

Public policy for biodiversity conservation: evaluating outcomes, opportunities and risks

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BA, BSc (Hons)



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Declaration by author

This thesis contains no material that has been accepted for the award of any other degree or diploma in any university. To the best of the author's knowledge and belief it contains no material previously published or written by another person, except where reference is made in the text or chapter statements of contribution.

A handwritten signature in black ink, appearing to read 'Megan Evans', with a long horizontal stroke extending to the right.

Megan Evans

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I benefited from several professional development opportunities during my PhD candidature, including an internship at the Australian Government’s Department of Sustainability, Environment, Water, Population and Communities in 2013, and at the Australian Academy of Science in 2016. I gained teaching experience at the Fenner School, represented HDR students on the School Board for 3 years, co-organised a Gender Equity in the Sciences discussion group and a student retreat, acted on the Board of two professional societies, and was appointed on the Editorial Board of two scientific journals. No wonder I took a while to write my thesis. However, I’m firmly of the view that these experiences have made me a better researcher.

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Papers forming the thesis by compilation

This thesis is a Thesis by Compilation, as set out in ANU's *Higher degree by research - thesis by compilation and thesis by creative works procedure* (https://policies.anu.edu.au/ppl/document/ANUP_003405).

The majority of the thesis was conceived, developed, and written by the candidate Megan Evans. Chapters Two, Three, Four, Five and Seven of this thesis correspond to the following papers listed below. All collaborating authors agree to the inclusion of papers listed below, and agree to the description of their contribution to papers (where applicable).

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Deforestation in Australia: drivers, trends and policy responses

Authors:	Megan C. Evans
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Journal:	<i>Pacific Conservation Biology</i>
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Contributions:	MCE conceived and developed the idea, analyzed the data, and wrote the paper
Signed:	



Megan Evans
28th May 2017

Chapter Three

A quantitative evaluation of regulatory policies for reducing deforestation in Queensland, Australia

Authors: **Megan C. Evans**, Grace Chiu, Andrew K. Macintosh, Philip Gibbons.

Status: In preparation

Journal: *Land Use Policy*

Contributions: PG and AKM conceived the idea; MCE and GC developed the idea, GC conceived and developed the statistical framework; MCE and GC implemented the statistical framework; MCE and GC analysed the data; MCE and GC wrote the paper; PG and AKM provided editorial input.

Signed:



Grace Chiu, on behalf of collaborating authors
23th June 2017

Chapter Four

Carbon farming in agricultural landscapes: assisted natural regeneration as a viable mechanism for restoring biodiversity and ecosystem services

Authors: **Megan C. Evans**, Josie Carwardine, Rod J. Fensham, Don Butler.,
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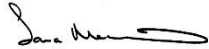
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MCE, JC, RJF, DB, KAW, HPP and TGM wrote the paper.

Signed:



Tara Martin, on behalf of collaborating authors
28th May 2017

Chapter Five

Discounting extinction: ecological time preference and biodiversity offsets

Authors: **Megan C. Evans**, Martine Maron, Philip Gibbons, Hugh P. Possingham

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Signed:



Martine Maron
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Philip Gibbons



Hugh Possingham

Chapter Seven

Opportunities and risks in the implementation of biodiversity offset policy in Australia

Authors: **Megan C. Evans**

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Journal: *Environmental Policy and Governance*

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A handwritten signature in black ink, appearing to be 'Megan Evans', with a long horizontal line extending to the right.

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28th May 2017

Publications during candidature

The following list incorporates all articles and reports published during candidature: including, but not limited to, those in the thesis.

Peer-reviewed publications

- Reside, A.E., Beher, J., Cosgrove, A.J., **Evans, M.C.**, Seabrook, L., Silcock, J.L., Wenger, A.S. & Maron, M. (2017). Ecological consequences of land-clearing and policy reform in Queensland. *Pacific Conservation Biology*.
- Lindenmayer, D.B., Crane, M., **Evans, M.C.**, Maron, M., Gibbons, P., Bekessy, S. & Blanchard, W. (2017). The anatomy of a failed offset. *Biological Conservation*, 210, Part A: 286–292.
- Evans, M.C.** (2016). Deforestation in Australia: drivers, trends and policy responses. *Pacific Conservation Biology*, 22(2): 130–150.
- Maron, M., Ives, C.D., Kujala, H., Bull, J.W., Maseyk, F.J.F., Bekessy, S., Gordon, A., Watson, J.E.M., Lentini, P.E., Gibbons, P., Possingham, H.P., Hobbs, R.J., Keith, D.A., Wintle, B.A. & **Evans, M.C.** (2016). Taming a Wicked Problem: Resolving Controversies in Biodiversity Offsetting. *BioScience*, 66 (6): 489–498.
- Evans, M.C.**, Tulloch, A.I.T., Law, E.A., Raiter, K.G., Possingham, H.P. & Wilson, K.A. (2016). Better planning outcomes requires clear consideration of costs, condition and conservation benefits, and access to the best available data: Reply to Gosper et al., 2016. *Biological Conservation*, 200: 242–243.
- Martin, N., **Evans, M.C.**, Rice, J., Lodhia, S. & Gibbons, P. (2016). Using offsets to mitigate environmental impacts of major projects: A stakeholder analysis. *Journal of Environmental Management*, 179: 58–65.
- Gibbons, P., **Evans, M.C.**, Maron, M., Gordon, A., Le Roux, D., von Hase, A., Lindenmayer, D.B. & Possingham, H.P. (2016). A Loss-Gain Calculator for Biodiversity Offsets and the Circumstances in Which No Net Loss Is Feasible. *Conservation Letters*, 9(4): 252–259.
- Evans, M.C.**, Tulloch, A.I.T., Law, E.A., Raiter, K.G., Possingham, H.P. & Wilson, K.A. (2015). Clear consideration of costs, condition and conservation benefits yields better planning outcomes. *Biological Conservation*, 191: 716–727.
- Maron, M., Bull, J.W., **Evans, M.C.** & Gordon, A. (2015). Locking in loss: Baselines of decline in Australian biodiversity offset policies. *Biological Conservation*, 192: 504–512.

-
- Evans, M.C.**, Carwardine, J., Fensham, R.J., Butler, D.W., Wilson, K.A., Possingham, H.P. & Martin, T.G. (2015). Carbon farming via assisted natural regeneration as a cost-effective mechanism for restoring biodiversity in agricultural landscapes. *Environmental Science & Policy*, 50: 114–129.
- Miller, K.L., Trezise, J.A., Kraus, S., Dripps, K., **Evans, M.C.**, Gibbons, P., Possingham, H.P. & Maron, M. (2015). The development of the Australian environmental offsets policy: from theory to practice. *Environmental Conservation*, 42(4): 306–314.
- Tulloch, V.J., Tulloch, A.I., Visconti, P., Halpern, B.S., Watson, J.E., **Evans, M.C.**, Auerbach, N.A., Barnes, M., Beger, M., Chadès, I., Giakoumi, S., McDonald-Madden, E., Murray, N.J., Ringma, J. & Possingham, H.P. (2015). Why do we map threats? Linking threat mapping with actions to make better conservation decisions. *Frontiers in Ecology and the Environment*, 13(2): 91–99.
- Tulloch, A.I.T., Tulloch, V.J.D., **Evans, M.C.** & Mills, M. (2014). The Value of Using Feasibility Models in Systematic Conservation Planning to Predict Landholder Management Uptake. *Conservation Biology*, 28(6): 1462–1473.

Edited proceedings

- Lo, A.Y., Pearson, L.J. & **Evans, M.C.** (2014). Opportunities for the Critical Decade: Enhancing well-being within Planetary Boundaries. Australia New Zealand Society for Ecological Economics 2013 Conference Proceedings. The University of Canberra and Australia New Zealand Society for Ecological Economics, Canberra, Australia.

Reports

- Evans M.C.** (2017). An evaluation of the efficacy of the Australian *Environment Protection and Biodiversity Conservation Act* (EPBC Act) 1999 Environmental Offsets Policy. Report prepared for the Department of the Environment and Energy.
- Nicol S, **Evans M**, Chadès I, Whitten S, Wolfe N, and Rough T. (2014). Developing a risk-based approach to environmental compliance and monitoring. CSIRO, Australia.
- Butler D.W., Halford J.J. and **Evans M.C.** (2014). Carbon farming and natural resource management in eastern Australia. Queensland Department of Science, Information Technology, Innovation and the Arts, Brisbane.

Popular articles

- Reside A, Cosgrove A.J., Silcock J.L., Seabrook L., **Evans M.C.** (2017) Land clearing on the rise as legal ‘thinning’ proves far from clear-cut. *The Conversation*, June 2017. URL: <http://theconversation.com/land-clearing-on-the-rise-as-legal-thinning-proves-far-from-clear-cut-79419>

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- Lindenmayer D., Maron M., **Evans M.C.**, Gibbons P. (2017). The plan to protect wildlife displaced by the Hume Highway has failed. *The Conversation*, May 2017. URL: <https://theconversation.com/the-plan-to-protect-wildlife-displaced-by-the-hume-highway-has-failed-78087>
- Evans M.C.** (2016). Australia needs better policy to end the alarming increase in land clearing. *The Conversation*, August 2016. URL: <http://theconversation.com/australia-needs-better-policy-to-end-the-alarming-increase-in-land-clearing-63507>
- Evans M.C.** (2016). Queensland moves to control land clearing: other states need to follow. *The Conversation*, May 2016. URL: <http://theconversation.com/queensland-moves-to-control-land-clearing-other-states-need-to-follow-58291>
- Evans M.C.**, Pulsford S., Foster C. (2015). Lifting the lid on unconscious bias and gender equity. *CEED News*. URL: <http://www.ceed.edu.au/ceed-news/36-news-2015/131-news-gender-equity.html>
- Evans M.C.** (2015). Science Leadership Training Not Just For Professors. *CEED Leaders Blog*. URL: <http://ceedleaders.weebly.com/blog/science-leadership-training-not-just-for-professors>
- Evans M.C.** (2015). Offsetting in the context of policy: what happens to the theory when it hits the real world? *Decision Point*, August 2015. URL: <http://decision-point.com.au/article/offsetting-in-the-context-of-policy/>
- Evans M.C.**, Renwick A., Carwardine J., Martin T. (2015). Farming carbon can be a win for wildlife, if the price is right. *The Conversation*, April 2015. URL: <http://theconversation.com/farming-carbon-can-be-a-win-for-wildlife-if-the-price-is-right-40088>
- Evans M.C.** and Maron M. (2013). Can we offset biodiversity losses? *The Conversation*, May 2013. URL: <http://theconversation.com/can-we-offset-biodiversity-losses-13805>

Conference presentations

**invited presentations*

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- Evans M.C.** (2016). Deforestation in Australia: drivers, trends and policy responses. Society for Conservation Biology - Oceania Conference, Brisbane, 6-9th July 2016. Conservation Oceania Style: Highlighting Oceania's unique approaches to conservation biology and on ground outcomes.
- Chiu G, **Evans M.C.**, Evans BJ, Macintosh AK, Gibbons P. (2016). Is no net loss possible? Evaluating policy instruments for reducing deforestation with a growing economy. The 26th Annual Conference of the International Environmetrics Society. Edinburgh, 18th-22nd July 2016
- *Evans M.C.** (2015). Applying a public policy lens to environmental decision making. 3rd Biannual Conference of the ARC Centre of Excellence for Environmental Decisions (CEED), 7-9 December, Australian National University, Canberra, Australia. *One of five ECRs invited to present in the opening research showcase - ANU representative.* [Slides](#)
- Evans M.C.**, Chiu G., Macintosh A., Gibbons P. (2015). Is no net loss possible? Evaluating policy instruments for reducing deforestation with a growing economy. Symposium: Conservation trade-offs in a resource-limited world. 27th International Congress for Conservation Biology, August 2-6 2015, Montpellier – France. [Slides](#)
- *Evans M.C.** (2014). An interdisciplinary approach to evaluating the outcomes of biodiversity offset policy. Biodiversity offsetting: challenges and solutions symposium. Ecological Society of Australia Conference, 3rd October 2014, Alice Springs, Australia.
- * Evans M.C.** (2014). An interdisciplinary approach to evaluating environmental policy: the case of biodiversity offsetting. North American Society for Conservation Biology Conference, 16th July 2014, Missoula, Montana USA.
- Evans M.C.**, Maron M., Gibbons P., Possingham H.P. (2013). A bird in the hand is worth two in the bush: ecological time preference and biodiversity offsets. 28 November, Australia New Zealand Society for Ecological Economics (ANZSEE) Conference 2013
- Evans M.C.**, Maron M., Gibbons P., Possingham H.P. (2013). A bird in the hand is worth two in the bush: ecological time preference and biodiversity offsets. 15th Annual BIOECON Conference, 18-20 September 2013, Kings College Cambridge, England.
- Evans M.C.** (2013). Is no net loss possible? Evaluating policy instruments for reducing deforestation with a growing economy. ARC Centre of Excellence for Environmental Decisions (CEED) Conference, 3-6 September 2013, Brisbane.

Abstract

The conservation of biodiversity is a daunting and complex public policy challenge. Over the past three decades, two clear themes have emerged in conservation science, policy and practice: greater experimentation with market-based policy instruments (MBIs); and an increased concern over the effectiveness of conservation policies. These two themes are interrelated, as a key driver of the rise in prominence of MBIs has been the promise of more effective, efficient and equitable conservation than that which is possible under ‘traditional’ regulatory approaches. However, scarce evidence is available on the efficacy of regulatory policies and MBIs alike, and it has been argued that “better theory, better methods, and better data” are required if conservation policies are to be more frequently and rigorously evaluated for effectiveness. This focus on the technical challenges of policy evaluation is incomplete, as effectiveness of conservation policy is influenced not only by the choice of policy instrument or combination thereof, but also the actors involved, the relevant institutional, social and political contexts, and decisions made at various stages of the policy process.

In this thesis, I investigate the challenges and complexities associated with conservation policy in Australia, an advanced and politically stable economy. Using an interdisciplinary, mixed methods approach, I consider regulatory and market-based policy responses to a major driver of biodiversity loss, deforestation, and evaluate what outcomes, opportunities and risks these policies present for conservation. In Chapter Two, I document the recent shift away from ‘command and control’ policy responses to deforestation in Australia, and towards self-regulation and MBIs. Despite this change in policy style, little is known of their efficacy. In Chapter Three, I use a spatially explicit bent-cable regression model to evaluate what effect regulatory policies have had on the rate of deforestation in Queensland, Australia. I find some evidence of a policy effect after adjusting for covariates, but extreme variation in regional deforestation trends reduces this effect at the state level. In Chapter Four, I present findings which confirm that carbon farming is economically viable in degraded Queensland agricultural landscapes under an estimated \$5 t CO₂e⁻¹ carbon price. In practice however, large-scale reforestation has not occurred despite being

the ‘rational’ option, in part due to policy complexity and political uncertainty. In the final three empirical chapters, I consider challenges in the design, implementation and evaluation of biodiversity offset policy. In Chapter Five I describe a mathematical framework used to underpin the Australian Environmental Offsets Policy, which was designed to deliver ‘no net loss’ outcomes for protected matters. I subsequently illustrate in Chapters Six and Seven that improvements to policy design do not necessarily lead to better policy outcomes, due to complexities that emerge through policy implementation in the context of multi-actor, multi-level environmental governance. I draw on qualitative data from interviews with key informants to describe potential risks to biodiversity outcomes under current offset policy settings, including: ambiguous responsibility for long term security and management, fragmentation within government departments at the federal and state levels, and a lack of transparency and public accountability. I conclude the thesis and provide future research directions in Chapter Eight.

Key words

Deforestation, evaluation, public policy, biodiversity conservation, environmental governance, market based instruments, environmental regulation, mixed methods, interdisciplinary research

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CHAPTER ONE:

Introduction

“Conservation is not rocket science; it is far more complex.”

Eddie Game et al. (2014) Conservation in a wicked complex world; challenges and solutions. *Conservation Letters* 7(3): 271–277

This thesis examines contemporary policy responses to biodiversity loss, and the challenges and complexities associated with the design, implementation, and evaluation of biodiversity policy instruments in an advanced and politically stable economy. Over the past few decades, two clear themes have emerged in the science, policy, and practice of biodiversity conservation. First, there has been increased interest in and experimentation with a diversity of policy instruments aimed at conserving and restoring biodiversity. The role and favour of state-based regulation has declined, and economic instruments and hybrid forms of governance have risen in prominence (Boisvert et al., 2013; Dovers, 2013; Gunningham and Young, 1997; Hutton et al., 2005; Lockie and Higgins, 2007; Lockwood and Davidson, 2010). Second, the effectiveness of conservation policies, programs and projects has become a key issue of concern for scientists, practitioners and policymakers. Debate and discussion about conservation effectiveness, evidence-based policy, and impact evaluation are now commonplace in ecology and conservation science journals (Adams and Sandbrook, 2013; Baylis et al., 2016; Ferraro and Pattanayak, 2006; Haddaway and Pullin, 2013; Miteva et al., 2012; Pullin and Knight, 2009; Sutherland et al., 2004, 2015).

These two themes are interrelated, as a key driver for the increased use and discussion of economic instruments in conservation has been the promise of more effective, efficient and equitable conservation than that which is possible under ‘traditional’ regulation administered by the state (Bishop et al., 2009; Kareiva and Fuller, 2016; Lockie, 2013). However, studies from the public policy, environmental politics, governance and economics literatures

(hereafter, the ‘policy sciences’¹) have long argued that institutional and political context, the interests, values, and motivations of policy actors, and the availability of resources to monitor and enforce policies will all influence policy outcomes; regardless of the choice of policy instrument type (Damiens et al., 2017; Dovers and Hussey, 2013; Gunningham and Sinclair, 1999; Hahn and Stavins, 1992; Jordan et al., 2013). Biodiversity conservation problems are frequently highly complex, and involve trade-offs between multiple objectives, values and interests. It is therefore necessary to consider a diverse range of perspectives and tools to comprehensively analyse conservation policy. This thesis draws on theory, principles and methods from the policy sciences and conservation science to investigate what opportunities and risks stem from the application of regulatory and economic policy instruments to the conservation of biodiversity.

In this chapter, I first summarise why policies have emerged to address biodiversity loss, describe the rise of diverse policy instruments and trends in their use, and then outline recent efforts to better understand the effectiveness of conservation policies. I provide a short background to key concepts and principles from the policy sciences, and discuss their relevance in the context of biodiversity conservation. I then introduce my study region of Australia: a ‘megadiverse’ nation which has experimented extensively with regulatory and market-based policy instruments over the past 30 years. Finally, I outline my research questions, methodological approach, and provide an overview of the structure of the thesis.

Conserving biodiversity in the Anthropocene

As the human enterprise shifts planet Earth into a new geological age (Crutzen, 2006; Lewis and Maslin, 2015; Steffen et al., 2007), transgresses several planetary boundaries (Rockstrom et al., 2009; Steffen et al., 2015), and drives the extinction of species at a rate that is orders of magnitude higher than what would occur without anthropogenic influence (Ceballos et al., 2015; Pimm et al., 1995, 2014; Pimm and Raven, 2000), the question of how society can

¹ The term *policy sciences* was first popularized by Lasswell to describe a “contextual, problem-oriented, multi-method” approach to understanding and analyzing public policy processes (Lasswell, 1971). However, in this thesis I use the term ‘policy sciences’ to more broadly refer to academic disciplines which employ a diversity of theories and policy analysis approaches – both quantitative and qualitative – to examine the design, implementation and evaluation of policy instruments.

effectively respond to these challenges remains daunting and complex (Colloff et al., 2017; Game et al., 2014).

Biodiversity, or biological diversity, encapsulates the diversity of ecosystems, species and genes which make up the natural world² (Groombridge and Jenkins, 2002). Human activities have long influenced the distributions and viability of species and ecosystems, beginning with the megafaunal extinctions during the Pleistocene (Burney and Flannery, 2005; Miller et al., 1999; Rule et al., 2012). However, the growth of the human economy since the industrial revolution has occurred at a pace and scale not seen at any other point in history (Steffen et al., 2011), and it is widely acknowledged that anthropogenic influence is driving the current global mass extinction event (Barnosky et al., 2011; Novacek and Cleland, 2001). New species require tens of thousands of years to evolve (Barnosky et al., 2011; Weir and Schluter, 2007), yet at least 38% of all known species are facing extinction in the near term (Vié et al., 2009), and whole ecosystems are also at risk of extinction (Keith et al., 2013; Nicholson et al., 2009).

Loss of habitat remains the principle driver of this trend (Brook et al., 2003; Evans et al., 2011; Sodhi et al., 2004), which is ultimately determined by global forces such as rising population and consumption (Lambin et al., 2001). Globally, deforestation is increasing: climbing from 2.7 million ha per annum between 1990 and 2000, to 6.3 million ha per annum between 2000 and 2005 (Lindquist et al., 2012). The most recent global assessment indicates 230 million ha of forest was lost between 2000 and 2012 (Hansen et al., 2013). Habitat loss not only contributes to species decline, but alters the provision of ecosystem services, and affects the ability of biological systems to support human needs (Millennium Ecosystem Assessment, 2005; Vitousek et al., 1997). Ecosystem services such pollination, nutrient cycling, water purification, climate regulation and pest and pathogen control are reliant on the maintenance of biodiversity (Cardinale et al., 2012; Dirzo et al., 2014; Hooper et al., 2012). Despite international efforts to slow habitat loss and protect biodiversity, to date these

² Article 2 of the Convention on Biological Diversity (1992) defines *biological diversity* as “the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystem.” <https://www.cbd.int/convention/articles/default.shtml?a=cbd-02>

efforts have not been sufficient to stem the decline (Butchart et al., 2010, 2015; Hoffmann et al., 2010).

Policy responses to biodiversity loss

Efforts to stem the decline of biodiversity and associated ecosystem services have traditionally been in the form of conserving areas of land and sea for protection in national parks and conservation areas (Margules and Pressey, 2000). The first national park in the world was Yellowstone National Park, declared by the United States Congress in 1872 (Haines, 1977). This marked the beginning of a trend in the protection of areas primarily for the preservation of the environment in a “natural state”, up until the present day where approximately 14.7% of the terrestrial and 4.12% of marine environments (UNEP-WCMC and IUCN, 2016) are designated as protected areas. The most recent commitment under the Convention for Biological Diversity called for 17% of each terrestrial biome and 10% of the marine realm to be protected by 2020 (Convention on Biological Diversity, 2011)

Over the past three decades, biodiversity conservation efforts have diversified beyond establishing protected areas, toward a larger selection of policy instruments (Barton et al., 2017; Gunningham and Young, 1997; Howlett et al., 2009; Jordan et al., 2013; Wurzel et al., 2013; Young and Gunningham, 1997). This trend has partly been driven by a recognition that effectively conserving biodiversity requires on-ground management across all landscapes and tenures (Fitzsimons, 2015; Wilson et al., 2010), as well as concerns that protection does not always equate to genuine conservation gains (Andam et al., 2008; Miteva et al., 2012; Pressey et al., 2015). The emergence of the modern environmental policy agenda in the 1970s marked the beginnings of this diversification in environmental policies (Jordan et al., 2013), on the heels of growing public concern about the state of the environment (Carson, 1962; Ehrlich, 1968; Meadows et al., 1972). This decade saw the first United Nations Conference on Environment and Development (UNCED) in 1972, the United States *Endangered Species Act* 1973 come into force, and the European Union was granted an environmental mandate for the first time, with the introduction of the first EU Environmental Action Programme (Knill and Liefferink, 2013). However, it was not until after the 1992 Convention on Biological Diversity (CBD) was adopted at the UNCED in Rio de Janeiro (the Earth Summit)

that interest and experimentation with diverse environmental policy instruments diffused into biodiversity conservation.

From government to governance: the diversification of policies for biodiversity conservation

Policies for environmental protection, including biodiversity conservation, have traditionally been administered by the state in the form of regulation (also known as ‘command and control’ policy). Regulatory instruments seek to control activities that are detrimental to biodiversity via explicit directives on how or to what extent activities should be carried out (Boisvert et al., 2013; Gunningham and Young, 1997; Pirard, 2012; Young and Gunningham, 1997). Government therefore has the primary role in creating rules, setting standards, and ensuring that industries comply with regulations by carrying out monitoring, enforcement and compliance checks.

With the rise of ‘new’ environmental policy instruments (NEPIs) from the 1970s onwards (Jordan et al., 2003b, 2005, 2013), the role of non-state actors in governing the environment has increased in prominence (Adger and Jordan, 2009; Lemos and Agrawal, 2006; Newell et al., 2012). Businesses, non-government organisations (NGOs), and community organisations increasingly operate alongside government to influence environmental actions and outcomes through the use of market-based and voluntary policy mechanisms. This transition from *government* to *governance* reflects a broader societal decline in the role of the state, and in the neoliberalisation of natural resource management and biodiversity conservation (Agrawal and Lemos, 2007; Boisvert, 2015; Lockie and Higgins, 2007; Lockwood and Davidson, 2010; Newell et al., 2012; Stavins, 2003). These changes have occurred alongside the increased importance of multi-level, adaptive, collaborative and networked forms of environmental governance (Benson et al., 2013; Folke et al., 2005; Wurzel et al., 2013; Wyborn, 2015). An understanding of how such governance systems operate is crucial for identifying how they can be strengthened for biodiversity conservation purposes (Agrawal and Lemos, 2007; Agrawal and Ostrom, 2006).

Of these NEPIs, market-based instruments (MBIs, also known as economic or incentive-based instruments) are perhaps the mostly widely discussed, studied, and divisive (Büscher,

2008; Ehrenfeld, 2008; Holmes et al., 2016; Muradian et al., 2013; Penca, 2013; Sandbrook et al., 2013; Stavins et al., 1998). The use of market-based instruments in conservation aims to facilitate biodiversity protection and management through financial self-interest, and may generate other environmental, social and economic co-benefits for firms and local communities. Examples include payments for ecosystem services (PES, Engel et al., 2008; Gómez-Baggethun et al., 2010; Muradian et al., 2013; Whitten et al., 2017; Wunder, 2007, 2013), biodiversity offsetting (Bull et al., 2013; Maron et al., 2016), individual transferable quotas (ITQs) for fisheries (Costello et al., 2008; Grafton, 1996), and carbon farming (S A Bekessy and Wintle, 2008; Evans et al., 2015).

Considerable scholarship exists within the conservation literature on the design and application of MBIs (Engel et al., 2008; Gonçalves et al., 2015; Jack et al., 2008; Pirard, 2012) and the potential economic, social and ecological opportunities delivered by such instruments (S A Bekessy and Wintle, 2008; Bishop et al., 2009; Evans et al., 2015; Gardner et al., 2013; Muradian et al., 2013; Robinson et al., 2016). MBIs for biodiversity conservation are now widely dispersed, yet like all policy interventions, there is frequently limited information on what outcomes are being delivered (Ferraro and Pattanayak, 2006; May et al., 2016; Miteva et al., 2012; Pattanayak et al., 2010). While the diversity of policy instruments applied to conserve biodiversity has increased dramatically over the past three decades, there remains considerable uncertainty about the effectiveness of conservation policies and programs.

Which conservation policies and programs are effective?

There has been substantial public and private investment in biodiversity conservation activities (McCarthy et al., 2012; Waldron et al., 2013), yet biodiversity as a whole continues to deteriorate. Now, as the world edges closer to the end of the 2011-2020 commitment period of the CBD (2010), there is increasing interest and concern over the scale and effectiveness of current policies and programs aimed to address biodiversity loss at international, national and regional scales (Butchart et al., 2015, 2016; Ferraro and Pattanayak, 2006; Miteva et al., 2012; Rands et al., 2010; Venter et al., 2014).

The Millennium Ecosystem Assessment (2005) contended that “Few well-designed empirical analyses assess even the most common biodiversity conservation measures”, and Ferraro and Pattanayak (2006) thereafter called for greater use of evaluation in conservation policy and practice. Without evidence, scientists and practitioners are said to rely on “myth, intuition and anecdote” to design and implement conservation policies and programs (Ferraro and Pattanayak, 2006; Pullin et al., 2004; Sutherland et al., 2004), which can lead to sub-optimal or even detrimental outcomes for biodiversity (Pullin and Knight, 2009). While an evidence base is building for the effectiveness of protected areas (Andam et al., 2008; Ferraro et al., 2013; Hockings et al., 2015; Joppa and Pfaff, 2011) and some on-ground management conservation actions (Sutherland et al., 2015), empirical evaluations, as a rule, continue to be rare (Curzon and Kontoleon, 2016; Ferraro and Hanauer, 2014; Miteva et al., 2012). Data scarcity, system complexity, large spatial and temporal scales, inadequate methodological techniques, and a lack of evaluation expertise amongst conservation scientists and practitioners have been cited as key barriers to evaluation in conservation (Baylis et al., 2016; Ferraro, 2009; Mickwitz, 2003). Yet policy scholars have long warned that the effectiveness of policies (and the evaluation thereof) is highly dependent on the social, political and institutional contexts in which they are developed and implemented (Dovers and Hussey, 2013; Gunningham and Young, 1997; Hahn, 1989; Hahn and Richards, 2013; Jordan et al., 2013; Primmer, 2017).

Applying a policy lens to biodiversity conservation

Conservation science and policy can benefit from research undertaken in fields such as public policy and administration, political science, environmental politics and environmental governance (Agrawal and Ostrom, 2006; Biesbroek et al., 2015; Cairney et al., 2016; Dovers, 1996; Dovers and Hussey, 2013; Jordan et al., 2013). Indeed, Dovers and Hussey (2013, p 22) have argued:

“A tendency to ignore basic policy knowledge from other areas (such as public policy, public administration, law, economics and political science) is, we believe, a weakness of much environment and sustainability policy thinking.”

In this thesis, I draw on theories and lessons from these disciplines to examine the effectiveness of policies for biodiversity conservation, and the context in which they are made. I describe a series of case studies which outline policy instruments for responding to a key driver of biodiversity loss: deforestation³. I consider regulatory and market based policy responses to deforestation in Australia, which is a useful study region for this analysis due to its richness in biodiversity, and high rate of human-induced forest loss despite the introduction of a series of policy interventions over the past 30 years. I demonstrate through these case studies how evaluation can be challenging or problematic due to various technical, governance and policy design issues, and illustrate where possible how these issues can be overcome to improve decision making. Although I recognize the continued importance of protected areas (Margules and Pressey, 2000; UNEP-WCMC and IUCN, 2016; Worboys et al., 2015) and community based conservation efforts (Borrini-Feyerabend and Hill, 2015), this thesis focusses on the design, implementation and evaluation of regulatory and market based policy instruments given these instruments have been a major contemporary focus of biodiversity policy in Australia.

The next section will provide background to key concepts from the policy sciences: namely, the policy process, policy instrument choice, and policy evaluation. I will then revisit these issues to examine how they translate to a biodiversity conservation context.

Background to the policy sciences

The policy sciences draw upon insights from multiple disciplines, including public policy and administration, law, economics, political science and governance. There are theoretical and methodological differences between the disciplines, and the choice of focus. For

³ Deforestation refers to the direct human-induced removal of forest cover whereby the land is subsequently converted to a non-forest use that is long-term or permanent (Watson et al., 2000). ‘Forest’ is defined under the UN Framework Convention on Climate Change (UNFCCC) classification system as vegetation that is at least 2 metres high, with a minimum of 20 percent canopy cover (Furby, 2002). I consider only native forest (not plantations) used for non-forestry purposes in this thesis. Although the loss of non-forest habitat such as grassland and shrubland is also a significant conservation issue, deforestation is still the dominant form of habitat loss in Australia and internationally. Further, satellite imagery of these non-forest vegetation types is still limited, rendering quantitative analysis of temporal and spatial trends difficult. Nevertheless, the policy instruments considered in this thesis – namely native vegetation regulations, carbon farming, and biodiversity offsetting – are still broadly applicable to non-forest vegetation such as grassland and shrubland.

example, economics tends to focus predominantly on the identification of a goal (problem formulation), and the means to achieve that goal (instrument choice) (Baumol and Oates, 1988; Goulder and Parry, 2008; Stavins et al., 1998) as the key components of policy making. However, scholars of political science, public policy and governance emphasise that policy making must be cognizant of the relevant social and political context, and satisfy multiple criteria, including equity and political acceptability (Dovers and Hussey, 2013; Howlett, 1991; Lasswell, 1971; Sterner, 2002).

The policy process

Public policy comprises of decisions made by individuals and organisations within government, which are influenced by actors⁴ operating within and outside government (Howlett et al. 2009). The purpose of public policy, as described by Dovers (2005, 2013), is to “change the behaviours of individuals, households, firms, communities, and governments engineering society for determined ends”. The practice of policy making is described as the *policy process*, or policy cycle (Althaus et al., 2012; Brewer and DeLeon, 1983; Bridgman and Davis, 2000; Cairney, 2012; Dovers and Hussey, 2013; Howlett et al., 2009; Lasswell, 1971; Weimer and Vining, 2010). Numerous models of the policy process exist, and vary marginally with respect to the number, order and names of the stages of the process (Howlett et al. 2009). Critics of the policy process argue that policymaking is rarely as simple and linear as the model implies (Bridgman and Davis, 2003; Everett, 2003), or even that there is no order or coherence to the process at all (as per the ‘garbage can’ model popularised by Kingdon (1984) and Cohen and colleagues (1972)). Nevertheless, the concept of a policy process serves as a useful tool by which to understand and analyse the messy, real-world complexities of policy making.

Figure 1.1 illustrates two representations of the policy process (Cairney, 2012; Howlett et al., 2009) which highlight not only its different stages, but also the participation of different policy actors within the process. *Agenda setting* (or problem-framing) is where all policy

⁴ In this thesis I refer to *policy actors* as any individual or organisation that interacts formally or informally with policy processes either directly or indirectly. *Policy actor* is used synonymously with *stakeholder*, as any individual or organisation with an interest or concern with a policy process. *Policymakers* are government actors who are directly involved in designing and implementing public policy.

actors comprehend and formulate the policy problem – that is, the problem which society wishes to solve in light of relevant social, environmental and economic goals (Dovers and Hussey; 2013; Howlett et al.; 2009). Biodiversity loss, unsustainable rates of deforestation and climate change are examples of broad policy problems. The *policy universe* refers to the broader community individuals and organisations both within and outside government (Howlett et al. 2009). Policy principles and goals are formulated by a smaller *policy subsystem*, which may be led by government actors who seek input from members of the community who are concerned with or have expertise in a policy problem. For example, the ‘no net loss’ principle of many biodiversity offset policies (Bull et al., 2016; Business and Biodiversity Offsets Programme (BBOP), 2012d; Maron et al., 2016) was informed and influenced by members of the scientific community, the business sector and regulated industries, environmental groups and broader political and societal trends.

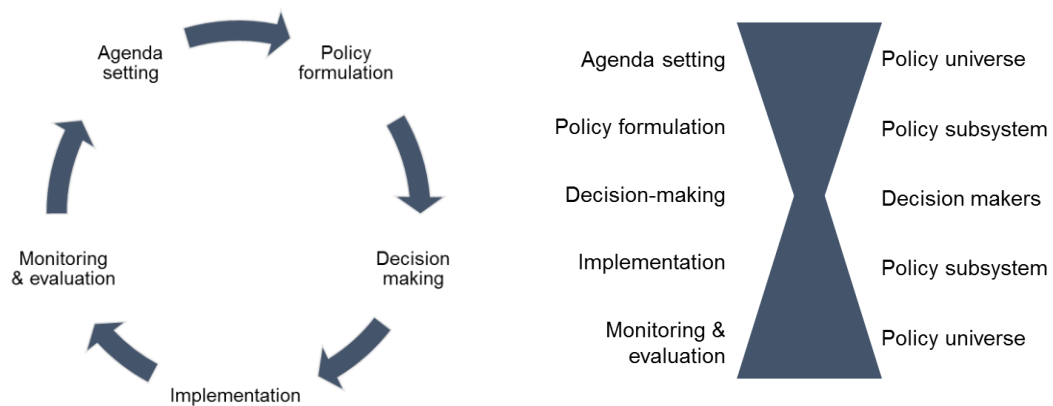


Figure 1.1. The policy cycle (left), and the policy cycle-actor hourglass (right), adapted from Cairney (2012) and Howlett et al. (2009) respectively.

Public policy decisions and the selection of policy instruments are ultimately made by a small number of government decision makers, but the implementation, monitoring and evaluation phases incorporate an increasing number of policy actors with diverse interests and motivations (Howlett et al., 2009).

Policy instrument choice

The selection and implementation of policy instruments is at the core of administering public policy (Hood, 2007; Jordan et al., 2013). Policy instruments are the ‘tools’ used by governments to implement policies and achieve policy goals, and a wide range of instruments and variations thereof exist. The selection of policy instruments should ideally be based on evidence of their relative suitability for addressing a particular issue, though ideological and political preferences also influence policy instrument choice (Dovers and Hussey, 2013; Hood, 1986). For the purpose of this thesis I focus only on a subset of instruments applied in the context of biodiversity conservation (Gunningham and Young, 1997; Miteva et al., 2012; Young et al., 1996), namely regulatory, and market-based instruments. It has been widely argued by economists (Cole and Grossman, 2002; Hahn, 1989; Stavins et al., 1998; Tietenberg, 1990), and increasingly by ecological and conservation scientists (Bishop et al., 2009; Gómez-Baggethun and Muradian, 2015; Kareiva and Fuller, 2016; Ring et al., 2010a), that economic instruments are a superior means of delivering environmental outcomes as compared to ‘command-and-control’ regulation. This assertion is informed partly by the perceived failure of regulation in controlling environmental degradation, as well as the popularization of neo-liberal economic rationalism over the last several decades.

Regulation is considered the default or ‘traditional’ type of policy instrument used for environmental protection (Gunningham and Sinclair, 1999). Regulatory instruments effectively specify both the outcome and the means to deliver it, and can be particularly well suited to situations where there is a high risk of detrimental and irreversible outcomes in the absence of policy intervention (such as in many biodiversity conservation problems, (Gunningham and Young, 1997; Perrings and Pearce, 1994). However, regulation is the least flexible form of governance, and critics argue it can be inefficient, intrusive, inequitable, and slow to respond to new information and changing conditions (Gunningham and Grabosky, 1998; Gunningham and Sinclair, 1999).

Market-based instruments have been particularly popularised in biodiversity conservation and natural resource management since the late 1990s (Hajkowicz, 2009; Lockie, 2013; Young et al., 1996). MBIs broadly aim to influence behaviour to meet policy objectives

through a ‘carrot’, as opposed to a regulatory ‘stick’ approach. Economic instruments encompass a diversity of policies, including eco-certification and labelling (termed as ‘market friction’), price-based mechanisms (such as pollution taxes), and cap-and-trade environmental markets (Dargusch and Griffiths, 2008; Lockie, 2013; Stavins, 2003). There is a tendency for MBIs to be described as an “opportunity” to deliver positive economic and social outcomes alongside biodiversity conservation instruments (Bishop et al., 2009; Kareiva and Fuller, 2016; Ring et al., 2010b), yet genuine “win-wins” are rare (Howe et al., 2014; Muradian et al., 2013; Phelps et al., 2012b) and likely only to occur where stakeholder interests align (Hahn and Hester, 1989).

Economic instruments are claimed to achieve either the same level of environmental protection at a lower cost than command and control alternatives; or deliver better environmental outcomes for the same cost (Hahn and Stavins, 1991; Hockenstein et al., 1997; Salzman and Ruhl, 2000; Stavins, 2003). However, this assertion is described as overly simplistic by scholars who argue that an understanding of the varied institutional and political contexts in which policy making occurs is fundamental (Boisvert, 2015; Hahn, 1989; Hahn and Hester, 1989; Jordan et al., 2013; Primmer, 2017), and ultimately influences the effectiveness and efficiency of any policy instrument. Indeed, Cole and Grossan (2002) contends that much of this literature “reflects a normative presumption that only such “economic” instruments have any possibility of producing an efficient outcome”. In reality, effective policy making generally requires a diversity of instruments to be implemented as part of a policy mix, while remaining cognisant of economic, political and social contexts (Dovers and Hussey, 2013; Gunningham and Sinclair, 1999; Hahn, 1989; Howlett et al., 2009; Young and Gunningham, 1997).

Policy evaluation

Evaluation is a crucial component of the policy process (Dovers and Hussey, 2013) which can give insight to the relative merits of different approaches (Bottrill and Pressey, 2012; Crabbé and Leroy, 2008; Mickwitz, 2003; Rossi et al., 2004; Vedung, 2000). Scriven (1991) provides the most general definition of evaluation, referring to it as “the process of determining the merit, worth, or value of something, or the product of that process”. Policy

instruments may be judged according to a wide range of criteria, including their cost, social or environmental outcomes, feasibility of implementation, equity implications, flexibility of use, information requirements and dependability (Dovers and Hussey, 2013; Gunningham and Young, 1997). In this thesis, I am primarily interested in the *environmental outcomes* delivered by policy responses to deforestation. I refer to *policy effectiveness* to describe whether a policy has capacity to or has delivered the environmental outcomes as per its stated objective.

In the context of the policy process, evaluation is most frequently referred to as the last “stage” (Figure 1.1), wherein the process of implementing a policy and the outcomes delivered by the intervention are retrospectively assessed with respect to the policy objective. However, “evaluation” may also be used to describe prospective analyses which aim to predict the potential opportunities, risks, and costs of an intervention during the policy formulation or design phase. I distinguish between these two broad approaches by describing the evaluation of the value or merit of a policy after it has been developed and implemented as *ex post* evaluation, and prospective policy analysis as *ex ante* evaluation (Crabbé and Leroy, 2008; Rossi et al., 2004). Since I consider both existing and emerging policy instruments for biodiversity conservation in this thesis, I draw on a range of *ex ante* and *ex post* evaluation methods.

Quantitative methods are commonly used for *ex ante* evaluation given their ability to predict the potential outcomes from policies under varying scenarios and assumptions. Examples of *ex ante* evaluation studies in the conservation literature include those which estimate the potential biodiversity benefits of protected area expansion (Venter et al., 2014; Watson et al., 2011) and reconfiguration (Fuller et al., 2010), quantify the carbon sequestration and biodiversity opportunities from forest governance interventions (Chapter Four, Carwardine et al., 2015; Evans et al., 2015), and determine the likelihood of biodiversity offset policies to achieve a ‘no net loss’ of biodiversity (Chapter Five, Bull et al., 2016; Gordon et al., 2011).

Of course, even the best designed *ex ante* evaluation cannot fully capture the realities of policy implementation, and so *ex post* evaluation is crucial for policy learning and

improvement. Both qualitative and quantitative methods may be used for *ex post* evaluation depending on its purpose, intended audience, the available data, and which component of the policy is being assessed (Margoluis et al., 2009; Rossi et al., 2004). Ferraro (2009) has argued that “the fundamental problem of evaluation” is in determining what would have occurred in the absence of the intervention i.e., the *counterfactual*. The counterfactual is frequently assumed, ignored or obfuscated in environmental policies (Maron et al., 2013; Maron et al., 2015), which makes *ex-post* evaluation extremely difficult. Establishing a counterfactual scenario is a pre-requisite for the use of experimental and quasi-experimental impact evaluation methods (Ferraro, 2009; Ferraro and Hanauer, 2014; Greenstone and Gayer, 2009; White, 2009) which aim to quantify the causal link between a policy intervention and a variable of interest. However, even in cases where it may not be possible or desirable to conduct a strict impact evaluation, counterfactual thinking can assist in the design of non-experimental and qualitative evaluations which seek to establish what effect a policy intervention has had on an outcome relative to other contributing factors (Ferraro, 2009; Margoluis et al., 2009; Rossi et al., 2004).

Recent debate in the conservation literature on the need for conservation policies and programs to be more frequently subject to *ex-post* evaluation has almost exclusively focussed on experimental and quasi-experimental designs (Margoluis et al., 2009). Some scholars have hypothesised that a lack of awareness and expertise in impact evaluation techniques is preventing more widespread knowledge of the effectiveness of conservation interventions (Baylis et al., 2016; Ferraro, 2009; Ferraro and Pattanayak, 2006; Miteva et al., 2012). However, as outlined below, this argument obscures two key issues.

First, there is an assumption that the barriers to *ex-post* policy evaluation are purely technical, and are simply a matter of “better theory, better methods, and better data” (Miteva et al., 2012). In a recent survey, Curzon and Kontoleon (2016) found that awareness of impact evaluation amongst conservation practitioners and policymakers was extremely high, but more widespread use of such evaluation methods was largely prevented by financial and time constraints.

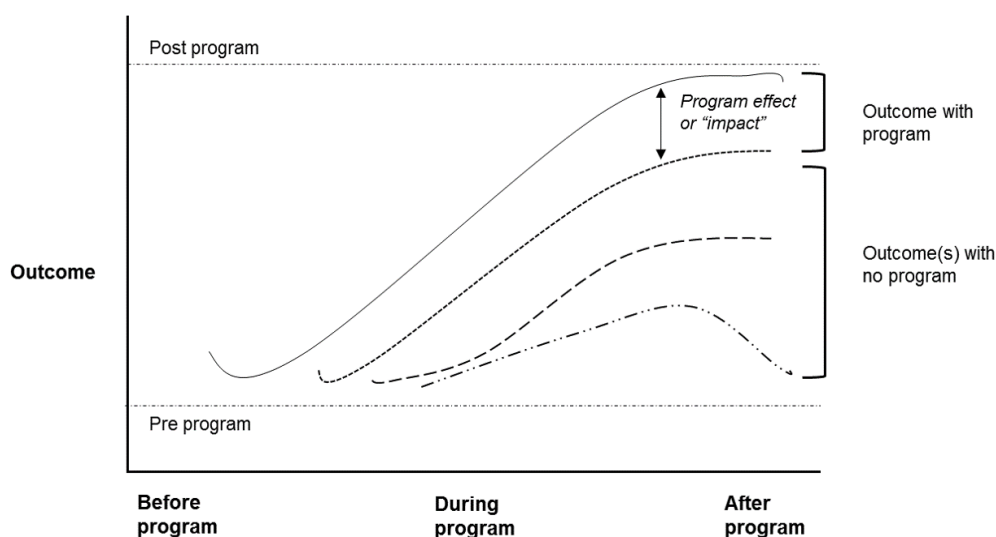


Figure 1.2 Illustration of an observed outcome under a policy or program (solid curve), and plausible counterfactual scenarios (dotted lines). The policy or program effect ('impact') is the difference between the observed outcome and the unobservable counterfactual. Adapted from Rossi et al. (2004)

Keene and Pullin (2011) acknowledged that the impediments to an 'effectiveness revolution' in conservation are not only technical, but cultural and political. Organisations that are responsible for designing and implementing conservation policies and programs, whether they be government or non-government, can be reluctant to conduct or publish evaluations in the event a 'failure' leads to political or financial repercussions (Dovers and Hussey, 2013; Keene and Pullin, 2011; Kleiman et al., 2000; Redford and Taber, 2000). Evaluation is also difficult when monitoring data is proprietary or otherwise inaccessible (Curtis et al., 1998; Lindenmayer et al., 2017).

Second, the characterisation of impact evaluation techniques as 'best practice' overlooks the value and rigor of well-designed non-experimental and qualitative evaluations. Non-experimental quantitative evaluation designs, such as those which measure a variable of interest before and after an intervention, have less capacity to infer causality but can be used in situations where the available data is simply not amenable to a quasi-experimental design (Margoluis et al., 2009). In **Chapter Three**, I use a hierarchical bent-cable regression model to quantify the effect of regulatory policies on the deforestation rate in Australia. Despite

using the most up to date spatial imagery of deforestation events across the continent, the complexity and ubiquity of policies regulating the clearing of native vegetation across all Australian jurisdictions meant that even a synthetic control impact evaluation approach (Abadie et al., 2010, 2015) was not feasible. Our results demonstrate that the bent-cable model is a promising technique for detecting changes in the rate of deforestation in response to policy interventions introduced over time, and indicate that caution should be taken to avoid any premature claims of policy effectiveness prior to conducting an *ex-post* evaluation.

While quantitative evaluation methods can estimate *what* the impact of a policy intervention is to varying degrees of confidence, they cannot easily explain *why* or *how* these effects may have occurred. Qualitative evaluation approaches allow a far deeper examination of a particular case study, which can suggest *why* a policy intervention may or may not be working (Margoluis et al., 2009; Yin, 2009). Techniques such as interviewing, surveys and focus groups can reveal crucial information on the institutional and political context in which a policy is implemented (Blaikie, 2009; Hay, 2010; Yin, 2009), and provide evidence in the form of perceptions, knowledge and behaviour that can be used to improve conservation policy and programs (Bennett, 2016; Bennett et al., 2017; Moon and Blackman, 2014; St. John et al., 2014). Particularly since conservation interventions are increasingly governed by multiple actors with varying motivations and objectives (Adger and Jordan, 2009; Lemos and Agrawal, 2006; Newell et al., 2012), qualitative methods can provide powerful insights into the whole policy process, and how these governance systems ultimately influence environmental outcomes. **Chapters Six and Seven** use qualitative methods to understand the governance ‘landscape’ of biodiversity offsetting in Australia, and the barriers and enablers to effective policy implementation. The lack of accessible data on the environmental outcomes being delivered by biodiversity offset policy in most jurisdictions worldwide (Brown et al., 2013; Lindenmayer et al., 2017; Maron et al., 2016; May et al., 2016; Pawliczek and Sullivan, 2011) makes a quantitative evaluation of policy ‘impact’ close to impossible.

Research approach

Study region

Australia is renowned not only for its diversity of endemic species and ecosystems (Mittermeier and Mittermeier, 1997), but also for its poor extinction record. For example, nearly half of all global mammalian extinctions in the last 200 years have occurred in Australia (Short and Smith, 1994), and this process of defaunation is ongoing (Dirzo et al., 2014; Woinarski et al., 2010, 2011, 2015). At least 1,257 endemic plant species, 480 animals and 74 ecological communities currently are listed as vulnerable, endangered or critically endangered under Australia's federal *Environment Protection and Biodiversity Conservation Act* (EPBC) 1999 (Cresswell and Murphy, 2016). The primary threats to Australia's biodiversity are habitat loss, invasive vertebrate and plant species, and changed fire regimes (Evans et al., 2011; Kingsford et al., 2009). Deforestation in Australia is globally significant (Evans, 2016; FAO, 2001; Keenan et al., 2015; Macintosh, 2012): close to 300,000 hectares of forest was cleared in the state of Queensland alone in 2014-15 (Department of Science, Information Technology, Innovation and the Arts, 2016).

Australia has a federal system of governance, with environmental matters primarily regulated by eight state and territory governments (Council of Australian Governments (COAG), 1992; Peel and Godden, 2005), except for 'matters of national environmental significance' which are under the jurisdiction of the federal EPBC Act. Regulatory controls on deforestation emerged in the 1980's, and by the mid-2000's were in place in each state and territory (Bricknell, 2010; Evans, 2016). Australian governments have also experimented extensively with MBIs (Dovers, 2013; Gunningham and Young, 1997; Lockie and Higgins, 2007), and have been early adopters of biodiversity offsetting (Gibbons and Lindenmayer, 2007; McKenney and Kiesecker, 2010; Miller et al., 2015) and carbon farming (Macintosh and Waugh, 2012; van Oosterzee et al., 2014). As an economically wealthy nation, with strong governance arrangements and relatively small population, Australia has the capacity to effectively conserve its biodiversity, should it choose to (Lindenmayer, 2015; McDonald et al., 2015; Ritchie et al., 2013; Woinarski et al., 2017)

Thesis aim and research questions

In this thesis, I investigate regulatory and market-based policy responses to a key driver of biodiversity loss, namely deforestation, and seek to evaluate what outcomes, opportunities and risks these policies present for conservation. I pursue this broad research aim by addressing the following sub-questions:

1. What are the contemporary drivers of, trends in, and policy responses to deforestation in Australia?
2. What effect have regulatory policies had on the rate of deforestation in Queensland, Australia?
3. What are the potential economic, biodiversity and climate mitigation benefits that could be delivered by an emerging market-based policy instrument (namely, carbon farming) for agricultural landscapes in Queensland, Australia?
4. What are the key challenges in the design, implementation and evaluation of biodiversity offset policy?
5. What are the barriers and enablers to delivering positive outcomes for biodiversity from the Australian biodiversity offset policy?

The three key case studies examined in this thesis – namely, regulatory controls on deforestation, carbon farming, and biodiversity offsetting in Australia - were chosen because of their potential to examine contemporary policy responses to biodiversity loss, and the challenges and complexities associated with the design, implementation, and evaluation of biodiversity policy instruments in an advanced and politically stable economy. Although the focus of this thesis is biodiversity conservation policy in Australia, there is scope for the findings to be transferable to other suitable social-ecological-institutional contexts.

Methodological approach

This thesis uses an interdisciplinary, mixed-methods approach (Creswell and Clark, 2011; Johnson and Onwuegbuzie, 2004; Tashakkori and Teddlie, 2003). Interdisciplinary research enables a more holistic perspective on a research question that a single discipline cannot fully appreciate, in this case through a rigorous investigation of all components of the policy

process: design, implementation and evaluation. I draw upon both positivist and constructionist methodologies (Moon and Blackman, 2014), in recognition of the need to use a combination of quantitative and qualitative methods to answer my research questions. The specific methods used in each part of the thesis are described in more detail within their respective Chapters, but I provide a brief overview here:

In **Chapter Two**, I use document analysis to generate a history of the native vegetation policies introduced across Australia over the past four decades. I then analyse remotely sensed forest extent and change imagery using the ‘raster’ package in the R statistical software and ArcGIS to quantify the rate and extent of deforestation in Australia from 1972 to 2014. Results of this quantitative analysis are subsequently used in **Chapters Three** and **Six**.

In **Chapter Three**, I use a nonlinear mixed-effects bent-cable regression model to determine the extent to which regulations have influenced the rate of deforestation in Queensland, Australia since 1988.

For **Chapter Four**, I develop and implement a spatially explicit discounted cash-flow analysis to determine the economic viability of carbon farming in the agricultural landscapes in Queensland, Australia. I use MATLAB to implement the cash-flow analysis and analyse the results using R statistical software and ArcGIS.

In **Chapter Five**, I describe the contributions I made to the design of an offset metric used as part of the Australian EPBC Act Environmental Offsets Policy (2012). Specifically, I outline the mathematical framework which underlies the ‘no net loss’ offset policy objective, and facilitates the quantification of biodiversity losses and gains from an offset exchange.

