The conservation value of Australia's Stock Route Network: A multi-taxonomic approach to management and planning

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Cover image: Stock on the move at the “Moora Moora West” TSR, Driftway Rd, Warroo, NSW. Image: P. Lentini.

Declaration

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

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Preface

This thesis is structured as a series of six papers which have been either published or are in review. Journal formatting has been retained throughout the thesis, and the corresponding references for each of the papers are listed on the title pages. These are as follows:

Paper I: Lentini, PE, Fischer, J, Gibbons, P, Lindenmayer, DB & Martin, TG (2011) Australia's Stock Route Network: 1. A review of its values, and implications for future management. *Ecological Management and Restoration* **12**, 119-127.

Paper II: Lentini, PE, Fischer, J, Gibbons, P, Lindenmayer, DB & Martin, TG (2011) Australia's Stock Route Network: 2. Representation of fertile landscapes. *Ecological Management and Restoration* **12**, 149-151.

Paper III: Lentini, PE, Martin, TG, Gibbons, P, Fischer, J & Cunningham, SA (2012) Supporting wild pollinators in a temperate agricultural landscape: maintaining mosaics of natural features and production. *Biological Conservation* **149**, 84-92.

Paper IV: Lentini, PE, Fischer, J, Gibbons, P, Hanspach, J & Martin, TG (2011) Value of large-scale linear networks for bird conservation: A case study from travelling stock routes, Australia. *Agriculture, Ecosystems and Environment* **141**, 302-309.

Paper V: Lentini, PE, Gibbons, P, Fischer, J, Law, BS, Hanspach, J & Martin, TG (in review) Corridors and unimproved pastures are wildlife-friendly farming measures that will benefit microbats. *PLoS ONE*.

Paper VI: Lentini, PE, Gibbons, P, Fischer, J, Carwardine, J, Drielsma, M & Martin, TG (in review) The effect of planning for connectivity on linear reserve networks. *Conservation Biology*.

Because these papers have been or will be published independently, there is some unavoidable repetition in nature of the background material covered, and in the case of the empirical papers, also the study design. It should also be noted that the terms ‘stock route’ and ‘paddock’ have been used throughout papers directed at an Australian audience, but these terms have been substituted respectively for ‘linear remnants’ and ‘fields’ in papers intended for international publication.

In line with the Australian National University’s College of Medicine, Biology and Environment guidelines for ‘PhD by Publication’, an Extended Context Statement has been provided at the beginning of the thesis. This is required to provide “the relation between all aspects of the research... [The context statement] will include an introduction to the field of study, and an account of the full research methodology employed... a summary of the outcomes of the project, a conclusion and a list of references cited.”

I carried out the majority of the work for the papers that form this thesis. This includes study design development, data collection, laboratory work, data entry and analysis, and writing and preparation of the manuscripts. I also took or prepared all of the images, figures and tables in the thesis unless otherwise acknowledged. My supervisors, Joern Fischer, Philip Gibbons, and Tara Martin provided advice for all of these components.

Other collaborators provided guidance for this thesis. David Lindenmayer was involved in developing Papers I and II, by identifying additional literature and providing feedback on the manuscripts. The design, analysis and manuscript preparation for Paper III were carried out under the advice of Saul Cunningham. Jan Hanspach provided assistance and R code for the study design and data analyses presented in Papers IV and V. Bradley Law provided guidance for the design and manuscript preparation for the microbat study in Paper V, and also developed the automated bat species identification key. Michael Drielsma developed the Landscape Value scores for each of the stock routes included in the spatial prioritisation exercise described in Paper VI, and provided feedback on manuscript preparation. Josie Carwardine also provided advice on the design and write-up of Paper VI. More specific assistance provided for each of the studies is acknowledged at the end of each paper.

Acknowledgements

I have been able to complete this thesis only with the support, guidance, good humour, and downright common sense of a very large number of people, and I am determined to acknowledge them all here.

Firstly, I would like to thank Joern Fischer, who for some reason agreed to take a chance on me after a very long coffee. I will forever be grateful for the reassurance and wisdom he has provided me, irrespective of whether I was in the office, up a tree, or on the other side of the world, and I think a big portion of the success of this project can be attributed to his vision from the get-go. Phil Gibbons has been endlessly encouraging and his feedback has always driven me to do better, and I am extremely grateful that he agreed to take over primary supervision responsibilities. Tara Martin has been generous and insightful, and her patience with me while I learnt the planning ropes allowed me to explore aspects of this project which I otherwise wouldn't have been able. I really lucked out in having three such approachable, constructive and helpful people as supervisors.

The contributions of several other collaborators really lifted the standard of the papers produced. Jan Hanspach was incredibly helpful during the process of data analysis, and generously housed and provided me with wonderful company during my stay in Germany. Saul Cunningham has been an absolute gun, and David Lindenmayer has provided insights that few others could. Michael Batley drove to Canberra to donate his considerable bee identification skills for what was a very long day, and I couldn't have carried out the bat project without Brad Law, who provided both his extensive expertise and his equipment.

This project received financial support from the CSIRO, the WildCountry Science Council, and the Paddy Pallin Foundation in partnership with the Royal Zoological Society of NSW – all of which helped greatly with the considerable associated field work costs. The ongoing interest and support of both the Pallins and the RZS members in particular helped keep me motivated.

Many landholders allowed me to stomp through their paddocks in the name of data collection. These were the Boland, East, Forde, Francis, Fuge, Gibb, Harper, Herbert, Kavanagh, Kelly, Kite, Knight, Knight, Markwort, Maslin, Matchett, McLachlan, Mitton, Naughton, Porritt, Robinson, Ryan, Sanderson, Smith, Sweeney, Sykes, Taylor, Thomas, Whyte, Worner, and Yerbury families, and Steve Brien at the Darley global breeding operation. These kinds of agroecology projects would not be possible without the cooperation of friendly, interested and willing folk, so thank you.

I'd also like to thank those that volunteered their time to help out with field work, which was often very hard going; Corey Bunnell, Madelaine Castles, Lesley Hook, Beth Noel, Gabbie Openshaw, Thomas O'Reilly, Mark Quinnell, Katherine Russell, and Jarom Stanaway. In particular, Mary Long, Lulu Lentini, and Marcus Salton came out to help on multiple occasions, travelling all the way from Melbourne, and I really couldn't have finished without them. Caragh Threlfall and Anna McConville demonstrated superhuman stoicism during the bat trapping trip – I would like to thank them for powering through the sleep deprivation!

Many others supplied knowledge, data, or equipment along the way just because they're lovely helpful people. These were Toni McLeish, Rainer Rehwinkel, Andrew Zelnick, Doug Mills, Leroy Gonsalves, John Stein, Nicki Munro, Deb Saunders, Jenny Newport, David Salt, Piers Bairstow, Cecile van der Burgh, Trudy O'Connor, and Bev Smiles.

I am going to dearly miss the friends that I have made whilst in Canberra, who made the whole venture eminently more do-able. That's you, Martin Westgate, Philip Barton, Laura Rayner, Chloe Sato, Ingrid Stirnemann, Dejan Stojanovic, John Evans, Alecia Carter, Brett Howland, Juliana Lazzari, Annabel Smith, and Ben Scheele. Katherine Selwood and Michael Kaplan provided me with a bed and some fantastic company at the beginning of my PhD, so I would like to thank them for the generosity. I would also particularly like to acknowledge my officemate Karen Stagoll/Ikin, another field work repeat-offender, who has helped me out in more ways than I could ever hope to list here.

Finally, to Alex Munro, who uprooted his life in the name of my academic pursuits - you have been a beacon of practicality and calm when I was a manic mess, and have contributed time, money, skills and sanity without complaint. Thanks for doing an excellent job of reminding me that there's more to life than work.

Abstract

Stock routes have been a feature of the Australian landscape since the mid-1800s. Originally established to provide corridors of forage and shelter for livestock droved ‘on the hoof’, the vegetation within them remained standing while vast tracts in the surrounding landscape were cleared to make way for agriculture. However, livestock are now more commonly transported in trucks, so managing authorities no longer receive adequate income from droving permits to offset costs. At the commencement of this PhD it was being suggested that some stock routes would be lost to freehold tenure, in spite of the fact that many scientists believed them to be of great value for conservation.

This project was structured into three distinct sections, aimed at providing evidence for the values of stock routes, and advice on how to best manage and plan for them into the future. The first section took the form of a literature review and spatial analysis, where I demonstrated that stock routes play a considerable role in biodiversity conservation, recreation, tourism, preserving Indigenous cultural heritage, and as a stock refuge. I also found that they occupy fertile, low-lying areas of the landscape, and contain associated vegetation communities which have been preferentially cleared for agriculture and are under-represented in protected areas.

In the second empirical section of the project I conducted surveys of three taxonomic groups which provide ecosystem services: woodland birds, wild bees, and insectivorous microbats. Data were collected from 32 stock routes which varied in width and vegetation condition, and in the fields adjacent to these stock routes. Statistical modelling was then used to quantify the response of each group to local and landscape

variables characterising the survey sites. Although specific responses were different for each taxonomic group, there were some consistencies in my findings. Scattered trees in fields had a positive effect on all taxa, and the value of both native pastures and formal conservation areas close to the stock routes was also a recurring pattern.

In the final section of my thesis I concentrated on the debate surrounding connectivity; whether it should be included as a goal in conservation planning, and whether this will result in trade-offs with habitat area and quality, or economic costs. It would seem logical to plan for connectivity if some stock routes are to be sold whilst others are retained for conservation purposes. Using them as a case study, I ran a Marxan minimum-set analysis which demonstrated that only certain connectivity approaches had effects on the costs and siting of reserves in conservation planning, and these effects were substantial only when conservation targets were set high.

At the time of writing, the future of the stock route network remains as uncertain as it was in 2009. However, through this work I have been able to provide a much more solid foundation for potential conservation decisions than was previously available. I have also demonstrated that the stock route network is an indispensible environmental heritage asset, which should be retained and managed for conservation in perpetuity.

Extended context statement

Introduction

Stock routes and associated stock reserves have been a feature of the Australian landscape since the mid-1800s. They were originally established to allow livestock to be transported ‘on the hoof’ with a drover, between properties or to markets. The livestock required shelter, forage and water on the journey, and could only be droved up to 16 km in a day. Because of this, a network of corridors (stock routes) and camps with access to water (reserves), many of which were wooded, developed across the states of New South Wales (NSW) and Queensland (McKnight 1977). Together, stock routes and stock reserves form what is known colloquially as “The Long Paddock”.

Intentionally or not, drovers tended to follow the path of least resistance in the landscape, which often happened to coincide with the same pathways that Australia’s Indigenous people had been using for thousands of years prior to European settlement (Spooner et al. 2010). In this way, the stock route network now follows Indigenous Song Lines and Dreaming Tracks in many areas (Kerwin 2006), as does the modern-day road network. However, there are a few key differences between stock routes and standard road reserves. Stock routes are often wider, stretching up to 1.6km in width (Spooner 2005), and have also been intermittently grazed by livestock over many decades (Hibberd et al. 1993). Because the stock routes were serving the purpose of droving, the vegetation within them was allowed to remain standing while much of the surrounding landscape was cleared to make way for agriculture.

The use of stock routes as a means of transporting stock has declined since the 1950s, and with this came a parallel decline in income from droving permits. As a result, the Rural Lands Protection Boards (RLPBs), who had been responsible for the NSW stock routes since the 1880s, could no longer manage them in a financially viable manner. In 2008, the RLPB system underwent an independent review, which recommended that the stock routes be ceded back to the NSW Department of Lands (Integrated Marketing Communications 2008). The Department of Lands stated that following this hand-back "disposal through sale may be an appropriate outcome for a restricted number and area of TSRs" (Land and Property Management Authority 2010, p31). The announcement was not taken lightly, and there was an outcry from environmental, agricultural, and scientific sectors of the community (Possingham & Nix 2008; The Wilderness Society Inc 2008). This is where the current PhD project commenced in 2009 – with talk of selling stock routes of 'lesser value' to private land holders, in spite of the fact that peer-reviewed ecological research on stock routes was very scarce. Because of the insufficient evidence base for decisions regarding which stock routes were most 'valuable', the overarching goal of this PhD project was to develop a better understanding of the role that stock routes play in biodiversity conservation, with a focus on the state of NSW. The resulting thesis is structured into three sections, aimed at providing both evidence for the values of stock routes, and advice on how to best manage and plan for them into the future.

The first section, comprising Papers I and II, aimed to cover and consolidate what was already known. This entailed drawing together some of the disparate information that was available with regards to the value of stock routes; as well as spatial data to quantify some simple but important metrics, such as the location of stock routes in the landscape. In particular, because the 2008 independent review of the system had focussed on traditional droving use, the intent was to highlight some of the emergent values with regards to biodiversity conservation, cultural heritage, rural communities, and Australian society as a whole.

During the course of the literature review it became evident that key pieces of information, central to stock route evaluation and management, were lacking. I therefore highlighted eight key knowledge gaps, which related to: (1) connectivity, (2) size and condition trade-offs, (3) landscape context, (4) representation, (5) indigenous heritage, (6) ecosystem services, (7) pricing schedules, and (8) recreational uses. Because the issues of recreation (knowledge gap 8) and indigenous culture (knowledge gap 5) would necessitate an evaluation of a social scientific nature, and pricing schedules (knowledge gap 7) could only be assessed through economic analyses, they were deemed to be outside the scope of the thesis. Instead, those central to conservation biology and ecology were addressed. Paper II explored disproportionate representation (knowledge gap 4) of vegetation communities, Papers III-V investigated the effects of stock route habitat quality and area (knowledge gap 2) and landscape context (knowledge gap 3) on wildlife communities that provide ecosystem services to farmland (knowledge gap 6), and Paper VI assessed the incorporation of connectivity (knowledge gap 1), into conservation planning for linear networks.

The second section of the thesis comprises Papers III-V, for which empirical data were collected on three separate taxa; woodland birds, wild bees, and insectivorous microbats. These corresponded to different spatial scales at which wildlife use the landscape, and all represent providers of mobile agent-based ecosystem services that are relevant in fragmented agricultural systems (Lundberg & Moberg 2003; Kremen et al. 2007). Bees provide pollination services and use the landscape at small scales (Steffan-

Dewenter & Tscharntke 1999; Gathmann & Tscharntke 2002), birds attract tourists to rural areas and use the landscape at moderate scales (Jones 2000; Doerr et al. 2011), and bats use the landscape at the broadest scales and provide natural control of crop pests (Entwistle et al. 1996; Goiti et al. 2003; Kalka et al. 2008). These groups were also chosen on the basis that they represent different Orders, can be rapidly surveyed, and are known to persist at least partially in agricultural environments.

The final section of the thesis, Paper VI, concerned conservation planning, and several factors determined the direction that it took. Although the original intent was to base the planning exercise on the data collected during field work, the question of whether 'connectivity' should be included as a goal during conservation planning arose naturally from discussions with stakeholders in both academia and policy. Numerous approaches for incorporating connectivity exist in the conservation planning literature (Hanski 1999; Moilanen & Nieminen 2002; Ball et al. 2009), but because these had not been systematically compared, it was unclear what the effects of using alternative connectivity approaches would be. Although linear networks such as stock routes are recognised for being valuable for conservation (Bennett 2003) and many recommend that they should be subject to systematic planning (Leon & Harvey 2006; Schmitz et al. 2007; Lundy & Montgomery 2010), few applied examples exist. Moreover, given that linear networks such as Australia's stock route network are inherently 'connected', they could feasibly contribute to representative reserve systems while providing connectivity at no additional cost. As multi-million dollar "connectivity conservation" projects continue to develop around the world (Worboys et al. 2010), it seemed timely that further investigation of systematic means for identifying priority 'connected' areas be carried out.

Methodology

The aim of the literature review, Paper I, was to collate all previous information pertaining to stock routes, but because much of this existed in the grey literature as government reports, it was difficult to carry out the review in a strictly systematic manner. Many reports and books were identified through discussions with organisations such as the NSW Department of Environment, Climate Change and Water, Catchment Management Authorities, Rural Lands Protection Boards, the CSIRO, and non-government organisations such as the National Parks Association. Additional literature was identified by conducting a broad literature search using the terms “travelling stock”, “stock route” and “stock reserve” in ISI Web of Knowledge (topic search– all results included), Google Scholar (whole documents – first ten pages of results included), and JSTOR (all disciplines, all content, all results included) up to October 2010.

ArcMap 9.3 (ESRI, Redlands, CA, USA) was used for the spatial analysis in Paper II, which incorporated a 41.1 million hectare area of the wheat-sheep belt of NSW. Three land tenures were compared: (i) individual stock routes, from the ‘TSR Conservation Values’ data set supplied by the NSW Department of Environment, Climate Change and Water 2010, (ii) protected areas in the National Reserve System, based on the “NSW National Parks and Wildlife Service Estate” layer from the DECCW data download website (http://mapdata.environment.nsw.gov.au), released on the 1st of April 2009, and (iii) all land. These three tenures were compared with regards to topography, using a multi-resolution valley bottom flatness (MRVBF) index raster at 250 m resolution (Gallant & Dowling 2003), and woody vegetation, from a dataset that had been developed as part of the National Carbon Accounting Systems Land Cover Change Project of the Australian Greenhouse Office (Furby 2002).

For the empirical component of the project, comprising Papers III-V, field surveys took place across a 100 km wide and 150 km long region in the NSW wheat-sheep belt: the aim was to cover as broad an area as logistically possible whilst still sampling the same vegetation types. In the region, paddocks form a mosaic interspersed with stock routes and planted vegetation, and in many cases large scattered trees also persist within the paddocks. Land use is dominated by dry cereal cultivation, including wheat, oats and barley, as well as both native and improved pastures for livestock grazing. Prior to European settlement, the area was covered predominantly by *Eucalyptus* woodlands, as well as some open forests and woodlands of *Callitris*, *Acacia* shrubland, and tussock grassland. However, it is now 84% cleared, with formal conservation reserves covering only 1.3% of the region (Pressey et al. 2000). Dominant canopy species include *Eucalyptus microcarpa*, *E. albens*, *E. melliodora*, *E. blakelyi*, and *E. populnea,* while *E. camaldulensis* is common along watercourses.

Thirty-two stock route sites and 24 adjacent paddocks were surveyed in the spring and summer of 2009/2010 (September to February) for birds and bees, and the following year this was extended to incorporate 32 paddocks for the bat surveys (November-February 2010/2011). Survey points were established in the paddocks at incremental distances, to see if isolation from the stock routes had an effect. These distances were at 100, 200 and 400 m from the stock routes for bees and birds, and 100 and 400 m for bats. Because bats can commute several kilometres in a night (Lumsden et al. 2002), it was expected that they would be less likely to be affected by proximity to the stock routes, so did not include the 200 m survey point. Instead, because bat activity is known to be heterogeneous at small scales (Fischer et al. 2009), they were surveyed in two separate points in the stock routes.

Standard survey techniques were employed for each group, and each was surveyed twice. Native bees were trapped using blue vane traps, which were suspended from a tree for one week before being collected. Bees were sorted to the morphospecies level following Michener (2000) and Dollin et al. (2000), and a reference collection provided by the Australian National Insect Collection. The correct sorting and identification of bees to the species level was then confirmed by Michael Batley (Australian Museum). For birds, following the standard method of the Birds Australia “Atlas” (Barrett et al. 2007), two hectare plots were actively surveyed for 20 minutes each, with all individuals seen or heard being recorded. For bats, because widespread surveying had not previously been carried out in the region, in the summer of 2009/2010 nine nights of harp trapping took place, to collect reference calls and confirm the presence or absence of some species. Then, in the following summer of 2010/2011, a full suite of bat surveys was conducted, using acoustic ‘Anabat’ detectors, placed on platforms strapped to trees. Bat calls were analysed using the ‘Anascheme’ program (Gibson & Lumsden 2003), which attributes ultrasonic pulses to individual taxa, using an identification key specifically designed for the study area. Arthropod sampling using 12 volt, 8 watt black-light traps, accompanied bat surveys. To determine arthropod biomass, samples were oven-dried at 60° C for eight hours, and then weighed using a laboratory balance to an accuracy of 0.001 g.

Most of the analyses for this field data were conducted using the “R” program for statistical computing (http://www.r-project.org/). This included ordination techniques, such as principal components analysis to reduce the number of explanatory variables, and non-metric multidimensional scaling to determine community-level patterns. In addition, regression modelling was used, including generalised linear models and generalised linear mixed-effects models. For each taxonomic group, explanatory variables were grouped according to the scale at which they occurred, and model selection followed an information-theoretic approach (Burnham & Anderson 2002). Species accumulation curves for the bee data were constructed using EstimateS software (Colwell 2006).

For the final paper (VI), systematic conservation planning was carried out using Zonae Cogito (vers. 1.22), a user interface for the Marxan conservation decision support tool (vers. 2.1.1). Our conservation objective was to represent a certain percentage (10, 30, 50 or 70%) of the original extent of each vegetation community in the stock routes and the protected areas, by protecting intact sections or restoring degraded sections, in each case at minimal cost. Spatial data for the state of NSW are generally poor, and the best available data came from national-level datasets. These were the “Australia - Estimated Pre-1750 Major Vegetation Groups - NVIS Stage 1, Version 3.0” spatial layer created by the Australian Government’s National Land and Water Resources Audit, 2002, and the “Vegetation Assets, States and Transitions, vers. 2” layer (Lesslie et al. 2010), used to create the conservation feature and cost files for the analysis. For each of the four representation targets we considered different connectivity approaches, and compared the alternative planning outcomes with a situation where connectivity was not considered. Three connectivity approaches were employed; two of which (“BLM” and “Euclidean Distance”) could be implemented by adjusting the “Boundary Length Modifier” parameter through Marxan. The third “Landscape Value” approach was developed by Michael Drielsma of the NSW Office of Environment and Heritage, by combining the Colonisation Potential and Neighbourhood Habitat Area approaches of Hanski (1999). All of the initial spatial analyses required to build the input files for Marxan were carried out in the ArcMap (vers. 9.3.1) geographic information system.

Summary of outcomes

Paper I. Australia's Stock Route Network: 1. A review of its values, and implications for future management

A total of 107 publications dealing with stock routes were found, including 68 journal articles, 13 government reports, 14 books, three consultancy reports, three conference papers, two government submissions, one independent report, one PhD thesis, one Honours thesis, and one letter. In many documents, stock routes were mentioned only in passing, and the majority of the literature related to the biodiversity conservation value of stock routes. It was found that stock routes form important habitat for, and support a wide range of, threatened species and communities. In addition, many stock routes hold heritage value for both Indigenous and non-Indigenous Australians. There are also other societal benefits arising from stock routes because they provide land for recreation and apiary sites, a source of seed for revegetation projects, a focus for rural tourism, as well as the traditional benefits of stock droving and emergency agistment. As mentioned previously, eight key knowledge gaps pertaining to the values of the stock route network were identified, and used to inform the research subsequently presented in this thesis.

Paper II. Australia's Stock Route Network: 2. Representation of fertile landscapes

Although stock routes are not pristine and are typically smaller and narrower than protected areas, the analysis revealed that they provide considerable representation of the landscape features and vegetation types not conventionally included the national reserve system. Around 55% of stock routes occur in low-lying valley portions of the landscape, compared to only 6% of protected areas. They support a wide range of threatened vegetation communities, and are not biased towards heavily forested areas.

White Box-Yellow Box-Blakeley's Red Gum woodland was recorded in 803 (or 17.5%) of the 4575 stock routes in the data set. It has been estimated that only 0.01% of this vegetation remains in relatively unmodified condition (Prober & Thiele 1995), and it is listed as critically endangered by the Australian Government. By comparison, only 10 of the 335 protected areas within the spatial study region are known to support "small occurrences" of this community.

Paper III. Supporting wild pollinators in a temperate agricultural landscape: maintaining mosaics of natural features and production

A total of 3,249 individual bees, representing four families and 36 species were collected in the blue vane traps. Nectar-bearing crop generally supported the most species-rich bee assemblages, and the highest abundance of individual species. The five smallest species collected (*Lassioglossum sexsetum, L. hemichalceum, L. aspratulum, Homalictus urbanus* and *H. sphecodoides*), which have estimated maximum homing distances of 20-62 m (based on the model presented in Greenleaf et al. 2007), were all trapped at 400 m from the stock routes. Hence, distance from stock routes does not appear to limit the body size of species occupying paddocks (up to 400 m). Bee assemblage richness was also positively correlated with the presence of conservation land nearby, or the number of remnant scattered trees surrounding traps in paddocks, and the abundances of several individual species responded in similar ways. This finding is important because scattered trees within cropped paddocks are often perceived negatively by farmers, but the small untilled areas beneath them may allow wild bees to persist in otherwise homogenous paddocks. Stock routes contributed to assemblage heterogeneity by adding unique bee species to the regional pool, and had marginally higher Shannon-Wiener diversity than paddocks ($t = -1.961$, $df = 96$, $p = 0.053$).

Paper IV. Value of large-scale linear networks for bird conservation: A case study from travelling stock routes, Australia

A total of 81 bird species were recorded in the surveys, and of these, 45 were classified as woodland species. Woodland birds were found to be particularly associated with stock routes and native pastures, and two species of conservation concern in NSW, *Pomatostomus temporalis* and *Climacteris picumnus*, were found exclusively in these areas. Stock routes supported significantly higher total species richness ($t = -4.07$, $df = 102$, $p < 0.001$) and woodland species richness ($t = -4.8893$, $df = 102$, $p < 0.001$) than paddocks. In the stock routes, vegetation structural complexity was a better predictor of woodland bird richness than stock route width. In the paddocks, the highest number of woodland species was found in native pastures, and there was also a positive association with the number of retained scattered trees. Paddocks which had narrower stock routes running next to them supported more woodland bird species. This suggests that as the width of the stock routes decreased, birds were using adjacent agricultural land as supplementary habitat to meet foraging and nesting requirements.

Paper V. Corridors and unimproved pastures are wildlife-friendly farming measures that will benefit microbats

The detectors recorded 91,969 bat calls, 17,277 of which could be attributed to one of the 13 taxa recorded, and 491 of the calls contained feeding buzzes. Stock routes supported higher bat activity than paddocks ($p = 0.044$, $t = -2.0283$, $df = 223$), but species richness and feeding activity were not significantly different between the two types of land use. A mean of 87.6 g ± 17.6 SE of dry arthropods was collected at each of the sites per night, but there were no differences in dry biomass between land use classes. Stock routes that were wider and contained intact native vegetation supported more bat species, as did those which ran next to unsealed, as opposed to sealed roads.

Paddocks of unimproved native pastures with retained scattered trees and associated hollows and logs supported the greatest bat species richness and activity. The negative effect of sealed roads also extended into the paddocks, having an apparent negative impact on bat activity. It should be noted that a considerable proportion of the landholders we spoke with indicated they were not aware that bats may use their paddocks for roosting or foraging – this strengthens the case for better communicating the persistence of these cryptic taxa in agricultural landscapes, especially given the positive indirect impact bats are likely to have on crop yield through pest predation.

Paper VI. What effect does planning for connectivity have on linear reserve networks?

Reserve networks that were designed using the "No Connectivity" and "BLM" approaches were generally similar, and the "Euclidean Distance" and "Landscape Value" approaches had the strongest influence on solutions, particularly when conservation targets were high. Reserve networks that were designed to meet high representation and connectivity targets covered a greater total area, and incorporated a larger number of smaller individual reserves. Because of this, 'connected' reserve systems generally cost more, and there were also fewer substitutions that could be made to meet the needs of stakeholders, so they were less flexible. However, these effects tended to occur only when representation targets were set at high levels (50-70%) rarely employed in planning, so concerns regarding undue costs arising from incorporating connectivity measures may not be warranted. It could be interpreted that reserve networks that seek to maximise both representation and connectivity compromise on larger contiguous areas and cost. However, in landscapes with linear habitat networks, connectivity measures serve to highlight important sections that 'link' the landscape, which may otherwise be overlooked.

Concluding remarks

The core goal of this thesis was to "provide evidence for the values of stock routes, and advice on how to best manage and plan for them into the future". The findings presented here demonstrate that stock routes protect some of the most threatened vegetation communities in Australia, and form habitat for a range of wildlife, including threatened species (Paper I). A range of potential management options to protect these values have also been suggested (Paper I), and the impact of decisions in conservation planning on biodiversity outcomes for the stock routes has been explored (Paper VI).

This research was framed in the context of eight key knowledge gaps; not all of these have been fully addressed, and some could constitute several theses unto themselves. Nonetheless, it has been clearly shown that stock routes protect features of the landscape not represented by the protected area system (Paper II, knowledge gap 4), and constitute continental-scale structural connectivity which may be enhanced further by restoration (Papers I and VI, knowledge gap 1). They are used by taxonomic groups which provide valuable ecosystems services to agricultural land (Papers III-V, knowledge gap 6); wild bees and woodland birds respond to the condition of the vegetation in stock routes, whereas microbat communities are affected by both habitat condition and area (Papers III-V, knowledge gap 2). Landscape context also plays a role; woodland birds and micobats are positively associated with native pastures, and the presence of formal conservation areas close to the stock routes is important for wild bees (Papers III-V, knowledge gap 2). A consistent, overarching message is that scattered trees in paddocks strongly affect the use of farming landscapes by wildlife. Forming a critical resource, the continued retention and management of these trees is vital for conservation of biodiversity across the wheat-sheep belt.

Due to government turnover and inaction during the course of this PhD project, as yet no 'hand-back' of stock routes to NSW Crown Land has taken place. However, when the new NSW government came into power in 2011 they fulfilled an election promise to carry out yet another review of (what was now called) the Livestock Health and Pest Authority System, which in February 2012 announced largely the same finding as the 2008 review:

"It should be an immediate priority to devolve the TSRs to the Crown as there is likely to be little benefit for ratepayers in ensuring a focus on core biosecurity issues" (Ryan 2012).

As of June 2012, when this thesis was being submitted, public comments on the decision are being considered. The future of the stock route network in NSW therefore remains unclear, and the issue is likely to be a matter of further discussion for some time. In this sense, although the thesis is complete, the research is still something of a work-in-progress, and how the issue of stock route management develops in the future will dictate what direction this will take. In particular, a union of the empirical and planning components of the project could prove valuable to determine whether the three taxa studied can act as surrogates for one another in conservation planning for the stock routes. However, most importantly, through the work presented here a much more solid foundation for potential conservation decisions has been provided than was previously available. It has been demonstrated that the stock route network is an indispensible environmental heritage asset, which should be retained and managed for conservation in perpetuity.

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Paper I. Australia's Stock Route Network: 1. A review of its values, and implications for future management

Because this section of "The Boundary" TSR in Cootamundra, NSW, has remained intact, it been converted into a native seed orchard, to provide seed for revegetation and restoration projects in the region. Image: P. Lentini.

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Abstract

The Stock Route Network (SRN) is a vast system of public land, comprising vegetated strips and small reserves across the eastern length of the Australian continent. Now predominantly following the road system, this network was historically established to allow for the movement of livestock prior to truck and railway transport. Due to declines in traditional uses, parts of the SRN may now be sold to private landholders, or put under long-term grazing leases, making them unavailable for other emerging uses. This is in spite of the fact that it is widely accepted by researchers, practitioners, graziers and agriculturalists that the SRN holds great natural and cultural value. We conducted a review of scientific and grey literature to determine the known values of the network for biodiversity conservation, cultural heritage, rural communities, and Australian society as a whole. We found that the majority of existing literature on the network focuses on New South Wales (NSW) and is of a conservation-based nature. The Stock Route Network supports a wide range of threatened species and communities, with considerable potential for many more to be discovered. The network also holds heritage value for both Indigenous and non-Indigenous Australians. The societal benefits from the SRN are numerous because it provides land for recreation and apiary sites, a source of seed for revegetation projects, a focus for rural tourism, as well as the traditional benefits for stock droving and emergency agistment. In light of our review, we identified key knowledge gaps pertaining to the values of the SRN, and propose a number of options for its future management. Appropriate governance and increased investment in SRN management is now urgently required to ensure that it continues to conserve its many environmental and social values in perpetuity.

Introduction

The Stock Route Network (SRN) is a vast system of strips and small blocks of remnant vegetation which stretches across the eastern length of Australia, covering approximately 4.5 million hectares (Figure 1). Now predominantly following the road system, this network incorporates a wide variety of vegetation types, topographies, geologies, and histories (Davidson 1999). Known as "the Long Paddock", the SRN was gazetted as public land early in Australia's pioneering history to allow for the movement of livestock prior to the advent of truck and railway transport (McKnight 1977). The SRN comprises the Travelling Stock Routes and Travelling Stock Reserves of New South Wales (NSW), and the Stock Route Network of Queensland (Qld), as well as numerous other parcels of Crown land (Hibberd *et al.* 1986). For the purpose of this review, all are referred to as 'stock routes', and the entire network of NSW and Qld are together referred to as the Stock Route Network (SRN).

The use of the NSW network as a means of transporting stock has declined since the 1950s (O'Connor 2004). Rural Lands Protection Boards (RLPBs) have been responsible for the management of NSW stock routes since 1880 (Cameron and Spooner 2010), but they have been increasingly subsidising stock route management due to this decline in use, which has in turn threatened their financial viability. In 2008, the RLPB system underwent an independent review, which recommended that the stock routes be ceded back to the NSW Department of Lands (Integrated Marketing Communications 2008). The Department of Lands (now called the Land and Property Management Authority) stated that following this hand back "disposal through sale may be an appropriate outcome for a restricted number and area of TSRs" (Land and Property Management Authority 2009, p31). Based on this suggestion, the future of the SRN in NSW is uncertain. The stock routes of Queensland are also experiencing a decline in use by travelling stock, with local councils responsible for their management struggling to administer them in an economically viable manner. In response to problems which arose during the drought of 2002-03, a review recommended that the Queensland network be classified into ‘active’ and ‘inactive’ sections, with ‘inactive’ stock routes being leased to private graziers and subjected to Annual Grazing Agreements (Department of Natural Resources and Water 2008)

Although the SRN is now used for droving less often than it formerly was, it is widely accepted by researchers, practitioners, graziers and agriculturalists that it holds great natural and cultural value. When it emerged that the future of the network was in question, 513 scientists signed the “Long Paddock Scientists Statement”, which highlighted the role the SRN plays in environmental and heritage protection, and in supporting traditional uses (Possingham and Nix 2008). The letter came at a critical point in the debate, but was primarily intended to be an advocacy tool, as opposed to a review of information collected. A more in-depth review, which presents evidence in support of the statements made, is both timely and much-needed to inform the debate concerning the future of stock route management. In this paper, we review and summarise information on a wide range of present-day values of the SRN, expanding on the themes introduced in the “Long Paddock Scientists Statement”, and identifying strengths and weaknesses in current knowledge. We then discuss the implications of our findings for the ongoing administration and management of the SRN.

Methods

We conducted a broad literature search to identify existing scientific and grey literature on the stock routes of NSW and Qld. Using the terms "travelling stock", "stock route" and "stock reserve", we searched ISI Web of Knowledge (topic search– all results included), Google Scholar (whole documents – first ten pages of results included), and JSTOR (all disciplines, all content, all results included) up to October 2010. We confined our search to literature relating to stock routes of NSW and Qld. Additional literature not identified during the search that we were made aware of through other discussions with stakeholders, was also included. We did not conduct a quantitative review of this literature (such as a meta-analysis), but instead completed a conventional, qualitative review.

Results

We found 107 publications dealing with stock routes, including 68 journal articles, 13 government reports, 14 books, three consultant's reports, three conference papers, two government submissions, one independent report, one PhD and one Honours thesis, and one letter (Table 1). Most did not focus solely on stock routes, and we may have failed to identify additional material in which the SRN is a small component of the work rather than its central focus.

We identified only seven peer-reviewed journal articles that focused specifically on the SRN (as opposed to roadsides or remnant vegetation in general). These were: Cameron and Spooner (2010), Davidson *et al.* (2005), Lentini *et al.* (2011a), Lentini *et al.* (2011b), Lindenmayer *et al.* (2010b), O'Connor (2004), and Spooner *et al.* (2010).

A total of 71% of all literature was specific to NSW, 16% to Qld, and 13% covered issues relating to both states. We grouped the information collected into five broad themes (biodiversity conservation, cultural heritage, supporting rural communities, and value to Australian society). For each theme, we summarised existing findings, then identified key knowledge gaps.

Biodiversity conservation

Some stock routes are in excellent condition because of the absence of pesticides, herbicides, fertilisers, irrigation and cultivation (Davidson 1999; Davidson *et al.* 2005; Hibberd *et al.* 1993), as well as long-term set-stocking by domestic livestock, which is commonplace on surrounding private land (Dorrough *et al.* 2004; Martin and McIntyre 2007). Stock routes are some of the oldest remnants of native vegetation in landscapes, and support a large number of old, hollow-bearing trees which provide vital habitat for hollow-nesting fauna (Law and Chidel 2006; Spooner and Smallbone 2009). Stock routes also contain some of the largest remnants of vegetation types currently underrepresented in the National Reserve System (Law and Chidel 2006; Prober and Thiele 2005; Prober *et al*. 2001).

There are many examples of threatened species occurring in stock routes. The native herb Wandering Peppercress (*Lepidium peregrinum*) was presumed extinct until it was rediscovered in 2001, and the majority of individuals were in a stock route (NSW Scientific Committee 2000). Another ground-dwelling herb, *Trioncinia retroflexa*, was also rediscovered in a stock route (Fensham *et al.* 2002). Mammal surveys conducted in three stock routes of the Mount Molloy area in Qld revealed a total of 56 species, of which nine were classed as rare or threatened. Finally, surveys of approximately 200 stock routes by the Queensland Department of Environmental and Resource

Management found at least 95 threatened species, of which 16 are federally listed as 'Endangered' under the Environment Protection and Biodiversity Conservation Act (EPBC Act, Walsh 2009). Several studies note the existence of listed threatened ecological communities in the SRN, including White Box (*Eucalyptus albens*) -Yellow Box (*Eucalyptus melliodora*) -Blakely's Red Gum (*Eucalyptus blakelyi*) woodland (Burrows 2000; Prober and Thiele 1995; Spooner and Lunt 2004), Coolibah (*Eucalyptus intertexta*) - Black Box (*Eucalyptus largiflorens*) woodland (Gibbons *et al.* 2008), natural grasslands of the Queensland Central Highlands (Fensham *et al.* 2002) and Inland Grey Box (*Eucalyptus microcarpa*) woodland (Lentini *et al.* 2011a; Lunt *et al.* 2006; Prober and Thiele 2004).

Cultural heritage

In many areas, the SRN overlaps with Indigenous trading paths, Song Lines and Dreaming Tracks (Department of Environment and Climate Change NSW 2008). As a result, stock routes contain places and objects of significance to Aboriginal people such as scarred or carved trees, camp sites, missions, quarries, axe-grinding grooves, ceremonial grounds, rock shelters, or burial sites (Nowland *et al.* 1997). According to the NSW Aboriginal Heritage Information Management System, in a 41 million ha section of the NSW wheat-sheep belt (see Lentini *et al.* 2011b, this issue), stock routes harbour around 1500 recorded Aboriginal heritage sites (pers. comm., DECCW Nov 2010).

The SRN also provides an important connection for European Australians to the development of modern Australia; "The romantic image of the tall, browned, Australian stockman or woman droving a mob of sheep or cattle has captured the world's imagination…" (O'Connor 2004, p1). This evocative image may be considered an important link to Australia's pioneering past (Figure 2).

Supporting rural communities

Perhaps the greatest value that modern-day stock routes provide to rural communities is fodder for livestock during times of drought, and emergency pasture following floods or fire (Duncan 1962; Heathcote 1991; Nowland *et al.* 1997). During some droughts, every stock route in particular districts has been used by livestock at some point (Hampton and NSW Crown Lands Office 1982).

Droving on the stock routes is now less common than in the past, but still occurs, and occasionally on quite large scales. The review of the Rural Land Protection Board system in NSW found that around 9.9 million head of stock used the stock routes annually. Surprisingly, Boards with highest usage were also suffering the largest financial losses (Integrated Marketing Communications 2008) which suggests that a primary problem is not one of lack of use, but rather costly management and inappropriate pricing schedules.

Stock routes also support rural economies in a variety of other ways. For example, NSW is currently the largest beekeeping state in Australia, and honey production is worth an estimated \$28.5 M yr^{-1} (NSW Government 2007). Because many stock routes incorporate large tracts of remnant vegetation, they are highly desirable sites for apiarists, and allow bees to be supplied with a diverse range of floral resources (NSW Agriculture and Rural Lands Protection Boards 2001, see Figure 2). Stock routes located next to flowering crops can provide bees with a source of forage, while landholders simultaneously reap the benefits of crop pollination (Breckwoldt 1986, Department of Agriculture, Fisheries and Forestry 2011).

Some farming communities use their local stock routes to attract tourists. The small town of Barraba in NSW was the first to establish a series of 'bird routes', marketing various stock routes as bird hotspots (Figure 1) which now attract enthusiast 'twitchers' from as far afield as Norway, Canada, Belgium and the USA (Jones 2000). The scheme proved highly successful, and several other districts have established bird trails of their own (Schultz and Valenzisi 2010).

Other values for Australian society

Stock routes provide important sites for recreation and scientific research, because most are easy and free to access, and incorporate a wide variety of vegetation and environmental conditions. Much of the scientific literature we found did not directly investigate stock routes, but rather used them as 'reference sites' (Gibbons *et al.* 2008; Lunt *et al.* 2006; Prober *et al.* 2002; Prober and Thiele 2004). This is because they are often the closest analogue to what grassland, woodlands and shrublands resembled prior widespread agricultural development (Martin and Green 2002). These same characteristics also make stock routes appealing to the general public. According to "The Long Paddock" directory, recreational activities that take place in stock routes include pony club and gymkhana events, fishing, bush walking, and picnicking (NSW Agriculture and Rural Lands Protection Boards 2001).

Another benefit of stock routes is that they may act as carbon sinks in otherwise cleared agricultural landscapes. A study in NSW found that significantly more carbon is stored in the aboveground components of the vegetation in stock routes and other roadside reserves compared to paddocks (Eldridge and Wilson 2004). Similarly, Miklos *et al.* (2010) found that a stock route stored higher levels of soil organic carbon (8.21 kg/m^2) than an adjacent cropping field (5.04 kg/m^2).

Finally, stock routes provide a valuable source of locally-adapted seed for restoration projects. Seed from stock routes is likely to be genetically fit and diverse, since the seed comes from relatively large, intact and well connected remnants that include a wide range of vegetation types. Burrows (2000) found seed collected from Yellow Box (*Eucalyptus melliodora*) trees in stock routes had significantly higher reproductive output than seed from trees in paddocks, and Prober *et al.* (1998) found that even small remnant populations in stock routes can help maintain genetic diversity of the rare Yam Daisy (*Microseris lanceolata*).

Knowledge gaps

Given the outcomes of this literature review, we have identified eight key knowledge gaps pertaining to the values of the Stock Route Network, which should be prioritised for future research.

1. Does the SRN provide connectivity and facilitate dispersal? The SRN provides substantial structural connectivity across hundreds of kilometres of the agricultural landscape, and this is possibly one of its greatest assets (Figure 1). Despite these likely benefits no study has empirically assessed this connectivity value to date.

2. Should we protect larger, or more intact stock routes? When considering which stock routes to prioritise for conservation, decision makers need to have a clear understanding of the outcomes of trade-offs between high-quality, small stock routes versus larger, but more degraded stock routes. These outcomes will change depending on the species being targeted for conservation

3. What impact does landscape context have on stock routes? Few studies on the ecological value of stock routes have considered the impact of the surrounding landscape, which may be important in determining habitat suitability. Surrounding land use is likely to have an important influence on the quality of the stock routes (Martin *et al.* 2006), and conversely, stock routes have been shown to influence fauna on adjacent agricultural land (Lentini *et al.* 2011a).

4. What is the value of the SRN compared with the National Reserve System? Some authors have expressed concern that because of their linear shape and association with roads, stock routes are susceptible to edge effects (Major *et al.* 1999a; Major *et al.* 1999b) and may act as conduits for weeds and pathogens (Lodge *et al.* 2005). Yet these negative effects may be offset by the fact that the SRN supports threatened ecological communities not represented in the National Reserve System (see Lentini *et al.* 2011b, for further exploration of this question.)

5. We do not know the extent to which the SRN preserves Indigenous sites. Many stock routes have not been subject to archaeological surveys, and hence more work is needed to determine the Indigenous heritage value of the network.

6. The ecosystem services that the SRN provides have not been quantified. Ecosystem services may include pollination and pest control by wildlife, water filtration, carbon sequestration and erosion mitigation. These and other likely services need to be taken into account when considering the financial costs and benefits of maintaining the SRN.

7. What permit pricing schedules would best reflect demand and management costs? John *et al.* (1997) use a derived-demand approach to investigate competitive pricing of the stock route permit system in Nyngan, NSW. They found that the cost of droving permits could be increased significantly from \$0.02 up to \$0.80/sheep/day before demand becomes elastic. This area of research warrants further attention, and case studies which cover broader areas in different regions of the SRN would be valuable.

8. How many people use the SRN each year? Permits are not required to use stock routes for many recreational activities, research, or seed collection, and data on recreational use are not aggregated consistently (Hampton and NSW Crown Lands Office 1982). This make it difficult to quantify the extent to which stock routes are used by the general public.

