the best option for efficiency gains.

When the benefits and, to a lesser extent, the costs are uncertain, delaying restoration to reduce uncertainty could allow achieving a more cost-effective allocation of funding across competing projects (Nelson et al. 2013). The benefit of delaying investment in the face of uncertainty could be assessed using real-option analysis (Majd & Pindyck 1987; Regan et al. 2015), which has been used to prioritize and rank conservation (Kassar & Lasserre 2004; Ben Abdallah & Lasserre 2012) and restoration projects (Leroux & Whitten 2014).

Once the costs and benefits have been appropriately measured, the choice among projects requires a metric, which is a formula or a model to translate the various parameters of a project (such as cost, effectiveness, and area) into a single score. Pannell and Gibson (2016) provide an empirical example of the importance of using a theoretically correct metric to select conservation projects. They found that environmental losses from a poorly designed metric could be up to 80% relative to the situation when a theoretically correct metric is used. The most costly metric errors are omitting information about environmental values, project costs, and effectiveness of management actions and using a weighted-additive decision metric for variables that should be multiplied.

The use of a rigorously designed metric is even more important when combining multiple benefits. Although restoration may generate ecological, economic, and social benefits, the relationship between these benefits may be complex and conflicting (Bullock et al. 2011). Restoration strategies that target these multiple benefits may therefore necessitate trade-offs in one or more of those values. Concessions may be required in the location, design, and complexity of restoration projects to achieve broader benefits. The acceptability of such a trade-off is likely to vary between restoration projects and depends on factors such as project outcomes specified by regulatory or funding bodies, threat status of the biodiversity asset, and value of the biodiversity asset to the community.

Long-Term Financing of Restoration Projects

Even when restoration benefits and costs have been correctly assessed and appropriate prioritization procedures employed, without adequate financial support failure is possible, particularly for long-term (decades) projects (Jones & Schmitz 2009). There are examples of long-running environmental programs (Conservation

Reserve Program in the United States was established over 30 years ago), but in most cases environmental programs have short funding time frames. For example, the environmental and restoration projects funded under market-based initiative programs (e.g., BushTender and EcoTender in Australia) lasted 3–5 years (Iftekhar et al. 2009). Therefore, it might be useful for agencies to consider innovative solutions to securing long-term funding, an issue considered by some as one of the greatest hurdles to restoration (Aronson & Alexander 2013). Long-term funding could be maintained by working within existing funding arrangements; developing synergy among existing programs; financing through property taxes; and developing public–private partnerships and through volunteerism.

Funding through existing government mechanisms could be a cost-effective way to fund long-term projects. For example, the Environmental Stewardship Program in Australia invested AU$152.3 million in long-term contracts (up to 15 years) with private landholders to provide ongoing agrienvironmental services (Lindenmayer et al. 2012). Implementation of long-term contracts presents a substantial departure from previous funding models for conservation in Australia, where conservation was characterized historically by the disbursement of public funds for conservation through regional and local government and nongovernment bodies (Hajkowicz 2009). Although programs are funded through existing government budgets, a key aspect of the Environmental Stewardship Program is that funds are secured beyond the forward estimates of the government. Securing funds through establishment of specific beyond-government accounts is challenging but, critically, allows for enduring action on long-term environmental programs.

Undertaking restoration programs in isolation may be costly. Agencies could consider restoration activities in combination with other activities (such as habitat protection, eradication of invasive species, carbon credits, etc.). For example, programs for eradication of invasive mammals may be combined with restoration of seabird populations (Kappes & Jones 2014), 2 programs that are commonly undertaken separately. Kappes and Jones (2014) showed that combining these programs improved effectiveness and enabled access to greater funding opportunities. Matzek et al. (2015) found that carbon credits alone can cover the establishment and maintenance costs of riparian restoration provided that sufficient effort is committed in the first few years of the program. However, it may not be sufficient to cover the opportunity costs of private landholders and funding from other sources may be required, further illustrating the importance of accounting for the full range of restoration costs.

Environmental restoration can improve the well-being of communities (Pressey et al. 2002). Such improvements are often reflected in increased house prices (Polyakov et al. 2016). Local governments typically collect property taxes based on property values. Implementation of restoration projects may result in the rise of local government revenues due to increase in house prices. Local governments could use this additional money to fund restoration programs. If the restoration program is large, local governments could borrow money against the expected increase in tax to finance restoration (Paull & Lewis 2008). However, for the restoration to have impact on property prices, it should be near residential areas.

Private investors can contribute substantially to restoration programs. Around 23% of the funding provided for the River Network, a U.S.-based association of 2000 organizations, was sourced from corporate funders (BenDor et al. 2015). Private investors and commercial enterprises invest in restoration programs to meet regulatory requirements, to meet corporate social responsibility, as an investment mechanism to earn profit, to save money, and to improve brand profile (Videras & Alberini 2000; BenDor et al. 2015). Being largely immune to the short-term political cycles and public pressures on competing policy priorities, corporate sponsors may provide a secure and flexible source of restoration funding.

Payment for ecosystem services (PES) is a concept often used to solicit private investment in restoration. New York City invested US$1.5 billion in watershed conservation to avoid large infrastructure projects (McPhearson et al. 2014). A wastewater utility in Oregon (U.S.A.) paid landholders to plant trees in riparian areas to reduce the warming effect from solar radiation (Bennett et al. 2014). Under the United Nations Reducing Emissions from Deforestation and Forest Degradation (REDD+) program, companies have formed partnerships with nongovernmental organizations, government agencies, and local communities to protect forests. These PES programs are not without risks. Long-term sustainability is a primary concern (Bullock et al. 2011), particularly when schemes are completed and landholders revert to original land uses.

Private landholders often undertake conservation programs on their land for personal reasons, such as a sense of stewardship. Landholders' intrinsic motivations have been identified as one of the primary reasons for their participation in environmental programs (Greiner 2015). People also contribute money and labor to many environmental programs such as revegetation (Langenfeld 2009). Crowdfunding, where individuals donate money to specific projects, has been recently tested to generate funding for environmental projects (Hörisch 2015). It may be possible for agencies to generate funding and manpower for restoration programs by appealing to the philanthropic nature of individuals.

Conclusions

There have been several attempts in recent years to highlight the benefits of incorporating economics into ecological restoration. To date, this literature has largely focused on the use of cost information in the spatial planning of restoration projects. We have identified 3 additional areas where economic principles and tools may be useful: assessment of benefits, project prioritization, and long-term funding. We found that the appropriate selection of a project depends on a rigorous assessment of the benefits and costs of the programs. Although ecological-benefit assessment tools are commonly used, proper assessment of economic and social benefits can also be important. Nonmarket valuation techniques may be useful in determining appropriate social values. We also found that even though some costs are generally included in decision making, others are not (such as transaction costs). The use of a rigorous prioritization tool that encompasses all relevant benefits and costs is thus very important. Failing to capture the full suite of benefits and costs, one risks undervaluing restoration and making poor investment decisions.

An additional challenge for conservation agencies is securing continuous or long-term funding to achieve restoration goals. Environmental programs of short duration may be inadequate to achieve restoration goals. The strategies we suggest could be used to secure additional or long-term funding. The suitability of different funding arrangements depends on the restoration program and needs to be examined before application. In essence, the sound application of economic principles and tools we discussed here can help in planning and successful implementation of restoration programs globally.

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