In **Chapter Six**, I discuss key findings from Martin et al. (2016), which used a structured content analysis of submissions made to a public inquiry on the effectiveness of biodiversity offsetting in Australia. I also describe results from Maron et al. (2015) which draws on estimates of deforestation in each Australian state and territory, as quantified in **Chapter Two**.

Chapter Seven draws on data I collected through semi-structured interviews with policymakers, practitioners and industry proponents who have had direct experience with the implementation of biodiversity offset policy in Australia. I analyse interview data thematically using MAXQDA software.

Thesis structure

The thesis is in the form of *compilation*, where I answer my research questions through the publication of articles in the peer-reviewed scientific literature.

In **Chapter Two** I introduce the broad policy problem examined in this thesis – deforestation –which is the major threat to biodiversity persistence in Australia and worldwide. Here, I provide a detailed history and critique of native vegetation policies introduced across Australia over the past four decades. I document the recent shift away from ‘command and control’ policies towards self-regulation and incentive-based policy instruments. Despite this change in preference in policy instrument type, very little is known of the effectiveness of regularity policy responses to deforestation in Australia, let alone more recent approaches such as biodiversity offsetting and carbon farming. The remainder of the thesis examines three of the policy instruments described in Chapter Two: ‘command and control’ regulatory policies, carbon farming, and biodiversity offsetting.

In **Chapter Three**, I use a spatially explicit bent-cable regression to collectively model deforestation for 50 local government areas (LGAs) across Queensland, to evaluate the effectiveness of regulatory policies introduced to reduce deforestation. I show for the first time what (if any) role policy has had on the historical rate of deforestation, relative to macroeconomic, climatic, institutional and biophysical drivers.

In **Chapter Four**, I consider a case study of carbon farming in Queensland, which holds significant promise for incentivising large-scale restoration of degraded agricultural land. I demonstrate that assisted natural regeneration is a viable land use for large parts of Queensland, and is considerably more cost-effective than the more commonly used method of environmental plantings. Although the economics are sound, it is likely that policy

complexity, uncertainty, and socio-cultural barriers have limited the adoption of carbon farming to below the level expected by some scholars.

In the final three chapters, I present a detailed analysis of challenges to the design, implementation and evaluation of biodiversity offsetting. A **Preamble** briefly introduces the history of biodiversity offset policy in Australia and internationally, and outlines the motivation for the remaining chapters. **Chapters Five and Six** synthesise and expand upon the contributions I made to five co-authored publications (see **Appendix Five**), which address key technical and governance issues affecting biodiversity offset policy, as summarised by Maron and colleagues (2016). I first present the rationale behind the selection of an appropriate discount rate when evaluating the equivalence of biodiversity losses and gains, that explicitly accounts for a species risk of extinction (**Chapter Five**). This research contributed to the design of the loss-gain metric described by Gibbons and colleagues (2016), and was adapted for use in the Australian Government's Environmental Offset Policy (2012).

In **Chapter Six**, I conduct a preliminary examination of the governance of biodiversity offsetting by drawing on published work and new analysis. I highlight key findings from Martin et al. (2016), which draws upon a recent Senate inquiry into the effectiveness of biodiversity offsetting in Australia to elicit the perceptions of key stakeholders on offset policy principles, implementation and monitoring and evaluation. Drawing on literature from the policy sciences, I then describe a number of governance challenges affecting biodiversity offsetting, including agency problems such as adverse selection and moral hazard, and challenges for monitoring and policy evaluation. I argue that agency theory is a useful lens through which to examine the efficacy of policy implementation in the context of multi-actor governance, and to understand the behavior of policy actors engaged in contractual relationships such as in biodiversity offset trades. I conclude Chapter Six by considering the findings of Maron and colleagues (2015), which critically analyse the baseline assumptions of offset policies in eight Australian jurisdictions by comparing their assumed rates of biodiversity loss with actual rates of deforestation (as per **Chapter Two**). I argue that the implausibly high baseline assumptions made by these policies demonstrates that adverse selection is undermining the potential environmental outcomes from biodiversity offsetting in Australia.

My final contribution to this case study, and to my thesis, examines the governance dimensions of biodiversity offsetting in greater detail. A key gap in the literature on biodiversity offsetting and environmental policy more broadly, is a focus on the socio-political context in which policy is developed and implemented. The policy actors involved, their motivations and objectives, and the institutional and organizational incentives in place all influence the capacity or inclination for environmental policy to be effectively, efficiently, and fairly implemented (**Chapter Seven**). My findings indicate that although improvements in offset policy and metric design has provided regulators and proponents with more guidance on how to implement offset policy, key barriers remain in the way of achieving positive biodiversity outcomes from offsetting: namely policy uncertainty, lack of capacity for effective oversight, limited public accountability, an inability to make offsetting commercially viable, and the use of offsetting in a piecemeal, non-strategic approach.

I conclude the thesis and provide future research directions in **Chapter Eight**.

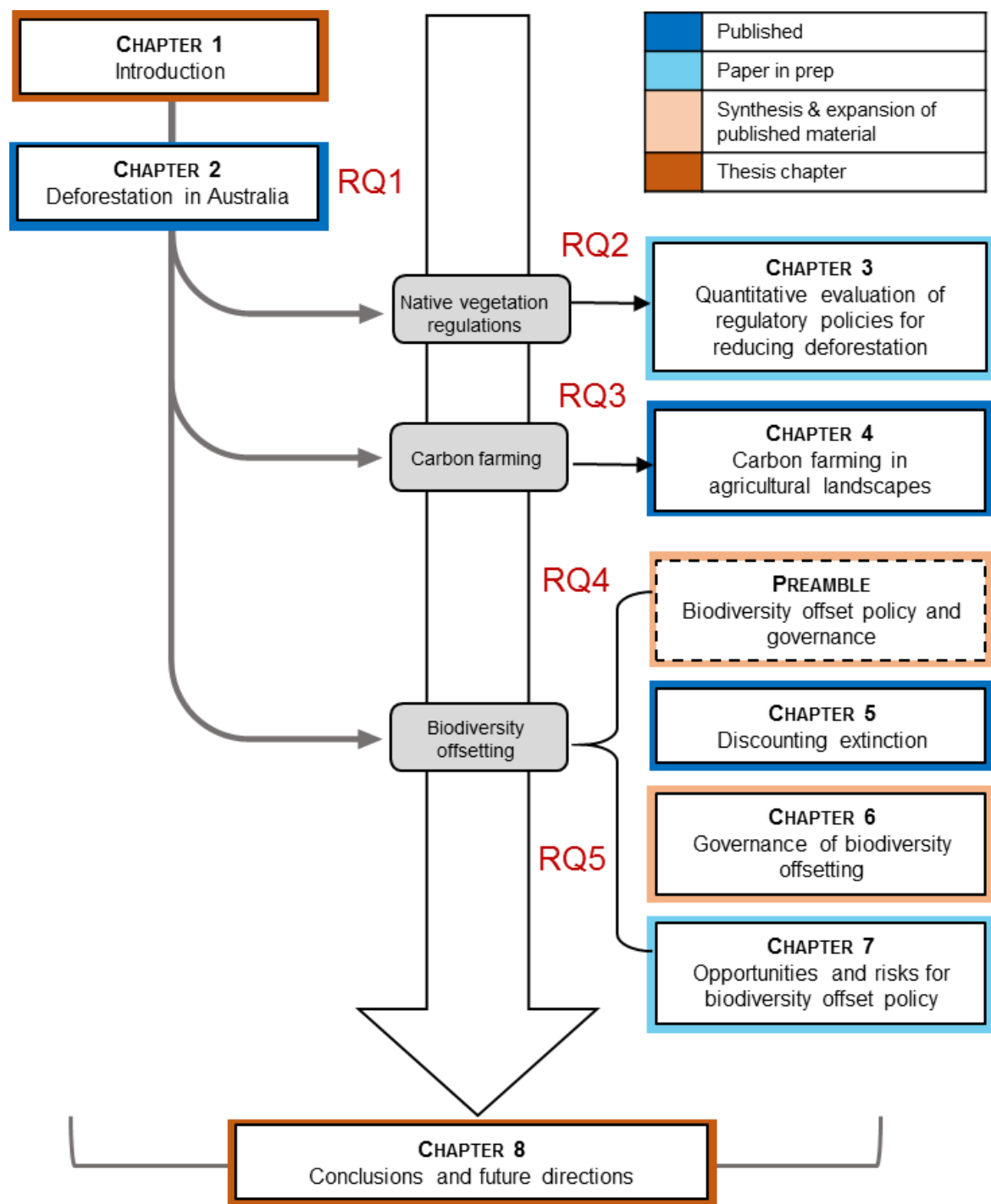


Figure 1.3. Thesis structure. Research sub-questions 1 to 5 and corresponding Chapters are indicated in red text.

Research significance

This thesis makes a significant contribution to knowledge and conservation policy through:

- The most comprehensive review and analysis to date of the drivers of, trends in, and policy responses to deforestation in Australia (Chapter Two);
- The development of new methodological approaches and analytical frameworks; specifically, the bent-cable regression model with spatial adjacency weightings as applied in Chapter 3, and the mathematical framework for evaluating the capacity for offsets to achieve ‘no net loss’ from the perspective of threatened species time preference in Chapter 5;
- New empirical findings which illustrate the effectiveness of regulatory policies for reducing deforestation in Queensland, Australia (Chapter Three), the potential effectiveness of carbon farming (Chapter Five), and the barriers and enablers to effective implementation of biodiversity offset policy in Australia (Chapter Seven);
- The consideration of biodiversity conservation in the context of key lessons from the policy sciences, including the application of agency theory to understanding the dynamics of policy actors as part of the governance of biodiversity offsetting (Chapter Six)
- The direct application of research in this thesis to the development (Chapter Five) and continual improvement (Chapter Seven) of biodiversity offset policy in Australia.

CHAPTER TWO:

Deforestation in Australia: drivers, trends and policy responses

A version of this chapter has been published in *Pacific Conservation Biology* as:

Evans, M.C. (2016). Deforestation in Australia: drivers, trends and policy responses.
Pacific Conservation Biology 22: 130–150.

Abstract

Australia's terrestrial environment has been dramatically modified since European colonisation. Deforestation – the clearing and modification of native forest for agricultural, urban and industrial development – has been, and remains a significant threat to Australia's biodiversity. Substantial policy reform over the last 40 years has delivered a range of policy instruments aimed to control deforestation across all Australian States and Territories. Despite these policy efforts - as well as strong governance and high institutional capacity - deforestation rates in Australia were nonetheless globally significant at the turn of this century. Legislation introduced in Queensland and New South Wales during the mid-2000s was at the time seen to have effectively ended broad-scale clearing; however, recent policy changes have raised concerns that Australia may again become a global hotspot for deforestation. Here, I describe the deforestation trends, drivers and policy responses in Australia over the last four decades. Using satellite imagery of forest cover and deforestation events across Australia between 1972 and 2014, I present a comprehensive analysis of deforestation rates at a fine resolution. I discuss trends in deforestation with reference to the institutional, macroeconomic and environmental conditions which are associated with human-induced forest loss in Australia. I provide a detailed history and critique of the native vegetation policies introduced across Australia over the last 40 years, including recent legislative amendments and reviews. Finally, I comment on future prospects for curbing deforestation in Australia, including the role of incentive-based policies such as carbon farming, private land conservation and biodiversity offsets. Despite being a highly active policy space, very little is known of the effectiveness of policy responses to deforestation in Australia, and whether the recent shift away from 'command and control' policies will necessarily lead to better outcomes. My analysis demonstrates the need for an effective policy mix to curb deforestation in Australia, including a greater focus on monitoring, evaluation and policy learning.

Introduction

Habitat loss is recognized as a major threat to biodiversity within the Oceania region (Kingsford et al., 2009). Globally and within Oceania, Australia is significant for both its megadiversity (Mittermeier and Mittermeier, 1997), and the extent to which its terrestrial species and ecosystems have been impacted by human activities (Evans et al., 2011; Mittermeier et al., 1999; Myers et al., 2000; Williams et al., 2011). Prior to European colonisation, approximately 30% of Australia's terrestrial area was covered in 'forest' (Barson et al. 2000; Bradshaw 2012) – defined as forest and woodland dominated by trees up to 2 metres high, with at least 20% canopy cover and a minimum area of 0.2 hectares (Furby, 2002). Since that time, around 40% of this original forest extent has been subject to deforestation: cleared or extensively modified for agricultural, urban or industrial development (Barson et al., 2000; Bradshaw, 2012; Graetz et al., 1995; Lindenmayer, 2005). Much of Australia's remaining forest, shrubland, grassland and open woodland ecosystems are now degraded or fragmented (Kirkpatrick, 1994; Norton, 1996; Tulloch et al., 2016).

As a developed nation, with strong governance arrangements, a high level of institutional capacity and a relatively small population, it might be expected that deforestation in Australia should be slowing towards a 'forest transition' –the cessation and eventual reversal of forest loss (Angelsen and Kaimowitz, 1999; Lambin and Meyfroidt, 2011; Rudel et al., 2005). Yet at the turn of the 21st century, Australia's deforestation rate was the sixth highest in the world (ACF, 2001; FAO, 2001), and the latest statistics suggest that Australia's deforestation may again become globally significant (Bulinski et al., 2016; Department of Science, Information Technology, Innovation and the Arts, 2015).

The environmental impacts of deforestation cannot be disputed. Clearing, modification and fragmentation of native vegetation erodes soil, contributes to salinity, and are key drivers in the decline of woodland birds, reptiles and mammals (McAlpine et al., 2002; Norton, 1996; Saunders, 1989). Land clearing, the local term for deforestation, has been repeatedly identified as the most significant threat to terrestrial biodiversity in Australia (State of the Environment 2011 Committee, 2011). Deforestation is also a major contributor to human induced climate change. In the base year of the Kyoto Protocol (1990), greenhouse gas

emissions due to deforestation in Australia equated to 132 Mt CO₂-e, or 25% of the country's total emissions (Australian Government, 2013; Macintosh 2012). However, deforestation in Australia is also a deeply political issue, and has been a prominent topic of debate between environmentalists, farmers and foresters over the last four decades (Lindenmayer, 2014). The drivers, trends and policy responses to deforestation cannot be fully understood without reference to its institutional⁵ and macroeconomic dimensions, in addition to its ecological impacts and limits.

The history of deforestation in Australia was most recently examined by Bradshaw (2012), who draws upon the first systematic nation-wide study of land cover change over 1990-1995 (Barson et al., 2000), as well as the Australian Bureau of Statistics (2005) and National Carbon Accounting System (NCAS, Australian Greenhouse Office, 2003) to document the trends in forest loss and degradation across Australia from European settlement up until 2005. Bartel (2004, 2008) provides the most detailed reviews of Australia's native vegetation policies to date, and highlights the importance of the use of satellite imagery to monitor deforestation and to evaluate policy effectiveness. However, significant changes have occurred in the policy landscape since the publication of Bradshaw (2012) and Bartel (2004, 2008). Legislation introduced from 2005 in the historically high deforestation States of Queensland and New South Wales had arguably marked the end of broad-scale land clearing in Australia (McGrath, 2007; Taylor and Dickman, 2014). Yet since 2010, a nationwide trend towards the relaxation of native vegetation regulations may be leading to increased deforestation (Bulinski et al., 2016), and so an up-to-date summary of deforestation trends, drivers and policy responses is needed.

Notwithstanding the extensive commentary that exists on Australia's protected areas (Taylor et al., 2011; Watson et al., 2011), natural resource management (Hajkowicz, 2009; Lockie and Higgins, 2007; Robins and Kanowski, 2011), and forestry policy (Kirkpatrick, 1998; Lane, 1999; Norton and Mitchell, 1993), there has been comparatively limited examination of the policy responses to deforestation (Bartel, 2003, 2004, 2008; Macintosh, 2012).

⁵ Institutions incorporate formal (laws, property rights) and informal (traditions, cultural and social norms) rules (North, 1991). In this paper I focus on formal institutions, though recognise that cultural factors are also important drivers of deforestation behaviour (Australian Greenhouse Office, 2000; Bartel and Barclay, 2011).

Although the impacts on biodiversity from the loss and degradation of native forests through commercial forestry operations are well documented (Lindenmayer, 2014), deforestation due to agricultural, urban and industrial development on private land, particularly since the 1970s, has had far more widespread impacts (Australian Government, 2013; Barson et al., 2000).

Here, I provide a comprehensive review of policy instruments aimed to control deforestation in Australia over the last four decades. I focus specifically from 1970 onwards for three reasons. First, the early history of deforestation in Australia has been covered extensively elsewhere (Bartel, 2004, 2008; Bombell and Montoya, 2014; Bradshaw, 2012; Rolfe, 2000; Seabrook et al., 2006), but there has been comparatively limited focus on its policy dimensions (c.f Bartel, 2004; 2008), and no analysis of contemporary policy trends from 2005 onwards. Second, in the context of Australia's history since European colonisation, government regulation of deforestation is only a fairly recent phenomenon. Deforestation for agricultural development has historically been incentivized by the Federal and state governments through low-cost finance, tax concessions, cheap land and lease conditions which required the removal and management of native vegetation (Australian Bureau of Statistics, 2002; Australian Greenhouse Office, 2000; Seabrook et al., 2006). The majority of these incentives were removed by the 1980s, when public concern over the environmental effects of deforestation began to rise (Council of Australian Governments, 1992). Finally, nationally-consistent spatial data on deforestation events developed as part of the National Carbon Accounting System (NCAS) are now available from 1972 up to 2014 (Australian Department of the Environment, 2015).

I first present a comprehensive analysis of deforestation rates at a fine resolution, by analysing satellite imagery of forest cover and deforestation events across Australia between 1972 and 2014. I discuss these statistics by State, land use and land tenure. Next, I provide a detailed history and critique of native vegetation policies across Australia, including recent legislative amendments and reviews. I conclude with an analysis of policy trends with reference to the broader macroeconomic trends over the last 40 years, and comment on future prospects for curbing deforestation in Australia, including the role of a more diverse range of policy instruments.

Deforestation trends and drivers

Data analysis and methodology

I draw upon the most recent national-scale spatial data to describe deforestation trends over time (Australian Department of the Environment, 2015). The Australian Government compiles fine-resolution data of land cover change as part of the NCAS (NCAS, Furby, 2002; Lehmann et al., 2013). The NCAS uses over 7,000 Landsat MSS, TM and ETM+ images to map forest extent and change at a 25 metre resolution across the Australian continent, excluding the treeless inland desert areas and grasslands. Note that these spatial data exclude native vegetation types which do not meet the height, canopy cover and area thresholds of ‘forest’ as defined by Furby (2002), meaning that the loss of grassland, shrubland and open woodland is not captured by this analysis. While I discuss deforestation trends specifically with reference to data on forest extent and change, I refer to the policy responses to deforestation as ‘native vegetation policies’, which recognises that clearing of non-forest vegetation is often (but not always) regulated in addition to the clearing of forest.

Data on forest extent and change are available for twenty-three epochs (instances in time) from 1972 to 2014 in the intensive land use zone only (Graetz 1995), where the majority of landscape modification has occurred. Forest change events were attributed to human intervention, meaning that “natural” forest change attributable to factors such as fire (and associated recovery), dieback, salinisation, drought and seasonal flushing were removed (Furby, 2002). Prior to 2004, annual data on deforestation events are not available within the NCAS, and are instead captured within multi-year epochs. For example, the 1972 epoch contains deforestation events over a five year period from 1972 to 1977. Following expert advice (Australian Department of the Environment, 2015; Reddy, S, pers. comm.), I converted deforestation events contained within the twenty-three epochs into annual values over forty-three years from 1972 - 2014. Further details are provided in the Supplementary Material (**Appendix One**).

The amount of arable land available to clear has been highlighted previously as an important factor influencing deforestation behavior (Australian Greenhouse Office, 2000; Bartel 2004; 2008). In particular, Bartel (2004) suggests that native vegetation policies introduced in

South Australia in the 1980s may have had little effect simply because there was scarce primary (remnant) forest remaining on land suitable for agricultural development. As such, an exploration of deforestation trends in the context of the amount of primary forest remaining intact is warranted. To derive an estimate of the amount of primary forest remaining over time, I assume that the forest extent in 1972 (the earliest epoch in the data set) is all primary forest. I then deduct the primary deforestation events each year from the remaining primary forest extent from the previous year. This calculation resulted in an estimate of primary forest remaining from 1972-2014. I derive an estimate of deforestation occurring on reforested land (regrowth deforestation) by considering deforestation events which occurred on land classified as non-forest in 1972, as well as land which was deforested, and subsequently reforested and deforested again over the 1972 – 2014 time period. In reality much of the forest extent in 1972 would in fact be regrowth forest, so the results I present here should be considered in the context of this simplifying assumption. While the total amount of deforestation would be unaffected by this assumption, the primary deforestation statistics should be regarded as an overestimate, and the regrowth deforestation as an underestimate.

I use the most recent national datasets (ABARES, 2010; Geoscience Australia, 1993) to summarise deforestation trends by land use and tenure. Note that land use and tenure data are not available over the full time series, so these summaries should only therefore be considered as an estimate. I use the ‘raster’ package (Hijmans and van Etten, 2012) in R Statistical software (R Development Core Team, 2017) for all raster processing.

Trends in deforestation

From 1972 to 2014, over 7.2 million hectares of primary forest was cleared across Australia. The total land forested in 1972 was 101 million hectares, hence the primary deforestation that has occurred up to 2014 represents a 7% reduction in this extent. An additional 9.5 million hectares of regrowth forest was cleared over this period (Figure 2.1). The majority of this deforestation has occurred in Queensland, where 9.7 million hectares of forest has been cleared, of which 3.6 million was primary deforestation.

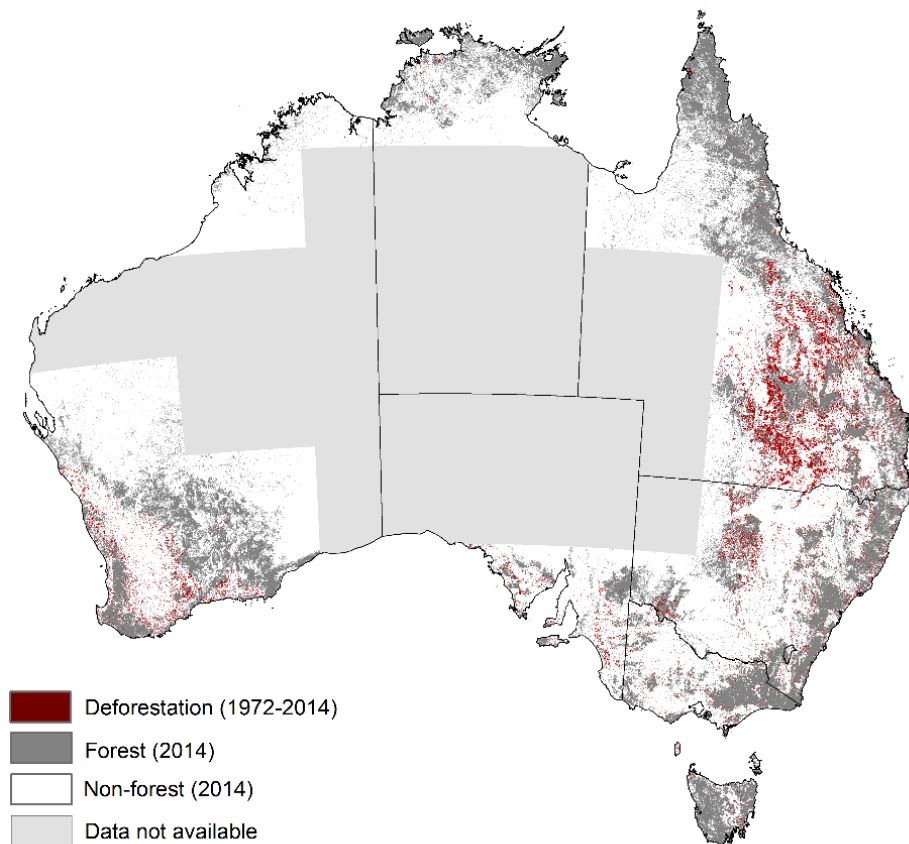


Figure 2.1. Distribution of total deforestation events (including primary and regrowth deforestation) attributed to human intervention from 1972 – 2014. Forest change attributable to “natural” factors such as fire, dieback, salinisation, drought and seasonal flushing is not shown. Data is sourced from the National Carbon Accounting System (NCAS), Australian Department of the Environment (2015).

The greatest overall deforestation occurred in the decade of 1980 – 1989; where close to 4.7 million hectares of native vegetation (including 2.4 Mha of regrowth) was cleared across the country (Figure 2.2). Total deforestation has declined in the following decades, however regrowth deforestation increased again in the 2000s, during which Queensland cleared 1.5 Mha of regrowth vegetation. The rate of primary deforestation has still substantially decreased since the 1970s, when extensive tracts of forest in southwestern Western Australia and Queensland were cleared for agricultural development (Barson et al., 2000; Graetz et al., 1995).

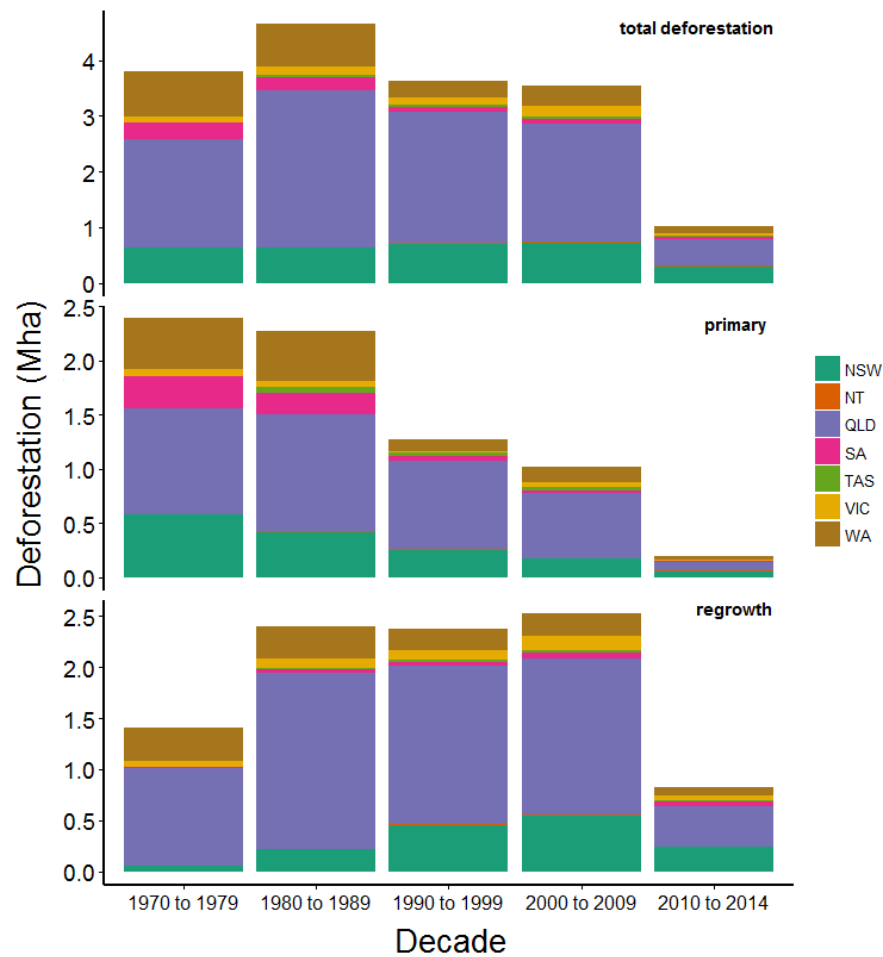


Figure 2.2 Amount of deforestation (total, primary and regrowth) per decade, from 1972-2014, for all Australian states and territories (excluding Australian Capital Territory).

As indicated by Figure 2.3a, the majority of deforestation has occurred for pasture, with much smaller percentages for cropping, forestry, urban development and mining. A surprisingly high percentage of clearing occurred in conservation areas and minimal use areas; however this may not be an accurate representation and should be regarded as an estimate only, given the use of the 2005-2006 land use layer (ABARES, 2010). Only a small percentage of deforestation has occurred on public land (2%, Figure 2.3b), with the remainder occurring on freehold (78% over 1972-2014) and leasehold land (20%). Deforestation has occurred disproportionately on freehold land, relative to the percentage of total land mass held in this tenure (31%, Geoscience Australia, 1993).

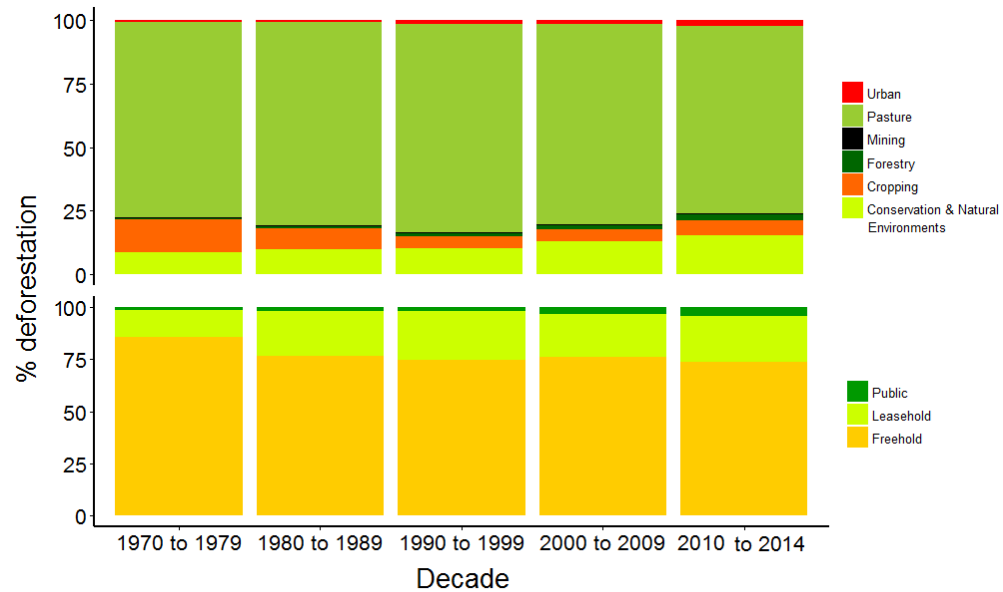


Figure 2.3. Percentage of total deforestation in each decade, by (a) land use (as of 2005-06, above) and (b) land tenure (as of 1993, below). Data are sourced from ABARES (2010) and Geoscience Australia (1993), respectively.

Relative to the amount of primary forest remaining, there has generally been a decline in primary deforestation in each State over time (Figure 2.4), although an increase in the rate of deforestation can be seen in several states in the early 1990s and early-mid 2000s. At the national scale, there is an overall declining trend in deforestation, and a link to the amount of primary forest remaining to clear is also apparent (Figures 2.5a,b). There appear to be some relationships between deforestation in Australia over time and key macroeconomic and climatic variables (Figure 2.5), though this requires further analysis to confidently attribute any change in these variables to the rate of deforestation. Similarly, a rigorous quantitative evaluation is needed to reliably establish what effect the introduction of native vegetation policies over the past four decades has had on deforestation in Australia.

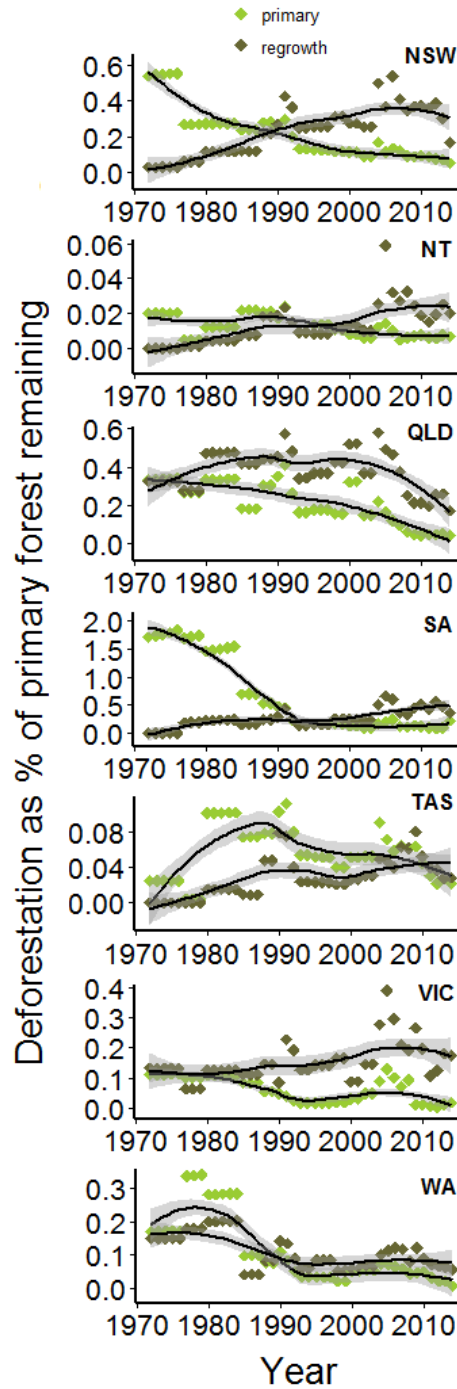


Figure 2.4. Deforestation as a percentage of primary forest remaining, with separate Loess (local regression, Cleveland and Devlin (1988)) curves for primary and regrowth deforestation, for all Australian states and territories (excluding Australian Capital Territory). Grey shading indicates a 95 % confidence interval around the Loess curve.

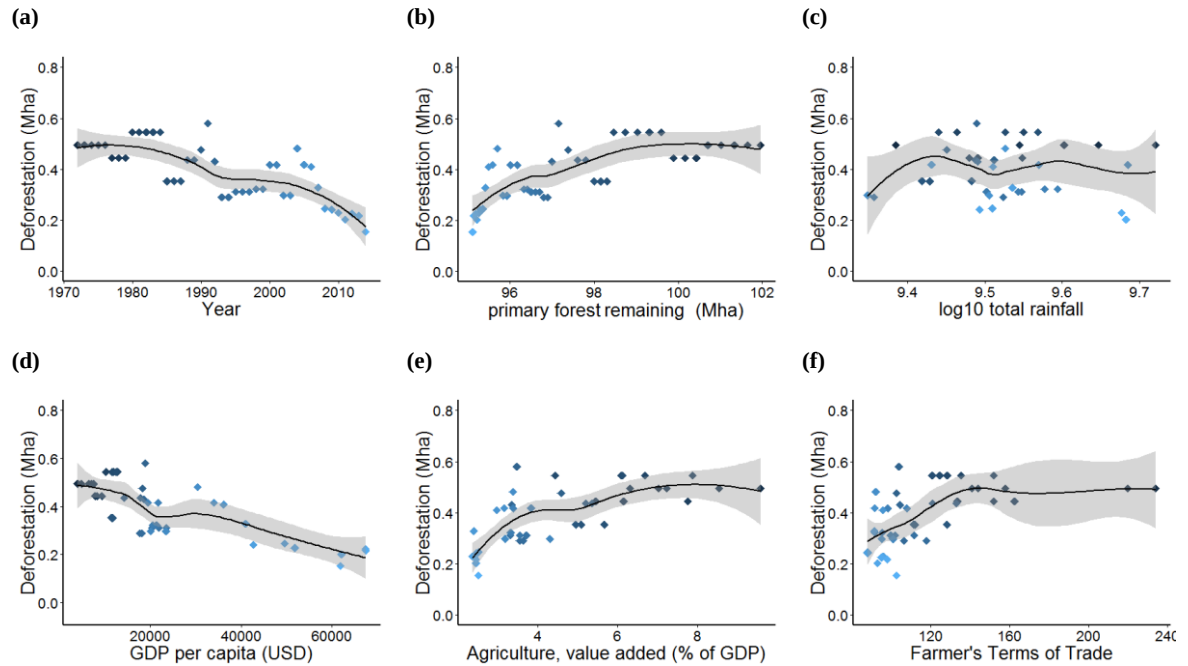


Figure 2.5. Trends in national-scale deforestation and key macroeconomic variables. Data points are shaded by year (1972: dark blue, to 2014: light blue). A Loess curve is fitted to each plot, and grey shading indicates a 95 % confidence interval. Plots are total deforestation versus: (a) Year, (b) Extent of primary forest remaining, (c) Log-transformed total rainfall (Evans, 2014), (d) Gross domestic product per capita (current USD) (The World Bank, 2015), (e) Agriculture, value added (% total GDP; value added is the net output of a sector after adding up all outputs and subtracting intermediate inputs) (The World Bank, 2015), (f) Farmer's terms of trade (ABARES, 2015).

A history of native vegetation policy in Australia

Deforestation is mainly regulated at the State level in Australia (Bricknell, 2010). Land clearing has been listed as a Key Threatening Process under the Federal Government's *Environmental Protection and Biodiversity Conservation Act (1999)* (EPBC Act) since 2001 (Department of the Environment, 2001; Lindenmayer, 2005). The Federal Government has limited jurisdiction over State environmental matters unless there are impacts on 'matters of national environmental significance' (MNES) such as a threatened species or ecological communities, or activities on Commonwealth land (Peel and Godden, 2005). This means that

vegetation communities generally receive no federal protection until they have already been extensively cleared (Tulloch et al., 2016).

Nonetheless, several attempts have been made to deliver a coordinated, national approach to the management of native vegetation. Since 1997, various commitments to address the decline in native vegetation have been made through nationally-funded programs and frameworks (Natural Resource Management Ministerial Council, 2001). The Natural Heritage Trust funding package aimed to deliver no net loss of native vegetation within Australia by July 2001. This goal was not met (Beeton et al., 2006). The most recent framework outlines a target for a net national increase in native vegetation extent and connectivity by 2020 (COAG Standing Council on Environment and Water, 2012). This framework, like others before it, is not prescriptive or binding.

The current environmental offsets policy under the EPBC Act aims to compensate for significant impacts on MNES relative to a ‘business as usual’ baseline (Australian Government, 2012; Maron et al., 2013), which as described by Maron et al. (2015) is one of ongoing biodiversity decline. Although not directly related to native vegetation policy and management in the State jurisdiction, the declining baseline assumed by the national environmental offsets policy suggests that the national target of a net increase in native vegetation is not expected to be met.

Protected areas can play a key role in reducing deforestation if genuine averted losses can be secured, and deforestation is not simply displaced elsewhere (Andam et al., 2008; Miteva et al., 2012; Pressey et al., 2015). Australia's National Reserve System is focused primarily on meeting goals for the conservation of biodiversity (NRMMC 2009; Watson et al., 2011), and it is not known whether the system as a whole has had an impact on deforestation in Australia. However, Pressey (2002) demonstrated that protected areas in New South Wales are biased towards steep and infertile public land, rather than on privately managed land where deforestation is generally higher. State-level policies designed to regulate deforestation on private (freehold and leasehold) land are therefore the main focus of this paper.

For the remainder of the paper, I focus primarily on policy reforms that have occurred in the historically high-deforestation states of Queensland, New South Wales, Western Australia,

South Australia and Victoria. The majority of clearing for agriculture in Tasmania occurred during the 1970s and 1980s (Kirkpatrick, 1991), and was not regulated until 2002 when the *Forest Practices Act 1985* was amended to prohibit non-commercial clearing of forest for agricultural purposes (Bricknell, 2010). Limited deforestation has occurred in the Northern Territory, as its rangelands are generally suitable for grazing in their natural state (Australian Greenhouse Office, 2000). Controls were introduced in 2002 under the *Planning Act 1999* and *Pastoral Land Act 1992* (Table 3). With a total area of 2,358 km², deforestation in the Australian Capital Territory is insignificant relative to other Australian jurisdictions, and policies dealing specifically with native vegetation have only recently been introduced (Bricknell, 2010).

In the following section, I describe how policy responses to deforestation have evolved in all Australian jurisdictions over the last four decades. I also provide a comprehensive summary of major native vegetation policies implemented at the Federal, State and Territory levels from 1970 to 2016 in Tables 2.1 to 2.4.

Table 2.1. Major native vegetation policies over 1970-1989

Jurisdiction	Year	Policy name	Policy intent	Instrument type (Gunningham and Sinclair, 1999)	Details	Source
South Australia	1980	Heritage Agreements under the <i>South Australian Heritage Act 1978</i>	To maintain or improve native vegetation with high conservation value on private land	Voluntarism, Economic	Legally binding agreements between the Crown and individual (current and future) landholders. Financial incentives provided to cover fencing costs, management assistance and advice; state and local overnment land charges are waived.	Harris, 2013; Slee 1997
	1983	Development Control Regulations under the <i>Planning Act (PA) 1982</i>	To curb native vegetation clearance	Command and control	Prescribed vegetation clearance as a class of development that required consent of the South Australian Planning Commission. Applied to clearing for agricultural purposes and commercial harvesting. Clearing controls removed from <i>PA 1982</i> and placed under <i>NVA 1985</i> . Compensation is paid to those refused clearing approval, and offered financial assistance to	Harris, 2013
	1985	<i>Native Vegetation Act (NVA) 1985</i>	To curb native vegetation clearance	Command and control, Economic	landholders prepared to enter a Heritage Agreement to protect remnant native vegetation.	AGO 2000; Harris, 2013; Slee 1997
Western Australia	1978	<i>Country Areas Water Supply Act (CAWSA), Part IIA, 1976</i>	To protect quality of water supply from salinity	Command and control	Vegetation clearance controlled in 6 south western catchments (about 5% of rural parts of the State)	Slee, 1997
	1986	<i>Soil and Land Conservation Act (SALCA)1945</i>	Conservation of soil and land resources, and mitigation of the effects of erosion, salinity and flooding	Command and control	In 1986, it became a requirement under the <i>SALCA 1945</i> to obtain a notice of intention to clear for areas of 1ha or more	Bennett, 2002; Slee, 1997
Victoria	1989	Amendment S16 to the <i>State Section of the Planning Scheme (SSPS)</i> under the <i>Planning and Environment Act 198</i>	Nature conservation and land management (soil, watercourses, greenhouse carbon loads, groundwater, dryland salinity control) End of broad-scale clearing in Victoria	Command and control	Planning permits are required to remove, destroy or lop native vegetation. Applied statewide to freehold, leasehold, Crown land (except Crown land used for forestry, and national parks). The Act specifically rules out the payment of compensation.	Slee, 1997

Table 2.2. Major native vegetation policies over 1990-1999

Jurisdiction	Year	Policy name	Policy intent	Instrument type (Gunningham and Sinclair, 1999)	Details	Source
Commonwealth	1999	<i>Environmental Protection and Biodiversity Conservation Act (1999)</i>	Protection of Matters of National Environmental Significance	Command and control	Limited remit over vegetation clearance, only if "significant impact"	
South Australia	1991	<i>Native Vegetation Act (NVA) 1991</i>	To retain native vegetation. End of broadscale clearing in South Australia	Command and control	End of compensation payments under NVA 198. Applies to freehold and leasehold land, with exception of certain urban areas.	Harris, 2013; Slee, 1997
New South Wales	1995	<i>State Environmental Planning Policy (SEPP) No. 46 under the Environmental Planning & Assessment (EP&A) Act 1979</i>	To prevent inappropriate native vegetation clearance	Command and control	Clearing of native vegetation (> 2 ha/annum) prohibited except with development consent of the Director-General of the Department of Land and Water Conservation (DLWC) and National Parks and Wildlife Service (NPWS)	Bombell and Montoya, 2014; Slee, 1998
	1998	<i>Native Vegetation Conservation Act (NVCA) 1997</i>	Conservation and management of native vegetation in accordance with the principles of ecologically sustainable development	Command and control, Economic	Brought all the clearing of all native vegetation in NSW under one regulatory regime (including leasehold land in Western Division). Regional vegetation management plan specified where clearing could occur. Provided some financial incentives for landholders to protect native vegetation.	Bombell and Montoya, 2014; Smith, 1999
Queensland	1995	<i>Land Act 1994</i>	Sustainable resource use and development	Command and control	Control of vegetation clearing on leasehold and State lands	AGO 2000; Slee, 1998
Victoria	1997	Victoria's Biodiversity Strategy 1997	Biodiversity conservation	Voluntarism, Information	State-wide target of 'no net loss' of native vegetation by 2001 (baseline 1999). 'Net gain' target from 2002	Department of Natural Resources and Environment, 1997

Table 2.3. Major native vegetation policies over 2000-2009

Jurisdiction	Year	Policy name	Policy intent	Instrument type (Gunningham and Sinclair, 1999)	Details	Source
Commonwealth	2001	National Framework for the Management and Monitoring of Australia's Native Vegetation	To reverse the long-term decline in the quality and extent of Australia's native vegetation cover by June 2001	Voluntarism, Information	Provided the framework for the implementation of the Natural Heritage Trust Partnership Agreement between the Commonwealth and the State and Territory Governments.	Natural Resource Management Ministerial Council, 2001
South Australia	2002	<i>Native Vegetation Act (NVA) 1991</i>	Permitted clearing conditional on achieving 'significant environmental benefits'	Command and control, Economic	Landholders could apply for financial assistance for delivering 'significant environmental benefits', enter into a Heritage Agreement or pay into a Native Vegetation Fund	Bricknell, 2010; South Australian Government 2002
New South Wales	2005	Native Vegetation Regulation 2005 under the <i>Native Vegetation Act (NVA) 2003</i>	To end broadscale clearing except where the clearing will improve or maintain environmental outcomes End of broad-scale clearing in New South Wales	Command and control, Economic	Permits required to clear vegetation, granted on condition of improving or maintaining environmental outcomes. Approval not required to clear regrowth vegetation or for routine agricultural activities	Productivity Commission, 2004
	2008	BioBanking under <i>Threatened Species Conservation (Biodiversity Banking) Regulation 2007</i>	To address the loss of biodiversity values from habitat degradation	Economic	Created a market where biodiversity credits could be bought and sold	Department of Environment and Climate Change 2007, Gibbons 2009
Queensland	2000	<i>Vegetation Management Act 1999</i>	To preserve endangered ecosystems, prevent land degradation, maintain biodiversity	Command and control	Regulates the clearing of vegetation on freehold land. Clearing of regrowth vegetation still allowed	Kehoe, 2009; McGrath, 2007
	2003	<i>Vegetation (Applications for Clearing) Act 2003</i>	To restrict vegetation clearing	Command and control	Imposed a moratorium on applications to clear remnant vegetation on freehold and leasehold land	McGrath, 2007
	2004	<i>Vegetation Management and Other Legislation Bill 2004</i>		Command and control		Kehoe, 2009; McGrath, 2007

	2006	<i>Vegetation Management and Other Legislation Bill 2004</i>	To end broadscale clearing of remnant vegetation. End of broadscale clearing in Queensland	Command and control	A ballot for clearing of 500,000 hectares was held in September 2004. All clearing permits issued under the ballot expired on 31 December 2006. Development applications guided by <i>State Policy for Vegetation Management</i> (2006). Provided \$150 million of financial assistance over five years	Giskes, 2004; McGrath, 2007
	2008	Queensland Government Environmental Offsets Policy (QGEOP) incorporating the Policy for Vegetation Management Offsets (2007)	To counterbalance unavoidable, negative environmental impacts that result from a development.	Command and control, Economic	A vegetation management offset may be secured to meet the performance requirements of a Regional Vegetation Management Code under the <i>Vegetation Management Act 1999</i> . Proponents may directly secure and manage an offset, engage a third party, or pay into an offset fund.	Queensland Government, 2008
	2009	<i>Vegetation Management and Other Legislation Amendment Act 2009</i>	To regulate clearing of regrowth vegetation	Command and control	Amendments to the VMA 1999 to protect 'high value regrowth'	Taylor, 2015
Victoria	2003	Victoria's Native Vegetation Management - A Framework for Action (2002)	To achieve a state-wide Net Gain in the extent and quality of native vegetation	Command and control, Information	Given regulatory force in 2003 when changes were made to the Victoria Planning Provisions (VPP). Framework must be considered when assessing proposals to clear vegetation	Department of Sustainability and Environment, 2010
	2006	BushBroker	To facilitate achievement of state-wide Net Gain target	Economic	Created a market where native vegetation credits could be bought and sold	Nemes et al., 2008; O'Connor, 2009; Stoneham et al., 2003
Western Australia	2004	Environmental Protection (Clearing of Native Vegetation) Regulations 2004 under the <i>Environmental Protection Act 1986</i>	Conservation, preservation, protection, enhancement and management of the environment	Command and control	Amendments introduced provisions for regulating the clearing of native vegetation on all land in Western Australia via a permit system. Approval conditions may include establishing a vegetation offset or contribution to an offset fund	Squelch, 2007
	2006	Environmental Offsets Position Statement no. 9	To achieve a 'net environmental benefit' from new developments	Command and control, Economic	Formalised offsetting provisions in Western Australia after being considered on an ad-hoc basis since at least 2000	Environmental Protection Authority, 2006, 2008; Hayes and Morrison-Saunders, 2007
Tasmania	2002	<i>Forest Practices Act 1985</i>	To regulate non-commercial clearing	Command and control	Amendments in 2002 prohibit non-commercial clearing of forest for agricultural purposes	Bartel, 2004; Bricknell, 2010
Northern Territory	2002	Land clearing guidelines 2002 under the <i>Planning Act (PA) 1999</i> and <i>Pastoral Land Act (PLA) 1992</i>	To minimise the impact of land clearing on natural resources	Command and control	Clearing on freehold, Crown and indigenous land regulated by PA 1999, where landholders are required to obtain a permit to clear more than 1ha of native vegetation. Consent required to clear on pastoral land under the PLA 1992	Northern Territory Government, 2005

Table 2.4. Major native vegetation policies over 2010-2016

Jurisdiction	Year	Policy name	Policy intent	Instrument type (Gunningham and Sinclair, 1999)	Details	Source
Commonwealth	2012	Australia's Native Vegetation Framework	To deliver a net national increase in native vegetation extent and connectivity by 2020	Voluntarism, Information	Framework to guide actions across government strategies, policies, legislation and programs related to native vegetation management	COAG Standing Council on Environment and Water, 2012
	2012	<i>EPBC Act Environmental Offsets Policy</i>	To maintain or improve viability of Matters of National Environmental Significance	Command and control, Economic	Limited remit over vegetation clearance, but relevant as assumed baseline trajectory of biodiversity decline runs counter to goal of 2012 Native Vegetation Framework	Australian Government, 2012
New South Wales	2013	Native Vegetation Regulation 2013 under the <i>NVA 2003</i>	To deliver a balanced regime of environmental protection and efficient agricultural management	Self-regulation, Information	Self-assessable codes to be made for certain common clearing activities without the need for a permit	Byron et al., 2014; Lane, 2013; Parker, 2013
	2014	NSW biodiversity offsets policy for major projects	To achieving long-term conservation outcomes while enabling development	Command and control, Economic	Currently applies only to state significant development and infrastructure, but recommended for expansion to all development activities by Byron et al., 2014	Byron et al., 2014;
Queensland	2011	Queensland Biodiversity Offset Policy (version 1)	To ensure that there is no net loss of biodiversity	Command and control, Economic	Offsets may be provided directly, through a third-party, or as a payment to a trust fund	Department of Environment and Resource Management, 2011
	2012	Vegetation Management Regulation 2012 under the <i>Vegetation Management Framework Amendment Act 2013</i>	To reduce red tape and regulatory burden, simplify vegetation management framework, and maintain sustainable vegetation clearing practices to protect native vegetation	Self-regulation, Information	Introduced a series of self-assessable codes for vegetation clearing, removed regulations on 'high value' regrowth clearing, introduced permitted clearing for necessary environmental clearing, high and irrigated high value agricultural clearing	Taylor, 2013; 2015
		Queensland Biodiversity Offset Policy Version 1.0 and Version 1.1 under the Environmental Offsets Regulation 2014 and <i>Environmental Offsets Act 2014</i>	To counterbalance significant residual impacts on matters of national, State or local environmental significance	Command and control, Economic	Offsets may be provided directly, through a third-party, or as a financial settlement	State of Queensland, 2014
	2014					

	2016	<i>Vegetation Management (Reinstatement) and Other Legislation Amendment Bill 2016</i>	To reduce deforestation rates and consequential carbon emissions	Command and control	If passed, the Bill would reinstate the <i>Vegetation Management Act 1999</i> as per the 2009 amendments. The protection of high value regrowth would be extended to three additional Great Barrier Reef catchments, and environmental offsetting would be required for all residual impacts on prescribed environmental matters rather than only significant residual impacts	State of Queensland, 2016
Victoria	2013	Permitted clearing of native vegetation – Biodiversity assessment guidelines (2013)	To improve and strengthen the regulatory system to deliver better outcomes for the environment and the community	Self-regulation, Command and control, Information	Replaces "Victoria's Native Vegetation – A Framework for Action" as incorporated document in the Victoria Planning Provisions (VPP). Permit required to clear vegetation only where there is a "high risk" to biodiversity	Department of Environment and Primary Industries, 2013a,b
South Australia	2013	<i>Native Vegetation (Miscellaneous) Amendment Act 2013</i>	To deliver more flexible arrangements for managing native vegetation	Command and control, Economic	Amendments to the NVA 1991 to introduce a the Third Party Significant Environmental Benefit Offsets Scheme	South Australian Government, 2013
Western Australia	2011	WA Environmental Offsets Policy	To ensure economic and social development may occur while supporting long term environmental and conservation values	Command and control, Economic	'Like for like' no longer required, offset must be 'proportionate' to the significance of the environmental value being impacted	Western Australian Government, 2011
	2013	Environmental Protection (Clearing of Native Vegetation) Regulations 2004 under the <i>Environmental Protection Act 1986</i>	To reduce unnecessary regulatory burden without compromising significant environmental values	Command and control	Amendment in 2013 allows landholders to clear up to 5 hectares per year on individual properties, and maintain cleared areas for pasture for up to 20 years without requiring a permit	Department of Environment Regulation, 2014

The beginnings of reform: 1970 - 1989

The beginnings of policy reform occurred in South Australia, which by 1975 had cleared 75% of the native vegetation within its agricultural zone (nearly 20% of the total State area; Australian Greenhouse Office 2000; Bartel 2004). The State's first policy effort in 1980 aimed to provide incentives for the retention of native vegetation on private property via legally binding agreements. Heritage Agreements were the first example of conservation covenants in Australia (Bartel, 2004), and provided financial assistance to cover land, fencing and management costs, but did not compensate for the opportunity costs agricultural production. Landholders who engaged with Heritage Agreements were those who held conservation-oriented values, whereas the policy did little to change the behaviour of landholders who were clearing extensively (Harris, 2013). Once it became clear that Heritage Agreements had failed to reduce deforestation, the State's first clearing controls were attached to the *Planning Act 1982*, which also included an extensive compensation scheme. In 1983, the requirement under the *Crown Lands Act 1929* to clear vegetation as a condition of lease was removed (Australian Greenhouse Office, 2000). With the introduction of the *Native Vegetation Management Act (NVMA) 1985*, South Australia became the first State in Australia to restrict the clearing of native vegetation on private property (Table 2.1, Bartel 2004). The introduction of the *Native Vegetation Act (NVA) 1991* removed the compensation arrangements which were part of the NVMA 1985. Clearing permits are granted under the NVA 1991 on the condition of vegetation being compensated elsewhere, making South Australia arguably the first State to adopt environmental offsetting. Despite being an early adopter of native vegetation policies, South Australia may be an example of a State where deforestation rates have declined simply because there was little land left to clear (Bartel 2004, Figure 2.4).