Discussion

Management issues

Through this literature review, as well as our parallel study which demonstrates that the SRN complements the National Reserve System (Lentini *et al.* 2011b), we have shown that the SRN provides a suite of values to Australian society, and also that further research is needed in several key areas. However, an increasingly apparent overarching problem is that the multiple-use nature of the SRN does not fit into the single-use management structures of NSW and QLD, whereby different departments manage land for different purposes. Therefore, it is unclear how management of the SRN can integrate traditional uses of grazing and droving with emerging uses, such as conservation and recreation.

To this end, several approaches might be feasible. For example, the Rural Lands Protection Boards and local councils could continue to manage stock routes, but in recognition of their responsibility to protect and conserve threatened species and communities they would receive additional financial support from State and Federal governments. Permit pricing schedules for use of the stock routes would need to be adjusted to better represent a user-pays system (see knowledge gaps 7 and 8, above). This would reflect not only the benefits derived by private enterprises which drove and agist livestock, but also the benefits to the rest of Australian society, including recreation and conservation.

Alternative SRN management options are available, but may restrict the use of the SRN to certain groups. For instance, management of portions of the network could be transferred to adjacent landholders, and market-based instruments (such as reverse auction) could be employed to promote the protection of stock route values. However, this would inhibit the accessibility of stock routes to the public. A similar approach has been successfully employed by the Australian Government in their Environmental Stewardship Program, which has made significant gains in the conservation of box-gum woodland on private land across south-eastern Australia via the implementation of a tender system (Zammit *et al.* 2010).

Another alternative for the future management of the SRN is to incorporate it into the current National Reserve System, which could be appropriate given its significance for biodiversity and cultural heritage. Transfer to the National Reserve System would help achieve regional and national conservation targets, and particularly conservation objectives relating to comprehensiveness and representativeness. However, transfer to the National Reserve System would also further strain an already over-stretched system, and would exclude any form of grazing on stock routes. Such an approach is therefore unlikely to receive support from traditional rural users. These are just a few of the options available, but clearly a funding model which better reflects contemporary uses of the SRN should be implemented.

An uncertain future for the Stock Route Network

It is difficult to fully quantify what Australian society might lose through the sale or long-term leasing of portions of the SRN. Although we know that it supports threatened species and ecological communities, and sites of cultural significance for Indigenous people, to date survey efforts in the SRN have been disparate. There is a chance that the loss of sections of the network will also mean the loss of these conservation and cultural assets before we fully understand their extent. Sustained high grazing pressure, which stock routes may be subjected to if sold or placed under long term leases, has been known to reduce diversity of herbaceous flora (Dorrough *et al.* 2004; McIntyre *et al.* 2003), inhibit tree regeneration (Fischer *et al.* 2009), and negatively affect woodland birds (Martin and McIntyre 2007). If sold or leased, these stock routes also will no longer be accessible to the general public as places for recreation and research.

The importance of maintaining and restoring connectivity across cleared landscapes is becoming increasingly recognised, with significant efforts being made by both public and private entities in this endeavour. The degradation of the stock routes through intensive grazing would run counter to current national and international multi-million dollar investments to restore large-scale connectivity (Department of Environment and Climate Change NSW 2007). When considering future options for individual stock routes, management decisions are restricted by both limited funding and time. However, both the Qld and NSW governments have a mandate to protect threatened ecosystem and species, Indigenous and European cultural heritage, and the interests of the agricultural communities through the EPBC Act (1999), the Crown Lands Act (1989), the NSW Aboriginal Land Rights Act (1983) and Qld Aboriginal Land Act (1991), the NSW Threatened Species Conservation Act (1995), Qld Nature Conservation Act

(1992), and the Rural Lands Protection Act (1998). These legal obligations are clearly relevant to the SRN.

We hope that this review helps draw attention to the fact that the stock routes are a significant public asset. Given that this natural infrastructure benefits not only rural communities, but all of society, improved governance and increased investment in SRN management is now urgently required to ensure that it continues to conserve its many environmental and social values in perpetuity.

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Tables and Figures

Table 1. Literature considered in the review, categorised according to theme, state which the material covers ('NA' refers to publications which are not state-specific), and the type of publication ('Book' refers to both books and book chapters).

Theme	State	Pub. type	Literature	Total
Conservation (47)	NSW	Book	Hibberd and Soutberg (1991), Williams *et al.* (1991)	2
		Conference	Davidson (1999)	1
		Journal	Brock *et al.* (1999), Debus *et al.* (2006), Dorrough *et al.* (2004), Driscoll (2004), Eldridge *et al.* (2006), Gill and Williams (1996), Law and Chidel (2006), Leavesley (2002), Lentini *et al.* (2011a), Lentini *et al.* (2011b), Lewis *et al.* (2008), Lindenmayer *et al.* (2010b), Major *et al.* (1999c), Major *et al.* (2003), McIntyre and Lavorel (1994b), Prober (1996), Prober and Thiele (1995), Prober and Thiele (2004), Prober and Thiele (2005), Prober *et al.* (1998), Prober *et al.* (2001), Sass (2003), Semple (1986), Spooner and Lunt (2004), Spooner and Smallbone (2009), Thompson *et al.* (2006)	26
		Report (Govt)	Drew *et al.* (2002), Freudenberger and Drew (2001), Mills (2000), NSW National Parks and Wildlife Service (2001), NSW National Parks and Wildlife Service (2002), NSW Scientific Committee (2000)	6
		Thesis (Hons)	Channing (2000)	1
	QLD	Journal	Fensham *et al.* (2002), McIntyre and Martin (2001), McIntyre *et al.* (2002), Smith *et al.* (2007)	4
		Report (Cons)	Burnett (2001)	1
		Report (Govt)	Walsh (2009)	1
	NA	Book	Breckwoldt (1986), Martin and Green (2002), Prober and Hobbs (2008), Lindenmayer *et al.* (2010a)	4
		Journal	Sutherst *et al.* (2007)	1
Cultural Heritage (19)	NSW	Journal	Cameron and Spooner (2010), King (1959), Lunt and Spooner (2005), No author (1860), O'Connor (2004), Smailes and Molyneux (1965), Spooner (2005)	7
		Report (Cons)	Heritage Concepts Pty Ltd. (2007)	1
		Report (Govt)	Department of Environment and Climate Change NSW (2008)	1
		Submission	Environmental Defenders Office NSW (2009)	1

Theme	State	Pub. type	Literature	Total
Cultural Heritage cont'd (19)	QLD	Journal	Moore *et al.* (1928), Knowles *et al.* (1946), Morphy & Morphy (1984), Saunders (1995)	4
	NA	Book	McKnight (1977)	1
		Journal	Spooner *et al.* (2010), Taylor (1926)	2
		Report	Pearson (1999)	1
		Thesis (PhD)	Kerwin (2006)	1
Society and Science (22)	NSW	Book	Schultz & Valensizi (2010)	1
		Journal	Burrows (2000), Dorrough *et al.* (2007), Gibbons *et al.* (2008), Jansen and Robertson (2001), Jones (2000), Lunt *et al.* (2006), Major *et al.* (1999a), Major *et al.* (1999b), Major *et al.* (2001), McIntyre and Lavorel (1994a), McIntyre *et al.* (1993), Miklos *et al.* (2010), Prober *et al.* (2002), Prober *et al.* (2005), Somerville and Nicholson (2005), Thompson and Eldridge (2005)	16
		Submission	NSW Government (2007)	1
	QLD	Journal	Fensham *et al.* (2007), Martin and McIntyre (2007), McIntyre *et al.* (2003)	3
	NA	Book	Pigram and Jenkins (2006),	1
Rural communities (6)	NSW	Book	Anderson (2006)	1
		Conference	John *et al.* (1997)	1
		Journal	Eldridge and Wilson (2004), No author (1915)	2
	QLD	Conference	Duncan (1962)	1
	NA	Journal	Heathcote (1991)	1
Management (5)	NSW	Report (Cons)	Integrated Marketing Communications (2008)	1
		Report (Govt)	Department of Environment and Climate Change NSW and Industry and Investment NSW (2010), Land and Property Management Authority (2009)	2
	QLD	Report (Govt)	Department of Natural Resources and Water (2008), Local Government Association of Queensland Inc. (2003)	2
Multiple values (8)	NSW	Book	Hibberd *et al.* (1986), Hibberd *et al.* (1993), NSW Agriculture and Rural Lands Protection Boards (2001), Nowland *et al.* (1997)	4
		Journal	Davidson *et al.* (2005)	1
		Report (Govt)	Hampton and NSW Crown Lands Office (1982)	1
	NA	Journal	Possingham (2008)	1
		Letter	Possingham and Nix (2008)	1
			TOTAL	107

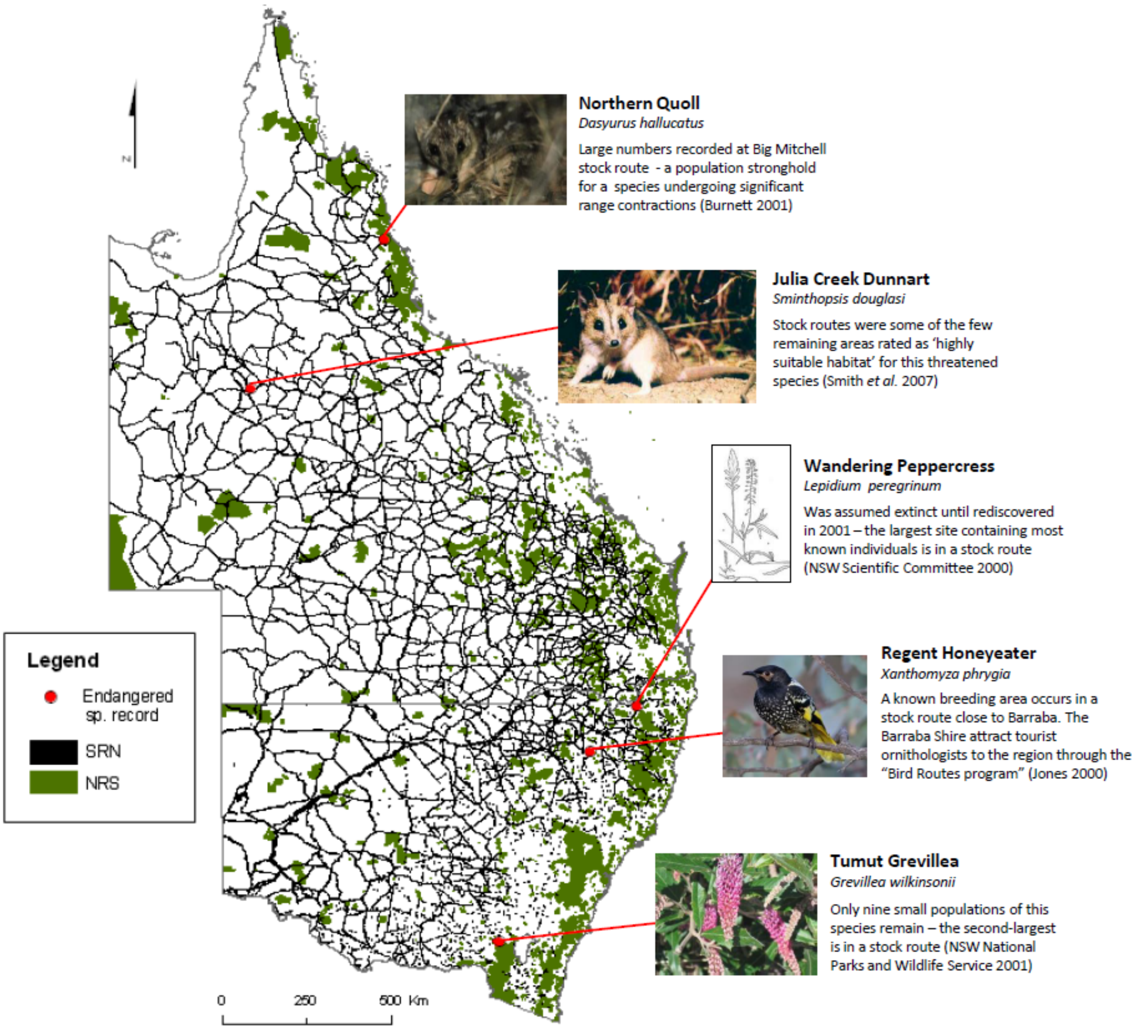

Figure 1. "Endangered species in the Long Paddock". Map shows: the Stock Route Network (SRN) of New South Wales and Queensland, the National Reserve System (NRS), and several examples of species which are listed by the Australian Government as Endangered, and that have been recorded in stock routes (photos top to bottom: Wildlife Explorer – Picasa Web Albums, Greg Mifsud, Catherine Wardrop, Chris Tzaros, Karen Hedley). Stock route spatial data accessed as the 'TSR Conservation Values' data set, supplied by the NSW Department of Environment, Climate Change and Water 2010, and the 'Stock Routes Queensland' data set, produced by the QLD Department of Environment and Resource Management 2011.

a) b)

Figure 2. a) The classic view of an Australian stock route: A drover with his mob of travelling sheep. “Australian stock route”, Harold Cazneaux, c. 1935 (Image courtesy of the Cazneaux family and the National Library of Australia) b) A modern-day stock route, supporting three purposes: An apiary, agistment, and conservation of nationally endangered Grey Box (*Eucalyptus microcarpa*) grassy woodland. “The Driftway”, Bogolong, 2009, NSW central-west slopes (Photo, Pia Lentini).

Paper II. Australia’s Stock Route Network: 2. Representation of fertile landscapes

Heavy rains in late summer 2010 transformed the low-lying “Fraters Speedway” TSR, in Narraburra NSW, into a wetland. Image: P. Lentini.

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Abstract

The Stock Route Network of New South Wales (NSW) and Queensland is a large-scale system of predominantly roadside remnant vegetation, which was established in the 1800s to allow livestock to be moved. Proposed changes to the management of the Stock Route Network (SRN) could result in some portions of it being sold to private landholders, or subjected to long-term set-stocking. This may have potentially negative impacts on some of the values of the SRN. One key feature of the SRN is that it covers low-lying parts of the landscape, which are poorly protected by national parks. To quantify this, we specifically analysed a 41 million hectare portion of the Stock Route Network which transects the NSW "wheat-sheep belt", characterising its representation of woody vegetation cover and topography, and contrasting this with the National Reserve System. Our analysis revealed that 55% of stock routes occur in low-lying valley portions of the landscape, compared to only 6% of the National Reserve System. The SRN supports a wide range of vegetation types, and unlike the National Reserve System, is not biased towards heavily forested areas. White Box-Yellow Box-Blakeley's Red Gum woodland, which is listed as critically endangered by the Australian Government, was recorded in 803 (or 17.5%) of the 4575 stock routes in our data set. In contrast, only 10 of the 335 reserves within our spatial study region are known to support small occurrences of this community. Our findings suggest that the protection of the SRN and National Reserve System together may fulfil the "representation" goal of systematic conservation planning far better than the National Reserve System on its own. Future research should quantify which stock routes in particular should receive priority for protection.

Introduction

Originally established in the 1800s for the purpose of moving livestock 'on the hoof', the Stock Route Network (SRN) of New South Wales (NSW) and Queensland forms an extensive system of connected linear remnant vegetation. However, proposed changes to the management of the SRN could see some parts of it sold to private landholders, or subjected to long-term set-stocking.

Given the uncertainty surrounding the future management of the SRN, Lentini *et al.* (2011) conducted a literature review to determine the known values of the network for biodiversity conservation, cultural heritage, rural communities, and for Australian society as a whole. They found that some authors had stated that stock routes should not receive priority for conservation, because their corridor-like configuration could make them susceptible to edge-effects, and may also facilitate the spread of weeds and invasive pests (for details and references, see Lentini *et al.* 2011). However, the unique history of stock routes also means that they capture vegetation types of a quality and topographical position conserved nowhere else. The SRN is likely to complement the existing National Reserve System, and may partly compensate for certain biases in the types of features typically protected in national parks (i.e. rugged landscapes with high tree cover rather than lowland vegetation). The degree to which this is the case has not yet been investigated, and Lentini *et al.* (2011) concluded that a key knowledge gap pertaining to the SRN was "*What is the value of the SRN compared with National Reserve System?*"

In response to this finding, and focusing on a well-documented region in NSW, we aimed to a) quantify if the SRN augments the National Reserve System with respect to woody vegetation cover and topography; and b) assess the degree to which Threatened Ecological Communities are represented within the SRN.

Methods

We compared land forms and woody vegetation cover across three land tenures within a 41.1 million hectare area of the wheat-sheep belt of NSW (Figure 1a). These were: (i) individual stock routes (ii) protected areas in the National Reserve System, and (iii) all land.

For the stock routes, we used the recently released 'TSR Conservation Values' data set (supplied by the NSW Department of Environment, Climate Change and Water 2010). This dataset defined the extent of the study area and comprises 4575 stock routes and reserves. These data also include information acquired from Rural Lands Protection Board rangers on perceived conservation values of each of the stock routes they manage, as well as the location of known Threatened Ecological Communities. For the National Reserve System, we drew on the "NSW National Parks and Wildlife Service Estate" layer from the DECCW data download website (http://mapdata.environment.nsw.gov.au), released on the 1st of April 2009. This includes National Parks and Nature Reserves, as well as State Conservation Areas, Historic Sites, Aboriginal Areas, Karst Conservation Areas, and Regional Parks.

Using ArcMap 9.3 (ESRI, Redlands, CA, USA), we 'clipped' two state-wide datasets to the stock route polygons, the National Reserve System, and the entire spatial study region. The first dataset represents land forms as a multi-resolution valley bottom flatness (MRVBF) index raster at 250 m resolution (Gallant and Dowling 2003), and contains four topographic classes: ‘valley flat’, ‘intermediate’, ‘erosional’, and ‘ridge flat’. The second dataset consists of a raster of 25 m resolution which classifies each cell as ‘wooded’ or ‘unwooded’. This dataset was developed as part of the National Carbon Accounting Systems Land Cover Change Project of the Australian Greenhouse Office (Furby 2002), is of a coarse nature, and does not capture areas with < 20% canopy cover (Gibbons *et al.* 2007). Therefore, it was not a comprehensive representation of woody vegetation cover in the Stock Route Network. We counted the number of cells in each of the classes within the SRN, the National Reserve System, and the spatial study region as a whole.

Results

The spatial study region was covered by 3.2 million ha of protected areas and 486,000 ha of stock routes. Of the area covered by stock routes, 55% occurred on low-lying ‘valley’ portions of the landscape, compared with only 6% in the National Reserve System. Conversely, 90% of the reserve system occupied sloping erosional areas (Figure 1b). The study area and the SRN contained similar proportions of woody vegetation (17% and 24% respectively), but substantially less than the reserve system, of which 76% was classed as ‘wooded’ (Figure 1c). Of the 4575 stock routes in the spatial study region, 1500 (or 32.8%) were known to contain Threatened Ecological Communities (TECs), with some containing more than one TEC (Table 1). White Box-Yellow Box-Blakely's Red Gum woodland was particularly well represented and was recorded in 803 or 17.5% of the stock routes in our data set.

Discussion

Our results demonstrate that the SRN contains a high proportion of low-lying valley areas. This is in a region that is otherwise dominated by intensive agriculture and grazing, and which contains very few other forms of public land. Fertile and naturally well-watered areas in valley floors may be more resistant to drought than elsewhere in the landscape, and may be important refuges for both livestock and wildlife (Martin and Green 2002). They can also provide varied, continuous floral resources for wild pollinators and the wider honey production industry, as well as wooded and aesthetically pleasing recreational and rest areas (Lentini *et al.* 2011).

Stock routes may represent all that is left of some threatened communities. Our analysis revealed that the White Box-Yellow Box-Blakely's Red Gum woodland community, which occupies these low-lying parts of the landscape and is listed as critically endangered under the Environment Protection and Biodiversity Conservation Act (EPBC Act, 1999), is particularly well represented in the SRN within our spatial study region. These woodlands once covered several million hectares of the wheat-sheep belt (Prober 1996), but were preferentially cleared by settlers, and are now mostly absent from the National Reserve System (Prober *et al.* 1998). Of the 335 reserves within our spatial study region, only 10 are known to support "small occurrences of White Box Yellow Box Blakely's Red Gum Woodland" (NSW Scientific Committee 2002). In support of the SRN's value to woodland conservation, Benson (2010) stated that "a single decision to increase funding to the authorities managing TSRs [travelling stock routes], would achieve as much as any other action in protecting the temperate grassy woodlands of NSW".

The value of stock routes is by no means limited to their role in conserving woodlands, and although our analysis has focussed on wooded vegetation, examples of a wide range of vegetation types are found in the SRN (Table 1). This may be because, unlike the National Reserve System, the location of stock routes is not biased towards heavily wooded areas. Instead, the SRN incorporates both wooded and unwooded portions of landscapes in an unbiased manner, containing comparable proportions to the wider study area (Figure 1c). In support of this, the NSW Government's Draft Biodiversity Strategy recognises that stock routes have provide an important refuge for native grasslands, and provide some of the best remaining examples of these ecosystems on public land (Department of Environment and Climate Change NSW and Industry and Investment NSW 2010).

Because the SRN incorporates vegetation types and topographical profiles poorly conserved on other forms of public land, the protection of the SRN and National Reserve System together would be an efficient means of improving "representation" within the conservation estate; a key goal of systematic conservation planning (Margules and Pressey 2000). It has also been proposed that providing representation and connectivity outside of protected area networks will be critical to the persistence of species under a changing climate (Lindenmayer *et al.* 2010). Although our results clearly demonstrate that the National Reserve System and SRN are complementary in the representation they provide, further work is needed to quantify which landscapes and communities in the SRN are most irreplaceable and vulnerable (sensu Pressey and Bottrill 2008) and should therefore receive priority for protection. Because the SRN provides a wide range of cultural, economic, and environmental values to Australia society (Lentini *et al.* 2011 this issue), as well as good representation of poorly protected landscapes, appropriate governance and investment to ensure its ongoing suitable management are needed, and warranted.

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Tables and Figures

Table 1. Threatened Ecological Communities, listed under the NSW Threatened Species Conservation Act (1995), known to exist within stock routes in the study region. Some stock routes support up to three of these communities.

Endangered ecological community	**Number of stock routes**
White Box-Yellow Box-Blakely's Red Gum woodland	803
Inland Grey Box woodland	590
Myall woodland	219
Coolibah–Black Box woodland	101
Fuzzy Box woodland	86
Allocasuarina luehmannii woodland	33
Sandhill Pine woodland	33
Native vegetation on cracking clay soils of the Liverpool Plains	21
Brigalow	16
Carbeen open forest	10
Semi-evergreen vine thicket	2
Cadellia pentastylis community	1
Montane peatlands and swamps	1
Tablelands basalt forest	1

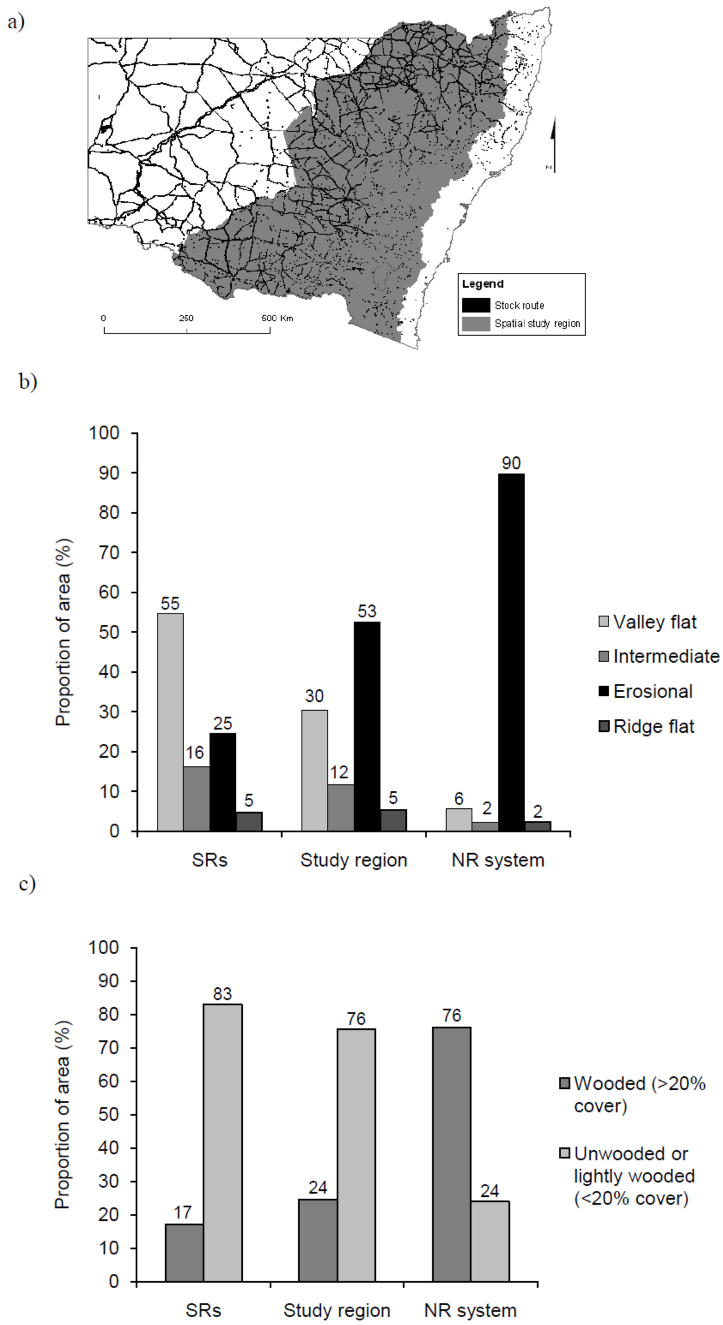

Figure 1. a) Map of New South Wales, Australia, showing the Stock Route Network and spatial study region used for our analysis b) Bar plot showing the proportion of each of the land tenures (SRs: stock routes, NR system: National Reserve System) occuring the four topographic profile classes c) Bar plot showing the proportion of each of the land tenures classed as ‘wooded’ or ‘unwooded’

Paper III. Supporting wild pollinators in a temperate agricultural landscape: maintaining mosaics of natural features and production

Blue vane traps on the back of the ute, ready to be set out to catch wild bees in the "Stanley" TSR, Wombat, NSW. Image: P. Lentini.

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Abstract

Pollination has received attention recently due to reported sharp declines of *Apis mellifera* in several locations, and it has been proposed that diverse native bee communities may be key for continued pollination of economically important crops. However, there is some inconsistency in the literature as to how these communities should best be managed. To address this issue, we collected bees from an intensively managed agricultural region in eastern Australia using blue vane traps. Both linear remnants of vegetation, which form part of a larger corridor network, and adjacent fields of native and exotic pastures, wheat, canola, and lucerne were sampled. A total of 3249 individual bees, representing four families and 36 species were collected. Highly modified environments of nectar-bearing crop supported the most species-rich bee assemblages, and the highest abundance of individual bee species. Distance from the remnants did not limit the body size of species occupying fields (up to 400 m). However, richness of bee assemblages also responded positively to the presence of conservation land in nearby areas, or the number of remnant native trees surrounding traps. Linear remnants of native vegetation contributed to assemblage heterogeneity by adding unique species to the regional pool. Our findings indicate that agricultural industries that currently rely on pollination by *A. mellifera* should ensure that intensive land use is complemented by untilled areas in the form of conservation land, or farm dams and scattered trees in fields, to support wild pollinators that may act as insurance against further future losses of managed hives.

Introduction

Bees (Hymenoptera: Apiformes) are the most important group of pollinators worldwide (Roubik, 1995; Kremen et al., 2004), and have been the centre of much recent debate (Ghazoul, 2005; Steffan-Dewenter et al., 2005). Given reported population declines of both *Apis mellifera* (European honey bee, De la Rua et al., 2009), and other pollinators (Potts et al., 2010), there has been growing concern for pollination services, and understanding how to best manage and boost populations of alternative wild pollinators has become a priority. Recent research has found that while agricultural fields provide an abundant source of forage for wild bees, they are also hostile nesting environments, with the proportion of untilled land in surrounding areas (Morandin and Winston, 2006; Morandin et al., 2007), diversity of weedy species (Winfree et al., 2008), and distance to natural areas (Ricketts et al., 2008; Krewenka et al., 2011) strongly influencing the diversity of wild bees in farmland. However, contrary lines of evidence suggest that under some circumstances, bees can readily persist in highly-modified anthropogenic habitats (Klein et al., 2002; Tommasi et al., 2004). For example, Winfree et al. (2007) found that in a mostly forested landscape, agricultural areas supported richer and more abundant bee assemblages than natural areas. In many cases, only the largest species are sampled a great distance from nesting sites (Steffan-Dewenter and Tscharntke, 1999; Gathmann and Tscharntke, 2002; Greenleaf et al., 2007), so many authors advise maintaining and building on networks of natural areas in the landscape to ensure continued visitation of bees to fields (Banaszak, 1992; Lagerlof et al., 1992; Morandin and Winston, 2005).

These recommendations are in line with increasing worldwide recognition that networks of natural areas can be beneficial to agricultural production (Tscharntke et al., 2005).

Examples of linear networks can be found in many countries, and may take the form of hedgerows (Ernoult and Alard, 2011), agricultural drainage ditches (Herzon and Helenius, 2008), riparian corridors (Sekercioglu, 2009), and railway right-of-ways (Tikka et al., 2001). In Australia, networks of roadside remnant native vegetation (known locally as ‘stock routes’) now transect some of the country’s most extensively cleared and intensively managed agricultural regions (Lentini et al., 2011b). It has been suggested that some of these ‘stock route’ remnants be sold to private landholders, necessitating the assessment of which sections to sell, and which to retain for conservation and other purposes (Lentini et al., 2011b). These sorts of conservation planning decisions are often based on well-studied groups such as birds and other vertebrates, for which responses to landscape changes can be more easily predicted (Kremen et al., 1993). Other groups, such as bees, may be equally or more important from a functional and economic perspective. One of the causes underlying the declines of *A. mellifera* is the parasitic mite *Varroa destructor* (Varroa mite, Ellis et al., 2010), which has now spread to all continents other than Australia. It is being assumed that it will eventually invade (Department of Agriculture Fisheries and Forestry, 2011), the economic consequences of which will be great: the predicted losses to Australian agriculture are between AUD $21 and 51 million annually (Cook et al., 2007).

In the absence of managed *A. mellifera*, it is likely that wild pollinator diversity will need to be maintained in order to fulfil pollination requirements (Kremen et al., 2002; Klein et al., 2003). Australia harbours approximately 1600 species of native bee, and the majority of these are solitary and nest in soil, hollow stems, or woody debris (Schwartz and Hogendoorn, 1999). However, the ecology of this group remains poorly understood (Batley and Hogendoorn, 2009). To inform the ongoing management of wild ‘free’ pollination in agricultural landscapes, we examined bee communities in linear remnants and in adjacent agricultural fields. Our study addressed two core questions:

1) What factors shape bee communities at landscape and local scales?
2) What management actions can be taken to encourage the persistence of wild bees in agricultural landscapes?

Materials and methods

Study area and design

Bees were surveyed across a 14,000 km^2 area of the inland agricultural region of New South Wales, Australia (33-34°S and 147-148°E, Fig. 1). Once covered by grassy *Eucalyptus*-dominated woodlands, the study region has since been cleared extensively to make way for cereal and livestock production. Cultivated fields and pastures now form a mosaic interspersed with linear remnant and planted vegetation, and in many cases large scattered trees also persist within the fields. Larger tracts of remnant native vegetation, in the form of reserves, are sparsely distributed across the region.

We sampled 104 points at 32 sites across the study region. Twenty four sites contained a trapping point in a remnant, and three additional trapping points in adjacent agricultural fields located at 100, 200, and 400 m from the remnant (Fig. 1). The remaining eight sites consisted of a trapping point in the remnant only, because adjacent fields did not contain an adequate number of trees. The 32 remnant points were stratified to represent the spectrum of vegetation condition and remnant widths within the study area – eight sites each of narrow-intact, narrow-degraded, wide-intact, and wide-degraded (widths ranged from 38 to 570 m; see Lentini et al., 2011a for details on site selection). The 24 agricultural field sites consisted of five native pastures, five improved pastures dominated by exotic grasses, five fields sown with lucerne (*Medicago sativa*) and/or clover (*Trifolium* spp.), six fields of wheat (*Triticum* spp.), and three fields of canola (*Brassica* spp.) All of the fields had some trees retained within them, ranging from 1 tree ha^{-1} in crops to 75 trees ha^{-1} in native pasture (μ = 5.6 trees ha-^{1}, see Lentini et al., 2011a for details).

Bee and vegetation surveys

We used blue vane traps (SpringStar Inc., Woodinville, USA) to conduct our bee surveys (Fig. 1), which are an efficient means of surveying wild bees in the presence of flowering resources (Stephen and Rao, 2007). Surveys were conducted in spring/summer between November 2009 and February 2010. For each survey, a single trap was hung at each trapping point for 1 week, after which the contents were collected. Each trapping point was surveyed twice, approximately two months apart, and crops in cultivated fields were harvested between these periods (though this did not appear to affect species richness– see Fig. S1, in supplementary material). At the 104 trapping points, traps were either hung from either (a) a tree branch (91 points); (b) a shelving bracket attached to a tree trunk (11 points); or (c) a shelving bracket attached to a post hammered into the ground (two points, both linear remnants). The average height of the hanging trap was 2.12 m (± 0.6 SD).

Vegetation data were collected at the same time as the bee surveys. The number of trees within a 100 m radius of the trapping point was counted, or calculated based on sampling described in Lentini et al. (2011a). This previous sampling also included measurements of diameter and length of logs at each trapping point, so the volume of logs (in cubic metres) was calculated from this data set. The per cent cover of plant species forming the ground-cover within a 100 m radius of each trapping point, and whether or not each species was flowering, was also recorded at the start of each survey. Using a geographic information system, we recorded the presence/absence of conservation land, farm dams, and farm or urban infrastructure (though not roads) within 1 km of each trapping point. These data were recorded using the "Land Use: New South Wales" spatial data set, dated 8th April 2011 and supplied by the NSW Department of Environment, Climate Change and Water. "Conservation land" includes National Parks, Nature Reserves, State Forests, and riparian areas, but not the linear remnants. Finally, we acquired information on the average daily rainfall for each of the survey periods, using the closest Bureau of Meteorology weather station to each of our trapping points (http://www.bom.gov.au/climate/data/, ranging from 1.5 to 26 km away). For all of these variables, we took the mean from data collected during the two surveys, for each trapping point to use in the analyses. Unfortunately, due to high winds 79 of the 208 (38%) traps fell from where they were hung, but in most cases still contained a large number of bees upon collection. To account for this, the number of traps remaining suspended for each trapping point was incorporated into the analyses (see below).

Following the survey period, all bees were sorted to morphospecies level, and then an expert was consulted (Michael Batley, Australian Museum) to verify the correct sorting of groups into species, and to assign them their correct scientific name. Once identification was complete, all individuals of a species, or 10 individuals for abundant species, were measured for intertegular span (IT, the distance between the wing bases, Cane, 1987) using a Nikon SMZ stereomicroscope and ocular micrometer, with a magnification between 0.75× and 2×. Then, using the model of Greenleaf et al. (2007), we used the IT measurements to calculate the predicted "maximum homing distance"

(MHD, the distance which an individual bee should be able to travel from and return to its nest) for each species.

Analyses

All analyses were carried out in 'R' ver. 2.10.0 (R project for statistical computing, http://www.r-project.org/), with exception of the species accumulation curves (see below). Based on an initial exploratory analysis (see Fig. S2), and our existing knowledge of bee biology, samples from native and exotic pastures were grouped into "pastures" (n = 10), canola and lucerne grouped into "nectar-bearing crop" (n = 8), and wheat was reclassified as "wind-pollinated crop" (n = 6). Exploratory analyses also revealed that landscape heterogeneity (number and proportion of surrounding land use types) did not influence species richness (see Fig. S3). For each trapping point, samples collected from the two separate trapping weeks were pooled for both species richness and abundance. All predictors in multivariate analyses were standardised to have a mean of zero and a standard error of one.

Habitat attributes that maximise bee diversity

a) Comparison between remnants and fields

We calculated the Shannon-Wiener (SW) index for bee communities recorded at each trapping point, and used equal-variance t-tests to test for any differences in the SW index and species richness between the remnant and field samples. Using EstimateS (Colwell, 2006), we then built species accumulation curves to compare data from the 24 remnant trapping points which had adjacent fields, with the three distance classes in the fields. This allowed us to determine the relationship between the number of bees caught, and the number of novel species detected in surveys.

b) *Species richness in remnants*

Because linear remnants and private agricultural properties are managed in isolation of one another, and we wished to make clear recommendations pertinent to both sets of managers, we chose to separate the analyses of the two components. Following Burnham and Anderson (2002) and Rhodes et al. (2009), we employed an information-theoretic approach to determine factors affecting species richness of bee assemblages in remnant sites. First, to reduce the number of variables used in the models, we created sets of predictors based on the scale at which they occurred (ordered here from smallest scale up): (1) the ground cover surrounding the trap, (2) the number of trees surrounding the trap, (3) the land use in the field adjacent to the remnant, and (4) the wider landscape context. A final group (5) accounted for conditions during the trapping period (Table 1a).

We then constructed generalised linear models which included all possible combinations of these five predictor sets (equating to 31 alternative models), assuming a Poisson distribution. To select the most parsimonious model, we considered Akaike's information criteria (AIC) and Akaike weights (w_i). Log likelihood (log$L$), which indicates how well the alternative models fit the data, was also considered. These are presented in Table 2a, also known as the '95% confidence table' as it lists the models that have a summed Akaike weight ≥0.95. In this way, models with lower predictive power are not included. The "relative importance" which is listed for each of the predictor sets was calculated by summing w_i for all of the models incorporating that predictor set.

c) *Species richness in farmland*

For this portion of the analysis, we considered only trap data from fields. We adapted the methods of Burnham and Anderson (2002) and Rhodes et al. (2009) once more, however, in this case we compared generalised linear mixed models (GLMMs) because of the nested nature of the data (i.e., multiple trapping points within individual fields). Using the R package glmmML version 0-18.3, models were constructed assuming a Poisson distribution, and with the field at which sampling took place treated as a random effect. Predictor variables were grouped into sets to represent (1) the ground cover surrounding the trap, (2) attributes of the field, (3) the width of the adjacent remnant, (4) the wider landscape context, and (5) the conditions during the trapping period (Table 1b.) All possible combinations of these variables were used to construct 31 alternative models, and model selection followed the same procedure as described above.

Effect of landscape modification on individual bee species abundance

Next, we addressed the question of how landscape modification affects the abundance of the most common bee species. First, we created two new variables to describe the level of modification at each of our trapping points using a principal components analysis (PCA). Per cent of ground covered by cropping, native vegetation, and exotic (non-crop) vegetation, as well as the volume of logs were included as predicator variables in the PCA, which was run using a correlation matrix to account for differences in variance of variables. These variables were used based on the assumption that ground cover of cropping would represent the most modified environments, likely to be subject to highest inputs of fertiliser and pesticides, and native and exotic ground cover would separate pastures along the spectrum of extensive to intensive management. The presence of logs is correlated with other structural attributes of less-modified natural areas, such as shrubs and leaf litter (Lentini et al., 2011a).

Principal Components 1 and 2 effectively separated the different land use types, and together explained 53% of the variance. Component 1 represented the level of modification, ranging from low scores in the linear remnants and native pastures, average scores in the exotic pastures and lucerne fields, to the highest scores in crops of wheat and canola. Component 2 separated sites with a pasture-like structure (linear remnants, exotic pastures, and lucerne/clover) from taller crops (wheat and canola; plot and loadings in Fig. S4 and Table S1).

Models were then constructed for the 15 most abundant and widespread species trapped (Table 2), to determine if there was any relationship between abundance and Components 1 and 2, as well as presence/absence of surrounding conservation land, number of trees in a 100 m radius, and the width of the remnant that the trap was in or next to. The falling of some traps during the trapping week led to the occurrence of more zeros in the data than can be accommodated in a standard Poisson or negative binomial distribution. To account for this, we modelled the data using zero-inflated mixture models (which mix two distributions, Martin et al., 2005) using the R package pscl. First, we modelled excess zeros against a distribution with a point mass at zero, and included the number of traps still suspended after the two separate trapping weeks (either zero, one or two traps) as a covariate. Then, a standard Poisson (or a negative binomial where overdispersion of non-zero values was apparent) model was used for the count data deriving from these distributions (Zuur et al., 2009). Model selection for all species started with the full model:

Abundance ~ Component 1 score + Component 2 score + conservation land + number of trees in 100m radius + width of linear remnant | number of traps remaining suspended

Following methods prescribed by Zuur et al. (2009), terms were sequentially dropped based on *z*-scores and *p*-values, until there was no longer a reduction in AIC. Pearson residuals were plotted against the fitted values to ensure that the assumption of homogeneity was not violated.

Results

Habitat attributes that maximise bee diversity

We trapped 3249 individual bees, representing four families and 36 species (Table 2). Of these species, five were found exclusively in remnants, 12 exclusively in remnants and untilled pastures, and four exclusively in crops. The five smallest species collected (*Lassioglossum sexsetum*, *Lassioglossum hemichalceum*, *Lassioglossum aspratulum*, *Homalictus urbanus* and *Homalictus sphecodoides*), which have estimated maximum homing distances of 20-62 m, were all trapped at 400 m from the remnant.

Comparison between remnants and fields

Traps in fields had a marginally greater mean species richness (μ = 5.53) than traps in remnants (μ = 4.16: t = -1.90, df = 96, p = 0.060). However, species accumulation curves showed that although field traps collected a greater abundance of bees, remnant traps added more unique species to the regional pool (Fig. 2). Consistent with this pattern, Shannon-Wiener diversity was marginally higher in remnant samples (μ = 1.32) than field samples (μ = 1.07: t = -1.961, df = 96, p = 0.053).

Species richness in remnants

The most parsimonious model of bee species richness in remnants incorporated variables relating to (in order of relative importance) the landscape context, ground cover, and the field adjacent to the remnants (Table 3a). However, the 95% confidence table included 10 out of 31 models compared and all of the predictor sets tested, so no one model or predictor set stood out as being the most parsimonious. The models indicated that bee species richness in remnants was positively correlated with the presence of conservation land in the surrounding landscape, and negatively with both the presence of built infrastructure, and the proportion of the ground cover which was flowering and/or native (Table 3b, and Fig. 3). Linear remnants adjacent to pastures were more species rich than those adjacent to nectar-bearing crops (Table 3b).

Species richness in farmland

Field variables, and the conditions under which trapping took place, had an important influence on bee species richness in fields, with both of these predictor sets being included in all of the models of the 95% confidence table, and each having a relative importance of 0.989 (Table 4a). The highest-ranked model also included the ground-cover predictor set, but once again all sets were included in the 95% confidence table and no one model stood out as the 'best'. Greatest bee species richness was recorded in fields containing nectar-bearing crop, dams, more trees within a 100 m radius of the trap, and where both traps remained suspended for the entire trapping period (Table 4b, and Fig. 4). As was the case in the remnants, there was a negative correlation with the proportion of native and flowering ground cover, and the amount of rainfall during trapping.

Effect of landscape modification on individual bee species abundance

Of the 15 species for which we modelled landscape modification effects on abundance, nine demonstrated a significant relationship with at least some of the predictor variables tested (Table S2). Six of these species responded positively to Component 1 of the PCA, or 'level of modification', indicating greater abundance in crops. However, these same species also responded positively to either the number of trees in a 100 m radius of the trap, or the presence of conservation land within a 1 km radius. The abundance of the native bee species *Lipotriches flavoviridis* and *H. urbanus* did not increase with modification, but instead responded positively to the number of trees as well as the width of the remnant. The only exception to these positive associations with landscape habitat complexity was *Amegilla chlorocyanea* (blue-banded bee) which responded negatively to the number of trees surrounding the trap.

Discussion

What factors at landscape and local scales influence bee communities?

Based on our findings relating to both the accumulation of unique species and assemblage heterogeneity, it appears that the linear remnants are playing a role in maintaining diversity at a regional scale. The maintenance of bee diversity is important not only for the sake of biodiversity conservation, but also for continued agricultural pollination (Brosi et al., 2007; Garibaldi et al., 2011). Within the study region, many land owners cultivate crops that benefit from insect pollination, including lucerne and canola. Lucerne is planted as a source of nitrogen-rich fodder for livestock, and is known to be inefficiently pollinated by *A. mellifera*, while other genera such as *Megachile* and *Nomia* are more effective (Cane, 2002). Canola crops constitute 1.9 M ha of Australian agricultural land and AU $1.1 B of annual exports (Cook et al., 2007), with growers currently relying on both managed and wild *A. mellifera* for pollination (Cunningham et al., 2002). Canola is also pollinated by many native bee species (Arthur et al., 2010), so these communities represent potentially critical insurance should *Varroa* mite enter Australia. Aside from agricultural production, most native plants rely on native pollinators, both in terms of outcrossing and the quantity of seed produced (Keller and Waller, 2002; Ashman et al., 2004).