Early policies in Western Australia were focused on soil conservation and the control of salinity (Table 2.1, Australian Greenhouse Office 2000). Statewide controls on the rate and extent of clearing were introduced in 1986 under the *Soil and Land Conservation Act (SALCA) 1942* which required landholders to obtain a permit to clear 1ha or more of native vegetation (Slee, 1998). In Queensland, deforestation was still strongly encouraged by Government. Concerns were raised by scientists in the early 1980s about the extent of

vegetation loss in the Brigalow Belt regions and its impacts on biodiversity, but this had little effect on the rate of deforestation (Bailey, 1984; Seabrook et al., 2006).

In 1989, the Victorian Native Vegetation Retention (NVR) controls under the *Planning and Environment Act 1987* introduced the requirement that landholders acquire a permit to remove, destroy or lop native vegetation. This arguably marked the end to broad-scale clearing in Victoria (Department of Natural Resources and Environment, 2002).

High rates of loss: 1990 - 1999

In response to rising public concern about environmental degradation and biodiversity loss (Council of Australian Governments, 1992), several state governments initiated major policy reforms to control deforestation (Table 2.2). In 1995, the Queensland *Land Act 1994* was introduced to control vegetation clearing on leasehold and State land (Rolfe, 2000). At this time, clearing on freehold land was still regulated by local governments under the *Local Government Act (LGA) 1993* and the *Planning and Environment Act 1990* (Slee, 1998).

Controls were also implemented in 1995 in New South Wales, with the introduction of the *State Environmental Planning Policy no. 46* (SEPP 46). The SEPP 46 aimed to “prevent inappropriate native vegetation clearance and to ensure that native vegetation is managed in the environmental, social and economic interests of the State” (Bombell and Montoya, 2014; Slee, 1998). SEPP 46 was soon replaced by the *Native Vegetation Conservation Act (NVCA) 1997*, which came into force in 1998. Under the NVCA, landholders were required to gain approval to clear native vegetation (Productivity Commission, 2004). In 1997, the Victorian government announced a Statewide target of ‘no net loss’ of native vegetation by 2000 as part of the State’s biodiversity strategy (Department of Natural Resources and Environment, 1997).

Despite these reforms, deforestation rates remained high (Figure 2.3). In 1999, the Queensland and New South Wales governments permitted the clearing of over 730,000 hectares of native vegetation (Australian State of the Environment Committee, 2001; Lindenmayer, 2005).

A decade of reform: 2000 – 2009

The high rates of deforestation seen in Queensland continued well into the 2000s. Regulations on vegetation clearing on freehold land came into force under the *Vegetation Management Act (VMA) 1999* in 2000. However, the deforestation rate increased after the introduction of the VMA to 528,000 ha/year over 2001 to 2003 (Department of Natural Resources and Mines, 2005). It was not until 2006 that amendments to the VMA phased out broad-scale clearing of remnant vegetation. A moratorium on clearing applications in May 2003 signaled the Government's intention to end broad-scale clearing of vegetation by 2006 (Kehoe, 2009; McGrath, 2007). This policy change has been credited with the national drop in deforestation from 2007 onwards (ABARES, 2014). An offsets policy was released in 2007 to assist proponents in meeting requirements under the amended VMA, which was incorporated into a broader environmental offsets policy in the following year (Queensland Government, 2008). Further amendments to the VMA came into force in 2009 which created protection for 'high value' regrowth (vegetation not cleared since 31 December 1989) in 'priority' Great Barrier Reef catchments (Macintosh 2012) (Table 2.3), after a temporary moratorium earlier in that year.

In 2005, the New South Wales *Native Vegetation Act (NVA) 2003* came into force, which prohibited of clearing native vegetation unless it could be demonstrated that it would "improve or maintain environmental outcomes" (Gibbons and Lindenmayer, 2007). An offset policy was formalized in 2008 with the introduction of the BioBanking scheme, which aimed to create a market for vegetation offsets in New South Wales (Gibbons et al., 2009). The NVA 2003 has been credited with the dramatic decline in approved clearing in NSW after 2005 (Taylor and Dickman, 2014). However, exempted and illegal clearing likely still occurred at a high rate (Bricknell, 2010; Gibbons, 2012) although these statistics are not publicly reported (Taylor and Dickman, 2014). A statutory review of the NVA 2003 in 2009 concluded that the Act remained valid and that no fundamental changes were necessary, though some stakeholders expressed concerns about lack of flexibility in restrictions, policy overlap and complexity, and the level of Government enforcement (Bombell and Montoya, 2014; Department of Environment, Climate Change and Water NSW, 2009).

Victoria revised its Statewide ‘no net loss’ goal in 2003 with the introduction of the Victorian Native Vegetation Management Framework, which aimed to achieve a Statewide net gain in vegetation extent and quality (Department of Natural Resources and Environment, 2002). However, the objective for ‘permitted clearing’ on private land was still to achieve a ‘no net loss’ (Department of Sustainability and Environment, 2012). Subsequent evaluations have indicated that neither the Statewide or permitted clearing goals been met (Dart and Grossek, 2007; Department of Sustainability and Environment, 2008). Amendments in 2006 to the Victoria Planning Provisions (VPP) aimed to simplify the permitting process for local councils and to provide consistency across the State (Department of Sustainability and Environment, 2010). The BushTender and BushBroker programs were initiated in 2007 to provide landholders opportunities to sell and purchase vegetation credits, respectively (Nemes et al., 2008; O’Connor, 2009; Stoneham et al., 2003).

Reforms also occurred in Western Australia, with the amendment of the *Environmental Protection Act 1986* (WA) to provide stricter and more consistent controls for clearing native vegetation across the State (Squelch, 2007). The Western Australian government also formalized an environmental offset policy in 2006 after releasing several guidance and position statements in the preceding years (Hayes and Morrison-Saunders, 2007).

The decade of reform saw the introduction of significant controls on deforestation in Queensland and New South Wales, and ambitious commitments in Victoria. Primary deforestation was substantially reduced across the country (Figure 2.2), and many heralded this time as the end of land clearing in Australia (McGrath, 2007; Squelch, 2007; The Wilderness Society, 2007). However, landholders have generally been opposed to top-down regulation (Australian Greenhouse Office, 2000; Bartel and Barclay, 2011), and concerns about policy duplication, inconsistencies and inefficiencies became more prominent over time (Productivity Commission, 2004).

Contemporary policy responses: 2010 to 2016

While the previous decade was marked by increasingly tight restrictions on deforestation across Australia, policy responses from 2010 have followed a trend of relaxing these controls (Table 2.4). In 2011, the newly elected government of New South Wales announced a

statutory review into the Native Vegetation Regulation 2005 made under the *Native Vegetation Act 2003* (Stoner and Parker, 2013) in an effort to “strike the right balance between sustainable agriculture and protecting the environment”. Following the release of the review’s independent report in 2013 (Lane, 2013), the NSW government introduced self-assessable codes which permitted landholders to undertake “low impact clearing activities” such as clearing of paddock trees, removal of invasive native species and native vegetation thinning without requiring approval. Concerns about the relaxation of native vegetation policies were raised by the environmental sector (Taylor and Dickman, 2014), while the changes were reported as being generally welcomed by landholders (Condon and Bryant, 2013). A comprehensive review of the *NVA 2003* and related biodiversity policies was announced in mid-2014, and the final report released December 2014 (Byron et al., 2014). In their report, Byron and colleagues recommended the repeal of the *NVA 2003*, and combining native vegetation regulations with other biodiversity policies under a single *Biodiversity Conservation Act*. They also argued that the ‘improve or maintain’ test under the *NVA 2003* is “unnecessary and burdensome at the site scale”, and that offsite, third-party biodiversity offsetting should be applied to all environmental impacts (rather than only to threatened species and communities), along with increased investment in conservation on private and public lands. At the time of writing (May 2016), it appears that the reforms recommended by Byron and colleagues have yet to be drafted into legislation (Druce and Foley, 2015).

The Victorian government initiated a review of the Native Vegetation Management Framework in 2012, in an effort to improve regulatory performance through the reduction of “red tape” (Department of Sustainability and Environment, 2012). The reforms introduced in 2013 provided a risk-based approach to the regulation of vegetation clearing, whereby only “moderate” or “high” risk clearing required on-site assessment, and offsetting of ecological impacts (Department of Environment and Primary Industries, 2013). The Statewide goal for native vegetation was again revised, this time to “No net loss in the contribution made by native vegetation to Victoria’s biodiversity”.

Following a series of reviews from 2009 to 2011, Western Australia’s native vegetation regulations were amended in late 2013 (Department of Environment Regulation, 2014).

Landholders are now permitted to clear up to 5 hectares per year on individual properties, and maintain cleared areas for pasture for up to 20 years without requiring a permit. The report from a recent senate inquiry into the gazetting of environmentally sensitive areas (ESAs) in Western Australia argues that the State's native vegetation regulations are "confusing" and "complex"; and financially disadvantage landowners (Standing Committee on the Environment and Public Affairs, 2015).

Perhaps the most environmentally significant policy change since 2010 has occurred in Queensland, where the latest data indicate that 266,191 hectares of forest was cleared in 2013-14⁶. This is the highest deforestation rate recorded in Queensland since the end of broad scale clearing permits in 2006 (Bulinski et al., 2016; Department of Science, Information Technology and Innovation, 2015). Amendments to the *Vegetation Management Act 1999* in 2013 permitted landholders to clear remnant native vegetation to establish for 'high value agriculture', removed restrictions on clearing 'high value' regrowth, and removed the requirement to obtain a permit under the *Water Act 2000* to clear native vegetation in watercourses. Existing investigations into alleged non-compliance with the VMA were put on hold (Cripps, 2012). In response, prominent Queensland ecologists issued a public statement which argued against the removal of clearing restrictions (Maron, Catterall, et al., 2013), and the World Wildlife Fund has warned that Australia may again become a global hotspot for deforestation (Taylor, 2013, 2015; WWF International, 2015). In 2015, the newly elected Queensland Government promised to reinstate the provisions of the *Vegetation Management Act 1999* which were removed as part of the 2013 amendments by the previous government led by Premier Campbell Newman. At the time of writing (May 2016), the *Vegetation Management (Reinstatement) and Other Legislation Amendment Bill 2016* (State of Queensland, 2016) has not yet been passed by the Queensland Parliament, but is due to be reintroduced later in the year. In an effort to prevent a surge in deforestation prior to the passage of the tightened regulation (so-called 'panic clearing'), clearing restrictions would

⁶ Note that this estimate is provided by Queensland's SLATS program (Department of Science, Information Technology and Innovation, 2015), which considers a broader definition of 'forest' and has historically reported higher estimates of deforestation than the NCAS (Bulinski et al., 2016; Macintosh, 2012)

be applied retrospectively to 17th March 2016 – when the Bill was first introduced to Parliament (Chambers, 2016).

Policy trends

From the preceding discussion of native vegetation policy reform over the last 40 years, some trends emerge. Up until the late 1980s, policies aimed to restrict deforestation were primarily framed around soil conservation and salinity prevention, rather than the protection of native vegetation itself (Table 1, Slee, 1998). However, increasing public concern for the environment in the 1990s and the realization of substantial gains in public land conservation saw a shift in focus to regulating native vegetation primarily to reduce environmental degradation and biodiversity loss (Table 2.1). From 2000, regulation in most States became increasingly ‘command and control’, and the use of satellite imagery for monitoring and compliance more widespread (Bartel, 2005, 2008). Offsetting arrangements, either as complementary policies or as conditions of approved clearance, were in place within most States and Territories by the mid 2010’s (Table 2.3, Maron et al. 2015).

Over the decade of reform, there was a sense of optimism that Australia’s globally significant rates of deforestation had come to an end. However, within 10 years of what was celebrated as the end of broad-scale land clearing, major legislative changes have been made which relax clearing regulations. This new wave of policy reform is being mirrored in all of the high-deforestation States except South Australia, where only minor amendments have been made (Table 2.4). Although not clearly reflected in the NCAS data presented in this paper, the most recent data from the SLATS program in Queensland suggests there has been a sharp rise in deforestation since the government first signaled legislative changes (Department of Science, Information Technology and Innovation, 2015; Queensland Audit Office, 2015). In the absence of a robust quantitative evaluation, it is not yet clear whether deforestation rates have significantly changed following other recent policy changes in New South Wales, Victoria and Western Australia.

The relaxation of State-level native vegetation policies from 2010 has marked a shift in emphasis from ‘command and control’, to voluntary compliance and self-regulation. This change has occurred in parallel with Federal level efforts to reduce “red tape” in

environmental approvals under the EPBC Act (Australian Government, 2014b; Standing Committee on the Environment, 2014), the “opening up” (and subsequent re-closing) of National Parks to cattle grazing (Beilharz and Taylor, 2015; Tlozek, 2015), as well as possible amendments to the EPBC Act to redress what is perceived by some as an imbalance between environmental protection and economic opportunity (Senate Legislation Committee Environment and Communications, 2015). The most recent announcement by the Queensland Government to revert back to ‘command and control’ regulation may suggest that the days of voluntary compliance and self-regulation are numbered (State of Queensland, 2016). However, no other state governments have so far indicated any intention to reinstate strict regulatory controls on deforestation. In no other state has such a significant increase in deforestation occurred over 2012-2014 as it has in Queensland, which has resulted in the release of carbon emissions almost equivalent to the amount secured through the Australian Government’s Emissions Reduction Fund (Bulinski et al., 2016). The scale of deforestation and its contribution to climate change has provided the Queensland Government a policy platform to reinstate the *Vegetation Management Act 1999* in its previous form, with the intent to reduce greenhouse gas emissions and agricultural run-off into the Great Barrier Reef (State of Queensland, 2016).

It is important to consider the broader macroeconomic environment when discussing trends in deforestation and policy responses over time (Figure 2.5). The drivers of deforestation are highly context specific, and cannot be easily generalized (Geist and Lambin, 2002). Many of the factors described by the international literature on deforestation, such as population growth, access to roads and shifting cultivation, are not relevant in Australia (Australian Greenhouse Office, 2000; Lindenmayer, 2005). Angelsen & Kaimowitz (1999) emphasises macroeconomic variables and policy instruments as key ultimate drivers of deforestation. Importantly, and rarely discussed in the literature, is the availability of suitable land which ultimately limits the amount of primary forest which can be cleared (Australian Greenhouse Office, 2000; Bartel, 2004).

Rainfall, commodity prices and terms of trade are widely known to influence landholder clearing decisions (Australian Government, 2013; Macintosh, 2012; Rolfe, 2002). The effects of rainfall are complex however (Figure 2.5c), as deforestation may be driven by high

rainfall as well as drought conditions – the latter due to the increased production required to be profitable (Australian Greenhouse Office 2000). The relationship between deforestation rates and farmer's terms of trade has been used to estimate historical clearing from 1940 to 1970 (Commonwealth of Australia, 2014), as well as to predict deforestation rates up to 2030 (Australian Government 2013). The Australian economy has undergone a restructure over the last several decades, leading to an increased contribution of the mining sector to economic growth, and unfavourable economic conditions for the agricultural sector (Connolly and Lewis, 2010; Corden, 2012; Gregory, 1976).

When considered in the context of these broader policy trends and the decline of the agricultural sector (Figure 2.5), the recent relaxation of native vegetation policies is not altogether surprising. 'Command and control' regulation is deeply unpopular amongst many rural landholders (Australian Greenhouse Office, 2000; Bartel and Barclay, 2011), who had historically held the right to clear vegetation without restriction, and indeed had been encouraged by Government to do so. Perceived and real impacts on farm productivity, inequitable impacts on landholders, and a large distance between the values and norms held by landholders and that of the Government and its policies mean that at least at the present time, strict regulations on deforestation are politically unpalatable (Bricknell, 2010; Chambers, 2016; Productivity Commission, 2004).

Future prospects for native vegetation policy in Australia

It is not yet apparent whether the current trend in deregulation will continue, or if it is simply a temporary pushback in the context of a long term trend of clearing decline, worsening economic conditions, and the increasing scarcity of primary forest available to clear (Figure 2.5). As highlighted by the previous section, relying too heavily on regulation can be politically costly, and may ultimately lead to policy failure (Bartel and Barclay, 2011). Acceptance and compliance with native vegetation policies has proven to be extremely difficult to achieve in Australia (Bartel, 2003; Bricknell, 2010).

A key recommendation made within recent reviews of State-level native vegetation policy is the need to consider incentive-based and educational policies in addition to regulatory enforcement, in order achieve positive environmental, social and economic outcomes (DSE

2012, Byron 2014). Arguments in favour of using a diversity of instruments to meet environmental policy goals are not new (Bartel, 2008; Bricknell, 2010; Commonwealth of Australia, 2009; Dovers and Hussey, 2013), but the strengths and weaknesses of all policy options must be clearly considered (Gunningham and Sinclair, 1999). The recent increased emphasis on policies such as biodiversity offsetting, private conservation agreements and carbon farming is analysed below.

Biodiversity offsetting

Biodiversity offsetting has been increasingly emphasized as an approach which can deliver environmental outcomes in a more flexible and efficient manner (Chapters Five to Seven, Byron et al., 2014; Commonwealth of Australia, 2009), and policies are now in place at the Federal and State levels. While offsetting can provide efficiencies over regulatory approaches, generally it can only maintain existing trajectories of deforestation and biodiversity loss, rather than slow or reverse the decline (Maron et al. 2015). As highlighted by Maron and colleagues, all Australian offset policies aim to achieve a ‘no net loss’ of biodiversity relative to a business-as-usual scenario. In fact, most policies assume a background rate of loss that is far higher than the existing rate of deforestation, meaning that offset policies have the potential to exacerbate biodiversity loss. This issue is one of a range of perverse outcomes that can occur as a result of widespread adoption of biodiversity offsetting (Gordon et al., 2015), hence regulation will still be necessary if deforestation is to be reduced or reversed. Indeed, regulation effectively sets the ‘cap’ on permitted environmental impacts, and thus is required to create the demand for a functioning environmental market (Salzman and Ruhl, 2000).

Private conservation agreements

The importance of providing incentives to protect native vegetation, wildlife and associated ecosystem services on private land is also regularly highlighted by commentators (Byron et al., 2014; Commonwealth of Australia, 2009; Fitzsimons, 2015; Hardy et al., 2016). Private land stewardship is a critical component of Australia’s biodiversity conservation efforts, given that the majority (74%) of the continent is freehold, leasehold or under Indigenous management (Geoscience Australia, 1993). It should be made clear, however, that increasing

the area of land privately (or indeed, publicly) managed for conservation does little to reduce the overall deforestation rate if they do not prevent the loss of forest (Maron et al., 2013; McDonald-Madden et al., 2009). As was the case in the early South Australian Heritage Agreements, landholders who enter into voluntary conservation agreements are generally already sympathetic to nature conservation, and the incentives provided are not enough to change land use decisions at a large scale. Landholders whose values do not align with conservation are not likely to change land practices unless it is economically profitable to do so – and even then, social and cultural norms can provide an additional barrier to participation (Bartel and Barclay 2011). As with other incentive-based programs, private conservation agreements are usually small-scale, prone to adverse selection (Ferraro, 2008), and subject to short-term funding cycles (Senate Environment and Communications References Committee, 2015). The efficacy of private conservation areas can also be compromised where land use conflicts are not resolved (Adams and Moon, 2013).

Carbon farming

Carbon farming also offers potential benefits for native vegetation protection and restoration, assuming a there is a market price on carbon emissions (Chapter Four, Crossman et al., 2011; Evans et al., 2015; Lin et al., 2013) and perverse impacts on biodiversity are avoided (Lindenmayer et al., 2012). Similar to private conservation agreements, factors such as high transaction costs, policy complexity and cultural norms can act as barriers to landholder participation in carbon farming projects (Macintosh, 2013). A key difference is that carbon farming can be more profitable than existing agricultural land uses, particularly in marginal areas where significant economies of scale exist (Evans et al., 2015). While unlikely to be influenced by reforestation and afforestation projects, the rate of deforestation can be reduced where genuine avoided loss can be secured. As with other incentive-based schemes, carbon farming can only genuinely prevent or reverse forest loss if regulatory controls on deforestation exist. At present, the Australian Government's carbon farming policy (Australian Government, 2014a) provides incentives for landholders to undertake avoided deforestation and reforestation, while State-level native vegetation policies have all recently been relaxed. This inconsistency in policy approach means that the environmental benefits generated by the Federal policy have largely been negated by recent increased deforestation

(Bulinski et al., 2016; Department of Science, Information Technology and Innovation, 2015) and creates significant policy uncertainty for landholders (Elks, 2016).

The need for an effective policy mix

Incentive-based policies such as those outlined above are attractive as they can afford flexibility and efficiencies that traditional regulation cannot provide. Although it is sensible to consider the potential benefits offered by a range of policy instruments, there can be a temptation to recommend them as alternatives, rather than complements to regulation, or without a clear assessment of their likely efficacy (Gunningham and Grabosky, 1998). A combination of ‘command and control’ regulation, self-regulation, incentive-based and educational instruments will generally perform better than any single instrument in meeting a policy objective (Dovers and Hussey, 2013; Gunningham and Sinclair, 1999). Based on the most recent deforestation trends and the history of native vegetation policy in Australia, it appears that a coordinated and mutually supportive policy mix has yet to be achieved with respect to effectiveness, equity and social and political feasibility.

Very little is actually known of the effectiveness of the various policy responses to deforestation over the last 40 years. Few government-sponsored evaluations are available (but see: Dart and Grossek, 2007; Department of Sustainability and Environment, 2008), and available data are often inadequate to conduct a rigorous evaluation (Byron et al., 2014). Environmental policies are notoriously difficult to evaluate, as environmental problems are generally complex, involve considerable uncertainties, and require detailed measurements and specialist methods to attribute a policy intervention to an observed response (Ferraro, 2009; Keene and Pullin, 2011; Mickwitz, 2003). The efficacy of policy responses to deforestation can only be reliably evaluated by considering observed deforestation rates (including regulated, exempted and illegal clearing), in addition to the other drivers of land management behaviour (Bartel 2004).

While it is recognized that macroeconomic, environmental and institutional arrangements all have an effect on deforestation rates (Angelsen and Kaimowitz, 1999; Geist and Lambin, 2002), how these variables interact and ultimately drive deforestation in Australia is poorly understood. Nonetheless, the reduction in deforestation since the 1990s has been attributed

to government intervention in several instances (Australian Government, 2013; Department of Environment and Resource Management, 2010; Garnaut, 2008). Macintosh (2012) argues that such suggestions are misleading, or at least incomplete, without explicitly considering the effects of commodity prices, terms of trade and rainfall on deforestation rates. A comprehensive evaluation of the impact of native vegetation policies on deforestation in Australia over time is needed, but this would require adequate data, appropriate methods and a willingness by relevant stakeholders to conduct such an analysis.

The lack of clear evidence for the historical effectiveness of Australia's native vegetation policies is extremely problematic, given the time and effort devoted to their design, implementation and review. Despite the introduction of a raft of policies aimed to reduce deforestation over the last 40 years, monitoring, evaluation and enforcement have been hampered by a lack of resources and information (Bartel, 2003; Bricknell, 2010; Nicol et al., 2014). The advent of satellite imagery was at one stage heralded as a new beginning that would enable greater monitoring and evaluation, and encourage compliance with clearing regulations (Bartel, 2005; Purdy, 2009). We have, however, yet to see a revolution in our understanding of native vegetation policy effectiveness in Australia. A key step required to deliver a more effective policy mix for addressing deforestation is therefore to invest a greater proportion of resources to monitoring, evaluation and compliance.

Conclusions

Native vegetation policy has been an extraordinarily active policy space in Australia over the last 40 years. Initially motivated by concerns around soil conservation and salinity, a growing interest around biodiversity conservation and ecologically sustainable development drove a wave of policy reforms over the 1990's and 2000's, which placed strict regulations on deforestation. An interest in providing landholders with flexibility and economic incentives to retain and restore vegetation saw the proliferation of offset policies from 2000 onwards. Since 2010, several Australian States have amended their native vegetation policies to place greater emphasis on self-regulation and voluntary compliance, in an effort to restore 'balance' between meeting environmental, social and economic objectives. The most recent

increase in deforestation in Queensland has triggered a potential shift back to ‘command and control’, but at present it does not appear that the other States will soon follow suit.

Other than in Queensland, it is not yet clear whether this shift to self-regulation has preceded an increase in deforestation. Deforestation results as a combination of institutional, macroeconomic and environmental factors; hence a change in the rate of deforestation cannot be attributed to any particular event without a rigorous evaluation. The long-term trend in Australia over 1972-2014 is of a gradual decline in the rate of deforestation relative to the amount of primary forest available to clear on suitable land. Faced with worsening economic conditions and the expansion of agriculture into increasingly marginal areas, deforestation for agricultural, urban and industrial development will likely cease being economically viable before all of the remaining primary forest is cleared. However, the raft of policies implemented over the last 40 years illustrates that there is a desire in the Australian community to limit deforestation for a range of environmental objectives. To be effective, native vegetation policies therefore need to induce a ‘forest transition’ before deforestation meets its economic and environmental limits (Angelsen and Kaimowitz, 1999; Lambin and Meyfroidt, 2011; Rudel et al., 2005). Ultimately, Australia has means to achieve this goal – it is a question of whether it is socially and politically feasible.

Environmental policy is made in the context of broader socio-political and economic trends. The recent shift towards self-regulation, flexibility and economic instruments reflects these broader societal trends – but this shift in focus on policy instrument *type* does not necessarily mean there will be a change in policy *effectiveness*. All environmental policy instruments, regardless of whether they are ‘command and control’, self-regulation, economic or informational require monitoring, evaluation and enforcement if they are to be effective (Gunningham and Sinclair, 1999). Historically, these crucial steps in the policy process have been poorly executed with respect to Australia’s native vegetation and biodiversity (Bartel 2003, Bricknell 2010). Ensuring that there is far greater capacity to monitor and evaluate the impacts of native vegetation policies will assist in delivering more effective, efficient and equitable outcomes.

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CHAPTER THREE:

Quantitative evaluation of regulatory policies for reducing deforestation in Queensland, Australia

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Abstract

Reducing and redressing the effects of deforestation is a complex public policy challenge, and evaluating the efficacy of such policy efforts is crucial for policy learning and adaptation. Deforestation in high-income nations can contribute substantially to global forest loss, despite the presence of strong institutions and high policy capacity. Here we employ a nonlinear mixed-effects statistical framework to evaluate what evidence may exist for the impact of recent successive regulatory policies aimed at reducing deforestation in Queensland, Australia. Over 5.5 million hectares of native vegetation has been cleared in Queensland since 1988, amounting to more than half of Australia's historical deforestation. We combine satellite imagery of forest loss with macroeconomic, land tenure, biophysical and climatic variables within a single spatially explicit bent-cable regression, and collectively model deforestation for 50 local government areas (LGAs) across Queensland. We find that annual % growth in GDP was the only clear driver of LGA-specific deforestation after adjusting for other covariate effects. Our model shows strong evidence of spatial contagion in deforestation, and this effect is influenced by the dominant land tenure type within each LGA. We find our model exhibits a "bend" mostly between 2000 and 2007, consistent with expectations, but the signal is not particularly strong due to extreme variation in deforestation trends between and within LGAs. Our results demonstrate that the bent-cable model is a promising technique for detecting system changes in response to policy interventions, but future work should be conducted at a national scale to provide more data points, and incorporate more LGA-specific data to improve model goodness-of-fit.

Highlights

- Deforestation can be globally significant in high-income nations with strong governance
- Over 5.5 million hectares of forest has been cleared in Queensland, Australia, since 1988
- A spatially explicit bent-cable regression was used to model deforestation
- Strong evidence for spatial contagion in deforestation influenced by LGA land tenure
- Some evidence of state-wide policy effect due to extreme variation in LGA deforestation responses

Key words: deforestation, regulation, environmental policy, bent-cable regression, Bayesian inference, hierarchical modeling, longitudinal data, spatial correlation

Introduction

Effective control of deforestation is crucial to ensure sustainable provision of ecosystem services and the conservation of biodiversity (Kirby et al., 2006; Lambin et al., 2001; Vitousek et al., 1997). Public policies such as protected areas (Andam et al., 2008; Ferraro et al., 2013; Gaveau et al., 2009; Joppa and Pfaff, 2011; Nolte et al., 2013), regulations and market intervention (Arima et al., 2014; Busch et al., 2015; Gaveau et al., 2012; Hargrave and Kis-Katos, 2013; Nepstad et al., 2014; Soares-Filho et al., 2014) and payments for ecosystem services (Arriagada et al., 2012; Liu et al., 2008; Sánchez-Azofeifa et al., 2007) can contribute to a reduction in forest loss. Not all policies have proved effective however, with some producing perverse outcomes (Brandt et al., 2016; Meyfroidt and Lambin, 2009) and many more policy interventions having not been evaluated for their efficacy (Ferraro, 2009; Ferraro and Hanauer, 2014; Miteva et al., 2012). An understanding of the efficacy of policy measures taken to control deforestation, and the varied institutional, social and political conditions in which they are adopted, implemented and enforced (Ferraro et al., 2011; Nolte et al., 2017), is critical for policy learning and adaptation.

Disentangling the effects of policy interventions from the varied drivers of deforestation is a complex evaluation challenge. Broader macroeconomic trends and policy drivers ultimately influence local market conditions and institutional settings, which in turn affect the deforestation behavior of agents (Angelsen, 2010; Angelsen and Kaimowitz, 1999; Geist and Lambin, 2002). The availability of arable land and forest resources also influences how much deforestation occurs locally, and how much is displaced elsewhere (Bartel, 2004; Evans, 2016; Meyfroidt and Lambin, 2009). How these variables interact and influence deforestation in a particular context cannot easily be generalized (Geist and Lambin, 2002), hence there is a need to evaluate the efficacy of a broad range of instruments across multiple locations and policy settings (Miteva et al., 2012).

Much of the published work on deforestation drivers and policy interventions has been in the tropics, perhaps due to the high rates of deforestation observed in this region (Hansen et al., 2013; Keenan et al., 2015), and the global significance of tropical forests for biodiversity, livelihoods, climate regulation and food production (Laurance, 1999; Laurance et al., 2012, 2014). Population growth (DeFries et al., 2010; Jha and Bawa, 2006), roads and access to markets (Kirby et al., 2006; Laurance, 2001), agricultural commodity prices and currency exchange rates (Barbier and Rauscher, 1994; Ewers et al., 2008; Richards et al., 2012), presence and strength of institutions

(Dolisca et al., 2007; Fearnside, 2001; Messina et al., 2006; Nolte et al., 2013; Robinson et al., 2014) and degree of policy enforcement (Arima et al., 2014; le Polain de Waroux et al., 2016) have all been found to influence tropical deforestation. However, less attention has been paid to deforestation drivers and policy responses in high income nations (Busch and Ferretti-Gallon, 2017), which have contributed substantially to global forest loss (Hansen et al., 2010, 2013) despite having comparatively strong governance (Bhattarai and Hammig, 2004; Deacon, 1994). Australia is one such example, where recent deforestation rates have been globally significant (Evans, 2016; FAO, 2001; Keenan et al., 2015; Macintosh, 2012), yet thus far there have been no quantitative evaluation of the effectiveness of policies introduced to reduce forest loss.

Here, we conduct such an evaluation to establish what, if any, effect the introduction of regulatory policies have had on the rate of deforestation in the state of Queensland, Australia. Forest loss in Australia at the turn of the century was the sixth highest in the world (FAO, 2001), and the vast majority (58%) of clearing over the last three decades has occurred in Queensland. Successive policies have been introduced in Queensland since 1995 in an attempt to control deforestation (Evans, 2016; Macintosh, 2012; Rolfe, 2000), with amendments in 2007 said to have signaled the “end of broad-scale land clearing” in Australia (Kehoe, 2009; McGrath, 2007). The national downturn in forest loss since the 1990’s has been attributed to state-level regulations on native vegetation clearing (Australian Government, 2013; Department of Environment and Resource Management, 2010; Garnaut, 2008), but with limited empirical basis (Macintosh, 2012). Here, we adapt the bent-cable regression model (Chiu et al., 2006; Chiu and Lockhart, 2010; Khan et al., 2009) to determine the impact of regulatory policies on deforestation in Queensland relative to macroeconomic, institutional, biophysical and climatic drivers. The bent-cable model generalizes the broken-stick (piecewise-linear) model by providing a more realistic and flexible representation of system changes (Chiu et al., 2006; Chiu and Lockhart, 2010; Toms and Lesperance, 2003), yet has so far been under-utilised for policy evaluation (but see Khan et al., 2009, 2012). In particular, suppose the policy introduced at time t unequivocally results in a reduction in deforestation. Then, even after adjusting for driver effects, the temporal trend in deforestation must exhibit a higher rate of decline sometime after t than the rate before time t . In reality, any observable impact of an environmental policy is rarely unequivocal. In such cases, the bend of an empirically fitted bent

cable would provide quantitative clues to the presence and nature of a change in deforestation rate before and after the introduction of policies.

In the remainder of this paper, we first document recent deforestation rates in Queensland, and describe how historical and present-date drivers have contributed to forest loss over time. Second, we briefly describe the key policy changes that have occurred in our study region in the past three decades, and highlight which interventions we expect to have influenced the rate of deforestation beyond the effects of macroeconomic, climatic, biophysical and institutional variables. Third, we provide details on the data used to model deforestation over time, and justify our selection of covariates used in the statistical analysis. We then outline our bent-cable regression model specification, and report on the results of our analysis. Finally, we discuss the implications of our findings for the design and implementation of policies aiming to control deforestation in Australia and other forested nations which rely on agriculture and other commodity exports for economic development.

Background

Study region and policy context

Queensland is Australia's second-largest state, covering 170 million hectares (23% of the continent). Close to 10 million hectares of forest has been cleared since 1988, of which 31% was primary (remnant) forest (Figure 3.1) Agriculture is the primary land use, with extensive grazing making up 85% of the state by area, and cropping and other agricultural industries (excluding forestry) comprising just 2%. Beef is the state's most important agricultural commodity, contributing AUD 3.3 billion in 2013–14 (Department of Agriculture, Fisheries and Forestry, 2014) which is close to half of Australia's total production. The majority of land is leasehold (63%), with only 25% privately owned (freehold). Leasehold land in Queensland may be held in perpetuity or for a fixed term (1-100 years), with tenure issued for a specific purpose (e.g. agriculture). Deforestation occurs disproportionately on freehold land in Queensland (Department of Science, Information Technology and Innovation, 2015) and in Australia as a whole (Evans, 2016).

Queensland holds the largest remaining area of forest Australia, and the availability of cheap land suitable for agriculture has been key historical driver of deforestation (Australian Greenhouse

Office, 2000; Evans, 2016). Native vegetation such as Brigalow (*Acacia harpophylla*) and Mulga (*Acacia aneura*) grow vigorously, and are generally re-cleared in 15 year cycles to maintain suitable pasture (Dwyer et al., 2009; Fensham and Guymer, 2009). Government land development schemes, access to cheap finance and tax concessions facilitated early agricultural expansion, but most of these incentives were removed by the 1980's (Australian Greenhouse Office, 2000; Seabrook et al., 2006). Fluctuations in commodity prices, terms of trade, rainfall and regulatory controls on clearing are considered the key contemporary drivers of deforestation in Australia (Australian Government, 2013; Australian Greenhouse Office, 2000; Evans, 2016; Marcos-Martinez et al., 2017).

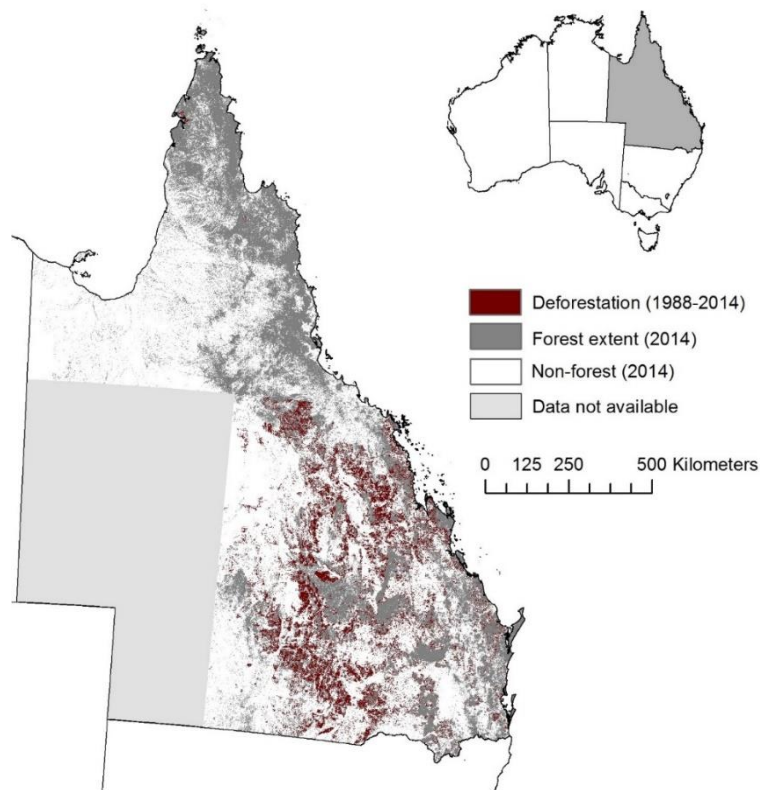


Figure 3.1. Deforestation events (including primary and regrowth deforestation) attributed to human intervention from 1988 – 2014. Data is sourced from the National Carbon Accounting System (NCAS), Australian Department of the Environment (2015, 2016).

Deforestation in Queensland has occurred at an average rate of 204,000 hectares per year since 1988 (Figure 3.2). State-wide regulation of native vegetation clearing on leasehold land was first introduced under the *Land Act 1994* (Figure 3.2 (a), Evans, 2016; Rolfe, 2000), whereas similar

controls on freehold properties were enacted five years later under the *Vegetation Management Act (VMA) 1999* (Figure 3.2 (b), Evans, 2016). The upturn in deforestation in 2000 has been attributed to “panic clearing” (McGrath, 2007), where landholders cleared substantial amounts of vegetation prior to the VMA coming into effect in an effort to avoid regulation. In 2004, a ballot was held for clearing permits totaling 500,000 ha which were all used or expired by 31 December 2006. This policy change is considered to have effectively reduced deforestation in Queensland to its lowest level in the last three decades. The relaxation of the VMA’s regulatory controls in 2013, and the subsequent rapid increase in clearing rates (Reside et al., 2017) has added further weight to this claim.

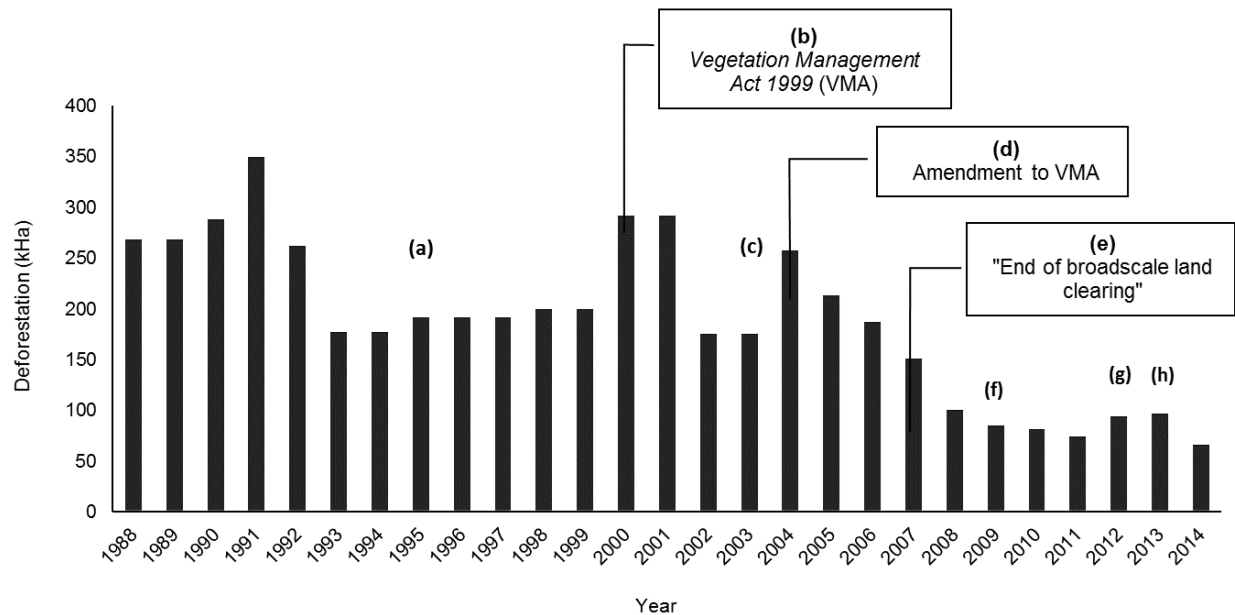


Figure 3.2. Annual rate of deforestation in Queensland from 1988-2014, and the introduction of key legislations which regulate the clearing of native vegetation. The central piece of legislation is the *Vegetation Management Act 1999* (b), which came into force in 2000 to regulate vegetation clearing on freehold land. Amendments to the VMA in 2004 (d) were said to led to the end of broad-scale land clearing by on 31st December 2006 (e). Other legislative changes corresponding to (a), (c) and (f) to (h) are described in the Supplementary Material (Appendix 2.1, Table A2.1). Data is sourced from the National Carbon Accounting System (NCAS), Australian Department of the Environment (2015, 2016).

Materials and Methods

Deforestation data

We used national-scale spatial datasets on forest extent and human induced forest change (Australian Department of the Environment, 2015, 2016) developed by the Australian Government as part of the National Carbon Accounting System (Evans, 2016; NCAS, Furby, 2002; Lehmann et al., 2013). The NCAS uses over 7,000 Landsat MSS, TM and ETM+ images to map forest extent and change from 1972 to 2014 at a 25 metre resolution across the Australian continent. Note that the NCAS classifies imagery according to a definition of ‘forest’ as vegetation with a minimum of 20% canopy cover, at least 2 metres high, and with a minimum area of 0.2 hectares.

Annual forest extent and deforestation data are not available within the NCAS until 2005. Prior to then, data are instead captured within multi-year epochs (instances in time), with some epochs (e.g 1972) containing data for five consecutive years (Australian Department of the Environment, 2015, 2016; Evans, 2016). We converted deforestation events contained within multi-year epochs into annual values by dividing the deforestation that occurred within that epoch by the number of years within the epoch (Australian Department of the Environment, 2015; Reddy, S, pers. comm.). For example, the 1972 epoch contains deforestation events for 1972 to 1977, hence the annual amount of deforestation in those years is the deforestation occurring in the 1972 epoch divided by five. To derive annual forest extent values, we simply assumed that the forest extent value in a multi-year epoch was equivalent to the corresponding annual values. For example, the area of forest extent in the 1988 epoch was assigned as the annual forest extent value for 1988 and 1989.

Preliminary modelling results suggested that an unreasonable amount of uncertainty in the model estimates was a consequence of (a) the lack of real observed data (as opposed to artificially imputed data) prior to 1988 and (b) the ambiguity in the definition of primary (remnant) forest extent. Therefore, for our final models, we discarded sparse data early in the time series by considering all deforestation events and forest extent (Australian Department of the Environment, 2015, 2016) from 1988 to 2014 only. By “all deforestation events and forest extent” we did not differentiate between primary and regrowth forest as per Evans (2016).

Using a national land use dataset (ABARES, 2010), we excluded protected areas and areas of commercial forestry from the analysis, and so considered deforestation only where the land use

was for residential and urban development, agriculture, grazing and mining. local government Areas (LGAs) were selected as spatial units of analysis due to their relatively small size and alignment with catchment boundaries, which have previously been used spatially differentiate native vegetation regulations in Queensland (Evans, 2016). State-level native vegetation legislation does not prevent a local planning restrictions on deforestation in an LGA, however the State law prevails in the event of inconsistencies (State of Queensland, 1999). We used the ‘raster’ package (Hijmans and van Etten, 2012) in R Statistical software (R Development Core Team, 2017) to summarise forest extent and deforestation data at the LGA level.

Macroeconomic data

We extracted key national level indicators (Table 3.1) including the annual percentage growth in gross domestic product (GDP, USD in real terms), the value added by agriculture to GDP, and terms of trade (The World Bank, 2015). Oil supply constraints are associated with changes in the rate of deforestation (Eisner et al., 2016), hence we considered the annual imported crude oil price (USD/barrel, U.S. Energy Information Administration, 2015). We used national scale agricultural commodity statistics (ABARES, 2015) as local surveys are not conducted annually.

Climate data

Rainfall is a key driver of rural land use decisions (Macintosh, 2012; Rolfe, 2002), and is expected to affect deforestation rates independently of broader macroeconomic variables (Australian Government, 2013). Queensland regularly experiences periods of drought, and these regional climatic extremes can influence observed deforestation rates due to changes in the availability and quality of fodder (Australian Greenhouse Office, 2000). Spatiotemporal variation in temperature and rainfall is also linked to the relative suitability of land for grazing and cropping (Marcos-Martinez et al., 2017). We obtained spatial data for key climatic variables (Table 3.1) from the Australian Bureau of Meteorology (Jones et al., 2009). Rainfall, vapour pressure (‘humidity’) and temperature were sourced as monthly averages (Bureau of Meteorology, 2017a, 2017c, 2017b). Quantities were calculated within each LGA polygon for each year from 1988 to 2014 using the Python Rasterstats package (Perry, 2015).

Biophysical data

We derived slope and elevation data from a 1 second Digital Surface Model (Geoscience Australia, 2017) to account for the influence of topography on agricultural productivity and deforestation (Angelsen and Kaimowitz, 1999; Marcos-Martinez et al., 2017; Pressey et al., 1996). Spatial variation in vegetation productivity was accounted for using the normalized difference vegetation index (Tucker et al., 2014). The mean, median and standard deviation was calculated for each LGA using the Rasterstats package (Perry, 2015).

Land tenure

We used the most recent national land tenure dataset (Geoscience Australia, 1993) to determine the proportion of each LGA held in leasehold, freehold or public tenure. We derived a spatial covariate L_i (Figure 3.3) to represent the extent to which an LGA i is held in leasehold or freehold tenure, according to:

$$L_i = \log \frac{(\% \text{ freehold} + 0.01)}{(\% \text{ leasehold} + 0.01)}$$

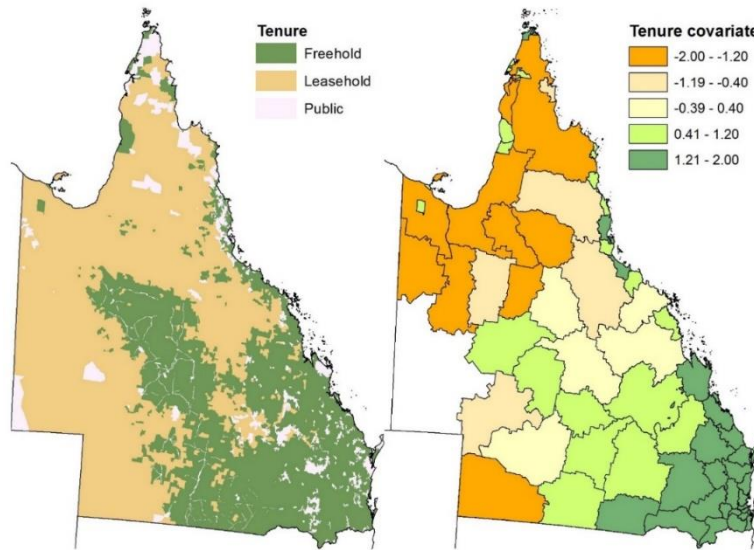


Figure 3.3. Land tenure in Queensland (left, Geoscience Australia, 2013) and corresponding tenure covariate L_i calculated for each local government areas (LGAs, $n = 74$ (total), $n = 50$ (included in regression analysis))

Table 3.1 Key variables considered in our regression analysis during covariate selection, and the covariates selected for inclusion in our final model. Details of the covariate selection process is provided in the Appendix 2.2.

Name	Type	Description	Spatial resolution and units		Temporal resolution		Contained in final model?	Data source	
			Original	Final	Original	Final			
Response variable									
Area of deforestation	Environmental	Extent of total forest loss attributed to human intervention	0.0002 degrees; 25m	Hectares, calculated for each LGA	Available for epochs: 1972, 1977, 1980, 1985, 1989, 1991, 1992, 1995, 1998, 2000, 2002, 2004, annually from 2005 to 2014	1988 to 2014 (annual)	-	Australian Department of the Environment (2015)	
Area of forest extent		Extent of vegetation classified as 'forest' (excluding protected areas and commercial forestry)					-	Australian Department of the Environment (2016)	
Proportion of deforestation (y_{it})		Area of deforestation relative to the area of forest available to clear in LGA I and year t					LGA; %	Yes	Derived
Explanatory variables									
Rainfall	Climatic	Total annual rainfall		mm, calculated for each LGA		1988 to 2014 (annual)	-	Bureau of Meteorology (2017a)	
		Minimum of mean monthly minimum observations							
		Maximum of mean monthly minimum observations							
Vapour pressure ('humidity')		Mean of mean monthly 9am estimates	0.05 degrees; 5km	hPa, calculated for each LGA	1900 to current	1989 to 2014 (annual)	Yes	Bureau of Meteorology (2017c)	
Temperature		Mean of mean monthly minimum observations					-		
		Minimum of mean monthly minimum observations		degrees celsius, calculated for each LGA		1990 to 2014 (annual)	-	Bureau of Meteorology (2017b)	
		Mean of mean monthly maximum observations					-		
		Maximum of mean monthly maximum observations					-		
Land tenure	Institutional	Extent of land tenures across Australia	Polygon	25m raster	-	-	-	Geoscience Australia (1993)	

Chapter Three: Quantitative evaluation of regulatory policies to reduce deforestation

Tenure covariate (L_i)		Log ratio of % freehold to % leasehold in each LGA	Index from -2 to 2. Calculated for each LGA				Spatial adjacency only	Derived
Name	Type	Description	Spatial resolution and units		Temporal resolution		Contained in final model?	Data source
			Original	Final	Original	Final		
<i>Explanatory variables</i>								
Slope	Biophysical	Mean, median, minimum, maximum	1 second (approx 30m)	m, calculated for each LGA	-	-	-	Geoscience Australia (2017)
Elevation		Mean of long term mean (estimate of productivity)					Yes (mean)	
NDVI		Standard deviation of long term mean (estimate of land cover variability)	10km	Calculated for each LGA	1981 to 2011	-	-	Tucker et al. (2014)
		Mean of long term standard deviation (estimate of production variability over time)						
% GDP growth (annual)	Macroeconomic	% growth in the value of all goods and services produced in a given year	National, %				Yes	
% value added to GDP by agriculture		Net output of agriculture after adding up all outputs and subtracting intermediate inputs	National, % of GDP		1972 to 2014		-	The World Bank (2015)
Inflation		Rate of price change in the economy as a whole	National, %					
Oil price		Annual average imported crude oil price	International, USD/barrel		1968 to 2015	1988 to 2014 (annual)	Yes	U.S. Energy Information Administration (2015)
Terms of trade (ToT)		Ratio of total export prices to import prices	National					
Farmer's terms of trade (FToT)		Ratio of prices received by farmers to prices paid by farmers	National		1972 to 2015		-	ABARES (2015)
Gross value of agricultural exports		Gross value of farm production, value of cereal exports, meat exports, wool exports, total exports	National, AUD					

Model specification

We developed a holistic model of deforestation trends in Queensland to model LGAs ($n = 74$) as the spatial units of analysis. Forest data were not available in 6 LGAs in the western part of the state, and we discarded a further 18 LGAs due to insufficient data (Appendix 2.3, Table A3.1), leaving 50 LGAs remaining to be analysed.

Our response variable y_{it} is calculated as the relative proportion of deforestation in LGA i at time step t , then log-transformed twice to reduce skewness:

$$y_{it} = \log \left\{ -\log \frac{(\text{Area of deforestation})_{it}}{(\text{Area of forest extent})_{it}} \right\}$$

The total forest extent in LGA i may decrease, increase or remain unchanged between sequential years, since the NCAS captures forest as it is cleared, regrown and re-cleared. This means that y_{it} similarly may decrease, increase or remain unchanged between time steps. Our specification of y_{it} controls for the area of forest available to clear (Australian Greenhouse Office, 2000; Evans, 2016), such that if no forest is present in time t , no deforestation may occur.

Our hierarchical (multilevel) regression framework collectively models deforestation across Queensland (i.e the population level) by drawing on LGA-specific bent-cable model estimates. The bent-cable function comprises two linear segments (the incoming and outgoing phase), connected by a quadratic bend (Figure 3.4). The linear segments are parametrized by an intercept α_{0_i} , incoming slope α_{1_i} , and outgoing slope $\alpha_{1_i} + \alpha_{2_i}$.

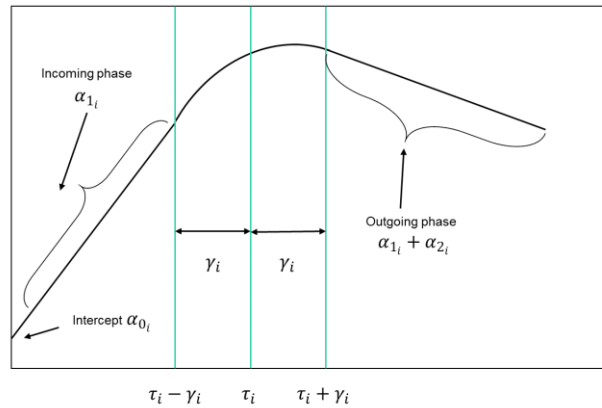


Figure 3.4 General depiction of the bent-cable function, adapted from Khan et al. (2009)

Our modeling framework below accounts for spatial correlation among LGAs, and the longitudinal nature of the data (i.e. each LGA has its own time series data):

$$\begin{aligned}
 & i = 1, \dots, n_{\text{LGA}} (= 50) \\
 & t = 1, \dots, n_T (= 27) \\
 & b_0, \mathbf{b}_1, \mathbf{b}_2, b_{15}, b_{23}: \text{population-level driver slope parameters (fixed effects)} \\
 & a_1, a_2, \mathcal{T}, \ell_\gamma: \text{population-level bent-cable parameters (fixed effects)} \\
 & v, \sigma_\tau, \sigma_\gamma, \sigma_1, \sigma_2, \sigma_{10}, \sigma_{20}: \text{population-level dispersion parameters (fixed effects)} \\
 & \mathbf{E}_t = (\text{macroeconomic data vector})_t \\
 & \mathbf{C}_{it} = (\text{climate data vector})_{it}
 \end{aligned}$$

$$\text{Level 1 : } \left\{ \begin{aligned}
 & y_{it} = (b_0 + \beta_{10i}) + b_{15}L_i + b_{23}(\text{elevation})_i \quad (\text{i.e., spatial-only covariates}) + \\
 & \quad \beta_{20t} + \mathbf{b}'_1 \mathbf{E}_t \quad (\text{i.e., temporal-only covariates}) + \\
 & \quad \mu_{it} \quad (\text{i.e., spatio-temporal term}) + \epsilon_{it} \\
 & \epsilon_{it} \sim N(0, v^2) \\
 & \mu_{it} = \mathbf{b}'_2 \mathbf{C}_{it} + \alpha_{1i}t + \alpha_{2i} \frac{(t - \tau_i + \gamma_i)^2}{4\gamma_i} 1\{|t - \tau_i| \leq \gamma_i\} + \alpha_{2i}(t - \tau_i) 1\{t > \tau_i + \gamma_i\} \\
 & \quad (\text{i.e., spatiotemporal covariates and LGA-specific bent cable})
 \end{aligned} \right.$$

$$\text{Level 2: } \left\{ \begin{aligned}
 & \alpha_{1i} \sim N(a_1, \sigma_1^2) \\
 & \alpha_{2i} \sim N(a_2, \sigma_2^2) \\
 & \tau_i \sim N(\mathcal{T}, \sigma_\tau^2) \\
 & \log \gamma_i \sim N(\ell_\gamma, \sigma_\gamma^2) \\
 & \beta_{10i} \sim \text{CAR}(\sigma_{10}^2) \text{ (i.e., random LGA-specific deviation} \\
 & \quad \text{from population intercept } b_0) \\
 & \beta_{20t} \sim N(0, \sigma_{20}^2) \text{ (i.e., random year-specific deviation f} \\
 & \quad \text{rom population intercept } b_0)
 \end{aligned} \right.$$

where “CAR” stands for the “conditional autoregressive” spatial correlation structure that we assume among LGAs. CAR is the spatial analogy of the lag-one autoregressive (AR(1)) temporal structure. It stipulates that given any LGAs i and j , their response values y_{it} and y_{jt} mutually influence each other only if i and j share borders (Earnest et al., 2007; Khan et al., 2012). Note that Khan and colleagues (2012) also employ the CAR structure under a spatial-longitudinal bent-cable framework to evaluate the impact of the Montréal Protocol on

the reduction of atmospheric chlorofluorocarbons. However, they only consider 8 spatial units under an ambiguous definition of spatial adjacency.

Statistical inference is made under the Bayesian paradigm. The posterior distribution from which Bayesian estimates are derived requires prior distributions to be specified for all population-level parameters (see Appendix 2.4 for details). Models are implemented in R using the R2WinBUGS package (Sturtz et al., 2005). WinBUGS numerically approximates the posterior distribution (Gelman et al., 2013; Kery, 2010; Lunn et al., 2000).

To explore relevance of land tenure on spatial adjacency, we draw on the approach taken by Earnest and colleagues (2007) and define spatial weighting according to:

$$w_{ij} = \frac{\mathbf{1}\{i, j \text{ share border}\}}{|L_i - L_j| + 0.00001}$$

Such weighting stipulates that neighbouring LGAs i and j would influence each other more than neighbouring LGAs i' and j' if L_i and L_j were more similar than $L_{i'}$ and $L_{j'}$.

We consider two key variations to our model framework described above, to explore:

- (a) Whether the “bend” occurs around the year 2000 at the introduction of the original VMA, or around 2007 after the VMA amendments came into effect; and
- (b) If weighting spatial adjacency by land tenure similarity makes a difference to model fit, as compared to unweighted spatial adjacency

Other minor model variations were considered to address model goodness-of-fit (Appendix 2.4)

Results

Having tested the influence of over 20 covariates on deforestation behavior, our model selection procedure revealed that annual % growth in GDP and year were the only clear predictor variables of LGA-specific deforestation in a combined regression model. In other words, GDP can be regarded as a clear driver of deforestation at the LGA level even in the presence of other predictor variables, while other macroeconomic and climate predictors are confounded with year. Land tenure and elevation were also not statistically important drivers. Appendix 2.5 provides justification for discarding these model covariates.

Our model shows strong evidence of spatial contagion in deforestation (Figure 3.5), and that this effect is strengthened by spatial weighting which accounts for land tenure similarity between each LGA (median deviance ~ -1500 with weighting, ~ -930 without weighting; a smaller deviance indicates better goodness-of-fit; Appendix 2.6, see also Figure A2.2)

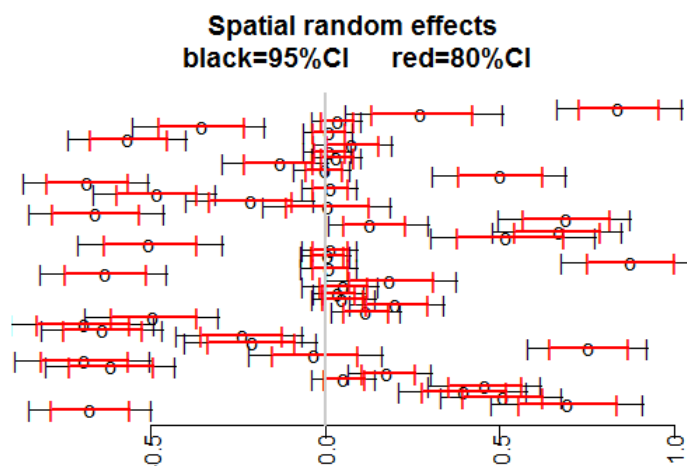


Figure 3.5 Estimates of β_{10i} ('o') and corresponding Bayesian confidence intervals (CIs) at the 80% level (black) and 95% level (red). Strong evidence of spatial contagion is suggested by the large number of CIs which exclude 0.

Our results also suggest that deforestation in Queensland exhibits a population-level (state-wide) “bend” mostly between 2000 and 2007, consistent with expectations (Figure 3.6). However, extreme variation is inherent in the deforestation time series (detrended by removing LGA-level driver effects from y_{it}) between and within LGAs, so that the signal for a state-wide bend is not particularly strong. Nevertheless, our results highlight there are broader macroeconomic and temporal explanations for deforestation at the state level, as supported by previous analyses (Chapter Two).

We found that LGA-level bent-cable estimates (Figure 3.7) can differ substantially. For some LGAs, deforestation increased over time to around 2000 then decreased (e.g Goondiwindi, Figure 3.7), as predicted by the timing of policy changes (Figure 3.2). Other LGAs followed the completely opposite trend (e.g Redland) and displayed an upturn in deforestation around 2007 which lasted until the end of the time series. Deforestation in some LGAs remained consistently high relative to the population-level trend (e.g Hope Vale) from 1988 to 2014,

or followed a general declining trend (e.g. Central Highlands). In yet more LGAs, deforestation peaked around 2007 and subsequently declined (e.g. Quilpie).

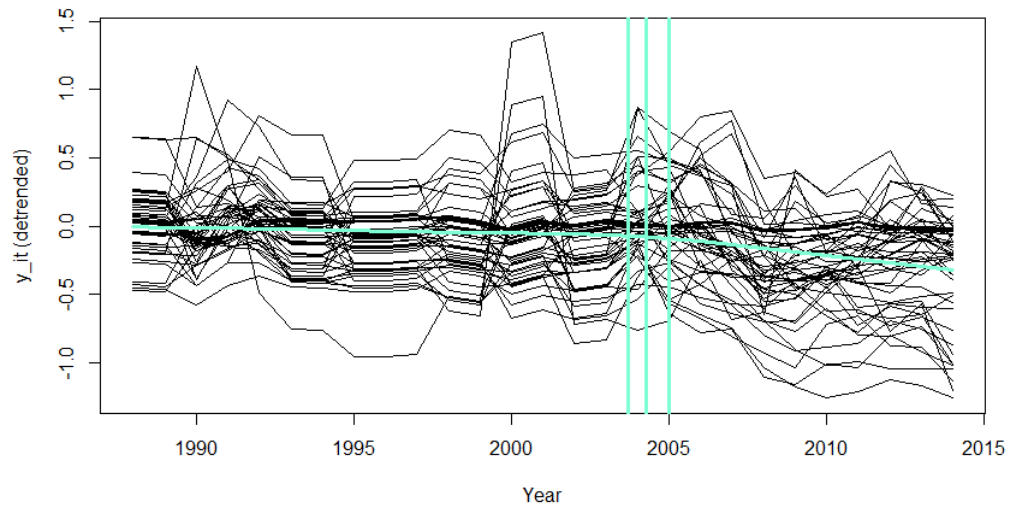


Figure 3.6. LGA-level time series y_{it} ($n = 50$) after detrending (black) and fitted population-level bent cable (green). “Detrending” refers to removing LGA-level driver effects from the y_{it} data. Vertical lines delimit the estimated bend (start, middle, and end of transition phase)

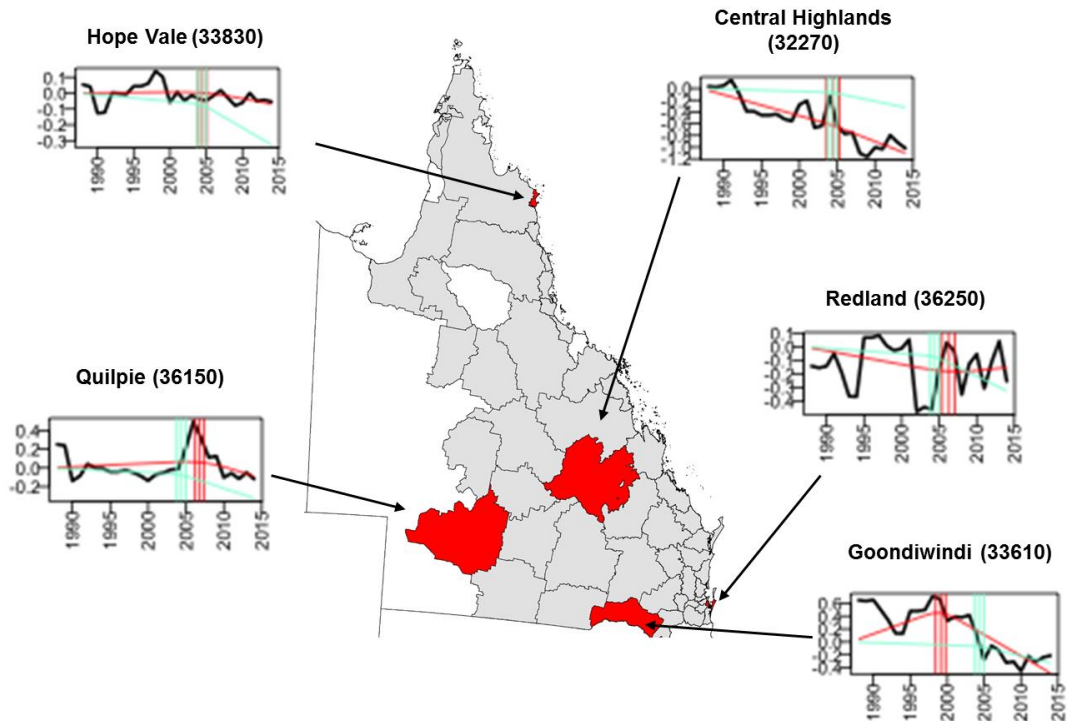


Figure 3.7. Examples of LGA-level time series y_{it} ($n = 50$) after detrending (black), with LGA-level bent-cable estimate (red) and fitted population-level bent cable (green). See also Appendix 2.8, Figure A2.3

Discussion

We found that the spatially explicit bent-cable model is a promising technique for detecting a policy effect simultaneously at the local government area (LGA) level and state level while controlling for driver variables; and there is some evidence of policy-induced shifts in the deforestation rate as expected in 2000 with the introduction of the original VMA, and with the subsequent amendments in 2007. However, due to an inadequate number of data points (LGAs) under extreme spatial variation in trends in, and responses to, deforestation within and between local government areas, the overall level of statistical confidence may not be overwhelmingly high. Nevertheless, our analysis quantitatively supports that while deforestation in some LGAs increased in 2000 and decreased in 2007 as predicted by the policy changes, other LGAs followed the completely opposite trend, while deforestation remained either consistently high or low throughout the time series (1988 to 2014).

A core challenge with any statistical analysis is for there to be sufficient data points which may permit detection of the effect of policy introduction amidst substantial variation inherent in the data. This is understandably difficult to achieve at the national/state scale, especially with limited temporal continuity in data points. Expanding our analysis to the national scale could improve the capacity to detect any effect of policy interventions on deforestation, but at the expense of additional complexities given the considerable variation in timing and scope of regulatory policies across different Australian states (Evans, 2016).

It should be noted that little improvement in our capacity to estimate a “bend” would be expected by considering the data at a finer spatial resolution than LGA (for example in an equal-area grid), as the total amount of information per region in a grid would be offset by the substantially higher spatial correlation across gridded regions.

Using LGAs as a spatial unit of analysis provides the benefit of being able to investigate plausible explanations for regional deforestation trends using data captured within our tested variables (Table 3.1), or additional contextual information we have far been unable to

quantify. For example, the Goondiwindi LGA is held entirely in freehold tenure (Table A2.2, excluding publicly managed protected areas). The LGA-specific “bend” around 2000 is consistent with the “panic clearing” phenomenon documented in 2000 (Evans, 2016; McGrath, 2007), where deforestation spiked prior to the introduction of the original VMA which imposed restrictions on deforestation on freehold tenure for the first time. Hope Vale is an Aboriginal Shire Council with a population of only 1,125 people (ABS, 2016), and several exemptions exist under the VMA to enable economic development on Indigenous land (State of Queensland, 2015). The upturn in deforestation around 2007 in the Redland LGA (Figure 3.7) can be explained by a spike in urban and residential development⁷. Exemptions under the VMA allow for native vegetation clearing to proceed for urban development (e.g. residential, industrial, sporting, recreational or commercial) in a regional ecosystem (Nelder et al., 2017) that is listed as ‘least concern’ (State of Queensland, 2015)⁸. Land tenure in the Quilpie LGA is equally split between freehold and leasehold (Table A2.2), and is dominated by Mulga (*Acacia aneura*) which is regularly “pushed” to maintain pasture, and to provide emergency feed for cattle during periods of drought (Dwyer et al., 2009; Fensham and Guymer, 2009; Gunders, 2017).

Our present analysis sought to quantitatively establish evidence for the impact of regulatory policies aimed at reducing deforestation in Queensland, which are widely regarded to have led to a state-wide and national decline in deforestation around 2007 (Australian Government, 2013; Evans, 2016; McGrath, 2007; Reside et al., 2017). Previous work has emphasized the overarching influence of macroeconomic variables (Angelsen, 2010; Angelsen and Kaimowitz, 1999; Geist and Lambin, 2002), agricultural commodity prices and terms of trade (Australian Government, 2013; Barbier and Rauscher, 1994; Ewers et al., 2008; Richards et al., 2012), rainfall and temperature (Australian Government, 2013; Marcos-Martinez et al., 2017), biophysical variables (Marcos-Martinez et al., 2017; Pressey et al., 1996) and institutions (Dolisca et al., 2007; Fearnside, 2001; Messina et al., 2006;

⁷ The koala is a charismatic species² which was once abundant throughout the Redland region, but has suffered an estimated 80.3% decline in population densities between 1996 and 2014, primarily due to urban development (Rhodes et al., 2006, 2015).

⁸ Note that the Queensland, New South Wales and Australian Capital Territory populations of the koala (*Phascolarctos cinereus*) were listed as Vulnerable under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) in 2012, and so would not have substantially influenced the observed deforestation trend in Redland.

Nolte et al., 2013; Robinson et al., 2014) in driving deforestation. We tested representative variables from each of these groups (Table 3.1), and our modelling framework (which combined driver variables with year as covariates within a single regression) revealed that most of these variables were highly correlated or provided no extra information in addition to year (Appendix 2.2), or did not improve model goodness-of-fit (Appendix 2.5).

The Australian Government uses a linear regression model (with farmers' terms of trade as a single covariate) to predict future deforestation up to 2035. This model informs national land use, land use change and forestry (LULUCF) emissions projections for reporting under the UN Framework Convention on Climate Change (UNFCCC) (Australian Government, 2013, 2015). Although this simple model may be adequate for making a national-level prediction of future deforestation trends, our spatially explicit analysis suggests that the assertion of a "drop in land clearing activity from 2007 onwards" (Australian Government, 2015: 20) based on a single-variable regression is premature. In future work, our quantitative evaluation approach could be strengthened by including spatially explicit information not usually considered in deforestation modelling, such as broad vegetation groups (Nelder et al., 2017), primary land use (ABARES, 2010), and social data which may reveal landholder compliance behaviours (Bartel and Barclay, 2011).

Our analysis used the 2014 version of human induced forest change data as developed by the Australian Government for the NCAS (Australian Department of the Environment, 2015, 2016). The Queensland Government also runs a system for detecting and analyzing land use change under the Statewide Landcover and Trees Study (SLATS) (Department of Science, Information Technology and Innovation, 2015). Recent work has identified substantial differences in the amount of deforestation estimated by the NCAS and SLATS systems (Bulinski et al., 2016), largely due to an inconsistent definition of 'forest'⁹ (Commonwealth of Australia, 2016b). To determine whether regulatory policies introduced in Queensland have affected local deforestation, the SLATS data would be more fit for purpose. However,

⁹ The NCAS adopts a Kyoto definition of 'forest' where canopy cover must be at least 20%. However, much of the historical and recent native vegetation clearing in Queensland is 'sparse woody vegetation' (e.g. Mulga and Brigalow ecosystems) which does not always meet the 20% canopy cover threshold. The NCAS is designed to monitor forest loss for the purpose of reporting Australia's emissions under the UNFCCC, whereas the SLATS program is used specifically to monitor for compliance under the Queensland VMA.

collectively modelling deforestation at the national scale would still require use of the NCAS system given the substantial differences in deforestation accounting methods used in each Australian state (Bulinski et al., 2016).

Policy evaluation in the conservation literature is currently dominated by the use of impact evaluation methods (Baylis et al., 2016; Ferraro, 2009; Ferraro and Pattanayak, 2006; Miteva et al., 2012), which aim to infer causal relationships through quasi-experimental techniques such as matching (Andam et al., 2008; Arima et al., 2014; Brandt et al., 2016; Ferraro et al., 2013; Joppa and Pfaff, 2011) and synthetic control (Abadie et al., 2010; Sills et al., 2015). Such methods rely on the ability to isolate a counterfactual from the available data, which must be of sufficient temporal and spatial resolution relative to the type and frequency of policy interventions to be evaluated. Impact evaluation methods are regarded by some scholars as ‘best practice’ for policy evaluation (Ferraro and Hanauer, 2014; Miteva et al., 2012), and have been used extensively to evaluate the impact of protected areas on deforestation in tropical nations (Andam et al., 2008; Ferraro et al., 2013; Gaveau et al., 2009; Joppa and Pfaff, 2011; Nolte et al., 2013). However, quasi-experimental techniques are not applicable to our Queensland case study, as a counterfactual could not be readily isolated since (i) the VMA and its amendments regulate the state-wide clearing of native vegetation, and any differences in compliance effort or legislative exceptions applied do not follow a clear or consistent pattern; and (ii) regulatory policies for reducing deforestation exist in each Australian state and territory, hence a synthetic control cannot be established. Arima et al. (2014) evaluated the impact of the second phase of Brazil’s Action Plan for Prevention and Control of the Legal Amazon Deforestation (PPCDAm ii), which led to a reduction in deforestation from 2008 despite an increase in the price of beef cattle and agricultural GDP (Arima et al., 2014; Richards et al., 2017). However, matching techniques were possible in this situation due to the differing levels of enforcement of PPCDAm ii across Brazilian municipalities.

Conclusions

Deforestation in Australia occurs at a rate and scale which is on par with well-known global deforestation hotspots (Hansen et al., 2013; Keenan et al., 2015), despite the presence of strong governance (Bhattarai and Hammig, 2004; Deacon, 1994) and the introduction of a

series of policies aimed to control the clearing of native vegetation over the past four decades (Evans, 2016). Given the substantial and widespread impacts of deforestation on biodiversity, ecosystem service provision, and climate regulation, a greater understanding of the efficacy of policy interventions in regions where deforestation may be assumed to be effectively “controlled” therefore warrants further attention. Although our population-level findings should be considered preliminary given the substantial “unexplained” variation in LGA-specific deforestation trends, our results nonetheless suggest a degree of caution should be taken before proclaiming the effectiveness of regulatory policies prior to conducting an *ex-post* evaluation. Data limitations and methodological challenges can impede the evaluation of policy impact in the presence of confounding variables, time-lags and misalignment of spatial and temporal data. Such challenges in evaluation must be overcome to ensure that policies effectively deliver environmental outcomes as anticipated.

Acknowledgments

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CHAPTER FOUR:

Carbon farming via assisted natural regeneration as a cost-effective mechanism for restoring biodiversity in agricultural landscapes

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Abstract

Carbon farming in agricultural landscapes may provide a cost-effective mechanism for offsetting carbon emissions while delivering co-benefits for biodiversity through ecosystem restoration. Reforestation of landscapes using native tree and shrub species, termed environmental plantings, has been recognized as a carbon offset methodology which can contribute to biodiversity conservation as well as climate mitigation. However, far less attention has been paid to the potential for assisted natural regeneration in areas of low to intermediate levels of degradation, where regenerative capacity still remains and little intervention would be required to restore native vegetation. In this study, we considered the economics of carbon farming in the state of Queensland, Australia, where 30.6 million hectares of relatively recently deforested agricultural landscapes may be suitable for carbon farming. Using spatially explicit estimates of the rate of carbon sequestration and the opportunity cost of agricultural production, we used a discounted cash flow analysis to examine the economic viability of assisted natural regeneration relative to environmental plantings. We found that the average minimum carbon price required to make assisted natural regeneration viable was 60% lower than that required to make environmental plantings viable (\$65.8 t CO₂e⁻¹ compared to \$108.8 t CO₂e⁻¹). Assisted natural regeneration could sequester 1.6 to 2.2 times the amount of carbon possible compared to environmental plantings alone over a range of hypothetical carbon prices and assuming a moderate 5% discount rate. Using a combination of methodologies, carbon farming was a viable land use in over 2.3% of our study extent with a low \$5 t CO₂e⁻¹ carbon price, and up to 10.5 million hectares (34%) with a carbon price of \$50 t CO₂e⁻¹. Carbon sequestration supply and economic returns generated by assisted natural regeneration were relatively robust to variation in establishment costs and discount rates due to the utilization of low-cost techniques to reestablish native vegetation. Our study highlights the potential utility of assisted natural regeneration as a reforestation approach which can cost-effectively deliver both carbon and biodiversity benefits.

Introduction

The carbon market has the potential to deliver significant outcomes for ecosystem restoration alongside the abatement of greenhouse gas emissions (Bradshaw et al., 2013). The demand for terrestrial carbon sinks is creating opportunities for avoided deforestation in tropical forests (Phelps et al., 2012a; Venter and Koh, 2011), as well as landscape-scale restoration through afforestation and reforestation (Galatowitsch, 2009; Peters-Stanley et al., 2013; Silver et al., 2000). There is particular interest as to whether the carbon market can deliver positive outcomes not only for the climate and local economies, but also for biodiversity (Bekessy and Wintle, 2008; Smith and Scherr, 2003). A too narrow focus on maximizing sequestration of carbon (such as the planting of monocultures) can lead to a range of negative ecological impacts (Lindenmayer et al., 2012; Pittock et al., 2013), and will miss opportunities for co-benefits derived through restoration of natural ecosystems (Bullock et al., 2011; Dwyer et al., 2009; Gilroy et al., 2014; Rey Benayas et al., 2009).

Carbon farming is a term that is used to describe land-based practices which either avoid or reduce the release of greenhouse gas emissions, or actively sequester carbon in vegetation and soils, primarily in agricultural landscapes. Several studies have examined the economics of carbon farming through establishment of monocultures or environmental plantings (Bryan et al., 2014; Bryan and Crossman, 2013; Crossman et al., 2011; Paterson and Bryan, 2012; Paul et al., 2013; Polglase et al., 2013). Environmental plantings are a mixture of locally indigenous tree and shrub species which are planted or seeded on cleared land, and are not normally harvested (Paul et al., 2013). The potential for environmental plantings to deliver biodiversity co-benefits alongside carbon abatement has been a focus of recent work (Bryan et al., 2014; Carwardine et al., 2015; Goldstein et al., 2006; Lin et al., 2013; Nelson et al., 2008; Pichancourt et al., 2014; Renwick et al., 2014). Yet given the high up-front costs of direct planting (Chazdon, 2008; Schirmer and Field, 2000), it is surprising that there has been limited assessment of the economic viability of carbon sequestration through assisted natural regeneration of vegetation, despite the large potential biodiversity and economic benefits of this approach (Birch et al., 2010; Bradshaw et al., 2013; Butler, 2009; Dwyer et al., 2009; Funk et al., 2014; Smith and Scherr, 2003; Trotter et al., 2005).

Assisted natural regeneration (ANR, also known as managed regrowth) is recognized as a cost-effective forest restoration method that can restore biodiversity and ecosystem services in areas of intermediate levels of degradation, while also providing income for rural livelihoods (Chazdon, 2008; Ma et al., 2014). ANR relies on residual seeds and plants at the site, or dispersed from vegetation nearby. ANR utilizes low-cost techniques to assist in the natural re-establishment of vegetation, such as: restriction of livestock grazing through fencing and direct stocking rate management; cessation of tree control practices like burning and disturbance with machinery; the use of vegetation thinning to reduce competition and promote growth, and; in some circumstances, supplementary planting of seedlings (Smith and Scherr, 2003). Although most frequently applied in tropical forests (Rey Benayas, 2007; Shono et al., 2007), ANR is gaining momentum as an important mechanism for restoring forests across a range of ecosystems (Chazdon, 2008; Gilroy et al., 2014; Shono et al., 2007).

Vegetation that is allowed to naturally regenerate has several advantages for biodiversity conservation over plantings, even when plantings are comprised of native species. First, under ANR, the vegetation is more likely to be comprised of native species adapted to local conditions, resulting in vegetation that is more resilient to local climate variation and disturbance. Second, natural regeneration can result in high species diversity including trees, shrubs, forbs and grasses, whereas under environmental planting, generally only tree species are planted. Third, ANR often provides superior habitat for local fauna as a result of the increased plant and structural diversity (Bloomfield and Pearson, 2000; Bowen et al., 2009; Bruton et al., 2013; R.J. Fensham and Guymer, 2009). Finally, under the right conditions, the cost of establishing vegetation through ANR is much lower than active planting (Sampaio et al., 2007; Schirmer and Field, 2000; Smith, 2002).

Despite the potential advantages of ANR, a lack of awareness of its benefits and demonstrative results means it remains underutilized (Shono et al., 2007). ANR falls under the definition of afforestation/reforestation (A/R) under the Kyoto Protocol and Clean Development Mechanism (Smith, 2002; Smith and Scherr, 2003), but has attracted little attention as a carbon sequestration methodology compared to mechanisms such as active planting or avoided deforestation (Niles et al., 2002). ANR has most potential in locations

that have not been intensively used (cropped or irrigated) or with a relatively short history of intensive land use. Across much of sub-tropical Australia most grassy eucalypt woodlands used for grazing land fall into this category (McIntyre and Martin, 2002). A window of opportunity therefore exists to achieve significant carbon and biodiversity outcomes through assisted natural regeneration across much of northern Australia (R.J. Fensham and Guymer, 2009; Martin et al., 2012), the Texas drylands (Asner et al., 2003), central Brazilian pastoral lands (Sampaio et al., 2007), the Gran Chaco in Argentina (Zak et al., 2004), degraded pastoral landscapes in Albania (Deichmann and Zhang, 2013) and in the mountainous Humbo region of Ethiopia (Biryahwaho et al., 2012).

The aim of this study was to evaluate the potential for carbon farming in the extensive agricultural landscapes of the state of Queensland, in north-eastern Australia, by examining the economic viability of ANR relative to environmental plantings. Commercial livestock grazing on pastures with dominant native species is the main land use across Queensland. The extensive, as opposed to intensive (McIntyre and Martin, 2002), nature of grazing in much of Queensland provides ideal conditions for carbon sequestration via ANR. Profitability (profitability at full equity) of grazing throughout Queensland is generally low with many farms losing money in recent years (ABARES, 2013). To determine whether carbon farming could be a viable land use in Queensland, we conducted a spatially explicit analysis of the minimum ('break-even') carbon price required for carbon farming to become profitable via environmental plantings and ANR. We also considered a range of hypothetical carbon prices and discount rates to estimate the carbon sequestration supply and profitability of carbon farming over a long (100 years) and medium (25 years) project duration. Finally, we tested the sensitivity of our results to variation in the establishment costs of each methodology.

Study region and policy context

Our case study region is in the state of Queensland, in north-eastern Australia (Figure 4.1). Agricultural development over the past 150 years has led to extensive landscape modification (Dwyer et al., 2009; McAlpine et al., 2002) with the most rapid development occurring in the vast Brigalow Belt bioregion within the latter half the 20th century (Seabrook et al., 2006).

As a result, around 34 million hectares of vegetation in Queensland (20% of the state's total vegetated area) is now considered non-remnant: heavily modified, secondary vegetation. Commercial grazing of livestock is the predominant land use across much of northern Australia, where it occurs in extensively managed grassy eucalypt and acacia woodlands and shrublands (Martin and McIntyre, 2007). Unlike southern parts of the continent, these northern landscapes have not been subject to broad scale intensification via sowing of exotic pastures, fertilization and irrigation. Despite broad scale clearing of trees and shrubs in some regions (Martin et al., 2012), much of the cleared land retains regenerative capacity via small trees and soil seed bank, e.g. Brigalow *Acacia harpophylla*) (Dwyer et al., 2009; Fensham and Guymer, 2009).

At present, clearing regrowth to maintain high quality forage for livestock represents a substantial management cost to graziers throughout Queensland (Gowen et al., 2012; McIntyre and Martin, 2002). Landholders not clearing regrowth would forgo some pasture, but could attract credits for carbon sequestered if the vegetation was left to regenerate (Commonwealth of Australia, 2013b, 2013a). Extensive restoration of vegetation in these agricultural landscapes is a high priority to avoid potential long term ecological impacts and future native species extinctions (Martin, 2010; McAlpine et al., 2002).

Australia's Carbon Farming Initiative (CFI, Commonwealth of Australia, 2011) and climate policies are currently under review (at time of writing, February 2015); however, there is broad political support for landholders to generate additional income through the provision of land-based carbon offsets in agricultural landscapes. We examine a range of carbon prices, project durations and establishment costs in order to gain an understanding of the economic viability of two key reforestation methodologies in our study region to help guide the carbon farming policy debate.

Methods

Land use data

We restricted the extent of our analysis to sub-bioregions in Queensland where at least 5% of the sub-bioregion is comprised of agricultural production landscapes (resulting in 73 of

130 sub-bioregions being considered). Our study extent encompasses 30.6 million hectares of agricultural landscapes potentially suitable for ANR or environmental plantings. To refine this extent to areas where ANR or environmental plantings are feasible, we used a state-wide vegetation coverage layer (Department of Environment and Resource Management, 2009) to delineate the extent of cleared land in Queensland (Neldner et al., 2005).

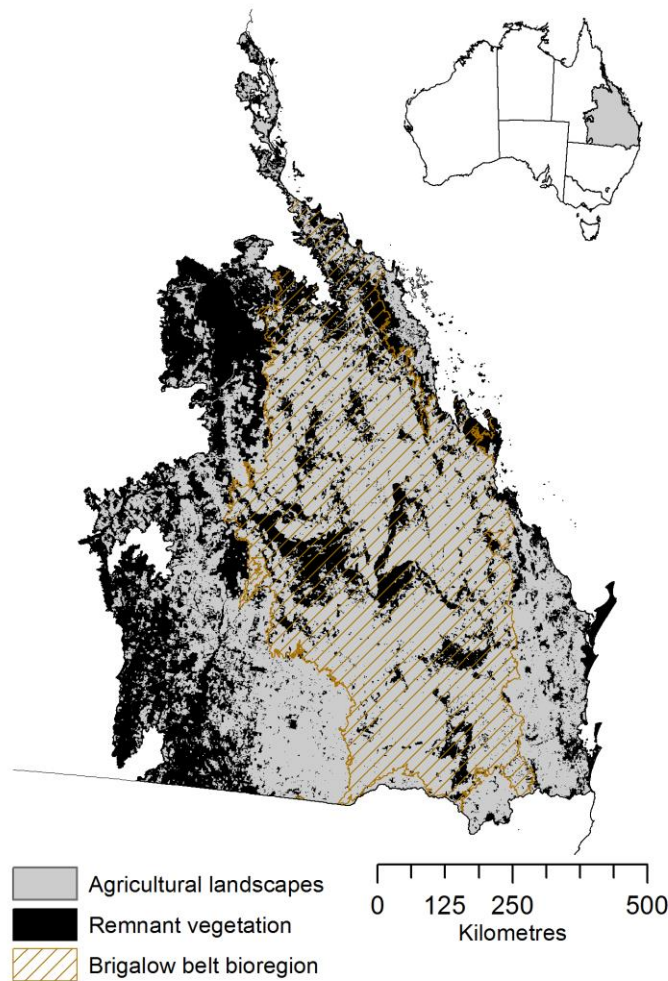


Figure 4.1. The study extent encompasses 73 sub-bioregions in Queensland, of which agricultural landscapes make up 30.6 million hectares. Areas of remnant vegetation (in black) are excluded from the analysis. The Brigalow Belt bioregion (hatched) covers an extensive part of the study extent.

We excluded areas of intensive land use (mines, urban areas), irrigated cropping, protected areas and water bodies from the analysis using a national land use dataset. Land uses included in our analysis were native pasture (88.7% study extent), non-irrigated cropping (hereafter, ‘cropping’, 3.5%), and modified pastures (6.1%). The native pasture category includes land where there has been limited or no deliberate attempt at pasture modification, and vegetation contains greater than 50 per cent dominant native species (ABARES, 2010). For ease of interpretation, we incorporated modified pastures within the ‘cropping’ land use category. Approximately 1.7% of the study extent is formerly rainforest where cropping is now the dominant land use, which is important to delineate given the higher costs of environmental plantings in these areas (Catterall and Harrison, 2006).

Environmental plantings were considered to be feasible across each of our three land use categories (native pasture, cropping and former rainforest). However, ANR is generally not a suitable carbon farming method on sites which have been cultivated, irrigated and sown to exotic pastures, due to lack of local regenerative capacity in native vegetation (Fischer et al., 2009; McIntyre and Martin, 2002). We therefore restricted our analysis of ANR to areas of native pasture.

Estimating the rate of carbon sequestration

The rate of carbon accumulation through forest growth varies temporally, and so understanding the cost-effectiveness of alternative carbon farming methodologies requires this dynamic variation in flows to be explicitly accounted for (Richards and Stokes, 2004). To capture this temporal variation, we emulated the core of the FullCAM forest growth model (Richards and Brack, 2004). We used a dataset known as the Maximum Potential Biomass layer (MaxBio, Department of Climate Change & Energy Efficiency, 2004) to derive estimates of the rate of carbon sequestration under the ANR and environmental plantings methodologies.

MaxBio and FullCAM are components of the National Carbon Accounting System (NCAS), which estimates Australia’s greenhouse gas emissions from land based activities in accordance with the international guidelines adopted by the United Nations Framework

Convention on Climate Change (UNFCCC). MaxBio is estimated from a forest productivity index (FPI, Kesteven and Landsberg, 2004) generated with a plant physiology model using bioclimatic parameters, and empirically related to above ground biomass in native forests (Richards and Brack, 2004).

The predicted above-ground tree biomass ($t\ ha^{-1}$) at time t is a function of MaxBio, M , and an estimated constant k that determines the rate of approach towards the maximum biomass (Richards and Brack, 2004) is:

$$M(t) = Me^{-k/t}. \quad (1)$$

Biomass generally accumulates more rapidly in tree species commonly used in environmental plantings compared to regrowth of native vegetation, hence we consider $k = 20$ for environmental plantings and $k = 24$ for ANR in this study, which reflects values used for Australia's National greenhouse gas accounts (Commonwealth of Australia, 2012a, 2014). We accounted for biomass allocation to coarse roots (root:shoot ratio) using the recommended fraction of 0.25 for acacia forest and woodland (Commonwealth of Australia, 2012a; Snowdon et al., 2000), hence the ratio of total biomass to above ground biomass is 5:4, or 1.25 times.

The long-term average annual increment in biomass accumulation (cumulative above- and below-ground biomass) between t and $t + 1$ years (I_t), using equation (1):

$$I_t = 1.25M(e^{-k/t} - e^{-k/(t-1)}). \quad (2)$$

The carbon content of tree biomass can range between 45-50% carbon, but this varies by species (Thomas and Martin, 2012) and tree component (Gifford, 2000). We adopted a conversion factor of 50% to be consistent with the Australian National Carbon Accounting System (Commonwealth of Australia, 2012a; Gifford, 2000).

The annual sequestration rate of carbon by vegetation (c_t , $t\ CO_2e\ ha^{-1}$), is therefore:

$$c_t = 0.5 \times 3.67(I_t - I_{t-1}), \quad (3)$$

where 3.67 is the ratio of the atomic masses of CO₂ and C.

For simplicity, we assumed that the carbon stock at project commencement ($t=0$) is zero. Current methodologies to credit carbon sequestration through ANR in Australia require that project areas have evidence of regeneration but little standing carbon stock at commencement. Our model is consistent with estimates from the CFI Reforestation Modelling Tool (RMT, Domestic Offsets Integrity Committee, 2011). FullCAM and the RMT model forest growth for user supplied point locations but our approach allowed us to generate estimates of carbon yield from ANR and environmental plantings over a large spatial extent rather than on a single project basis (see Supplementary Material in Appendix Three for further details).

Costs of carbon farming

Opportunity costs of agriculture were derived from the most current map of agricultural profit for Australia, which was based on data for the year 2005/2006 (Marinoni et al., 2012). The map is a grid of profitability at full equity (*PFE*, \$ ha⁻¹) at a 1km² resolution across the Australian continent, using data on production, revenues and costs for 23 irrigated and rain-fed agricultural commodities, combined with data on land use (2005/2006) and yield estimates. *PFE* is a measure of profit which is calculated as the difference between revenue from the sale of agricultural commodities and all fixed and variable costs (Bryan et al., 2009; Marinoni et al., 2012). The system developed by Marinoni and colleagues will enable the production of a more current map of agricultural profitability once the latest land use data set for Australia (2010/2011) is finalized. As this dataset is not yet available, we adjusted *PFE* to present day values based on a 2.7% annual rate of inflation between 2006 and 2013 (Reserve Bank of Australia, 2014).

We considered a mid-range once-off (incurred at $t=0$) on-ground establishment cost of \$2,000 ha⁻¹ for environmental plantings (tube stock, fencing, weed management, labour), but also conducted sensitivity analyses using high and low cost estimates (\$3,000 ha⁻¹ and \$1,000 ha⁻¹) adopted in previous studies (Crossman et al., 2011; Polglase et al., 2013; Schirmer and

Field, 2000). Environmental plantings in areas of former rainforest incurred an establishment cost of \$8,000 ha⁻¹ (Catterall & Harrison 2006).

Ceasing the routine re-clearing of regrowth vegetation is likely to be sufficient to allow regeneration in many parts of our Queensland study region. The balance between re-clearing costs and production income are part of *PFE*, hence we considered an establishment cost of \$0 ha⁻¹ for ANR in our analysis. However, in areas with a longer history of intensive land use, it may be necessary to restrict livestock access to the carbon farming project site to facilitate regeneration of vegetation (Comerford et al., 2011; Prober et al., 2011; Vesk and Westoby, 2001; Witt et al., 2011). We therefore conducted a sensitivity analysis by considering an establishment cost for ANR to cover the cost of erecting fences (including materials, labour, and transport). We derived an estimate for the establishment cost for an ANR project using estimates from Schirmer and Field (2000). This estimate was adjusted to present day values based on a 2.9% annual rate of inflation between 2000 and 2013 (Reserve Bank of Australia, 2014), to reach a final estimate of \$460 ha⁻¹ (Figure A3.1).

Finally, for both environmental plantings and ANR, we derived annual on-ground management costs from comparable studies (Comerford et al., 2011; Polglase et al., 2008; Schirmer and Field, 2000) and adjusted to 2013 prices (Reserve Bank of Australia, 2014) to reach an estimate of \$45 ha⁻¹ year⁻¹. Market participation costs included an initial project establishment cost (\$100 ha⁻¹), as well as annual monitoring and auditing costs of \$10 ha⁻¹ year⁻¹, and transaction costs of \$10 ha⁻¹ year⁻¹ (Bryan et al., 2014; Comerford et al., 2011; Paul et al., 2013).

Economic viability of carbon farming

To determine the economic viability of carbon farming, we used a discounted cash flow analysis to calculate the minimum price on carbon required to generate an economic return via environmental plantings or ANR. We generated a 1km² vector grid covering the extent of the study area, resulting in 707,530 planning units *i*. The average annual sequestration rate of carbon by ANR and environmental plantings and opportunity cost of carbon farming was calculated for each planning unit.

The net present value (*NPV*) of carbon sequestration in each site *i* is:

$$NPV_i = PVB_i - PVC_i, \quad (4)$$

where PVB_i is the present value of the benefits at site *i*, calculated according to a carbon price p (\$ t CO₂e⁻¹), discount rate r , and accounting for c_{ti} the rate of carbon sequestration at time t in site *i*:

$$PVB_i = \sum_{t=0}^T \frac{pc_{ti}(1 - d_R - d_{T=25})}{(1 + r)^t} \quad (5)$$

For Australian Government cost-benefit analyses, it has been proposed that discount rates over a range of 3 to 10 per cent should be tested (Harrison, 2010). Previous studies have evaluated the economic potential of carbon farming using discount rates ranging from 0% to 12% (Bryan et al., 2014; Funk et al., 2014; Paul et al., 2013; Polglase et al., 2013; Renwick et al., 2014). Unless otherwise indicated, we present our results where a moderate 5% discount rate has been applied throughout the manuscript. We also test the sensitivity of our findings to rates of 1.5% and 10% to enable comparison to the results of relevant key studies.

We accounted for a risk of reversal buffer (d_R) of 5%, which is deducted as a percentage of generated carbon credits in order to insure the CFI scheme against residual risks (Commonwealth of Australia, 2012c). Carbon farming policy in Australia currently requires carbon sequestration projects to remain in place for 100 years to meet permanence obligations (Commonwealth of Australia, 2012c; Macintosh and Waugh, 2012). An option for landholders to adopt a 25-year contract for a carbon farming project is currently under consideration (Australian Government, 2014a), but under this permanence option 20% of carbon credits would be deducted to reflect the potential cost to government of replacing carbon stores if 25-year projects are discontinued. Hence, we considered two project durations T of 100 and 25 years, and applied a 20% discount ($d_{T=25}$) to the credits earned if $T=25$.

The present value PVC_{ij} of the costs of carbon sequestration at site *i* is:

$$PVC_i = EC + \sum_{t=0}^T \frac{MC + TC + PFE_i}{(1+r)^t}, \quad (6)$$

where EC is the sum of the initial establishment and costs (\$ ha⁻¹), MC is the annual on-ground management cost (\$ ha⁻¹ year⁻¹), TC is the sum of transaction and monitoring costs (\$ ha⁻¹ year⁻¹) and PFE_i is the profitability at full equity (\$ ha⁻¹) of the current agricultural land use in site i (Marinoni et al., 2012).

Finally, we converted the NPV in each site to the equal annual equivalent:

$$EAE_i = NPV_i \frac{r(1+r)^T}{(1+r)^T - 1}.$$

Spatial analyses were conducted using ArcMap version 10 (ESRI, 2011), the discounted cash flow analysis was implemented using MATLAB version 7.10.0.499 (2010), and results were analysed using the R statistical package version 2.15.0 (R Development Core Team, 2012).

Results

Break-even carbon prices

We calculated the minimum price required for carbon farming to become profitable to gain an understanding of the size of payment necessary (\$ t CO₂e⁻¹) to encourage landholder adoption of ANR or environmental plantings in our study region. The ‘break-even’ carbon price p occurs when the NPV (Equation 4) is equal to 0. A positive break-even price indicates that an annual payment is required to stimulate conversion from agriculture to carbon farming, whereas a negative break-even price signifies that the current agricultural land use is producing negative economic returns and a conversion to carbon farming could occur at no cost.

Overall, the average site-level break-even carbon price ($T=100$) was considerably lower for ANR (\$65.8 t CO₂e⁻¹) as compared to environmental plantings (\$108.8 t CO₂e⁻¹, Figure 4.2). The break-even price varied according to land use (Figure A3.2), whereby environmental

plantings were generally more economically viable on sites where cropping was the dominant land use (\$99.9 t CO₂e⁻¹) compared to sites on native pasture (\$109.5 t CO₂e⁻¹). Environmental plantings on former rainforest sites broke even for \$153.0 t CO₂e⁻¹ on average. These averaged estimates mask much of the spatial heterogeneity in the break-even carbon price across the study extent (Figure 4.2). Low break-even prices were more frequent in the relatively productive east of the study region, and several areas in the central eastern coast have similar break-even prices under either methodology. Over the 25 year project duration, average break-even estimates for ANR increased to \$76.1 t CO₂e⁻¹, and to \$141.5 for environmental plantings.

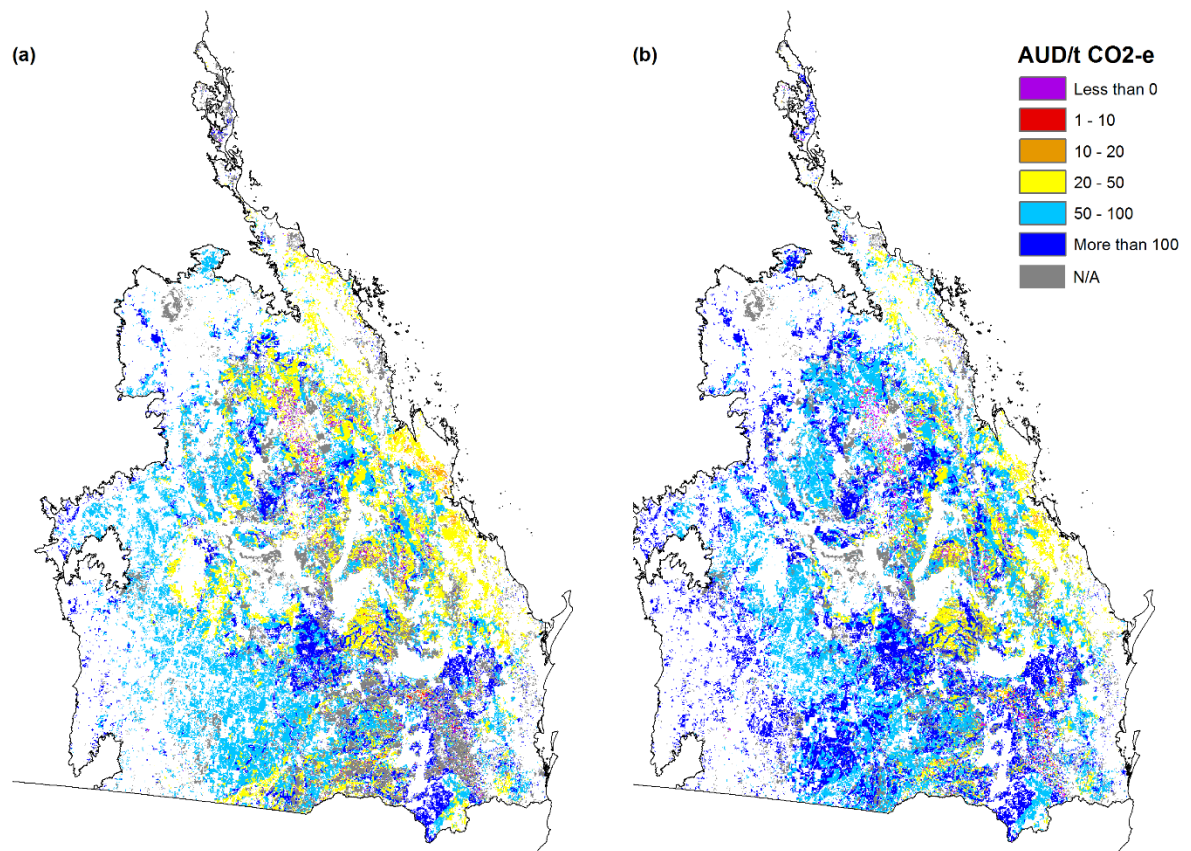


Figure 4.2. Break-even carbon prices (100 year project duration and 5% discount rate) for a) assisted natural regeneration (ANR) and b) environmental plantings. Areas where carbon farming is not available are shaded in grey.

Carbon sequestration

Using the break-even prices estimated previously for all sites in our study extent, we generated supply curves for carbon sequestration using ANR and environmental plantings under the two project durations (Figure 4.3). We also present a third ‘least cost’ curve which is derived by selecting the methodology with the lowest break-even price in each site. This portfolio comprises of sites where environmental planting is the only available carbon farming methodology (where the land use is either cropping or former rainforest), in addition to sites where ANR is the more cost-effective of the two possible carbon methodologies.

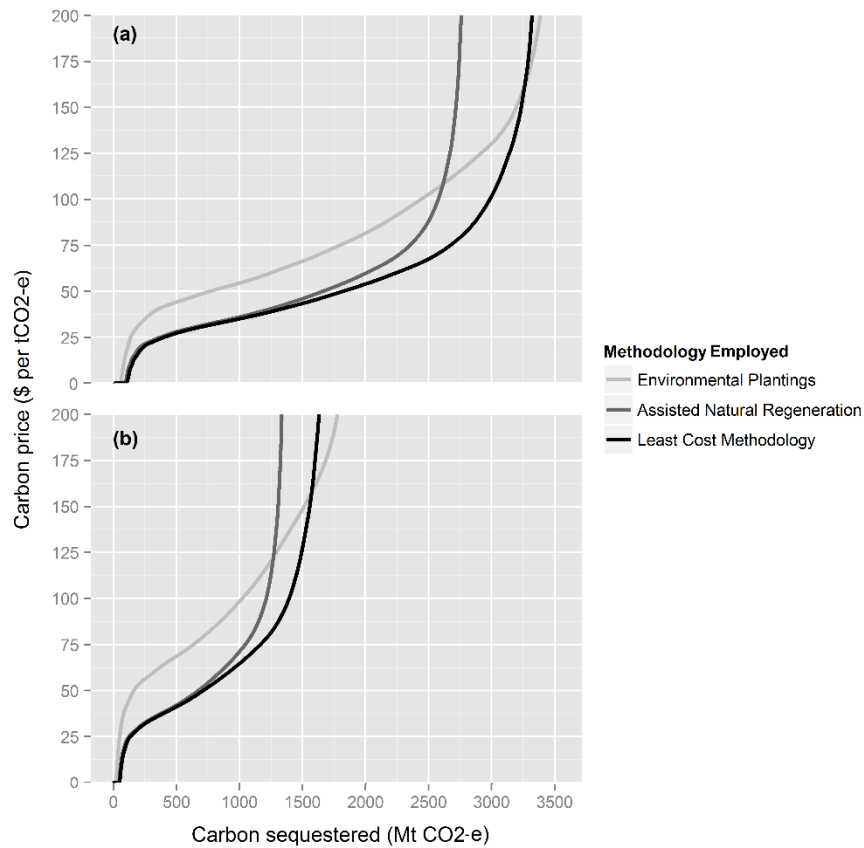


Figure 4.3. Carbon sequestration supply curves for ANR, environmental plantings and ‘least cost’ methodology (where the methodology with the lowest break-even price at each site is assumed to be adopted) for a) 100 year and b) 25 year project durations. The y-axis is restricted to \$200 per t CO₂e⁻¹ or less for ease of interpretation.

With a low carbon price of \$5 t CO₂e⁻¹ (similar to the currently trading price in the European market), only 63 Mt CO₂e⁻¹ could be sequestered over 100 years by considering environmental plantings alone (Figure 4.3a). ANR could supply 110 Mt CO₂e⁻¹ at this price, and a total of 123 Mt CO₂e⁻¹ could be sequestered if the ‘least cost’ methodology was adopted in each site (Table 4.1).

Table 4.1. Key results for the ‘least cost methodology’ scenario, assuming the most plausible establishment costs (\$2000/ha for environmental plantings, \$0/ha for ANR), 100 year project duration and three hypothetical carbon prices. Note that it is assumed the methodology (ANR or environmental plantings) with the lowest break-even price is adopted in each site.