Interestingly, when considering more localised scales, our modelling revealed that six of the most abundant species trapped in the study responded positively to increasing levels of landscape modification, and cultivated fields were also more species rich than other land uses. This is not surprising, given that nectar-bearing crops offer a pulse of highly attractive and abundant forage while in flower, which in turn should attract a large number of pollinators (Westphal et al., 2003; Otieno et al., 2011). However, these crops also have a limited flowering season which may not extend long enough to support bee species throughout their life cycle (Carvalheiro et al., 2010). Crops are also not a diverse source of forage, so supplementary areas such as pastures and remnants allow for dietary variation (Morandin and Winston, 2005). Nectar-bearing crops and other ground-layer flowering species appeared to attract bees away from traps in remnants, and it is also possible that traps in open fields may have been more visible to bees than those in remnants, where tree cover may have obscured the traps. Given that we know so little about wild Australian bee communities, even the basics such as which species are ‘common’, and which may be rare is still unclear, so these findings should be considered with some caution. It should be borne in mind that we were only able to model abundance for the most common species, which are also likely to be the least vulnerable to human land use. In addition, although blue vane traps have been shown to effectively sample more species than other survey methods (Stephen and Rao 2007), our data may be biased towards those which are especially attracted to blue.

Along with the positive effect of landscape modification, several other key factors appear to shape bee assemblages. The presence of conservation land proved important for both species richness in remnant vegetation, and individual species abundance. In fields, the number of trees also had a positive impact on both bee diversity and individual species abundance. Retained scattered trees are a phenomenon of agricultural landscapes worldwide (Mountford and Peterken, 2003; Plieninger et al., 2004; Gibbons et al., 2008), and have received recognition for their role in providing habitat for a wide variety of biota, increasing landscape connectivity and permeability (Manning et al., 2009), and are also associated with a variety of ecosystem services (Manning et al., 2006). Trees within cropping fields are often perceived negatively by farmers, because they compete directly with crops for water and nutrients (Huth et al., 2010), and reduce yield in the area directly surrounding them, resulting in a halo of non-cropped and untilled area under the canopy of each tree (see Fig. S5). However, these small untilled areas may be critical in providing alternative nectar sources, nesting sites, and shade or shelter for wild bees in otherwise homogenous fields. Therefore, the cost of crop yield losses from scattered trees should be weighed up against the potential pollination benefits gained. This additional function in providing refuge for pollinators warrants further attention. Finally, the presence of watering points had a positive effect on bee species richness in fields, possibly because either access to water is a limiting factor, or because bare dam banks were also providing additional nesting sites for ground-nesting species.

What management actions can be taken to encourage the persistence of wild bees in agricultural landscapes?

Based on these results, it appears that wild bees have the ability to persist in highly modified landscapes, provided appropriate refugia such as linear remnants and scattered trees are available. This finding is in concordance with other studies which have found that small land concessions for suitable nesting sites and alternative sources of forage allow for the continued provision of pollination services by bees (Banaszak, 1992; Gathmann and Tscharntke, 2002). Our results include some positive news for the management of bee populations in modified landscapes. Firstly, smaller (and highly abundant) bee species, which we would not expect to be trapped so far into the fields if they were dispersing from outside, were in fact present. This indicates that they may be nesting within the fields by taking advantage of small untilled areas such as the bases of trees or “no-till” systems, or simply are not limited by distance to nesting sites as some other species may be. What is also encouraging is that some potential alternative pollinator candidates appear quite resistant to modification. For example, *A. chlorocyanea* was captured at 70 of the 104 trapping points, in all land use types and at all distance classes, and unusually showed a negative response to the number of trees near traps. It is one of the larger-bodied species, is capable of sonication (‘buzz pollination’), is partially gregarious, and 25 species of this genus are known to occur across Australia (Dollin et al., 2000). It has also already been shown to be an effective pollinator of tomatoes in greenhouse experiments (Hogendoorn et al., 2006).

In terms of landscape management, remnants that are closer to other conservation reserves and further from infrastructure will support more diverse bee communities. Tree cover will be important for ensuring the abundance of individual species, so wooded linear remnants are themselves an important component of agricultural landscapes. Land holders who wish to encourage visitation of bees to their properties should take action to retain or plant scattered trees, establish or retain patches of remnant vegetation, and also maintain a watering point in the close vicinity. Ultimately, further research into the native bee communities of rural Australia is warranted because even basic information on distribution and abundance is still lacking. Specifically, it would be beneficial to gain a more in-depth understanding of how remnant vegetation (including scattered trees in agricultural fields) fulfils the nesting requirements of common versus rare species. Also, visitation by bees does not necessarily equate to pollination, as pollination efficiency varies greatly between bee species and is dependent upon the target plant (Cane, 2002). Hence, studies that directly assess pollination by wild bees are needed. On the basis of current information and the findings of this study, the proposed conversion of remnants to farmland would cause a reduction in regional diversity of wild bees, as well as local abundance and species richness. This in turn may have a negative impact on surrounding farmland, with the loss of services from ‘free’ unmanaged pollinators. We therefore recommend, for the future productivity of agricultural industries that currently rely on pollination by *A. mellifera*, the maintenance of mosaics where cropping and native remnant vegetation coexist.

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Tables and Figures

Table 1. Sets of predictor variables used in a) Generalised linear modeling of bee species richness in linear remnants (see Table 3) and b) Generalised linear mixed modeling of bee species richness in fields (see Table 4)

a) Bees in remnants	
Set name	Variables in set
GROUND	Per cent flowering ground cover + Per cent native ground cover
TREES	Trees in 100m radius
FIELD	Land use in the adjacent field
CONTEXT	Proportion conservation land in 1km radius + Proportion infrastructure in 1km radius
COND	Average rainfall (mm) + Number of traps remaining suspended
b) Bees in fields	
Set name	Variables in set
GROUND	Per cent flowering ground cover + Per cent native ground cover
FIELD	Land use in the field + Presence/absence of dam + No. of trees in 100m radius
REMNANT	Width of adjacent remnant
CONTEXT	Proportion conservation land in 1km radius + Proportion infrastructure in 1km radius
COND	Average rainfall (mm) + Number of traps remaining suspended

Table 2. Bee species collected in the study, including measured average intertegular span (IT), and predicted 'maximum homing distance' (MHD). The abundance of species in each of the land use types, as well as the total number collected, is listed. * indicates species for which abundance was modelled using ZINB regression, and ** ZIP regression. See Table S3 for an expanded version of this table, which includes distances.

SPECIES	IT (mm)	MHD (m)	Rem	Pasture	Nectar crop	Wind crop	Tot.
Apidae							
*Amegilla (Notomegilla) chlorocyanea**	3.68	3,494	93	83	180	40	396
Amegilla (Zonamegilla) asserta	3.42	2,723		1			1
*Apis (Apis) mellifera**	3.26	2,322	9	24	60	2	95
Braunsapis diminuta	1.25	92	1				1
Thyreus waroonensis	3.03	1,802				1	1
Colletidae							
Hylaeus (Pseudhyaeus) albocuneatus	1.58	202		1			1
Megachilidae							
Megachile (Austrochile) sp.	2.50	947	1				1
Megachile (Eutricharaea) captionis	2.67	1,181	3		4		7
Megachile (Eutricharaea) sericauda	2.63	1,126			1		1
Megachile (Hackeriapis) canifrons	3.00	1,750	2	3			5
Megachile (Hackeripis) oblonga	1.84	339	5	6			11
Megachile (unplaced) atrella	2.57	1,034	1	1			2
Megachile (unplaced) callura	1.51	175	1		1		2
Megachile (unplaced) heriadiformis	2.19	609	5				5
Megachile (unplaced) semiluctuosa	4.21	5,477	2			1	3

SPECIES	IT (mm)	MHD (m)	Rem	Pasture	Nectar crop	Wind crop	Tot.
Halictidae							
*Homalictus (Homalictus) sphecodoides**	0.96	38	13	9	27	8	57
*Homalictus (Homalictus) urbanus***	1.00	43	37	4	4		45
*Lassioglossum (Chilalictus) aspratulum***	1.11	62	3	7	11	4	25
*Lassioglossum (Chilalictus) cambagei**	1.64	230	196	93	114	69	472
*Lassioglossum (Chilalictus) clelandi***	1.94	404	5	6	4		15
*Lassioglossum (Chilalictus) cognatum**	1.31	108	150	113	65	41	369
*Lassioglossum (Chilalictus) ebeneum***	1.77	297	12	2	28	2	44
*Lassioglossum (Chilalictus) expansifrons**	1.53	179	40	47	492	38	617
Lassioglossum (Chilalictus) helichrysi	1.50	170	2	4	5		11
*Lassioglossum (Chilalictus) hemichalceum***	1.07	54	45	57	36	15	153
*Lassioglossum (Chilalictus) imitator**	1.45	151	6	30	175	1	212
*Lassioglossum (Chilalictus) lanarium**	2.03	467	136	89	223	65	513
Lassioglossum (Chilalictus) mundulum	1.35	119			3		3
Lassioglossum (Chilalictus) sexsetum	0.80	20			2		2
Lassioglossum (Chilalictus) speculatum	1.58	202		1			1
Lassioglossum (Chilalictus) willsi	1.66	238	3	3	3	3	12
Lassioglossum (Parasphecodes) sulthicum	1.78	300	1	1	1		3
Lipotriches (Austronoma) australica	2.43	866	2				2
*Lipotriches (Austronoma) cf flavoviridis***	1.45	150	10	4			14
*Lipotriches (Austronoma) cf moerens**	1.76	292	56	36	35	19	146
Lipotriches sp.	1.71	264	1				1

Table 3. Outcomes of generalised linear modeling of bee species richness in linear remnants a) 95% confidence set of models b) Coefficient estimates for the final model. See Table 1a for variables included in variable sets.

a) 95% confidence model set								
Model Rank	CONTEXT	GROUND	FIELD	COND	TREES	Log(*L*)	AIC	w_i
1	✓	✓	✓			-63.50	140.99	0.208
2	✓	✓	✓		✓	-62.85	141.69	0.146
3	✓	✓		✓		-63.00	141.99	0.126
4	✓	✓	✓	✓		-61.07	142.19	0.117
5	✓	✓	✓	✓	✓	-60.37	142.69	0.087
6	✓		✓	✓	✓	-62.53	143.09	0.074
7	✓	✓		✓	✓	-62.58	143.19	0.070
8	✓			✓	✓	-64.85	143.69	0.054
9	✓		✓		✓	-66.13	144.29	0.041
10	✓	✓				-64.85	144.79	0.031
Relative imp.	0.955	0.786	0.673	0.529	0.472			

b) Coefficient estimates (CONTEXT + GROUND + FIELD)		
Term	Coefficient	Standard error
Intercept	1.244	0.180
Conservation land	0.500	0.164
Adjacent land – pasture	0.353	0.187
Infrastructure	-0.261	0.299
Adjacent land – nectar-bearing	-0.251	0.284
Flowering ground cover	-0.222	0.090
Native ground cover	-0.156	0.076

Table 4. Outcomes of generalised linear mixed modeling of bee species richness in fields a) 95% confidence set of models b) Coefficient estimates for the final generalised linear mixed model. See Table 1b for variables included in variable sets.

a) 95% confidence model set								
Model Rank	FIELD	COND	GROUND	REMNANT	CONTEXT	Log(*L*)	AIC	w_i
1	✓	✓	✓			-54.62	131.2	0.241
2	✓	✓				-56.89	131.7	0.185
3	✓	✓	✓	✓		-54.12	132.2	0.146
4	✓	✓		✓		-56.10	132.2	0.150
5	✓	✓	✓	✓	✓	-52.66	133.3	0.086
6	✓	✓		✓	✓	-54.88	133.7	0.069
7	✓	✓	✓		✓	-54.01	134	0.060
8	✓	✓			✓	-56.13	134.2	0.054
Relative imp.	0.989	0.989	0.533	0.450	0.268			

b) Coefficient estimates (FIELD + COND + GROUND)		
Term	Coefficient	Standard error
Intercept	0.704	0.222
Land use – nectar-bearing crop	0.779	0.200
Land use - pasture	0.623	0.187
Traps remaining suspended – two	0.453	0.142
Traps remaining suspended - none	-0.428	0.191
Dam	0.202	0.171
Trees in 100m radius	0.134	0.073
Flowering ground cover	-0.112	0.072
Native ground cover	-0.098	0.074
Rainfall (mm)	-0.035	0.075

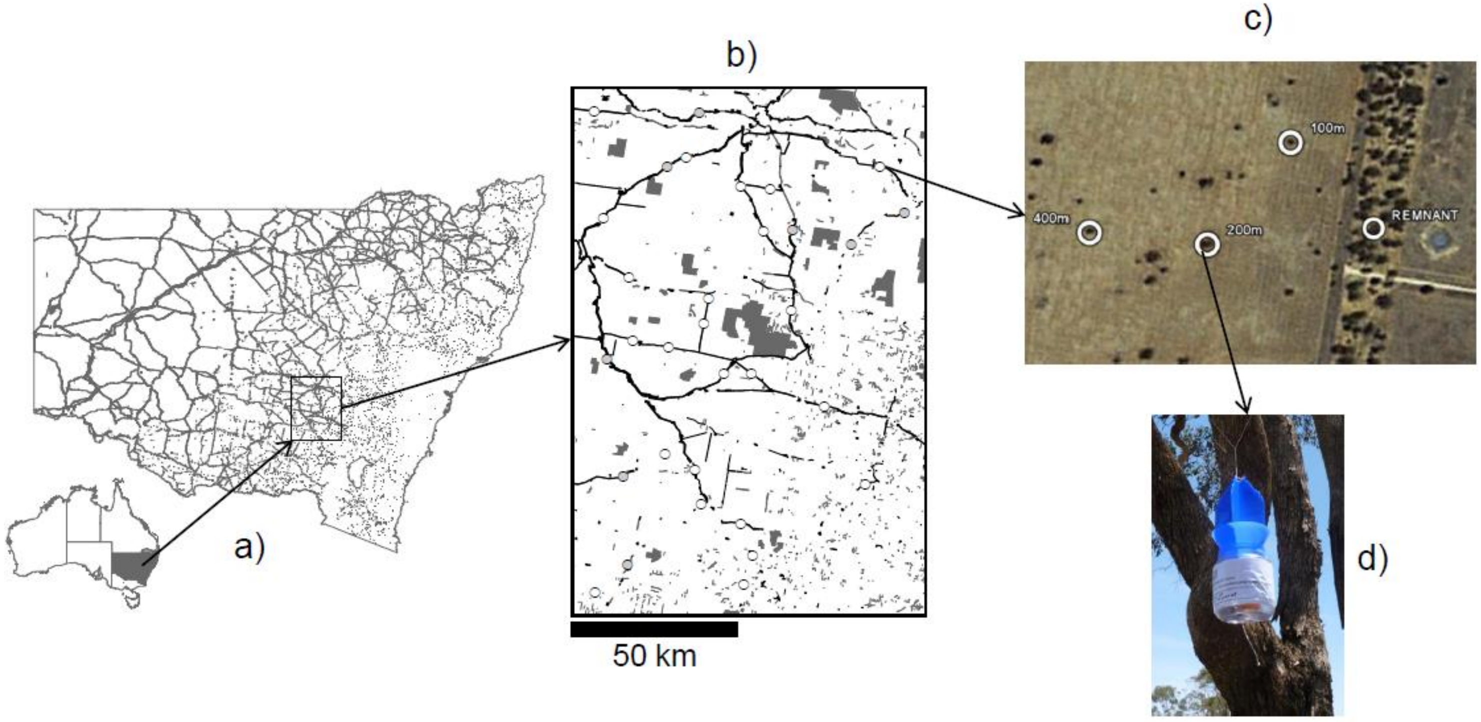

Figure 1. Schematic showing a) the extent of the linear remnant network across New South Wales, b) the position of study sites within the landscape, with remnants shown in black. Light-grey circles indicate traps were placed only in the remnants, and white circles that traps were in both the remnant and the adjacent field. Conservation areas are dark grey, c) study site showing trapping point in the remnant and at 100, 200 and 400m into the field, d) blue vane trap suspended from a tree branch.

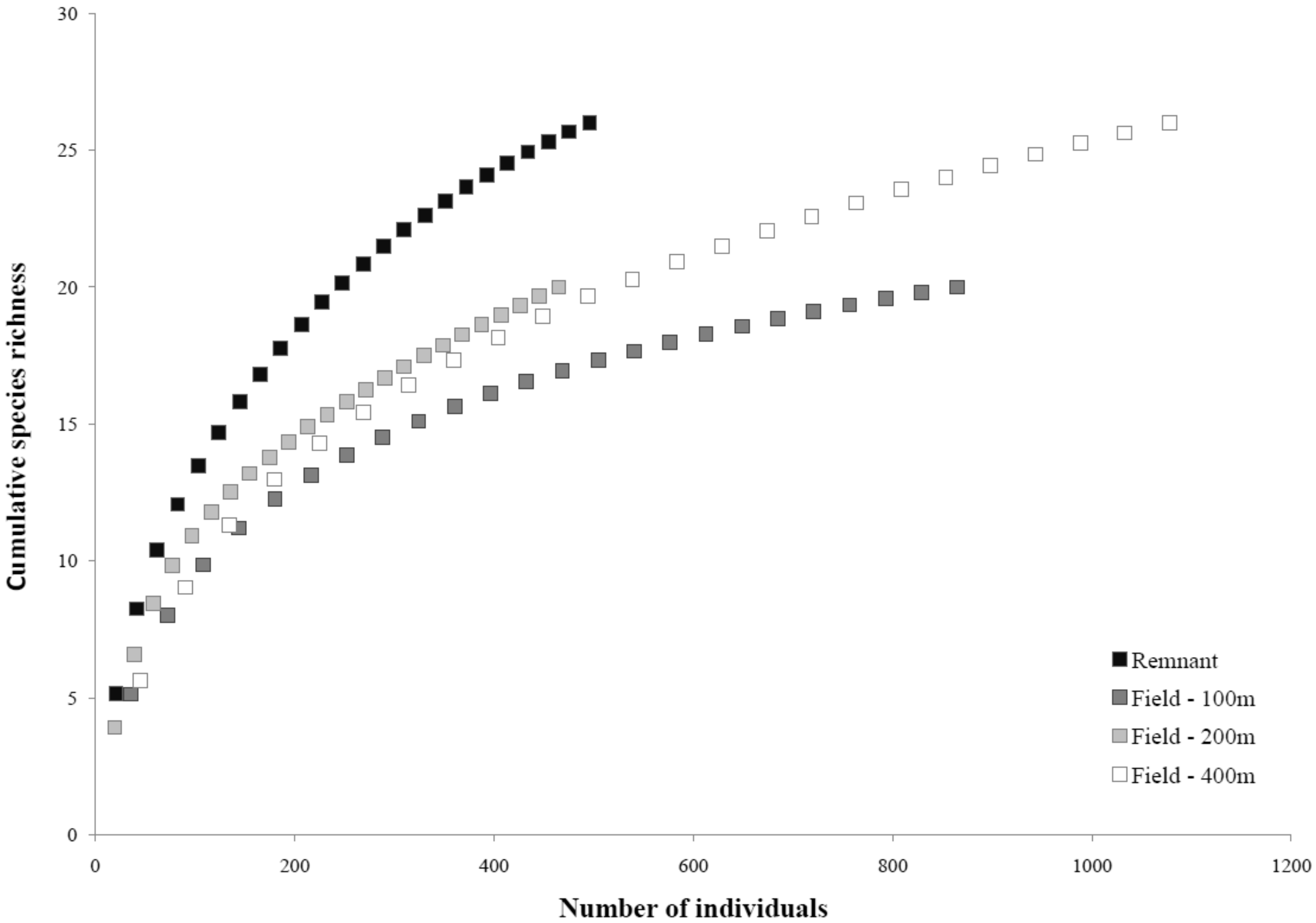

Figure 2. Species accumulation curves based on 24 trapping points in remnants, and 24 at 100, 200 and 400m in the adjacent fields.

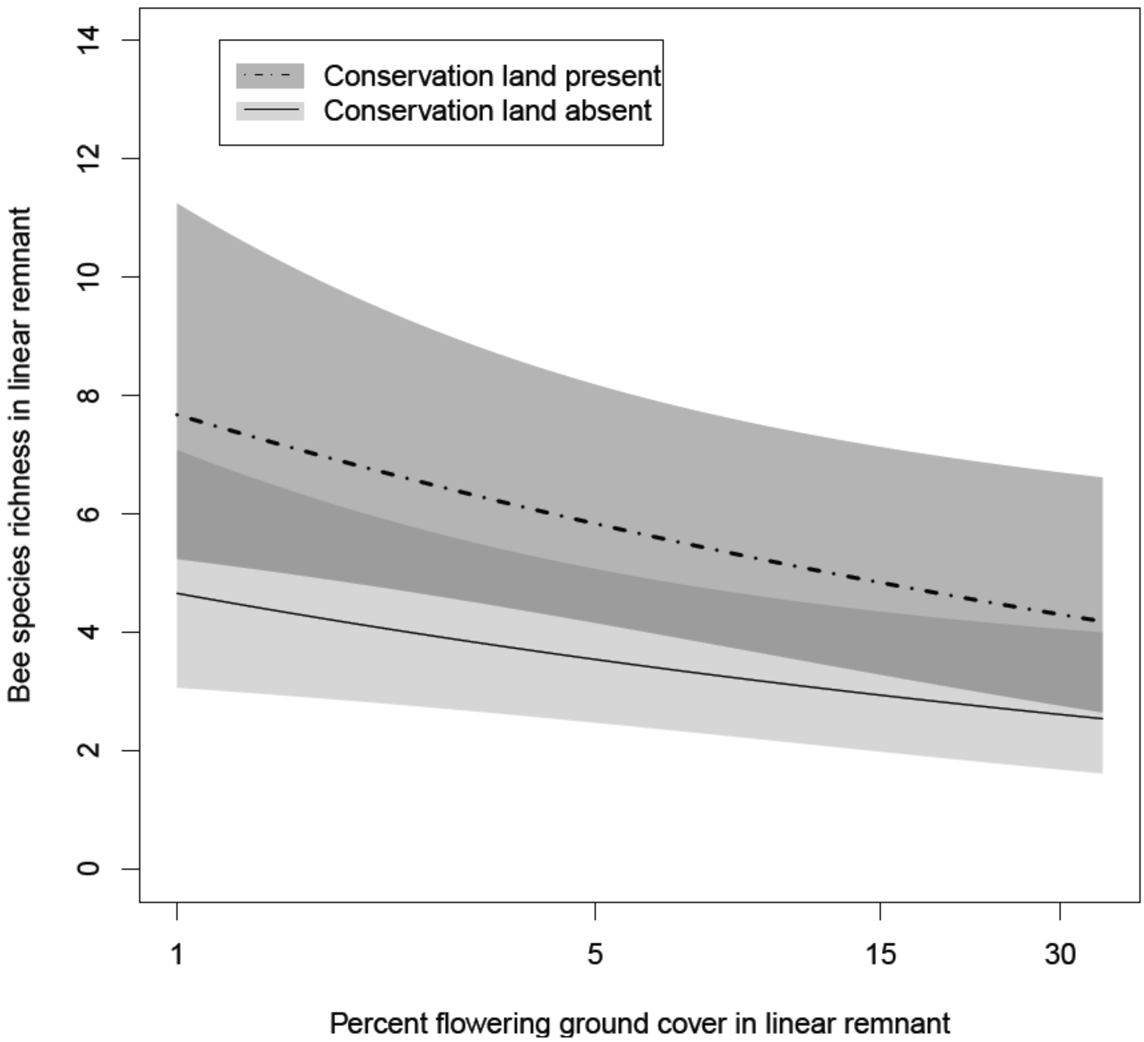

Figure 3. Negative relationship of bee species richness in linear remnants with percent flowering ground cover in a 100m radius. Model coefficients are shown in Table 3b. Curves depict remnants where conservation land is present within a 1km radius, and those where it is absent. Transparent polygons represent 95% confidence intervals.

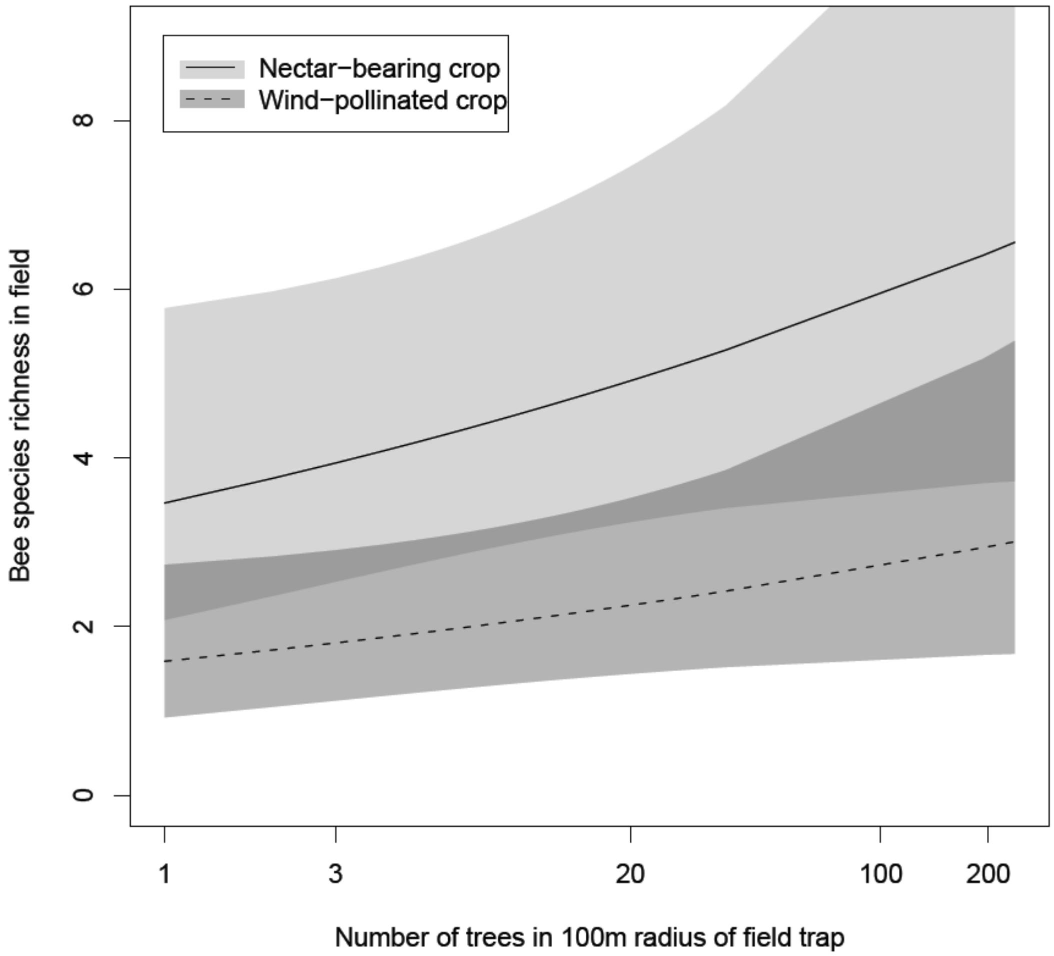

Figure 4. Positive relationship of bee species richness in fields with the number of scattered trees in a 100m radius. Model coefficients are shown in Table 4b. Curves depict nectar-bearing crop (canola and lucerne) samples and wind-pollinated crop (wheat) samples. Transparent polygons represent 95% confidence intervals.

Supplementary material

Table S1. Loadings of component 1 and component 2 from the principal components analysis. These two components were subsequently used as new variables to describe site modification.

Variable	Component 1	Component 2
Volume of logs (m^3)	-0.551	
Native ground cover	-0.621	
Exotic ground cover	0.415	-0.684
Ground covered by cropping	0.371	0.371

Table S2. Model estimates and parameters for each of the variables included in final species abundance models, taken from the zero-inflated regression analysis (analysis 2.3.2.; “Individual species abundance”.) Component 1 is a proxy for level of landscape modification, ranging from low scores in the linear remnants to the highest scores in cultivated fields. Component 2 separates sites with a pasture-like structure from taller crops.

	Lassioglossum lanarium				***Homalictus sphecodoides***				***Lassioglossum hemichalceum***			
	Est	SE	z	*P*	Est	SE	z	*P*	Est	SE	z	*P*
Intercept	0.68	0.43	1.58	0.114	-1.77	0.70	-2.53	0.011	-0.92	0.58	-1.58	0.115
Component 1 score	0.56	0.19	2.99	0.003	0.76	0.31	2.45	0.015	0.31	0.09	3.66	<0.001
No. trees in 100m	0.31	0.14	2.23	0.026	0.46	0.21	2.16	0.031	0.20	0.07	3.04	0.002
Conservation Land									0.65	0.19	3.38	<0.001
Component 2 score									-0.12	0.08	-1.42	0.157

	Apis mellifera				***Lassioglossum ebeneum***				***Lassioglossum expansifrons***			
	Est	SE	z	*P*	Est	SE	z	*P*	Est	SE	z	*P*
Intercept	-0.69	0.44	-1.59	0.112	0.36	0.34	1.06	0.288	0.47	0.38	1.25	0.212
Component 1 score	0.55	0.22	2.49	0.013	0.32	0.13	2.53	0.011	0.62	0.18	3.38	<0.001
Conservation Land	1.29	0.63	2.03	0.042	0.84	0.36	2.34	0.019	1.45	0.63	2.32	0.02

	Amegilla chlorocyanea				***Lipotriches flavoviridis***				***Homalictus urbanus***			
	Est	SE	z	*P*	Est	SE	z	*P*	Est	SE	z	*P*
Intercept	1.86	0.27	6.81	<0.001	-9.10	2.41	-3.78	<0.001	-12.65	3.61	-3.50	<0.001
No. trees in 100m	-0.18	0.09	-2.08	0.038	0.64	0.14	4.65	<0.001	0.78	0.10	7.48	<0.001
Width of remnant					0.91	0.42	2.16	0.031	0.78	0.10	7.48	<0.001
Component 2 score	-0.27	0.12	-2.20	0.028	-0.78	0.53	-1.49	0.138				

Table S3. Bee species collected in the study, listed in family groups. Table includes measured average intertegular span (IT), and predicted 'maximum homing distance'. The abundance of species in each of the land use types, as well as the total number collected, is listed according to the distance class at which they were trapped.

	TOTAL	IT (mm)	Max homing dist (m)	Rem	Native past			Exotic past			Lucerne			Canola			Wheat		
				0m	**100**	**200**	**400**	**100**	**200**	**400**	**100**	**200**	**400**	**100**	**200**	**400**	**100**	**200**	**400**
Apidae																			
Amegilla (Notomegilla) chlorocyanea	396	3.68	3,494	93	16	23	13	9	11	11	57	23	69	18	2	11	22	3	15
Amegilla (Zonamegilla) asserta	1	3.42	2,723							1									
Apis (Apis) mellifera	95	3.26	2,322	9	3	1				20		11		38	8	3	2		
Braunsapis diminuta	1	1.25	92	1															
Thyreus waroonensis	1	3.03	1,802														1		
Colletidae																			
Hylaeus (Pseudhyaeus) albocuneatus	1	1.58	202							1									
Megachilidae																			
Megachile (Austrochile) sp.	1	2.50	947	1															
Megachile (Eutricharaea) captionis	7	2.67	1,181	3									1	2		1			
Megachile (Eutricharaea) sericauda	1	2.63	1,126									1							
Megachile (Hackeriapis) canifrons	5	3.00	1,750	2						3									
Megachile (Hackeripis) oblonga	11	1.84	339	5			6												
Megachile (unplaced) atrella	2	2.57	1,034	1			1												
Megachile (unplaced) callura	2	1.51	175	1										1					
Megachile (unplaced) heriadiformis	5	2.19	609	5															
Megachile (unplaced) semiluctuosa	3	4.21	5,477	2														1	
Halictidae																			
Homalictus (Homalictus) sphecodoides	57	0.96	38	13			5	1		3	2		5	15	2	3	7		1
Homalictus (Homalictus) urbanus	45	1.00	43	37	1	3							4						
Lassioglossum (Chilalictus) aspratulum	25	1.11	62	3		2	5						4		3	4	4		
Lassioglossum (Chilalictus) cambagei	472	1.64	230	196	19	18	7	2	11	36	46	1	15	8	33	11	63	3	3
Lassioglossum (Chilalictus) clelandi	15	1.94	404	5			1	1	1	3				1		3			

Halictidae cont'd																			
Lassioglossum (Chilalictus) cognatum	369	1.31	108	150	20	48	23	14	4	4	23	1	10	2	14	15	38	1	2
Lassioglossum (Chilalictus) ebeneum	44	1.77	297	12		1			1		4		4	4	5	11	2		
Lassioglossum (Chilalictus) expansifrons	617	1.53	179	40	12		4	3	24	4	8		16	53	35	380	36	1	1
Lassioglossum (Chilalictus) helichrysi	11	1.50	170	2	3					1			3		2				
Lassioglossum (Chilalictus) hemichalceum	153	1.07	54	45	1	7	7	10	22	10	11		2	9	8	6	5	8	2
Lassioglossum (Chilalictus) imitator	212	1.45	151	6	5			24		1			1	36	1	137	1		
Lassioglossum (Chilalictus) lanarium	513	2.03	467	136	1	3	8	11	24	42	69	4	20	22	61	47	62		3
Lassioglossum (Chilalictus) mundulum	3	1.35	119													3			
Lassioglossum (Chilalictus) sexsetum	2	0.80	20										2						
Lassioglossum (Chilalictus) speculatum	1	1.58	202						1										
Lassioglossum (Chilalictus) willsi	12	1.66	238	3			1	2			1		2				1		2
Lassioglossum (Parasphecodes) sulthicum	3	1.78	300	1						1		1							
Lipotriches (Austronoma) australica	2	2.43	866	2															
Lipotriches (Austronoma) cf flavoviridis	14	1.45	150	10	1		1	1	1										
Lipotriches (Austronoma) cf moerens	146	1.76	292	56	2	8	6	9	3	8	4		10	5	10	6	11	5	3
Lipotriches sp.	1	1.71	264	1															

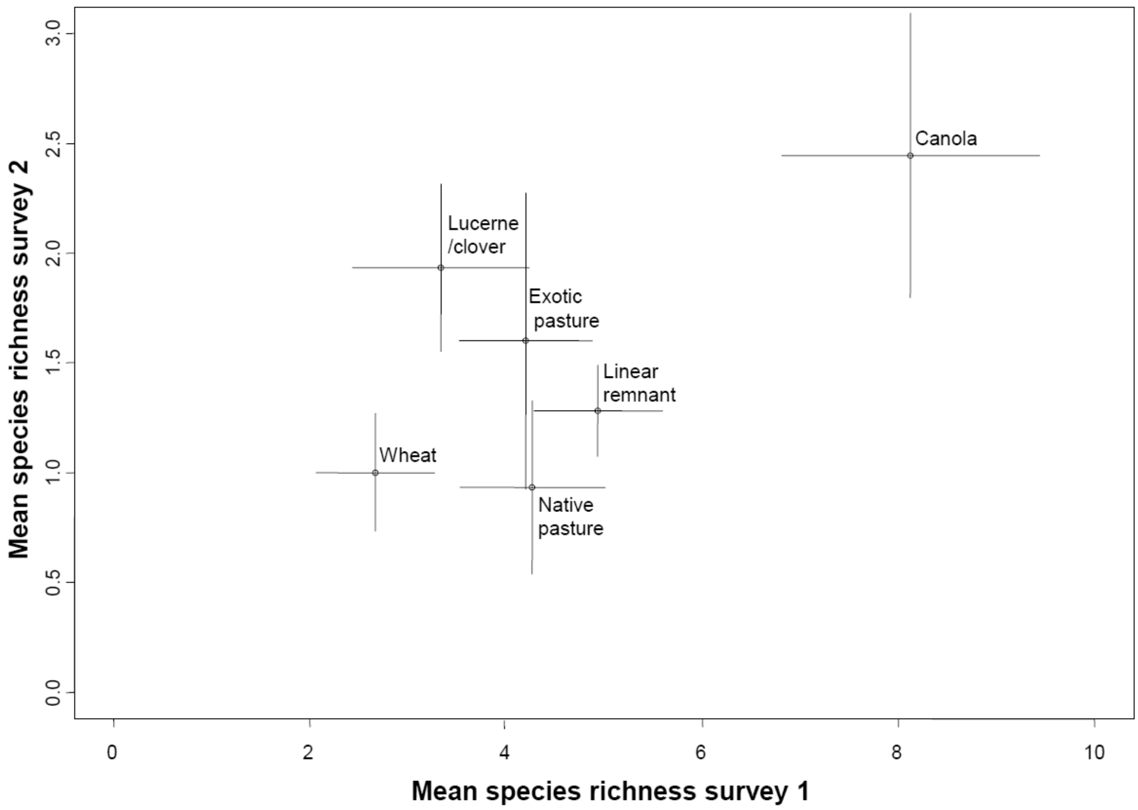

Figure S1. Plot showing mean species richness recorded at trapping points in each of the different land use types during survey 1 (November-December 2009) and survey 2 (January-February 2010), ± SE. Wheat and canola fields were harvested between the two survey periods.

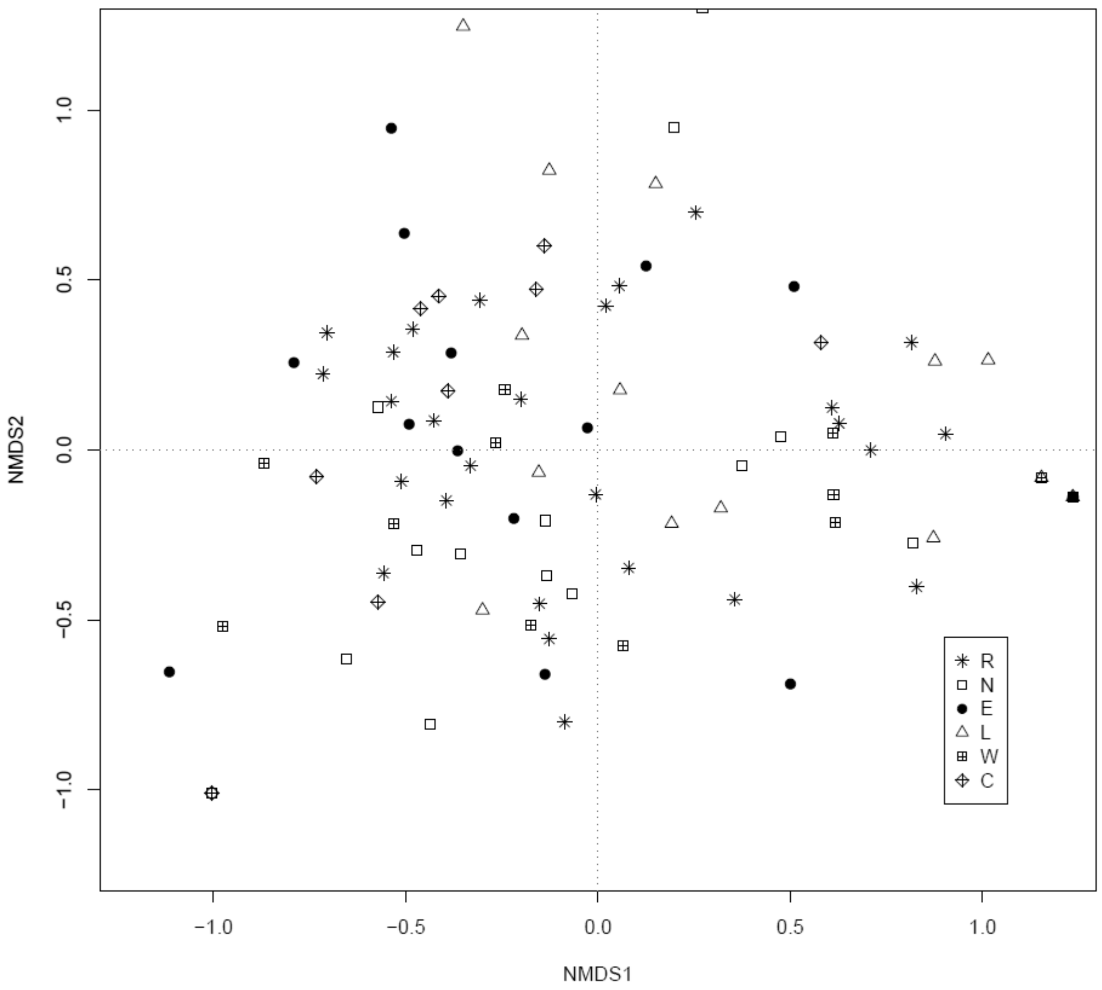

Figure S2. Plot resulting from non-metric multidimensional scaling (NMDS) of bee community composition, showing the spread of site types. The NMDS was based on a square-root transformed abundance matrix of species. Land use codes correspond to: R = linear remnant, N = native pasture, E = exotic pasture, L = lucerne, W = wheat and C = canola. Stress = 21.42.

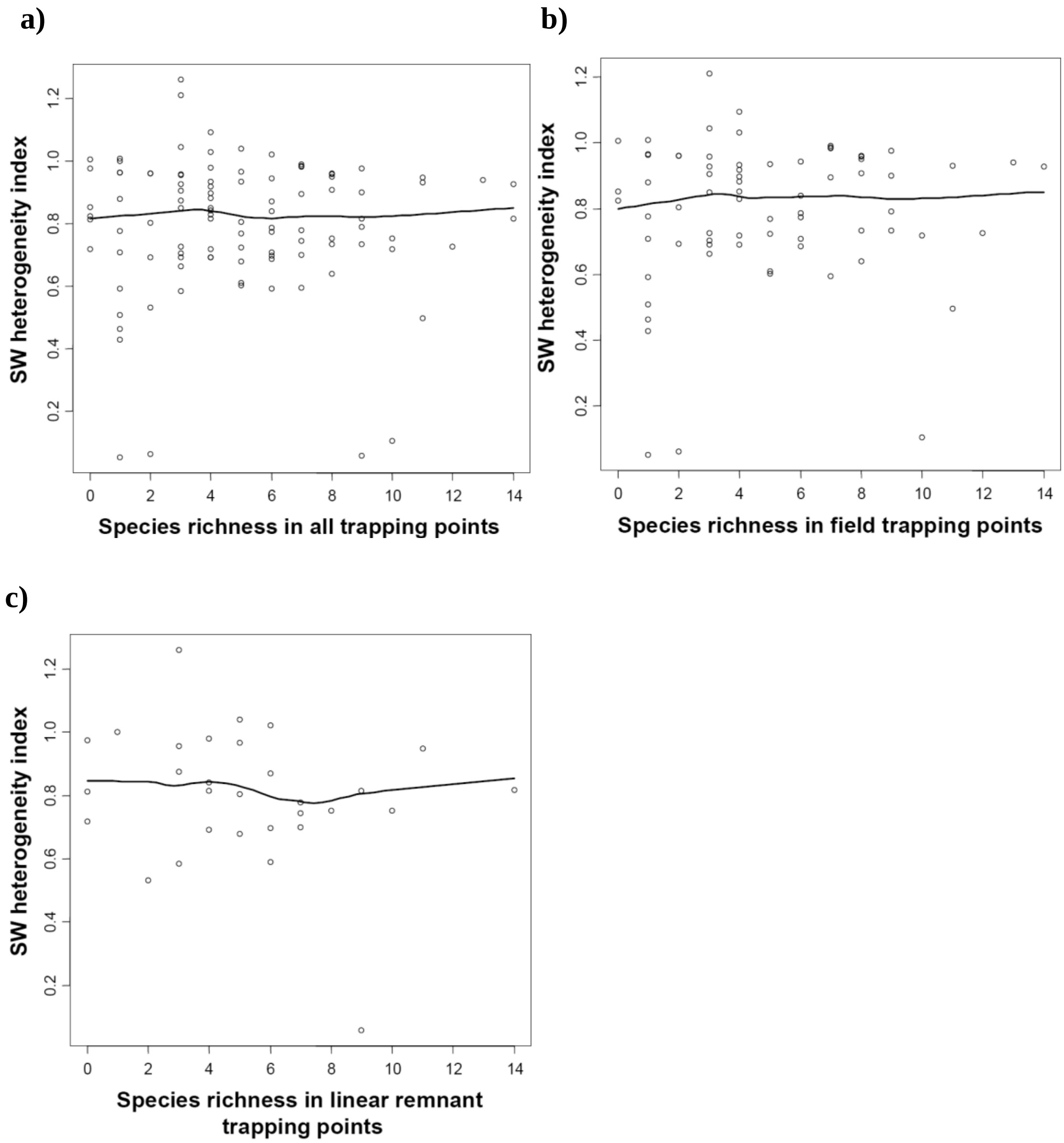

Figure S3. Plots showing the relationship between Shannon-Wiener (SW) heterogeneity and species richness recorded at a) all trapping points b) trapping points in fields and c) linear remnant trapping points. Landscape heterogeneity was quantified using a geographic information system, and the "Land Use: New South Wales" spatial data set, dated 8th April 2011 and supplied by the NSW Department of Environment, Climate Change and Water. A 1km buffer was drawn around each of the trapping points, and within this buffer the proportion of area covered by the following land uses was recorded: conservation area, cropping, grazing, horticulture, intensive animal production, mining and quarrying, river and drainage system, farm infrastructure, transport and other corridors, tree and shrub cover, and urban. The Shannon-Wiener heterogeneity index was then calculated for each trapping point based on these proportions.