Carbon price	Discount rate		
	1.5%	5%	10%
\$5 per t CO₂e⁻¹			
Area of carbon farming (ha)	762,700	696,700	622,700
Area (% study extent)	2.5%	2.3%	2.0%
Total carbon sequestered (Mt CO ₂ e ⁻¹)	136	123	110
Net present value (\$M)	4,230	1,513	691
Equal annual equivalent (\$M/year)	82	76	69
% NPV from ANR	84%	89%	94%
\$20 per t CO₂e⁻¹			
Area of carbon farming (ha)	1,135,300	1,212,200	984,400
Area (% study extent)	3.7%	4.0%	3.2%
Total carbon sequestered (Mt CO ₂ e ⁻¹)	223	244	194
Net present value (\$M)	5,805	2,252	983
Equal annual equivalent (\$M/year)	112	113	98
% NPV from ANR	83%	89%	94%
\$50 per t CO₂e⁻¹			
Area of carbon farming (ha)	8,038,400	10,508,600	8,106,100
Area (% study extent)	26.3%	34.3%	26.5%
Total carbon sequestered (Mt CO ₂ e ⁻¹)	1,532	1,825	1,489
Net present value (\$M)	19,720	10,956	4,089
Equal annual equivalent (\$M/year)	382	552	409
% NPV from ANR	85%	91%	96%

*Study extent is 30.6 million hectares

At a moderate carbon price of \$20 t CO₂e⁻¹, it is feasible for around 243 Mt CO₂e⁻¹ to be sequestered by a mixture of ANR and environmental plantings over 100 years. Carbon farming becomes more viable under a high carbon price of \$50 t CO₂e⁻¹ (comparable to the estimated price required to induce significant cuts in emissions, Pezzey and Jotzo, 2013), with around 1,825 Mt CO₂e⁻¹ that could be supplied using a combination of ANR and environmental plantings. ANR however could viably sequester 1,664 Mt CO₂e⁻¹ at this price,

whereas carbon farming via environmental plantings alone could supply 770 Mt CO₂e⁻¹ over 100 years. Carbon sequestration supply over the 25 year project duration was less than half of what could be achieved over 100 years for each of our hypothetical carbon prices. A total of 710 Mt CO₂e could be sequestered with a of \$50 t CO₂e⁻¹ carbon price with a combination of ANR and environmental plantings (Figure 4.3b). Supply of carbon sequestration via ANR was relatively insensitive to discounting due to negligible establishment costs, whereas a high discount rate (10%) increased the disparity evident in the economic viability of environmental plantings relative to ANR (Figure A3.3).

Economic returns under hypothetical carbon price scenarios

Carbon farming was competitive with agriculture over a fairly limited spatial extent under low and moderate carbon prices (Figure 4.4). Environmental plantings were viable over 372,500 ha (1.2% study extent), while ANR viable was over approximately twice that area (626,400 ha) with \$5 t CO₂e⁻¹. Increasing to a moderate \$20 t CO₂e⁻¹, the area viable for carbon farming increased marginally to 568,700 ha and 1,088,200 ha for environmental plantings and ANR respectively.

With a \$50 t CO₂e⁻¹ carbon price, carbon farming via ANR alone was competitive with agriculture over 9.8 million hectares, or 32.3% of the study extent, and could generate \$503 M per year over 100 years. Environmental plantings alone were viable across 3.1 million ha (10.2% study extent) and generated less than half of the economic returns possible under ANR (\$212 M per year). When we considered the ‘least cost’ methodology in each site, ANR generated the vast majority of economic returns, holding the market share of between 83 and 96% of total net present value over 100 years, under each of our carbon price scenarios and discount rates (Table 4.1). Assuming the ‘least cost’ methodology was adopted in each site, the total area viable for carbon farming and carbon sequestered at this price was actually comparable under both 1.5% and 10% discount rates with a carbon price of \$20 t CO₂e⁻¹ or more.

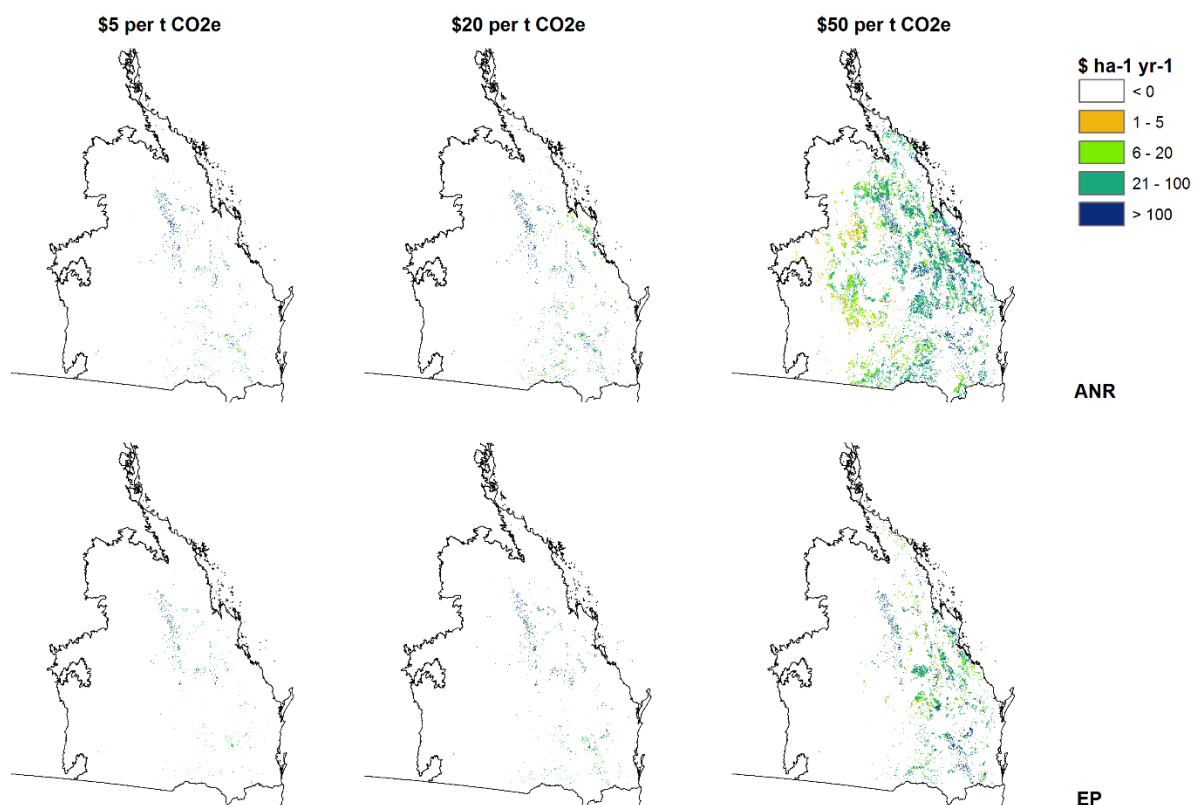


Figure 4.4. The impact of alternative establishment costs on the economic viability of carbon farming. (a) A high establishment cost for ANR (\$460/ha) results in a shift up for both the ANR and ‘least cost’ methodology supply curves. (b) Low (\$1000/ha) and high (\$3000/ha) establishment costs for environmental plantings have a large impact on the carbon supply curve for environmental plantings alone, but (c) makes minimal change to the shape of the ‘least cost’ methodology curve.

Environmental plantings were more attractive with a lower discount rate, and subsequently made up a greater proportion of the market (Table 4.1). However, a high discount rate resulted in the ‘least cost’ supply curve shifting upwards, reducing the overall viability of carbon farming (Figure A3.3).

Annual economic returns over the 25 year project duration ranged between 69% and 94% of what could be achieved over 100 years for carbon prices of \$50 and \$5 t CO₂e⁻¹ respectively. The proportion of economic returns via ANR was slightly higher over the shorter project duration (Table 4.1). However, the total area viable for carbon farming was largely unaffected by project duration (Figure A3.4).

Impact of variation in establishment costs

When we considered a high establishment cost for ANR, the average break-even price for this methodology increased from \$65.8 to \$80.0 t CO₂e⁻¹. However, this was still less than the average break-even price for environmental plantings with both low (\$83.1 t CO₂e⁻¹) and high establishment costs (\$134.4 t CO₂e⁻¹).

If a high establishment of \$460ha⁻¹ was incurred for ANR across all eligible sites, the supply of carbon via this methodology would decrease from 1,664 to 1,251 Mt CO₂e (25%) over 100 years, assuming a \$50 t CO₂e⁻¹ carbon price (Figure 4.5a). The environmental plantings supply curve was highly sensitive to variation in establishment costs (Figure 4.5b), with an increase from \$2,000ha⁻¹ to \$3,000ha⁻¹ leading to a reduction in carbon supply from 770 to 342 Mt CO₂e⁻¹ (56%) over 100 years.

However, when the ‘least cost’ methodology was considered in each site (Figure 4.4c), variation in the establishment cost for environmental plantings had a minimal impact on the overall supply of carbon, since ANR was the most viable methodology in the majority of sites in our study region. At a \$50 t CO₂e⁻¹ carbon price, a high establishment cost for environmental plantings reduced overall carbon supply from 1,824 to 1,752 Mt CO₂e⁻¹ (4%). ANR retained the market share when we considered an optimistic low establishment cost for environmental plantings (83-84% for all carbon price scenarios), and also under a high ANR establishment cost (87% for all carbon price scenarios).

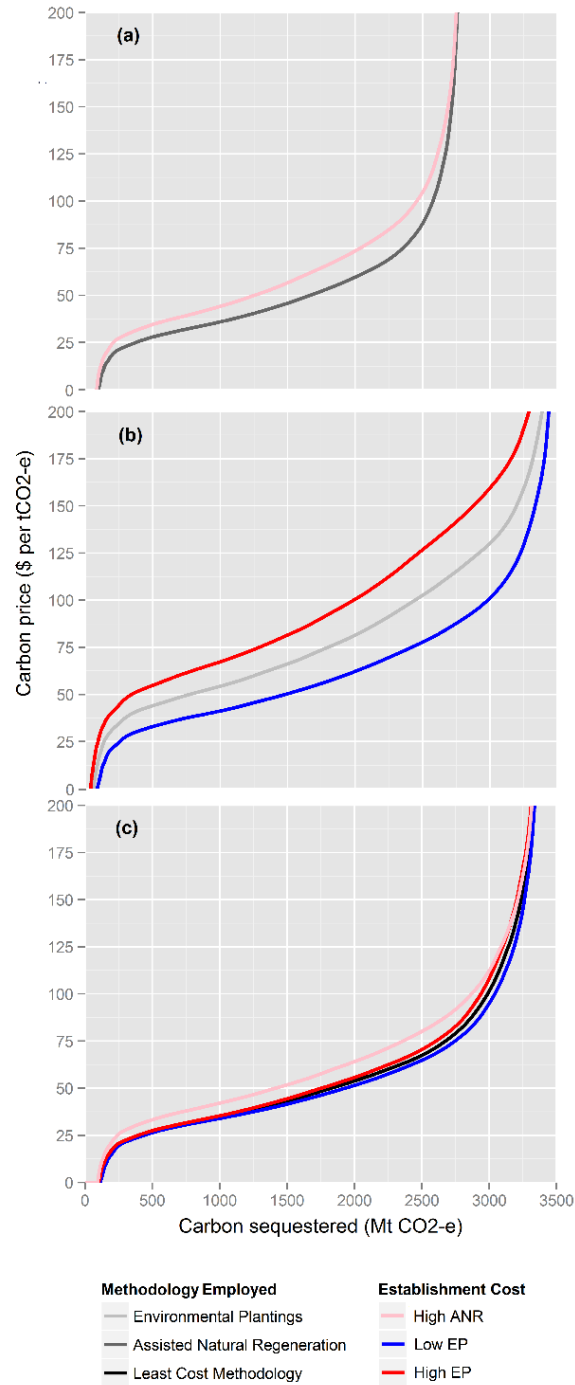


Figure 4.5. Equivalent annual returns (\$ per ha per year) for (a) ANR and (b) environmental plantings, under hypothetical carbon prices (\$5, \$20 and \$50) and 5% discount rate.

Discussion

Carbon farming in agricultural landscapes presents an important opportunity to deliver biodiversity, financial and social co-benefits alongside terrestrial carbon abatement (Lin et al., 2013). Identifying low-cost options for carbon abatement which can contribute to biodiversity conservation and other co-benefits is a high priority (Bryan et al., 2014; Gilroy et al., 2014; Nelson et al., 2008; Phelps et al., 2012a). Our study has highlighted the potential for carbon farming to establish as a viable land use in agricultural landscapes in north-eastern Australia. In particular, our research illustrates the ability of ANR to provide a cost-effective alternative to environmental plantings for sequestering carbon and providing biodiversity co-benefits in areas of intermediate levels of degradation.

In our Queensland study region, we found that carbon farming was a viable alternative to agricultural production across a fairly limited spatial extent under low and moderate carbon prices. Nonetheless, this is still a significant result as it highlights the marginal economic nature of the dominant grazing land use in some parts of the landscape. The level of payments delivered either by an appropriate incentive scheme or a market price on carbon would only need to be minimal to stimulate adoption of carbon farming in such areas, but would provide an alternative viable land use as well as deliver environmental and biodiversity co-benefits. Under a scenario where carbon is priced at a level needed to stimulate significant cuts in global greenhouse gas emissions (\$50 t CO₂e⁻¹, Pezzey and Jotzo; 2013), carbon farming could become a viable land use across up to 10.5 million hectares of agricultural landscapes (34% of our study extent), and sequester 1825 Mt CO₂e over 100 years. These findings demonstrate the importance of gaining an understanding of future land use change across a range of possible scenarios, in order to inform the development of policies which will have implications for climate mitigation, agriculture and biodiversity conservation.

Our study is the first to quantify the economic and carbon sequestration opportunities derived from assisted natural regeneration of vegetation in Australian agricultural landscapes, and one of few internationally (Birch et al., 2010; Funk et al., 2014; Gilroy et al., 2014). We found that where it is possible, ANR was almost always a more cost-effective methodology for sequestering carbon than the direct planting of trees. Environmental plantings were

competitive with ANR on areas of native pasture only when the discount rate was very low, or in the situation where the cost of establishing environmental plantings is very low (\$1000 ha⁻¹), and a high establishment cost (\$460 ha⁻¹) is assumed for ANR. It is unlikely that the establishment cost of an ANR project would be as high as \$460 ha⁻¹, particularly in our study region where regeneration can be facilitated simply by ceasing to re-clear regrowth vegetation (Dwyer et al., 2009; Fensham and Guymer, 2009).

Previous studies which have examined the economics of carbon farming via environmental plantings have found that the viability of this methodology is highly sensitive to variation in establishment cost (Polglase et al., 2013). Under one particular scenario evaluated by Polglase and colleagues (\$20 t CO₂e⁻¹, 5% discount rate), the area profitable for environmental plantings across Australia declined from 32 M ha to only 1M ha when the establishment cost was increased from \$1000 ha⁻¹ to \$3000 ha⁻¹. Our results were also sensitive to changes in the establishment cost, but the overall viability of carbon farming was affected minimally when accounting for the option of ANR. In our study region, we found that the area viable for carbon farming using environmental plantings alone halved under the same circumstances (868,600 ha to 387,700 ha). However, when we considered environmental plantings in combination with ANR, the reduction in total viable area was just under 10% (1.3 M ha to 1.2 M ha), as ANR was the more viable methodology in the vast majority of sites and contributed the greatest proportion of economic returns. Since ANR establishment costs are largely negligible, carbon sequestration supply and economic returns generated using this methodology are likely to be more robust to variable economic conditions.

The amount of carbon which could profitably be sequestered using ANR was roughly twice the amount possible if only environmental plantings were considered. A \$50 t CO₂e⁻¹ carbon price could incentivize the sequestration of 1,664 Mt CO₂e using ANR, whereas environmental plantings alone could only supply 770 Mt CO₂e over 100 years. Environmental plantings are still an important methodological option for carbon farming, particularly in areas where natural regenerative capacity has been diminished to the point where ANR is not viable. However, our findings do indicate that ANR holds considerable

potential for restoring vegetation in agricultural landscapes, particularly when the costs associated with establishing environmental plantings are high or uncertain. Our study contributes to a growing body of work which demonstrates the potential for ANR as a low-cost reforestation methodology which can benefit biodiversity conservation alongside carbon sequestration (Birch et al., 2010; Funk et al., 2014; Gilroy et al., 2014), in a literature which has so far been dominated by studies focused predominantly on environmental plantings and fast-growing monocultures (Bryan et al., 2014; Bryan and Crossman, 2013; Carwardine et al., 2015; Paul et al., 2013; Polglase et al., 2013; Renwick et al., 2014). While the biodiversity value of regrowth or secondary forests may generally not be as high as in unmodified forest (Álvarez-Yépiz et al., 2008; Gibson et al., 2011; Sampaio et al., 2007), ANR has a key role to play as a pragmatic forest restoration method which can cost-effectively sequester carbon and restore biodiversity in landscapes of intermediate levels of degradation (Gilroy et al., 2014; Shono et al., 2007). Restoration of deforested landscapes can provide crucial habitat for highly threatened species (Bowen et al., 2009; Butler, 2009; Munro et al., 2007), by supplementing important refugia (Shoo et al., 2011) and enhancing structural complexity (Munro et al., 2009; Woinarski et al., 2009). The outcomes of restoration are highly dependent on geographic and historical context (Suding, 2011), and it should be noted that ANR is most suitable for restoring areas where some level of natural succession is in progress (Chazdon, 2008; Shono et al., 2007). In our study region, it has been shown that management history can affect density of regrowth and rates of recovery in Brigalow forest (Dwyer et al., 2010a). Management of fire and grazing will play an important role in forest regeneration. In some instances fire may be useful for thinning which has been demonstrated to enhance growth rates in some forest types in Australian rangelands (Dwyer, et al., 2010b). Likewise, the management of grazing pressure will be important to allow early establishment of trees. In regions which also contain high fuel load exotic grasses, grazing may also be necessary to manage fire risk.

It is important to consider some caveats to our approach. We have assumed that monitoring costs are incurred annually, whereas carbon farming projects often have a defined crediting period (15 years for reforestation projects under the CFI) (Commonwealth of Australia, 2012c). Monitoring costs could be reduced by undertaking measurements of carbon stocks

at longer intervals (Cacho et al., 2012). Carbon farming offers considerable economies of scale (Cacho et al., 2013; Charnley et al., 2010) which we have not accounted for here. While some establishment and management costs are proportional to project size (tube stock, pest management), many components of market participation and transaction costs are fixed. It is therefore likely that carbon farming projects will be more profitable over large areas, for example where several landholders could collaborate, thereby reducing management and transaction costs (Polglase et al., 2013). Such economies of scale may be particularly significant for the ANR methodology, given there is no need for intensive restoration of vegetation and where fencing is needed, the cost will scale in proportion to project area (Schirmer and Field, 2000).

We have also assumed the full opportunity cost of agricultural production is incurred to establish carbon farming on a property, but this is likely an overestimate. Grazing by livestock is permitted on sites with environmental plantings 3 years after project establishment (Commonwealth of Australia, 2012b), and on ANR sites once forest cover has been re-established (Commonwealth of Australia, 2013a) or earlier if evidence can be provided that grazing has not prevented the regrowth of native forest (Commonwealth of Australia, 2013b). Future analyses should factor in a model of diminishing returns from grazing as a function of vegetation growth rate (Scanlan, 1991), so as to better understand the costs and benefits of ANR versus environment plantings. We have also not accounted for the cost savings associated with ceasing re-clearing of regrowth vegetation in this study, which could make ANR an even more attractive option in landscapes with high natural regenerative capacity (Dwyer et al., 2009; Gowen et al., 2012). Such an analysis will need to take into consideration the variation in regrowth clearing costs, which are dependent on the local dominant vegetation.

A potential source of uncertainty in our results is the error contained within the agricultural profitability layer (Marinoni et al., 2012), which we used as a proxy for the opportunity cost of carbon farming. Two main sources of uncertainty are inherent in spatial estimates of agricultural profitability: mapping uncertainty, and estimation uncertainty (Bryan et al., 2009). Mapping uncertainty emerges due to inaccuracies in the underlying land use data

layer, as well as the use of NDVI mapping as a proxy for agricultural yield. Estimation uncertainty is mainly due to the temporal and spatial variability of individual parameters of the agricultural commodity profit function, particularly costs, which cannot be fully captured over large geographical areas and for multiple commodities (Bryan et al., 2011). It should also be noted that Marinoni et al. (2012) derived estimates of agricultural profitability for the year 2005/2006, which was a time of drought in our Queensland case study region. The viability of the beef industry in particular was affected during this time (ABARE, 2009), resulting in some negative estimates of profitability mainly in the south-west of our study extent (Marinoni et al., 2012). Our central finding of the economic viability of ANR relative to environmental plantings should be robust to this source of uncertainty, given that 89% of our study extent is devoted to livestock grazing on native pastures, and any price volatility due to drought would affect this landscape fairly evenly. We therefore expect that the overall impacts of uncertainties in agricultural profitability estimates are low, and do not change the general conclusions of our study.

We considered the influence of project duration on the viability of carbon farming, as it is clear that in addition to policy risk and market uncertainty, long-term contracts can present a significant barrier to private landholder participation (Ando and Chen, 2011; Charnley et al., 2010; Mitchell et al., 2012). In our analysis, we found that the total area viable for carbon farming and the annual economic returns over a 25 year project duration did not differ substantially to what was expected over 100 years. Although this result was of course influenced by discounting, this effect was diminished by the predominance of the low-cost ANR methodology. Our findings suggest that the 25 year contract option would offer a similar degree of financial benefit compared to a long term contract, despite the proposed 20% discount on credits earned over a 25 year project duration (Australian Government, 2014a). However, a key consequence of this shorter contract option is that approximately half as much carbon would be sequestered relative to a 100 year carbon farming project. Reducing barriers to landholder participation in carbon farming schemes does not necessarily mean that only short contracts should be offered, as establishing a carbon farming project requires long term investment, and the carbon sequestration and biodiversity benefits from restoring vegetation increase over time. Alternatives to strict long-term permanence

obligations such as insurance policies and premiums (van Oosterzee et al., 2012) and arrangements where the contract duration is selected based on a sliding scale with the estimated risk of project reversal (Macintosh, 2013) deserve further investigation.

In our analysis, we have demonstrated the economic viability of ANR relative to environmental plantings in Australian agricultural landscapes for the first time. An important future extension to this work would be to explicitly consider the expected biodiversity benefits derived from ANR to understand how the supply of carbon sequestration and contribution to biodiversity conservation can be jointly maximized. Targeted payments to areas of high conservation value could augment economic returns from carbon farming to facilitate ‘win-win’ carbon and biodiversity outcomes (Bryan et al., 2014; Carwardine et al., 2015; Crossman et al., 2011; Nelson et al., 2008; Phelps et al., 2012b). However, trade-offs exist between biodiversity and carbon sequestration potential over both space and time. While above-ground carbon storage generally increases in a monotonic fashion as stands age and mature (Law et al., 2001; Stephenson et al., 2014), it can take substantially more time for regrowth vegetation to provide habitat values similar to mature remnant vegetation (Hatanaka et al., 2011; Woinarski et al., 2009). Carbon sequestration potential, biodiversity values and opportunity costs are unevenly distributed throughout landscapes (Crossman et al., 2011; Nelson et al., 2008). An important area of future research would be to determine how biodiversity co-benefits can be delivered alongside the economic and carbon sequestration benefits generated by the carbon market, while taking into account these potential trade-offs. In particular, future work should specifically focus on how the cost efficiencies of ANR could deliver improved outcomes for biodiversity relative to what is possible with environmental plantings, as examined by previous studies (Bryan et al., 2014; Carwardine et al., 2015; Crossman et al., 2011).

Despite the requirement for carbon sinks to remain in place over a very long timeframe, consideration of future global change and associated risks to carbon farming projects are noticeably absent in current Australian carbon farming policy. We have calculated the economic viability of carbon farming using a discounted cash flow analysis, but such a deterministic methodology is unable to account for the uncertainties inherent over long time

frames in the face of climate change (Dobes, 2008; Stafford Smith et al., 2011). Future carbon prices are subject to high uncertainty and fluctuations, as evidenced by the 2012 crash of the carbon price in the European market and recent climate policy changes in Australia. The costs and benefits of offsets are rarely considered within a climate adaptation framework, and in particular, the projected climate impacts on offset projects which aim to mitigate the effects of climate change are often unaccounted for. Unless analyzed appropriately, mitigation responses to climate change could ultimately prove to be maladaptive in the future. A key research gap therefore exists on how to analyze the costs and benefits of offset projects in the face of uncertainty.

Conclusions

Carbon farming in agricultural landscapes presents an important opportunity to deliver biodiversity, financial and social co-benefits alongside carbon abatement. Although carbon farming is only one of many policy options available to stimulate abatement of greenhouse gas emissions, it is an active policy space in Australia (Australian Government, 2014; Bradshaw et al., 2013; Bryan et al., 2014), New Zealand (Funk et al., 2014; Trotter et al., 2005), Canada (Anderson et al., 2014; van Kooten, 2000) and internationally (Benítez et al., 2007; Gilroy et al., 2014; Ma et al., 2014).

We have presented the first spatially explicit assessment of potential carbon supply via assisted natural regeneration relative to environmental plantings, in a region which is significant for its biodiversity values as well as its rural history (Dwyer et al., 2009; McAlpine et al., 2002; Seabrook et al., 2006). Our findings show that carbon farming is a viable alternative to agricultural production in the marginal areas within our study region even with low and moderate carbon prices, whereas a \$50 t CO₂e⁻¹ carbon price could make over 10 million hectares of land attractive for carbon sequestration projects.

Crucially, the vast majority of carbon sequestration and economic potential of carbon farming in our study region is derived from assisted natural regeneration. In addition to providing a low-cost option for terrestrial carbon sequestration, there is considerable potential for ANR to make an important contribution to biodiversity conservation within modified agricultural

landscapes (Bowen et al., 2009; Butler, 2009; Martin and McIntyre, 2007; McAlpine et al., 2002).

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PREAMBLE

Biodiversity offset policy and governance

...I want to ask you today what the generations to follow will say of us 40 years from now. It could be they'll report the loss of many million acres more, the extinction of species, the disappearance of wilderness and wildlife; or they could report something else. They could report that sometime around 1989 things began to change and that we began to hold on to our parks and refuges and that we protected our species and that in that year the seeds of a new policy about our valuable wetlands were sown, a policy summed up in three simple words: "No net loss."

— President George HW Bush, June 8, 1989

This thesis has so far covered native vegetation regulations (**Chapters Two and Three**) and carbon farming (**Chapter Four**). Here I introduce the third and final policy instrument examined in this thesis: biodiversity offsetting. Biodiversity offsetting is a highly contested and controversial policy instrument, which is nonetheless increasing in popularity worldwide. Like carbon farming, biodiversity offsetting is often framed in a very optimistic sense, as an “opportunity” that will provide “win-wins” for biodiversity and the economy. It also shares similarities with carbon farming as a market-based policy instrument, which aims to change behaviour by encouraging “correct” behavior (environmental protection and conservation of biodiversity) rather than discouraging or punishing poor behavior (c.f native vegetation regulations). Both instruments rely heavily on the selection of a baseline from which to measure benefits, and are subject to agency problems that can result in lower environmental outcomes than what may have been anticipated.

I began my research on biodiversity offsetting while working at the University of Queensland in early 2012, prior to beginning my PhD. I worked as part of the Australian National Environmental Research Program (NERP) Environmental Decisions Hub with Phil Gibbons, Martine Maron and Hugh Possingham to provide advice on the development of the Australian Environmental Offsets Policy under the *Environment Protection and Biodiversity Conservation (EPBC) Act 1999*. The final policy was

published by the Australian Government's Department of Sustainability, Environment, Water, Population and Communities (DSEWPaC) in October 2012.

Upon commencing my PhD in March 2013, I began a part-time internship within DSEWPaC and worked alongside Katherine Miller, James Trezise and Stefan Kraus for 9 months. During this time, I co-authored a paper (Miller et al., 2015) which describes the history behind, policy rationale and development of the current Australian EPBC Act Environmental Offsets Policy. I also contributed to the design and testing of the offset metric used to implement the policy, as outlined in **Chapter Five**. My time at the Department (now Department of the Environment and Energy) gave me an invaluable insight into the realities of policy design, implementation and evaluation from the perspective of policy makers and administrators. My PhD research on biodiversity offsetting was inspired and facilitated by this time spent within the Department.

This preamble briefly describes the history of biodiversity offset policy, and two of the broad contested issues encountered in its design, implementation and evaluation. I then introduce my case study of the biodiversity offsetting in Australia, which provides context for the remainder of this thesis.

Introduction to biodiversity offsetting

Human activities contribute to the loss of biodiversity through activities such as urban and residential expansion, built infrastructure, mining and agriculture. Concerns about the impacts of continued development on threatened species and ecosystems have prompted a range of policy responses in an effort to avert biodiversity loss (Chapters One to Four of this thesis). One such policy is biodiversity offsetting – also known as environmental offsetting

Biodiversity offsetting is an increasingly widespread policy mechanism which aims to compensate for the ecological impacts of human development (Gardner et al., 2013; McKenney and Kiesecker, 2010; ten Kate et al., 2004). In 1989, the United States adopted a goal of 'no net loss' of wetlands, which was popularized by George H.W. Bush during his 1988 presidential campaign (Boisvert, 2015; Robertson, 2000; Salzman and Ruhl, 2010). This shift in policy framing and approach for wetland management (Burgin, 2010; Kentula, 2000; Zedler, 1996) alongside the rise of market-based instruments for biodiversity conservation, were central to the emergence of biodiversity offsetting

(Boisvert, 2015; Burgin, 2008b, 2008a). Now, at least 69 countries have biodiversity offset policy in place or in development (Maron et al., 2016), and the majority of schemes have emerged within the last 10 years (Ives and Bekessy, 2015).

Despite its popularity, biodiversity offsetting is highly controversial and faces several technical, governance, social and ethical dilemmas (Maron et al., 2016). There is also little evidence available which can demonstrate what outcomes are being delivered in any jurisdiction (but see Lindenmayer et al., 2017; May et al., 2016; Pickett et al., 2013). Nevertheless, biodiversity offsetting remains a policy instrument of choice for governments worldwide seeking to balance competing economic and environmental demands, and is becoming firmly entrenched within regulatory and voluntary systems. There is therefore a need to resolve these issues wherever possible in order to deliver better outcomes for biodiversity (Maron et al., 2016).

In this thesis, I consider only two of these broad challenges described by Maron et al. (2016). *Technical* issues in biodiversity offsetting primarily concern the selection and use of metrics to represent ‘biodiversity’, how to appropriately account for biodiversity losses and gains, how to capture uncertainty and time lags, and questions around baselines and counterfactual assumptions. *Governance* issues include agency problems which may emerge as a result of counterfactual assumptions, varying motivations of policy actors, and limited capacity or incentive for monitoring, evaluation and auditing. Australia is an enthusiastic adopter of biodiversity offsetting, and so provides fertile ground for research into these challenges.

Biodiversity offsetting in Australia

In Australia, biodiversity offsetting has been used both formally and informally (i.e, in absence of a documented policy) for the last 20 years, and formal policies are now in place at the federal level and in most states and territories (Maron et al. 2015; Table P.1, Figure P.1). The 2009 Hawke review of the EPBC Act recommended that Australia “develop a national biodiversity banking (biobanking) system and standards” and that the EPBC Act should be amended to “facilitate and promote the use of biobanking as part of project approvals; and facilitate the operation of a national biobanking scheme.” (Commonwealth of Australia, 2009).

Table P.1. Key details of offset policies in Australia. Adapted from Maron et al. (2015)

	Jurisdiction								
	Commonwealth of Australia	Victoria	Queensland	Western Australia	South Australia	New South Wales		Australian Capital Territory	Southern Tasmania
Policy name	EPBC Act Environmental Offsets Policy	Native Vegetation Permitted Clearing Regulations	Queensland Government Environmental Offsets Policy	Western Australia Environmental Offsets Policy	Significant Environmental Benefit	BioBanking Assessment Methodology	Biodiversity Offsets Policy for Major Projects	ACT Environmental Offsets Policy	Southern Tasmanian Councils Authority Guidelines for the use of Biodiversity Offsets
Current policy year	2012	2013	2014	2011 - policy; 2014 - guidelines	2015	2014	2014	2015	2013
First policy year	2007 (draft)	2007	2008	2006	2002	2008	-	-	-
Policy scope	Significant impacts on MNES*	Most vegetation clearing permits	Significant impacts on threatened species or ecological communities	Significant residual impacts	Most native vegetation clearance	Threatened species and ecological communities	Threatened species and ecological communities	Significant residual adverse impacts on MNES or ACT protected matters	Residual impacts, on protected biodiversity value/s
Intended net biodiversity outcome**	“Offsets must... deliver an overall conservation outcome that improves or maintains the viability of the protected matter as compared to what is likely to have occurred under the status quo, that is if neither the action nor the offset had taken place.”	“No net loss in the contribution made by native vegetation to Victoria’s biodiversity”	“Offsets must achieve an equivalent or better environmental outcome”	Offsets “provide environmental benefits which counterbalance the significant residual environmental impacts or risks of a project or activity.”	“The Significant Environmental Benefit must outweigh the value of retaining the vegetation”	“Offsets should aim to result in a net improvement in biodiversity over time. Enhancement of biodiversity in offset areas should be equal to or greater than the loss in biodiversity from the impact site.”	Not specified	“...an overall conservation outcome that improves or maintains the viability of the aspect of the environment that is within the scope of the Policy and is impacted by the proposed action.”	“When taken as a whole, the benefit of the offset actions must be greater than the scope of the adverse impacts on biodiversity value”

* Matters of National Environmental Significance

** As per Maron et al. 2015, but updated for South Australia's 2015 Significant Environmental Benefit policy



Figure P.1. The jurisdictions across Australia in which each of the eight offset policies and guidelines examined apply. Map appears in Maron et al. (2015) and was developed by M. Evans.

Although the EPBC Act has not yet been amended to explicitly require biodiversity offsetting as part of project approvals, a formal offset policy and guide was developed over 2011-2012 to support Commonwealth regulation of impacts to Matters of National Environmental Significance (Australian Government, 2012; Miller et al., 2015).

The EPBC Act Environmental Offsets Policy (2012) was designed in response to stakeholder dissatisfaction with the Australian Government's ad-hoc approach to offsetting in use since 2001 (Miller et al., 2015). The policy was designed to adhere to the Business and Biodiversity Offsets Programme's principles of biodiversity offsets (2012), and requires that offsets deliver an overall conservation outcome that improves or maintains the viability for a protected matter as compared to what is likely to have occurred under the status quo (Maron et al., 2013; Miller et al., 2015).

A recent Australian Senate inquiry into the effectiveness of biodiversity offsetting in 2014 attracted 97 submissions from individuals and organisations, with divergent views expressed from different stakeholders on what is needed to for an effective offset policy (Martin et al., 2016). A conclusion of the Senate committee overseeing the inquiry was that there is a “...lack of evidence that offsets are effective and actually achieving their intended outcomes” (The Senate Environment and Communications References Committee, 2014). A key principle of the Australian environmental offset policy (2012) is that of transparency in decision making, and this was central in the policy design process. However, it is not yet clear whether transparency in the “front end” of policy implementation has translated to transparency in outcomes or policy evaluation.

The following three Chapters describes work I have undertaken which attempt to resolve or clarify key technical and governance challenges in biodiversity offsetting. In **Chapter Five**, I highlight the key contributions I made to its design (and in particular, the *technical* aspects of metric design, as discussed within Miller, Gibbons and colleagues (2015)). I describe the rationale behind the selection of an appropriate discount rate when determining ecological equivalence between biodiversity losses and gains, that explicitly accounts for a species risk of extinction.

Chapter Six then examines the *governance* of biodiversity offsetting. I first draw on co-authored work (Martin et al. 2016) which analysed the perceptions of stakeholders who made submissions to a Senate Inquiry into the effectiveness of biodiversity offsetting in Australia. I then introduce agency theory as a lens which may be used to explain stakeholder perceptions and behaviours, and outline how perverse outcomes are predicted to emerge for biodiversity under offset policy arrangements. I present evidence which suggests adverse selection is undermining the anticipated environmental outcomes from biodiversity offsetting in Australia (Maron et al. 2015), and consider whether moral hazard also has the potential to occur.

In **Chapter Seven**, I examine stakeholder perspectives of the efficacy of biodiversity offsetting in Australia, with a particular focus on how the policy is interpreted and implemented in practice. I draw on data collected through semi-structured interviews with

policymakers, practitioners and industry proponents who have had direct experience with the Australian Environmental Offsets Policy (2012), to examine how policy actors interact with the offset policy process. I identify a series of factors within the process through which biodiversity offsets are designed, assessed, implemented and evaluated under the EPBC Act which may enable or inhibit good environmental outcomes. I conclude by discussing what prospects exist for improved policy outcomes under current institutional and political arrangements.

CHAPTER FIVE:

Discounting extinction: ecological time preference and biodiversity offsets

A version of this Chapter was originally drafted as:

Evans, M.C., Maron, M., Gibbons, P. & Possingham, H.P. (2013). A bird in the hand is worth two in the bush: ecological time preference and biodiversity offsets. Paper prepared for the 15th Annual BIOECON Conference, Kings College, Cambridge, United Kingdom, 18 – 20th September 2013.

It also draws upon and expands on contributions made by the Candidate to the following publications, provided in **Appendix Five** of this thesis.

Gibbons, P., **Evans, M.C.**, Maron, M., Gordon, A., Le Roux, D., von Hase, A., Lindenmayer, D.B. & Possingham, H.P. (2016). A Loss-Gain Calculator for Biodiversity Offsets and the Circumstances in Which No Net Loss Is Feasible. *Conservation Letters*, 9, 252–259.

Miller, K.L., Trezise, J.A., Kraus, S., Dripps, K., **Evans, M.C.**, Gibbons, P., Possingham, H.P. & Maron, M. (2015). The development of the Australian environmental offsets policy: from theory to practice. *Environmental Conservation*, 42, 306–314.

Abstract

Biodiversity offsetting is an increasingly popular yet contentious policy instrument used to compensate for the ecological impacts of development. Much of the research focus on biodiversity offsetting has been on developing metrics for quantifying the amount of offset required to achieve a ‘no net loss’ or ‘net gain’ of biodiversity. Although the majority of metrics in use take into account the spatial dimension of impact, few consider the time required for an offset to fulfil its compensation requirements to a particular species or ecosystem. Given that the future prospects of biodiversity are inextricably linked with time, surprisingly little work has been undertaken to quantify the offset requirements for threatened species and ecosystems which face varied degrees of threat to their persistence. In this study, we introduce a novel approach to comprehensively estimate the offset requirements for biodiversity impacts over time and space, by incorporating the species’ annual probability of extinction as a discounting factor within our loss-gain metric. We find that accounting for ‘ecological time preference’ within our loss-gain metric results in greater offset requirements for threatened species and ecosystems, but offset burden can be reduced if compensation can be delivered more rapidly. Our approach can utilise readily available estimates of the probability of extinction related to the IUCN threat status of Red Listed species and ecosystems, but could benefit from refined estimates of annual extinction probabilities for specific biodiversity elements.

Introduction

The overall goal of biodiversity offsetting is to achieve a ‘no net loss’ or ‘net gain’ in biodiversity after accounting for development impacts (Bull et al., 2013; Business and Biodiversity Offsets Programme (BBOP), 2012d). This goal is usually operationalized through the use of metrics, which facilitate the quantification of biodiversity losses in the development area as well as estimating the offset requirements commensurate with the impacts.

Adequately accounting for biodiversity in offset calculations requires consideration of equity in type, time and space in order to meet the ‘no net loss’ policy objective (Salzman and Ruhl, 2000). Although the inherent uncertainties associated with measuring biodiversity (Regan et al., 2002) inevitably mean that surrogates (such as the area of habitat impacted) are often used to quantify the biodiversity losses and gains associated with offsetting, much work has been done to provide guidance on how these estimates can be strengthened (Business and Biodiversity Offsets Programme (BBOP), 2012b). Generally, it is recommended that loss-gain calculations for biodiversity offsetting should consider:

- (i) the type of biodiversity being impacted, such that ‘like-for-like’ or similar components of biodiversity are provided in compensation (Dunford et al., 2004; Quétier and Lavorel, 2011),
- (ii) the difference between status quo and the impact and offset scenario, such that offset outcomes are additional to what would have occurred anyway (Gibbons and Lindenmayer, 2007; Maron, et al., 2013),
- (iii) the likelihood of the proposed offset meeting its stated requirements (McKenney and Kiesecker, 2010; Moilanen et al., 2009),
- (iv) the threat status or vulnerability of the impacted biodiversity components (Business and Biodiversity Offsets Programme (BBOP), 2012b), and
- (v) the time delay between the ecological impact and delivery of the offset (Bekessy et al., 2010; Salzman and Ruhl, 2000).

Although there is general consensus on the importance of these factors, no ‘standard’ approach exists to quantify the ecological impacts from development, and the offset required to compensate for these impacts (Gibbons et al., 2016; McKenney and Kiesecker, 2010). The majority of existing methods used to quantify biodiversity offset requirements account in some way for exchanges in type and across space (Business and Biodiversity Offsets Programme (BBOP), 2012c; McKenney and Kiesecker, 2010), but the remaining (ii) – (v) factors are considerably less developed.

Typically, offset requirements will be adjusted according to a ‘multiplier’, to reflect those factors not explicitly accounted for in the loss-gain calculation (McKenney and Kiesecker, 2010). Multipliers are commonly used to increase offset requirements in an effort to better account for risk, species or ecosystem threat status, and time delays (Business and Biodiversity Offsets Programme (BBOP), 2012a). However, the danger in the use of offset multipliers is that their selection is arbitrary, and may lead to over- or under-estimation of offset requirements (Moilanen et al., 2009).

Several authors have introduced approaches for explicitly considering time delays and the associated uncertainty in delivering offset outcomes. A time discounting factor can be incorporated into the offset calculation to ensure that a delayed offset results in no net loss from the perspective of the present time (Denne and Bond-Smith, 2012). The discount rate can also reflect (either individually or as a composite) a range of factors including pure time preference, the balance of supply and demand, risk and change in the marginal value of investment returns (Business and Biodiversity Offsets Programme (BBOP), 2012c; Moilanen et al., 2009; Overton et al., 2013).

Moilanen et al. (2009) introduced the concept of applying a discount rate within offset calculations, to account for the loss to biodiversity during the time-lag between development impact and the offset being delivered. Overton et al. (2013) suggested that discounting should be incorporated into calculation of the ‘net present biodiversity value’ to address the risk of an offset not being delivered. Yet despite the general agreement on the concept of discounting for biodiversity offsets, no clear guidance exists for the selection of an appropriate discount rate. This is crucial, given the large impact discounting can have on

calculated offset requirements (Gibbons et al., 2016; Moilanen et al., 2009).

Time discounting has been applied in a number of biodiversity offset policies. A 3.5% discount rate is currently being applied in calculations within the UK biodiversity offsetting pilot scheme (DEFRA, 2012) to reflect societal time preference. Denne and Bond-Smith (2012) recommend a 1% discount rate for the New Zealand biodiversity offsets program. Further, they contend that that pure time preference is the only appropriate reason to apply discounting to biodiversity offsets, as “the assessment of what is an adequate offset is concerned only with the biodiversity outcome, not with the financial costs of retaining or replacing it”.

It could be argued then, that the appropriate discount rate should not necessarily just reflect just human preference for the timing of offset delivery, but rather the adequacy of a time-delayed offset for biodiversity which is (in most cases) declining over time. A critically endangered species is unlikely to utilise an offset in a future time point beyond which it is expected to remain extant. It is surprising then that the discussion around time-discounting and biodiversity offsets has not extended to how the threat status of species and ecosystems should be explicitly considered in offset calculations, to avoid the use of arbitrary multipliers.

In this study, we propose that the use of discounting in biodiversity offset calculations should be informed by species or ecosystem threat status, which can be quantified as an annual probability of extinction. We define ‘ecological time-preference’ as the use of time-discounting to reflect the heightened offset requirements of species or ecosystems which are highly vulnerable to future extinction.

We demonstrate our approach using a loss-gain metric developed by Gibbons and colleagues (2016), which has been adopted as the basis for the Australian Government’s Environmental Offsets Policy (Australian Government, 2012). Our approach can utilise readily available estimates of the probability of extinction related to the IUCN threat status of Red Listed species and ecosystems, but could benefit from refined estimates of annual extinction probabilities for specific biodiversity elements.

Methodology

Loss-gain metric

We present our approach for calculating offset requirements based on the loss-gain metric detailed in Gibbons et al. (2016). The loss-gain metric is similar to the discrete-time cumulative ‘net present biodiversity value’ presented by Overton et al. (2013), but is simplified to consider only the instantaneous losses and gains, rather than the cumulative sum of marginal losses and gains over time.

We assume that prior to calculating offset requirements, we have:

- 1) followed the mitigation hierarchy (Business and Biodiversity Offsets Programme (BBOP), 2012d) and considered any thresholds of impact beyond which offsets should not be considered (Gibbons et al., 2009; Pilgrim et al., 2013), and
- 2) have identified a ‘credit profile’ of the ecological characteristics of the impact site which must be matched on the offset site, to ensure a ‘like-for-like’ trade (Department of Environment and Climate Change (DECC), 2008; Gibbons et al., 2009).

For simplicity, we assume that the likelihood of success of the offset meeting its requirements is 100%.

We first present the discrete-time cumulative value of an offset detailed by Overton et al. (2013). We assume the impact occurs at $t = 0$, and the full value of the offset is delivered at $t = T$. No net loss is achieved when the net present value (NPV) of the difference between the discounted sum of marginal values of the offset (MV_o) and the discounted sum of marginal values of the impact (MV_i) is equal to zero:

$$NPV = \sum_{t=0}^T MV_o(t) \times D(t) - \sum_{t=0}^{\infty} MV_i(t) \times D(t),$$

where $NPV = 0$ represents ‘no net loss’.

The discounting function $D(t)$ applies a discounting factor d to the value of impacts or

offsets that occur at time t into the future:

$$D(t) = (1-d)^t.$$

We consider an exponential discounting function in this case, but other forms such as the hyperbolic (Gowdy, 2008; Henderson and Sutherland, 1996) and gamma (Weitzman, 2001) functions are possible.

The loss-gain metric developed by Gibbons and colleagues (2016) and adapted for the Australian Government's Environmental Offset Policy (2012) does not consider the discounted cumulative value of the impact and offset over time; but rather their respective values at the end-point T . This simplification means that the impacts are potentially over-valued, and offset under-valued over the time period $t = 0$ to T . However, for the purposes of the policy principles¹⁰, it was necessary to assume that biodiversity impacts are instantaneous, and that the value of the offset is not considered until T .

We calculate the value of an offset (V_o) by taking the difference between two counterfactual scenarios (Maron et al., 2013): the future value of the offset site under baseline or status quo conditions (V_b), and the future value of the offset site under the impact-and-offset scenario (V_a):

$$V_o = V_a - V_b.$$

As in the discrete-time cumulative version (Overton et al., 2013) the impact occurs at $t = 0$ and the offset is delivered at the end-time point T . The discounting factor d is applied to V_o at $t = T$. We only consider discounting on the offset side of the equation, but impacts could also be discounted based on human preference for biodiversity conservation (Gowdy et al.,

¹⁰ In particular, Policy Principle 7 requires the calculation of offsets to be “efficient, effective, timely, transparent, scientifically robust and reasonable”. Calculating the NPV cumulatively over the full time period was considered too complex for inclusion in the offset assessment guide spreadsheet: <http://www.environment.gov.au/epbc/publications/environmental-offsets-policy.html>

2010; Overton et al., 2013). No net loss is achieved when the discounted value of an offset ($V_o \times D(t)$) is equivalent to the magnitude of the original impact (V_i).

Discounting function

We account for ‘ecological time preference’ by applying a discounting factor d which is informed by the impacted species’ threat status¹¹. Extant threatened species and ecological communities listed as protected under the Australian Environmental Protection and Biodiversity Conservation Act (EPBC) Act (1999) are categorised as Vulnerable, Endangered and Critically Endangered.

Similar to the United States *Endangered Species Act 1973*, there are no guidelines under the EPBC Act which quantify the probability of extinction for species listed under different threat categories (Regan et al., 2013). We therefore adopted the IUCN (2001) criteria for each threatened species category (Table 5.1), and converted the probability of extinction in the wild into our annual discounting factor d by taking the geometric mean:

$$d = 1 - \sqrt[y]{P},$$

where P = the minimum probability of extinction in y years.

The geometric mean is commonly used to model species population growth curves (Bascompte et al., 2002), and so provides a better approximation of the annual probability of extinction than the arithmetic mean.

¹¹ The Australian Government’s EPBC Act Environmental Offsets Policy requires offsets to be “in proportion to the level of statutory protection that applies to the protected matter” (Policy Principle 3)

Table 5.1. IUCN (2001) criteria translated into the minimum annual probability of extinction for each major threatened species category, used as the discounting factor d for calculation of biodiversity offsets

Threat Status	IUCN Criteria for Probability of Extinction in the Wild	Annual Probability of Extinction (Geometric mean)
Critically Endangered	At least 50% in 10 years	6.7%
Endangered	At least 20% in 20 years	1.1%
Vulnerable	At least 10% in 100 years	0.1%

Case study

To illustrate our approach, we present a hypothetical case study for a threatened species native to Australia: the red-tailed black cockatoo (*Calyptrorhynchus banksii graptogyne*), which listed as Endangered under the EPBC Act. The key concern for this species is habitat loss and fragmentation driven by agricultural expansion in south-eastern Australia, which has impacted on the availability of its preferred food resources (Maron et al., 2010; Maron and Fitzsimons, 2007).

Consider an impact where 20 hectares of feeding habitat for *C. banksii graptogyne* is removed as part of a development. The impact occurs on a site which encompasses 100 hectares of suitable habitat in total. Ongoing pressures on the red-tailed black cockatoo's habitat mean that it is expected to decline at a rate of 5% per year, based on estimates of land conversion in the surrounding landscape. The impact is to occur at $t = 0$ and it is proposed that an offset is initiated at this time by planting suitable vegetation at an adjoining site. We assume that the value of the offset site increases according to a power function, $A_x(t) = A_x \times t^z - A_x(T)$ such that it reaches its maximum value at $t = T$, and the growth parameter $z = 0.10$ (Figure 5.1).

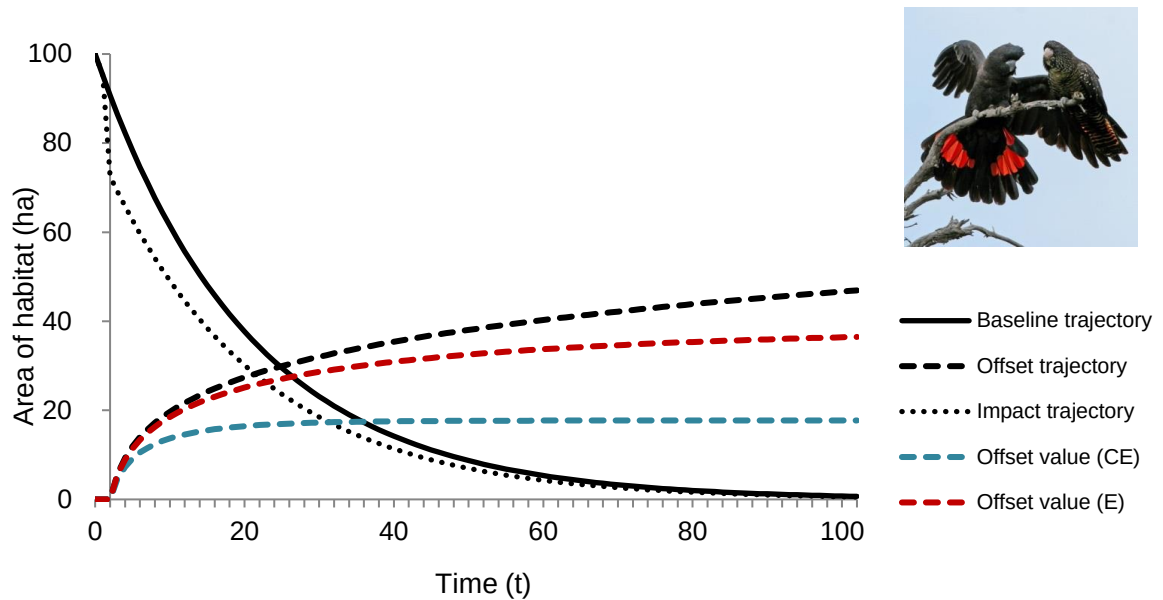


Figure 5.1. Representation of the cumulative impact and offset value for our hypothetical case-study of the red-tailed black cockatoo (*Calyptorhynchus banksii graptogyne*, see inset). Under the baseline/status quo scenario, feeding habitat is lost at a rate of 5% per year due to external pressures. The time of impact is $t=0$, where 20 hectares of feeding habitat is removed due to development. Note that we do not apply a discounting factor to the impact trajectory, although this is possible. The figure illustrates the cumulative value of the offset after it is first planted at $t = 0$, and the trajectories as discounted according to whether the species is endangered (EN, $d = 1.1\%$) or critically endangered (CE, $d = 6.7\%$).

We consider two scenarios: first, where the value of the proposed offset for *C. banksii graptogyne* is discounted according to its current threat status (Endangered, $d = 1.1\%$). Secondly, the offset value is calculated according to the Critically Endangered threat status ($d = 6.7\%$) to illustrate the impact on our findings. We calculate the offset requirements based on (1) our loss-gain metric (Gibbons et al. 2016) (2) the discrete-time cumulative function (Overton et al., 2013), and (3) a standard ‘multiplier’ approach.

Results

Using our loss-gain metric, we found that the 20 ha impact on endangered *C. banksii graptogyne* habitat could be offset to a no-net loss standard after 100 years by planting 60 ha

of land with new vegetation (Table 5.2). When the cumulative value of the impact and offset were taken into account (Overton et al., 2013), we found that a slightly larger offset area of 80 ha was recommended. When we considered the case where *C. banksii graptogyne* was actually categorised as Critically Endangered, our results based on the alternative approaches differed substantially. The offset required to achieve no-net loss as calculated by our loss-gain metric exceeded 10,000 ha in this case, which would likely be considered infeasible. Based on the discrete-time cumulative calculation, the offset requirement was 170 ha. The offset areas recommended by the standard ‘multiplier’ approach were lower than the estimated requirements based on the loss-gain metric as well as the discrete-time cumulative calculation (Table 5.2).

Discussion

We have presented a theoretical framework for explicitly accounting for the threat status of species and ecosystems in the calculation of biodiversity offset requirements. Offset policies will often adjust offset requirements for threatened biodiversity according to an arbitrary multiplier (McKenney and Kiesecker, 2010), which may seem intuitive but ultimately has little scientific or economic basis. Time discounting has been incorporated into some biodiversity offset policies (DEFRA, 2012; Denne and Bond-Smith, 2012), but is generally used to reflect pure-time preference or the risk of offset failure without differentiating whether the biodiversity components in question are threatened, and to what degree.

Considering a discount rate for biodiversity offsetting which is informed by species’ annual probability of extinction offers a several advantages over the other approaches seen in policies so far. Our approach explicitly considers ‘ecological time preference’ for offset delivery: specifically, the risk that a species will go extinct prior to an offset becoming available for use. Arguably, human time preference for offset delivery should be accounted for separately to the species’ time preference for the offset, given that the first consideration in biodiversity offset calculations should be as to whether an offset adequately satisfies a ‘no net loss’ criterion (Business and Biodiversity Offsets Programme (BBOP), 2012d; Denne and Bond-Smith, 2012).

Table 5.2. Results from our hypothetical case study of the red-tailed black cockatoo (*Calyptorhynchus banksii graptogyne*), as calculated using the loss-gain metric, discrete-time cumulative function and basic multiplier approach. Offset requirements are calculated over 100 years, under scenarios where the species is either endangered (EN) or critically endangered (CE). Cells in grey indicate that the variable (row) is not considered in the corresponding offset calculation (column).

		Loss-gain metric (Gibbons et al. 2016)		Discrete-time cumulative function (Overton et al. 2013)		"Multiplier" approach	
		EN ($d=1.1\%$)	CE ($d=6.7\%$)	EN ($d=1.1\%$)	CE ($d=6.7\%$)	EN ($A_1 \times 2$)	CE ($A_1 \times 3$)
Impact	A_1	20.0	20.0	20.0	20.0	20.0	20.0
Cumulative impact value	$\sum_{t=0}^{\infty} MV_i(t) \times D(t)$			36.1	36.1		
Offset value required for NNL	A_0	60.0	infeasible	80.0	170.0	40.0	60.0
Discounted offset value	$A_0 \times D(t)$	20.1	-				
Discounted cumulative offset value	$\sum_{t=0}^T MV_o(t) \times D(t)$			36.5	38.0		

Pure time preference for biodiversity offsets could still be considered, but it first must be understood whether the biodiversity in question will be adequately compensated in order to remain extant over the time in which ecological impacts are incurred. Differentiating the discounting factor based on threat status could also reflect human time preference; given the higher value which society places upon species and ecosystems at risk. Applying a generic discount rate consistently across all biodiversity impacts suggest that humans place no greater preference over a critically endangered species compared to a common species. Of course, public preferences for biodiversity are influenced by a range of factors (Gowdy et al., 2010), but arguably these considerations should be accounted for after offset requirements are quantified.

Higher discount rates for a threatened species could further incentivise efforts to avoid, mitigate and restore impacts prior to choosing to offset¹². Alternatively, the threatened species' 'preference' for timely compensation will require the development to provide an offset (which is scaled to the degree of species threat) which would *ceteris paribus* be larger than for a non-threatened species. In particular, a high discount rate could incentivise a more rapid delivery of an offset. While reducing the time taken to deliver an offset in itself only increases the offset requirement, if the rate of restorative works or securing an averted loss offset happens more rapidly during this reduced timeframe, this could result in considerable cost savings for the proponent, and provide compensation for the threatened species sooner. Gibbons et al. (2016) found that a low discount rate was a key factor in contributing to whether a development is likely to achieve no net loss. Based on our approach, this translates to offsets being more viable for species with a lower level of endangerment.

We have adopted the IUCN criteria on species probability of extinction to inform the discounting factor calculation of offset requirements, given this is the international benchmark for threatened species categorisation (IUCN, 2001). However, locally specific estimates of species extinction probability, or estimates informed by population viability

¹² Anecdotal evidence from the implementation of the Australian Government's Environmental Offsets Policy suggest that a greater emphasis is being placed on threatened species listings, to ensure species are placed in the most appropriate threat category; given the considerable influence the discount rate (particularly for Critically Endangered species) has on biodiversity offset requirements

analysis (Boyce, 1992; McCarthy et al., 2003), or expert opinion (Carwardine et al., 2012; Regan et al., 2013) could easily be incorporated if they were readily available. In contrast to a basic ‘multiplier’ approach, offset requirements discounted according to species probability of extinction can readily updated as improved information on species risks emerge. This may be difficult to achieve if specific multipliers were set in policy according to categorical criteria only.

Our case study illustrated the magnitude of difference in offset requirements for biodiversity where discounting is informed by species’ probability of extinction. The multiplier approach recommended offsets much smaller in size than what was calculated under the loss-gain and discrete-time cumulative approaches, and importantly, there was no way to determine whether these quantities would result in a no-net-loss outcome. The net present value of the offset could be estimated using the loss-gain and discrete-time cumulative approaches, but each differed in their treatment of impact and offset values. The offset recommended by each approach was similar in magnitude when the discounting factor was relatively small ($d = 1.1\%$, endangered status), but the loss-gain metric was particularly sensitive to the critically endangered discounting factor ($d = 6.7\%$). We considered offsetting over a 100 year time period in this case, but limiting consideration of offsets to a shorter timeframe could represent a more realistic scenario, and reduce sensitivities in the loss-gain metric¹³.

However, certain offsetting scenarios may actually be more accurately represented by the instantaneous loss-gain metric, whereby the value of an offset is close to zero until a future time point. For example, the red-tailed black cockatoo may not be able to utilise feeding trees until they reach maturity around 100 years after they are planted (Maron et al., 2010), so the cumulative benefit function may not be appropriate in this case. More work is needed to fully explore a range of impact and offset value functions (Arponen et al., 2005), and the implications for ‘offsetability’ (Pilgrim et al., 2013) under different circumstances.

Biodiversity offsetting is a policy which has come under fire due to the poor reported performance to date (Brown et al., 2013; Quigley and Harper, 2006), the lack of transparency

¹³ The maximum period of time over which offsets can be considered under the the Australian Government’s Environmental Offsets Policy is 20 years.

and evaluation of outcomes (Edgar et al., 2005; Madsen et al., 2010) and the widespread use of metrics that are not fit for purpose (Maron et al., 2013). Many of the challenges presented by biodiversity offsetting will not be solved by improved metric design (Walker et al., 2009), yet the metric used to calculate offset requirements has such a fundamental role in determining whether the ‘no net loss’ objective can be achieved. Accounting for an ecologically relevant discounting factor in the calculation of offset requirements would address the need to explicitly consider the interaction between time and species endangerment, which has so far eluded biodiversity offset policy.

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We wish to thank James Tresize and Stefan Kraus from the Australian Government’s Department of Sustainability, Environment, Water, Population and Communities for helpful discussions around this work. This research was conducted with the support of funding from the Australian Government’s National Environmental Research Program. M.C.E was supported by an Australian Postgraduate Award and a CSIRO Climate Adaptation Flagship scholarship.

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CHAPTER SIX:

Governance of biodiversity offsetting

This Chapter draws upon and expands on contributions made by the Candidate to the following publications, provided in **Appendix Five** of this thesis.

Martin, N., **Evans, M.C.**, Rice, J., Lodhia, S. & Gibbons, P. (2016). Using offsets to mitigate environmental impacts of major projects: A stakeholder analysis. *Journal of Environmental Management* 179: 58–65.

Maron, M., Bull, J.W., **Evans, M.C.** & Gordon, A. (2015). Locking in loss: Baselines of decline in Australian biodiversity offset policies. *Biological Conservation* 192: 504–512.

Maron, M., Ives, C.D., Kujala, H., Bull, J.W., Maseyk, F.J.F., Bekessy, S., Gordon, A., Watson, J.E.M., Lentini, P.E., Gibbons, P., Possingham, H.P., Hobbs, R.J., Keith, D.A., Wintle, B.A. & **Evans, M.C.** (2016). Taming a Wicked Problem: Resolving Controversies in Biodiversity Offsetting. *BioScience* 66(6): 489–498.

Introduction

“The process of initiating multiactor governance is not politically neutral, nor does it exist in a vacuum. It rather reflects competing interpretations of the performance of the polity: its effectiveness, efficiency, equity, and attempts by political actors to influence the direction of political change”

Peter Newell, Philipp Pattberg, and Heike Schroeder (2012). Multiactor Governance and the Environment, *Annual Review of Environment and Resources* 37, 365–387

Environmental governance refers to the institutions and organisations that guide the implementation of environmental policies, as well as the actors that influence and are influenced by these institutions and organisations (Adger and Jordan, 2009; Agrawal and Lemos, 2007; Lemos and Agrawal, 2006; Lockwood et al., 2010; Newell, 2008). Effective biodiversity conservation requires analysis of the incentives and actions of multiple actors, and an understanding of how these arrangements play out across multiple scales and contexts (Agrawal and Ostrom, 2006). Thus far, discussion of biodiversity offsetting in the literature has primarily focused on policy principles and metric design, with far less attention on political and social dimensions (Apostolopoulou and Adams, 2017; Benabou, 2014; Maron et al., 2016; Penca, 2013, 2015). The governance of biodiversity offsetting ultimately determines how, and to what extent, policy objectives are met (Lemos and Agrawal, 2006; Newell, 2008), and so is a crucial area for research.

As discussed in **Chapter One**, ‘new’ environmental policy instruments such as biodiversity offsetting operate through the participation of multiple actors, each with their own objectives and interests (Agrawal and Lemos, 2007; Jordan et al., 2005; Lemos and Agrawal, 2006; Liverman, 2004; Newell et al., 2012). How successful and efficient these governance arrangements are in a particular jurisdiction will be highly dependent on local context – in terms of who is involved, and how power and authority are distributed between different actors. In developing nations such as Madagascar, Mongolia and Uzbekistan, businesses and non-government organisations have driven offset policy design, and market participation is on a voluntary basis (Benabou, 2014; Bidaud et al., 2015, 2017; Bull et al., 2014; Business and Biodiversity Offsets Programme (BBOP),

2012d). In contrast, biodiversity offsetting in Australia and other developed nations operates within an existing regulatory environment, where businesses are required by government to deliver offsets as conditions of approval for development activities (Fox and Nino-Murcia, 2005; Miller et al., 2015).

Although market-based instruments (MBI's) such as biodiversity offsetting are often promoted as more efficient or 'easier' than traditional 'command and control' regulation, in reality they may require as much, if not more, regulatory settings than regulation on its own. Moreover, the number of actors involved in MBI's can increase the transaction costs *vis-à-vis* the direct 'regulator-regulated' relationship. Far from a simple trade in ecological values from one place to another, biodiversity offsetting operates within highly complex governance arrangements (Boisvert, 2015; Coggan et al., 2013a; Whitten et al., 2012), and should be analysed accordingly. Direct observation of how actors interact within a particular institutional and political context is difficult, hence I draw on a combination of stakeholder perceptions (Martin et al. 2016), quantitative estimates of policy scenarios (Maron et al. 2015) and a widely used theoretical framework (Eisenhardt, 1989; Laffont and Martimort, 2009; Shapiro, 2005) to deliver a preliminary description of biodiversity offset governance.

In this Chapter, I first introduce some of the policy actors involved in biodiversity offsetting, by drawing on work co-authored with Martin et al. (2016) which analysed stakeholder perceptions captured in submissions to an Australian Senate Inquiry in 2014. I then use agency theory as a framework for understanding the perceptions and behaviours of policy actors who are engaged with biodiversity offset policy. I outline how perverse outcomes can emerge for biodiversity under conditions predicted by agency theory, including adverse selection and moral hazard. I then present evidence first documented in Maron et al. (2015) which suggests adverse selection is undermining the potential environmental outcomes from biodiversity offsetting in Australia, by comparing the baseline assumptions of offset policies in eight Australian jurisdictions with estimated rates of deforestation. I conclude by considering how moral hazard may occur under common biodiversity offset policy arrangements: specifically, where there is asymmetric information and insufficient monitoring and public scrutiny.

Stakeholder perceptions of biodiversity offset policy

There is evidently a need to better understand the governance of biodiversity offsetting from a holistic perspective (Boisvert, 2015), by investigating the motivations, incentives, and challenges faced by diverse policy actors. However, the academic literature on biodiversity offsetting has so far paid little attention to the stakeholders involved in designing, implementing and evaluating offset policies (Bidaud et al., 2017; Coggan et al., 2013a; Kaplowitz et al., 2008). Exceptions to this include work by Hayes and Morrison-Saunders (2007), who surveyed government agencies, regulators, consultants and industry proponents involved with biodiversity offsetting in Western Australia. Respondents in this study generally supported the use of offsets in-principle, but expressed doubts over the likelihood of achieving a ‘no net loss’ or ‘net gain’ outcome in practice. Brown et al. (2014) found similar results in New Zealand where ecological practitioners expressed strong support for the use of offsetting, but raised several organizational and informational barriers to effective implementation. Respondents also expressed concern about the lack of regulatory follow-up and long-term security of offsets (Brown et al., 2014).

Given the complexity of biodiversity offset governance, it is useful to analyse these arrangements in a structured way which recognizes the varying degrees of interest and participation that diverse actors have in different stages of the offset policy process. In our published paper (Martin et al., 2016), we made use of a recent Australian Senate inquiry into the effectiveness of environmental offsets to investigate the views of stakeholders who had made written submissions. The Senate inquiry specifically focused on “the history, appropriateness and effectiveness of the use of environmental offsets in federal environmental approvals in Australia”, incorporating: 1) policy principles, 2) the development of offset proposals, 3) processes used to assess and approve offsets, 4) the adequacy of monitoring and evaluation, and 5) outcomes and results delivered by approved offsets (The Senate Environment and Communications References Committee, 2014). A total of 97 submissions were made by individuals and organizations, including 47 non-government organisations (NGOs; conservation, education, environment legal services, indigenous bodies), 9 industry peak bodies (business, mining, farming and aquaculture), 7 government organisations (political party, city council, regional

development agencies, indigenous land and advisory, environment), and 5 business firms (consultants, developers), with 29 individuals providing written submissions.

We used a structured content analysis (Denzin and Lincoln, 2005; Miles et al., 2013) to examine the arguments made in stakeholders' submissions, and coded to five broad themes reflecting broad stages in the offset policy process (Table 6.1). Within each theme, statements were then coded to subthemes which reflected support for key offset principles drawn from the literature (e.g. additionality, no net loss; see Figure 1, Martin et al. (2016) **Appendix 5**), considerations which stakeholders considered should or should not be included within offset policy processes (e.g. include high quality scientific information, exclude existing protected areas as possible offsets), and any comments relating to known offset success or failure. Despite the constrained nature of the public inquiry and the unbalanced response rate from different stakeholder groups (e.g. 47 NGO responses versus 14 from industry peak bodies and business firms), some useful insights emerged.

First, there was a perception by NGO and individual (often scientists) respondents that the mitigation hierarchy was not being stringently considered in development decisions, with offsets being favoured over avoidance, minimization and rehabilitation measures (Clare et al., 2011; Clare and Krogman, 2013). In their final report, the Senate committee expressed concern over evidence presented by submissions that the mitigation hierarchy “is not being rigorously applied and that there is insufficient emphasis on avoidance and mitigation measures” (The Senate Environment and Communications References Committee, 2014). Many environmental stakeholders submitted that an assessment should be conducted as part of offsets pre-testing to assure that all environmental impact avoidance, minimization and restoration options have been rigorously canvassed prior to offsets planning.

Second, the need for greater consistency between local, state and federal environmental offset policies was emphasized, in particular by regulated industries. The National Farmers' Federation described that better policy alignment “will avoid the current confusion of separate offset requirements by the different jurisdictions.” (Submission 15, NFF, 3rd April 2014, pg. 2). The Chamber of Minerals and Energy Western Australia stated that “any offsets requirements imposed under both State and Commonwealth

Table 6.1. Summary of stakeholder statements relating to offsets. From Martin et al. (2016)

Theme	Number of statements by stakeholder group (n = 742)						Theme total (%)
	Businesses	Business peak bodies	Government	NGO	Individuals	Theme total (no.)	
1. Principles and Policy	8	24	29 ^b	162 ^{a b}	41	264 ^c	36%
2. Proposal Development and Submission	11 ^b	25 ^b	13	147 ^a	45 ^b	241	32%
3. Proposal Assessment and Approval	1	5	7	48 ^a	19	80	11%
4. Monitoring and Evaluation	4	7	6	90 ^a	19	126	17%
5. Outcomes and Results	1	2	4	18 ^a	6	31	4%
Stakeholder Total (No.)	25	63	59	465	130	742	
Normalized Total (No.)	5.00	7.00	8.43	9.89	4.48	7.65	
Stakeholder Total (%)	3%	8%	8%	63%	18%		100%

Notes:

^a Indicates lead stakeholder in theme.

^b Indicates stakeholder primary area of interest.

^c Indicates leading theme.

legislation should be complementary and should not impose additional costs on industry” (Submission 33, CME, April 2014, pg. 6).

There were however concerns expressed about the Australian Government’s proposed ‘one stop shop’ for environmental approvals, which was intended to reduce policy duplication and reduce regulatory burden for industry (Australian Government, 2014b; Standing Committee on the Environment, 2014). A number of stakeholders criticized the government’s plan to accredit state and territory planning processes under the EPBC Act, since these are of a lower standard than the federal policy.

Third, submissions from peak bodies, NGOs and individuals argued for a more ‘strategic’ approach to offsetting, which can account for cumulative impacts from developments over larger spatial and temporal scales than are currently considered under the standard piecemeal, ‘project by project’ approach. Achieving this holistic view of offsetting would require some harmonization of offset policies across jurisdictions (as per above), and the integration of offsetting within the broader context of strategic landscape planning. There is a role for government to provide guidance and information to facilitate this strategic approach, which the The Environment Institute of Australia and New Zealand argued would assist with “the identification and delivery of offsets that provides proponents with greater certainty of their required contributions” (Submission 88, EIANZ, April 2014, pg. 8).

Finally, all stakeholder groups expressed concern over the current lack of transparency and availability of information relating to offset assessments, approvals and delivery of environmental outcomes. NGOs such as Friends of the Earth argued that the current approval process prevents rigorous public scrutiny, since “...[offsets] are imposed as post-approval conditions and are therefore not subject to review and public input” (Submission 58, Friends of the Earth Australia, April 2014, pg. 6). Once offsets are approved and presumably established, there is similarly limited information the environmental performance of offsets, due to the absence of a public register of offsets and “systemic failures in monitoring and performance” (Submission 72, Environmental Defenders Office (Victoria), 10th April 2014, pg. 6). The Outcomes and Results theme in our analysis attracted the lowest number of stakeholder statements (4%), which is due in part to it being “too early” to tell whether offsets

approved since the introduction of the 2012 policy had generated environment outcomes (Submission 88, EIANZ, April 2014, pg. 9). However, since environmental offsetting has been in operation informally since the introduction of the EPBC Act in 2000 (Miller et al., 2015), the chronic lack of information available to conduct a meaningful evaluation was criticized by several stakeholders. An independent audit concluded in 2014 that:

“...compliance monitoring undertaken by the department has, generally, been insufficient to provide an appropriate level of assurance of proponents' ongoing compliance with their conditions of approval” (Australian National Audit Office (ANAO), 2014).

Environmental NGOs and industry peak bodies alike supported the development of a publicly available offsets register, which suggests that greater transparency and information could provide benefits for multiple interests. As noted by the Senate committee (2014), the Australian Environmental Offsets Policy (2012) states that all offsets (including their locations and details of protected matters and management activities) will be registered and information made publicly available, which will “allow strategic planning, and streamline processes with state and territory requirements and schemes”. However, such a system has not yet become publicly available.

Understanding the behavior of policy actors in biodiversity offset governance

A limitation with the use of data that captures ‘perceptions’, particularly in the context of a public inquiry, is that the statements made by stakeholders do not necessarily match their true intent or behavior (Bennett, 2016). It is notable that businesses (3%, Table 6.1) and business peak bodies (8%) made up only a small subset of the total number of submissions, hence it is likely the views of a number of businesses subject to biodiversity offset policy were not explicitly captured in the public inquiry. More in-depth research would be necessary to capture this data (such as through semi-structured interviews, e.g Brown et al. 2014), but a theoretical framework can also assist in predicting and explaining the likely behavior of policy actors.

MBI's such as biodiversity offsetting aim to align the incentives of policy actors to encourage policy compliance (Jordan et al., 2003a; Lemos and Agrawal, 2006), in contrast to 'command and control' regulation where compliance is in theory more fervently enforced by government. Principal-agent theory (Laffont and Martimort, 2009), otherwise known as agency theory (Eisenhardt, 1989; Shapiro, 2005), provides a useful lens through which to examine the efficacy of policy implementation in the context of multi-actor governance (Howlett et al., 2009) and has been widely used to analyse the behavior of actors within a contractual relationship.

A biodiversity offset involves a full or partial transfer of rights and responsibilities from one party (the developer, or offset buyer) to another (the offset seller). This transfer is codified in a legal contract, and compliance is enforced by government (the regulator). In this simplified situation, the developer is the principal of contract between it and the offset seller (the agent). The regulator is the principal of the relationship between it and the developer, hence the developer is simultaneously a principal and an agent (Figure 6.1). Economists have typically drawn on principal-agent theory to analyse contractual arrangements between a small number of actors (Laffont and Martimort, 2009), whereas agency theory broadens this perspective to incorporate a wider range of relationships involving non-monetary incentives (Eisenhardt, 1989; Shapiro, 2005).

Kiser (1999) defines agency theory as "a general model of social relations involving the *delegation of authority*, and generally resulting in *problems of control*, which has been applied to a broad range of substantive contexts". Agency problems can arise in any contractual arrangement between one or more actors, due to asymmetric access to information, uneven sharing of risks between these parties, and the diverging interests between the principal(s) and agent(s). Biodiversity offsetting is thus exposed to agency problems, including adverse selection and moral hazard (Maron et al., 2016).

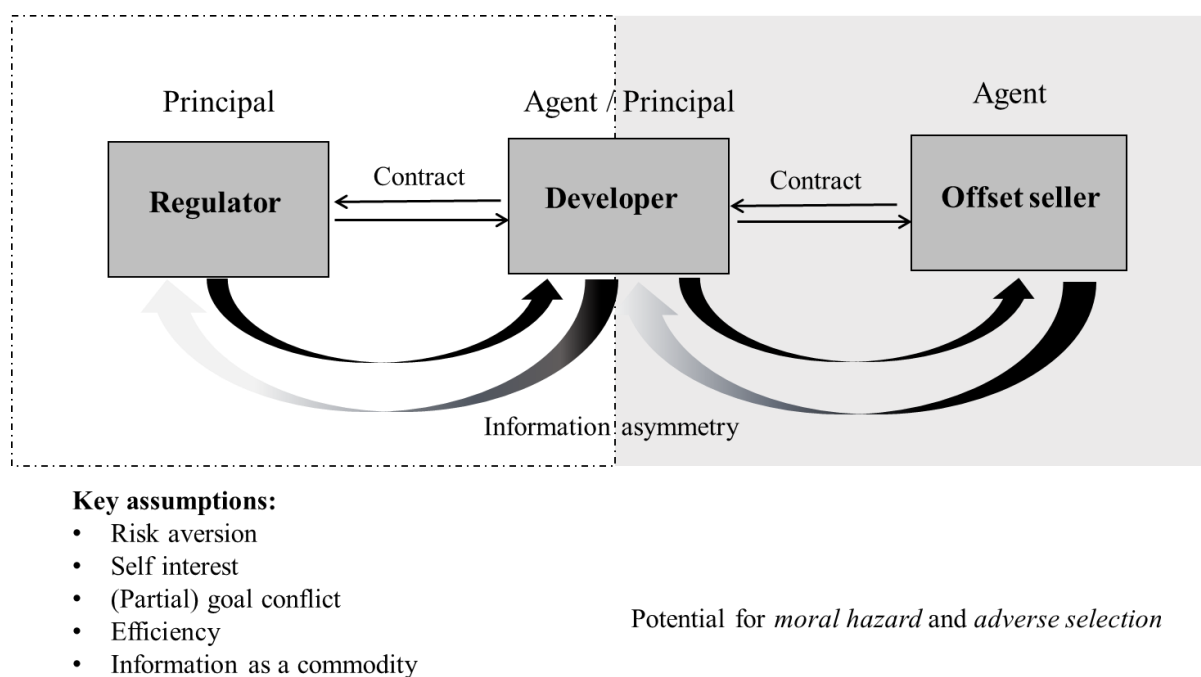


Figure 6.1. Simple representation of a principal-agent relationship and its key assumptions.

Market based instruments aim to select the units where the greatest benefit per unit cost can be delivered – that is, where the benefits delivered by the market are *in addition* or *additional* to what would have happened in absence of the policy. Determining what is additional requires an estimate of the counterfactual scenario, which is by definition not observable (Ferraro, 2009), and therefore has the potential to be gamed (Maron et al., 2016; Salzman and Ruhl, 2010). Adverse selection has been documented in carbon offset markets (Burke, 2016; Bushnell, 2010), payments for ecosystem services schemes (Ferraro, 2008) and species banking in the United States (though additionality is not a stated policy goal in this case, Fox and Nino-Murcia, 2005; U.S. Fish & Wildlife Service, 2003). In the case of biodiversity offsetting, there's an incentive for the agent to inflate the counterfactual scenario, such that it appears that they are delivering a higher environmental outcome offset than in reality. Where the counterfactual scenario, or 'crediting baseline', assumes a future trajectory of biodiversity decline, the intended net outcome of a biodiversity offset is maintenance of that declining trajectory. If the rate of decline of the crediting baseline is implausibly steep, biodiversity offsetting can exacerbate biodiversity decline (Figure 6.2).

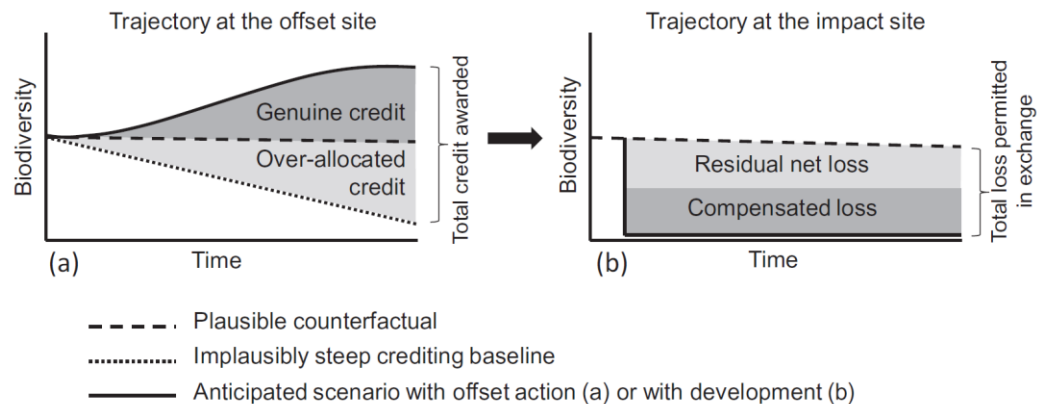


Figure 6.2. Example of an offset trade in which the use of an implausibly steep crediting baseline (fine dotted line) results in inflation of offset credit (a) which is then exchanged for an equivalent loss (b). For simplicity, we assume the plausible crediting baseline and debiting baselines are identical. The total credit allocated in (a) comprises both the genuine gain compared to the plausible counterfactual (dark grey) and the overallocated credit (light grey), resulting in a residual net loss equivalent to the amount of credit that was overallocated (light grey shading in (b)). From Maron et al. (2015).

Evidence of adverse selection occurring under Australian biodiversity offset policies

In our published paper (Maron et al., 2015), we examine the often-implicit assumptions used to inform counterfactual scenarios in Australian offset policies. Specifically, we (1) explore the extent to which a baseline from which to measure offset gains is clearly articulated for nine offset policies in eight Australian jurisdictions; and (2) for cases where a baseline is not explicitly stated, we explore whether a declining baseline is implied based on the rules specified for calculating offset credit under that policy and/or actual offset trades that have occurred, and if so, estimate the implied rate of decline; and (3) compare the baseline rates of decline to recent rates of vegetation loss for each of the jurisdictions to identify the degree to which the decline assumptions inherent in the policy correspond to available evidence of vegetation loss.

To evaluate the plausibility of assumed crediting baselines in offset policies, we compared the rates of future change implied by the crediting baselines built into each credit calculation approach with recent rates of change in woody vegetation extent in each jurisdiction (as per methods in **Chapter Two**). A comparison of implied crediting baselines with rates of change in extent allows an examination of whether there is any relationship between the two, as might be expected if crediting baselines were drawn from the assumption that the future would reflect recent trends.

We found that the crediting baselines used in policy implementation varied among and within the jurisdictions from 0.36% and 4.2% decline per annum. In comparison, recent rates of woody vegetation loss were generally much lower and varied between 0% and 0.5% loss per annum (Figure 6.3). On average, crediting baselines were over five times the mean observed rate of forest loss, ranging from between 1.6 (Queensland) and 16 (South Australia) times higher than the observed rates of change (excluding ACT, with zero observed woody vegetation loss). There was no correlation between a jurisdiction's observed rate of forest loss and its crediting baseline (Pearson's $r = 0.39$); therefore, for crediting baselines to relate predictably to recent rates of change, prevailing processes of condition decline would have to vary considerably among jurisdictions.

Our results show that vegetation decline scenarios used as crediting baselines in Australian biodiversity offset policies are much higher than recent rates of vegetation loss. Thus, not only are biodiversity declines entrenched as the net outcome across impact-offset trades, but those trades also risk exacerbating background rates of decline by overestimating averted loss offset credit which can be exchanged for equivalent losses. This assumption of ongoing biodiversity decline is at odds with federal and state government commitments to reduce or reverse the decline in native vegetation extent and quality (COAG Standing Council on Environment and Water, 2012; Evans, 2016).

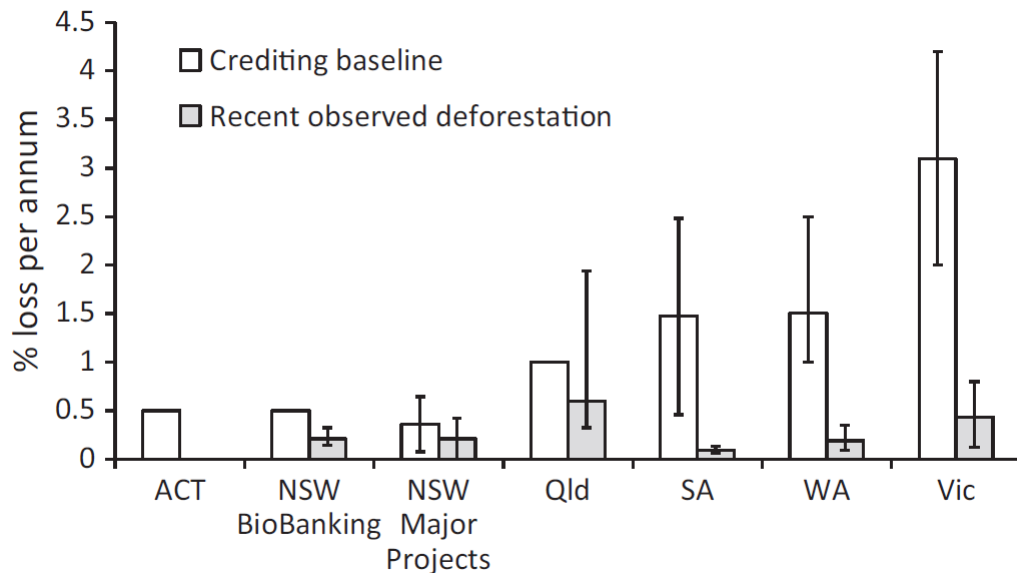


Figure 6.3. Counterfactual annual rates of native vegetation decline (area per se, or condition $\times$ area) used for calculating offset credit in (L-R) the Australian Capital Territory, two New South Wales policies, Queensland, South Australia, Western Australia and Victoria (open bars), and observed mean annual rates of loss for each jurisdiction for the period 2002-2012 (grey bars). Error bars depict range of values. From Maron et al. (2015).

Our findings provide evidence of adverse selection occurring under Australian biodiversity offset policies. Further work is needed to ascertain how it manifests in practice, but is likely a result of: unstated or ambiguous counterfactual assumptions; inadequate or inaccurate data on background rates of biodiversity loss; and an incentive to overestimate the conservation benefits from a potential offset site by those who incur the cost of offsetting.

Is there a risk of “hidden action” in biodiversity offsetting?

Integration of monitoring, evaluation and reporting within the long term management plan of an offset is considered a core ‘best practice’ component of biodiversity offsetting (Australian Government, 2012; Business and Biodiversity Offsets Programme (BBOP), 2012d). Unfortunately, there are widespread reports of inadequate monitoring and evaluation of offsets worldwide, as well as a lack of enforcement by regulators (Bernhardt et al., 2005; Brown et al., 2013; May et al., 2016; Nature Conservation Council of NSW, 2016; Treweek

and ten Kate, 2014; U. S. Government Accountability Office, 2005). In the absence of monitoring and evaluation, it of course cannot be known what outcomes are being delivered by offsetting or any other on-ground conservation management action, nor can deficiencies be identified or rectified. Inadequate monitoring may also incentivise developers or third party offset providers to underdeliver offset requirements (Whitten et al., 2012).

Similarly, moral hazard, otherwise known as ‘hidden action’ (Arrow, 1984; Ferraro, 2008) describes a situation where an agent may judge it is cost-effective to shirk its contract with the principal, since the risk of detection is low (Eisenhardt, 1989; Shapiro, 2005). For example, a developer may choose to not meet its offset obligations if it is known the regulator has limited capacity to monitor their activities for compliance (Whitten et al., 2012; Xepapadeas, 1991). Moral hazard can also arise with the involvement of third-party offset providers, particularly in cases where the liability of offset failure still rests with the developer. Lack of resources or institutional capacity to monitor and evaluate environmental policies is an ongoing challenge faced by regulating agencies (Brown et al., 2013; Whitten et al., 2012). There may also be a more overt interest in minimizing scrutiny of individual offsets due to the high costs involved (Salzman and Ruhl, 2000; Whitten et al., 2012), and the knowledge that the benefits of biodiversity conservation are dispersed and not universally understood by the public (as the final principal).

Public scrutiny can prove to be financially or politically costly for governments (Dovers and Hussey, 2013; Keene and Pullin, 2011; Salzman and Ruhl, 2000), hence monitoring and evaluation may not necessarily be a high priority. Although claims of agency (or regulatory) capture are typically associated with more flagrant forms of corruption or bias, a regulator’s incremental actions may over time tend to favor regulated parties without an explicit intention to do so. This “bureaucratic slippage” (Freudenburg and Gramling, 1994) has been observed in the wetland compensation program in Alberta, Canada, where it was found that the mitigation hierarchy was routinely skipped over in favour of compensation, and compensation ratios approved by the regulator were frequently less than those required by stated policy guidelines (Clare and Krogman, 2013). Direct or indirect political influence, pressure to make rapid decisions with limited information, conflicting organisational goals

and the use of bureaucratic discretion can all influence individual decisions made by policy administrators, which over time cumulatively lead to policy outcomes which fail to meet the originally-stated policy goals (Clare and Krogman, 2013; Macintosh and Waugh, 2014).

Conclusions

This Chapter has provided a preliminary examination of the governance of biodiversity offsetting. Our structured analysis of stakeholder submissions (Martin et al., 2016) provided useful data which describes how different policy actors perceive the efficacy of biodiversity offsetting, and opportunities for policy improvement. Greater policy consistency, transparency, and a need to implement offsets ‘strategically’ over larger spatial and temporal scales were identified by stakeholders as key issues requiring attention to improve policy effectiveness. Drawing on agency theory, literature review and new empirical findings (Maron et al., 2015), I argue that adverse selection and moral hazard may be reducing the environmental outcomes from biodiversity offsetting below what would be required to achieve a ‘no net loss’ policy outcome. Further research is needed to establish what institutional, organisational, and informational incentives drive the behavior of state and non-state actors within biodiversity offset governance, how these interactions play out at different stages of the offset policy process, and what the implications are for biodiversity conservation.

CHAPTER SEVEN:

Opportunities and risks in the implementation of biodiversity offset policy in Australia

This Chapter draws upon contributions made by in the following report:

Evans M.C. (2017). An evaluation of the efficacy of the Australian *Environment Protection and Biodiversity Conservation Act* (EPBC Act) 1999 Environmental Offsets Policy. Report prepared for the Department of the Environment and Energy.

A version of this Chapter is in preparation for publication and may be cited as:

Evans M.C. (In preparation). Opportunities and risks in the implementation of biodiversity offset policy in Australia. *Environmental Policy & Governance*.

Abstract

Biodiversity offset policies have been embraced worldwide as a mechanism for reconciling economic development and environmental protection. Much of the research into biodiversity offsetting to date has been focused on developing and refining offset metrics, and evaluating the environmental outcomes of offsets at the site scale. However, there has been limited analysis of the biodiversity offset policy process as a whole, and how the interpretation and implementation of the policy in practice may ultimately impede or enable environmental outcomes. In this paper, I present the perspectives of policymakers, practitioners and industry proponents on how improved outcomes for biodiversity could be delivered under the Australian *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) Environmental Offsets Policy (2012). Australia is an early adopter of biodiversity offsetting, with policies in place in most state, territory and federal jurisdictions. Drawing on semi-structured interviews with key informants, I found three major results. First, the ‘durability’ of offsets over time – either because of limited regulatory oversight or because of insecure land tenure arrangements – was considered to be a major policy limitation. Second, the potential for offsetting to be effective and efficient is severely constrained by the fragmentation that exists within government departments at the federal level, as well as between state and territory governments. Third, stakeholders expressed concern at the “piecemeal” outcomes resulting from current policy arrangements, and argued that offsets should be designed and implemented at the landscape scale to ensure both long-term environmental outcomes and economic efficiency. The paper concludes with prospects for how these challenges may be addressed within the current institutional and political environment.

Introduction

“...they say that offsets are 90% other stuff and 10% ecology”
Industry respondent 3

“I think there’s a very large focus on the technical aspects of an offset...the metrics and whether the habitat meets all that, which is great, it has to be. I don’t think there’s much attention given to the legal, financial and governance side of offsets at all”

Broker respondent 1

The continued erosion of biodiversity and associated ecosystem services is a key public policy issue globally. Governments are increasingly turning towards biodiversity offsetting as a mechanism to counterbalance the impacts from economic development, with the goal being an environmental outcome that is “no net loss” or “net gain” (Boisvert et al., 2013; Boisvert, 2015; Bull et al., 2013; Gardner et al., 2013; Maron et al., 2016; ten Kate et al., 2004). The number of offset schemes worldwide has increased dramatically over the last two decades (Ives and Bekessy, 2015), and there are now at least 69 countries with national policies in place, in addition to international policies under the International Union for the Conservation of Nature (IUCN, 2016) and the International Finance Corporation (IFC, 2012).

Considerable scholarship exists on the principles underpinning biodiversity offsetting (Bull et al., 2013; Gardner et al., 2013; McKenney and Kiesecker, 2010), the design of metrics used to quantify biodiversity losses and gains through offset trades (Bull et al., 2014; Gibbons et al., 2016; Gonçalves et al., 2015; Quétier and Lavorel, 2011), and the ethical and social justice issues raised by the “commodification of nature” as required by such policies (Apostolopoulou and Adams, 2017; Bidaud et al., 2017; Ives and Bekessy, 2015; Moreno-Mateos et al., 2015; Spash, 2015; Spash and Aslaksen, 2015; Sullivan and Hannis, 2015; Taherzadeh and Howley, 2017). Up until recently, very few studies have examined how offset policy is administered, implemented and evaluated following the initial design phases (Brown et al., 2013, 2014; Clare and Krogman, 2013; Lukey et al., In press).

There is now a growing recognition within the conservation literature that the institutional and political drivers which influence the behavior of policy actors ultimately determine

the biodiversity outcomes from offsetting (Damiens et al., 2017; Gordon et al., 2015; Maron et al., 2016; Salzman and Ruhl, 2000, 2010). Such an insight is crucial, given that a central concern of scientists, scholars and members of civil society is the current lack of evidence that offsetting is delivering its promised outcomes for biodiversity (Curran et al., 2013; Gibbons and Lindenmayer, 2007; Lindenmayer et al., 2017; Maron et al., 2012, 2016; May et al., 2016). For example, the United States Species Banking program has been in operation since 1995, yet it is still not known to what extent impacts to endangered species have been compensated by bank sites (Bunn et al., 2014; Fox and Nino-Murcia, 2005). Similarly in Australia, an early adopter of biodiversity offsetting (Miller et al., 2015), there is a “...lack of evidence that offsets are effective and actually achieving their intended outcomes” (The Senate Environment and Communications References Committee, 2014).

In this paper, I examine stakeholder perspectives of the efficacy of biodiversity offsetting in Australia, with a particular focus on how the policy is interpreted and implemented in practice. Biodiversity offsetting has been used both formally and informally in Australia for the last 20 years, and formal policies are now in place at the federal level and in most states and territories (Maron et al. 2015). Here, I focus specifically on the federal Environmental Offsets Policy (2012), which was designed in response to stakeholder dissatisfaction with the Australian Government’s ad-hoc approach to offsetting in use since 2001 (Miller et al., 2015). The Environmental Offsets Policy (2012) applies specifically to ‘matters of national environmental significance’ (MNES) which are under the jurisdiction of the federal *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). Offsets under the EPBC Act are required to deliver an overall conservation outcome that improves or maintains the viability for a protected matter as compared to what is likely to have occurred under the status quo (Maron, et al., 2013; Miller et al., 2015). The policy is underpinned by a loss-gain calculator which is used by proponents, consultants and the regulator to estimate offset requirements¹⁴ (Chapter Five, Gibbons et al., 2016; Miller et al., 2015).

Environmental impact assessment (EIA) in Australia frequently crosses regulatory processes at federal, state and local levels (Macintosh, 2010b, 2010a), hence a federal approval under the EPBC Act will often still require interactions with state and local

¹⁴ Available at: <http://www.environment.gov.au/epbc/publications/epbc-act-environmental-offsets-policy>

regulatory bodies. Contractual arrangements required to deliver biodiversity offsets within this regulatory environment are therefore highly complex, and specialist intermediaries such as offset brokers have emerged in response (Coggan et al., 2013a). A range of other third parties may also participate in offset governance (Martin et al., 2016), such as environmental consultants and offset providers (e.g. landholders and non-government organisations). An understanding of the efficacy of biodiversity offsetting therefore requires the perspectives of a number of stakeholder groups to be considered, along with an appreciation of how each group experiences policy implementation.

Specifically, this research aimed to understand:

- The roles and experiences of policy actors involved in the process through which biodiversity offsets are designed, assessed, implemented and evaluated under the EPBC Act;
- Factors which may enable or inhibit good policy outcomes as part of this process;
- How improved policy outcomes could be delivered.

Methodology

This research uses qualitative data from semi-structured interviews with 30 policymakers, practitioners and industry proponents who have had direct experience with the implementation of biodiversity offset policy in Australia, and specifically the EPBC Act Environmental Offsets Policy (2012). It also draws upon a review of relevant academic and grey literature, policy documentation and legislation.

I recruited participants using a snowball sampling approach (Biernacki and Waldorf, 1981; Blaikie, 2009) through informal networks and via their workplaces. Sampling began with known professional contacts, and further participants were identified by asking interviewees to nominate other potential respondents who could provide additional insights into the biodiversity offset policy process in Australia. The interviewing process continued until no additional themes or insights emerged during individual interviews.

A key aim of this research was to understand the diversity of individuals and organisations involved in the assessment, implementation, and evaluation of biodiversity offsets in Australia, hence I selected interview respondents from a number of stakeholder groups:

(1) government staff; (2) industry proponents; (3) offset brokers, (4) other specialist knowledge intermediaries (legal and financial advice); (5) environmental consultants; (6) non-government environmental organisations (NGOs) (Table 7.1). Stakeholder groups were identified iteratively through the snowball sampling procedure and the literature review. Interviews with non-government participants (groups 2 to 6) were conducted between December 2015 and March 2016, and interviews with staff from the Australian Government’s Department of the Environment and Energy (hereafter “the Department” or “the regulator”) were during April and May 2016. Government staff within the Department’s Environmental Standards Division are responsible for the assessment, approval, monitoring and compliance of biodiversity offsets under the EPBC Act. Executive-level staff within this Division responded to interview requests and identified up to four staff from their Branch¹⁵ to participate in the study. A sample interview schedule (Appendix 4.1) and participant information sheet (Appendix 4.2) was provided to all respondents prior to the interview.

Table 7.1 Stakeholder groups, descriptions and abbreviations used to refer to specific interview data in the main body of the Chapter

Stakeholder group	Description	n	Abbreviation
Government	Australian Federal Government staff working in offset assessments, approvals, compliance and enforcement.	13	G [1,2,3]
Industry	Representatives from companies who have experience in delivering offsets as part of federal environmental approvals for developments (mining, gas, urban)	4	I [1,2,3]
Brokers	Intermediary contracted by the industry proponent (offset buyer) or the offset provider to mediate an offset transaction (as per Coggan et al. 2013a)	3	B [1,2,3]
Legal & financial advice	Intermediary contracted by the industry proponent or the offset provider to provide independent legal or financial advice. Work with brokers but do not mediate the transaction	4	LF [1,2,3]
Consultants	Ecological specialists with experience in conducting Environmental Impact Assessments for developments, contracted by the industry proponent (for impact assessment) or the offset provider (for assessment of offset suitability)	3	C [1,2,3]
NGO	Environmental non-government organisation, either directly involved in offset transactions as an offset provider, and/or through policy advocacy	3	N [1,2,3]

The interview schedule was used to structure questioning, but was flexibly altered or reordered to preserve the flow of the interview and to facilitate in-depth exploration of

¹⁵ The current organisational structure of the Australian Department of the Environment and Energy is available in its latest annual report (2016) or may be accessed at: <http://www.environment.gov.au/about-us/departamental-structure>

topics raised within the interviews. Participants were asked to provide some contextual information on their role in the organization and their contact with biodiversity offsets, and who else within or outside their organization they interacted with as part of their workflow.

The core of the interview asked participants for their perceptions of the barriers and enablers within the offset process that influence the environmental outcomes delivered by biodiversity offsets. Finally, participants were asked to summarise what they considered to be the key issues that needed to be addressed to improve biodiversity offsetting in Australia. Interviews lasted for up to one hour, and were digitally recorded with the permission of the participant, or otherwise transcribed by hand during the interview. Handwritten notes were also taken during each interview. Digital recordings were professionally transcribed between May and July 2016. Interview transcripts were subsequently provided to all participants, who had the opportunity to check the transcript for inaccuracies or ambiguities and make any necessary revisions.

I analysed interview data by coding passages of text from interview transcripts into thematic categories corresponding to stages in the offset policy process (adapted from Martin et al. 2016): (1) design and proposal, (2) assessment and approval; (3) implementation; (4) monitoring and evaluation; and (5) outcomes. Within each of these stages, I further coded responses into broad domains adapted from Whitten et al. (2012) which reflect the components that underpin and influence the design, assessment, approval and implementation of biodiversity offsets (Figure 7.1).

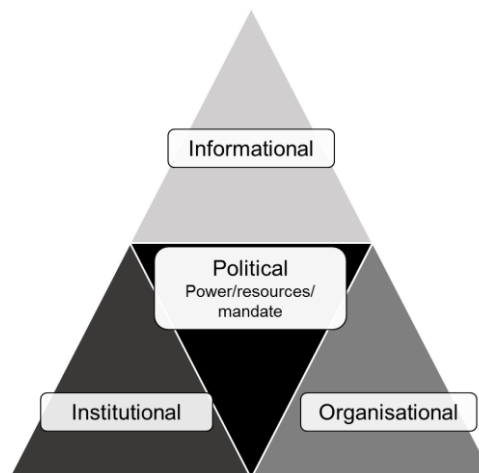


Figure 7.1. Domains which underpin an effective biodiversity offset market, after Whitten et al. (2012).

Results and discussion

Policy actors

Offset transactions under the EPBC Act broadly follow the ‘broker’ model as identified by Coggan and colleagues (2013a), whereby contracts and negotiations between the regulator, the industry proponent and the offset provider may be (but not always) facilitated by an offset broker. Here, I build upon the model outlined by Coggan et al. (2013a) to illustrate how policy actors behave and are influenced by policy settings under the EPBC Act Environmental Offsets Policy within Australia’s federal system of governance (Figure 7.2). The interactions between these actors, as shaped by informational, organizational, institutional and political incentives (Figure 7.1), will ultimately influence policy performance. Biodiversity offsetting is prone to agency problems including adverse selection and moral hazard (Chapter Six, Maron et al., 2016), which may arise due to ambiguous or inaccurate counterfactual assumptions (Maron et al., 2013; Maron et al., 2015; Salzman and Ruhl, 2010; Whitten et al., 2012), inadequate data to inform the estimation of counterfactuals (Maseyk et al., 2017), or ‘bureaucratic slippage’ (Clare and Krogman, 2013; Freudenburg and Gramling, 1994) resulting from incremental discretionary decisions made by regulators (Braithwaite, 2011; Coslovsky et al., 2011; Macintosh and Waugh, 2014).

Environmental approvals under the EPBC Act follow a self-referring system, whereby industry proponents are required to report actions which may have a significant impact on a MNES to the federal regulator¹⁶ (Macintosh, 2004, 2009b). Should the Minister (or delegated authority) decide an action requires approval under the Act, it becomes a ‘controlled action’¹⁷ and is moved into the assessment and approval process under the Act¹⁸. Proponents then subcontract out the ecological work required for the environmental impact assessment (EIA) to third-party consultants. Biodiversity offsets only apply to ‘controlled action’ decisions (Australian Government, 2012) and must not be considered

¹⁶ EPBC Act, Part 7 Section 68:
http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s68.html

¹⁷ EPBC Act, Part 7 Section 75:
http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s75.html

¹⁸ EPBC Act, Part 9

at the referral stage¹⁹ (but see Macintosh and Waugh, 2014). Should offsets be included as a condition of approval²⁰, the proponent will either conduct the offset design and negotiation with third party offset providers in-house, or they will engage with a broker to facilitate the process [I2, I4, G5].

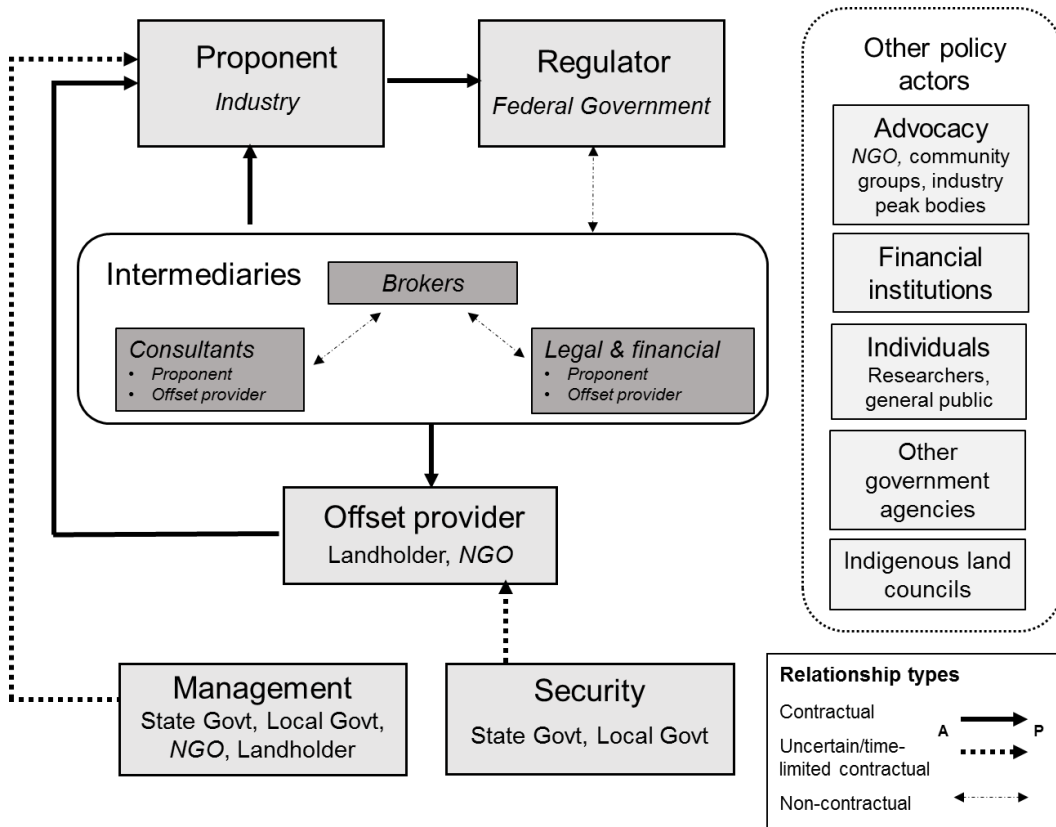


Figure 7.2 Depiction of biodiversity offset governance under the EPBC Act, as derived from literature review and analysis of interview data. Solid single arrows indicate contractual relationships, with the arrow pointing away from the agent (A) towards the principal (P). The agent is the liable party in the contract. Dashed single arrows indicate contractual relationships which are uncertain, insecure or time-limited. Dotted double arrows indicate non-contractual relationships (including working relationships among intermediaries working for either the proponent or the landholder). Actors in *italics* were interviewed as part of the analysis (as per Table 7.1), whereas other actors were mentioned by respondents but not interviewed.

¹⁹ EPBC Act, Part 7, Division 2, Section 75(2)(b):
http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s72.html

²⁰ EPBC Act, Part 9, Division 1, Section 134:
http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s134.html

Brokers can fulfill several functions (Coggan et al., 2013a), including assisting with determining offset requirements, identifying and negotiating with offset providers, facilitating discussions with the regulator, and conducting monitoring and reporting of the offset after it is secured [B1, LF2]. Government-created and owned offset brokers operate at the state level in Australia (e.g New South Wales Biobanking, Victoria's Bushbroker and the former EcoFund in Queensland), however the brokers operating at the federal level are commercial enterprises.