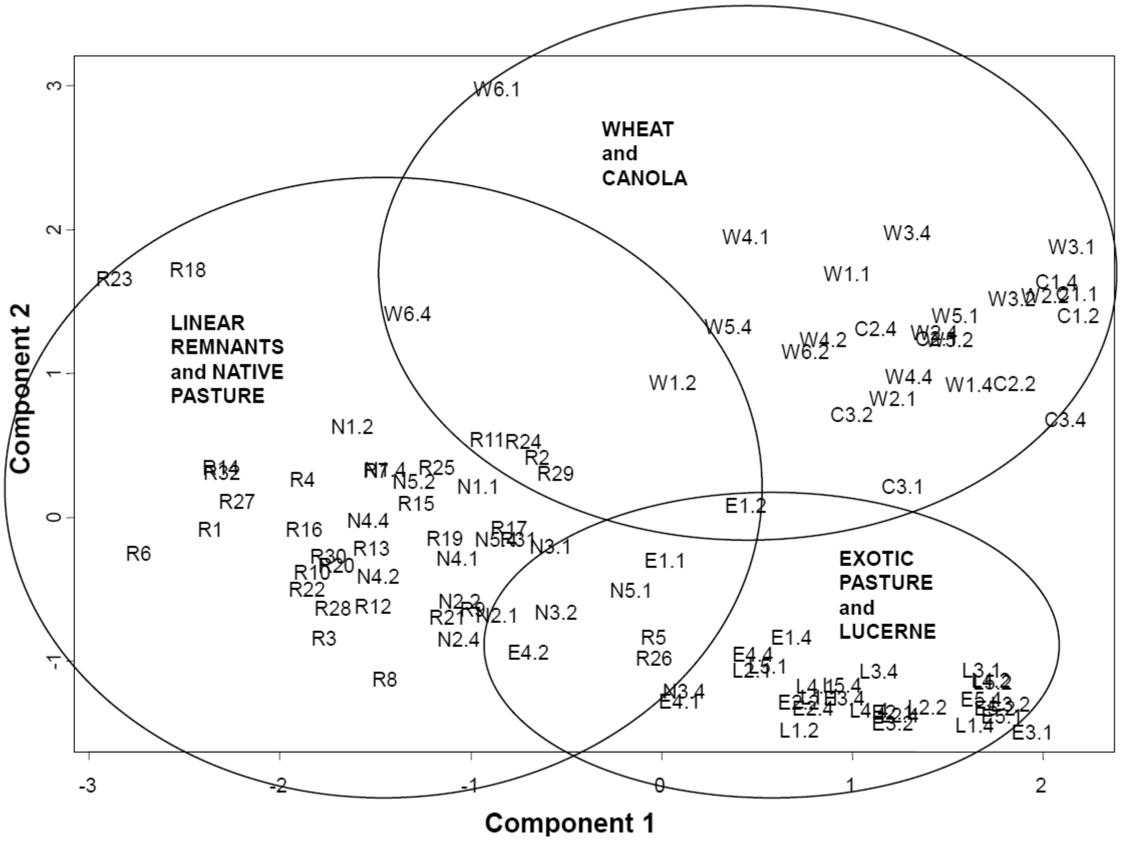

Figure S4. Plot resulting from the principal components analysis, which effectively separated different land use types along components 1 and 2 according to level of modification.

a)

b)

Figure S5. Images showing untilled area underneath scattered trees in cultivated fields of a) wheat and b) canola. This untilled area is less conspicuous in the canola field as it has been colonised by other weedy Brassicaceae species (Rocket). Other common flowering species occurring under these trees included *Echium plantagineum* (Patterson's curse), *Capsella bursa-pastoris* (Shepherd's purse), *Marrubium vulgare* (Common horehound), and *Hypochoeris radicata* (Cat's ear), which may have provided an additional source of forage for bees after fields had been harvested.

Paper IV. Value of large-scale linear networks for bird conservation: A case study from travelling stock routes, Australia

View from the “Beena” wheat field in Piney Range, NSW - waiting for the sun to rise over the Weddin Mountains National Park, so a bird survey can commence. The “Causes Driftway” TSR can be seen in the background. Image: P. Lentini.

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Abstract

We investigated the potential role of the travelling stock route network, Australia, in the conservation of declining birds. We surveyed 32 linear remnants and 24 adjacent agricultural fields of crop, native pasture or exotic pasture, for woodland birds. Compared to surrounding agricultural fields, linear remnants provided better habitat for woodland birds. Within the remnants, vegetation structural complexity was a better predictor of woodland bird richness than remnant width. In the fields the highest number of species was found in native pastures, and there was also a positive association with the number of scattered trees retained. Interestingly, there was a negative association with the width of the stock route running next to the field, with narrower linear remnants providing a greater source of avian visitors to farmland. Our findings suggest that investments in woodland bird habitat may be best spent protecting smaller, better quality remnants, or enhancing structural complexity of the vegetation already present. Existing networks of linear remnants present a low cost opportunity for regional scale conservation across extensively cleared agricultural landscapes.

Introduction

Worldwide, arable lands have been cleared extensively of native vegetation to make way for livestock production and agriculture (Foley et al., 2005; Hoekstra et al., 2005). This has caused a shift in the composition of ecological communities, with those species limited to remnant vegetation facing the synergistic effects of multiple threatening processes such as invasive species, habitat degradation and microclimatic changes (Fahrig, 2003; Saunders et al., 1991). Wildlife corridors have been suggested as a possible means of ameliorating the effects of habitat fragmentation (Beier and Noss, 1998; Levey et al., 2005), but the benefit of establishing new corridors is disputed (Bennett, 2003; Hodgson et al., 2009; Simberloff et al., 1992). In addition to improving connectivity, wildlife corridors may enhance ecological communities in adjacent farmland as individuals disperse or 'spill-over' to the surrounding areas (Brudvig et al., 2009; Hess and Fischer, 2001).

In some cases, linear elements that might function as wildlife corridors already exist in the landscape as a result of historical land use and transportation networks. Examples include hedgerows (Fuller et al., 2001), abandoned railways (Poague et al., 2000), and drainage ditches (Geert et al., 2010), all of which can be found across Europe and North America, and which have received recognition for their role in biodiversity conservation (Marshall and Moonen, 2002; Vickery et al., 2009). In Australia, travelling stock routes form an extensive network of connected linear habitat (see supporting information). This system of remnants of native vegetation spans the eastern length of the continent and was established prior to broad-scale clearing for agriculture, for the purpose of droving livestock between pastures and to markets. Comprising over 3.2 million ha in the states of Queensland and New South Wales (NSW) alone, the network is made up of

(i) the stock routes, historically used for daytime stock movements, and (ii) stock reserves, which are larger patches where stock were kept overnight (McKnight, 1977). Most stock routes have a sealed or unsealed road running through them (Spooner, 2005). This study focuses on the stock routes (referred to as “linear remnants” hereafter) of the wheat-sheep belt of NSW, which traverse low-lying and fertile areas. The condition of these remnants across the state is highly variable, but many remain intact and some comprise several threatened ecological communities which are poorly represented in the protected area system (Prober and Thiele, 1995).

Historically, stock routes have been administered by Rural Land Protection Boards (RLPBs), which managed them for both agricultural and conservation purposes (Davidson et al., 2005). However, in 2008 an independent review of the structure, administration and services offered by the RLPB system found that stock are now often transported by other means, calling into question whether the network should be maintained. The review recommended that the network be handed over to the NSW Land and Property Management Authority (Integrated Marketing Communications, 2008). The resulting change in management now places uncertainty over the future of the network as the authority considers selling off portions to private landholders (Land and Property Management Authority, 2010). To date, little research has quantified the value of the network from a wildlife conservation perspective, and in particular which sections are most valuable (but see Channing, 2000; Freudenberger and Drew, 2001; Lindenmayer et al., 2010).

Woodland birds are of particular conservation concern in southeastern Australia, having experienced population declines for several decades as a result of habitat loss, invasive species and changes to land use (Ford et al., 2001). These birds occupy both remnants of woodland as well as the matrix formed by agricultural fields (though to a lesser extent). We investigated how woodland birds use the network of linear remnants, and also whether the network influences bird communities on adjacent agricultural land. Specifically, we posed three questions: (1) What is the habitat value of linear remnants for woodland birds compared with the surrounding agricultural matrix? (2) Which characteristics of linear remnants are related to high woodland bird richness? (3) How do linear remnants influence bird use of adjacent farmland?

Methods

The study took place in the South Western Slopes Bioregion of the state of NSW, Australia, within an area covering approximately 15,000 km^2 and bounded by 33-34°S and 147-148°E. The Bioregion is located within the wheat-sheep belt, which is dominated by livestock grazing and crop production (Briggs et al., 2007). Much of the woody vegetation in the region was cleared following European settlement, and although deforestation has now mostly ceased, today only approximately 16% of native vegetation cover remains (Pressey et al., 2000). Most remnant vegetation occurs on less fertile ridges in national parks or state forests, and as scattered trees and small patches on private land (Fischer et al., 2010; Gibbons and Boak, 2002). Travelling stock routes are an exception from this general pattern, because they often traverse valleys and fertile parts of the landscape which have otherwise been cleared for agriculture.

The dominant vegetation assemblages of the study area are grassy box woodlands, including inland grey box (*Eucalyptus microcarpa*), but also the white box (*Eucalyptus albens*)/yellow box (*Eucalyptus melliodora*)/Blakely's red gum (*Eucalyptus blakelyi*) communities, which are listed as endangered ecological communities under the NSW

Threatened Species Conservation Act. Grassy box woodlands are characterised by an open canopy of up to 60% cover, a sparse mid-storey of shrubs (e.g. *Acacia* and *Dodonaea*), and a high diversity of grasses and forbs (Prober and Thiele, 1995).

The agricultural landscape is made up of a mixed matrix of both pasture and cropping, with landholders often producing both livestock and grain. Many pastures used for sheep and cattle production are dominated by exotic annual grasses, or are sown with clover (*Trifolium* spp.) and/or lucerne (*Medicago sativa*), and are commonly subject to inputs of fertilizers and herbicides. Pastures of native perennial grasses and saltbush (Chenopodiaceae) are less common, but also present. Cropped fields are predominantly sown with either canola (*Brassica napus*) or wheat (*Triticum* spp.*).*

Study design and site selection

Initially, 72 points along the length of the network within our study region were picked at random to form a pool of potential sites. Each of these points were rapidly assessed for vegetation condition, based on variables previously identified as important in determining habitat quality for woodland birds. During these rapid surveys the extent of tree cover, presence of shrubs and fallen timber, weediness of the ground layer and evidence of past grazing were noted. Following these assessments each route was scored subjectively out of ten for 'intactness'; those with a score <5 were deemed 'degraded' and those with a score >5 'intact'. This initial classification was later refined using quantitative measurements (see below). Linear remnant width was determined from the "Land Use: New South Wales" spatial data set supplied by the NSW Department of Environment, Climate Change and Water. Farmland plots were required to contain at least one tree, so fields which did not contain adequate tree cover were removed from the potential pool of sites. Final sites were selected to ensure an even representation of the different land use types, linear remnant width (38-570m, average width 190m) and condition.

Our final design focused on 32 linear remnants of two width ("narrow" i.e. ≤100m versus "wide" i.e. >100m) and two condition levels ("intact" and "degraded"): eight narrow-intact, eight wide-intact, eight narrow-degraded and eight wide-degraded. The linear remnants chosen incorporated the range of vegetation types typical of our study region, from densely wooded sections of the routes that included stands of white cypress-pine (*Callitris glaucophylla*), to treeless sections of natural and derived native grassland. Each 'site' consisted of a single plot in the linear remnant, or a plot in the remnant as well as three plots in the adjacent field (see Figure S1, supplementary material). The remnant plots on average incorporated 180 trees, 5% shrub cover and 15% of the ground cover was native. In the fields, an average plot had 11 trees, 4% native ground cover, and 2% shrub cover.

All survey plots covered 2 ha, but the shape of the plots in the remnants varied depending on their width. As the routes became narrower, more of the road was incorporated into the plots, to account for the fact that birds in narrower stock routes are likely to experience greater impacts from the road. 12 plots measured 100m × 200m, and 20 were 400m × 50m. We also established plots in farmland adjacent to 24 of the remnants, including 5 native pastures, 10 exotic pastures (including lucerne and clover), and 9 crop fields (wheat or canola; Table 1). Plots were established at 100, 200, and 400m from the linear remnant; and all measured 50m × 400m running parallel to the remnant. The approximate area of each of the fields was 50 ha.

Bird and vegetation surveys

Bird surveys were carried out (by PL) on fine days between late September and early November 2009, in the four hours following sunrise. This spring period corresponds to territory establishment and breeding activity for most of the bird species being surveyed. On two separate occasions spaced three weeks apart, each 2 ha plot was searched for a total of 20 min. The abundance of all bird species either sighted or heard, and exhibiting nesting behaviour or other use of the habitat (such as foraging) was recorded. Birds flying >25m overhead that had no association with the vegetation were not included. A two ha active search technique was employed because it allows the observer to effectively determine presence/absence of species within the area, and is the standard method recommended by Birds Australia for "Atlas" surveys (Barrett et al., 2007).

We also undertook vegetation surveys of each 2 ha plot. We recorded the number of *Eucalyptus* seedlings (less than 130cm in height); and for all trees, their diameter at breast height, species, and presence of hollows and peeling bark. All logs >10cm in diameter were measured for length and diameter. Using a measuring tape reel, two 50m transects were established within the plot and running perpendicular to the stock route, and ground cover was noted at points every meter along the tape (i.e. native or non-native vegetation, rock, bare ground, litter, presence/absence of livestock dung). The measurements from the two transects were then pooled, and the proportion of each ground cover category was calculated from the 100 points.

Although a grazing permit is required to place stock in a linear remnant, non-permitted grazing and agistment is quite common, especially during the study period when a 9-year drought had caused forage levels on private properties to be very low. Therefore, a count of the number of permits issued for a given remnant would likely have been an inaccurate reflection of actual grazing pressure. For this reason, we used the vegetation measures outlined above (which are likely to respond to grazing) as a proxy of grazing intensity.

The vegetation surveys in linear remnants were conducted in the same manner as in the fields, unless there were >50 trees in the plot. If this was the case, tree and log measurements were taken from two 0.1 ha sub-plots which were marked out using a measuring tape and flagging tape, and results were scaled up represent to the whole plot.

We analysed land use composition in a geographical information system. Using the "Land Use" dataset described above, 500m, 1 and 1.5km radius buffers were drawn around each of the linear remnant sites, and the proportion of the landscape allocated as cropping, pasture, infrastructure, water, linear remnant, road, and conservation land in each of these buffers was calculated, as well as the Shannon-Wiener (SW) heterogeneity index of these land uses.

Following Silcocks et al. (2005), we classified birds according to habitat association, considering "grassland species", "woodland species" and "generalists" (which were listed under both grassland and woodland guilds). A list of observed species, the site types in which they were detected, and their habitat association is provided in the online supplementary material (Table S1). We also carried out a search of the NSW Wildlife Atlas (http://wildlifeatlas.nationalparks.nsw.gov.au) to identify all woodland bird species that have been recorded in the field area previously, in order to determine which species from the regional pool were not found in the surveys (only species with greater than 10 records were considered to be a regular part of the regional pool.)

We acknowledge that in some cases, variation in detectability of species due to differences in environmental variables or observers can lead to errors in inference, if not accounted for (Rosenstock et al., 2002). However, all surveys for this study were conducted by a single observer within a single season, where environmental conditions remained stable. Also, only presence/absence data were analysed, and the open nature of the woodlands and fields in which surveys took place meant that bird species present were easily observed or heard. Hence, we consider our estimates of species presence to be reliable indices of what was actually present at the time of the surveys (Johnson, 2008), and for this reason, have not corrected for detectability.

Data analysis

Statistical analyses were divided into three parts, in line with our key questions, and were conducted in R 2.7.0 (R project for statistical computing, http://www.r-project.org/). Data from the two surveys for each plot were pooled and converted to presence/absence.

The habitat value of remnants compared to the matrix

We used principal components analysis (PCA) to determine whether the 12 habitat variables measured in our vegetation surveys could be reduced to describe ‘habitat condition’. The PCA was run using a correlation matrix, accounting for differences in variance in the predictor variables. Principal component 1 effectively separated ‘intact’ and ‘degraded’ sites, and explained 38% of the variance. On this basis, this component was considered to represent a gradient of habitat condition for the remainder of the analyses; and for ease of interpretation, the scores were inverted so that higher scores corresponded to more intact sites (see Table 2). We then used a Spearman rank correlation term to confirm that the condition scores from our initial rapid assessment of vegetation correlated with our new index. Component 1 scores of the PCA were significantly correlated with scores from the initial rapid assessment of vegetation ($S = 8539.24$, $rho = -0.57$, $p<0.001$), suggesting the initial classifications of condition were reasonable.

Non-metric multidimensional scaling (NMDS) was used on the presence/absence matrix of species to examine bird species composition in relation to the different site types. Finally, a pooled variance t-test was used to assess differences in woodland bird species richness between field and linear remnant sites.

Favorable linear remnant habitat for woodland birds

For this question, we only considered woodland birds in linear remnants. Explanatory variables were standardized to have a mean of zero and a standard deviation of one. In an initial exploratory analysis, generalised linear models were constructed containing the proportions of land uses in 500 m, 1 and 1.5 km buffers around each site. These analyses indicated that the proportion of conservation land (state reserves, national parks) in these buffers was related to species richness in linear remnants, and the effect appeared strongest at the 1.5 km scale; hence this scale was used in subsequent analyses (see next paragraph).

Next, we tested if variables describing the linear remnants, or other characteristics relating to surrounding land use, heterogeneity and landscape context were most strongly related to species richness. To do this, explanatory variables were grouped according to the scale at which they occurred. The “REM” group of variables included the width and the condition scores for each of the remnants. The “LOC” group of variables described ‘local’ landscape variables, including land use type in adjacent fields, and the SW heterogeneity of land use types in the 500 m buffer surrounding them. The “LAND” group of variables described landscape scale variables, including the proportion of conservation land and the SW heterogeneity index in the 1.5 km buffer. Finally, to capture possible interactions between linear remnant width and adjacent land with remnant condition, we created an ‘interactions’ group of variables, “INT”, which included the interactions width×condition and land use×condition (Table 3a). The ability of 11 alternative generalised linear models, which included all possible combinations of these groups of variables, was tested to explain the variability of woodland species richness in the linear remnants, assuming a Poisson error distribution.

For model selection, we used an information theoretic approach (Burnham and Anderson, 2002), adapting the methods applied by Rhodes et al. (2009), using the “car” package. Models are presented in a 95% confidence table, which lists the log likelihood (L), Akaike’s information criteria (AIC) and Akaike weights (w_i, which can be interpreted as the relative likelihood) for each model. Final model selection was based on AIC and w_i, but where two models were ranked closely as ‘best’ the “relative importance” of each of the variable groups was also taken into account (calculated by summing w_i for all of the models incorporating that variable group) (Burnham and Anderson, 2002).

The influence of adjacent linear remnants on farmland

For this analysis, we only considered woodland birds in agricultural sites. Again, the methods of Rhodes et al. (2009) were adapted, this time using the glmmML package version 0-18.3.

Nine alternative generalised linear mixed models were compared assuming a Poisson error distribution. All possible combinations of variable groups were tested, and the field at which sampling took place was treated as a random effect. Model selection procedure was the same as described in 2.3.2. Explanatory variables were grouped once again, this time addressing the question of whether birds in farmland are responding to the characteristics of the fields themselves, or if the adjacent linear remnants exert an effect. Variable groups were: “ADJ”, which included the width, condition, and woodland species richness of the adjacent linear remnant; and “FIELD”, which included the number of trees, the distance from the linear remnant, and land use type of the plot. Again, to account for possible interactions, we considered two additional groups: “DIST”, interactions with distance from the linear remnant, and “L.USE”, interactions with the land use type in the field (see Table 4a).

Results

The habitat value of remnants compared to the matrix

A total of 81 bird species were recorded in the surveys, and of these, 45 were classified as woodland species (see Table S1, online supplementary material). Our search of the NSW Wildlife Atlas revealed that 36 woodland bird species known to exist within our study region were not found in our surveys. Linear remnants contained significantly more bird species than the fields - both for all species ($t = -4.07$, $df = 102$, $p < 0.001$) and for woodland species ($t = -4.8893$, $df = 102$, $p < 0.001$). The number of woodland species ranged from 0 to 16 in the remnants ($\mu = 7.4$) and 0 to 7 in the fields ($\mu = 3.8$). While linear remnants and agricultural fields supported different bird communities, species composition was similar between linear remnants and native pastures (Fig. 1). These two land use types supported an assemblage of woodland birds, such as the grey-crowned babbler (*Pomatostomus temporalis)* and brown treecreeper (*Climacteris picumnus)*, which are listed as species of conservation concern in NSW. Conversely, exotic pastures and cropping sites contained fewer woodland species, especially with increasing distance from linear remnants (Fig. 1).

Favorable linear remnant habitat for woodland birds

The characteristics of the linear remnant itself were having the strongest effect on woodland bird richness, with 'condition' (structural complexity) a better predictor than width (Fig. 2). Support was very similar for the two highest ranked models (Table 3b). The model ranked first included both linear remnant and local landscape variables, whereas the second incorporated only remnant variables (Table 3b). However, the relative importance of the local landscape effects variable group was 0.49, compared to 0.98 for linear remnant variables.

The influence of adjacent linear remnants on farmland

Both field and adjacent linear remnant variables had an influence on woodland species richness in the fields; the model incorporating both of these groups was clearly the most parsimonious (Table 4b). Woodland bird richness was positively associated with the number of trees in field plots, and negatively associated with the width of the adjacent linear remnant (Fig. 3). Fields of native pasture supported the highest number of woodland species, followed by cropping and exotic pasture (Table 4c).

To account for a potential confounding of remnant width and the number of trees in adjacent farmland, we tested whether these variables were correlated. We found no significant correlation ($S = 57057.04$, $rho = -0.042$, $p = 0.730$, see Figure S2, supplementary material).

Discussion

Our findings clearly demonstrated that existing linear remnants provide habitat for woodland birds, many of which are declining nationally (Ford et al., 2001). However, linear remnants do not function as habitat for birds independently from the agricultural matrix. They are influenced by the intensity of land use in the surrounding areas (Martin et al., 2006), and in return can shape farmland bird communities.

The value of linear remnants for woodland birds

We found that linear remnants provided better habitat for woodland birds than adjacent fields, especially remnants with a complex vegetation structure. This finding supports previous work which suggests that structural elements such as logs and shrubs are important for a range of woodland birds (Briggs et al., 2007; Seddon et al., 2003). Nevertheless, remnants lacking tree and shrub cover may still have conservation value, because some ecological communities do not naturally support these structures, such as threatened grasslands (McIntyre and Lavorel, 1994).

Interestingly, the effect of the width of a remnant was secondary to its condition. This finding is important, because much of the past literature has emphasized the need to prioritise the conservation of larger tracts of remnant vegetation (Diamond, 1975; Major et al., 2001). We argue that, based on our case study, conservation investments in woodland bird habitat may be better spent protecting smaller but better quality remnants, or enhancing structural complexity of the vegetation already present as opposed to increasing the width of an existing remnant. As the width of a given remnant decreases, habitat in the surrounding farmland becomes more important, so incentives to promote lower chemical inputs and the establishment of trees within fields should also be considered as part of a coordinated strategy.

A total of 36 woodland bird species known to exist within our study area were not found during our surveys of linear remnants and fields. This includes 10 species of honeyeater (Meliphagidae), 5 species of robin (Petroicidae), 5 species of thornbills and gerygones (Acanthizidae), 3 cuckoos (Cuculidae) and 3 finches (Estrildidae). The lower than expected number of honeyeaters may be explained in part by the fact that 2009 was the end of a long drought for this region, and only 2% of the trees surveyed were in flower, meaning little forage was available for these primarily nectarivorous species. The smaller species such as the robins, thornbills and finches may rely on structurally complex tracts of habitat, but also possibly larger areas of contiguous habitat than our largest remnant, which was 570 m wide. Large conservation areas are sparsely distributed throughout this landscape, but probably play an important role in supporting populations of these species not found in the remnants.

Debate over the value of corridors, and whether conservation investments would be better spent on more connected habitat or higher quality habitat (Hodgson et al., 2009), has been heightened given their potential role in facilitating movement as species respond to climate change (Halpin, 1997). Although we did not attempt to determine whether stock routes functioned as movement conduits, it is clear that many of them provide suitable habitat links between formal conservation areas, with potential benefits for functional connectivity (Levey et al., 2005).

Bird communities adjacent to linear remnants

Land use had an influence on farmland avifauna in our study, with native pastures supporting the most woodland species, and being most similar to linear remnants in species composition. The case for the value of native pastures is further strengthened by the result that many woodland species, including the threatened brown treecreeper and grey-crowned babbler, were only observed in native pastures and linear remnants.

Surprisingly, bird richness in fields was highest near narrow linear remnants. A plausible explanation for this is the process of landscape supplementation, which occurs if "the population in a focal patch [is]… increased if that patch is … in a portion of the landscape that contains additional available resources" (Dunning et al., 1992). It appears that as the width of a remnant decreased, birds used adjacent land as supplementary habitat to meet foraging and nesting requirements. Birds were more likely to use adjacent farmland when there was greater cover of scattered trees, suggesting their maintenance is important to maintain farmland bird diversity (Manning et al., 2006). The visitation of birds to agricultural areas is desirable in Australia because they may assist in the pollination of scattered eucalypt trees (Ottewell et al., 2009; Southerton et al., 2004), reduce populations of crop pests (Gámez-Virués et al., 2007), and are generally appreciated by farmers (Seddon et al., 2003).

In contrast to our expectations, we found no strong distance effect on woodland species richness in fields adjacent to linear remnants. This was surprising, because Tubelis et al. (2004b) and Robertson and Radford (2009) both found that increased distance from forests caused a decrease in species richness, and willingness of birds to cross gaps respectively. The difference in findings may be caused by the fact that the contrast between our woodland remnants and the adjacent farmland was softer than those investigated in the other studies, as our fields contained scattered trees which may act as 'stepping stones' facilitating bird movements (Fischer and Lindenmayer, 2002; Gillies and St. Clair, 2010). Given this presence of stepping stones, the distance classes in our design may not have reached far enough into the fields to capture movement thresholds. Also, our surveys were conducted during the height of the breeding season, when individuals may be more willing to move into suboptimal areas to meet increased foraging requirements (Tubelis et al. 2004a), or as territories within remnants are more fervently defended, forcing birds out into the matrix (Robertson and Radford, 2009). Despite a lack of change in richness with distance, there did appear to be changes in community composition, suggesting the most sensitive woodland species were not moving great distances into fields.

The value and future management of the travelling stock route network

Travelling stock routes, like other types of linear landscape elements, pose a challenge because they need to be managed for multiple values and stakeholders. They are valuable features of the landscape which can benefit not only ecological but also human communities through the provision of ecosystem services, as recreations areas, and places of cultural significance to both European and Indigenous people (Spooner, 2005; Spooner et al., 2010). A well-managed travelling stock route network would allow for the occasional droving of stock and refuge during fire and floods, as well as substantial conservation benefits. Certain stock routes have remained in such good condition to date because previous RLPB rangers strategically grazed at certain periods to maintain a diverse ground layer (Davidson et al., 2005). If portions of the network are sold to private land holders, it is likely that they will be subject to higher grazing pressure, which limits *Eucalyptus* regeneration (Fischer et al., 2009) and may negatively impact woodland birds sensitive to grazing (Martin and McIntyre, 2007).

Travelling stock routes contain some of the most intact examples of what pre-European box woodlands may have resembled as a result of their exclusion from continuous livestock grazing (Prober and Thiele, 1995). At present stock routes are not represented in the Australian reserve network, yet within NSW they may contain a substantial proportion of high-priority conservation vegetation (Pressey et al., 2000).

As nations grapple to find cost-effective conservation strategies in the face of multiple threatening processes, management of existing networks such as travelling stock routes represent a potentially valuable and relatively inexpensive conservation option. There is enormous potential to share the costs of management between agencies and adjacent private landholders as the multiple benefits of these areas are clear. Remnant patches of vegetation within the network can act as foci from which to link regional scale conservation plans. Finally, management of the entire network would tie in with the increasingly accepted notion that the future of biodiversity conservation in agricultural areas lies in our ability to 'scale-up' and view the landscape as a whole (Fischer et al., 2010; Tscharntke et al., 2005).

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Tables and Figures

Table 1. Study design matrix showing the number of plots at each level of stratification (average remnant width, and number of trees shown in parentheses.)

		Linear remnant	Cropping	Native pasture	Exotic pasture
Narrow	Intact	8 (375 trees, 100 m)	12 (4 trees)	0 (NA)	6 (9 trees)
	Degraded	8 (91 trees, 81 m)	3 (6 trees)	3 (12 trees)	12 (6 trees)
Wide	Intact	8 (236 trees, 337 m)	3 (5 trees)	6 (74 trees)	9 (3 trees)
	Degraded	8 (23 trees, 242 m)	9 (3 trees)	6 (10 trees)	3 (5 trees)
Total		32	27	15	30

Table 2. Loadings of component 1 from the PCA of habitat surveys.

Covariate	**Loading of component 1**
Proportion of leaf litter	-0.404
Proportion of trees with peeling bark	-0.400
Total basal area	-0.367
Volume of logs	-0.333
Number of stumps	-0.288
Percent shrub cover	-0.282
Number of seedlings	-0.269
Proportion of trees with hollows	-0.228
Proportion of native ground cover	0.000
Proportion of road	0.000
Proportion of non-native ground cover	0.246
Proportion of bare ground	0.287

Table 3. Outcomes of analysis 2.3.2., of woodland birds in remnants. (a) Variable groups used to build generalised linear models. (b) The resulting 95% confidence set of models. (c) Coefficient estimates for the final generalised linear model.

a) Variable groups

Group name	Variables in group
REM	Remnant width + remnant condition
LOC	Adjacent land use + Shannon-Wiener heterogeneity (500 m)
LAND	Shannon-Wiener heterogeneity (1.5 km) + presence/absence of conservation land (1.5 km)
INT	(remnant condition * remnant width) + (remnant condition * adjacent land use)

b) 95% confidence model set

Model	Log(L)	AIC	w_i	dAIC
REM + LOC	-63.81	139.6	0.393	0
REM	-67.00	140	0.324	0.4
REM + INT	-62.92	141.8	0.130	2.2
REM + LOC + INT	-62.78	143.5	0.055	3.9
REM + LOC + LAND	-62.99	143.9	0.045	4.3
REM + LAND	-66.25	144.5	0.034	4.9

c) Coefficient estimates (REM + LOC)

Term	Coefficient	Standard error
Intercept (land use - cropping)	1.851	0.116
Remnant condition	0.422	0.094
Land use - native pasture	-0.150	0.272
Land use - exotic pasture	0.125	0.148
Remnant width	0.051	0.078
Land use heterogeneity, 500m radius	0.040	0.077

Table 4. Outcomes of analysis 2.3.3, of woodland birds in fields. (a) Variable groups used to build generalised linear mixed models. (b) The resulting 95% confidence set of models resulting from the analysis. (c) Coefficient estimates for the final, best-ranked generalised linear mixed model.

a) Variable groups

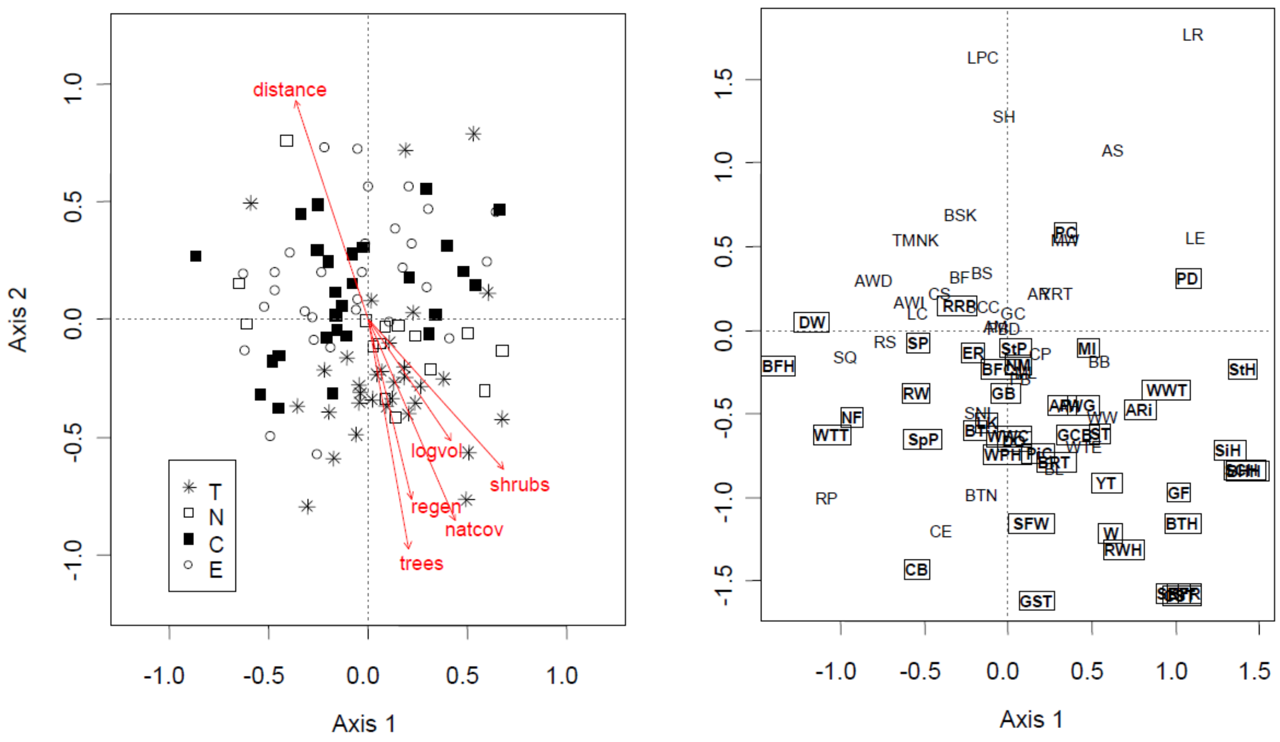
Group name	Variables in group
ADJ	Adjacent remnant width + remnant condition + remnant species richness
FIELD	Farm land use + number of trees in field plot + distance from remnant
DIST	(Distance*width) + (distance*condition) + (distance*land use) + (distance*number of trees)
L.USE	(Land use*width) + (land use*condition) + (land use*distance) + (land use*number of trees)

b) 95% confidence model set

Model	Log(L)	AIC	w_i	dAIC
ADJ + FIELD	-26.82	71.65	0.858	0
FIELD + DIST	-25.43	76.85	0.063	5.2
ADJ + FIELD + DIST	-24.78	71.55	0.045	5.9

c) Coefficient estimates (ADJ + FIELD)

Term	Coefficient	Standard error
Intercept (land use - cropping)	3.046159	0.583085
Remnant width	-0.43046	0.134501
Land use - exotic pasture	-0.39159	0.156127
Land use - native pasture	0.29447	0.190866
Number of trees in field	0.186634	0.063367
Remnant species richness	0.033549	0.02483
Remnant condition	-0.02523	0.040937
Distance from stock route	-0.00026	0.000513



F

Figure 1. Plot resulting from the NMDS, showing the spread of site types on the left and bird species on the right, with woodland species boxed in bold typeface. T = travelling stock route, N = native pasture, C = crop, and E = exotic pasture; bird species codes are listed in the online supplementary material. Significant vegetation covariate vectors: ‘distance’ = field plot distance from the remnant, ‘natcov’ = cover of native vegetation, ‘shrubs’ = presence of shrubs, ‘trees’ = number of trees, ‘logvol’ = volume of logs (m^3), ‘regen’ = number of regenerating seedlings. Stress = 28.13.

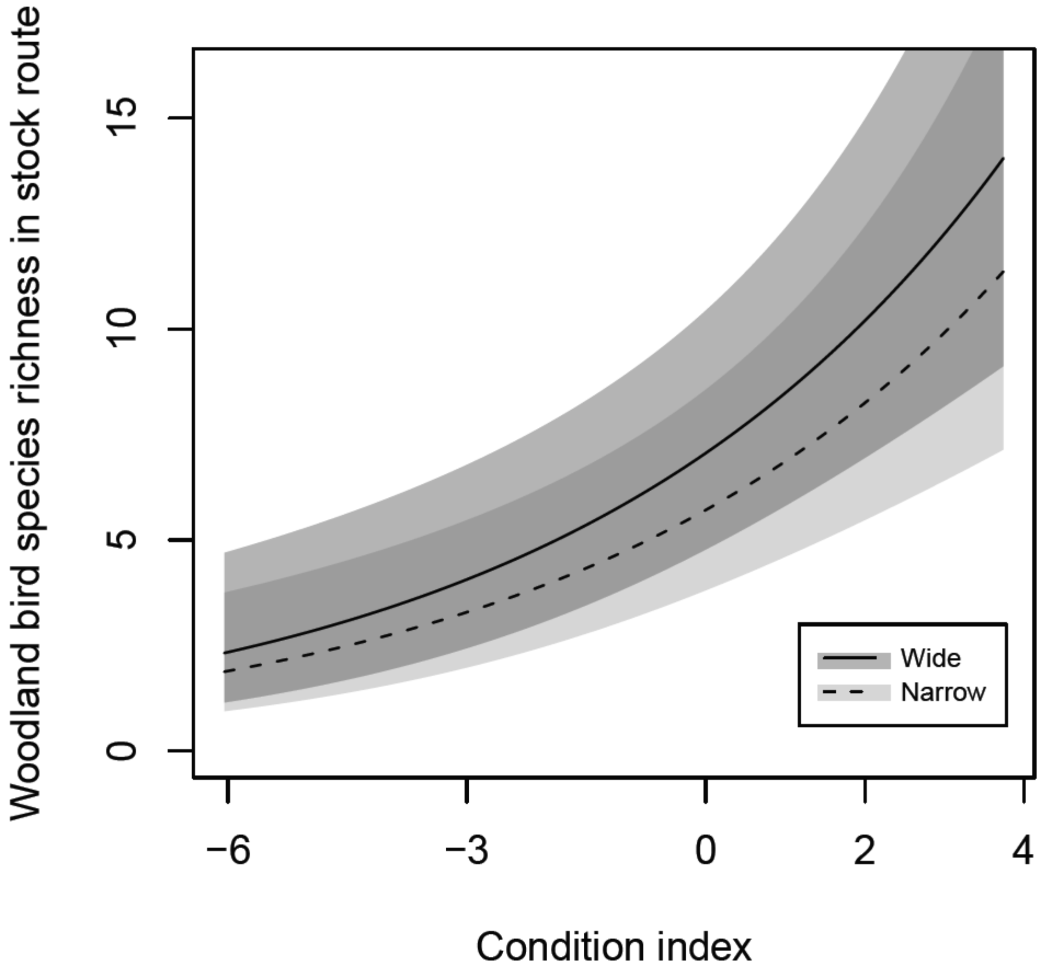


Figure 2. Relationship of woodland bird species richness in linear remnants, with remnant condition. Model coefficients are shown in Table 3c. Curves depict minimum (38 m) and maximum (570 m) remnant widths, and 95% confidence intervals are shown as polygons of different transparencies.

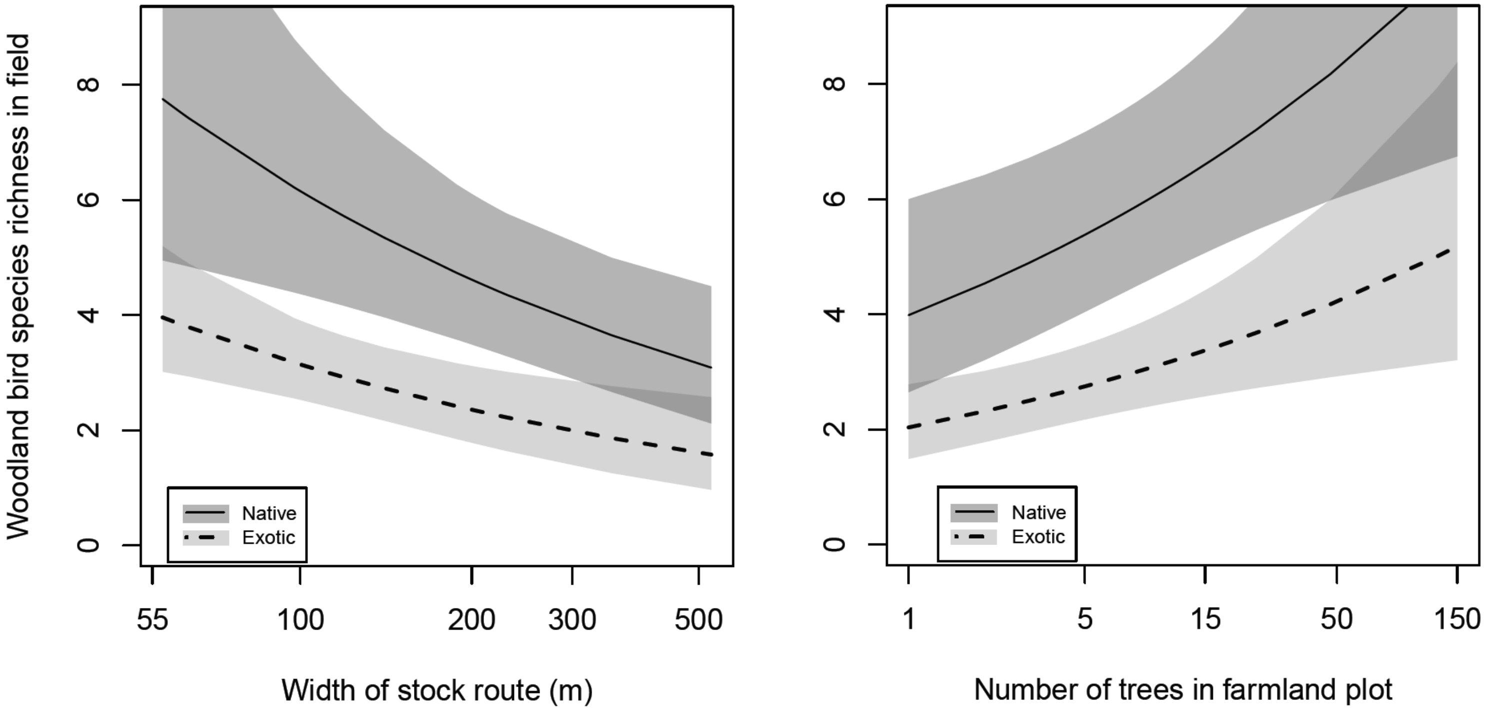


Figure 3. Relationship of woodland bird species richness in field plots with adjacent remnant width (left) and the number of trees (right). Model coefficients are shown in Table 4c. Curves depict native and exotic pastures, and 95% confidence intervals are shown as polygons of different transparencies.

Supplementary material

Table S1. List of all bird species identified in the surveys, including common and scientific names, and codes used in Figures 2. Species are listed in descending order according to the proportion of observations made in linear remnants. 'Guild' refers to whether the birds were categorised as woodland species (W), generalists (G), or open-area/grassland species (O). Columns five to eight list the proportion of the four different site types that each species was observed in, and listed below each of the land-use categories in parenthesis are the number of plots surveyed in that category. Column nine shows the total number of plots that the species was observed in. Species which were found solely in the linear remnants are shaded in light grey, and those found only in linear remnants and native pastures in dark grey.