Brokers are generally contracted by experienced, well-resourced proponents who are highly exposed to financial or reputational risk, or who have otherwise decided that engaging an expert broker will be more cost-effective than designing the offset entirely in-house or by negotiating with multiple intermediaries [I2, I4]. Developing an offset proposal requires environmental, legal and financial expertise, and several contracts between the regulator, proponent, offset provider, and the parties responsible for protection and management of the offset site (Figure 7.2). The proponent and the offset provider may both engage with separate intermediaries to provide independent advice, though this is dependent on the financial resources available to each party [LF1, LF3]. The presence of a broker can significantly reduce the transaction costs incurred during the offset policy process and particularly for large developments which require offsets for multiple MNES (Coggan et al., 2010 Coggan et al., 2013a; 2013b).

The offset provider may be a private landholder who is contracted by the proponent to protect and manage part or all of their property for the MNES requiring offsets. Landholders who negotiate with proponents for offset delivery may seek their own legal and financial advice, with or without the assistance of a broker. Should a landholder hold a mortgage over their property, the financial institution issuing the mortgage must approve the conditions placed on the property as part of the offset contract [LF1, LF2, LF4, B2]. In rare circumstances, a proponent may hold private property which has been purchased specifically for the provision of offsets, and will manage the site with in-house expertise rather than engaging with third party offset providers [I1, LF3]. Some NGOs also act as offset providers, by purchasing land using a revolving fund mechanism, securing the property with a covenant (Fitzsimons and Carr, 2014; Hardy et al., 2016), and then on-selling the property to new owners who are likely to abide by the conditions of the covenant.

Under the EPBC Act, the proponent is liable for the purchase, management and security of a biodiversity offset until the expiry of the management plan attached to the conditions of approval for an action²¹. The Minister specifies the period for which the approval has effect at their discretion²², meaning that the conditions of approval and their required duration are highly bespoke [G13]. Interview data indicated that the duration of management required under approved conditions (including biodiversity offsets) in practice can vary anywhere between two and twenty years [LF3, G9, G10]. Security is achieved by a legally binding agreement, such as a covenant (Fitzsimons, 2015; Fitzsimons and Carr, 2014; Hardy et al., 2016), between the offset provider and the authority (usually state or local government) administering the covenant. The proponent may sub-contract out the on-ground management of the offset to another third-party (e.g. an NGO), if it is deemed that the offset provider lacks the sufficient expertise to manage the offset to the standard required by regulator. Following the expiry of the proponent's EPBC Act conditions of approval, responsibility for management of the offset may be vested to the state or local government, or to the offset provider under the terms of its agreement with the covenanting authority [N1, LF4, B2, G1, G2, G9, G10]. Unless specifically stipulated in the EPBC Act conditions of approval or the terms of the covenant, monitoring and evaluation of outcomes delivered by the offset are not guaranteed (May et al., 2016).

Policy process

Here describe factors within each stage of the policy process which I identified from interview data as influencing the efficacy of biodiversity offsetting under the EPBC Act. Figure 7.3 summarises key factors over each of the following policy stages (1) to (5) (as adapted from Martin et al., 2016).

1. Design and proposal

Brokers and legal and financial intermediaries considered that the commercial viability of an offset proposal was fundamental to its success. A commercially viable offset

²¹ EPBC Act, Part 9, Division 2, s 142(1): A person whose taking of an action has been approved under this Part must not contravene any condition attached to the approval.

http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s142.html

²² EPBC Act, Part 9, Division 1, s 133(2)(e):

http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s133.html

arrangement provides mutually beneficial financial outcomes for a proponent and the landholder, whilst still meeting regulatory requirements for compliance. For example, an offset can be designed such that a landholder may continue operate their business (e.g cattle grazing), while also making a financial return from protecting and managing parts of the property for biodiversity conservation [B1]. Landholders with a mortgage on their properties must obtain mortgagee consent from the financial institution issuing the loan before entering into an offset contract:

“...where there’s a mortgage on the property, often that is the biggest obstacle to getting the offset secured” [LF2]

Respondents indicated that financial institutions frequently decline mortgagee consent, as offset agreements are perceived to detract from the value of a property [LF1, LF2, B1, B2]. Government staff had limited engagement with this part of the policy process, and there was a perception held by other stakeholders that there was a lack of understanding within government about the “commercial reality” of biodiversity offsetting [B2]. This issue appeared to be a source of conflict once the offset proposal was presented to government assessments and approval staff:

“...we’re trying to achieve an environmental outcome, the proponent’s trying to achieve a project that is economic ...so you end up with this butting heads” [G6]

Some government respondents expressed an appreciation of the issue, but noted that ‘win-win’ situations between the environmental and economic values of a property are not always possible or desirable [G11, G12]. The issue of mortgagee consent was said to be not yet “on the radar” within the regulator [G1].

Intermediaries and NGO offset providers emphasized the importance of identifying landholders who had sufficient understanding of biodiversity offsets, the expertise required to undertake the necessary on-ground management, and who could be trusted to abide by the conditions of approval [B3, N1, LF1]. Understanding and trust between proponents and offset providers was similarly required, to ensure that each party received a mutually beneficial arrangement [LF3]. Some legal and financial intermediaries noted that the asymmetry in resources and information between large proponents, the proponent’s intermediaries, and landholders (often farmers) have led to detrimental financial outcomes for landholders [LF1, LF4].

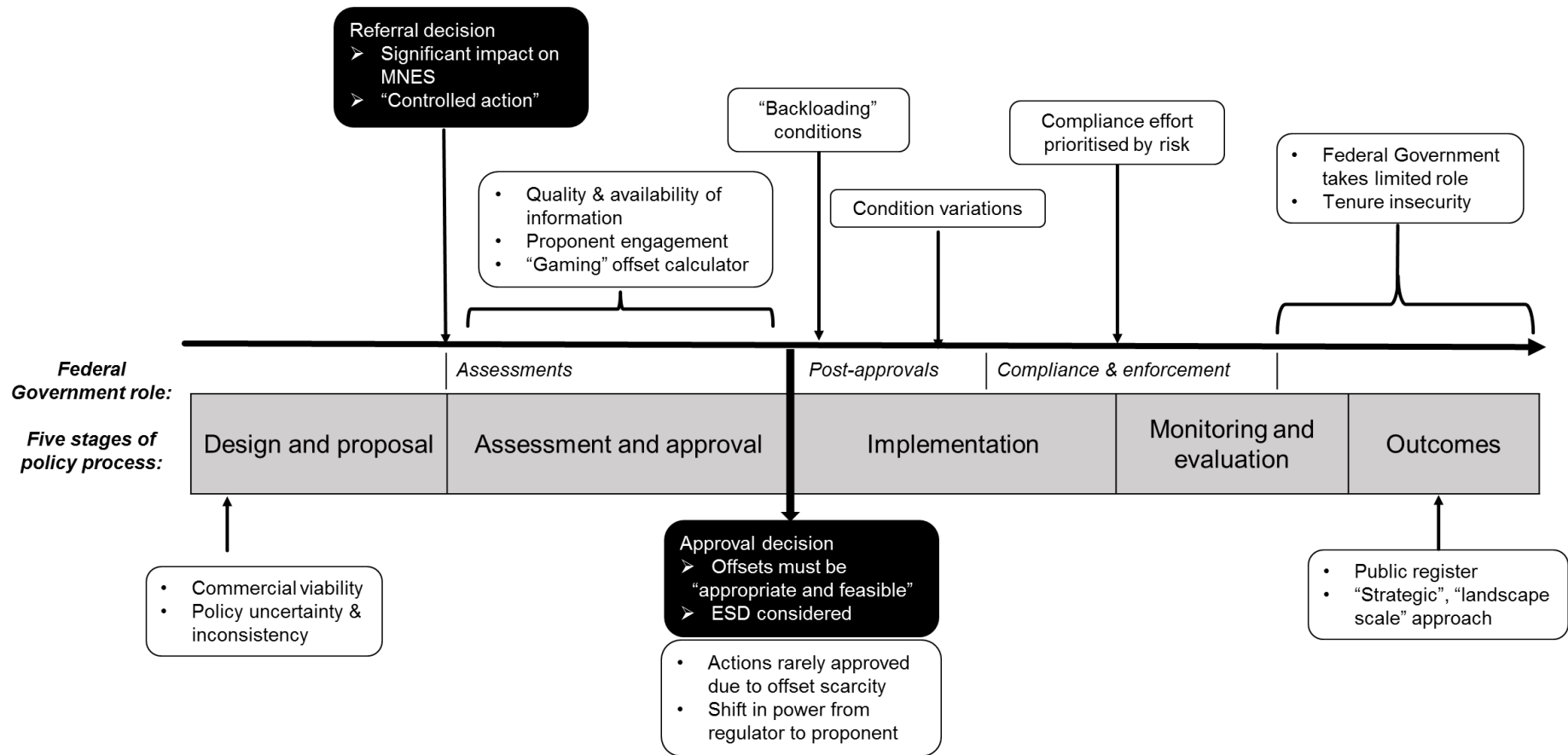


Figure 7.3: Biodiversity offset policy process under the Australian EPBC Act Environmental Offsets Policy (2012), with key factors influencing its efficacy identified through the current research

A lack of alignment between Federal and State biodiversity offset policies, and changes in policy requirements was said to create uncertainty for proponents trying to understand their regulatory requirements [I1, N2, G9, G12, B2]. Bilateral agreements are in place between the Federal Government and most states to ensure there is no unnecessary duplication in conditions placed on the proponent by federal and state regulators (Australian Government, 2014b; Standing Committee on the Environment, 2014). However, the overall policy goals and definitions of what is a ‘suitable offset’ differ between federal and state biodiversity offset policies (Maron et al., 2015), hence these differences still add complexity to the process [G9]. Industry, consultant and broker respondents indicated general support of the EPBC Act Environmental Offset Policy (2012) framework, though frequent changes in state-level policy requirements over the course of the offset design phase still caused delays and additional expense [I1, I2, C2, B2].

Industry respondents with responsibility for offset design and negotiation indicated that their workflow is facilitated by support of upper management within a business [I1]. This willingness to maintain good relationships with the regulator and ensure compliance with environmental conditions appeared to be relevant for proponents who required a ‘social license’ to operate [C2, G1, G3, G4, G11] (Kagan et al., 2003).

2. Assessment and approval

The degree of co-operation, mutual understanding and trust between proponents, government staff and consultants was considered to strongly influence the speed and ease of the assessment and approval process [G1, G3, G5, G9]. Government respondents frequently cited an engaged, policy literate proponent with high ecological capacity (either in-house or contracted to a consultant); and the early provision of high quality ecological information as key enabling factors in this stage:

“...the confidence you have in the offset being delivered, for me is largely based on the information that the proponent’s providing and their engagement in the assessment process.” [G3]

The capacity and policy literacy of proponents was said to be “highly variable” [G5], but influenced by the frequency of their exposure to biodiversity offsets (Coggan et al., 2013b) and business size. Residential developers were generally considered by

government respondents to be the least engaged [G1, G3, G7, G10], while some (but not all) larger companies with in-house ecological expertise were said to display high capacity and engagement [G5]. Government respondents indicated that proponents with low policy literacy or those who failed to negotiate in good faith took up considerable time in their workflow, to the point where sometimes “you might as well do it [the EIA and offset proposal] yourself” [G1].

It was noted that the proper assessment of development impacts on MNES and the associated offset proposal is ultimately reliant on the information provided by the proponent to the regulator [G4, C2]. The inevitability of asymmetric information was acknowledged [G4], as well as potential for this to be exploited [N1, C2, G1, G8]:

“There is a lot of scope for gaming that particular tool [the loss-gain calculator], and a lot of consultants will seek to find an outcome favourable to their client.” [C2]

Government respondents indicated they draw on multiple sources of information to cross-check and validate the information provided by proponents, including: expertise of colleagues within the Division or elsewhere within the Department (e.g wildlife, parks, marine), published scientific data²³ (when accessible) and occasionally direct contact with academic researchers. Although it was noted that the Department has a high level of in-house ecological expertise, the capacity to access this information is limited by short deadlines, physical separation between Branches, and the loss of corporate knowledge due to high staff turnover and recent voluntary redundancy rounds [C2, G1, G2, G5, G6].

The environmental outcomes eventually delivered by a biodiversity offset largely hinges on the details specified within the conditions of approval. The workflow of government staff within the Department is largely taken up by drafting conditions, negotiating conditions with proponents, and varying conditions post-approval. Conditions were identified as a source of conflict and frustration for government staff responsible for assessments, post-approvals and compliance and enforcement, and for non-government respondents. Government staff working on assessments have limited time and

²³ The Department of the Environment and Energy has no access to paywalled scientific journals, though some government interviewees indicated they occasionally sourced journal articles from colleagues who had library access through their University enrolment, or by contacting the Department of the Prime Minister and Cabinet [G7].

information available to write conditions, and they often face intense negotiations with proponents, and received variable feedback on the clarity and enforceability of conditions from colleagues working in post-approvals and compliance and enforcement [G7, G11, G12]. It was however noted that condition writing had improved since the introduction of the Environmental Offsets Policy in 2012, the outcomes-based conditions policy (Commonwealth of Australia, 2016a), and informal “communities of practice” had facilitated interaction and feedback between staff in different Branches [G5, G6].

Non-government respondents expressed concern over a lack of clarity and consistency in conditions of approval [N2, N3, I2], which they perceived to be due to a lack of interaction within the Department, and an inadequate understanding of how conditions translated to on-ground actions:

“...all our offset conditions are all completely different, every single one of them. So just a lack of consistency with, it depends on who in the approvals section you talk to [within the regulator]. It’s hard to come up with a process when you’ve got to adapt it to every different approval you’ve got” [I3]

Government respondents indicated a preference for as many details about the offset as possible to be “locked down” [G8] within conditions prior to approval, but this appeared to not be standard practice. Since “not all assessments get done in the time available” [G1], conditions of approval may simply require a proponent to “develop an offset management plan” [G1], and specific offset requirements may instead be tied to project commencement rather than approval:

“I’ve heard across Branches where approval is given with certainly an amount of work needed to be done on that offset side of things” [G7]

In addition, proponents who rely on federal approval to access project finances may request that specific offset requirements be organised post-approval. This practice of “backloading conditions” [G1, G6, G8, N3] is problematic as it limits the ability for the Department to keep track of the environmental outcomes expected from a project approval, and it reduces the degree of power and leverage held during negotiations with proponents:

“... when it gets to the pointy end of timelines and statutory timelines are running out and decisions have to be made and there’s multi-billion or million dollar projects on the line, often it’s easier for delegates and others to give proponents the benefit of the doubt and assume that they’ll come around and follow through on what’s agreed.” [G8]

Biodiversity offsets are now routinely required as part of conditions of approval under the EPBC Act ²⁴. If significant residual impacts on a MNES are likely to remain after avoidance and mitigation measures are taken, and offsetting is judged to be “appropriate and feasible”, the Minister or delegated authority will approve the action with offsets as part of its conditions if it is considered acceptable with “regard to the likely impact on environmental matters protected, together with economic and social factors” (Australian Government, 2012). Government respondents indicated that “...under the Act it’s very hard to refuse a project on the basis that you can’t find an offset” [G7], due to this requirement to make approval decisions in the context of ecologically sustainable development (ESD, Pittock et al., 2015; Productivity Commission, 1999). However, ESD is meant to maximise economic and social welfare *within a sustainability* constraint, meaning that this interpretation of the EPBC Act is inconsistent with its legislative intent (Macintosh, 2015).

Some government respondents expressed concern about the current or impending scarcity of suitable land available for direct offsets (“we’re literally just going to run out of offsets” [G4]), and suggested the availability of direct offsets for an MNES is not usually known prior to approval:

“...the more and more we become comfortable with using the offsets policy, the assessment officers are saying yes you can impact that if you offset this somehow, approved, handed over to the post approvals officers...[the proponent] may then turn around and say actually there’s no offsets” [G12]

It was also indicated that in situations where it is known there is a scarcity of land available for offsets prior to approval, the offset policy may not be strictly adhered to²⁵:

²⁴ 80% of controlled actions approved since October 2012 require offsets as part of the conditions of approval (K Miller, pers comm).

²⁵ The EPBC Act Environmental Offsets Policy requires an offset package to comprise minimum of 90% direct offsets, maximum of 10% other compensatory measures. *Direct offsets* must “provide a measurable conservation gain for an impacted protected matter” and typically involve the protection and management

“...if something is unoffsettable in direct terms you can look to do virtually all of your offset in indirect terms and compensatory terms [as a financial contribution].” [G8]

3. Implementation

Following regulatory approval of the action, proponents may commence their development and the ‘on ground’ activities required to deliver the biodiversity offset. There may be considerable delay between approval and project commencement, as commodity price fluctuations can put a development on hold for several years or indefinitely [C1, LF1, G4]. Many approved projects and associated offsets therefore do not proceed, though the Department doesn’t currently have data which can differentiate between conditioned impacts to MNES and offset requirements, and the actual impacts and offsets which eventuate (Department of the Environment and Energy, 2016).

Non-government respondents expressed frustration over the tendency for conditions of approval to focus disproportionately on quantification of offset requirements, rather than the complexities associated with implementation [I3, C3, B1, N2]:

“...there’s so much focus on the numbers game and then in two words [the conditions] say oh it’s got to be a secure “something”. Well that process is actually really the most important part.” [I3]

In cases where conditions of approval cannot be feasibly translated on ground, due to ambiguity or changed circumstances, conditions may be varied by the regulator in its post-approvals function²⁶. In many cases the circumstances are genuine, and engaged proponents will seek alternative offset arrangements and present the new proposal to the regulator for approval. However, interview data suggested there was a perception that some proponents lobbied for condition variations in an attempt to shirk regulatory burden [G6, G7, G8, N1].

of land. *Other compensatory measures* are financial contributions which are “anticipated to lead to benefits for the impacted protected matter, for example funding for research or educational programs”

²⁶ EPBC Act, Part 9, Division 1, s 143:

http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s143.html

Some respondents expressed concern that condition variations can lead to diminished environmental outcomes, either due to inadequate internal discussions within the regulator or direct lobbying from proponents:

“...we only hear anecdotally when things are varied... often our conditions are the best we can get from a proponent...I haven’t heard of an instance where it’s necessarily better [for the MNES]. It’s just, I don’t know, easier rather than better” [G7]

“...it’s just death by a thousand cuts where [proponents] keep coming back to [the regulator] and I just saw this one that took 2 years. [The proponent] finally got existing parkland accepted as an offset” [N1]

The shift in power from the regulator to the proponent after approval was noted again, as it was suggested that the compliance and enforcement Branch “have limited statutory power about what they can and can’t push back on”²⁷ should a proponent try to “weasel out” of their regulatory obligations [G8].

4. Monitoring and evaluation

A key criticism of biodiversity offsetting is its lack of demonstrable outcomes in Australia and internationally (Curran et al., 2013; Maron et al., 2016; Martin et al., 2016; May et al., 2016; The Senate Environment and Communications References Committee, 2014). Previous studies have highlighted inadequate monitoring and reporting (Bernhardt et al., 2005; Hornyak and Halvorsen, 2003; May et al., 2016; Reiss et al., 2009; U. S. Government Accountability Office, 2005), non-compliance with conditions and under-delivery of offset requirements (Ambrose et al., 2007; Brown et al., 2013; Brown and Veneman, 2001; May et al., 2016; National Research Council (U.S.) and Committee on Mitigating Wetland Losses, 2001; Robertson and Hayden, 2008; Sudol and Ambrose, 2002; Tischew et al., 2010), and difficulties with accessing data in order to conduct an evaluation (Lindenmayer et al., 2017; May et al., 2016). A key part of each interview was therefore aimed at establishing the extent to which the respondent was involved with the

²⁷ The EPBC Act provides sufficient statutory power under ss 142-142B and 143-145A, but using this power may be politically difficult

monitoring and evaluation of offsets, and their perspectives on the efficacy of this stage of the offset process (Appendix 4.2.)

Some respondents stated that monitoring and reporting of offsets occurred routinely and as required by the regulator [LF3, I4, C2, N1, B1]:

“...it has been made very clear to us by the regulators and our consultants that the monitoring and reporting are essential components” [I4]

Respondents who held a positive view of current offset monitoring and reporting arrangements were often directly involved in this process [B1], or worked closely with proponents who were eager to visibly demonstrate the environmental outcomes being delivered by offsets [I1, I2, I4, C2, LF2, LF3]. Some respondents who had close experience with offset monitoring and reporting were highly critical of its adequacy:

“You will have proponents that have never had an offset that’s been visited by a monitoring or compliance person.... they know [the monitoring report] just goes to the department and that never gets audited and cross-referenced...if you’re a savvy proponent could just be writing something on a bit of paper, submitting it and that would just never get picked up.” [N3]

Non-government respondents indicated that compliance and enforcement may not occur due to inadequate resources [I2, C3] or an unwillingness to pursue instances of non-compliance linked to large, highly politicized projects and proponents [N1, N3, LF2]:

“...[there’s a] perception that there’s probably not sufficient resources put into compliance from the regulators” [I2]

The resourcing, information and staff capacity required to monitor offsets and other environmental conditions under the EPBC Act is daunting, given the sheer number of listed MNES and the historical accumulation of approved projects²⁸ [G1, G8]. Individual projects may have over 100 conditions, and several management plans attached to conditions that are thousands of pages long. Following the tabling of an ANAO audit

²⁸ At the time of data collection (February to March 2017), it was reported that around 4,500 approved projects are prioritised for compliance and enforcement effort by the Department, of which approximately 1,000 are controlled actions (and so may require offsets as conditions of approval) [G10]. The 2013-2014 ANAO audit reported that at September 2013 there were 635 approved controlled actions under the EPBC Act, with around 8000 conditions attached to them to protect 1,282 MNES.

into the Department's management of compliance under the EPBC Act (ANAO, 2014) a risk based approach to prioritizing compliance and enforcement effort was introduced (Nicol et al., 2014). Prior to this, it was suggested that "our [the Department's] monitoring function didn't really exist" [G10]²⁹. Government respondents support for the risk based approach used to prioritise compliance and enforcement effort, and indicated there had been substantial improvements since its introduction [G5, G10]. However, focusing on "high risk" projects may give "good" proponents and offset providers a perception that the Department's compliance effort "doesn't exist" [N1]:

"...we shot our first annual report off...didn't even hear back from them [the regulator]" [N1]

Government respondents working in an assessments capacity noted that although they may receive feedback on the quality and enforceability of approval conditions, they rarely learned of what (if any) environmental outcomes the conditions had ultimately delivered [G2, G3, G7]. Monitoring reports delivered by proponents to the Department technically contain the information required to evaluate offset outcomes, but this data is not readily accessible [G3, G5, G7, G8]. It was said that the resources required to collate this information into a centralised database would exceed the Department's current capacity.

5. Outcomes

Several respondents suggested it was simply too early to know what environmental outcomes have been delivered by many approved offsets given the time lags associated with the approval process, project commencement, offset establishment and the ecological responses to management [LF3, C2, G3, G6, G12]. Notwithstanding this caveat, the interview data suggests that the Department currently does not have the capacity to evaluate the collective efficacy of biodiversity offsets approved under the EPBC Act.

Many government respondents expressed concerns about the long-term security and management of approved offset sites [G1, G7, G9, G11, G12]. It was emphasized that while conditions of approval can regulate impacts to MNES and require the proponent to

²⁹ The findings of the 2002-03 ANAO audit (2003) originally led to the creation of the compliance and enforcement team. The 2006-07 ANAO report (2007) also found deficiencies in compliance and enforcement.

identify an offset site and ensure it is secured, the Federal Government had limited role beyond the expiry of the offset management plan attached to the conditions:

“...there isn’t necessarily great formal management after that. And there certainly isn’t tracking in outcomes terms I don’t think” [G9]

Government respondents indicated that long-term responsibility for offsets is typically taken on by a state or local government, either by being transferred into public conservation estate [G9, G12] or secured on private land via a covenant [G2]. Covenanted programs vary substantially across State and Territories, are administered by different legislations, and provide varying degrees of permanence (Fitzsimons, 2015). The Federal Government cannot force another jurisdiction to agree to or honour a conservation covenant, and so government respondents indicated there was a reliance on negotiating offset agreements between proponents and a state or local government where the likelihood of a covenant being broken is low:

“...what we need is our assessment people [to] make sure that they have certainty that the regulatory environment is going to work for that particular outcome...they must have a commitment from council that council are actually going to deal with it, they are going to enforce the covenant” [G10]

Perversely, this practice is likely reducing the additionality and permanence of outcomes delivered by terrestrial offsets by reducing the averted loss component of the conservation gain (Maron et al., 2013; Maron et al., 2015; Maseyk et al., 2017). It was said to be “really common” [G12] for proponents to have difficulties in securing an offset site because the state or local government won’t agree to a covenant, and as such become non-compliant with their conditions through no fault of their own. In these cases, a new offset site would need to be identified, or else the proponent may be conditioned to provide monetary compensation in lieu of a direct offset [G12]. Hardy and colleagues (2016) found only limited cases where covenant agreements were released or breached, but the findings presented here suggest that a bigger problem may be the reluctance or inability to secure offsets with a covenant in the first place.

Respondents expressed very low confidence in state and local government’s ability to protect and manage offset sites in the long term, due to a lack of resources or a desire to

not impose tenure restrictions which may limit future economic development on the site (Adams and Moon, 2013):

“...enforcement [of the covenant] lies with the council and so some councils won’t even agree to them...they run off the smell of an oily rag, some of them. And often conservation’s not a priority for many councils like they just don’t care.” [G7]

“...no covenant is secure... so when we say you must protect the area in perpetuity, there’s no such thing. So yeah covenants get removed, councils choose not to enforce, state government choose not to enforce. That’s the reality” [G10]

Interviewees suggested that the lack of Federal control over offset secured under State or Local legislation may risk long term environmental outcomes:

“... in the short term it’s probably fine, in the medium term your risks grow and in the longer term you’ve got no idea what’s going to happen with it...It is just a systemic failure that would build over time.” [G9]

Although the EPBC Act provides statutory power for the Commonwealth to enter into legally binding conservation agreements with proponents for “protection and conservation” of MNES ³⁰, it was indicated that these agreements are not necessarily secure from actions taken by a state or local government [G10]³¹.

Persistent risks, and opportunities for improvement

This research identified a range of factors which enable or inhibit the environmental outcomes delivered by the biodiversity offset policy process under the Australian EPBC Act Environmental Offsets Policy (2012). Most of these factors (Table 7.2) had little to do with policy design, or with the specific details of the loss-gain calculator used estimate offset requirements (Chapter Five, Gibbons et al., 2016; Miller et al., 2015). The calculator can be “gamed” to produce a more favourable outcome for proponents, and further refinements and guidance may reduce the scope for its misuse (Maseyk et

³⁰ EPBC Act, Part 14, ss 304 to 312

http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s304.html

³¹ EPBC Act, Part 14, s 312(1): “A provision of a conservation agreement has no effect to the extent (if any) to which it is inconsistent with a law of the Commonwealth, or of a State or Territory.”

http://www.austlii.edu.au/au/legis/cth/consol_act/epabca1999588/s311.html

al., 2017). However, there are several other factors and critical junctures within the offset policy process which can lead to detrimental environmental outcomes (Figure 7.3, Table 7.2).

Interviewees revealed that financial institutions can be a major barrier to the successful negotiation of offset agreements, and suggested this was due to a lack understanding of biodiversity offsets within the financial sector [LF2]. Legal and financial intermediaries indicated that they are increasingly negotiating biodiversity offset agreements for clients who had previously operated in the carbon market:

“...there was a lot of hope and potential around the carbon farming initiative, when that sort of tanked then those businesses were looking at other business models and biodiversity offsets is one of them” [LF4]

Given this shift in market activity and the commercial potential of biodiversity offsetting (under certain circumstances), there is an opportunity for the financial sector to improve its bottom line through increased knowledge of biodiversity offset agreements.

Several respondents argued that greater emphasis needed to be placed on the legal, financial and governance components of biodiversity offsetting [B1], and suggested that focusing primarily on the “front end” assessment of ecological impacts and offset requirements [I3, C3, G4] can be to the detriment of environmental outcomes delivered through implementation:

“You’ll spend \$100, 000 on a biodiversity assessment to make sure the numbers on one side match the numbers on the other side, whereas that money should really just go into managing a property” [C3]

The Department is under considerable pressure to move projects through the assessment phase quickly so that proponents can commence their developments (Department of the Environment and Energy, 2016; Macintosh and Waugh, 2014; Standing Committee on the Environment, 2014).

Table 7.2. Summary of factors influencing the environmental outcomes delivered by the biodiversity offset policy process under the EPBC Act Environmental Offsets Policy (2012), as derived from interview data.

Informational	Organisational	Institutional	Political
Asymmetric information	Degree of support of business higher management	Federal government has limited role in offset security	Capacity for offset loss-gain calculator to be gamed
Quality of environmental impact assessment	Social license to operate	Policy uncertainty and inconsistency	Consideration of ESD in offset decisions
Inadequate access to spatial data and information systems within the government	Interaction between government staff	"Focal species approach" of EPBC Act	Pressure on regulator to provide approval to ensure project financing
Public register of offsets	High government staff turnover	Lack of covenant security and enforcement by States	Shift in power from regulator to proponent after approval
Government compliance effort not visible to all proponents	Short timeframes and high government staff workload	Liability rests on proponent not offset provider	Condition backloading and variations
Financial institutions have limited understanding of offsets		Disproportionate focus on assessment stage	Inequitable resourcing between government assessment and compliance functions

The Department invests most of its resources into the assessments function rather than compliance, but total staffing has decreased over the last financial year (Senate Standing Committee on Environment and Communications Legislation Committee, 2015, 2016)^{32,33}. The complexities and inefficiencies of environmental impact assessment in Australia is well understood (Macintosh, 2010b, 2010a), and it appears unlikely that a greater focus on implementation and outcomes will occur in the near term. The Federal Government has released a policy on outcomes-based conditions (Commonwealth of Australia, 2016a) which specify the environmental outcome required for compliance, but at this stage the use of such conditions it is not mandatory.

The most common suggestion from respondents to improve biodiversity offsetting in Australia was for a “landscape scale” or “strategic” approach. These two terms were used interchangeably, but were generally used to describe an approach to offsetting which required:

- Operation across larger spatial and temporal scales than current arrangements;
- Coordination, alignment and/or strategic partnerships between multiple parties external to the Department, including state governments, local governments, proponents, NGOs and management agencies;
- Improved coordination and integration within the Department;
- Long term strategic oversight and coordination;
- Better alignment between federal, state and local legislation which governs environmental protection and planning.

³² The staffing level for compliance and enforcement sections of the Division was budgeted at 29.5 (full time equivalent) for 2015–16 financial year. The budgeted staffing level for the 2016–17 financial year is 30.7 (full time equivalent). The staffing level for the assessment and approval sections of the Division was budgeted at 61.7 (full time equivalent) for the 2015–16 financial year. The budgeted staffing level for the 2016–17 financial year is 60.7 (full time equivalent).

³³ As at 31 April 2015 there were 140.1 (Full Time Equivalent) staff engaged in assessment, investigation, compliance and enforcement, and the Department will continue to maintain capabilities across the function of environmental assessments and approvals.

A “strategic” approach to biodiversity offsetting was said to ensure both long-term environmental outcomes [B1, N1, C2, G3, G6, G9, G11], and exploit economies of scale [B1, I3]. The EPBC Act Environmental Offsets Policy (2012) applies only to listed MNES, hence offsets are not required to improve or maintain outcomes for biodiversity “as a whole” [N1, G2, G9]. Respondents noted that the “focal species approach” (Lambeck, 1997; Lindenmayer et al., 2002; Possingham et al., 2002) of the Act constrained the ability to deliver conservation benefits at the landscape scale, in favour of “piecemeal” outcomes [N1, I3, C3, B1]. Nevertheless, several government staff, brokers, industry proponents and consultants reported that a “good” offset delivered positive outcomes for MNES in addition to species and communities not listed under the EPBC Act, and attempted to co-locate offsets alongside existing protected areas to improve landscape connectivity. Government staff reported that their efforts to achieve such outcomes were often hampered by a lack of easy access to spatial data which could indicate the locations of existing offsets, protected areas, critical habitat and wildlife corridors [G4, G8]. Proponents and brokers indicated that Australian governments could assist in enabling a “strategic” approach through improved coordination and guidance:

“...the nirvana would be strategic leadership from the regulators with a shopping list of already approved and higher priority areas to secure that you can basically pick off the shopping list” [I2]

However, some respondents indicated they are already designing and implementing large-scale, “strategic” offsets in a commercially viable matter in absence of government intervention, and in collaboration with businesses who are otherwise industry competitors [B1, I3].

The Australian Government has stated that information on registered offsets (e.g spatial data, information relating to MNES and ongoing management actions) would be made publicly available “where it is appropriate to do so”, and would “allow strategic planning, and streamline processes with state and territory requirements and schemes” (Australian Government, 2012). A Senate Committee report recommended in 2014 that the Department “expedite the development of a publicly available nationally coordinated register of

environmental offsets.” (The Senate Environment and Communications References Committee, 2014). However, such a register has yet to be developed. Interview respondents were generally supportive of a public register of offsets, as it would promote transparency and accountability [N1, N3, G5, I2], enable a “strategic” approach [G7, G13], and enhance the legitimacy of biodiversity offsetting in the eyes of the Australian public [G1, G3]:

“I think it’s useful for everyone...it’s less work for the proponents, less work for consultants. Landholders who want to put an offset up can see what the market value is and how that changes over time. And it’s better for us and a better environment outcome. It ticks all the boxes.” [G7]

Some risks of a public register were noted, including landholder privacy concerns, land price speculation [B1, G4], unintentional or deliberate misinterpretation of information held in the register [G3, G8], and government exposure to political repercussions resulting from public awareness of offset performance [G3, G5].

Another suggested mechanism for enabling a “strategic” or “landscape scale” approach was a national offset trust fund, which could be paid into by proponents (whose offset liability would then be severed) [G10]. However, industry proponents and brokers were skeptical of the Federal Government’s ability to effectively and efficiently operate such a fund [I3, B1, B3, G10], based on experience in other jurisdictions and a perception that the money would shift into consolidated revenue. One government respondent suggested that a national offset trust fund could be used to strategically purchase around Australia:

“So what we do as the federal government, we take responsibility for it. We establish these corridors, we set up these places and so on. It’s like a national reserve scheme.” [G10]³⁴

However, such an approach would shift offset liability and risk on the Federal Government, which “wouldn’t be looked on very favourably” [G10]. Interviewees expressed support for an independent authority to administer such a fund [B1, N2]. Some respondents argued than

³⁴ Federal Government investment in the expansion of the National Reserve System ceased in 2013. <http://www.environment.gov.au/land/nrs/management>

an offset fund must be liquid if it is to be effective, which would require that the ‘like for like’ offset policy requirement be relaxed³⁵ [N1]. This approach may lead to additional risks as the connection between the impact and the promised compensation would be diminished [G1, G9] (Maron et al., 2016; Salzman and Ruhl, 2000), but such risks could be mitigated by effective oversight from an independent authority.

Conclusions

Biodiversity offsetting in Australia operates within a highly complex system of governance, with diverse policy actors operating across multiple jurisdictions. The current institutional, organisational and political environment incentivizes poor environmental outcomes from biodiversity offsetting, due to persistent information asymmetry, inadequate (real or perceived) regulatory oversight, ambiguous responsibility for long term offset security management, and lack of transparency and public accountability. The EPBC Act provides sufficient statutory power to overcome many of the issues identified in this research, including: outcomes based offset conditions specified prior to approval; decisions which ensure offsets truly are “appropriate and feasible” before an impact on a MNES is deemed acceptable; effective long-term security and management, and effective compliance and enforcement of conditions. Given the plethora of research which has previously highlighted the disconnect between the Act’s legislative intent and its interpretation by decisionmakers (Macintosh, 2004, 2009a, 2015; Macintosh and Waugh, 2014), there appears to be little political appetite to use this power to improve outcomes for Australia’s environment.

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³⁵ According to the EPBC Act Environmental Offsets Policy (2012), “Suitable offsets must deliver an overall conservation outcome that improves or maintains the viability of the aspect of the environment that is protected by national environment law and affected by the proposed action”

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CHAPTER EIGHT:

Conclusions and future directions

“During the past two decades we have been lulled into complacency by the allure of “win-win” solutions and we have assumed rather too quickly that simple policy tools can solve complex policy problems”

Roldan Muradian et al. (2013). Payments for ecosystem services and the fatal attraction of win-win solutions. *Conservation Letters* 6(4): 274–279.

“For those concerned with making positive changes towards a less unsustainable world, the state has to be engaged with, its positive potentials explored and developed, its flaws examined for signs of possible improvement, and its dynamics and characteristics understood in their particularity.”

John Barry and Robyn Eckersley (2005) W(h)ither the Green State?
The State and the Global Ecological Crisis

The conservation of biodiversity is a complex, ‘wicked’ problem which is frequently at odds with the growth of the human economy (Czech, 2008; Game et al., 2014). This seemingly irreconcilable conflict has motivated the search for ‘win-win’ policy solutions (Muradian et al., 2013; Ostrom et al., 2007; Phelps et al., 2012b) where biodiversity can be maintained alongside economic development, instead of restricting human activities that are detrimental to the environment. While it can be possible to simultaneously achieve positive social, environmental and economic outcomes, these situations are highly context specific and will generally rely on a combination of policy instruments (Dovers and Hussey, 2013; Gunningham and Sinclair, 1999; Jordan et al., 2013; Muradian et al., 2013; Young and Gunningham, 1997). As policy diversity increases, so does the number and diversity of actors participating in environmental governance (Agrawal and Lemos, 2007; Jordan et al., 2005; Lemos and Agrawal, 2006; Liverman, 2004; Lockwood et al., 2010; Newell et al., 2012),

hence policy implementation and *ex post* evaluation becomes increasingly complex and detached from any simple expectations held during policy design and *ex ante* evaluation.

This thesis has examined the challenges and complexities associated with the design, implementation and evaluation of policies which aim to reduce or redress the effects of deforestation: a major driver of biodiversity loss globally. I considered regulatory and market-based policy instruments implemented in Australia, an advanced and politically stable economy, and sought to evaluate what outcomes, opportunities and risks these policies present for biodiversity conservation.

Chapter Two described the contemporary drivers of, trends in, and policy responses to deforestation in Australia, by drawing on existing literature and new empirical findings. I showed that deforestation is primarily driven by the expansion of pasture for livestock grazing, but this expansion is mediated by broader macroeconomic and climatic trends. At a national level, there has generally been a decline in deforestation in Australia over time, however this is tightly linked to the amount of primary forest remaining and availability of forest regrowth.

Close to 17 million hectares of forest has been cleared across Australia since 1972, with the vast majority (58%) occurring in the state of Queensland. Over the past four decades, there has been a shift in the intent, framing, and style of policies introduced to combat deforestation in Australia. Previously incentivized by governments (Australian Greenhouse Office, 2000; Seabrook et al., 2006), deforestation was recognized to be detrimental to soil health in the 1980's, and regulatory policies to reduce deforestation were first introduced in an effort to combat salinity and desertification. Concerns about ecologically sustainable development and biodiversity conservation motivated the introduction of further regulatory controls over the 1990's and 2000's

However, since 2010, most Australian states have wound back regulatory controls on deforestation in favour of self-regulation and voluntary compliance. This trend towards deregulation has continued after the publication of Chapter Two as Evans (2016). The current (mid-2017) Queensland Government attempted to re-strengthen the *Vegetation Management Act 1999* (VMA) after it was relaxed in 2013, but the Bill failed to pass through Parliament

in 2016 (Reside et al., 2017). The New South Wales Government has repealed the *Native Vegetation Act 2003* and introduced a new *Biodiversity Conservation Act 2016* and the *Local Land Services Amendment Act 2016*. The new policy framework relies heavily on biodiversity offsets and voluntary compliance under self-assessable clearing codes, and permits high risk clearing activities without regulatory oversight (Evans and Maron, 2016). Although the science is clear on the need to reduce deforestation in Australia (Evans, 2016; Reside et al., 2017), the politics are certainly not. Further, there is scarce evidence of the effectiveness of various policy responses to deforestation over the last three decades, whether they be regulatory or market-based instruments.

The marked drop in the rate of deforestation in Queensland following amendments to the VMA in 2007 was hailed as the “end of broad scale land clearing” in Australia (Kehoe, 2009; McGrath, 2007). Yet how much of this trend can be attributed to the policy change relative to macroeconomic, climatic, biophysical and institutional drivers has never been empirically tested. In Chapter Three, I conducted such an evaluation using a novel statistical framework that has a greater capacity than traditional regression methods to detect changes in complex systems. I found that the bent-cable model is a promising technique for detecting a policy effect while controlling for other drivers, and there is some evidence of a policy-induced shifts in the deforestation rate as expected in 2000 with the introduction of the original VMA, and with the subsequent amendments in 2007. However, the current model fits are not sufficiently robust to support these findings with confidence, due to an inadequate number of data points (local government area, LGAs) and extreme spatial variation in trends in, and responses to, deforestation within and between local government areas. While deforestation in some LGAs increased in 2000 and decreased in 2007 as predicted by the policy changes, other LGAs followed the completely opposite trend. In yet more LGAs, deforestation remained either consistently high or low throughout the time series (1988 to 2014).

The level of detail and methodological sophistication required to quantitatively estimate policy impact in Chapter Three provides a hint as to why *ex-post* evaluation of conservation policies is still rare. Although the technical barriers to evaluation are well understood (Baylis et al., 2016; Ferraro, 2009; Ferraro and Pattanayak, 2006; Miteva et al., 2012), the cultural and political impediments to an “effectiveness revolution” in conservation are not as widely

acknowledged (Curzon and Kontoleon, 2016; Keene and Pullin, 2011; Kleiman et al., 2000). There is also a tendency to move onto designing and implementing new, “innovative” policy instruments before it is known whether existing policies are effective.

In the late 1990’s and early 2000’s, there was a surge of interest in market-based instruments for conservation in Australia (Evans, 2016; Gunningham and Young, 1997; Young et al., 1996). Biodiversity offset policies are now in place across the country (Maron et al., 2015; Miller et al., 2015), and carbon farming was established in 2011 soon after the introduction of a national carbon pricing scheme (Commonwealth of Australia, 2011; Evans et al., 2015; Macintosh, 2013). The high level of policy activity and the allure of ‘win-win’ outcomes stimulated academic interest in how such policies could be designed to maximise potential biodiversity outcomes (Bekessy and Wintle, 2008; Carwardine et al., 2015; Crossman et al., 2011; Evans et al., 2015; McKenney and Kiesecker, 2010; Nelson et al., 2008; ten Kate et al., 2004).

Chapter Four of this thesis examined the potential for carbon farming policy to incentivize the restoration of degraded agricultural landscapes in Queensland, where much of the grazing is on marginal land (ABARE, 2009; Marinoni et al., 2012) and threatened ecological communities such as Brigalow (*Acacia harpophylla*) have been reduced to less than 10% of their original extent. Assisted natural regeneration (ANR) of native vegetation that is subject to regular, costly clearing for pasture in largely unprofitable areas presented as a likely ‘win-win’ outcome. We found that ANR was a far more cost-effective carbon farming methodology than the more commonly considered environmental plantings (Carwardine et al., 2015; Crossman et al., 2011; Paul et al., 2013), and is the superior option for biodiversity (Dwyer et al., 2009; Fensham and Guymer, 2009). Although we sought to make more realistic assumptions about market conditions and establishment costs than previous studies (Crossman et al., 2011; Paul et al., 2013; Polglase et al., 2013), our quantitative analysis nevertheless assumes “rational” economic behavior and policy stability, and ignores the socio-cultural barriers to adoption of carbon farming policy (Kragt et al., 2017; Macintosh, 2013; Torabi et al., 2016). Policy changes and political events over the past five years have clearly illustrated why future work needs to focus on the social and institutional barriers to participation in voluntary environmental schemes.

The initial work on Chapter Four began in 2012, when the Australian carbon price was set at AUD\$23 t CO₂e⁻¹, and excitement over the potential of the carbon market to deliver for biodiversity was at fever-pitch. By the time Evans et al. (2015) was accepted for publication, Australia was onto its third Prime Minister in three years, and the national carbon pricing mechanism had been abolished (Bailey et al., 2012). The Carbon Farming Initiative (CFI) transitioned into the Emissions Reduction Fund (ERF), and a reverse auction mechanism is now used to distribute payments from a total pool of AUD\$2.55 billion (Australian Government, 2014). Individuals and organisations may submit bids that outline an emissions reduction project (which may follow vegetation methodologies approved under the CFI), and the lowest bids are selected by the Clean Energy Regulator (CER). The ERF has been roundly criticized as ineffective and inefficient (Blakers and Considine, 2016; Burke, 2016), as the majority of successful bids are low-cost, avoided deforestation projects where the additionality is highly questionable. Adverse selection of low-quality projects through the ERF (Burke, 2016) is common in environmental markets, including agricultural subsidy schemes (Ferraro, 2008) and biodiversity offsetting (Chapter Six, Maron et al. 2015).

My carbon farming case study highlights how even with the best intentions, the outcomes anticipated during policy design may not survive the rest of the policy process. Biodiversity offsetting (Chapters Five to Seven) is another good example. In Chapter Five, I described the mathematical framework that underpins the Australian Government's EPBC Act Environmental Offsets Policy, which was designed to give effect to its 'no net loss' policy goal. The offset metric described in Chapter Five is rare in its explicit quantification of biodiversity gains relative to a counterfactual scenario (Ferraro, 2009; Maron et al., 2013), and so correctly estimates the benefit gained by an offset action in addition to what would have occurred in the absence of the impact and offset. By considering a discount rate which reflects the probability of species extinction rather than pure societal time preference, offset requirements calculated using the metric scale up as the degree of threat faced by a species increases, thus incentivizing greater avoidance, mitigation and offset measures for endangered species and ecosystems. This work was adopted as federal policy in Australia (Australian Government, 2012; Miller et al., 2015), and informal discussions with government staff suggest that the large impact the 'ecological time preference' discount rate

has on offset calculations is influencing threatened species listing decisions. Whether the higher offset requirements calculated using the new metric are translating into better outcomes for threatened species and ecological communities, however, is much less clear.

Inspired by my glimpse into the messiness of real-world policymaking (Lubell, 2013, 2015) during my time spent with the Australian Department of the Environment, I sought to examine biodiversity offsetting through a policy lens. Chapter Six draws on published work (Maron et al., 2015, 2016; Martin et al., 2016) and agency theory (Eisenhardt, 1989; Kiser, 1999; Shapiro, 2005) to develop an understanding of the behavior of policy actors in biodiversity offset governance. I argued that the implausible crediting baselines employed by Australian biodiversity offset policies not only risk entrenching or worsening biodiversity decline, but constitutes evidence of adverse selection (Burke, 2016; Ferraro, 2008). Policies that rely on historical or unobservable counterfactual baselines are at risk of being gamed (Salzman and Ruhl, 2010) such that the economic cost borne by proponents is reduced, and similarly the political pressure on regulators is lessened (Clare and Krogman, 2013; Freudenburg and Gramling, 1994). The unfortunate outcome for biodiversity is a lower than expected ecological compensation for approved development impacts. To what extent the promised biodiversity outcomes from offsets translate to what happens on the ground is, however, frequently unknown.

The final empirical Chapter of this thesis (Chapter Seven) aimed to build on the preliminary investigation of biodiversity offset governance in Chapter Six, by drawing on qualitative data from semi-structured interviews with key informants who have direct experience with the implementation of the EPBC Act Environmental Offsets Policy (2012) in Australia. I identified a range of informational, institutional, organizational and political factors which enable or inhibit the environmental outcomes delivered by the current biodiversity offset policy process. The majority of these factors had little to do with policy design, or the specifics the loss-gain calculator (Chapter Five, Gibbons et al., 2016; Miller et al., 2015). Information asymmetry, inadequate regulatory oversight, uncertain long-term offset security and management, and lack of transparency and public accountability all contribute to an institutional and political environment which incentivizes the under-delivery of environmental outcomes from offsetting.

Biodiversity offset policy is promoted as a market-based solution which can deliver environmental outcomes more effectively and efficiently than ‘traditional’ regulation (Boisvert et al., 2013; ten Kate et al., 2004). However, the information requirements, transaction costs, and sheer complexity of the governance arrangements required to give effect to the ‘no net loss’ goal of biodiversity offsetting casts this claim into considerable doubt. Biodiversity loss is irreversible and subject to highly uncertain threshold effects, hence requires a precautionary policy approach (Gunningham and Young, 1997; Perrings and Pearce, 1994). A policy mix which is primarily underpinned by ‘traditional’ regulation may well be a more effective and efficient option than current biodiversity offset policy arrangements.

Future research directions

This thesis has described the evolution of contemporary policy responses to deforestation in Australia, and in doing so has illustrated that the intent, framing, and style of policies has generally shifted in each of the last four decades. Deforestation increasingly imperils threatened species and ecological communities, endangers marine ecosystems with pollution and sedimentation, and worsens climate change (Evans, 2016; Reside et al., 2017). The current trend towards deregulation and self-regulation of native vegetation clearing is at odds with the severity and irreversibility of these ecological impacts. Future work should examine the social, political and environmental consequences of this policy style over the next decade, and the prospects for a policy mix which is effective, efficient and equitable.

Evaluating the effect of regulatory policies on deforestation in Queensland was hampered by the limited number of available data points, and the considerable variation in deforestation trends and responses within and between local government areas that was unexplained by the macroeconomic, climatic, biophysical and institutional variables tested in the analysis. Expanding our analysis to the national scale will likely improve the capacity to detect any effect of policy interventions on deforestation, though may present additional complexities given the considerable variation in timing and scope of regulatory policies across different Australian states. Spatially explicit information on the broad vegetation groups (Nelder et al., 2017), primary land use (ABARES, 2010) and landholder compliance behaviours (Bartel and

Barclay, 2011) within each local government area could provide further insight into the ‘unexplained’ variation in deforestation behavior and strengthen our quantitative evaluation approach.

The environmental outcomes delivered from biodiversity offsetting is influenced not by metric design alone, but by a range of informational, institutional, organizational and political factors throughout the policy process. Future work should examine the biodiversity offset policy process and the actions and incentives of policy actors in other jurisdictions, to establish what lessons from Australia may generalize elsewhere, and in contrast, what aspects may be specific to a particular institutional, social and political context. A further useful contribution would be to investigate how financial institutions perceive biodiversity offsetting, and what may lead them to withhold mortgagee consent for an offset agreement. Although several interviewees suggested that financial institutions lack sufficient knowledge and understanding of biodiversity offsetting, it is possible that experiencing the rise and fall of the Australian carbon market has caused financial institutions to become more risk-averse to new and emerging markets.

The relaxation of regulatory controls on deforestation and concurrent rise in ‘new’ environmental policy instruments in Australia has occurred in the context of declining public investment in environmental protection (ACF, 2016). With this change in policy style, the governance of natural resources has increasingly been shifted from being the exclusive domain of the state, towards the private sector and voluntary community groups. This governance transition is occurring largely without the awareness of many non-state actors, and without an understanding of what risks and opportunities this new regime presents for the conservation of biodiversity. The current biodiversity offset policy settings in Australia are likely entrenching or worsening biodiversity decline, while the capacity and willingness of the state to engage in environmental protection appears to be deteriorating. Whether the private and voluntary sectors are sufficiently motivated to make up for this investment shortfall is not yet clear. My findings suggest widespread stakeholder interest in securing biodiversity offsets that delivers improved environmental and economic outcomes at the landscape scale. While the concept of “strategic” offsetting has been discussed in the

academic literature and policy for some time, there are few examples of how it may translate in practice. Several respondents from the private sector described how they are already designing and implementing offsets to be commercially viable, and establishing large-scale offsets in collaboration with businesses who are otherwise industry competitors. As such, the innovation and self-organisation displayed by private enterprise to deliver strategic offset outcomes warrants future research attention.

REFERENCES

- Abadie A, Diamond A, and Hainmueller J (2010) Synthetic control methods for comparative case studies: Estimating the effect of California's Tobacco Control Program. *Journal of the American Statistical Association* 105(490): 493–505.
- Abadie A, Diamond A, and Hainmueller J (2015) Comparative politics and the Synthetic Control Method. *American Journal of Political Science* 59(2): 495–510.
- ABARE (2009) *Australian beef 09.1*. Canberra: Commonwealth of Australia.
- ABARES (2010) *Land Use of Australia, Version 4, 2005/2006*. Canberra: Australian Bureau of Agricultural and Resource Economics – Bureau of Rural Sciences (ABARE–BRS).
- ABARES (2013) *Australian farm survey results 2010–11 to 2012–13*. Canberra: Australian Bureau of Agricultural and Resource Economics and Sciences.
- ABARES (2014) *Australia's State of the Forests Report 2013*. Available from: <http://www.daff.gov.au/ABARES/forestsaustralia/Pages/SOFR/sofr-2013.aspx>.
- ABARES (2015) *Agricultural commodity statistics 2015*. Canberra: Australian Bureau of Agricultural and Resource Economics and Sciences.
- ABS (2016) *3218.0 – Regional Population Growth, Australia, 2014–15*. Canberra: Australian Bureau of Statistics. Available from: <http://www.abs.gov.au/AUSSTATS/abs@.nsf/DetailsPage/3218.02014-15?OpenDocument> (accessed 28 June 2017).
- ACF (2001) *Australian Land Clearing, A Global Perspective: Latest Facts & Figures*. Fitzroy, Victoria: Australian Conservation Foundation (ACF).
- ACF (2016) *Funding for environment in drastic decline – MYEFO set to make things worse*. Canberra: Australian Conservation Foundation (ACF). Available from: https://d3n8a8pro7vhmx.cloudfront.net/auscon/pages/1526/attachments/original/1481681664/Federal_enviro_spending_brief_14-12-16.pdf?1481681664 (accessed 28 June 2017).
- Adams VM and Moon K (2013) Security and equity of conservation covenants: Contradictions of private protected area policies in Australia. *Land Use Policy* 30(1): 114–119.
- Adams WM and Sandbrook C (2013) Conservation, evidence and policy. *Oryx* 47(03): 329–335.