Common Name	**Scientific name**	**Species code**	**Guild**	**Crop (27)**	**Exotic (30)**	**Native (15)**	**Linear remnant (32)**	**Total no. of plots**
Australian hobby	*Falco longipennis*	AH	G	0	0	0	1	1
Australian ringneck	*Barnardius zonarius*	ARi	W	0	0	0	1	1
Black-tailed native hen	*Gallinula ventralis*	BTN	O	0	0	0	1	1
Brown thornbill	*Acanthiza pusilla*	BTH	W	0	0	0	1	1
Brown-headed honeyeater	*Melithreptus brevirostris*	BHH	W	0	0	0	1	1
Cattle egret	*Ardea ibis*	CE	O	0	0	0	1	1
Common bronzewing	*Phaps chalcoptera*	CB	W	0	0	0	1	2
Crested shrike-tit	*Falcunculus frontatus*	CST	W	0	0	0	1	1
Dollarbird	*Eurystomus orientalis*	DO	W	0	0	0	1	1
Eastern yellow robin	*Eopsaltria australis*	EYR	W	0	0	0	1	1
Grey fantail	*Rhipidura albiscapa*	GF	W	0	0	0	1	1
Grey shrike-thrush	*Colluricincla harmonica*	GST	W	0	0	0	1	2

Common Name	Scientific name	Species code	Guild	Crop (27)	Exotic (30)	Native (15)	Linear remnant (32)	Total no. of plots
Little raven	*Corvus mellori*	LR	G	0	0	0	1	1
Peaceful dove	*Geopelia striata*	PD	W	0	0	0	1	2
Pied currawong	*Strepera graculina*	PiC	W	0	0	0	1	1
Restless flycatcher	*Myiagra inquieta*	RF	W	0	0	0	1	1
Rufous whistler	*Pachycephala rufiventris*	RWH	W	0	0	0	1	4
Sacred kingfisher	*Todiramphus sanctus*	SK	W	0	0	0	1	1
Spiney-cheeked honeyeater	*Acanthagenys rufogularis*	SCH	W	0	0	0	1	1
Straw-necked ibis	*Threskiornis spinicollis*	SNI	O	0	0	0	1	1
Striated thornbill	*Acanthiza lineata*	ST	W	0	0	0	1	4
Superb fairy-wren	*Malurus cyaneus*	SFW	W	0	0	0	1	2
Wedge-tail eagle	*Aquila audax*	WTE	G	0	0	0	1	1
Weebill	*Smicrornis brevirostris*	W	W	0	0	0	1	3
Western gerygone	*Gerygone fusca*	WG	W	0	0	0	1	5
White-winged triller	*Lalage sueurii*	WWT	W	0	0	0	1	2
Yellow thornbill	*Acanthiza nana*	YT	W	0	0	0	1	3
Grey-crowned babbler	*Pomatostomus temporalis*	GCB	W	0	0	0.25	0.75	20
White-plumed honeyeater	*Lichenostomus penicillatus*	WPH	W	0	0.25	0	0.75	4
White-winged chough	*Corcorax melanorhamphos*	WWC	W	0.11	0	0.16	0.74	19
Willie wagtail	*Rhipidura leucophrys*	WW	G	0	0.27	0.09	0.64	11
Apostlebird	*Struthidea cinerea*	AP	W	0.10	0	0.28	0.62	29
Buff-rumped thornbill	*Acanthiza reguloides*	BRT	W	0.2	0	0.2	0.6	5
Laughing kookaburrah	*Dacelo novaeguineae*	LK	W	0.14	0	0.29	0.57	7
Magpie lark	*Grallina cyanoleuca*	ML	G	0.21	0.09	0.16	0.55	44
Brown treecreeper	*Climacteris picumnus*	BT	W	0	0	0.5	0.5	2
Singing honeyeater	*Lichenostomus virescens*	SiH	W	0	0	0.5	0.5	2
White-throated treecreeper	*Cormobates leucophaeus*	WTT	W	0	0	0.5	0.5	2
Spotted pardalote	*Pardalotus punctatus*	SpP	W	0.5	0	0	0.5	2
Grey butcherbird	*Cracticus torquatus*	GB	W	0.08	0.19	0.27	0.46	37

Common Name	**Scientific name**	**Species code**	**Guild**	**Crop (27)**	**Exotic (30)**	**Native (15)**	**Linear remnant (32)**	**Total no. of plots**
Pacific black duck	*Anas superciliosa*	PBD	O	0.27	0.27	0	0.46	11
Pied butcherbird	*Cracticus nigrogularis*	PB	G	0.17	0.26	0.13	0.44	23
Striated pardalote	*Pardalotus striatus*	StP	W	0.25	0.19	0.132	0.43	53
Little corella	*Cacatua sanguinea*	LC	G	0	0.14	0.429	0.43	7
Black-faced cuckoo shrike	*Coracina novaehollandiae*	BFC	W	0.33	0.12	0.12	0.42	33
Sulphur-crested cockatoo	*Cacatua galerita*	SCC	G	0.17	0.33	0.08	0.42	12
Cockatiel	*Nymphicus hollandicus*	C	O	0.21	0.21	0.18	0.41	39
Red wattlebird	*Anthochaera carunculata*	RW	W	0	0.6	0	0.4	5
Noisy miner	*Manorina melanocephala*	NM	W	0.25	0.22	0.17	0.37	65
Crested pigeon	*Ocyphaps lophotes*	CP	O	0.24	0.25	0.18	0.34	68
Eastern rosella	*Platycercus eximius*	ER	W	0.26	0.288	0.11	0.34	80
Dusky woodswallow	*Artamus cyanopterus*	DW	W	0	0.333	0.33	0.33	3
Australian white ibis	*Threskiornis molucca*	AWI	O	0	0.667	0	0.33	3
Australian magpie	*Gymnorhina tibicen*	AM	G	0.21	0.326	0.15	0.31	80
Superb parrot	*Polytelis swainsonii*	SP	W	0.25	0.312	0.125	0.31	48
Galah	*Cacatua roseicapilla*	G	G	0.26	0.288	0.149	0.31	101
Australian wood duck	*Chenonetta jubata*	AWD	O	0.29	0	0.429	0.29	7
Yellow-rumped thornbill	*Acanthiza chrysorrhoa*	YRT	G	0.28	0.32	0.12	0.28	25
Australian raven	*Corvus coronoides*	AR	G	0.26	0.306	0.153	0.28	72
Red-rumped parrot	*Psephotus haematonotus*	RRP	W	0.37	0.22	0.146	0.27	41
Blue bonnet	*Northiella haematogaster*	BB	O	0.29	0.071	0.429	0.21	14
Brown songlark	*Cincloramphus cruralis*	BS	G	0.13	0.625	0.042	0.21	24
Noisy friarbird	*Philemon corniculatus*	NF	W	0.2	0.6	0	0.2	5
Rufous songlark	*Cincloramphus mathewsi*	RS	G	0.31	0.5	0	0.19	16
Common starling	*Sturnus vulgaris*	CS	G	0.39	0.27	0.18	0.16	44
Nankeen kestrel	*Falco cenchroides*	NK	O	0.11	0.67	0.11	0.11	9

Common Name	**Scientific name**	**Species code**	**Guild**	**Crop (27)**	**Exotic (30)**	**Native (15)**	**Linear remnant (32)**	**Total no. of plots**
Banded lapwing	*Vanellus tricolor*	BL	O	0	0	1	0	1
Striped honeyeater	*Plectorhyncha lanceolata*	StH	W	0	0	1	0	1
Mistletoebird	*Dicaeum hirundinaceum*	MI	W	0.33	0.33	0.33	0	3
Tree martin	*Hirundo nigricans*	TM	G	0.57	0.21	0.21	0	14
Brown falcon	*Falco berigora*	BF	G	1	0	0	0	2
Australian shelduck	*Tadorna tadornoides*	AS	O	0	1	0	0	1
Blue-faced honeyeater	*Entomyzon cyanotis*	BFH	W	0	1	0	0	1
Little pied cormorant	*Phalacrocorax melanoleucos*	LPC	O	0	1	0	0	1
Masked woodswallow	*Artamus personatus*	MW	G	0	1	0	0	1
Pallid cuckoo	*Cuculus pallidus*	PC	W	0	1	0	0	1
Richard's pipit	*Anthus novaeseelandiae*	RP	O	0	1	0	0	1
Stubble quail	*Coturnix pectoralis*	SQ	G	0	1	0	0	2
Swamp harrier	*Circus approximans*	SH	O	0	1	0	0	1
Black-shouldered kite	*Elanus axillaris*	BSK	O	0.5	0.5	0	0	6
Little eagle	*Hieraaetus morphnoides*	LE	G	0.5	0.5	0	0	2

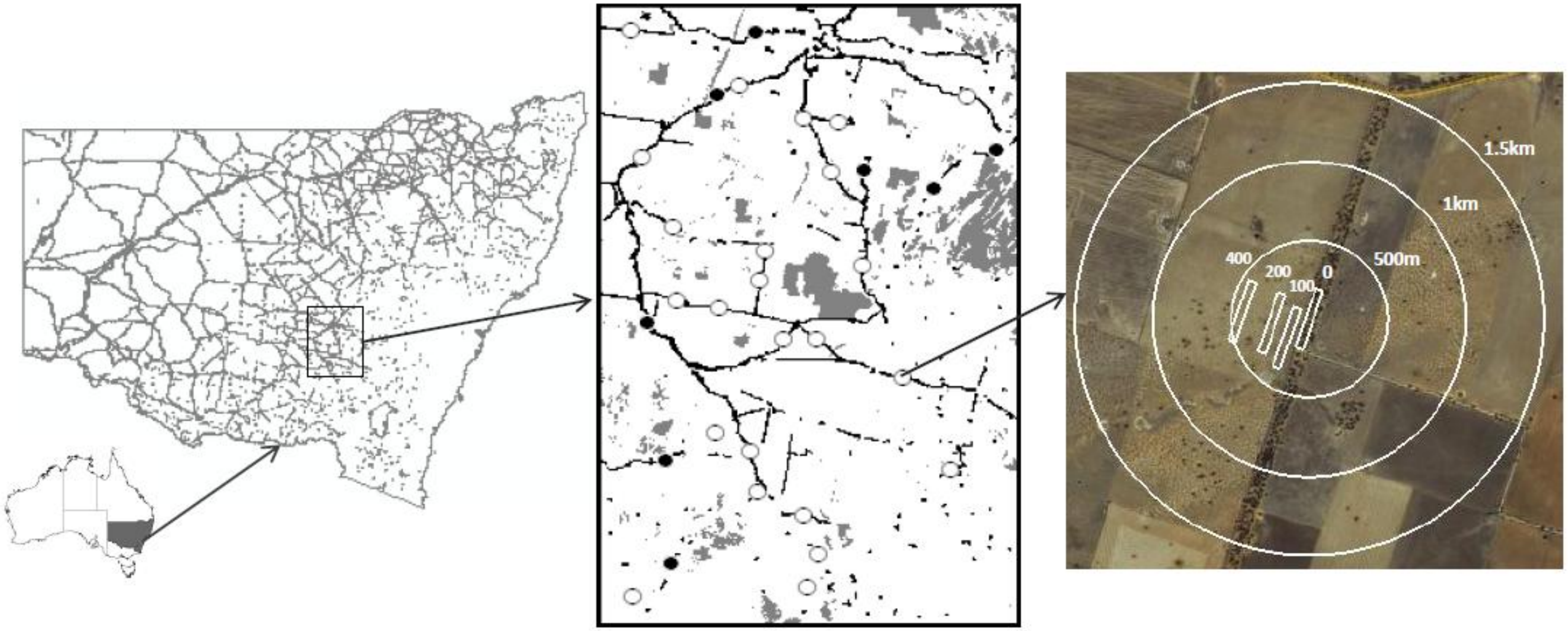


Figure S1. Schematic showing the full extent of the travelling stock route network across the state of New South Wales, the position of study sites within the landscape, and the layout of plots and buffers in and around each of the linear remnants. A site coloured in black indicates plots were only located within the remnant, whereas white colouration signifies plots were in both the remnant and the adjacent field.

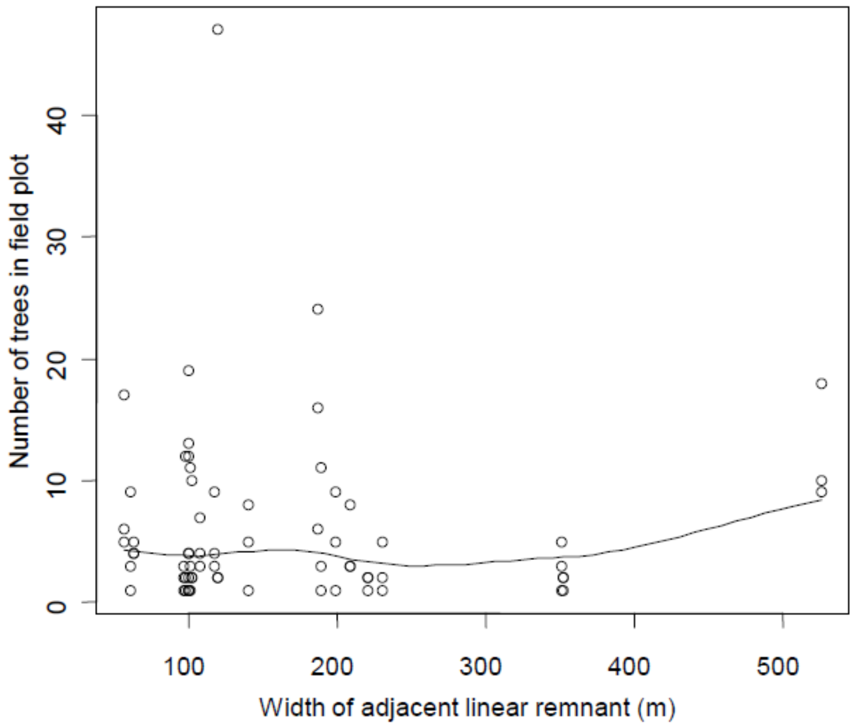


Figure S2.- Scatterplot showing the number of trees in each of the surveys plots in the fields, and the width of the adjacent remnant. After removing three outlier points, a weak and non-significant negative correlation can be seen.

Paper V. Corridors and unimproved pastures are wildlife-friendly farming measures that will benefit microbats

A little forest bat, *Vespadelus vulturnus*, waiting to be released at “Mortons Lane” TSR in Grogan, NSW, where it has been trapped. Its echolocation calls will be recorded as it flies away, and will form a reference for our automated species identification key. Image: P. Lentini.

This research paper is in review: Lentini, PE, Gibbons, P, Fischer, J, Law, BS, Hanspach, J & Martin, TG (in review) Corridors and unimproved pastures are wildlife-friendly farming measures that will benefit microbats. *PLoS ONE.*

Abstract

Schemes that aim to make agricultural landscapes less hostile to wildlife have been questioned because target taxa do not always benefit from prescribed measures. Although microbats are rarely targeted by such schemes, they often persist in agricultural landscapes and exert important top-down control of crop pests. Here, we investigated how bats respond to the presence and condition of linear remnants in a heavily modified agricultural landscape, as well as to the type of land use in adjacent fields, to derive management prescriptions for their ongoing conservation. We used acoustic detectors to quantify bat species richness, activity, and feeding in 32 linear remnants and adjacent fields across an agricultural region of New South Wales, Australia. Linear remnants varied in width and vegetation condition, and land use in fields consisted of native pastures, exotic pastures, and cereal or canola crop, each containing some scattered trees. Nocturnal arthropods were simultaneously trapped using black-light traps. We recorded 91,969 bat calls, 17,277 of which could be attributed to one of the 13 taxa recorded, and 491 calls contained feeding buzzes. The linear remnants supported higher bat activity than the fields, but species richness and feeding activity did not significantly differ. We trapped a mean 87.6 g (± 17.6 g SE) of arthropods per night, but found no differences in biomass between land use. Wider linear remnants with intact native vegetation supported more bat species, as did those adjacent to unsealed, as opposed to sealed roads. Fields of unimproved native pastures, with more retained scattered trees and associated hollows and logs, supported the greatest bat species richness and activity.

Synthesis and applications: The juxtaposition of linear remnants of intact vegetation and scattered trees in fields, coupled with less-intensive land uses such as unimproved pastures will benefit bat communities in agricultural landscapes, and should be

incorporated into agri-environment schemes. In contrast, sealed roads may act as a deterrent. The “wildlife friendly farming” vs “land sparing” debate has so far primarily focussed on birds, but here we have found evidence that the integration of both approaches could particularly benefit bats.

Introduction

Agricultural intensification and associated habitat fragmentation are key threatening processes for wildlife (Tilman *et al.* 2001). To mitigate negative effects, Agri-Environment Schemes (AES) have been established which offer farmers financial incentives to plant and protect vegetation, use fewer agrochemicals, or employ alternative grazing regimes (Kleijn & Sutherland 2003). Because many native species do not necessarily benefit from such 'wildlife-friendly farming' measures (Kleijn *et al.* 2001; Vickery *et al.* 2004), some propose that these investments would be better spent establishing separate conservation reserves; the so-called 'land-sparing' approach (Green *et al.* 2005; Phalan *et al.* 2011). However, most existing work has focused on birds, and some more cryptic groups may respond differently. For example, being highly mobile, many microbats are able to exploit patchily-distributed resources and retained features in the landscape, and therefore often constitute a large component of the mammalian fauna in agricultural environments (Lumsden, Bennett & Silins 2002a). In addition, bats exert top-down natural control of arthropod pests that have considerable impacts on crop yield (Lee & McCracken 2005; Kalka, Smith & Kalko 2008).

To date, there is a lack of consensus as to how to best manage for bats in agricultural environments. In Europe, AES which are primarily designed to support birds, invertebrates, and plants (Kleijn & Sutherland 2003) bring varied benefits for microbats. For example, Wickramasinghe *et al.* (2003) recorded higher levels of bat activity on organic compared to conventional farms, but in contrast, Fuentes-Montemayor, Goulson & Park (2011) concluded that AES-participating farms supported lower activity of two *Pipistrellus* species and their invertebrate prey. These conflicting results may be partly attributed to the fact that bats often use complementary habitats to fulfil life-history requirements. Whereas undisturbed remnants with many old, hollow-bearing trees might be favoured for roosting (Lumsden, Bennett & Silins 2002a), foraging activity is often higher in near trees in open areas and along edges (Downs & Racey 2006; Law & Chidel 2006) because vegetation clutter can inhibit flight for some species (Brigham *et al.* 1997). 'Roosting' and 'foraging' habitats thus can be quite different, and can be located several kilometres apart (Lumsden, Bennett & Silins 2002a) despite potential energetic costs of commuting (Tuttle 1976; Ransome 1990; Walsh & Harris 1996). Several AES target hedgerows, which are broadly analogous to other linear features that transect agricultural landscapes around the world, including living fences (Leon & Harvey 2006), treelines (Russ & Montgomery 2002), and road reserves (van der Ree 2002). Managed well, such linear features may reduce the energetic cost of commuting for bats by providing suitable roosts closer to open foraging sites, or by functioning as corridors for movement (Russo, Jones & Migliozzi 2002; Murray & Kurta 2004).

An Australian example of linear features is 'stock routes', which extend across the eastern portion of the continent, and form roadside corridors of remnant vegetation. They were originally established for the transport of livestock 'on the hoof', and were placed in low-lying, fertile portions of the landscape close to freshwater (Lentini *et al.* 2011b). The preference of bats for foraging and roosting in fertile geologies or in close proximity to water (Oakeley & Jones 1998; Rainho & Palmeirim 2011; Threlfall, Law & Banks 2012), coupled with many stock routes supporting old trees (Lentini *et al.* 2011a), suggests that stock routes should constitute valuable bat habitat. Because stock routes ("linear remnants" hereafter) vary greatly in width, vegetation condition, the roads they run adjacent to, and intensity of surrounding land use, they provide an excellent opportunity to explore what kinds of environmental measures can be implemented in agricultural landscapes for bat conservation. We aimed to establish (1) how linear remnants and surrounding fields differed in habitat value for bats; (2) what kinds of linear remnants were most important for bat conservation; and (3) what kinds of 'wildlife friendly' measures made fields better habitat for bats.

Methods

Study region and design

We studied a 15,000 km^2 area of the "wheat-sheep belt" of New South Wales, Australia (Fig. 1a). Land use is dominated by dry cereal cultivation, including wheat, oats and barley, as well as both native and improved pastures for livestock grazing. Prior to European settlement, the area was covered predominantly by *Eucalyptus* woodlands, but it is now 84% cleared, with formal conservation reserves covering only 1.3% of the area, and occurring mostly on ridgelines and unproductive areas (Pressey *et al.* 2000). Other remnant vegetation occurs as small patches or individual scattered trees in fields on private land, or in the public land system as linear remnants.

Our study design incorporated 32 sites (Fig. 1b); nested within each were two survey points in a linear remnant, and two in an adjacent field (totalling 128 surveys points). The two remnant survey points ('Remnant 1' and 'Remnant 2') were spaced at least 100 m apart, and the two field survey points were spaced approximately 100 m ('close') and 400 m ('far') from the remnant (Fig. 1c). Remnants ranged from narrow (38 m) to wide (570 m), and the condition of the vegetation within them from 'intact' (little evidence of anthropogenic disturbance) to 'degraded' (evidence of considerable grazing pressure or clearing). Four of the 32 remnants in this study could be classified as 'riparian', in they had a small stream or creek (~2m wide) running through them (see Appendix S1, Supporting information). Fields represented locally common land-uses, namely 12 cereal fields (wheat, barley or oats), 11 improved pastures (exotic annual grasses or lucerne/clover), five unimproved native pastures (largely perennial species), and four fields of canola (*Brassica* sp.). All remnants and fields contained at least two large trees (see below). Although the region had been in drought in previous years, rainfall was higher than average in 2010-2011, restricting access to some sites. Therefore, we collected data from 114 of the 128 points only (59 remnants and 54 fields).

Surveys

Bats

Microbat data were collected twice in summer 2010-2011: (1) the "maternal survey period" from 22 Nov to 22 Dec 2010 to represent the time when female bats usually have dependent young, and (2) the "juvenile survey period" from 21 Jan to 14 Feb 2011, when the young had become volant. We used Anabat ultrasonic detectors (models SD I, and SD II with ZCAIM storage units, Titley Electronics, Ballina) to conduct acoustic surveys. Detectors were calibrated following Larson & Hays (2000) and set in weatherproof boxes with a cut-out for a microphone funnel. With these funnels, the detectors were set at an angle of 45° from the horizontal. In both fields and remnants, we placed boxes on wooden platforms strapped to trees approximately 2 m above ground (Fig. 1d). We surveyed four sites at a time, and in each placed one detector at a remnant survey point and one at a field point for two consecutive nights (total eight detectors per night). In this way, at each site two of the points were surveyed for two nights in the maternal survey period, and the other two points were surveyed for two nights in the juvenile survey period. Detectors were set to turn on at least one hour before sunset (1800 hrs), and off again one hour after sunrise (0700 hrs).

Arthropods

We collected flying nocturnal arthropods at each survey point using 12 volt, 8 watt black-light (ultraviolet) traps (Australian Entomological Supplies Pty. Ltd., Coorabell, Fig. 1e). Because these traps may deter *Nyctophilus* species (Adams, Law & French 2005), they were set out for only one of the two consecutive detector nights. We did not sample fields when livestock were present to prevent damage to equipment. Traps were placed approximately 10 m from each survey point, on the opposite side of the tree to the detector. These were set out during the day, and collected the next morning, but were fitted with light/dark relay switches (Ozitronics, Melbourne) so they would switch on at dusk, and off again at dawn. Arthropod samples were stored in methylated spirits, and following the field season, samples were oven-dried at 60° C until desiccated. Dried samples were weighed using a laboratory balance to an accuracy of 0.001 g, and this figure was recorded as the “dry biomass”.

Habitat

Vegetation surveys were conducted within a 1-ha circle at each of the survey points during the two-day detector period. For trees, we determined the species, the diameter at breast height (DBH), the presence of hollows, bark type, and stage of senescence (the last two measures were recorded only for *Eucalyptus* species; based on Rayner (2008); Table S1 and Fig. S1 in Supporting Information). The number of *Eucalyptus* seedlings <130 cm tall was recorded to quantify tree regeneration, an important component of landscape function (Weinberg *et al.* 2011). Where tree cover was extremely dense, the area around the survey point was reduced to 0.283 ha (30m radius) or 0.126 ha (20 m radius), and these estimates were later scaled up to represent 1 ha. We visually estimated the percent cover of shrub species, and noted whether there was evidence of recent grazing by livestock (dung or foot prints). The length and diameter of every log (fallen timber >1m long and ≥10cm in diameter) was also measured. Two 50 m point-intercept transects were run from either side of the base of the survey tree, and at every metre we recorded the nature of the ground cover (native vegetation, non-native vegetation, rock, bare ground, leaf litter, water, cryptograms, cow dung). Finally, we recorded the type of road running adjacent to each of the linear remnants (multi-lane major highway, a single-lane sealed road, or unsealed but graded laneway).

Weather

At each site, rain gauges (Nylex Rain gauge 500, Pakenham) were inserted into the ground at the fenceline in between the remnant and field survey points, and were checked daily to estimate overnight rainfall. An anemometer (Vortex Hand-Held Anemometer Pro-1200, Inspeed, Sudbury) was taped to the top of the fence to record the maximum overnight wind speeds and we also checked these daily. Finally, we used iButton thermochron loggers (model no. DS1921G, Maxim, Sunnyvale) to record ambient temperature at each site. These were set to log readings every five minutes, were tied into the finger of a latex glove for weatherproofing, then also taped to the top of the fence in an area exposed to the sun. Although this will have caused the loggers to read artificially high temperatures, the bias was consistent. Data was downloaded off the logger once a week.

Bat call analysis

Call files recorded during the acoustic surveys were analysed using AnaScheme software, vers 1.0 (Gibson & Lumsden 2003; Adams, Law & Gibson 2010). Anascheme reads sound files recorded by Anabat detectors, and identifies bat pulses using a regional identification key; ours was built by BL, based on keys developed for Law & Chidel (2006) and Hanspach *et al.* (in press, further information in Appendix S2). It included 14 species (Table 1), and of these only *Rhinolophus megaphyllus* was not recorded during the surveys. Calls of *N. geoffroyi* and *N. gouldi* cannot be reliably distinguished, therefore the two species were pooled as "*Nyctophilus* sp.". This was also the case for *Vespadelus darlingtoni* (40-45 kHz) and *V. regulus* (40-45 kHz), which were pooled as "*Vespadelus darlingtoni*/*regulus*". *V. regulus* is known to also produce a higher-frequency (HF) call (54-55 kHz) around large water courses in the field area (Law, Reinhold & Pennay 2002) so Anascheme identified these separately as "*Vespadelus regulus* HF". We set Anascheme so that if >50 % of pulses could not be allocated to the same species, the file was identified as an "Unknown sp.". All files identified as containing bat calls also were separately filtered for feeding buzzes, using a filter developed by BL. Any files flagged as containing feeding buzzes were manually and audibly checked.

Based on the above call analysis, we considered three bat responses at each survey point for each night: (1) species richness, the number of species identified each night, not including the 'Unknown' calls; (2) total activity, the number of files containing bat calls, irrespective of identification; and (3) feeding buzzes, an index of the number of files containing feeding activity, irrespective of identification.

Data analysis

Do linear remnants and fields differ in habitat value for bats?

All analyses were conducted using 'R', vers 2.13.1 (http://www.r-project.org/). We first compared the three bat responses (species richness, activity, and feeding buzzes) between remnants and fields, by log-transforming the species richness and activity data, and running equal-variance t-tests. The feeding buzz data could not be transformed to fit a normal distribution, so we used a non-parametric Wilcox rank-sum test instead. To test for differences in bat species composition between land-use classes (canola crop, cereal crop, exotic pasture, native pasture and remnants), we used non-metric multidimensional scaling on the activity matrix of species (excluding *Chalinolobus picatus* and *Saccolaimus flaviventris*, rarely recorded) using the ‘metaMDS’ function in the ‘vegan’ package.

Fewer arthropod samples were collected than planned. High rainfall meant that some samples were washed away (14 of 98 samples), and trap number two may have been faulty, as it collected significantly smaller arthropod samples (18 samples, see Fig. S2). In addition, some traps appeared to not have switched on reliably, collecting few insects in some nights (<1.0g dry biomass). In total, 54 samples were considered reliable and could be analysed (Fig. 2). Dry biomass from these samples was log-transformed, and we ran a one-way ANOVA to test for differences in arthropod biomass between land-use classes. For each of our bat responses from both remnants and fields, we used Spearman rank correlation to test whether the relationships with arthropod biomass were significant.

What kinds of linear remnants are most important for bat conservation?

We used generalised linear mixed-effects models (GLMMs) to model the responses of bats (species richness, activity, feeding buzzes) in the linear remnants. Because they are highly mobile, we were interested in the scale at which variables would impact on our responses. Therefore, we grouped our predictor variables according to whether they occurred locally to the trapping point, in the area directly adjacent, or within the wider landscape context.

With regards to ‘local’ effects, we were limited in the number of explanatory variables we could include in our models. We chose five that we considered most relevant to bats (see Appendix S3), namely total basal area of trees, number of trees with hollows, volume of logs, percent ground cover that was native, and percent cover of shrubs, each within a given 1ha site. Using principal components analysis (PCA), we reduced these five variables to two components, which together explained 60% of variance in habitat data (Fig. 3; Table S2). Habitat component 1 ranged from ‘Intact’ at the negative end of the scale (more trees, hollows, logs, shrubs, and native ground cover) to ‘Degraded’ at the positive end of the scale (low values for these variables). For component 2, sites that structurally resembled ‘shrub/grassland’ (more native ground cover and shrubs) scored negatively, whereas those that resembled ‘grazed/cropped woodland’ (more trees, hollows and logs) scored positively. The width of the linear remnant, and the type or road running next to it, were also included in the local variable group (Table 2).

The ‘adjacent’ variable used was the land use in the adjoining field. ‘Landscape’ variables included the distances of each survey point to the nearest natural water body or farm dam with a surface area >1 ha (‘distance to water’, see Appendix S1) and nearest conservation area (‘distance to conservation area’; based on data supplied by NSW Office of Environment and Heritage “Land Use: New South Wales”). Finally, ‘conditions’ variables accounted for weather, presence or absence of a black-light trap, and the survey period. Skewed explanatory variables were log-transformed prior to the analyses, and continuous variables were standardised to have a mean of zero and a standard deviation of one.

Combinations of these explanatory variable groups (local, adjacent, landscape, and conditions) resulted in 15 alternative models, and we also tested a 16th ‘null model’ made up of random effects only, to determine if the explanatory variables predicted any more than our study design alone (Table S3). The random effect structure used in the models differed for each response, and this was based on visual inspection of the influence of each random effect (study site, survey point, and survey night) on responses, and also statistical methods outlined in Zuur *et al.* (2009). We used 'study site' as the random effect for the species richness data, because the survey point did not appear to influence the data. However, for feeding buzz data, the survey point did appear influential, and hence we used 'study site/survey point' in this case. There was evidence of overdispersion in the activity data, and to correct for this we added the random effect "night" ('study site/survey point/night'). In this way, every level in the random effect structure corresponded to a data point from an individual detector night. Each of the 16 alternative GLMMs were applied to each of the three bat response variables (species richness, activity, and feeding) assuming a Poisson distribution and using a log-link function in the 'glmer' function in the 'lme4' package for R.

For model selection we used an information-theoretic approach as implemented in the 'AICcmodavg' package. For each response we constructed 95% confidence tables, which list all models of the potential 16 tested with summed corrected Akaike weights ≥0.95 ('cw_i', which corrects for small sample sizes). To pick a 'final model' which best explained the patterns in our data, we compared corrected Akaike's Information Criteria ('AICc'), log-likelihood ('Log(L)') and cw_i for each of the models in the table. After choosing our final model, we judged which of the explanatory variables were having a strong influence by the magnitude of the coefficient estimate, and also whether the 95% confidence intervals included zero.

What kinds of 'wildlife friendly' measures make fields better habitat for bats?

Analysis for this question closely followed that described above – we again used GLMMs to predict the three bat responses, this time using the data collected in fields. Our 16 candidate models were the same as for the remnants, however some of the variables switched between groups to reflect the change of survey location. The 'local' variable group consisted of land use, the distance of the survey point from the remnant, and habitat components 1 and 2, and the 'adjacent' group contained the width of the adjacent remnant, and road type. The 'landscape' and 'conditions' groups remained the same (Table 2). The random effect structure for a given response and model selection was the same as described for the remnant analysis.

Results

Across 228 detector nights and 2,475 survey hours, we recorded 1,193,152 sound files. Of these, 91,969 (7.7%) were confirmed as bat calls (403 passes/night), and 17,277 (19% of bat calls) could be identified. Although the filter matched 3,031 files as containing feeding buzzes, only 491 files were confirmed as buzzes when manually checked (2.8% of bat calls). A total of 13 taxa were recorded (Table 1), and of these, *V. vulturnus* and *C. gouldii* were the most common, present at 98% and 91% of the survey points respectively. The two species listed as threatened were the least common, namely *C. picatus* (n = 6), and *S. flaviventris* (n = 1, Table S4).

Do linear remnants and fields differ in habitat value for bats?

There were no significant differences between remnants and fields with regards to bat species richness ($p = 0.434$, $t = 0.7842$, $df = 223$, remnant mean = 5.02, field mean = 5.38) or the number of feeding buzzes recorded ($p = 0.178$, $W = 6893$, remnant mean =

1.1, field mean = 3.4).). However, total bat activity in the remnants was double that of the fields ($p = 0.044$, $t = -2.0283$, $df = 223$, remnant mean = 609.84, field mean = 257.74). No clear differences in community composition were apparent between land use classes (Fig. S3). Arthropod biomass also did not significantly differ between land use classes ($p = 0.603$, $df = 4$, $F = 0.688$, Fig. 2). There were significant correlations between bat species richness, and also bat activity and arthropod biomass in both fields and remnants, but no such relationship was evident for feeding buzzes (Fig. 4).

What kinds of linear remnants are most important for bat conservation?

The 'adjacent' variable group was ranked as the best predictor of bat species richness in the linear remnants (relative importance 0.95, Table 3a), but the 95% confidence intervals for land use categories included zero (Table 4a), indicating they were not having a very strong effect, and estimates were of a low magnitude.

Because the 'local' variable group (relative importance 0.39) also appeared to have a strong effect on species richness data, we selected the second-highest ranked model ('local' + 'adjacent'; Table 3a) for plotting and interpretation. Wider linear remnants were the most species rich, as were those that ran next to unsealed laneways and had a more intact vegetation structure (Table 4a, Fig. 5a). For both bat activity and feeding in the remnants, the 'adjacent' and 'conditions' variable groups constituted the highest-ranked models (Table 3b and 3c), however, once again land use did not appear to have a very strong effect (Table 4a). Both activity and feeding levels were higher in the juvenile survey period (Fig. 5b and 5c), though temperature was a better predictor of activity data, and wind speed of feeding data. The large number of models included in the 95% confidence table for feeding in remnants (Table 3c), which includes Model 16 (the 'null model'), indicated that there was a very high degree of uncertainty in predicting bat feeding behaviour.

What kinds of 'wildlife friendly' measures make fields better habitat for bats?

The highest-ranked model for bat species richness in fields included the 'local' variable group only, with a relative importance of 0.95 (Table 3d). Fields containing native, unimproved pastures supported the most species-rich communities of bats compared with other land use categories (exotic pasture, canola, or cereal crop, Fig. 5d), and a positive effect of habitat component 2 indicated that bat species richness increased with a greater number of trees, number of hollows and log volume (Fig. 3, 5d). A large number of variables strongly predicted for bat activity in the fields – the 'local', 'adjacent', and 'conditions' groups were all included in the highest-ranked model (Table 3e). Again, there was a positive effect of native pastures, as well as those with higher values of habitat component 2 ('grazed/cropped woodlands', a. 5e). In concordance with our findings from the remnants, we recorded marginally higher bat activity in fields next to unsealed laneways (Fig. 5f). Finally, only "conditions" affected the number of feeding buzzes recorded in fields, as the model of this variable group alone was very highly weighted (cw_i = 0.83, Table 3f). The presence of a light trap in particular led to higher levels of feeding activity (Table 4b).

Discussion

Do linear remnants and fields differ in habitat value for bats?

Surprisingly, the only detectable difference between linear remnants and fields was higher bat activity in remnants. Because arthropod samples from the different land uses were similar in mass (Fig. 2), this is not due a greater availability of prey. An alternative explanation may be that there were more active roosts in the remnants, and our vegetation data support this. If we assume that bats prefer to roost in trees >70cm in diameter with hollows (based on Lunney *et al.* 1988; Lumsden, Bennett & Silins 2002b), typical remnants supported on average 6.53 (±0.67 SE) potential roost trees per hectare, compared with 2.22 (±0.35 SE) in fields. This figure for fields within the study region is likely to be an over-estimation, because we only chose sites with trees to conduct our surveys. Also, while bat activity is high around scattered trees, they are uncommonly used as roosts (Law, Chidel & Turner 2000; Lumsden, Bennett & Silins 2002b). The provision of suitable roosting trees is likely to become more important in the future, especially as scattered trees in fields are declining through senescence and a lack of natural regeneration (Gibbons et al. 2008).

Perhaps even more surprisingly, we did not detect differences in community composition between land uses (Fig. S3). This may be because the woodland communities that naturally occur in this region, and which are retained in the linear remnants, are quite open and thus do not preclude foraging by open-area species. Similarly, clutter-tolerant species are not necessarily limited to foraging in cluttered areas (Webala *et al.* 2011), and we also sampled fields with scattered trees, providing edge-space bats with suitable habitat. Analyses relating to the requirements of individual species would be necessary to determine more subtle effects of land use on the occupancy of remnants and fields by different bat fauna.

What kinds of linear remnants are most important for bat conservation?

Our surveys indicated that wider remnants, composed of intact vegetation that includes a variety of structures, form the best habitat for diverse bat communities (Fig. 5a, Table 4a), which is very much in line with conventional conservation wisdom (Mortelliti, Amori & Boitani 2010). Remnants next to sealed roads supported lower bat species richness than those next to unsealed roads, although there was not a great difference between single-lane roads and multi-lane highways (Fig. 5a). This might suggest that perhaps it is not the level of traffic itself that is deterring bats, but rather the nature of the bitumen surface. Potential causes of lower richness with the sealed road surface need to be explored further, particularly in relation to their effects on individual species. In their study of the BAB3 motorway in Germany, Kerth & Melber (2009) found that a 'clutter-tolerant' bat species was more vulnerable to the effects of the road than an open-area adapted bat species, so it would be valuable to determine the role that ecomorphology plays in these circumstances.

What kinds of 'wildlife friendly' measures make fields better habitat for bats?

Scattered trees in fields, and other structures associated with them such as logs and hollows were found to be important in maintaining high bat activity, most likely because they provide a source of forage and shelter from predators (Lumsden & Bennett 2005; Fischer, Stott & Law 2010). The importance of these findings needs to be reinforced to private land managers, because scattered trees are being lost from fields globally (Gibbons *et al.* 2008). This is especially the case in cropping environments, because trees compete for water and nutrients and are an obstacle for large equipment (Ozolins, Brack & Freudenberger 2001), but these fields are also the most likely to both benefit from bat predation services on pests. In spite of uncertainty regarding the value of more 'wildlife friendly' land uses to bats (Fuentes-Montemayor, Goulson & Park 2011), we found a clear positive effect of unimproved native pastures, on both bat species richness and activity in fields. This cannot be explained by prey availability, however, because we did not detect any differences in arthropod biomass between land use classes (Fig. 2). The lack of difference might be because although lower pesticide inputs in native pastures should benefit arthropod diversity, higher nutrient loads in more intensively managed areas can also lead to outbreaks in a small number of herbivorous insect species (Landsberg, Morse & Khanna 1990), resulting in a similar arthropod biomass. If prey availability alone cannot explain patterns of bat use in the fields, then the habitat native pastures provide in forming a ‘softer’ matrix is likely to also be playing a role.

How can agricultural landscapes best be managed for bat conservation?

Plausible management goals in our study area would be to maximise bat richness in remnants (for conservation purposes) and maximise bat activity in fields (for pest control purposes). For these goals, across all land use categories, the retention of more natural structures such as trees will be important, as is an understorey not strongly modified by grazing impacts or cropping. Our results imply that conservation actions are likely to be more successful if conducted in areas close to unsealed, rather than sealed roads. Unfortunately, it is less clear how to manage for bat feeding specifically, because responses were not as strong and mostly related to conditions during surveys. However, other studies in agricultural areas have found higher rates of feeding over more fertile geologies (Law, Chidel & Penman 2011), suggesting that conserving remnant vegetation in productive parts of the landscape is important for bats. The juvenile survey period also saw a considerable increase in activity, highlighting the importance of maintaining structures that allow for successful breeding close to fields, such as linear remnants and large retained trees.

It was somewhat surprising that the bats in our study did not respond to the proximity of water or protected areas. Given the mobility of bats, all of our survey points may have been within easy commuting distance from important resources such as a bodies of water (maximum distance 6.9km) or conservation areas (maximum distance 11.5km). Alternatively, in many cases bats may have been accessing roosts and water in the linear remnants themselves, and therefore the distances we measured were irrelevant. We may also not have found an effect of distance to water because of the high rainfall during the survey period, which resulted in free-standing water being present across much of the landscape (Appendix S1, Fig. S4). It is probable that in years such as this, water resources are of lesser importance to bat communities than other factors, such as roost or forage availability. However, in drier years streams in remnants and dams in fields are likely to form important resources for bats (Wickramasinghe *et al.* 2003; Lundy & Montgomery 2010).

It should be noted that a considerable proportion of the landholders we spoke with indicated they were not aware that bats used their fields for either roosting or foraging. This further strengthens the case for better communicating the persistence of these cryptic taxa in agricultural landscapes, especially given the positive indirect impact bats are likely to have on crop yield through pest predation (Lee & McCracken 2005; Lundy & Montgomery 2010). Furthermore, the two types of areas surveyed in this study (linear remnants and fields) have historically been managed as separate entities by separate actors, yet variables relating to areas adjacent to the survey points were strong predictors of bat responses in many of our models (Table 3). A more integrated approach to landscape planning and management, which takes into account not only the individual features or fields but also the surrounding landscape (Martin *et al.* 2006), is therefore required.

This study has revealed some key features that can be manipulated to conserve bats in agricultural landscapes, which should be further incorporated into AES. In particular, linear elements can support high bat activity if managed appropriately. Moreover, by harbouring bat communities and having high edge to area ratios, linear elements have the potential to provide pest predation ecosystem services to a greater number of fields than more remote or isolated reserves alone. Finally, there appears to be no clear-cut answer as to whether “wildlife friendly farming” (the integration of wildlife-friendly features into fields) or “land sparing” (the protection of designated areas) is preferable for bats. So far most arguments for “land sparing” have focused on birds (Green *et al.* 2005; Phalan *et al.* 2011). However in our study system, we found evidence that the integration of both approaches could be useful because both conditions in fields and linear remnants influenced bat communities.

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Tables and Figures

Table 1. Bat species recorded in the surveys, listed in descending order according to total activity across all sites. “C” is “Count”, the number of survey points that each species was recorded at, with the total listed in parenthesis in the header row. “A” represents “Activity”, the number of calls recorded, and “F” represents “Feeding buzzes”. An expanded version of this list is supplied in Table S4.

		Remnants (118)			Fields (107)			Total (225)	
Species	**C**	**A**	**F**	**C**	**A**	**F**	**C**	**A**	**F**
Unknown sp.	116	57,490	6	105	17,202	30	221	74,692	36
Vespadelus vulturnus	108	2,881	33	96	3267	72	204	6,148	105
Chalinolobus gouldii	86	1,507	31	73	2344	164	159	3,851	195
Mormopterus sp. 4	72	730	11	71	1477	42	143	2,207	53
Scotorepens greyii	51	1,879	7	42	320	7	93	2,199	14
Scotorepens balstoni	56	511	13	49	318	28	105	829	41
Mormopterus sp. 2	33	279	20	41	354	5	74	633	25
Tadarida australis	40	246	0	50	233	1	90	479	1
Vespadelus darlingtoni/regulus	43	144	2	41	235	9	84	379	11
Chalinolobus morio	44	109	1	31	132	0	75	241	1
Nyctophilus sp.	28	52	2	43	138	3	71	190	5
Vespadelus regulus (HF)	22	59	3	26	55	1	48	114	4
Chalinolobus picatus	3	3	0	3	3	0	6	6	0
Saccolaimus flaviventris	1	1	0	0	0	0	1	1	0
Rhinolophus megaphyllus	0	0	0	0	0	0	0	0	0
Total		65,891	129		26,078	362		91,969	491

Table 2. Groups of explanatory variables used to construct the alternative generalised linear mixed models predicting bat species richness, activity and feeding (see Table 3, Table S3).

Variable group name	Variables in remnant models	Variables in field models
1. Local habitat "LOC"	• Remnant width • Road type • Habitat component 1 • Habitat component 2	• Distance from the remnant • Land use in the field • Habitat component 1 • Habitat component 2
2. Adjacent habitat "ADJ"	• Land use in adjacent field	• Width of the adjacent remnant • Adjacent road type
3. Landscape context "LCSP"	• Distance to conservation area • Distance to water body	• Distance to conservation area • Distance to water body
4. Survey conditions "COND"	• Presence/absence of rain • Maximum wind speed • Maximum temperature • Presence/absence of light trap • Survey period	• Presence/absence of rain • Maximum wind speed • Maximum temperature • Presence/absence of light trap • Survey period

Table 3. 95% confidence tables for bat response analyses, listing the variable groups included in the models, corrected Akaike's Information Criteria (AICc), corrected Akaike Weights (cw_i), and log-likelihood (Log(L)). Variable groups are described in Table 2, and model numbers are as defined in Table S3. Underlined models were used for plotting, and are further described in Table 4.

Model no.	**LOC**	**ADJ**	**LSCP**	**COND**	**AICc**	**cw_i**	**Log(L)**
a) Remnants - species richness							
2		X			176.94	0.39	-82.06
<u>4</u>	X	X			177.13	0.35	-76.19
9		X		X	179.46	0.11	-77.36
5		X	X		180.89	0.05	-81.72
11	X	X		X	181.48	0.04	-71.75
Relative importance	0.39	0.95	0.05	0.15			
b) Remnants - activity							
<u>9</u>		X		X	747.01	0.84	-359.86
12		X	X	X	750.60	0.14	-359.04
Relative importance	0	0.95	0.14	0.95			
c) Remnants - feeding							
<u>9</u>		X		X	220.34	0.39	-97.79
15				X	221.79	0.19	-102.22
11	X	X		X	223.31	0.09	-92.67
8	X			X	224.45	0.05	-97.42
4	X	X			224.55	0.05	-99.90
12		X	X	X	224.61	0.05	-97.37
1	X				224.81	0.04	-103.73
2		X			225.12	0.04	-106.15
10			X	X	225.55	0.03	-101.72
16 (NULL)					225.65	0.03	-109.72
14	X	X	X	X	227.19	0.01	-91.75
Relative importance	0.24	0.63	0.09	0.81			
d) Fields - species richness							
<u>1</u>	X				143.42	0.50	-62.93
8	X			X	144.05	0.36	-56.94
6	X		X		147.55	0.06	-62.55
13	X		X	X	149.26	0.03	-56.80
Relative importance	0.95	0	0.09	0.39			
e) Fields - activity							
<u>11</u>	X	X		X	599.60	0.72	-279.11
8	X			X	602.06	0.21	-284.59
14	X	X	X	X	604.82	0.05	-278.72
Relative importance	0.99	0.85	0.05	0.99			
f) Fields - feeding							
<u>15</u>				X	276.05	0.83	-129.24
10			X	X	279.95	0.12	-128.75
Relative importance	0	0	0.12	0.95			

Table 4. Model parameters predicting bat species richness, activity and feeding in a) remnants and b) fields, showing the coefficient, standard error (SE), and lower and upper 95% confidence intervals (CI upp and CI low respectively) for each variable in the final model. The ‘landscape’ variable group was not included in any of the final models, so is not listed here.

a) REMNANTS		**RICHNESS**				**ACTIVITY**				**FEEDING**			
Var. group	**Term**	**Est.**	**SE**	**CI low**	**CI upp**	**Est.**	**SE**	**CI low**	**CI upp**	**Est.**	**SE**	**CI low**	**CI upp**
	Intercept	1.732	0.238	1.256	2.208	4.374	0.600	3.174	5.574	-1.670	0.796	-3.262	-0.078
Local habitat “LOC”	Remnant width	0.169	0.075	0.019	0.319								
	Road - Major	-0.263	0.214	-0.691	0.165								
	Road - Sealed	-0.338	0.156	-0.65	-0.026								
	Hab. comp. 1	-0.090	0.063	-0.216	0.036								
	Hab. comp. 2	-0.008	0.068	-0.144	0.128								
Adjacent habitat “ADJ”	L.use - Cereal crop	-0.058	0.203	-0.464	0.348	0.305	0.644	-0.983	1.593	-0.045	0.844	-1.733	1.643
	L.use - Exotic past	-0.088	0.197	-0.482	0.306	0.490	0.645	-0.8	1.78	-0.059	0.845	-1.749	1.631
	L.use - Native past	-0.061	0.247	-0.555	0.433	0.309	0.757	-1.205	1.823	0.583	0.974	-1.365	2.531
Survey conditions “COND”	Wind speed					-0.216	0.143	-0.502	0.07	-0.381	0.139	-0.659	-0.103
	Temperature					0.320	0.156	0.008	0.632	-0.223	0.184	-0.591	0.145
	Rain - Present					-0.151	0.291	-0.733	0.431	0.412	0.312	-0.212	1.036
	Light trap - Present					-0.292	0.207	-0.706	0.122	-0.228	0.208	-0.644	0.188
	Survey period - juvenile					0.859	0.311	0.237	1.481	1.370	0.451	0.468	2.272

b) FIELDS		**RICHNESS**				**ACTIVITY**				**FEEDING**			
Var. group	**Term**	**Est.**	**SE**	**CI low**	**CI upp**	**Est.**	**SE**	**CI low**	**CI upp**	**Est.**	**SE**	**CI low**	**CI upp**
	Intercept	1.546	0.146	1.254	1.838	3.693	0.470	2.753	4.633	-0.821	0.432	-1.685	0.043
Local habitat “LOC”	Land use - Cereal crop	0.075	0.160	-0.245	0.395	1.002	0.401	0.2	1.804				
	L.use - Exotic past	0.026	0.170	-0.314	0.366	0.694	0.422	-0.15	1.538				
	L.use - Native past	0.518	0.260	-0.002	1.038	1.712	0.687	0.338	3.086				
	Hab. comp. 1	-0.105	0.064	-0.233	0.023	-0.214	0.156	-0.526	0.098				
	Hab. comp. 2	0.256	0.064	0.128	0.384	0.706	0.149	0.408	1.004				
	Distance into field	0.053	0.047	-0.041	0.147	-0.010	0.108	-0.226	0.206				
Adjacent habitat “ADJ”	Road - Major					-0.932	0.450	-1.832	-0.032				
	Road - Sealed					-1.198	0.347	-1.892	-0.504				
	Remnant width					0.124	0.142	-0.16	0.408				
Survey conditions “COND”	Wind speed					-0.063	0.107	-0.277	0.151	-0.230	0.122	-0.474	0.014
	Temperature					-0.085	0.127	-0.339	0.169	0.205	0.179	-0.153	0.563
	Rain - Present					-0.287	0.236	-0.759	0.185	0.089	0.158	-0.227	0.405
	Light trap - Present					0.180	0.196	-0.212	0.572	0.611	0.148	0.315	0.907
	Survey period - juvenile					1.575	0.230	1.115	2.035	-0.476	0.377	-1.23	0.278

Fig. 1. Study design diagram, showing a) the study area within the state of New South Wales, Australia b) the position of the 32 study sites within the study area, with the linear remnant network shown in white c) an example of the layout of four survey points nested within a study site d) a bat detector, in a weatherproof box and with microphone funnel attached, on a platform strapped to a survey tree, and e) a black-light trap on the ground.

Fig. 2. Mean dry biomass of arthropod samples collected per night in each of the land-use classes, with error bars representing 95% confidence intervals, and sample sizes listed below each of the plotting points.

Fig. 3. PCA biplot showing the loading of the five vegetation measures on habitat components 1 and 2. Habitat component 1 separates sites according to condition, ranging from 'intact' (more negative scores) to 'degraded' (more positive scores) Habitat component 2 related to site structure: 'shrub/grassland' in the negative values to 'grazed/cropped woodland' in the positive values. Survey points are plotted according to the land use class that they occur within.

Fig. 4. Plots showing the relationship of bat species richness, activity and feeding with arthropod biomass, in both remnants and fields. Parameters from the Spearman-rank correlation analysis of each relationship are listed on the plots, and a smooth curve has been fitted for visualisation.

Fig. 5. Plots showing the effect of the most influential predictor variables on a) bat species richness in remnants, b) bat activity in remnants, c) bat feeding in remnants, d) bat species richness in fields, e) and f) bat activity in fields. More positive values of

habitat component 2 indicate that a site has a structure which closer resembles a grazed or cropped woodland (mote trees, hollows and logs), as opposed to a shrub/ grassland. Model parameters are listed in Table 4a) and 4b). Semi-transparent polygons depict 95% confidence intervals.

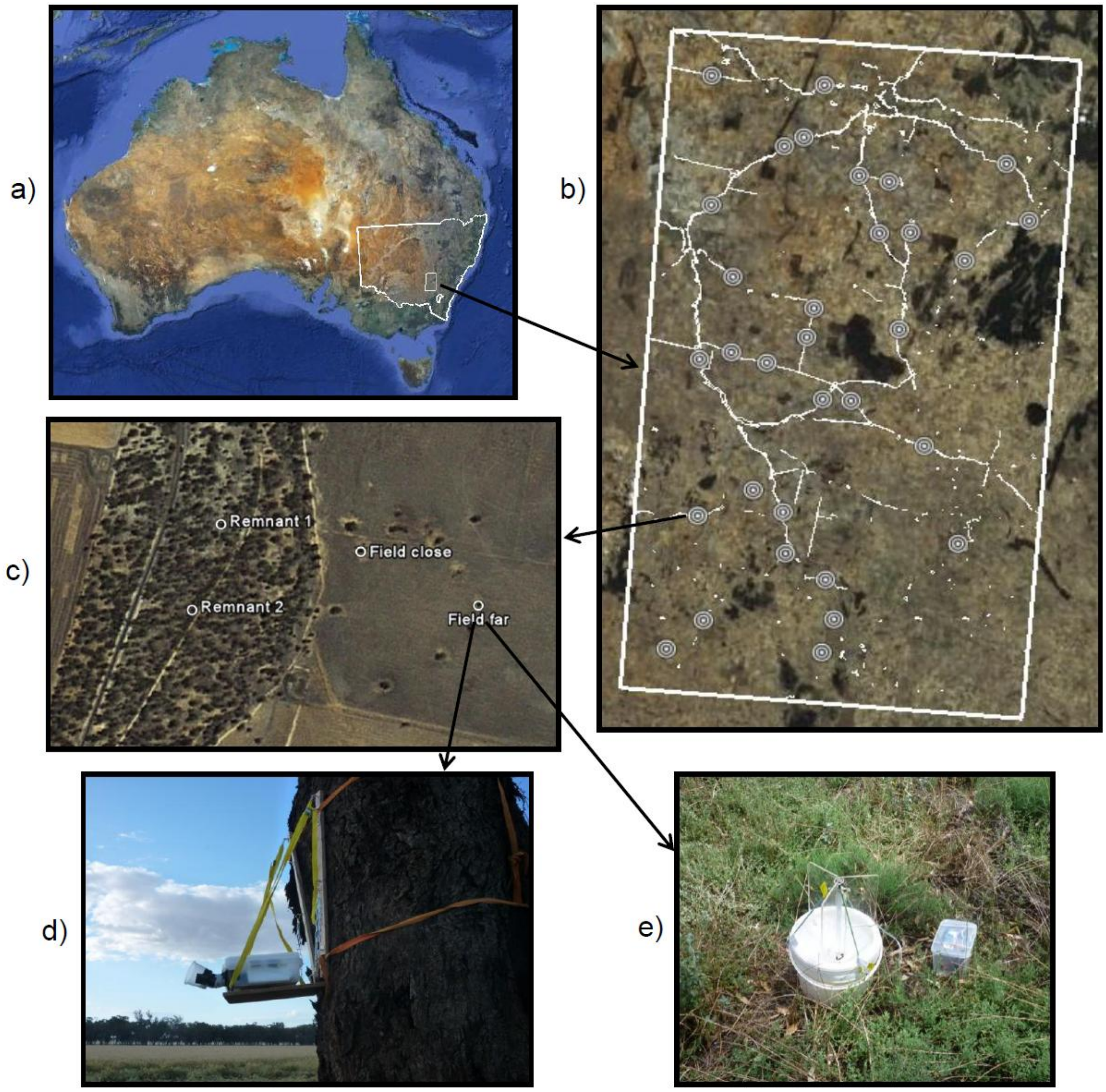

Fig. 1.

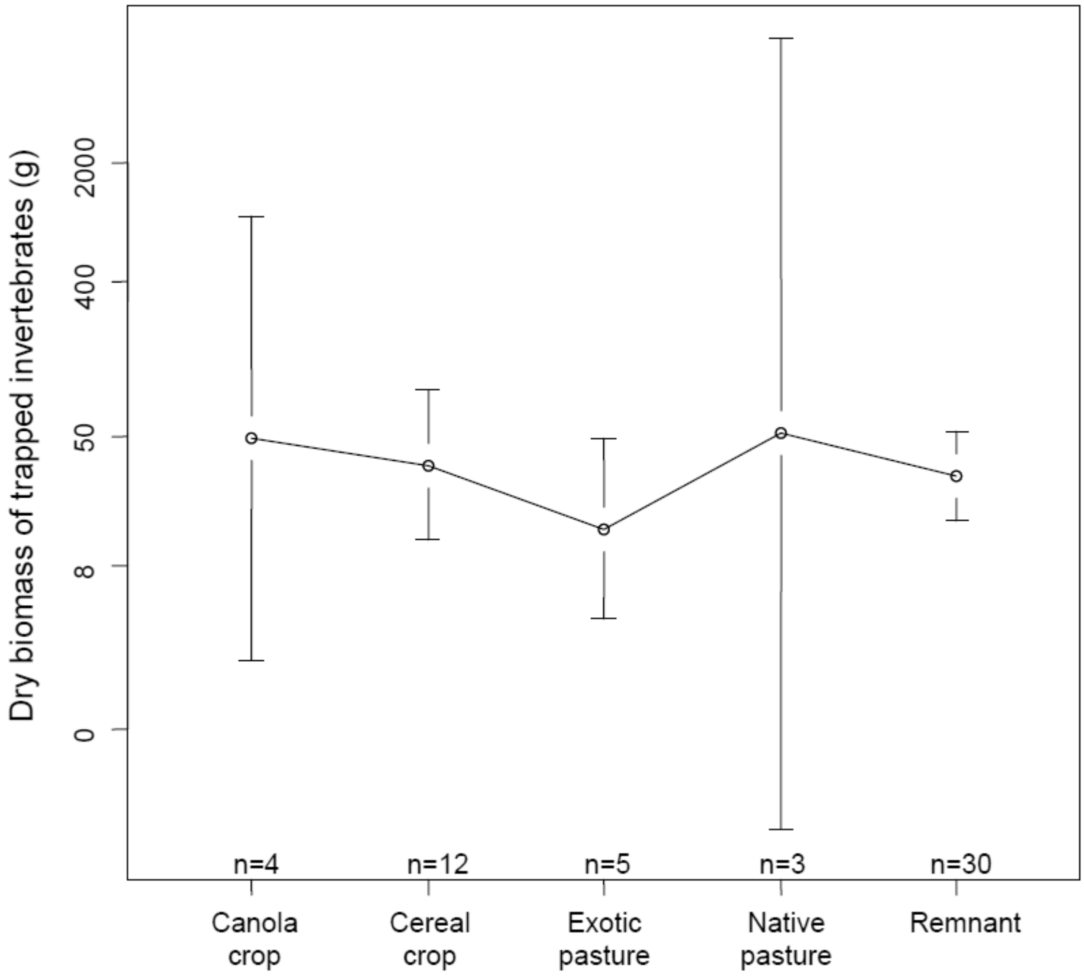
Dry biomass of trapped invertebrates (g)
2000
400
50
8
0
n=4
n=12
n=5
n=3
n=30
Canola crop
Cereal crop
Exotic pasture
Native pasture
Remnant

Fig. 2.

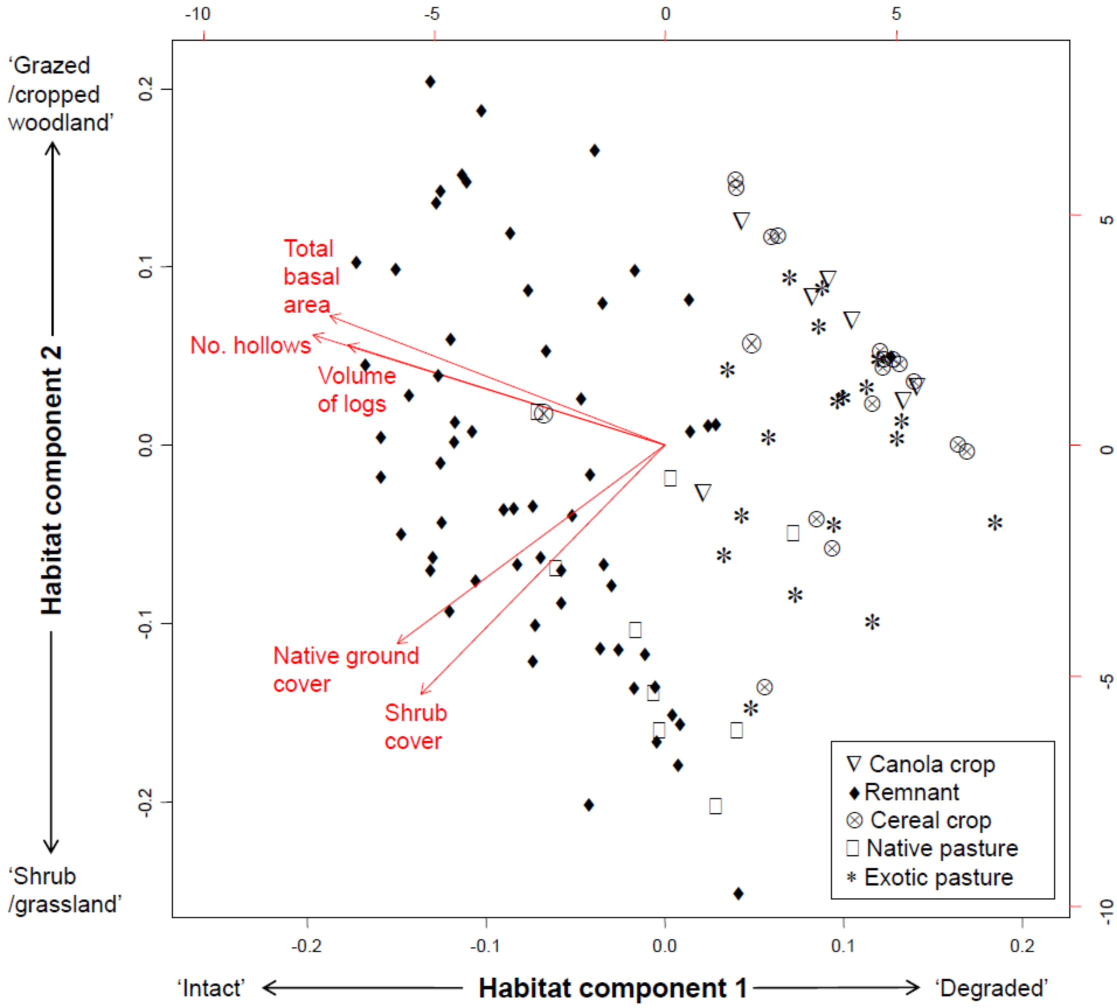
'Grazed /cropped woodland'
'Shrub /grassland'
Habitat component 2
'Intact'
Habitat component 1
'Degraded'
Total basal area
No. hollows
Volume of logs
Native ground cover
Shrub cover
Canola crop
Remnant
Cereal crop
Native pasture
Exotic pasture

Fig. 3.

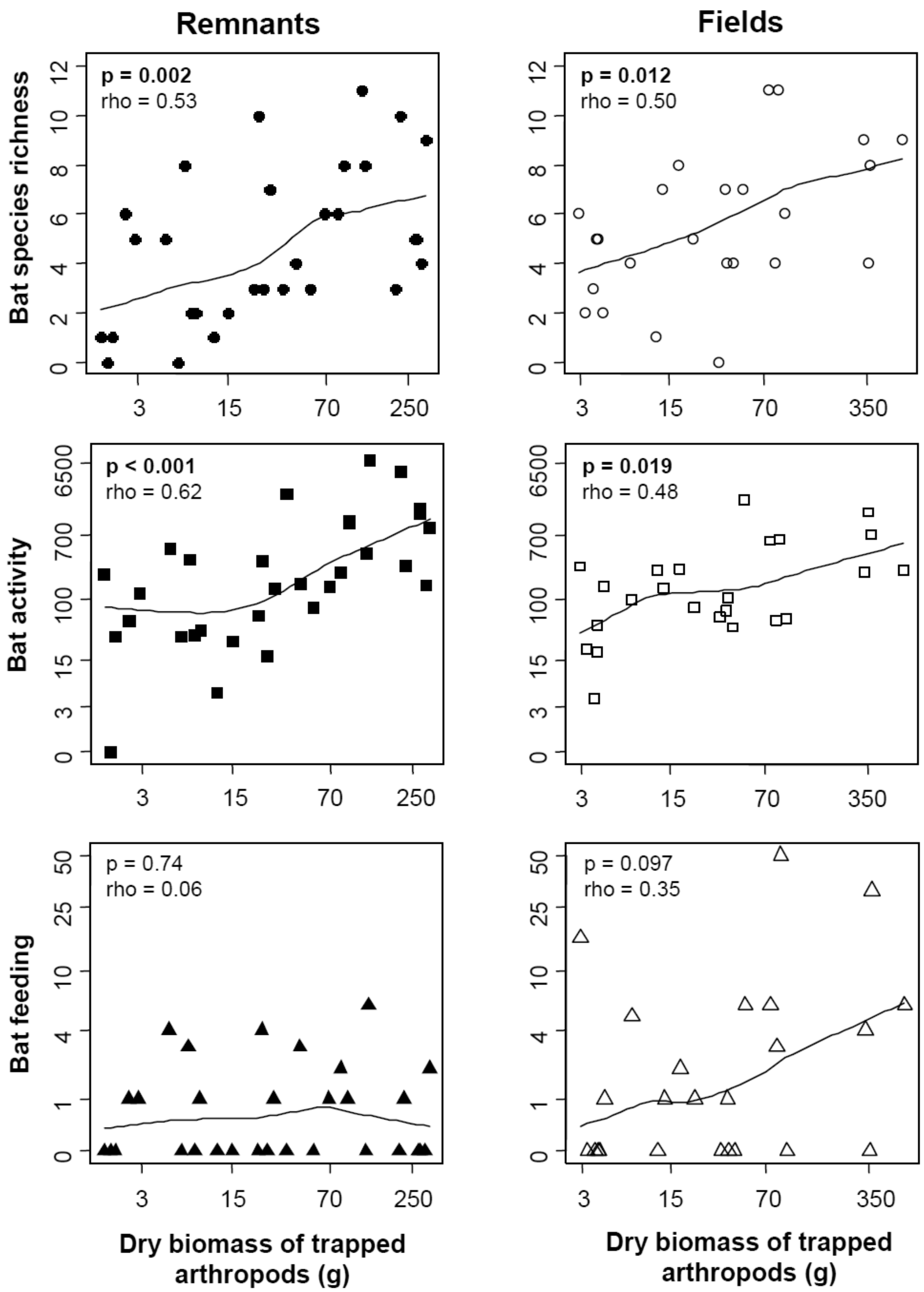
Remnants
Fields
Bat species richness
Bat activity
Bat feeding
p = 0.002
rho = 0.53
p = 0.012
rho = 0.50
p < 0.001
rho = 0.62
p = 0.019
rho = 0.48
p = 0.74
rho = 0.06
p = 0.097
rho = 0.35
Dry biomass of trapped arthropods (g)
Dry biomass of trapped arthropods (g)

Fig. 4.

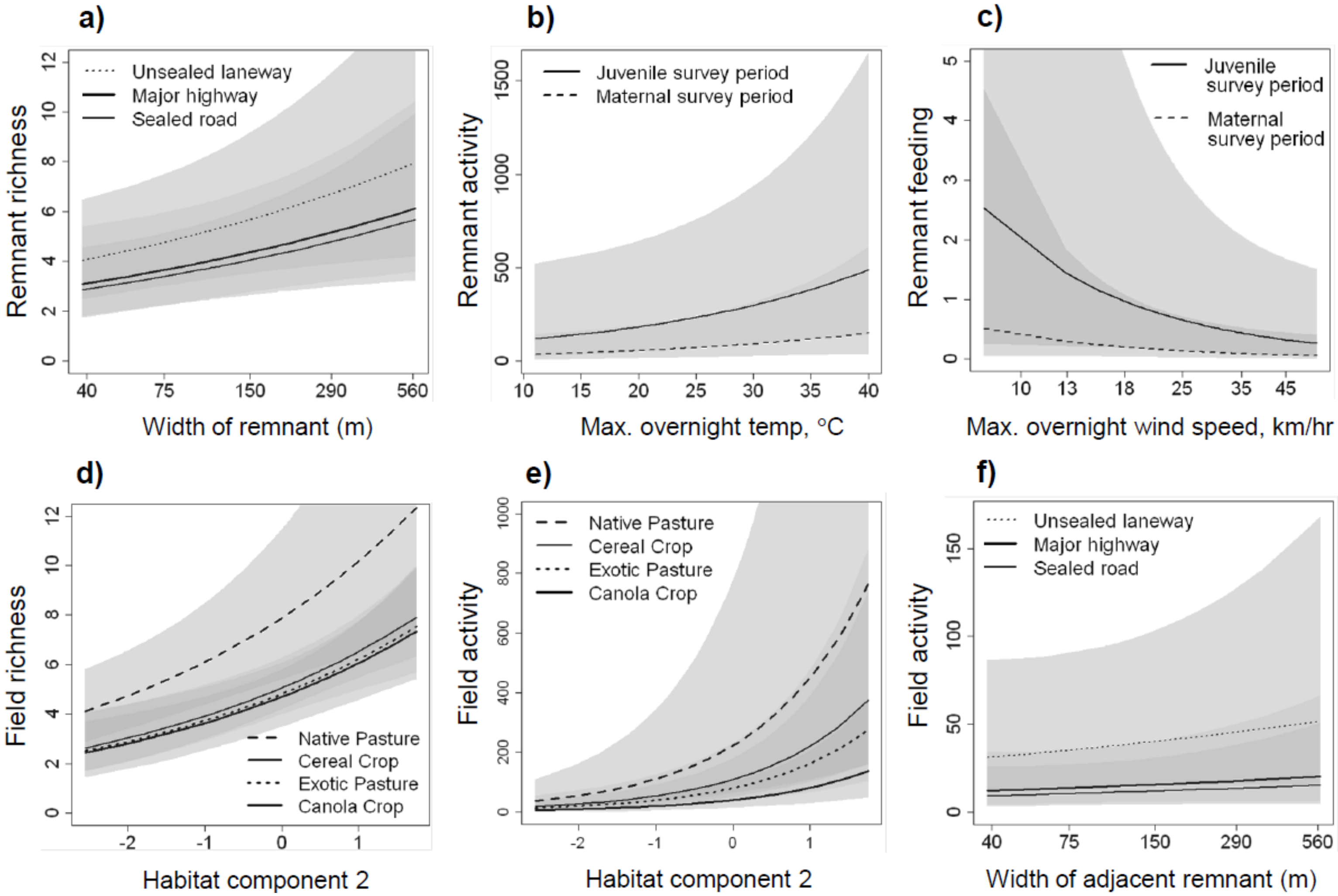
a)
Unsealed laneway
Major highway
Sealed road
Remnant richness
0
2
4
6
8
10
12
40
75
150
290
560
Width of remnant (m)
b)
Juvenile survey period
Maternal survey period
Remnant activity
0
500
1000
1500
10
15
20
25
30
35
40
Max. overnight temp, °C
c)
Juvenile survey period
Maternal survey period
Remnant feeding
0
1
2
3
4
5
10
13
18
25
35
45
Max. overnight wind speed, km/hr
d)
Native Pasture
Cereal Crop
Exotic Pasture
Canola Crop
Field richness
0
2
4
6
8
10
12
-2
-1
0
1
Habitat component 2
e)
Native Pasture
Cereal Crop
Exotic Pasture
Canola Crop
Field activity
0
200
400
600
800
1000
-2
-1
0
1
Habitat component 2
f)
Unsealed laneway
Major highway
Sealed road
Field activity
0
50
100
150
40
75
150
290
560
Width of adjacent remnant (m)

Fig. 4.

Supplementary material

Appendices

Appendix S1 – The importance of water and riparian areas

Many previous studies have found water resources, both natural and anthropogenic, to be important for bats (Law, Anderson & Chidel 1998; Wickramasinghe *et al.* 2003; Lundy & Montgomery 2010). Four of the 32 remnants in this study could be classified as ‘riparian’, in they had a small stream or creek (~2m wide) running through them. These four remnants did not support higher bat species richness compared with non-riparian remnants (5.2±0.54, cf 4.95±0.29), or bat activity (373.13±157.62, cf 625.68±145.8), or feeding buzzes (0.6±0.25, cf 1.18±0.25).

Using a GIS, we measured two separate distances for each survey point to water – the first was to any form of mapped water (streams, rivers, farm dams, or irrigation channels), irrespective of their size. The second was the distance to any natural water body or large farm dam, >1 ha in surface area. The two measures were highly correlated, and as we could only include one in our modelling, both were plotted against our bat responses. The second measure appeared to fit the data better, so we used this in our analyses. It is likely that neither the ‘riparian’ nature of the remnants, nor the distance to water, had a strong effect on the outcomes of our surveys because, at the time, water was abundant across much of the landscape. As noted in our Methods section, there was high rainfall in the summer that our surveys took place, so not only were rivers, streams and dams full or overflowing, but many fields were ankle-deep in free-standing water (Fig. S4). The extent of this free-standing water was quantified during the habitat surveys, as part of the ground-cover transects. However, if emergent vegetation was present at each point, this was recorded and ‘native’ or ‘non-native’ instead of ‘water’, so not enough ‘water’ data points were recorded to be meaningful in the analyses. Finally, we trialled adding the elevation of each of the survey points, to act as a proxy for the patterns of water accumulation across the landscape, to our models, but this did not strengthen them. We therefore concluded that, in years with very high rainfall, water resources are of lesser importance to bat communities than other factors, such as roost or forage availability.

Appendix S2 - Bat call analysis in Anascheme

Call files recorded during the acoustic surveys were analysed using AnaScheme software, vers 1.0 (Gibson & Lumsden 2003; Adams, Law & Gibson 2010). Anascheme reads sound files recorded by Anabat detectors, and models individual bat search-phase pulses using regression analysis. Pulses are then identified, using a regional identification key; ours was built by BL, and was based on keys developed for Law & Chidel (2006) and Hanspach *et al.* (in press). The key was tested on a library of 78 reference calls collected from the study region during pilot trapping surveys, and was found to correctly identify 72.4% of calls, 27.6% were deemed “Unknown”, and none were identified incorrectly. Our key included 14 species (Table 1), and of these only *Rhinolophus megaphyllus*, a cave-dwelling bat, was not recorded during the surveys. Calls of *N. geoffroyi* and *N. gouldi* cannot be reliably distinguished, therefore the two species were pooled to form the “*Nyctophilus* sp.” complex. This was also the case for *Vespadelus darlingtoni* (40-45 kHz) and *V. regulus* (40-45 kHz), which were pooled as “*Vespadelus darlingtoni*/*regulus*”. *V. regulus* is known to also produce a higher-frequency (HF) call (54-55 kHz) around large water courses in the field area (Law & Chidel 2006), so Anascheme identified these separately as “*Vespadelus regulus* HF”.

We set Anascheme to identify call files only when a minimum of three pulses, and >50 % of the pulses within a pass, were classified as the same species. If > 50 % of pulses could not be allocated to the same species, the file was allocated as an "Unknown sp.". Calls from the surveys which were identified as *Chalinolobus picatus* or *Saccolaimus flaviventris,* which are listed as threatened in the region, were manually checked, as were those of *Vespadelus regulus* "HF". All files which Anascheme identified to contain bat calls were then separately filtered for feeding buzzes, using a filter developed by BL that recognises short sequences of steep linear calls produced in rapid repetition, which typifies feeding buzzes. Any files flagged as containing feeding buzzes were also manually and audibly checked to exclude non-feeding buzzes using Anabat 6 software (vers 6.3, Chris Corben, www.hoarybat.com)

Appendix S3 - Data exploration and potential correlation of fixed effects

Many habitat measures were taken at each survey point, and we were limited in the number of explanatory variables we could include in our models. Some of the bat species which occur in our study region roost under decorticating bark (Lumsden, Bennett & Silins 2002), and trees in later stages of senescence are more likely to have developed hollows (Bennett, Lumsden & Nicholls 1994). It is also known that some bat species are sensitive to high levels of habitat 'clutter' (Law & Chidel 2002; Adams, Law & French 2009). For these reasons, we suspected that variables relating to trees would be most important for bats, so we first trialled a Principal Components Analysis with the following variables: tree species, stage of senescence, the number of tree stems, and bark types. However, when the components resulting from this ordination were plotted against the bat responses, it appeared that they had little power in explaining the observed patterns. We therefore repeated the process with measures that we felt reflected the degree of anthropogenic disturbance at each site, and habitat structure, based on measurements commonly incorporated into condition assessments for these woodlands (Gibbons *et al.* 2005; Department of Environment Water Heritage and the Arts 2008). These were the variables presented in the final analysis: the total basal area of trees, the number of trees with hollows, the volume of logs, the percent of the ground cover which was native, and the percent cover of shrubs across the 1 ha site. The two new variables produced from this ordination, "habitat component 1" and "habitat component 2", appeared to be more effective in explaining patterns in our bat responses.

We checked for correlation between all pairs of explanatory variables using the 'cor' function in R, and by visually inspecting pairwise plots. Explanatory variables appeared to be completely independent, with two exceptions. First, survey season was correlated with some of the weather variables: the juvenile season was hotter ($p < 0.001$, $df = 204$, $t = -7.9$) and windier ($p < 0.048$, $df = 222$, $t = -2.0$). The average maximum overnight temperature in the maternal season was 23.4°C, and the maximum wind speed was 24.7 km/hr. By comparison, in the juvenile season the average maximum overnight temperature 29.6°C, and the maximum wind speed was 26.8 km/hr. We chose to leave all three variables (season, temperature and wind speed) in the analysis because 'season' captures important life-history information that the weather variables do not. We anticipated an increase in the three bat responses in the juvenile survey period because of the increase in the number of volant individuals in the population, irrespective of weather, and this is what we found.

The second correlation relates to land use classes and habitat variables. Native pastures had lower values of habitat component 1 than other land use classes (Fig. S5a), though this component did not prove to have a very strong effect in final field models (Table

4b). All land use classes differed from each another with regards to habitat component 2, with the exception of the two crop classes (cereals and canola, Fig. S5b). Once again, we chose to leave all three variables (land use, habitat component 1 and habitat component 2) in the analysis, because certain factors associated with land use class, other than habitat structure, may affect our bat responses. Crops are associated with cultivation practices such as tillage and harvesting, and both cropped fields and exotic pastures are subjected to higher inputs of pesticides, fertilisers, or herbicides which may affect bat prey. Native pastures are also less likely to have been set-stocked for long periods of time, and it is not known how this may indirectly affect bats.

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Supporting Tables and Figures

Table S1. Descriptions of the seven types of bark encountered in field surveys of *Eucalyptus* trees, with example species.

Bark type	**Description**	**Typical groups**	**Example species**
Smooth	Smooth bark over entire tree	Mallees	*Eucalyptus rubida*, *E. rosii*
Flaked	Annual bark shedding, residual bark in flakes, underlying bark is smooth	Red gums	*E. mannifera*, *E. blakelyi*
Rough base	Smooth bark over branches and upper trunk, rough or ribbon bark collects at trunk base	Ribbon gums	*E. melliodora*
Fibrous	Long-fibred, partially furrowed, spongy bark that can be stripped manually in long pieces	Stringy barks	*E. macroryncha*
Sub-fibrous	Bark fibres of short-medium length, narrow longitudinal fissures, possibly with shaggy base	Box species	*E. microcarpa*, *E. populnea*, *E. albens*
Furrowed	Bark is thick, hard, widely furrowed and retained on tree	Ironbarks	*E. sideroxylon*
Tessalated	Dead bark retained on tree in small, short-fibred plates/tiles	Bloodwoods	*E. gummifera*

Table S2. Loadings of variables on habitat components 1 and 2, from the principal components analysis of vegetation measures taken from a 1 ha area around each survey point.

Covariate	**Loadings of habitat component 1**	**Loadings of habitat component 2**
Number of trees with hollows	-0.514	0.294
Total basal area of trees	-0.490	0.347
Volume of logs (m^3)	-0.463	0.268
Per cent ground cover which is native	-0.392	-0.531
Per cent cover of shrubs	-0.357	-0.663

Table S3. Combination of variable groups used in the 16 alternative generalised linear mixed models, used to predict bat species richness, activity and feeding. An ‘X’ indicates that a variable group was included in the model.

Model no.	**Local habitat “LOC”**	**Adjacent habitat “ADJ”**	**Landscape context “LCSP”**	**Survey conditions “COND”**	**Random effects(s)**
1	X				X
2		X			X
3			X		X
4	X	X			X
5		X	X		X
6	X		X		X
7	X	X	X		X
8	X			X	X
9		X		X	X
10			X	X	X
11	X	X		X	X
12		X	X	X	X
13	X		X	X	X
14	X	X	X	X	X
15				X	X
16 (null)					X

Table S4. Expanded list of species recorded in each of the land use classes. "C" is "Count", the number of survey points that each species was recorded at, with the listed in parenthesis in the header row. "A" represents "Activity", the total number of calls recorded, and "F" represents "Feeding buzzes".

	Remnant (118)			**Cereal crop (39)**			**Exotic pasture (37)**			**Canola crop (15)**			**Native pasture (16)**			**Total (225)**		
Species	C	A	F	C	A	F	C	A	F	C	A	F	C	A	F	C	A	F
Unknown sp.	116	57,490	6	39	13,351	15	35	2,312	4	15	841	7	16	698	4	221	74,692	36
Vespadelus vulturnus	108	2,881	33	36	909	17	31	1,500	37	13	330	8	16	528	10	204	6,148	105
Chalinolobus gouldii	86	1,507	31	23	764	99	22	940	17	15	524	34	13	116	14	159	3,851	195
Mormopterus sp. 4	72	730	11	25	653	25	21	467	11	12	188	3	13	169	3	143	2,207	53
Scotorepens greyii	51	1,879	7	20	112	2	10	102	2	3	36	3	9	70	0	93	2,199	14
Scotorepens balstoni	56	511	13	17	98	13	14	151	2	8	41	8	10	28	5	105	829	41
Mormopterus sp. 2	33	279	20	21	182	2	7	19	1	5	14	0	8	139	2	74	633	25
Tadarida australis	40	246	0	21	94	0	15	32	1	6	26	0	8	81	0	90	479	1
Vespadelus darlingtoni/regulus	43	144	2	15	31	1	14	153	6	6	36	2	6	15	0	84	379	11
Chalinolobus morio	44	109	1	14	75	0	9	39	0	3	5	0	5	13	0	75	241	1
Nyctophilus sp.	28	52	2	12	47	2	16	57	0	5	12	1	10	22	0	71	190	5
Vespadelus regulus (HF)	22	59	3	12	34	0	4	9	1	2	2	0	8	10	0	48	114	4
Chalinolobus picatus	3	3	0	0	0	0	2	2	0	0	0	0	1	1	0	6	6	0
Saccolaimus flaviventris	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0
Rhinolophus megaphyllus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Total		65,891	129		16,350	176		5,783	82		2,055	66		1,890	38		91,969	491

Fig. S1. The eight stages of *Eucalyptus* senescence, taken from Rayner (2008). (1) Immature tree, branches upright, (2) mature, adult tree, branches spread and intact with healthy crown, (3) mature tree with signs of senescence, some large broken branches, crown thinning (<50%), (4) live adult tree, largely bare, but small patches of canopy or areas of regrowth, (5) dead stag with majority of branches (>50%) intact, (6) dead stag with <50% branches remaining, (7) upright, dead stag with no major branches remaining, and (8) broken or cut stump.

Fig. S2. Mean dry biomass of nocturnal arthropod samples collected in each of the black-light traps, with error bars representing 95% confidence intervals, and sample sizes listed below each of the plotting points.

Fig. S3. Plots showing the arrangement of survey points of different land-use classes on axes 1, 2, and 3 from non-metric multidimensional scaling (NMDS), based on an abundance matrix of species.

Fig. S4. Images depicting the free-standing water and pools that were present in many of the remnant and field sites, following high rainfall during the study period.

Fig. S5. Differences in the mean levels of the two habitat components between the different land use classes: a) habitat component 1, and b) habitat component 2. Land use class contrasts are shown on the y-axes.

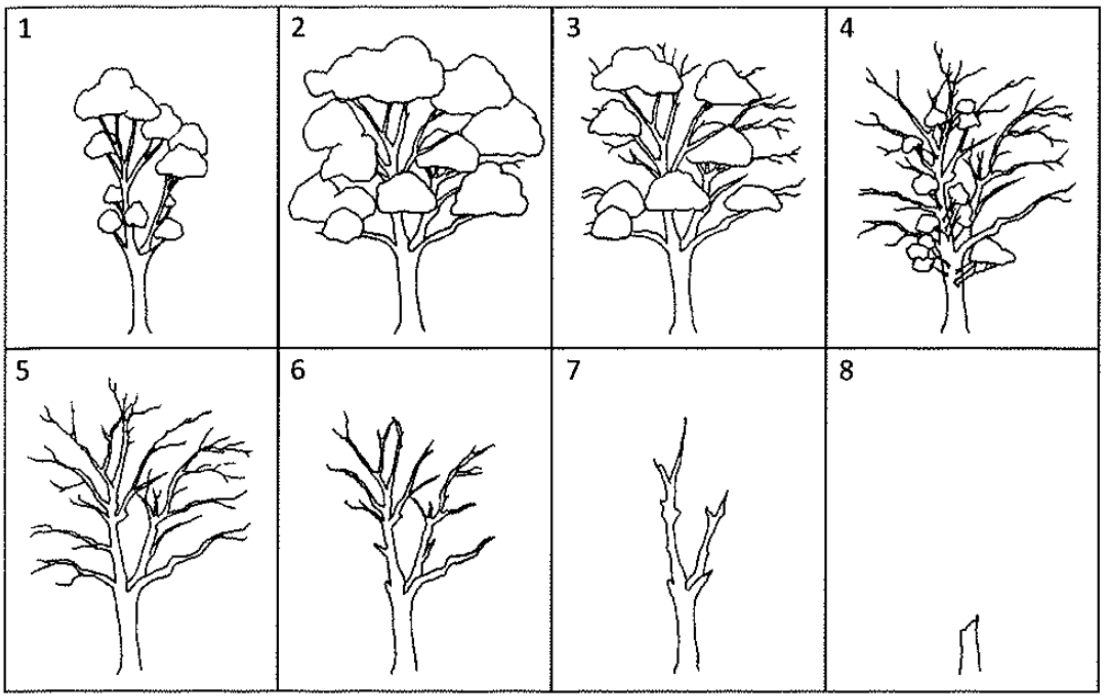


Fig. S1.

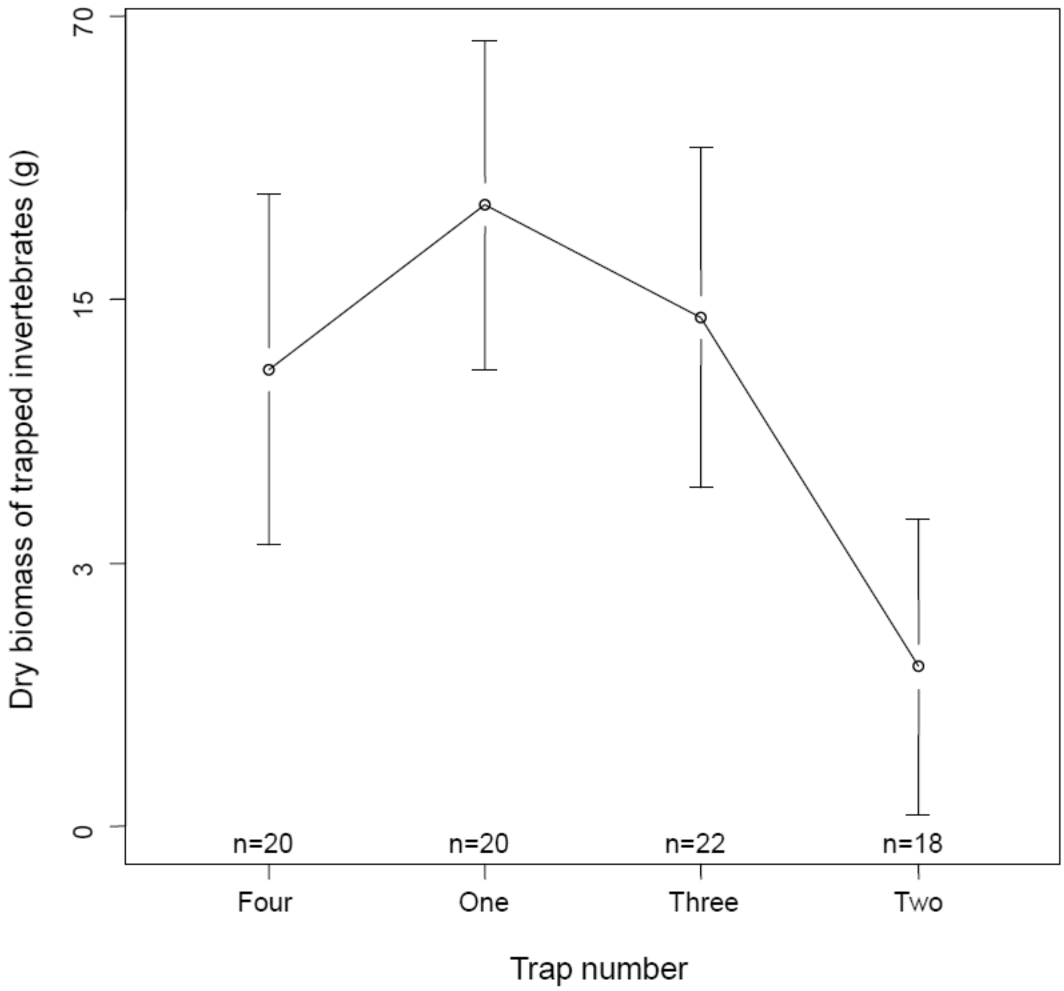

70
15
3
0
Dry biomass of trapped invertebrates (g)
n=20
n=20
n=22
n=18
Four
One
Three
Two
Trap number

Fig. S2.

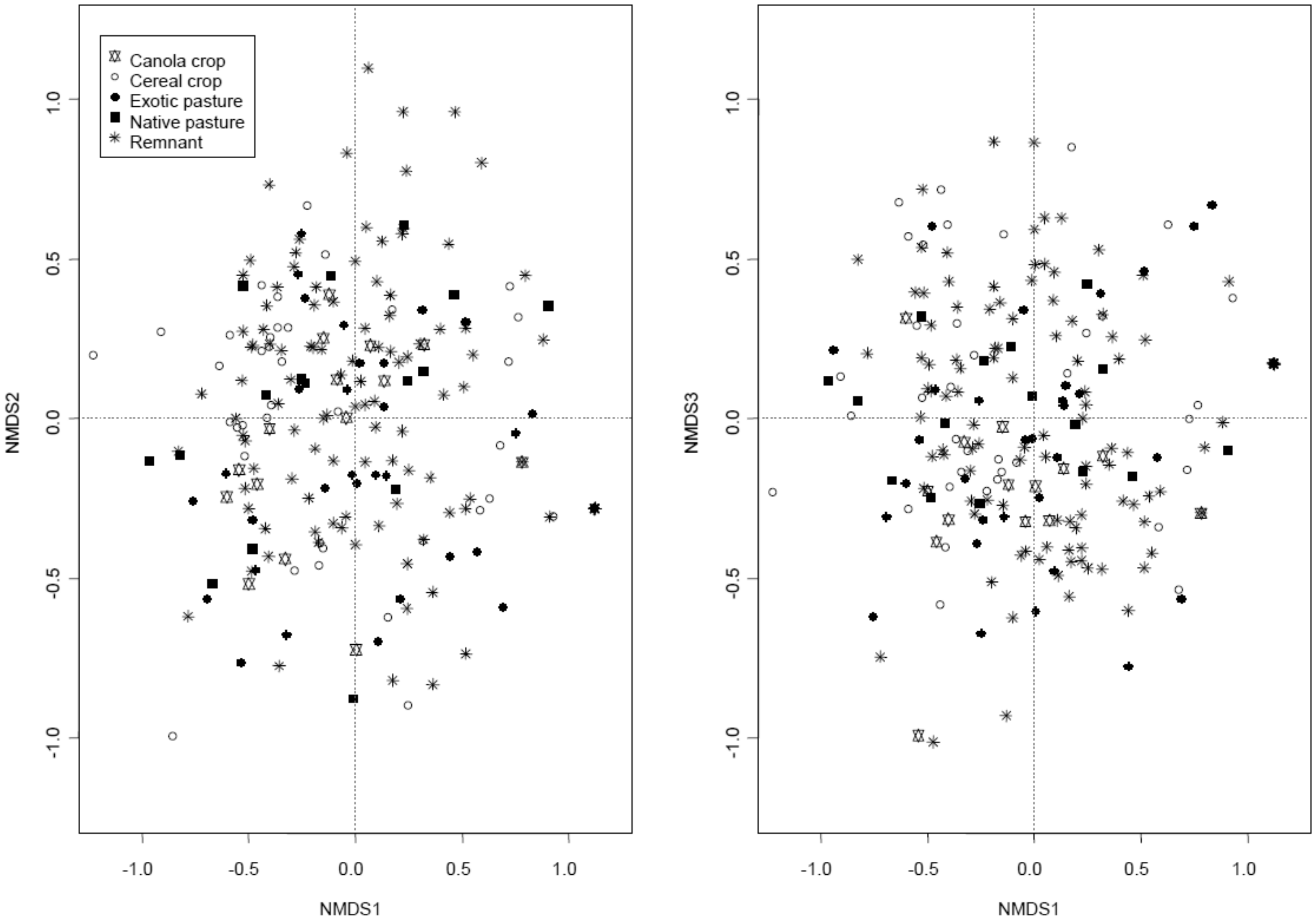
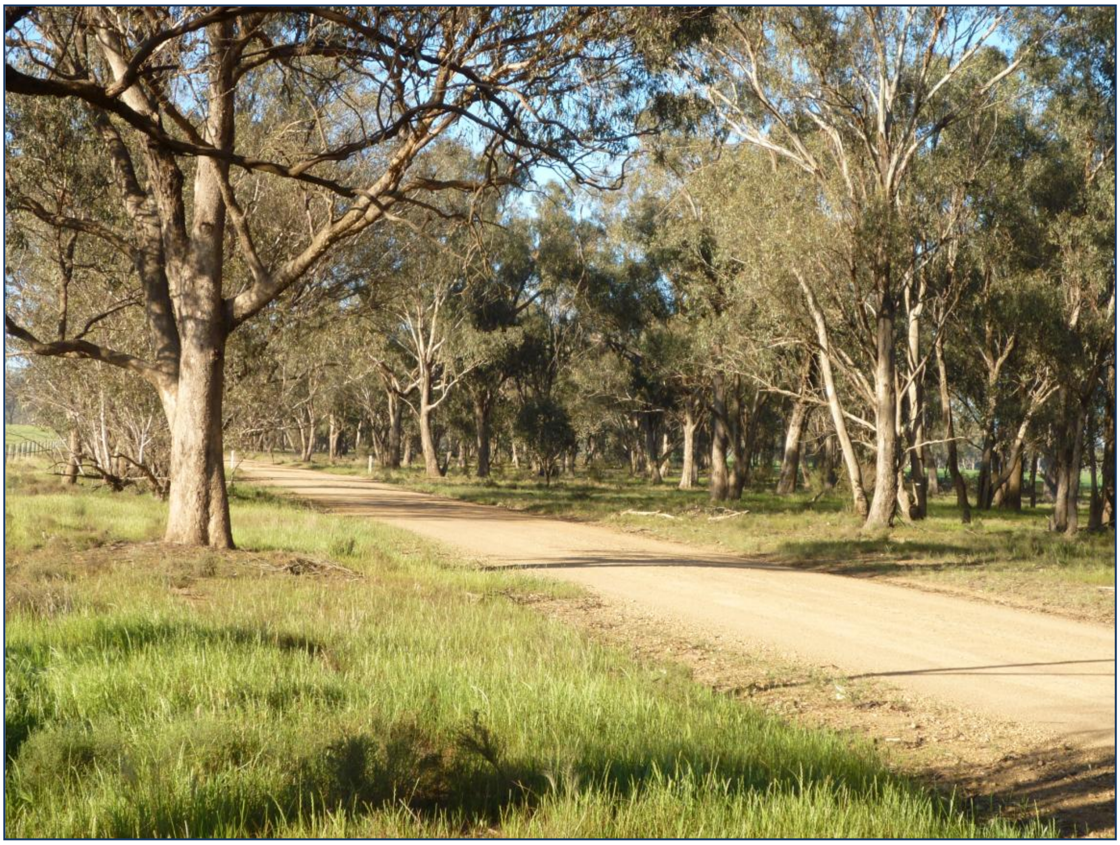

Canola crop
Cereal crop
Exotic pasture
Native pasture
Remnant
NMDS2
NMDS3
NMDS1
NMDS1
-1.0
-0.5
0.0
0.5
1.0

Fig. S3.

Fig. S4.

Paper VI. The effect of planning for connectivity on linear reserve networks



Grey box (*Eucalyptus microcarpa*) woodland in the “Eualdrie” TSR, Bogolong, NSW. Travelling stock routes such as this which form corridors of remnant vegetation play an important role in providing connectivity in otherwise highly fragmented landscapes. Image: P. Lentini.

This research paper is in review: Lentini, PE, Gibbons, P, Fischer, J, Carwardine, J, Drielsma, M & Martin, TG (in review) The effect of planning for connectivity on linear reserve networks. *Conservation Biology.*

Abstract

Although the concept of connectivity in conservation is now decades old, it remains both poorly understood and defined. Some argue that metrics known to have a strong relationship with biodiversity (such as habitat quality and area) should take precedence in conservation planning. However, many heavily fragmented landscapes are characterised by features such as streams and hedgerows which are inherently linear, so for these strategic planning that enhances both representation and connectivity may be possible with little effect on the cost, or area and quality of the reserve network. We assessed how alternative approaches for accounting for connectivity affect planning outcomes for linear habitat networks, using the stock route network of Australia as a case study. Our planning objective was to represent vegetation communities across the network at a minimal cost. We ran scenarios with a range of representation targets: 10, 30, 50 and 70% of the original extent of each vegetation community in the linear remnants and nearby protected areas, using three different approaches for accounting for connectivity (Boundary Length Modifier, Euclidean Distance, and Landscape Value). We found that decisions regarding the specific target and connectivity approach used may affect the siting of reserve systems: at conservation targets ≥50% networks designed using the Euclidean Distance and Landscape Value connectivity approaches consisted of a greater number of small reserves. Hence, by maximizing both representation and connectivity, these networks compromised on larger contiguous areas. However, targets this high are rarely employed in real-world conservation planning, so concerns regarding undue costs of connectivity appear unwarranted. Approaches for incorporating connectivity into the planning of linear reserve networks which take into account not only the spatial arrangement of reserves, but also the characteristics of the intervening matrix, highlight important sections that ‘link’ the landscape, and which may otherwise be overlooked.

Introduction

The concept of connectivity originally appeared in the conservation literature in the 1980s (Merriam 1984) and has been the subject of much debate since, particularly in relation to habitat corridors (Bennett 2003; Crooks & Sanjayan 2006). Part of this is due to the fact that its definition is somewhat fluid and varies between sub-disciplines: in metapopulation ecology, connectivity is measured between individual patches, whereas landscape ecologists often classify it as the ability of an entire landscape to impede or facilitate movement (Moilanen & Hanski 2001). Recently, some authors have further emphasised the need to improve connectivity in the landscape due to shifts in distribution of species with climate change (Hannah et al. 2002).

A large number of metrics have been developed which attempt to quantify connectivity (Crooks & Sanjayan 2006; Kindlmann & Burel 2008), but it is extremely context-specific, being a function of both the species of interest (habitat preferences and dispersal abilities) and the spatial arrangement, area, and quality of patches (Tischendorf & Fahring 2000). Therefore, these metrics are associated with a high degree of uncertainty, and some argue that other measures more strongly related to biodiversity, such as habitat area and quality (Turner 2005), should take precedence over connectivity when determining the location of conservation areas (Hodgson et al. 2009). Despite a lack of consensus about whether and how connectivity matters, large investments continue to be made by both government and private institutions to enhance it in the landscape. Projects such as “Gondwana Link” in Australia, “Yellowstone to Yukon” in North America, and the “Mesoamerican Biological Corridor” of central America represent multi-million dollar endeavours, together constituting hundreds of partner organisations (Worboys et al. 2010). It is currently not known whether these investments could be better spent, and systematic methods for identifying cost-effective connected reserve networks are not widely applied in these examples.

A range of decision support tools have been developed to facilitate the systematic siting of reserves (Pressey et al. 2005; Moilanen & Kujala 2006; Ball et al. 2009). These tools address the fact that socio-economic factors compete with the establishment of conservation areas (Carwardine et al. 2008), and use mathematical approaches to ensure that as many species as possible are protected whilst minimising conflict with other human demands. They take into account not only what is to be conserved, but also how much – a measure known as the 'conservation target'. Targets are frequently dictated by policy (Svancara et al. 2005), or are 'evidence-based', in that they represent empirically-derived critical thresholds in population size or habitat area (Drielsma & Ferrier 2009). Expert elicitation can also be used to determine appropriate targets (Airame et al. 2003). In some cases, authors provide little or no justification as to how and why their targets were set (Carwardine et al. 2009). If solutions are sensitive to targets, their arbitrary allocation could greatly influence our understanding of how certain parameters (such as connectivity) affect planning outputs.

Species inhabiting highly modified and fragmented landscapes may benefit most from enhanced connectivity (Donald & Evans 2006), and these landscapes are often dominated by linear habitat networks such as streams, hedgerows and roadside reserves (McCollin et al. 2000; Hermoso et al. 2012). The configuration of these linear landscape elements is often inherently 'connected', so strategic planning that enhances both representation and connectivity may be possible with little effect on the cost, or area and quality of the reserve network. However, examples of systematic conservation planning for these networks are scarce, so the potential effects of alternative connectivity approaches on costs are not known. To explore these issues further, we used the stock route network of Australia as a case study. Stock routes constitute roadside strips of remnant vegetation and form a large-scale habitat network across the state of New South Wales (Fig. 1). The management of this publicly owned and managed system is currently under review, with the potential that some sections could be sold to private landholders for agricultural production, and some retained for conservation (Lentini et al. 2011a). Against this background, we aimed to determine (1) the implications of including connectivity as a goal in conservation planning for linear networks; and (2) whether the level of ambition in representation targets interacts with the effects of connectivity approaches.

Methods

We used Zonae Cogito (vers. 1.22), a user interface for the Marxan conservation decision support tool (vers. 2.1.1), for our analyses. Marxan addresses the "minimum set" problem, which is to meet a set of predefined targets for a minimum cost. Targets are set for each of the 'conservation features' which are anything that a planner is aiming to protect within their conservation reserve network, such as the area of habitat for an individual species, vegetation community, or other features of the landscape. Marxan minimises an objective function via a process of simulated annealing, to select candidate reserves from a pool of potential areas (or 'planning units'), taking into account planning unit costs and the locations of the conservation features for protection (Ball et al. 2009). Each alternative set of runs of Marxan, where input parameters such as targets or conservation features are altered, are known as 'scenarios'.

Planning units – the stock route network and protected areas

Our analysis was carried out on stock routes of the "wheat-sheep" region of New South Wales, Australia (Fig. 1), which covers 41M ha of the state and has been heavily cleared for agricultural production. Remnant vegetation primarily exists as isolated protected areas, scattered trees in fields, and as stock routes, which cover 485,818 ha (1.2%) of the study area (Lentini et al. 2011b).

We used the "TSR Conservation Values" spatial layer (provided by the NSW Department of Environment, Climate Change and Water, DECCW, 2010) as the basis for our planning unit layer, because the stock route polygons in this had already been subdivided into sections as used by managers. This initial set of 4,865 individual planning units ranged from 0.2 - 4,300 ha (μ=99.8ha $\pm$ 1.43 SE) in area. Stock routes often occur in low-lying fertile portions of the landscape, and represent vegetation communities that are characteristically under-represented in the protected area system (Lentini et al. 2011b). The differing levels of representation of vegetation communities should affect their priority for reserve expansion. Therefore, we also included the 333 protected areas which occur in the region in our analysis, using the 'NSW National Parks and Wildlife Service Estate' layer from the DECCW data download website (http://mapdata.environment.nsw.gov.au), released on the 1st of April 2009. Protected areas were 'locked in', so solutions always included these 333 planning units (and the vegetation within them). Hence, in total 5,198 planning units were considered.

Conservation features and targets

We used two data layers to account for the major vegetation classes, and the heterogeneity of these across the study region. The first was the "Australia - Estimated Pre-1750 Major Vegetation Groups - NVIS Stage 1, Version 3.0" spatial layer created by the Australian Government's National Land and Water Resources Audit, 2002. Based on this, 21 vegetation classes which had historically occurred were identified. We then stratified each vegetation class according to the "landscape type" that they fell within, using the "Mitchell Landscape V3" layer (Eco Logical Australia 2008). Each stock route was then classified into one of 1,452 'vegetation communities', which were used as conservation features. For example, "Mallee Woodlands and Shrublands/Goonoo Slopes Landscape" (Table 1, see Appendix S1 in the Supplementary Material for further detail). We also chose to test a range of conservation targets: 10, 30, 50 or 70% of the current extent of each conservation feature in the stock routes and the protected areas. This was not intended to reflect what an 'adequate' reserve network would be, but rather that it has been proposed that some stock routes will be sold and some protected, with no comment on what the extent of either of these actions may be.

Cost

A number of costs were combined to determine the overall cost of taking conservation action in each of the planning units. Because stock routes are already publicly owned, there would be no purchase cost to incorporate them into the national reserve system. However, there would be an opportunity cost of reserving a stock route which may otherwise be sold (i.e. a lost sale price). For each planning unit, we estimated this 'opportunity cost' using an unimproved land value layer, obtained from the Australian State land valuation offices, and aggregated to the scale of local government areas.

We also accounted for differences in the condition of vegetation within stock routes remnants by adding a restoration cost to degraded planning units. We assumed that once this cost had been committed, and restoration activities were complete, these planning units would have equal conservation value to those which were intact. We used the "Vegetation Assets, States and Transitions, vers. 2" layer (Lesslie et al. 2010) to determine the current condition of each of our planning units, and restoration cost estimates for each vegetation class were based on information provided by Greening Australia (pers. comm., Graham Fifield, Greening Australia Capital Region, Gibson-Roy et al. 2010: see Appendix S2 for detail of restoration calculations). No restoration costs were set for protected areas, or 'Unknown', 'naturally bare' and 'Aquatic' vegetation types. The final cost of each planning unit was therefore the unimproved value, plus restoration costs, which ranged from \$31.50 for 0.2 ha of open eucalypt forest, to \$6,414,372 for a stock route covering 4,675 ha of degraded tussock grassland and chenopod shrubland (μ= \$207,171 ± \$5,174 SE, all costs in \$AUD).

Incorporating connectivity approaches

We analysed 16 different scenarios in Marxan, testing the effect of every combination of four connectivity approaches: No Connectivity, Boundary Length Modifier, Euclidean Distance, and Landscape Value with one of the four targets: 10, 30, 50 or 70% of the current extent of each conservation feature in the stock routes and protected areas.

No Connectivity

The first set of scenarios used no measure of connectivity ('No Connectivity'). The siting of reserves was based purely on the representation of conservation features in the most cost-effective way possible, irrespective of their spatial configuration.

Boundary Length Modifier (BLM), standard

The second set of scenarios employed the BLM approach, which is commonly used in Marxan to adjust the degree of 'clumping' of reserves in solutions (Stewart et al. 2003). It takes into consideration the fact that two planning units adjacent to one another, if reserved at the same time, will practically form one reserve and the boundary between them will dissolve. The length of the perimeter, or the boundary 'cost' of adjacent units reserved together is therefore lower than two planning units of the same size which are reserved apart from one another. The 'Boundary Length Modifier' adds a cost to the objective function in Marxan for every additional length of boundary or perimeter, and causes Marxan to more frequently select units adjacent to one another in order to minimise this cost. The approach assumes that more clustered solutions are more connected. Detail of the calibration of the BLM modifier for these scenarios, and also Euclidean Distance scenarios, can be found in Appendix S3.

Euclidean Distance, using BLM

In many circumstances, closer planning units will be more functionally connected than distant ones, even if they are not spatially touching. For example, sections of stock routes may be transected by roads, yet are still only metres apart and functionally connected for the majority of taxa. However, the BLM approach described above will not pick up on this. An alternative approach is to calculate the straight line (Euclidean) distance between pairs of sites, and aim to minimise the overall distance between all pairs of sites (Moilanen & Nieminen 2002). We calculated the distance between all pairs of sites within 50 km of each other, and treated these distances as boundary lengths. We then adjusted the BLM in our analyses, so that sites closer together would be prioritised, assuming that sites which are closer are also more connected.

Landscape Value

Finally, we tested the "Landscape Value" metric, developed by NSW Office of Environment and Heritage. This addresses a shortcoming of the BLM-type approaches, which is that they do not take into account the nature of the landscape between the patches/planning units. Landscape Value (LV) mapping aims to highlight 'linking' areas, where conservation of existing vegetation, condition improvement of degraded vegetation, or rehabilitation of cleared areas are most likely to contribute to maintaining or enhancing functional connectivity across a region.

LV was derived, using a graph theoretical approach (Urban & Keitt 2001), by systematically modelling habitat linkages across a broad range of ecological scales. The connectivity measures employed were Colonisation Potential (Drielsma et al. 2007b) and Neighbourhood Habitat Area (Hanski 1999; Drielsma et al. 2007a). The LV was calculated for each planning unit based on the vegetation it supported (Table 1), and these ranged from 0.0-844.25 (μ= 133.83 ± 1.08 SE) – further detail of this can be found in Appendix S4. To incorporate these into the Marxan analyses, "Landscape Value" was treated as a separate conservation feature, and was set the target appropriate for each scenario (i.e. if the conservation target for vegetation communities was 30%, then the target for Landscape Value was also 30%).

Analyses in Marxan

For each scenario, Marxan was run 1,000 times, with each run producing a near-optimal solution. The 'best solution' for each scenario was that which met all objectives at minimum cost. 'Irreplaceable' planning units were those which were included in all 1,000 solutions for a scenario, and were therefore essential for meeting representation targets. When there are more 'irreplaceable' planning units in a scenario, there are also fewer options to swap and substitute areas, so 'irreplaceability' acts as an indicator of scenario flexibility. When assessing the outputs of each of the scenarios and for plotting, we present data only for the stock route planning units selected, as protected areas were 'locked to every solution.

Results

Number and area of planning units

We found that conservation reserves increased in area with the conservation target, but a proportionally greater area was required to meet the 50-70% targets for the Euclidean Distance and Landscape Value scenarios (Fig. 2a). This pattern was also found for the number of individual reserves/planning units in the network, with more planning units required to meet targets for the Euclidean Distance scenario at targets ≥50%, and to a lesser extent this was also the case for the Landscape Values approach at targets ≥30% (Fig. 2b).

It appears that this increase in the number of planning units required to meet targets for the Euclidean Distance and Landscape Value scenarios was being met by incorporating smaller individual parcels into the reserve network. The average planning unit included in the reserve network was smaller at targets ≥50% for the Euclidean Distance scenarios, and at targets ≥30% for Landscape Values scenarios (Fig. 3a). It is interesting to note that this is not simply because all of the large planning units had been exhausted – at targets ≥50%, the average planning unit excluded from solutions for Euclidean Distance/Landscape Value was much larger than that for No Connectivity/BLM (Fig. S1).

Cost

Patterns in the financial cost of the solutions closely follow that of the number and area of planning units. Overall, relative costs remained almost unchanged across the connectivity approaches where the target was ≤30%, but this pattern changed in the 50-70% target scenarios (Fig. 2c). The best solution for the Euclidean Distance approach increased from being 4% more expensive than No Connectivity at the 10% target, to 20% more expensive at the 70% target (Table 3). By comparison, at the 70% target level, the Landscape Value scenario was only 5% more expensive than the No Connectivity approach.

Both targets and connectivity affected the cost of the individual planning units being selected. At 10-30% targets, the Euclidean Distance approach selected the more expensive planning units, but this switched at targets ≤50% (Fig. 3b). The Landscape Value approach consistently resulted in the selection of cheaper planning units, irrespective of targets. Finally, the way in which funds were allocated, in relation to management actions, was only subtly affected by connectivity. All approaches generally allocating equal amounts to protection and restoration in each scenario (Fig. 3c), with the exception that the Euclidean Distance approach resulted in proportionally greater investment in protection than restoration at the 70% target.

Irreplaceable planning units

Only the inclusion of the Euclidean Distance approach had a strong effect on the number of planning units which were completely irreplaceable, and this was only at conservation targets ≥50% (Table 2, Fig. S2a). However, when looking at the average number of times each planning unit was selected in the 1,000 near-optimal solutions, we found that from the 30% target level the Landscape Value approaches were also generally less flexible, with a greater proportion of the planning units being needed in more solutions to meet targets (Table 2, Fig. S2b).

Spatial arrangement and overlap of planning units

We investigated potential changes to the spatial arrangement of reserves using a map of the planning units selected in the best solutions for each of the No Connectivity, Euclidean Distance and Landscape Values scenarios, for an 85×75 km section of the study region (Fig. S3). The number of planning units and total area selected increased with the size of the conservation target, particularly at the 50% target level for the Euclidean Distance scenario. We also quantified the percent overlap of individual planning units selected in the best solution for each scenario, and found that as the conservation target increased, the connectivity approaches selected increasingly different planning units (Fig. 4). Those in the No Connectivity and BLM scenarios were most similar, and the BLM and Euclidean Distance scenarios were the least similar.

Discussion

Should connectivity be incorporated, and what impact does it have?

Decisions regarding which specific approach to use to incorporate connectivity into linear reserve planning may affect the siting of the final reserve system. The Boundary Length Modifier, which is one of the most commonly-used approaches for adjusting connectivity in Marxan, had almost no effect on solutions. This is likely to be because stock routes are not spatially continuous, being transected by roads and streams, and what boundaries they do share are very narrow in width in proportion to the length of the remnant. Therefore, the standard BLM may not be appropriate when planning for connectivity in linear networks where many planning units that are effectively connected are not directly adjacent, or share only very narrow boundaries. In these situations, the BLM should be used in more sophisticated ways, but examples of this in the literature so far are rare (though see Hermoso et al. 2012). Here, we addressed this issue by using the Euclidean Distance approach, which resulted in a subsequent increase in the aggregation of reserves, and particularly at 50-70% target levels (Fig. S3). However, this came at the cost of both monetary resources and reduced size of the remnants selected (Figs. 2-3). Aggregation of suitable habitat is important in low-cover fragmented landscapes (Radford et al. 2005) such as our study region, but this will only boost functional connectivity if reserves are spaced close enough to be within the dispersal thresholds of target species (Doerr et al. 2011). Because these thresholds were only taken into account in the calculation of Landscape Values, which also considers vegetation outside of the planning units, it may be most effective in promoting actual functional connectivity. Moreover, although the Landscape Value approach increased the cost of reserve networks at 50-70% representation targets, these were of a lower magnitude than Euclidean Distance scenarios (Fig. 2c).

An important question is therefore how much connectivity should be used to drive reserve siting, given that we identified a potential trade-off with total cost and individual reserve area. This appears to go against conventional wisdom in conservation (e.g. Bender et al. 1998), but in our example, these smaller parcels of land were more spatially connected, potentially creating functionally larger clumps. When executed with sufficient consideration, planning that accounts for connectivity should therefore provide conservation benefits above and beyond just representation, by ensuring adequate performance of the reserve networks. For example, Olds et al. (2012) reported that fish abundance in marine protected areas differs from unprotected areas only if they are sufficiently connected to adjacent mangrove habitat, so without connectivity, the reserves are not serving their purpose. Although we know that larger conservation reserves are often needed to ensure the persistence of viable breeding populations (Nicholson et al. 2006), if a single catastrophic event causes the extinction of populations within isolated reserves (Possingham et al. 2000), they cannot be recolonised from other source populations (Fahrig & Merriam 1994). For this reason, Doerr et al. (2011) argue that planners need to account for both 'habitat for settlement' and 'habitat for dispersal', with smaller patches effectively linking core habitat patches. In highly fragmented landscapes, such as our study area, it has been shown that small parcels of land (Geert et al. 2010; Gillies & St. Clair 2010), even to the level of individual trees (Fischer & Lindenmayer 2002a), can increase functional connectivity, so a balance of both large and small remnants may be optimal (Fischer & Lindenmayer 2002b).

Do the effects of connectivity metrics interact with conservation targets?

Our findings demonstrate that the largest differences between accounting, and not accounting for connectivity occurred when 50 or 70% of conservation features were targeted for inclusion in the reserve network. There was also comparatively higher 'irreplaceability' in the Euclidean Distance and Landscape Value scenarios at the 50-70% level (Fig. S2a,b), indicating that there were fewer options available to meet both the vegetation representation and connectivity requirements. Because of this, Marxan was forced to select planning units that were smaller and less cost-effective (Fig. 3a,b). In reality, targets this high are rarely employed: in their review of the conservation planning literature Svancara et al. (2005) found that the average policy-driven target is set at 13.3%, those based on conservation assessments at 30.6%, and those on threshold analysis at 41.6%. Of course, these targets refer to the entire historical extent of vegetation communities or other conservation features, and in our example, we assessed only the stock routes and protected areas. In our study region stock routes cover only 1.2% of the land (and together with protected areas 8.8%), so even if all were set aside as reserves, we would still not meet a typical policy-based landscape target– let alone reach an ecologically adequate level of representation. But if unambitious targets such as those listed above are the norm, concerns regarding effects of connectivity metrics on reserve network costs may not be warranted. This is not to say that targets well below 50% are adequate for this system, or indeed any other system (Noss et al. 2012) – in our example, a 50% conservation target for stock routes also corresponds to a potential 50% loss.

Applying conservation planning to real-world linear networks

This study occurred in a developed country which is well-studied. Despite this, we found spatially-explicit data suitable for our purposes to be scarce: both vegetation community and condition data were only accessible at a national scale (although this is not the case for other Australian states). Based on the knowledge of the authors who have conducted field studies in the region (PL, PG), it was clear that some of the areas reported as degraded actually had high conservation value, and that some vegetation communities had been misclassified. For this reason, solutions described here are not intended to be prescriptive, but rather present an exploration of how connectivity can alter the results of planning exercises for linear networks. However, this highlights an important issue. There are often calls in the literature for ‘less data, more action’: it is argued that resources currently allocated to collecting data could be better spent on implementing conservation actions instead (Knight et al. 2010). Sometimes this assessment is fair, especially when not taking action risks the future viability of species or reserves (Martin et al. 2012). In our example though, the ‘action’ to be taken is not only the protection, but also the sale of what are potentially the most intact examples of the most highly threatened vegetation communities in Australia. In this case, better spatial data or ground-truthing is needed to complement these decisions.

A common limitation in systematic conservation planning is that landscapes and threats are dynamic, and usually there are not enough resources available to purchase all selected reserves at once (Possingham et al. 2000). While a reserve network is being established, species may be lost, or sites may become degraded, and it is difficult to predict where and when this will happen (Carwardine et al. 2008). This is not the case for stock routes: because all are currently publicly owned and do not require purchasing, the allocation of funds over a protracted period does not have to be taken into account. Instead, the primary costs involved are lost opportunities in selling the land and ongoing management. With an ever-increasing emphasis on connectivity when planning for global change, stock routes offer a compelling example of where we can gain both connectivity and representation of threatened vegetation in highly fragmented landscapes for little additional financial cost, for typical reservation targets. This scenario is the same for many of the pre-established linear networks around the world. Many authors have noted that these linear features hold value for biodiversity conservation and should be the subject of landscape planning (Leon & Harvey 2006; Lundy & Montgomery 2010). This has partially been taken on board, as linear features are now often integrated into conservation programs. For example, hedgerows and field margins are now incorporated into multi-million dollar agri-environmental schemes across Europe (Kleijn & Sutherland 2003). However, examples of systematic conservation planning for linear networks in the literature are scarce (but see Burel & Baudry 1995; Hirt et al. 2011).

Here, we found that incorporating connectivity in planning for linear networks when representation targets are up to 30% is unlikely to have strong effects on the financial cost of solutions, and may provide additional benefits in the form of improved reserve performance. In situations where targets are to be set higher, planners will have to carefully weigh up the benefits of connectivity with the area needs of target species and budgets for prescribed actions, which may be costly (such as restoration). Connectivity metrics, such as the "Landscape Value" approach employed here, which take into account not only the spatial location of potential reserves but also the characteristics of the intervening matrix, involved moderately greater cost only at high (50-70%) representation targets, and also highlighted smaller 'linking' areas which may otherwise be overlooked in systematic conservation planning.

Acknowledgements

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Tables and Figures

Table 1. Vegetation classes included in analyses, and the number of separate landscape types that each occurs in.

Vegetation	**Number of landscapes**	**Landscape Value score used**
Eucalypt woodlands	295	WL
Eucalypt open forests	278	OF
Tussock grasslands	128	WL
Eucalypt open woodlands	101	WL
Callitris forests and woodlands	86	OF
Eucalypt tall open forests	80	OF
Other grasslands, herblands, sedgelands and rushlands	66	WL
Casuarina forests and woodlands	61	OF
Mallee woodlands and shrublands	61	WL
Acacia forests and woodlands	53	OF
Other shrublands	49	WL
Heathlands	45	WL
Rainforests and vine thickets	44	CF
Chenopod shrublands, samphire shrublands and forblands	35	WL
Naturally bare - sand, rock, claypan, mudflat	16	WL
Eucalypt low open forests	15	OF
Acacia shrublands	12	WL
Inland aquatic - freshwater, salt lakes, lagoons	9	WL
Acacia open woodlands	8	WL
Other forests and woodlands	5	OF
Unknown/no data	5	WL
Total Conservation Features	1,452	

*The 'Landscape Value' score of each planning unit was calculated three times: (1) for taxa which require woodland/grassland (WL) vegetation (2) for taxa which require open forest (OF) and (3) for taxa which require closed forest (CF). Therefore the dominant vegetation class/es of each planning unit determined which of the scores it was allocated. The score allocated for each of the vegetation classes is listed in the third column.

Table 2. Outline of each of the scenarios run in Marxan, and the associated outcomes; protected area planning units are not included in the values listed.

INPUTS			OUTPUTS								
			Best solution					Irreplaceability			
Scenario	**Target**	**Connectivity measure**	**Cost solution $M**	**$M spent purcha-sing**	**$ M spent restor-ing**	**No. PUs**	**Total area (ha)**	**Mean Cost $ per PU**	**Av. Area each PU (ha)**	**Mean frequency of PUs in solutions**± SE	**No. Irreplace-able PUs**
1	0.1	No Connectivity	111.951	60.93	51.02	831	64,778	224,801	130.08	102±6.6	169
2	0.1	BLM	112.350	60.46	51.88	827	65,381	227,430	132.35	101±6.5	168
3	0.1	Euclidean	116.652	63.19	53.45	766	67,442	269,405	155.76	87±6.1	168
4	0.1	Landscape Value	112.112	60.63	51.47	842	64,922	220,260	127.55	104±6.6	169
5	0.3	No Connectivity	218.289	109.70	108.59	1285	122,983	229,295	129.18	196±7.7	211
6	0.3	BLM	218.771	111.32	107.45	1249	122,482	238,833	133.71	193±7.5	211
7	0.3	Euclidean	226.129	113.67	112.46	1154	124,823	275,432	152.04	169±7.1	211
8	0.3	Landscape Value	221.468	109.44	112.03	1702	124,552	161,774	90.98	283±9.0	223
9	0.5	No Connectivity	373.171	189.27	183.90	1721	199,677	268,855	143.86	284±8.7	291
10	0.5	BLM	375.938	188.95	186.99	1673	198,392	280,551	148.05	282±8.6	291
11	0.5	Euclidean	436.821	214.57	222.25	2782	225,565	178,367	92.10	508±12.0	1,706
12	0.5	Landscape Value	390.321	192.53	197.79	2562	207,938	175,111	93.29	460 ± 9.6	343
13	0.7	No Connectivity	568.728	290.37	278.36	2260	291,418	295,137	151.23	409±9.7	445
14	0.7	BLM	571.652	291.31	280.34	2299	292,331	290,769	148.69	405±9.6	444
11	0.7	Euclidean	687.043	324.53	362.51	4214	342,991	177,027	88.38	804±9.6	3,301
12	0.7	Landscape Value	599.068	300.47	298.60	3461	308,421	191,518	98.60	641±8.9	561

[a]All costs are in $AUD
[b]‘PU’ stands for ‘planning unit’
[c]Average outputs across the 1,000 near-optimal solutions for each scenario are listed in Table S1

Figure 1. a) The distribution of stock routes (in black) across the state of New South Wales, Australia, with the study region for the analysis shaded in darker grey, the protected area network in darkest grey, and the black square showing the area used to draw Fig. S3, and b) an image depicting a typical stock route on the ground.

Figure 2. Characteristics of the reserve network designed for scenarios with different conservation targets (10, 30, 50 and 70%) and using four approaches to connectivity (No Connectivity, BLM, Euclidean Distance, and Landscape Value), averaged across 1000 near-optimal solutions: (a) mean area incorporated into solutions (b) mean number of planning units in solutions and (c) mean cost of solutions. Error bars depict 95% confidence intervals, though these are very small and difficult to see.

Figure 3. Characteristics of the reserve network designed for scenarios with different conservation targets (10, 30, 50 and 70%) and using four approaches to connectivity (No Connectivity, BLM, Euclidean Distance, and Landscape Value), averaged across 1000 near-optimal solutions: (a) mean area of each planning unit (b) mean cost of each planning unit and (c) mean amount spent on planning units which require protection only, compared with those that also require restoration. Error bars depict 95% confidence intervals, though these are extremely small for c).

Figure 4. Comparison of the planning units selected in the best solution for different connectivity approaches at different targets. Values range from 0% (completely different planning units were selected) to 100% (exactly the same planning units were selected).

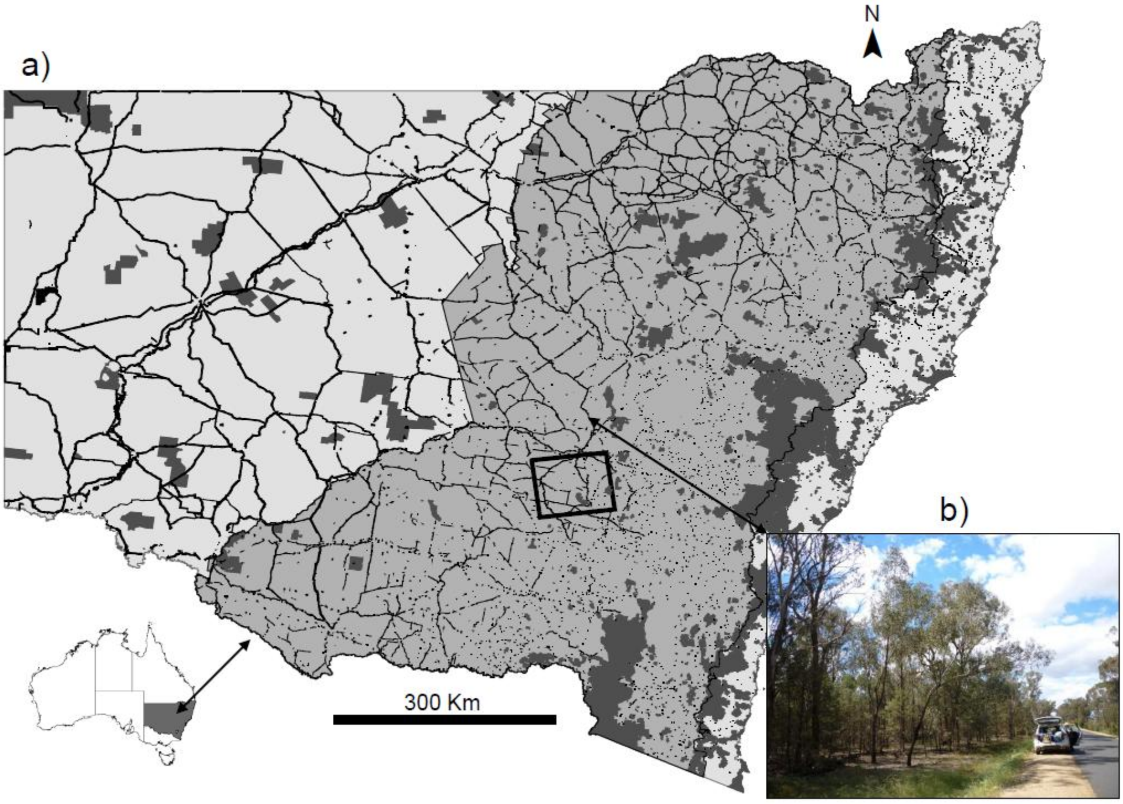
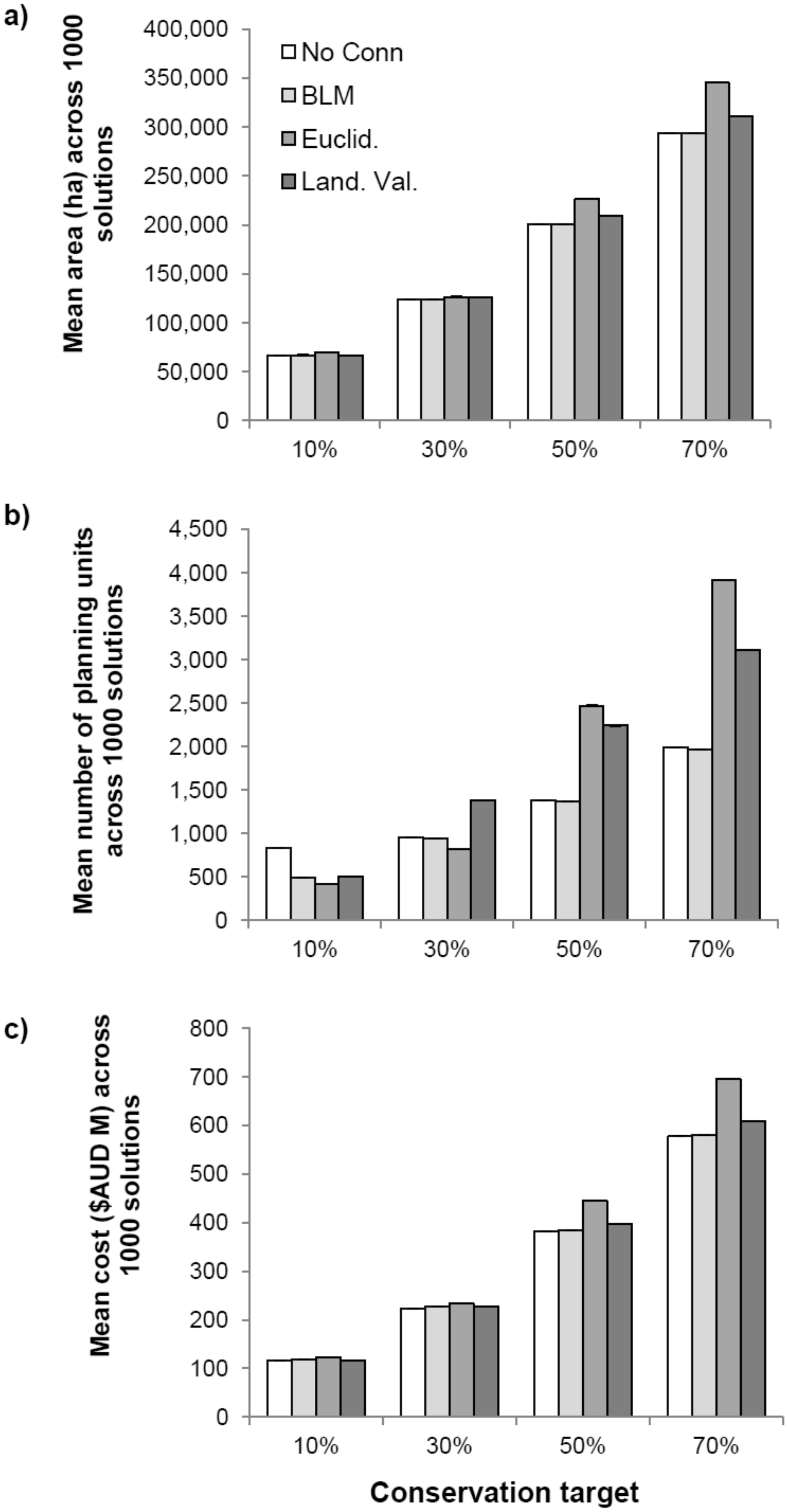


Figure 1.



a)
Mean area (ha) across 1000 solutions
400,000
350,000
300,000
250,000
200,000
150,000
100,000
50,000
0
No Conn
BLM
Euclid.
Land. Val.
10%
30%
50%
70%
b)
Mean number of planning units across 1000 solutions
4,500
4,000
3,500
3,000
2,500
2,000
1,500
1,000
500
0
10%
30%
50%
70%
c)
Mean cost ($AUD M) across 1000 solutions
800
700
600
500
400
300
200
100
0
10%
30%
50%
70%
Conservation target

Figure 2.

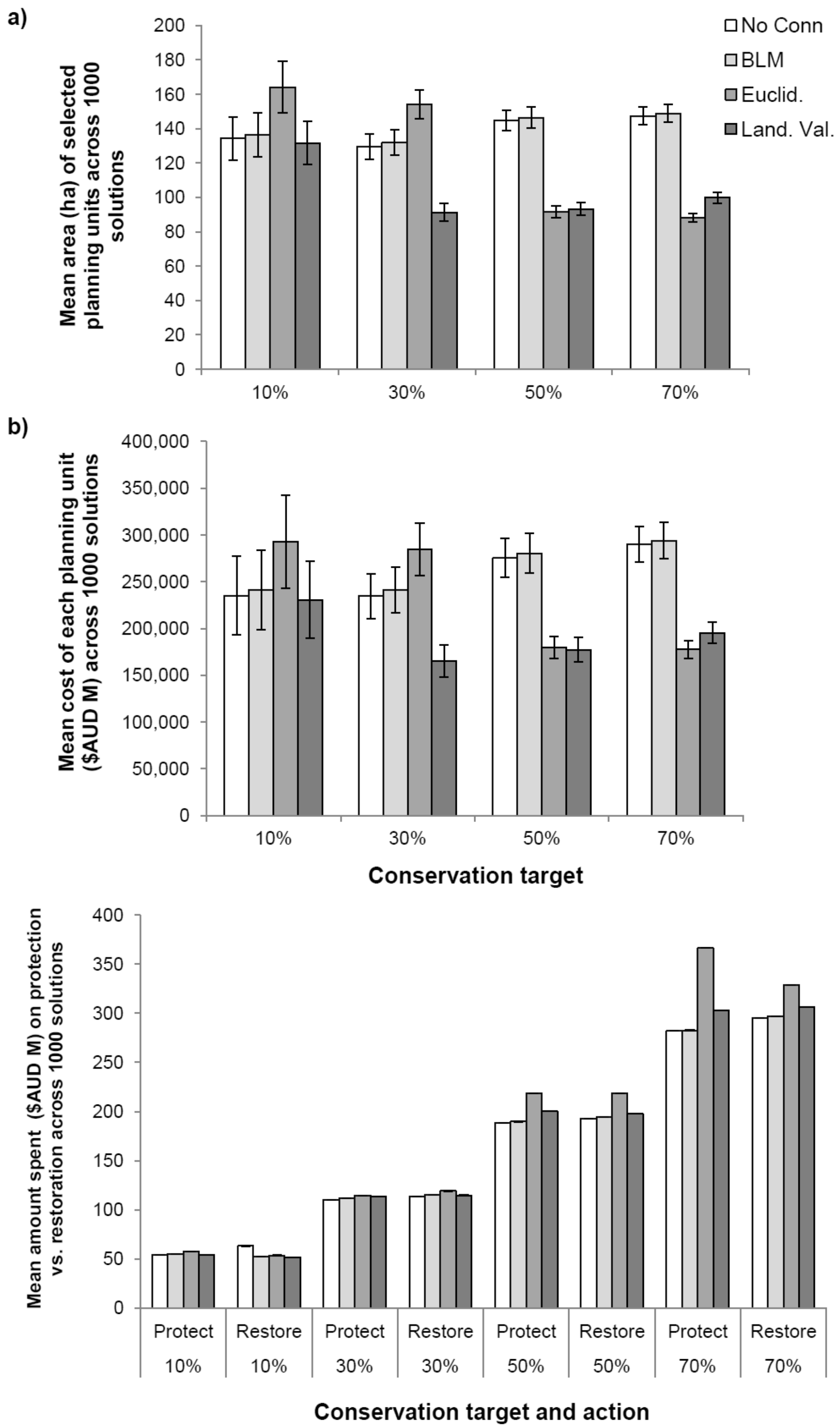
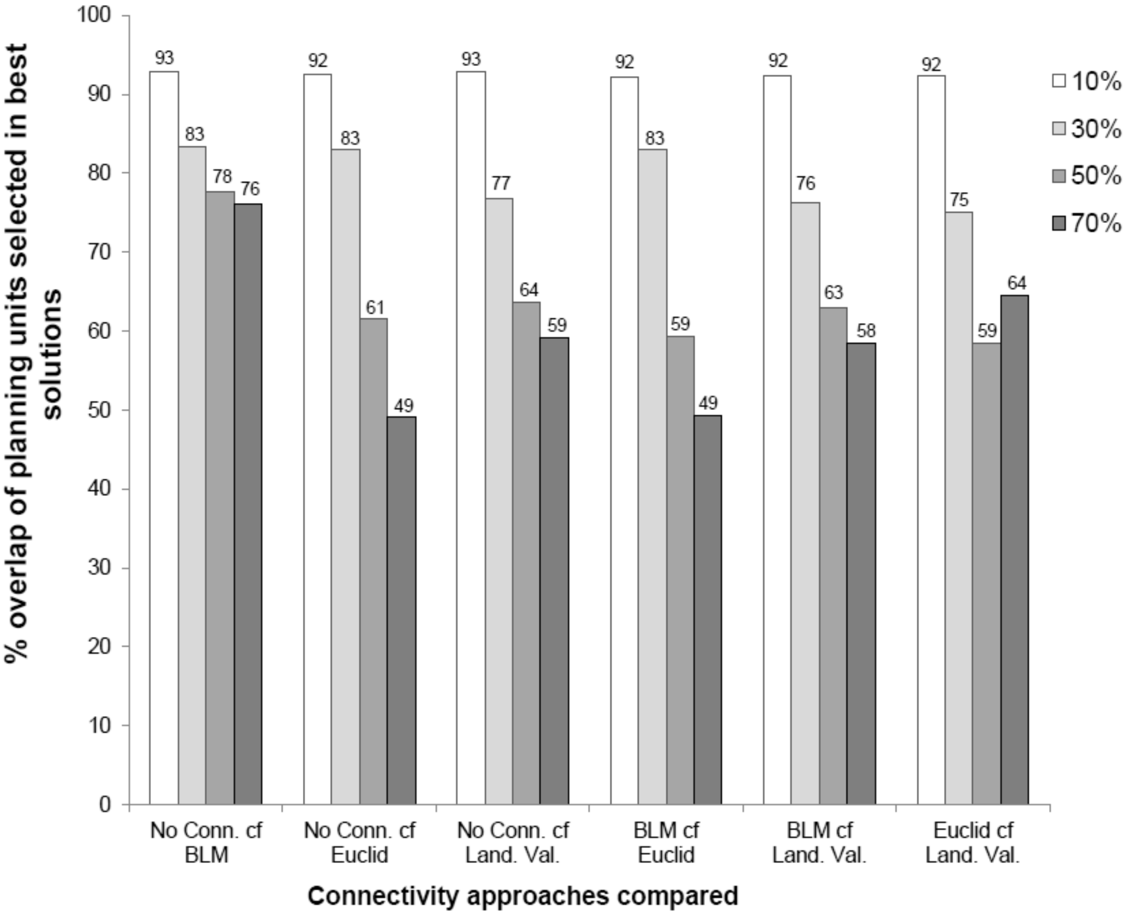

a)
Mean area (ha) of selected planning units across 1000 solutions
200
180
160
140
120
100
80
60
40
20
0
10%
30%
50%
70%
No Conn
BLM
Euclid.
Land. Val.
b)
Mean cost of each planning unit ($AUD M) across 1000 solutions
400,000
350,000
300,000
250,000
200,000
150,000
100,000
50,000
0
10%
30%
50%
70%
Conservation target
c)
Mean amount spent ($AUD M) on protection vs. restoration across 1000 solutions
400
350
300
250
200
150
100
50
0
Protect 10%
Restore 10%
Protect 30%
Restore 30%
Protect 50%
Restore 50%
Protect 70%
Restore 70%
Conservation target and action

Figure 3.



% overlap of planning units selected in best solutions
100
90
80
70
60
50
40
30
20
10
0
93
83
78
76
92
83
61
49
93
77
64
59
92
83
59
49
92
76
63
58
92
75
59
64
10%
30%
50%
70%
No Conn. cf BLM
No Conn. cf Euclid
No Conn. cf Land. Val.
BLM cf Euclid
BLM cf Land. Val.
Euclid cf Land. Val.
Connectivity approaches compared

Figure 4.

Supplementary material

Appendices

Appendix S1 - Conservation features and targets

The objective of our planning was to conserve pre-clearing vegetation communities in the stock routes across New South Wales. To determine which major vegetation classes occurred across our region, we used the "Australia - Estimated Pre-1750 Major Vegetation Groups - NVIS Stage 1, Version 3.0" spatial layer created by the Australian Government's National Land and Water Resources Audit, 2002. Based on this, 21 vegetation classes which historically occurred were identified.

To ensure that the final reserve network captured more regional heterogeneity in vegetation across the study region, we stratified each vegetation class according to the landscape type that they fell within. "Landscape types" were classified using the "Mitchell Landscape V3" layer (Eco Logical Australia 2008). There are 359 landscape types across the study region, classified according to geomorphological features, and each type is intended to represent a separate ecosystem. The vegetation class polygons were intersected with the landscapes layer to classify each stock route according to combinations of vegetation classes and landscape types ('vegetation communities'): for example, "Mallee Woodlands and Shrublands/Goonoo Slopes Landscape" (Table 1). A total of 1,452 of these 'vegetation communities' were therefore used as conservation features.

Appendix S2 – Calculation of estimates used for restoration costs

Estimates of restoration costs (all provided in Australian dollars, AUD), are based on Greening Australia's "Whole of Paddock Rehabilitation" (WOPR) program for over-and mid- storey restoration (Streatfield et al. 2010), and the “Grassy Groundcover Restoration Project” (Gibson-Roy et al. 2010) for groundcover restoration. Because the stock route remnants are linear, and the WOPR program focuses on rehabilitating fields with planted strips, it was easy to adapt the learning’s from this program to costs for our particular problem.

We used the “Vegetation Assets, States and Transitions, vers. 2” layer (Lesslie et al. 2010) to determine the current condition of each of our planning units, and based our assumptions of restoration requirements on the classifications that accompanied this dataset:

1) Bare – Areas where native vegetation does not naturally persist
2) Residual - Native vegetation community structure, composition, and regenerative capacity intact, no significant perturbation from land use/land management practice
3) Modified - Native vegetation community structure, composition and regenerative capacity intact - perturbed by land use/land management practice
4) Transformed - Native vegetation community structure, composition and regenerative capacity significantly altered by land use/land management practice
5) Replaced – (a) Adventive. Native vegetation replacement – species alien to the locality and spontaneous in occurrence (b) Managed. Native vegetation replacement with cultivated vegetation
6) Removed – Vegetation removal

We assumed that because ‘Bare’ stock routes do not naturally support vegetation, and ‘Removed’ areas were excessively degraded, they would not contribute to meeting conservation feature targets. ‘Residual’ and ‘Modified’ areas are relatively intact, so had no additional associated restoration cost. ‘Transformed’ areas had moderate restoration requirements, such as seeding and herbicide sprays, and ‘Replaced’ areas needed more intensive restoration, such as scalping, so were more expensive still. These calculations are detailed below.

Woodland restoration

a) ‘Transformed’ woodlands = $450/ha

For these woodlands the key objective of the restoration was to re-establish dominant structural components, in the form of trees and shrubs. Direct seeding for woodland systems costs approximately $200/km, including the cost of seed and wages for the seeding operator. For each hectare, we assumed the 20 rows spaced 5 m apart would be required, which equates to 2 km of seeding. Therefore, seeding alone would cost $400/ha. An additional $50/ha was added to cover the cost of preparing the site for restoration, which includes ripping up seeding lines, and an initial herbicide spray to inhibit weeds which may compete with seedlings.

b) ‘Replaced’ woodlands = $1,950/ha

Because the ‘Replaced’ state implies a high level of exotic groundcover invasion, we assumed that additional weed removal measures would be required to restore these areas. On top of the $450/ha calculated for ‘Transformed’ sites (see above), a further $1,000/ha is required for site preparation, in the form of slashing and/or manual weed removal. $500/ha was then added for 25 rows of groundcover species seeding.

Forest restoration

a) ‘Transformed’ forest = $550/ha

Restoration of ‘Transformed’ forest involved the same costs as the woodland, with the exception that the final plot should have denser tree cover. Therefore, we budgeted for 25 rows of over- and mid-storey species spaced 4 m apart ($500/ha), plus the $50/ha site preparation cost.

b) 'Replaced' forest = $2,050/ha

As with 'Replaced' woodland, we came to this estimate by adding $1,500/ha for ground-layer restoration in addition to the $550/ha required for over- and mid-storey seeding.

Grassland restoration

a) 'Transformed' grassland = $1,500/ha

This estimate is the figure that was added to the 'Replaced' woodland and forest sites, and includes $1,000/ha for site preparation and weed removal, and $500/ha for the seeding of 25 rows of groundcover species spaced 4 m apart.

b) 'Replaced' grassland = $4,000/ha

Restoration of highly degraded grassland is difficult and expensive. The only effective and practical means of reversing advanced weed invasion is to 'scalp' the site, which involves the removal of all above ground biomass, as well as the topmost layer of soil to prevent weed reestablishment from the seed bank. A modest estimate of the cost of this scalping process is $3,000/ha. Because newly-scalped sites are completely devoid of propagules, we budgeted for subsequent dense seeding of native grass species – 50 rows spaced 2 m apart, costing $1,000/ha.

Appendix S3 – Calibration of BLM values

We used the BLM calibration tool in Zonae Cogito to determine a BLM multiplier which was of a magnitude which would encourage clumping, but not blow out the cost of solutions. We tested the 30% target scenario with 20 BLM values, ranging from 0-100, and plotted the outputs to see at what value we could get the shortest total boundary at the lowest increase in cost. This process was repeated and refined with another 20 BLM values ranging from 0-5 (although the boundary did not vary greatly within this range, see results and discussion). From this calibration, we chose to use a BLM of 2.0 for all scenarios.

For the Euclidean Distance scenarios, the same BLM calibration technique as described above was used. We first tested 20 BLM values ranging from 0-100, then another 20 ranging from 0-5, and then a final 20 ranging from 0-0.05. Based on this, a BLM of 0.02 was employed for these scenarios.

Appendix S4 - Calculation of Landscape Value

The "Landscape Value" (LV) metric has been developed by NSW Office of Environment and Heritage (Drielsma et al. 2007a; Drielsma et al. 2007b). LV mapping aims to highlight 'linking' areas, where conservation of existing vegetation, condition improvement of degraded vegetation, or rehabilitation of cleared areas are most likely to contribute to maintaining or enhancing functional connectivity across a region. LV was derived, using a graph theoretical approach (Urban & Keitt 2001), by systematically modelling habitat linkages across a broad range of ecological scales, from local (i.e. connections that affect the day-to-day movements of fauna with limited dispersal ability, such as small birds and reptiles), to regional (i.e. dispersal and/or migration across tens or hundreds of kilometres). The connectivity measures employed were Colonisation Potential and Neighbourhood Habitat Area (Hanski 1999). The process was repeated for three broad vegetation structures: open forest, closed forest, and woodland/grassland.

In this way, the LV of each planning unit was calculated for taxa dependent on open forest, for taxa dependent on closed forest, and for taxa dependent on woodland/grassland, and we allocated each planning unit the LV which conformed to the vegetation it supported (Table 1). So if a planning unit supported “Eucalypt open forest” it was allocated the open forest (OF) Landscape Value score. LVs ranged from 0.0-844.25 (μ= 133.83 ± 1.08 SE). To incorporate these into the Marxan analyses, “Landscape Value” was treated as a separate conservation feature, and was set the target appropriate for each scenario (i.e. if the conservation target for vegetation/landscape classes was 30%, then the target for Landscape Value was also 30%). It should be noted that because LV is dependent on not only the characteristics of each stock route, but also the surrounding landscape, the value of each planning unit may change in time if, for example, clearing of native vegetation takes place nearby.

References for Appendices

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Supplementary Tables and Figures

Table S1. Restoration cost estimates used to calculate the total cost of restoring the vegetation within each planning unit (see Appendix S2 in the Supplementary material for further detail).

Vegetation type	Cost/ha for 'Residual'/ 'Modified'	Cost/ha for 'Transformed'	Cost/ha for 'Replaced'
Eucalypt woodlands	0	450	1,950
Eucalypt open forests	0	550	2,050
Tussock grasslands	0	1,500	4,000
Eucalypt open woodlands	0	450	1,950
Callitris forests and woodlands	0	450	1,950
Eucalypt tall open forests	0	550	2,050
Other grasslands, herblands, sedgelands and rushlands	0	1,500	4,000
Casuarina forests and woodlands	0	450	1,950
Mallee woodlands and shrublands	0	450	1,950
Acacia forests and woodlands	0	450	1,950
Other shrublands	0	450	1,950
Heathlands	0	450	1,950
Rainforests and vine thickets	0	600	3,000
Chenopod shrublands, samphire shrublands and forblands	0	1,500	4,000
Naturally bare - sand, rock, claypan, mudflat	0	0	0
Eucalypt low open forests	0	550	2,050
Acacia shrublands	0	450	1,950
Inland aquatic - freshwater, salt lakes, lagoons	0	0	0
Acacia open woodlands	0	450	1,950
Other forests and woodlands	0	450	1,950
Unknown/no data	0	0	0

* Costs are listed as $AUD

Table S2. Outline of each of the scenarios run in Marxan, and the associated outcomes across 1,000 near optimal-solutions (mean ± standard error): for all of the outcomes listed, protected area planning units are not included.

INPUTS					OUTPUTS Averages across 1000 solutions						
Scenario	**Target**	**Connectivity measure**	**SPF**	**BLM**	**Av. total area (ha)**	**$M spent purchasing**	**$M spent restoring**	**Cost solution $M**	**No. PUs**	**Cost ($M) per PU**	**Area each PU (ha)**
1	0.1	No Connectivity	70	0	66,693±810	53.8± 0.029	62.9±0.036	116.8±0.054	830±12	235,228±21,068	134.2±12.7
2	0.1	BLM	70	2	66,989±655	54.6± 0.031	63.9±0.039	118.5±0.059	491±12	241,368±21333	136.4±12.8
3	0.1	Euclidean	70	0.02	69,242±934	57.0± 0.035	66.5± 0.042	123.6±0.063	422 ±9	293,173±24,879	164.2±14.9
4	0.1	Landscape Value	65	0	66,720±808	53.8± 0.03	62.9± 0.036	116.8±0.054	507 ±7	230,598±20,650	131.7±12.4
5	0.3	No Connectivity	100	0	123,688±845	110.2± 0.045	113.6± 0.052	223.9±0.074	955±17	234,573±12,121	129.6±7.2
6	0.3	BLM	100	2	124,039±873	111.4± 0.048	115.4± 0.055	226.8±0.079	941±18	241,079±12,316	131.8±7.3
7	0.3	Euclidean	100	0.02	126,507±1,021	114.4± 0.052	118.9± 0.061	233.4±0.09	820±16	284,597±14,194	154.2±8.4
8	0.3	Landscape Value	75	0	125,828±860	113.1± 0.046	114.5± 0.053	227.6±0.075	1378±10	165,210±8,386	91.3± 5
9	0.5	No Connectivity	300	0	200,231±905	188.4± 0.047	192.8± 0.065	381.3±0.081	1383±19	275,620±10,572	144.7± 6
10	0.5	BLM	300	2	200,558±924	189.5±0.048	194.57±0.07	384.1±0.088	1370±18	280,422±10,719	146.4± 6
11	0.5	Euclidean	300	0.02	226,302±1029	218.4±0.065	218.4±0.065	444.3± 0.083	2471±25	179,830±5,927	91.6±3.3
12	0.5	Landscape Value	275	0	208,852±988	200.0±0.048	197.4±0.068	397.45±0.086	2239±12	177,537±6,472	93.3±3.7
13	0.7	No Connectivity	400	0	293,370±917	281.8±0.042	295.3±0.066	577.2±0.082	1989±19	290,169±9,562	147.5±5.1
14	0.7	BLM	16	2	293,482±886	282.±0.043	296.6±0.065	579.2±0.079	1971±20	293,932±9,689	148.9±5.1
11	0.7	Euclidean	400	0.02	345,280±972	366.2±0.042	328.4±0.062	694.5±0.079	3909±12	177,666±4,702	88.3±2.5
12	0.7	Landscape Value	300	0	310,716±1,366	303.1±0.053	306.2±0.082	609.2±0.102	3115±12	195,562±5,764	99.7±3.1

[a]'BLM' is the Boundary Length Modifier
[b]'SPF' the Species Penalty Factor, used to ensure targets are met
[c]All costs are in $AUD
[d]'PU' stands for 'planning unit'

Figure S1. The mean area (ha) of planning units either included in (“IN”) or excluded from (“OUT”) the best solutions for each of the scenarios. Error bars depict 95% confidence intervals

Figure S2. Flexibility of solutions for scenarios with different conservation targets (10, 30, 50 and 70%) and using four approaches to connectivity (No Connectivity, BLM, Euclidean, and Landscape Value): a) total number of “irreplaceable” planning units (PUs), which were included in all of the 1,000 near-optimal solutions and b) mean frequency of planning units in solutions – higher values indicate planning units had to be included more often, making solutions less flexible.

Figure S3. Examples of changes in the reserve network, based on different connectivity metrics and conservation targets - areas included in the solution are shown in black, and protected areas locked into the solution are shown in green. Because the No Connectivity and BLM outputs were so similar, BLM scenarios have not been presented here. The area of NSW which is mapped is shown in Fig. 1.

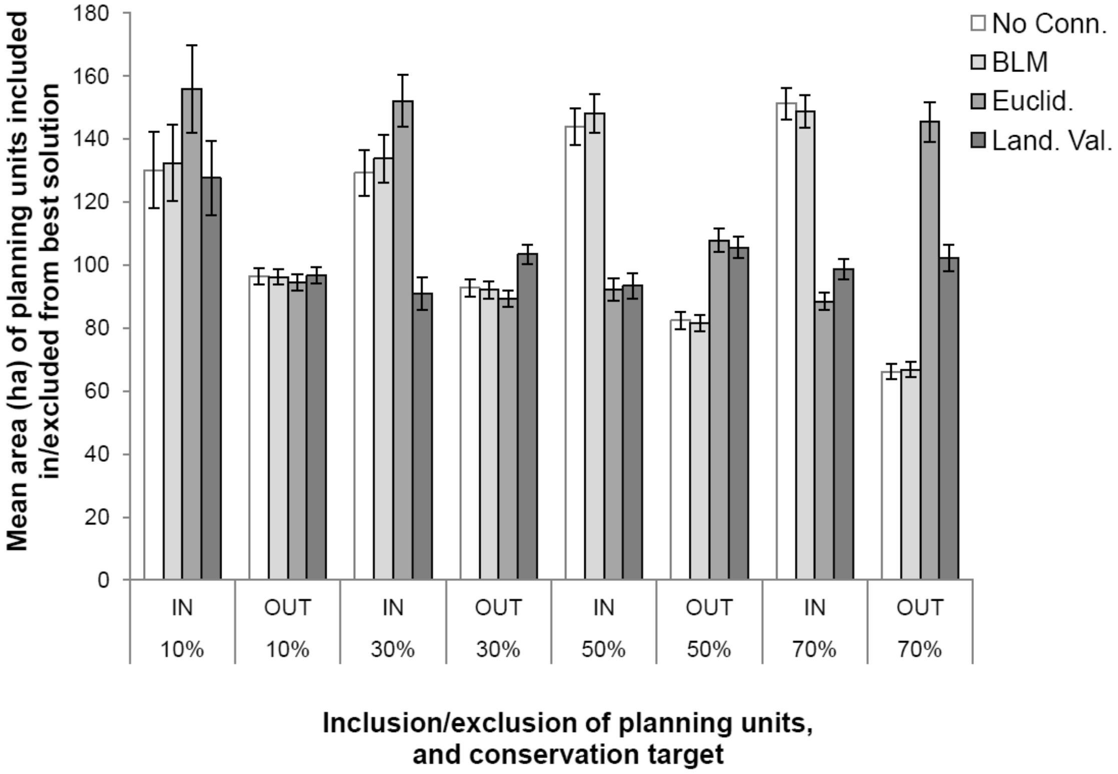
No Conn.
BLM
Euclid.
Land. Val.
Mean area (ha) of planning units included in/excluded from best solution
180
160
140
120
100
80
60
40
20
0
IN 10%
OUT 10%
IN 30%
OUT 30%
IN 50%
OUT 50%
IN 70%
OUT 70%
Inclusion/exclusion of planning units, and conservation target

Figure S1.

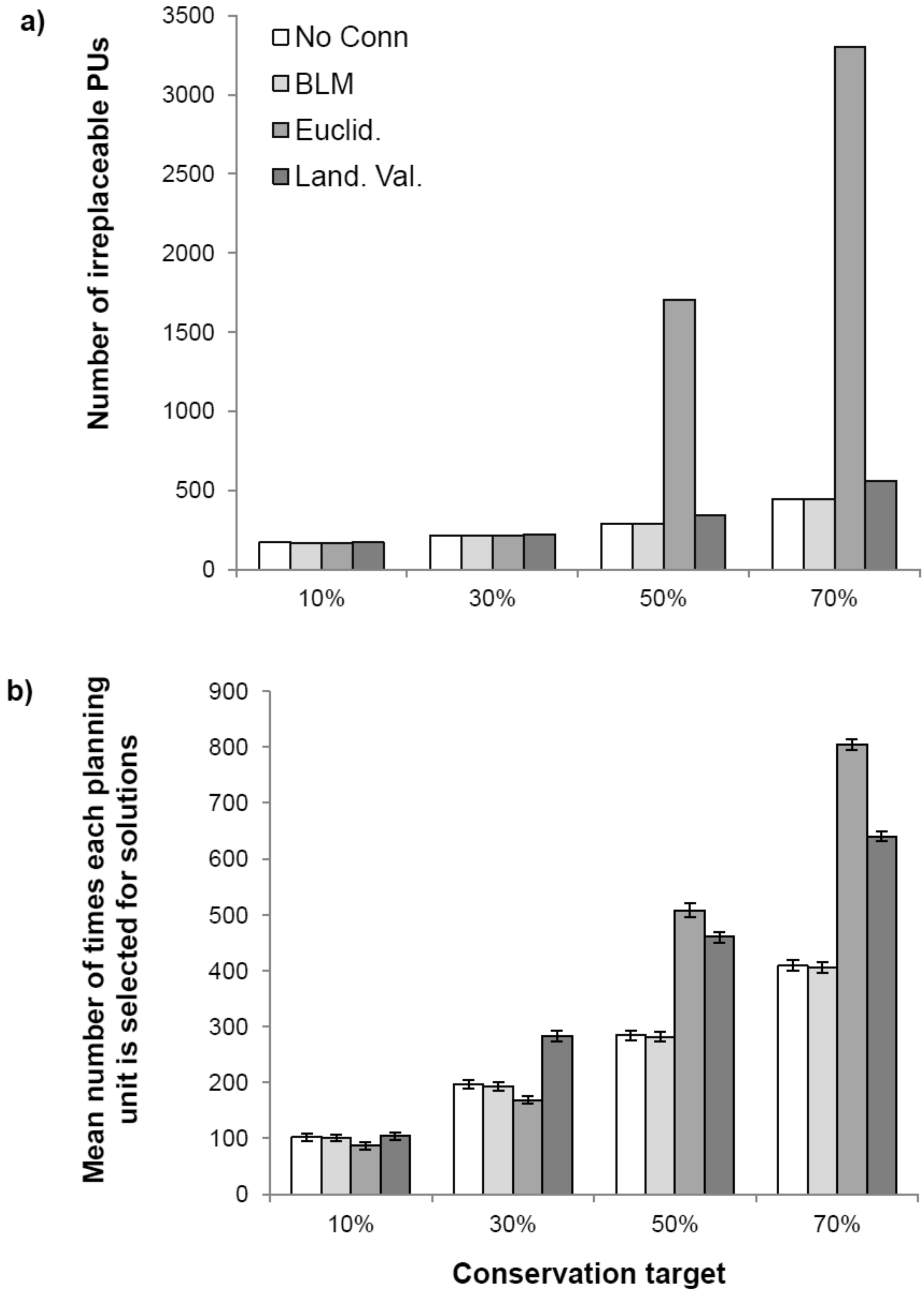
a)
Number of irreplaceable PUs
3500
3000
2500
2000
1500
1000
500
0
No Conn
BLM
Euclid.
Land. Val.
10%
30%
50%
70%
b)
Mean number of times each planning unit is selected for solutions
900
800
700
600
500
400
300
200
100
0
10%
30%
50%
70%
Conservation target

Figure S2.

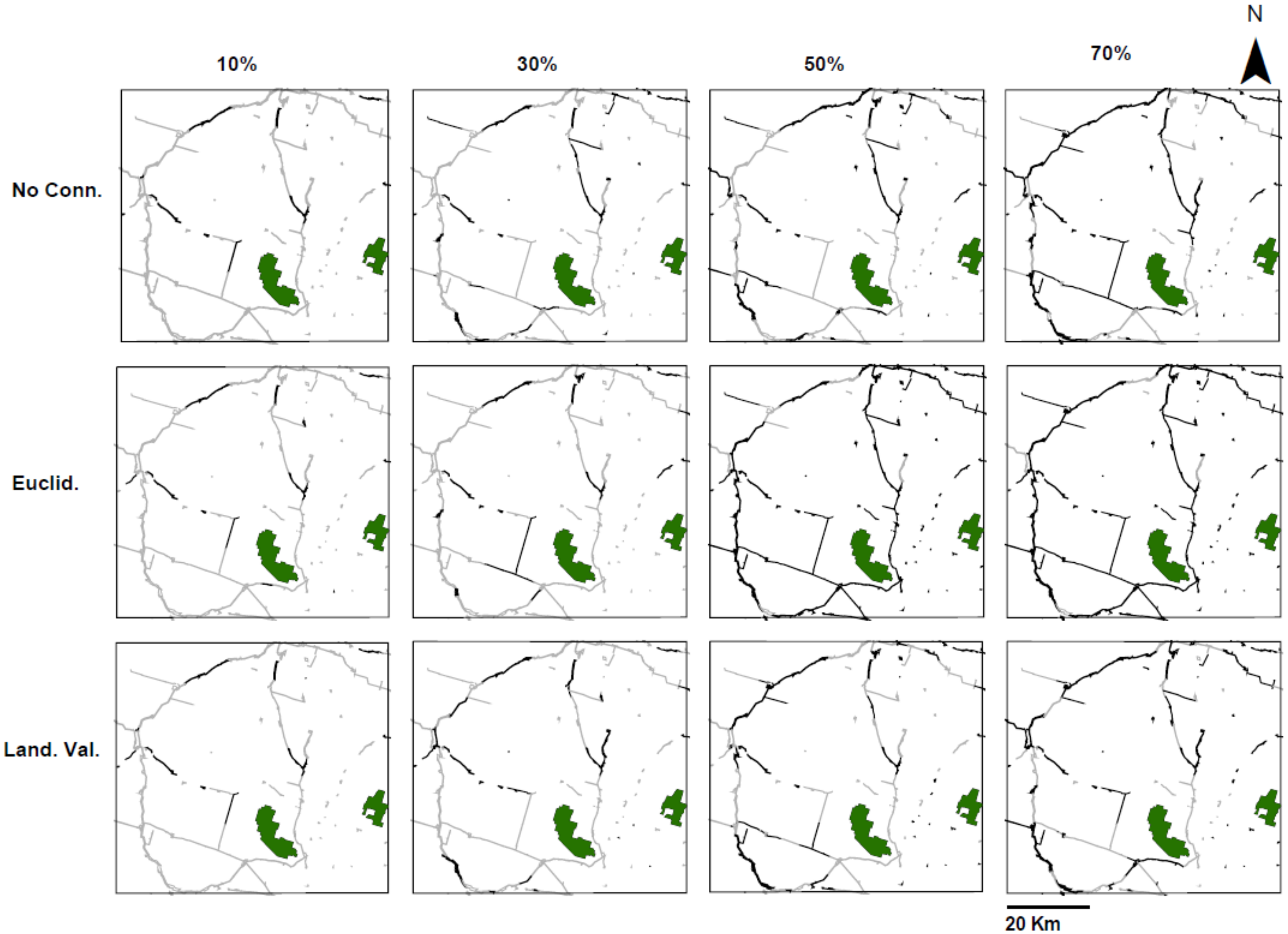

Figure S3

Appendices. Non-refereed material

The following material was produced during the course of the project, with a view to communicating the importance of stock routes to managers and the general public who may not otherwise be able to access the information.

The author, excited at experiencing her first interaction with real, live stock on a stock route. "Budgery" TSR, in Thuddungra NSW. Image: M. Salton.

A. Lentini, P. E. (2011) Managing travelling stock routes and private property for the persistence of woodland birds across the SW slopes. *Woodland Wanderings* **8**, 1-3.

B. Lentini, P. E. (2012) A road to nowhere? What next for the Stock Route network? *Decision Point* **59**, 4-5.

A.

Woodland Wanderings

Newsletter of the Grassy Box Woodlands Conservation Management Network

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Managing Travelling Stock Routes and Private Property for the Persistence of Woodland Birds across the SW Slopes

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Communities IN LANDSCAPES

The travelling stock route (TSR) network of New South Wales is a large-scale system of vegetation corridors which criss-cross some of the country's most extensively cleared and intensively managed agricultural regions. Gazetted early in Australia's pioneering history, it allowed for the movement of livestock prior to the advent of truck and railway transport. Suggested changes to the management of the TSR system, which may result in the loss of sections to freehold tenure, warrant the assessment of its value for biodiversity conservation, and the potential flow-on benefits it provides to surrounding farmland. Conservation planning decisions will need to be made regarding which sections of the network to sell or retain, so it is important that we have a clear understanding of what types of TSRs have highest biodiversity value, in order to inform these decisions.

Field surveys being conducted in Causes North TSR, approximately 30km west of Grenfell. Photo: Pia Lentini.

In response to these issues, 24 properties and 32 travelling stock routes (TSRs) across the south-western and central slopes were surveyed for birds as well as vegetation attributes in the spring of 2009. The study region covered 1,400 ha, stretching between Forbes in the north to Cootamundra in the south. Sites incorporated the full spectrum of vegetation conditions and types found in stock routes across the region, and included routes ranging from 38 to 570m wide. The survey paddocks also represented a range of land use types, including native and exotic pastures, lucerne and clover, and crops of wheat or canola. These surveys form part of a larger Australian National University PhD project. Additional surveys have been conducted which focussed on insectivorous microbats, which are likely act as important biological controls against crop pests, and native bees, which provide pollination services to nectar-bearing crops such as canola and lucerne. It is hoped that the outcomes of these surveys will be published in the near future.

Native pastures such as this form important supplementary habitat for woodland birds in the wheat-sheep belt. Photo: Pia Lentini.

Persistence of Woodland Birds

Although all birds were recorded in the surveys, the final publication focuses on the woodland-dependent species. Woodland birds are of particular conservation concern in south-eastern Australia, having experienced population declines for several decades as a result of habitat loss, invasive species, and changes to land use. Iconic woodland birds across the region attract tourism, and provide pollination and biological pest control services to private land. Because they are widespread and conspicuous, woodland birds have also helped to stimulate community interest in conservation issues, making them a particularly useful focal group for research.

The Grey-crowned Babbler, which is listed as vulnerable in NSW, was found only in travelling stock routes or paddocks containing native pasture. Image courtesy of Dejan Stojanovic.

Key lessons deduced from the woodland bird study were:

1. It is very important to maintain structural complexity in the stock routes to ensure a diversity of woodland bird species. This includes increasing the amount of logs and leaf litter on the ground, cover of shrubs, and the number of large trees. In particular, trees with peeling bark are important, particularly for insectivorous species.
2. When it comes to enhancing woodland bird communities, the effect of TSR width, or 'size' is secondary to structural complexity. This means that conservation efforts should concentrate on smaller, better quality TSRs, or improving structural complexity of the vegetation already present. However, some woodland species known to occur in the region were not found in either the TSR or paddock surveys, so are likely to be persisting in only the largest remnant patches in the landscape. For this reason, the continued protection of contiguous vegetation in the national reserve system is very important.
3. Native pastures form an important source of supplementary habitat for woodland birds. Two species of conservation concern in NSW, the Grey-crowned Babbler (*Pomatostomus temporalis*) and the Brown Treecreeper (*Climacteris picumnus*), were only found in either travelling stock routes or native pastures. The bird communities found in native pastures also most closely resembled those found in the stock routes. This may be explained by the fact that native pastures are usually subject to less intensive management practices and lower inputs, which is in turn likely to have a positive effect on the insect communities upon which many of these birds rely.
4. The retention of scattered trees on farmland is important. There was a significant increase in the number of woodland bird species in paddocks with a higher number of scattered trees. These trees provide shelter and nesting sites for hollow-dependent species, such as the vulnerable Superb Parrot (*Polytelis swainsonii*). They also act as 'stepping stones', making otherwise cleared paddocks more permeable to species which require tree cover and protection from aerial predators.

The retention of scattered paddock trees in the landscape encourages the visitation of woodland birds to properties. Photo: Pia Lentini.

5. Narrow stock routes may support a lower diversity of woodland birds, however these narrow routes appear to act as a source of bird visitors to farmland. Specifically, paddocks located adjacent to narrow stock routes supported a higher diversity of woodland bird species than those next to the widest routes. This is likely to be caused by the fact that birds in narrow routes will 'spill over' to adjacent farmland in search of additional feeding and nesting resources. As mentioned above,

native pastures with plenty of scattered of trees are likely to provide the best 'spill over' habitat for woodland birds.

The above findings and recommendations for woodland bird conservation in the TSR network will be compared to those deduced from the microbat and native bee surveys, to see how the three taxonomic groups respond differently to local habitat and landscape factors. The outcomes of this project, due to be completed in early 2012, will therefore be guidelines as to how to manage the TSR network and landscape to maximise total diversity of these three beneficial groups, and which TSRs specifically should receive highest priority for protection.

Acknowledgements: This study would not have been made possible without the support of the land holders who granted access to their properties, individuals who generously volunteered their time to assist with field work, and staff from both OEH and the Lachlan CMA who provided valuable information. The project received financial support from the Paddy Pallin Foundation, the Wilderness Society's WildCountry Science Council, and an Australian Postgraduate award and CSIRO top-up scholarship to the author.

Further reading:

Lentini, P. E., Fischer, J., Gibbons, P., Hanspach, J., and Martin, T. G. (in press) Value of large-scale linear networks for bird conservation: a case study from Travelling Stock Routes, Australia. **Agriculture, Ecosystems and Environment.** doi:10.1016/j.agee.2011.03.008

Atlas of Living Australia http://www.ala.org.au/

Have you always wondered what wildlife might be living on you property or in your region? Then why not explore the two great resources mentioned in this newsletter, the Atlas of Living Australia and the Atlas of Wildlife NSW. These tools offer you not only some useful wildlife information but also give you an opportunity to contribute to all our collective knowledge of Australias natural heritage.

B.

A road to nowhere?

What next for the Stock Route network?

By Pia Lentini (EDG, Australian National University)

Eastern Australia stands to possibly lose one of its greatest environmental and heritage assets, and many of us are not even aware of it.

Stock Routes and Reserves have been a feature of the Australian landscape since the mid-1800's, and are now most prominent throughout New South Wales and Queensland. For those not familiar with them, they basically form a large-scale network of linear connected roadside remnant vegetation. Often they are wider that your usual roadside reserve – over a kilometre, in some cases. Stock routes were established to provide corridors for livestock as they were walked 'on the hoof' between properties, complete with watering points, forage, shade, and shelter. During the expansion of agriculture following European settlement in Australia, the vegetation within the stock routes was allowed to remain standing whilst vast tracts around them were cleared.

More than a long paddock

Although the network of stock routes is often referred to as 'The Long Paddock', which superficially seems to fit, there are a few of very important differences between them and the average paddock. Aside from the obvious feature that they possess a greater cover of native vegetation, stock routes have never been subjected to management inputs common on agricultural lands, such as fertilisers and pesticides. These inputs, we now know, negatively impact on native fauna and the regeneration of eucalypts, which should form the canopy. The stock routes have also traditionally only been 'crash-grazed', or intensively grazed for short periods of time. We now know that this is also a more conservation-sympathetic form of pasture management.

Due to these factors, the emergent conservation, recreational and heritage values of the stock routes have, in some cases, superseded their pastoral role. And given that livestock are now usually transported in large trucks rather than on the hoof, authorities in NSW tasked with their management are no longer receiving adequate income from droving permits to cover the costs of managing this land.

Reviews of stock route management in 2008, for both NSW and Queensland, recommended big changes. In NSW, a hand-back of stock route to the state Crown Lands department was announced, which would see sections which they deemed to be of 'lesser value' sold off to private land holders. In Queensland, it was proposed that sections of stock routes that no longer supported high droving activity would be put under 'annual grazing agreements', negating all the value these remnants had accumulated from only being crash-grazed.

> "It would appear that the interim efforts to raise awareness, provide evidence, and suggest alternative options have been in vain."

Assembling the evidence

Following the first announcement that the stock routes might be lost from the public land system back in 2009, we commenced a project aimed at demonstrating their conservation values, with a focus on the network of stock routes in New South Wales. We collected all the information, data and literature we could find, and synthesised it to provide clear evidence of how stock routes benefit not only biodiversity conservation but rural communities and Australian society as a whole.

We were able to demonstrate that stock routes contain a high proportion of landscape features (namely valleys) and vegetation types currently severely underrepresented in the National Reserve System.

The project also included a large field-based component, in which we surveyed three fauna groups associated with the provision of ecosystem services: woodland birds (tourism), native bees (pollination) and microbats (pest control). Not surprisingly, stock routes often supported more diverse or abundant communities of these groups than the surrounding landscape. However, they also had a positive influence on the native fauna in surrounding agricultural land – it appeared that the habitat resources that stock routes provide allowed for the persistence of these beneficial communities, which then 'spill over' into adjacent paddocks.

Disparate stakeholders

During the course of the project, the political climate surrounding the stock routes has changed (as political climates tend to do). An outcry from both environmental and agricultural sectors temporarily spared the stock routes from the fate of 'disposal' and there have been changes in state government in New South Wales and Queensland. Successful lobbying by the Stock Routes Coalition and other groups in Queensland led to the 2011 Stock Route Network Management Bill, which represents

Matchett's TSR near Bogolong. So intact and 'natural' is this site that it is used as a reference site of pre-European veg for restoration. (Photo by Pia Lentini)

'The Driftway' TSR in Bogolong, NSW, supporting three purposes: An apiry, agistment, and conservation of nationally endangered grey box (Eucalyptus microcarpa) grassy woodland.(Photo by Pia Lentini)

a compromise between those with production and conservation interests.

There has been less success in New South Wales. In November 2011, a stock route conference was staged that brought together around 100 representatives from local, state and federal government, Catchment Management Authorities, the Livestock Health and Pest Authorities (these are the groups formally charged with managing the stock routes), Aboriginal Land Councils, Landcare, bird watching, and shooters and hunters groups, landholders, NGOs, and academics. Despite so many different positions and values being brought to the table, participants managed to agree upon five sensible key priorities for the stock route network:

1. Establish a central authority with oversight of stock routes that has stable and adequate resourcing for the task

2. Make stock route data accessible, and provide more information than is currently available, in a more coordinated manner

3. Ensure that there is representative management, that brings together the various values and interests

4. Establish educational programs to raise awareness of the wide importance of stock routes

5. Assess the economic significance of stock routes

It would appear that the interim efforts to raise awareness, provide evidence, and suggest alternative options have been in vain. In spite of both the additional science, and sensible requests from key stakeholders, the latest 2011 review of the NSW Livestock Health and Pest Authority (LHPA) system (under which stock routes fall, called the 'Ryan Review') reached the same conclusion as that of 2008 – that NSW stock routes are to be handed back to Crown Lands. Unfortunately, there's a very good chance that the intentions to sell off sections of 'lesser value' still exist.

The Ryan Review states that LHPAs can retain some stock routes where they can present a clear business case for doing so, which means management of the network may become fragmented. There is also no mention of what is to become of the stock route rangers, whose collective management experience is as unique and irreplaceable as the stock routes themselves.

The rhetoric of connectivity

A glimmer of hope may be present in the form of the recent National Wildlife Corridors Plan (NWCP). A glance of the map provided by Carina Wyborn in Decision Point #58 of existing large Australian connectivity projects reveals a gaping hole west of the Great Dividing Range. This is exactly where the stock routes come into their own.

The NWCP acknowledges the role that stock routes play in providing

The "Causers" TSR in Thuddungra, NSW, supports a large stand of white box (Eucalyptus albens). Federally listed as threatened, less than 0.01% of the original white box woodlands remain, with very little in protected areas. However, at least 803 stock routes in NSW wheat-sheep belt support these communities. (Photo by Marcus Salton)

Latest review report open to submissions

The draft report for the Ryan Review was released on the 24th of March. The key finding for the NSW Travelling Stock Routes was that "It should be an immediate priority to devolve the TSRs to the Crown as there is likely to be little benefit for ratepayers in ensuring a focus on core biosecurity issues if resources are continuing to be diverted to their management". Submissions in response to the draft report are due Wednesday 23rd May 2012. http://www.dpi.nsw.gov.au/aboutus/about/legislation-acts/review/lhpa

both cultural and ecological connectivity, and there is a strong emphasis on both collaboration amongst a variety of interest groups, and connectivity conservation not being restricted solely to conventional conservation reserves. Given this, stock routes seem to be a prime candidate for nomination as 'National Wildlife Corridors'.

But these aren't corridors that need to be built or restored, as is the case with most of the NWCP initiatives – the stock routes are already there. No extra money is needed to buy or restore them. No need to pay a farmer to manage them in a more sympathetic manner. There just needs to be an effort to not sell them, and manage them as they have been for the last 150-odd years.

Attempting to promote connectivity and healthy, functioning landscapes on one hand, while simultaneously selling off sections of an irreplaceable corridor network in some of the most fragmented areas of the country makes no sense whatever. Surely this situation will be acknowledged and rectified. Hopefully, before it's too late.

More info: Pia Lentini pia.lentini@anu.edu.au

References

Lentini PE, J Fischer, P Gibbons, DB Lindenmayer, & TG Martin (2011). Australia's Stock Route Network: 1. A review of its values, and implications for future management. *Ecological Management and Restoration* 12: 119-127.

Lentini PE, J Fischer, P Gibbons & DB Lindenmayer (2011). Australia's Stock Route Network: 2. Representation of fertile landscapes. *Ecological Management and Restoration* 12: 148-151.

A long time since the Long Paddock Statement

"At the end of August [2008] more than 450 ecologists and wildlife scientists called on the premiers of NSW and Queensland to protect the 3.2 million hectare travelling stock route (TSR) network. Why all the fuss? (After all, it's not often you get a roll call of Australia's best and brightest environmental minds standing up as one and calling for the urgent protection of a paddock.) The stock routes, also known as the Long Paddock, are an irreplaceable biodiversity treasure. They're a legacy of our grazing history, and one of the few land assets we have that enhance the landscape's capacity to cope with climate change. They provide refuge for endangered species and in many cases are the best remaining examples of native vegetation in a highly cleared landscape."

So began a story in Decision Point #22 (September 2008) on the value of the TSRs. Since then, NSW has changed premiers three times and government once, and Queensland has also recently changed government. Since then, the evidence of the conservation value of network of stock routes has been confirmed and strengthened yet the future of the network seems just as uncertain (in NSW anyway).

Have a look at the Long Paddock Scientists' Statement and consider how difficult it is to protect the non-production values of this unique and irreplaceable resource.

http://www.wilderness.org.au/files/long-paddock-scientists-statement.pdf