- Adger WN and Jordan A (2009) Sustainable development: exploring the processes and outcomes of governance. In: Adger WN and Jordan A (eds.), *Governing Sustainability*, Cambridge: Cambridge University Press.
- Agrawal A and Lemos MC (2007) A greener revolution in the making? Environmental governance in the 21st Century. *Environment: Science and Policy for Sustainable Development* 49(5): 36–45.
- Agrawal A and Ostrom E (2006) Political Science and Conservation Biology: a dialog of the deaf. *Conservation Biology* 20(3): 681–682.
- Akerlof GA (1970) The market for ‘lemons’: Quality uncertainty and the market mechanism. *The Quarterly Journal of Economics* 84(3): 488–500.
- Althaus C, Bridgman P, and Davis G (2012) *The Australian Policy Handbook*. Sydney: Allen & Unwin.
- Álvarez-Yépiz JC, Martínez-Yrizar A, Búrquez A, et al. (2008) Variation in vegetation structure and soil properties related to land use history of old-growth and secondary tropical dry forests in northwestern Mexico. *Forest Ecology and Management* 256(3): 355–366.
- Ambrose RF, Callway JC, and Lee SF (2007) *An Evaluation of Compensatory Mitigation Projects Permitted Under Clean Water Act Section 401 by the California State Water Resources Control Board, 1991-2002*. Final Report, Prepared for: California State Water Resources Control Board.
- Andam KS, Ferraro PJ, Pfaff A, et al. (2008) Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences of the United States of America* 105(42): 16089–94.
- Anderson JA, Long A, and Luckert MK (2014) A financial analysis of establishing poplar plantations for carbon offsets using Alberta and British Columbia’s afforestation protocols. *Canadian Journal of Forest Research* 45(2): 207–216.
- Ando AW and Chen X (2011) Optimal contract lengths for voluntary ecosystem service provision with varied dynamic benefit functions. *Conservation Letters* 4: 207–218.
- Angelsen A (2010) Policies for reduced deforestation and their impact on agricultural production. *Proceedings of the National Academy of Sciences of the United States of America* 107(46): 19639–44.
- Angelsen A and Kaimowitz D (1999) Rethinking the causes of deforestation: Lessons from economic models. *The World Bank Research Observer* 14(1): 73–98.
- Apostolopoulou E and Adams WM (2017) Biodiversity offsetting and conservation: reframing nature to save it. *Oryx* 51(1): 23–31.

-
- Arima EY, Barreto P, Araújo E, et al. (2014) Public policies can reduce tropical deforestation: Lessons and challenges from Brazil. *Land Use Policy* 41: 465–473.
- Arponen A, Heikkinen RK, Thomas CD, et al. (2005) The value of biodiversity in reserve selection: Representation, species weighting, and benefit functions. *Conservation Biology* 19(6): 2009–2014.
- Arriagada RA, Ferraro PJ, Sills EO, et al. (2012) Do payments for environmental services affect forest cover? A farm-level evaluation from Costa Rica. *Land Economics* 88(2): 382–399.
- Arrow KJ (1984) *The Economics of Agency*. Stanford: Institute for Mathematical Studies in the Social Sciences, Stanford University.
- Asner GP, Archer S, Hughes RF, et al. (2003) Net changes in regional woody vegetation cover and carbon storage in Texas Drylands, 1937–1999. *Global Change Biology* 9: 316–335.
- Australian Bureau of Statistics (2002) *Measuring Australia's Progress, 2002*. Canberra: Commonwealth of Australia. Available from: <http://www.abs.gov.au/ausstats/abs@.nsf/94713ad445ff1425ca25682000192af2/1c36c09104a4765aca256bdc001223fe!OpenDocument> (accessed 29 November 2015).
- Australian Department of the Environment (2015) *Human Induced Forest Extent & Change (version 11)*. Canberra: Commonwealth of Australia.
- Australian Department of the Environment (2016) *Australian National Inventory System Forest Extent Data (version 12)*. Canberra: Commonwealth of Australia.
- Australian Government (2012) *Environment Protection and Biodiversity Conservation Act 1999 Environmental Offsets Policy*. Department of Sustainability, Environment, Water, Population and Communities. Canberra: Commonwealth of Australia. Available from: <http://www.environment.gov.au/epbc/publications/environmental-offsets-policy.html>.
- Australian Government (2013) *Australian National Greenhouse Accounts: Australian Land Use, Land Use Change and Forestry Emissions Projections to 2030*. Canberra: Commonwealth of Australia.
- Australian Government (2014a) *Emissions Reduction Fund - White paper*. Canberra: Commonwealth of Australia. Available from: <http://www.environment.gov.au/climate-change/publications/emissions-reduction-fund-white-paper> (accessed 20 September 2014).
- Australian Government (2014b) *Environment Protection and Biodiversity Conservation Amendment (Bilateral Agreement Implementation) Bill 2014*. Canberra: Commonwealth of Australia. Available from:

- http://www.aph.gov.au/Parliamentary_Business/Bills_Legislation/Bills_Search_Results/Result?bId=r5231 (accessed 5 December 2015).
- Australian Greenhouse Office (2000) *Land clearing: a social history. National Carbon Accounting System Technical Report No. 4*. Canberra: Commonwealth of Australia.
- Australian Greenhouse Office (2003) *Greenhouse gas emissions from land use change in Australia : results of the National Carbon Accounting System 1988-2001*. Canberra: Australian Greenhouse Office.
- Australian National Audit Office (ANAO) (2003) *Referrals, Assessments and Approvals under the Environment Protection and Biodiversity Conservation Act 1999*. Audit Report No. 38 2002-03. Canberra: Commonwealth of Australia. Available from: <http://www.anao.gov.au/Publications/Audit-Reports/2013-2014/Managing-Compliance-with-EPBC-Act-1999-Conditions-of-Approval>.
- Australian National Audit Office (ANAO) (2007) *The Conservation and Protection of National Threatened Species and Ecological Communities*. Performance Audit Report No.31 2006–07. Canberra: Commonwealth of Australia. Available from: <http://www.anao.gov.au/Publications/Audit-Reports/2013-2014/Managing-Compliance-with-EPBC-Act-1999-Conditions-of-Approval>.
- Australian National Audit Office (ANAO) (2014) *Managing Compliance with EPBC Act 1999 Conditions of Approval*. Performance Audit Report No. 43 2013-14. Canberra: Commonwealth of Australia. Available from: <http://www.anao.gov.au/Publications/Audit-Reports/2013-2014/Managing-Compliance-with-EPBC-Act-1999-Conditions-of-Approval>.
- Australian State of the Environment Committee (2001) *State of the Environment 2001 (SoE 2001)*. Canberra: Commonwealth of Australia. Available from: <https://www.environment.gov.au/science-and-research/state-environment-reporting/soe-2001> (accessed 29 November 2015).
- Bailey AA (ed.) (1984) *The Brigalow belt of Australia*. Brisbane: Royal Society of Queensland.
- Bailey I, MacGill I, Passey R, et al. (2012) The fall (and rise) of carbon pricing in Australia: a political strategy analysis of the carbon pollution reduction scheme. *Environmental Politics* 21(5): 691–711.
- Barbier EB and Rauscher M (1994) Trade, tropical deforestation and policy interventions. *Environmental and Resource Economics* 4(1): 75–90.
- Barnosky AD, Matzke N, Tomiya S, et al. (2011) Has the Earth’s sixth mass extinction already arrived? *Nature* 471(7336): 51–57.
- Barson M, Randall L, and Bordas V (2000) *Land cover change in Australia: results of the collaborative Bureau of Rural Sciences - State agencies’ project on remote sensing*

- of agricultural land cover change*. Canberra: Bureau of Rural Sciences. Available from: <http://adl.brs.gov.au/brsShop/data/report.pdf>.
- Bartel R (2008) The formal and informal orders in land clearance regulation in Australia. In: *8th International Conference on Environmental Compliance and Enforcement*. Available from: <http://www.inece.org/conference/8/proceedings/proceedingsfinal.pdf#page=445> (accessed 11 September 2014).
- Bartel R and Barclay E (2011) Motivational postures and compliance with environmental law in Australian agriculture. *Journal of Rural Studies* 27(2): 153–170.
- Bartel RL (2003) Compliance and complicity: an assessment of the success of land clearance legislation in New South Wales. *Environmental and Planning Law Journal* 116: 116–141.
- Bartel RL (2004) Satellite imagery and land clearance legislation: A picture of regulatory efficacy? *Australasian Journal of Natural Resources Law and Policy* 9(1): 1–31.
- Bartel RL (2005) When the heavenly gaze criminalises: Satellite surveillance, land clearance regulation and the human-nature relationship. *Current Issues in Criminal Justice* 16: 322.
- Barton, D.N., Ring, I., Rusch, G.M., 2017. Policy Mixes: Aligning instruments for biodiversity conservation and ecosystem service provision. *Environmental Policy and Governance*. 27, 397–403. doi:10.1002/eet.1779
- Bascompte J, Possingham H, and Roughgarden J (2002) Patchy populations in stochastic environments: Critical number of patches for persistence. *American Naturalist* 159(2): 128–137.
- Baumol WJ and Oates WE (1988) *The Theory of Environmental Policy*. Cambridge: Cambridge University Press.
- Baylis K, Honey-Rosés J, Börner J, et al. (2016) Mainstreaming impact evaluation in nature conservation. *Conservation Letters* 9(1): 58–64.
- Beeton R, Buckley KI, Jones GJ, et al. (2006) *State of the Environment 2006 (SoE 2006)*. Independent report to the Australian Government Minister for the Environment and Heritage. Available from: <https://www.environment.gov.au/topics/science-and-research/state-environment-reporting/soe-2006> (accessed 29 November 2015).
- Beilharz N and Taylor H (2015) Victorian Government bans alpine national park grazing ‘forever’. *ABC Rural*, 6th May. Available from: <http://www.abc.net.au/news/2015-05-06/alpine-grazing-ban-victoria-fishers-shooters-party/6448608> (accessed 5 December 2015).

- Bekessy S A and Wintle BA (2008) Using carbon investment to grow the biodiversity bank. *Conservation Biology* 22(3): 510–513.
- Bekessy SA, Wintle BA, Lindenmayer DB, et al. (2010) The biodiversity bank cannot be a lending bank. *Conservation Letters* 3(3): 151–158.
- Benabou S (2014) Making up for lost nature? A critical review of the international development of voluntary biodiversity offsets. *Environment and Society* 5(1). Available from: <http://berghahnjournals.com/view/journals/environment-and-society/5/1/air-es050107.xml> (accessed 9 April 2017).
- Benítez PC, McCallum I, Obersteiner M, et al. (2007) Global potential for carbon sequestration: Geographical distribution, country risk and policy implications. *Ecological Economics* 60(3): 572–583.
- Bennett NJ (2016) Using perceptions as evidence to improve conservation and environmental management: Perceptions and Conservation. *Conservation Biology* 30(3): 582–592.
- Bennett NJ, Roth R, Klain SC, et al. (2017) Conservation social science: Understanding and integrating human dimensions to improve conservation. *Biological Conservation* 205: 93–108.
- Benson, D., Jordan, A., Smith, L., 2013. Is Environmental Management Really More Collaborative? A Comparative Analysis of Putative “Paradigm Shifts” in Europe, Australia, and the United States. *Environment and Planning A* 45, 1695–1712. doi:10.1068/a45378
- Bernhardt ES, Palmer MA, Allan JD, et al. (2005) Synthesizing U.S. river restoration efforts. *Science* 308(5722): 636–637.
- Bhattarai M and Hammig M (2004) Governance, economic policy, and the environmental Kuznets curve for natural tropical forests. *Environment and Development Economics* 9(03): 367–382.
- Bidaud C, Hrabanski M, and Meral P (2015) Voluntary biodiversity offset strategies in Madagascar. *Ecosystem Services* 15: 181–189.
- Bidaud C, Schreckenberg K, Rabeharison M, et al. (2017) The sweet and the bitter: Intertwined positive and negative social impacts of a biodiversity offset. *Conservation and Society* 15(1): 1 - 13.
- Biernacki P and Waldorf D (1981) Snowball sampling: Problems and techniques of chain referral sampling. *Sociological Methods & Research* 10(2): 141–163.
- Biesbroek R, Dupuis J, Jordan A, et al. (2015) Opening up the black box of adaptation decision-making. *Nature Climate Change* 5(6): 493–494.

- Birch JC, Newton AC, Aquino CA, et al. (2010) Cost-effectiveness of dryland forest restoration evaluated by spatial analysis of ecosystem services. *Proceedings of the National Academy of Sciences of the United States of America* 107(50): 21925–30.
- Biryahwaho B, Misiko M, Tefera H, et al. (2012) *Institutional innovations in African smallholder carbon projects. Case Study: Humbo Ethiopia Assisted Natural Regeneration Project*. CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS). Available from: <http://r4d.dfid.gov.uk/pdf/outputs/ccafs/africanagcarbon-casestudy-humbo.pdf> (accessed 23 January 2014).
- Bishop J, Kapila S, Hicks F, et al. (2009) New business models for biodiversity conservation. *Journal of Sustainable Forestry* 28(3–5): 285–303.
- Blaikie N (2009) *Designing Social Research*. Cambridge: Polity Press.
- Blakers M and Considine M (2016) *Mulga bills won't settle our climate accounts: An analysis of the Emissions Reductions Fund*. Canberra: The Green Institute.
- Bloomfield J and Pearson HL (2000) Land use, land-use change, forestry, and agricultural activities in the clean development mechanism: Estimates of greenhouse gas offset potential. *Mitigation and Adaptation Strategies for Global Change* 5(1): 9–24.
- Boisvert V (2015) Conservation banking mechanisms and the economization of nature: An institutional analysis. *Ecosystem Services* 15: 134–142.
- Boisvert V, Méral P, and Froger G (2013) Market-based instruments for ecosystem services: Institutional innovation or renovation? *Society & Natural Resources* 26(10): 1122–1136.
- Bombell A and Montoya D (2014) *Native vegetation clearing in NSW: a regulatory history*. NSW Parliamentary Research Service.
- Bottrill MC and Pressey RL (2012) The effectiveness and evaluation of conservation planning. *Conservation Letters* 5: 407–420.
- Bowen ME, McAlpine CA, Seabrook LM, et al. (2009) The age and amount of regrowth forest in fragmented Brigalow landscapes are both important for woodland dependent birds. *Biological Conservation* 142(12): 3051–3059.
- Boyce MS (1992) Population Viability Analysis. *Annual Review of Ecology and Systematics* 23(1): 481–497.
- Bradshaw CJA (2012) Little left to lose: deforestation and forest degradation in Australia since European colonization. *Journal of Plant Ecology* 5(1): 109–120.

-
- Bradshaw CJA, Bowman DMJS, Bond NR, et al. (2013) Brave new green world – Consequences of a carbon economy for the conservation of Australian biodiversity. *Biological Conservation* 161: 71–90.
- Braithwaite J (2011) The essence of responsive regulation. *The University of British Columbia Law Review* 44(3): 475–520.
- Brandt JS, Nolte C, and Agrawal A (2016) Deforestation and timber production in Congo after implementation of sustainable forest management policy. *Land Use Policy* 52: 15–22.
- Brewer, G.D., DeLeon, P., 1983. The foundations of policy analysis. Brooks/Cole, Pacific Grove.
- Bricknell S (2010) *Environmental crime in Australia*. AIC Reports: Research and Public Policy Series, Canberra: Australian Institute of Criminology.
- Bridgman P and Davis G (2000) *The Australian Policy Handbook*. Sydney: Allen & Unwin.
- Bridgman P and Davis G (2003) What use is a policy cycle? Plenty, if the aim is clear. *Australian Journal of Public Administration* 62(3): 98–102.
- Brook BW, Sodhi NS, and Ng PKL (2003) Catastrophic extinctions follow deforestation in Singapore. *Nature* 424(6947): 420–6.
- Brown M, Clarkson BD, Barton BJ, et al. (2013) Ecological compensation: an evaluation of regulatory compliance in New Zealand. *Impact Assessment and Project Appraisal* 31(1): 34–44.
- Brown M, Clarkson B, Barton B, et al. (2014) Implementing ecological compensation in New Zealand: stakeholder perspectives and a way forward. *Journal of the Royal Society of New Zealand* 44(1): 34–47.
- Brown SC and Veneman PLM (2001) Effectiveness of compensatory wetland mitigation in Massachusetts, USA. *Wetlands* 21(4): 508–518.
- Bruton MJ, McAlpine CA, and Maron M (2013) Regrowth woodlands are valuable habitat for reptile communities. *Biological Conservation* 165: 95–103.
- Bryan BA and Crossman ND (2013) Impact of multiple interacting financial incentives on land use change and the supply of ecosystem services. *Ecosystem Services* 4: 60–72.
- Bryan BA, Hajkowicz S, Marvanek S, et al. (2009) Mapping economic returns to agriculture for informing environmental policy in the Murray–Darling Basin, Australia. *Environmental Modeling & Assessment* 14(3): 375–390.

- Bryan BA, King D, and Ward JR (2011) Modelling and mapping agricultural opportunity costs to guide landscape planning for natural resource management. *Ecological Indicators* 11(1): 199–208.
- Bryan BA, Nolan M, Harwood TD, et al. (2014) Supply of carbon sequestration and biodiversity services from Australia's agricultural land under global change. *Global Environmental Change* 28: 166–181.
- Bulinski J, Enright R, and Tomsett N (2016) *Tree clearing in Australia: Its Contribution to Climate Change*. CO2 Australia Limited.
- Bull JW, Suttle KB, Gordon A, et al. (2013) Biodiversity offsets in theory and practice. *Oryx* 47(03): 369–380.
- Bull JW, Milner-Gulland EJ, Suttle KB, et al. (2014) Comparing biodiversity offset calculation methods with a case study in Uzbekistan. *Biological Conservation* 178: 2–10.
- Bull JW, Gordon A, Watson JEM, et al. (2016) Seeking convergence on the key concepts in 'no net loss' policy. *Journal of Applied Ecology* 53(6): 1686–1693.
- Bullock JM, Aronson J, Newton AC, et al. (2011) Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology & Evolution*. Available from: <http://dx.doi.org/10.1016/j.tree.2011.06.011>.
- Bunn DA, Moyle PB, and Johnson CK (2014) Maximizing the ecological contribution of conservation banks: Ecological Contribution of Conservation Banks. *Wildlife Society Bulletin* 38(2): 377–385.
- Bureau of Meteorology (2017a) *Average annual & monthly maximum, minimum, & mean temperature*. Available from: <http://www.bom.gov.au/climate/austmaps/about-temp-maps.shtml> (accessed 28 June 2017).
- Bureau of Meteorology (2017b) *Average annual, seasonal and monthly rainfall*. Available from: <http://www.bom.gov.au/climate/austmaps/about-vprp-maps.shtml> (accessed 28 June 2017).
- Bureau of Meteorology (2017c) *Daily 9am vapour pressure for Australia*. Available from: <http://www.bom.gov.au/climate/austmaps/about-vprp-maps.shtml> (accessed 28 June 2017).
- Burgin S (2008a) BioBanking: an environmental scientist's view of the role of biodiversity banking offsets in conservation. *Biodiversity and Conservation* 17(4): 807–816.
- Burgin S (2008b) Biodiversity offsets : Lessons from the American experience. *Australian Zoologist* 35(3): 544–550.

- Burgin S (2010) ‘Mitigation banks’ for wetland conservation: a major success or an unmitigated disaster? *Wetlands Ecology and Management* 18(1): 49–55.
- Burke PJ (2016) Undermined by adverse selection: Australia’s Direct Action Abatement Subsidies. *Economic Papers: A journal of applied economics and policy* 35(3): 216–229.
- Burney DA and Flannery TF (2005) Fifty millennia of catastrophic extinctions after human contact. *Trends in Ecology & Evolution* 20(7): 395–401.
- Busch J and Ferretti-Gallon K (2017) What drives deforestation and what stops it? A Meta-Analysis. *Review of Environmental Economics and Policy* 11(1): 3–23.
- Busch J, Ferretti-Gallon K, Engelmann J, et al. (2015) Reductions in emissions from deforestation from Indonesia’s moratorium on new oil palm, timber, and logging concessions. *Proceedings of the National Academy of Sciences of the United States of America* 112(5): 1328–1333.
- Büscher BE (2008) Conservation, Neoliberalism, and Social Science: a critical reflection on the SCB 2007 Annual Meeting in South Africa. *Conservation Biology* 22(2): 229–231.
- Bushnell JB (2010) *The Economics of Carbon Offsets*. SSRN Scholarly Paper, Rochester, NY: Social Science Research Network. Available from: <http://papers.ssrn.com/abstract=1667338> (accessed 23 October 2014).
- Business and Biodiversity Offsets Programme (BBOP) (2012a) *Biodiversity Offset Design Handbook-Updated*. Washington, D.C.: Forest Trends.
- Business and Biodiversity Offsets Programme (BBOP) (2012b) *Guidance Notes to the Standard on Biodiversity Offsets*. Washington, D.C.: Forest Trends.
- Business and Biodiversity Offsets Programme (BBOP) (2012c) *Resource Paper: No Net Loss and Loss-Gain Calculations in Biodiversity Offsets*. Washington, D.C.: Forest Trends. Available from: http://bbop.forest-trends.org/guidelines/Resource_Paper_NNL.pdf.
- Business and Biodiversity Offsets Programme (BBOP) (2012d) *Standard on Biodiversity Offsets*. Washington, D.C.: Forest Trends. Available from: <http://bbop.forest-trends.org/guidelines/Standard.pdf>.
- Butchart SHM, Walpole M, Collen B, et al. (2010) Global biodiversity: Indicators of recent declines. *Science* 328(5982): 1164–1168.
- Butchart SHM, Clarke M, Smith RJ, et al. (2015) Shortfalls and solutions for meeting national and global conservation area targets. *Conservation Letters* 8(5): 329–337.

- Butchart SHM, Di Marco M, and Watson JEM (2016) Formulating smart commitments on biodiversity: Lessons from the Aichi Targets. *Conservation Letters* 9(6): 457–468.
- Butler DW (2009) Planning iterative investment for landscape restoration: Choice of biodiversity indicator makes a difference. *Biological Conservation* 142(10): 2202–2216.
- Byron N, Craik W, Keniry J, et al. (2014) *A review of biodiversity legislation in NSW: Final report*. Sydney: State of NSW and the Office of Environment and Heritage.
- Cacho OJ, Hean RL, and Wise RM (2012) Carbon-accounting methods and reforestation incentives. *The Australian Journal of Agricultural and Resource Economics* 47(2): 153–179.
- Cacho OJ, Lipper L, and Moss J (2013) Transaction costs of carbon offset projects: A comparative study. *Ecological Economics* 88: 232–243.
- Cairney P (2012) *Understanding Public Policy: Theories and Issues*. Houndmills, Basingstoke, Hampshire; New York: Palgrave Macmillan.
- Cairney P, Oliver K, and Wellstead A (2016) To bridge the divide between evidence and policy: Reduce ambiguity as much as uncertainty. *Public Administration Review* 76(3): 399–402.
- Cardinale BJ, Duffy JE, Gonzalez A, et al. (2012) Biodiversity loss and its impact on humanity. *Nature* 486(7401): 59–67.
- Carson R (1962) *Silent Spring*. 1st edition. Boston; Cambridge, Mass.: Houghton Mifflin Co.
- Carwardine J, O'Connor T, Legge S, et al. (2012) Prioritizing threat management for biodiversity conservation. *Conservation Letters* 5(3): 196–204.
- Carwardine J, Hawkins C, Polglase P, et al. (2015) Spatial priorities for restoring biodiverse carbon forests. *BioScience* 65(4): 372–382.
- Catterall CP and Harrison DA (2006) *Rainforest Restoration Activities in Australia's Tropics and Subtropics*. Cairns, Australia: Cooperative Research Centre for Tropical Rainforest Ecology and Management. Rainforest CRC.
- CBD (2010) COP 10 Decision X/2. In: Nagoya, Japan. Available from: <https://www.cbd.int/decision/cop/?id=12268> (accessed 28 May 2017).
- Ceballos G, Ehrlich PR, Barnosky AD, et al. (2015) Accelerated modern human-induced species losses: Entering the sixth mass extinction. *Science Advances* 1(5): e1400253–e1400253.

-
- Chambers G (2016) Land-clearing laws retrospective. *The Australian*, 29th March. Available from: <http://at.theaustralian.com.au/link/7130d5c4c081a31eec6bfe409b069d03?domain=theaustralian.com.au> (accessed 1 April 2016).
- Charnley S, Diaz D, and Gosnell H (2010) Mitigating climate change through small-scale forestry in the USA: Opportunities and challenges. *Small-scale Forestry* 9(4): 445–462.
- Chazdon RL (2008) Beyond deforestation: restoring forests and ecosystem services on degraded lands. *Science* 320(5882): 1458–60.
- Chiu G, Lockhart R, and Routledge R (2006) Bent-Cable Regression theory and applications. *Journal of the American Statistical Association* 101(474): 542–553.
- Chiu GS and Lockhart RA (2010) Bent-cable regression with autoregressive noise. *Canadian Journal of Statistics* 38(3): 386–407.
- Clare S and Krogman N (2013) Bureaucratic slippage and environmental offset policies: The case of wetland management in Alberta. *Society & Natural Resources* 26(6): 672–687.
- Clare S, Krogman N, Foote L, et al. (2011) Where is the avoidance in the implementation of wetland law and policy? *Wetlands Ecology and Management* 19(2): 165–182.
- Cleveland WS and Devlin SJ (1988) Locally weighted regression: An approach to regression analysis by local fitting. *Journal of the American Statistical Association* 83(403): 596–610.
- COAG Standing Council on Environment and Water (2012) *Australia's Native Vegetation Framework*. Canberra: Australian Government, Department of Sustainability, Environment, Water, Population and Communities.
- Coggan A, Whitten SM, and Bennett J (2010) Influences of transaction costs in environmental policy. *Ecological Economics* 69(9): 1777–1784.
- Coggan A, Buitelaar E, Whitten SM, et al. (2013a) Intermediaries in environmental offset markets: Actions and incentives. *Land Use Policy* 32: 145–154.
- Coggan A, Buitelaar E, Whitten SM, et al. (2013b) Factors that influence transaction costs in development offsets: Who bears what and why? *Ecological Economics* 88: 222–231.
- Cohen, M.D., March, J.G., Olsen, J.P., 1972. A Garbage Can Model of Organizational Choice. *Administrative Science Quarterly* 17, 1–25. doi:10.2307/2392088

-
- Cole DH and Grossman PZ (2002) Toward a total-cost approach to environmental instrument choice. *Research in Law and Economics* 20: 223–241.
- Colloff MJ, Lavorel S, van Kerkhoff LE, et al. (2017) Transforming conservation science and practice for a post-normal world. *Conservation Biology*.
- Comerford E, Norman P, and Le Grand J (2011) *Report on the Project: Economics of Carbon Forestry in Queensland*. Brisbane: Queensland Department of Environment and Resource Management.
- Commonwealth of Australia (2009) *The Australian Environment Act: Report of the Independent Review of the Environment Protection and Biodiversity Conservation Act 1999 - Final report*. Canberra: Commonwealth of Australia. Available from: <http://www.environment.gov.au/epbc/review/publications/final-report.html> (accessed 8 July 2013).
- Commonwealth of Australia (2011) *Carbon Credits (Carbon Farming Initiative) Act 2011*. Canberra: Commonwealth of Australia. Available from: <http://www.comlaw.gov.au/Details/F2013L02036> (accessed 16 December 2013).
- Commonwealth of Australia (2012a) *Australian National Greenhouse Accounts: National Inventory Report 2010 Volume 2*. Canberra: Commonwealth of Australia.
- Commonwealth of Australia (2012b) *Carbon Farming (Quantifying Carbon Sequestration by Permanent Environmental Plantings of Native Species using the CFI Reforestation Modelling Tool) Methodology Determination 2012 - F2012L01340*. Canberra, Australia: Department of Climate Change and Energy Efficiency. Available from: <http://www.comlaw.gov.au/Details/F2013L01189>.
- Commonwealth of Australia (2012c) *The Carbon Farming Initiative Handbook*. Canberra: Commonwealth of Australia.
- Commonwealth of Australia (2013a) *Carbon Credits (Carbon Farming Initiative) (Human Induced Regeneration of a Permanent Even-Aged Native Forest—1.1) Methodology Determination 2013 - F2013L01189*. Canberra: Department of the Environment. Available from: <http://www.comlaw.gov.au/Details/F2013L01189>.
- Commonwealth of Australia (2013b) *Carbon Credits (Carbon Farming Initiative) (Native Forest from Managed Regrowth) Methodology Determination 2013 - F2013L02036*. Canberra: Department of the Environment. Available from: <http://www.comlaw.gov.au/Details/F2013L02036>.
- Commonwealth of Australia (2014) *Australian National Greenhouse Accounts: National Inventory Report 2012 Volume 2*. Canberra: Commonwealth of Australia.
- Commonwealth of Australia (2016a) *Outcomes-based conditions policy and guidance*. Canberra.

-
- Commonwealth of Australia (2016b) *Quarterly Update of Australia's National Greenhouse Gas Inventory: September 2015*. Canberra: Commonwealth of Australia.
- Condon M and Bryant S (2013) NSW Farmers welcome change in native veg policy. *ABC Rural*, 14th June. Available from: <http://www.abc.net.au/news/2013-06-14/native-vegetation-nsw-friday-stoner-nationals-confernece/4755492> (accessed 18 December 2015).
- Connolly E and Lewis C (2010) Structural change in the Australian economy. *Reserve Bank of Australia Bulletin* (September Quarter).
- Convention on Biological Diversity (2011) Aichi Biodiversity Targets. Available from: <http://www.cbd.int/sp/targets/> (accessed 5 August 2013).
- Corden WM (2012) Dutch disease in Australia: policy options for a three-speed economy. *Australian Economic Review* 45(3): 290–304.
- Coslovsky S, Pires R, and Silbey S (2011) The Pragmatic Politics of Regulatory Enforcement. In: Levi-Faur D (ed.), *Handbook on the Politics of Regulation*, Cheltenham, UK; Northampton, MA, USA: Edward Elgar Publishing.
- Costello C, Gaines SD, and Lynham J (2008) Can catch shares prevent fisheries collapse? *Science* 321(5896): 1678–1681.
- Council of Australian Governments (1992) *National Strategy for Ecologically Sustainable Development*. Canberra: Prepared by the Ecologically Sustainable Development Steering Committee. Available from: <http://www.environment.gov.au/about-us/esd/publications/national-esd-strategy> (accessed 3 December 2015).
- Council of Australian Governments (COAG) (1992) *Intergovernmental Agreement on the Environment*. Canberra.
- Crabbé A and Leroy P (2008) *The handbook of environmental policy evaluation*. London; Sterling, VA: Earthscan. Available from: <http://search.ebscohost.com/login.aspx?direct=true&scope=site&db=nlebk&db=nlabk&AN=250552> (accessed 11 September 2013).
- Cresswell ID and Murphy H (2016) Biodiversity: Biodiversity. In: *Australia state of the environment 2016*, Canberra: Australian Government Department of the Environment and Energy. Available from: <https://soe.environment.gov.au/theme/biodiversity> (accessed 15 June 2017).
- Creswell JW and Clark VLP (2011) *Designing and Conducting Mixed Methods Research* (2nd ed.). Thousand Oaks, CA: SAGE.
- Cripps A (2012) *Minister Orders Review of Vegetation Clearing Enforcement Processes*. Media release, Brisbane: Queensland Government. Available from: <http://statements.qld.gov.au/Statement/Id/79000> (accessed 5 December 2015).

- Crossman ND, Bryan BA, and Summers DM (2011) Carbon payments and low-cost conservation. *Conservation Biology* 25(4): 835–45.
- Crutzen PJ (2006) The ‘Anthropocene’. In: Ehlers PDE and Krafft DT (eds), *Earth System Science in the Anthropocene*, Springer Berlin Heidelberg, pp. 13–18. Available from: http://link.springer.com/chapter/10.1007/3-540-26590-2_3 (accessed 4 January 2017).
- Curran M, Hellweg S, and Beck J (2013) Is there any empirical support for biodiversity offset policy? *Ecological Applications* 24(4): 617–632.
- Curtis A, Robertson A, and Race D (1998) Lessons from recent evaluations of natural resource management programs in Australia. *Australian Journal of Environmental Management* 5(2): 109–119.
- Curzon HF and Kontoleon A (2016) From ignorance to evidence? The use of programme evaluation in conservation: Evidence from a Delphi survey of conservation experts. *Journal of Environmental Management* 180: 466–475.
- Czech B (2008) Prospects for reconciling the conflict between economic growth and biodiversity conservation with technological progress. *Conservation Biology* 22(6): 1389–1398.
- Damiens FLP, Mumaw L, Backstrom A, et al. (2017) Why politics and context matter in conservation policy. *Global Policy* 8(2): 1–4.
- Dargusch P and Griffiths A (2008) Introduction to special issue: a typology of environmental markets. *Australasian Journal of Environmental Management* 15(2): 70–75.
- Dart J and Grossek G (2007) *Evaluation of the Victorian Biodiversity Strategy - Summary Report*. Available from: <http://www.clearhorizon.com.au/wp-content/uploads/2011/05/final-summary-of-bse-november-2007.pdf>.
- Deacon RT (1994) Deforestation and the rule of law in a cross-section of countries. *Land Economics* 70(4): 414–430.
- Department of DEFRA (2012) *Biodiversity Offsetting Pilots. Technical Paper: the metric for the biodiversity offsetting pilot in England*.
- DeFries RS, Rudel T, Uriarte M, et al. (2010) Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geoscience* 3(3): 178–181.
- Deichmann U and Zhang F (2013) Farms and Forests. In: *Growing Green: The Economic Benefits of Climate Action*, Washington: World Bank Publications, 341–385.

-
- Denne T and Bond-Smith S (2012) *Discounting for Biodiversity Offsets*. Report prepared for NZ Department of Conservation: Covec. Available from: <http://www.covec.co.nz/pdf/Covec-Biodiversity-Offsets-Discounting.pdf>.
- Denzin NK and Lincoln YS (2005) *The SAGE Handbook of Qualitative Research*. Thousand Oaks, CA: SAGE.
- Department of Agriculture, Fisheries and Forestry (2014) *State of Queensland agriculture report*. Brisbane: State of Queensland.
- Department of Climate Change & Energy Efficiency (2004) *Maximum Potential Biomass (Maxbio)*. Canberra.
- Department of Environment and Climate Change (DECC) (2008) *BioBanking Assessment Methodology*. Sydney: NSW Government.
- Department of Environment and Primary Industries (2013) *Permitted clearing of native vegetation: Biodiversity assessment guidelines*. Melbourne: The State of Victoria Department of Environment and Primary Industries.
- Department of Environment and Resource Management (2009) *Vegetation Cover 2006b for Queensland Version 6.0b (November 2009)*.
- Department of Environment and Resource Management (2010) *Analysis of Woody Vegetation Clearing Rates in Queensland: Supplementary report to Land cover change in Queensland 2008–09*. Brisbane: State of Queensland.
- Department of Environment, Climate Change and Water NSW (2009) *Review of the Native Vegetation Act 2003*. Sydney: State of New South Wales.
- Department of Environment Regulation (2014) *A guide to the exemptions and regulations for clearing native vegetation*. Perth: State of Western Australia.
- Department of Natural Resources and Environment (1997) *Victoria's Biodiversity Strategy*. Melbourne: State of Victoria.
- Department of Natural Resources and Environment (2002) *Victoria's Native Vegetation Management: A Framework for Action*. Melbourne: State of Victoria.
- Department of Natural Resources and Mines (2005) *Land Cover Change in Queensland 2001-2003, incorporating 2001–2002 and 2002–2003 change periods*. Statewide Landcover and Trees Study (SLATS) report, Brisbane: State of Queensland.
- Department of Science, Information Technology and Innovation (2015) *Land cover change in Queensland 2012–13 and 2013–14*. Statewide Landcover and Trees Study, Brisbane: State of Queensland.

-
- Department of Science, Information Technology, Innovation and the Arts (2015) *Land cover change in Queensland 2012–13 - Preliminary Report*. Statewide Landcover and Trees Study Report, Brisbane: State of Queensland.
- Department of Science, Information Technology, Innovation and the Arts (2016) *Land cover change in Queensland 2014-15*. Statewide Landcover and Trees Study report, Brisbane: State of Queensland.
- Department of Sustainability and Environment (2008) *Native Vegetation Net gain accounting first approximation report*. East Melbourne: State of Victoria.
- Department of Sustainability and Environment (2010) *Native vegetation: Policy and planning*. East Melbourne: State of Victoria. Available from: http://www.dse.vic.gov.au/__data/assets/pdf_file/0017/124622/Native-Vegetation-Policy-and-Planning-Context_20110909.pdf.
- Department of Sustainability and Environment (2012) *Future directions for native vegetation in Victoria*. Melbourne: State of Victoria.
- Department of the Environment (2001) Land clearance. Available from: <http://www.environment.gov.au/node/14579> (accessed 29 November 2015).
- Department of the Environment and Energy (2016) *Annual Report 2015-16*. Available from: <https://www.environment.gov.au/annual-report-2015-16/downloads> (accessed 3 January 2017).
- Dirzo R, Young HS, Galetti M, et al. (2014) Defaunation in the Anthropocene. *Science* 345(6195): 401–406.
- Dobes L (2008) Getting real about adapting to climate change : Using ‘real options’ to address the uncertainties. *Agenda* 3: 55–69.
- Dolisca F, McDaniel JM, Teeter LD, et al. (2007) Land tenure, population pressure, and deforestation in Haiti: The case of Forêt des Pins Reserve. *Journal of Forest Economics* 13(4): 277–289.
- Domestic Offsets Integrity Committee (2011) *Endorsement of the Methodology for Quantifying Carbon Dioxide Sequestration by Permanent Environmental Plantings of Native Species using the CFI Reforestation Modelling Tool*. Available from: <http://www.climatechange.gov.au/government/initiatives/carbon-farming-initiative/methodology-development/approvedmethodologies/environmental-plantings.aspx>.
- Dovers S (2005) Clarifying the imperative of integration research for sustainable environmental management. *Journal of Research Practice* 1(2): Article M1.
- Dovers S (2013) The Australian environmental policy agenda. *Australian Journal of Public Administration* 72(2): 114–128.

- Dovers S and Hussey K (2013) *Environment & sustainability: a policy handbook*. Second edition. Annandale, NSW: The Federation Press.
- Dovers SR (1996) Sustainability: Demands on policy. *Journal of Public Policy* 16(3): 303–318.
- Druce A and Foley M (2015) Native veg reforms stalled until 2016. *The Land*. Available from: <http://www.theland.com.au/story/3449346/native-veg-reforms-stalled-until-2016/> (accessed 12 November 2015).
- Dunford RW, Ginn TC, and Desvousges WH (2004) The use of habitat equivalency analysis in natural resource damage assessments. *Ecological Economics* 48(1): 49–70.
- Dwyer JM, Fensham RJ, Butler DW, et al. (2009) Carbon for conservation: Assessing the potential for win–win investment in an extensive Australian regrowth ecosystem. *Agriculture, Ecosystems & Environment* 134: 1–7.
- Dwyer JM, Fensham RJ, and Buckley YM (2010a) Agricultural legacy, climate, and soil influence the restoration and carbon potential of woody regrowth in Australia. *Ecological applications : a publication of the Ecological Society of America* 20: 1838–50.
- Dwyer JM, Fensham R, and Buckley YM (2010b) Restoration thinning accelerates structural development and carbon sequestration in an endangered Australian ecosystem. *Journal of Applied Ecology* 47(3): 681–691.
- Earnest A, Morgan G, Mengersen K, et al. (2007) Evaluating the effect of neighbourhood weight matrices on smoothing properties of Conditional Autoregressive (CAR) models. *International Journal of Health Geographics* 6: 54.
- Edgar PW, Griffiths RA, and Foster JP (2005) Evaluation of translocation as a tool for mitigating development threats to great crested newts (*Triturus cristatus*) in England, 1990–2001. *Biological Conservation* 122(1): 45–52.
- Ehrenfeld D (2008) Neoliberalization of conservation. *Conservation Biology* 22(5): 1091–1092.
- Ehrlich PR (1968) *The Population Bomb*. Cutchogue, N.Y.: Buccaneer Books.
- Eisenhardt KM (1989) Agency theory: An assessment and review. *Academy of Management Review* 14(1): 57–74.
- Eisner R, Seabrook LM, and McAlpine CA (2016) Are changes in global oil production influencing the rate of deforestation and biodiversity loss? *Biological Conservation* 196: 147–155.

- Elks S (2016) To clear or not to clear: farmers in the dark on new laws. *The Australian*, 28th March. Available from: <http://www.theaustralian.com.au/news/inquirer/to-clear-or-not-to-clear-farmers-in-the-dark-on-new-laws/news-story/a24bfd12faf4057c1a2086f21b140b93> (accessed 2 April 2016).
- Engel S, Pagiola S, and Wunder S (2008) Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics* 65(4): 663–674.
- ESRI (2011) *ArcGIS Desktop: Release 10*. Redlands, CA: Environmental Systems Research Institute.
- Evans B (2014) *Annual precipitation: eMAST-R-Package 2.0, 0.01 degree, Australian Coverage, 1970-2012*. Melbourne: Macquarie University. Available from: http://datamgt.nci.org.au:8080/geonetwork/srv/en/metadata.show?uuid=eMAST_yr_prec_v1m1_1970_2012.
- Evans MC (2016) Deforestation in Australia: drivers, trends and policy responses. *Pacific Conservation Biology* 22(2): 130–150.
- Evans MC and Maron M (2016) *Submission to the NSW biodiversity legislation review*. Available from: https://mcevanresearch.files.wordpress.com/2014/06/evans_maron_submission-to-nsw-biodiversity-act.pdf.
- Evans MC, Watson JEM, Fuller RA, et al. (2011) The Spatial Distribution of Threats to Species in Australia. *BioScience* 61(4): 281–289.
- Evans MC, Carwardine J, Fensham RJ, et al. (2015) Carbon farming via assisted natural regeneration as a cost-effective mechanism for restoring biodiversity in agricultural landscapes. *Environmental Science & Policy* 50: 114–129.
- Everett S (2003) The policy cycle: Democratic process or rational paradigm revisited? *Australian Journal of Public Administration* 62(2): 65–70.
- Ewers RM, Laurance WF, and Souza CM (2008) Temporal fluctuations in Amazonian deforestation rates. *Environmental Conservation* 35(04): 303–310.
- FAO (2001) *Global Forest Resources Assessment 2000*. United Nations Food and Agriculture Organization (UNFAO). Available from: <http://www.fao.org/docrep/004/Y1997E/Y1997E00.HTM>.
- Fearnside PM (2001) Land-tenure issues as factors in environmental destruction in Brazilian Amazonia: The case of Southern Pará. *World Development* 29(8): 1361–1372.
- Fensham RJ and Guymer GP (2009) Carbon accumulation through ecosystem recovery. *Environmental Science & Policy* 12(3): 367–372.

-
- Ferraro PJ (2008) Asymmetric information and contract design for payments for environmental services. *Ecological Economics*, 65(4): 810–821.
- Ferraro PJ (2009) Counterfactual thinking and impact evaluation in environmental policy. *New Directions for Evaluation* 2009(122): 75–84.
- Ferraro PJ and Hanauer MM (2014) Advances in Measuring the Environmental and Social Impacts of Environmental Programs. *Annual Review of Environment and Resources* 39(1): 495–517.
- Ferraro PJ and Pattanayak SK (2006) Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biology* 4(4): e105–e105.
- Ferraro PJ, Hanauer MM, and Sims KRE (2011) Conditions associated with protected area success in conservation and poverty reduction. *Proceedings of the National Academy of Sciences of the United States of America* 108(34): 13913–13918.
- Ferraro PJ, Hanauer MM, Miteva DA, et al. (2013) More strictly protected areas are not necessarily more protective: evidence from Bolivia, Costa Rica, Indonesia, and Thailand. *Environmental Research Letters* 8(2): 025011.
- Fischer J, Stott J, Zerger A, et al. (2009) Reversing a tree regeneration crisis in an endangered ecoregion. *Proceedings of the National Academy of Sciences of the United States of America* 106: 10386–10391.
- Fitzsimons JA (2015) Private protected areas in Australia: current status and future directions. *Nature Conservation* 10: 1–23.
- Fitzsimons JA and Carr CB (2014) Conservation covenants on private land: Issues with measuring and achieving biodiversity outcomes in Australia. *Environmental Management* 54(606): 00 - 00.
- Folke, C., Hahn, T., Olsson, P., Norberg, J., 2005. Adaptive Governance of Social-Ecological Systems. *Annual Review of Environment and Resources* 30, 441–473. doi:10.1146/annurev.energy.30.050504.144511
- Fox J and Nino-Murcia A (2005) Status of species conservation banking in the United States. *Conservation Biology* 19(4): 996–1007.
- Freudenburg WR and Gramling R (1994) Bureaucratic slippage and failures of agency vigilance: The case of the environmental studies program. *Social Problems* 41(2): 214–239.
- Fuller RA, McDonald-Madden E, Wilson KA, et al. (2010) Replacing underperforming protected areas achieves better conservation outcomes. *Nature* 466(7304): 365–367.
- Funk JM, Field CB, Kerr S, et al. (2014) Modeling the impact of carbon farming on land use in a New Zealand landscape. *Environmental Science & Policy* 37: 1–10.

- Furby S (2002) *Land cover change: specifications for remote sensing analysis*. National Carbon Accounting System Technical Report No. 9. Canberra: Commonwealth of Australia.
- Galatowitsch SM (2009) Carbon offsets as ecological restorations. *Restoration Ecology* 17(5): 563–570.
- Game ET, Meijaard E, Sheil D, et al. (2014) Conservation in a wicked complex world; challenges and solutions. *Conservation Letters* 7(3): 271–277.
- Gardner TA, Von Hase A, Brownlie S, et al. (2013) Biodiversity offsets and the challenge of achieving no net loss. *Conservation Biology* 27(6): 1254–1264.
- Garnaut R (2008) Chapter 22: Transforming Rural Land Use. In: *The Garnaut Climate Change Review*. Cambridge University Press.
- Gaveau DLA, Curran LM, Paoli GD, et al. (2012) Examining protected area effectiveness in Sumatra: importance of regulations governing unprotected lands. *Conservation Letters* 5, 142–148. Available from: <http://doi.wiley.com/10.1111/j.1755-263X.2012.00220.x>.
- Gaveau DLA, Epting J, Lyne O, et al. (2009) Evaluating whether protected areas reduce tropical deforestation in Sumatra. *Journal of Biogeography* 36(11): 2165–2175.
- Geist HJ and Lambin EF (2002) Proximate causes and underlying driving forces of tropical deforestation. *BioScience* 52(2): 143–143.
- Gelman A, Carlin JB, Stern HS, et al. (2013) *Bayesian Data Analysis, Third Edition*. CRC Press.
- Geoscience Australia (1993) *Australian Land Tenure 1993*. Canberra: Commonwealth of Australia. Available from: http://www.ga.gov.au/metadata-gateway/metadata/record/gcat_42340.
- Geoscience Australia (2017) *Shuttle Radar Topographic Mission (SRTM) Level 2 Elevation Data: 1 second Digital Surface Model (SRTM-DSM Level 2) v1.0*. Canberra: Commonwealth of Australia. Available from: <http://data.aims.gov.au/metadataviewer/uuid/67b96045-2dae-47fb-90b9-4f6c5325764e> (accessed 9 June 2017).
- Gibbons P (2012) *Submission on the Draft Native Vegetation Regulation 2012*. Available from: <http://www.environment.nsw.gov.au/resources/vegetation/subs/NVRegSub394.pdf>.
- Gibbons P and Lindenmayer DB (2007) Offsets for land clearing: No net loss or the tail wagging the dog? *Ecological Management & Restoration* 8(1): 26–31.

- Gibbons P, Briggs SV, Ayers D, et al. (2009) An operational method to assess impacts of land clearing on terrestrial biodiversity. *Ecological Indicators* 9(1): 26–40.
- Gibbons P, Evans MC, Maron M, et al. (2016) A loss-gain calculator for biodiversity offsets and the circumstances in which no net loss is feasible. *Conservation Letters* 9(4): 252–259.
- Gibson L, Lee TM, Koh LP, et al. (2011) Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature* 478(7369): 378–381.
- Gifford RM (2000) *Carbon Contents of Above-Ground Tissues of Forest and Woodland Trees. National Carbon Accounting System Technical Report No. 22*. Canberra: Australian Greenhouse Office.
- Gilroy JJ, Woodcock P, Edwards FA, et al. (2014) Cheap carbon and biodiversity co-benefits from forest regeneration in a hotspot of endemism. *Nature Climate Change* 4(6): 503–507.
- Goldstein JH, Daily GC, Friday JB, et al. (2006) Business strategies for conservation on private lands: Koa forestry as a case study. *Proceedings of the National Academy of Sciences of the United States of America* 103(26): 10140–10145.
- Gómez-Baggethun E and Muradian R (2015) In markets we trust? Setting the boundaries of Market-Based Instruments in ecosystem services governance. *Ecological Economics* 117: 217–224.
- Gómez-Baggethun E, de Groot R, Lomas PL, et al. (2010) The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics* 69(6): 1209–1218.
- Gonçalves B, Marques A, Soares AMVDM, et al. (2015) Biodiversity offsets: from current challenges to harmonized metrics. *Current Opinion in Environmental Sustainability*, Open Issue 14: 61–67.
- Gordon A, Langford WT, Todd JA, et al. (2011) Assessing the impacts of biodiversity offset policies. *Environmental Modelling & Software* 26(12): 1481–1488.
- Gordon A, Bull JW, Wilcox C, et al. (2015) Perverse incentives risk undermining biodiversity offset policies. *Journal of Applied Ecology* 52(2): 532–537.
- Goulder LH and Parry IWH (2008) Instrument choice in environmental policy. *Review of Environmental Economics and Policy* 2(2): 152–174.
- Gowdy J, Howarth RB, and Tisdell C (2010) Discounting, ethics, and options for maintaining biodiversity and ecosystem integrity. In: Kumar, P (ed.) *The Economics of Ecosystems and Biodiversity: The Ecological and Economic Foundations*. London, Washington: Earthscan. Available from: <http://www.economics.rpi.edu/workingpapers/rpi1008.pdf> (accessed 14 May 2013).

- Gowdy JM (2008) Behavioral economics and climate change policy. *Journal of Economic Behavior & Organization* 68(3–4): 632–644.
- Gowen R, Rolfe J, and Donaghy P (2012) A bioeconomic model of carbon trading within an Australian grazing enterprise. Presented at the AARES 56th Annual Conference, Fremantle, Western Australia, February 7–10, 2012.
- Graetz R, Wilson MA, and Campbell SK (1995) Landcover Disturbance over the Australian Continent: a Contemporary Assessment. In: *Biodiversity Series, Paper No. 7*, Canberra, Australia: Department of the Environment, Sport and Territories, Biodiversity Unit.
- Grafton RQ (1996) Individual transferable quotas: theory and practice. *Reviews in Fish Biology and Fisheries* 6(1): 5–20.
- Greenstone M and Gayer T (2009) Quasi-experimental and experimental approaches to environmental economics. *Journal of Environmental Economics and Management* 57(1): 21–44.
- Gregory RG (1976) Some implications of the growth of the mineral sector. *Australian Journal of Agricultural Economics* 20(2): 71–91.
- Groombridge B and Jenkins M (2002) *World Atlas of Biodiversity: Earth's Living Resources in the 21st Century*. Berkeley and Los Angeles: University of California Press.
- Gunders J (2017) Weed and feed: Graziers push for mulga reclassification as woody weed and stock feed. *ABC Rural*. Available from: <http://www.abc.net.au/news/rural/2017-06-19/graziers-want-mulga-reclassified-as-woody-weed-and-drought-feed/8624052> (accessed 28 June 2017).
- Gunningham N and Grabosky PN (1998) *Smart Regulation: Designing Environmental Policy*. Oxford: Clarendon Press.
- Gunningham N and Sinclair D (1999) Regulatory pluralism: Designing policy mixes for environmental protection. *Law & Policy* 21(1): 49–76.
- Gunningham N and Young MD (1997) Toward optimal environmental policy: The case of biodiversity conservation. *Ecology Law Quarterly* 24: 243.
- Haddaway N and Pullin AS (2013) Evidence-based conservation and evidence-informed policy: a response to Adams & Sandbrook. *Oryx* 47(03): 336–338.
- Hahn R and Richards K (2013) Understanding the effectiveness of environmental offset policies. *Journal of Regulatory Economics* 44(1): 103–119.
- Hahn RW (1989) Economic prescriptions for environmental problems: How the patient followed the doctor's orders. *Journal of Economic Perspectives* 3(2): 95–114.

-
- Hahn RW and Hester GL (1989) Marketable permits: Lessons for theory and practice. *Ecology Law Quarterly* 16: 361.
- Hahn RW and Stavins RN (1991) Incentive-based environmental regulation: A new era from an old idea. *Ecology Law Quarterly* 18: 1.
- Hahn RW and Stavins RN (1992) Economic incentives for environmental protection: Integrating theory and practice. *The American Economic Review* 82(2): 464–468.
- Haines AL (1977) *The Yellowstone story: a story of our first national park, volume 1*. Boulder, CO: Colorado Association University Press.
- Hajkowicz S (2009) The evolution of Australia's natural resource management programs: Towards improved targeting and evaluation of investments. *Land Use Policy* 26(2): 471–478.
- Hansen MC, Stehman SV, and Potapov PV (2010) Quantification of global gross forest cover loss. *Proceedings of the National Academy of Sciences of the United States of America* 107(19): 8650–8655.
- Hansen MC, Potapov PV, Moore R, et al. (2013) High-resolution global maps of 21st-century forest cover change. *Science (New York, N.Y.)* 342(2013): 850–3.
- Hardy MJ, Fitzsimons JA, Bekessy SA, et al. (2016) Exploring the permanence of conservation covenants: Permanence of conservation covenants. *Conservation Letters* 10(2): 221–230.
- Hargrave J and Kis-Katos K (2013) Economic causes of deforestation in the Brazilian Amazon: A panel data analysis for the 2000s. *Environmental and Resource Economics* 54(4): 471–494.
- Harris C (2013) Native vegetation heritage: South Australian initiatives to better manage the State's native vegetation. *A history of agriculture in South Australia*. Available from: http://www.pir.sa.gov.au/aghistory/left_nav/natural_resources/native_vegetation_conservation/national_parks (accessed 5 December 2015).
- Harrison M (2010) *Valuing the Future: the social discount rate in cost-benefit analysis - Productivity Commission Visiting Researcher Paper*. Canberra: Productivity Commission. Available from: <http://www.pc.gov.au/research/visiting-researcher/cost-benefit-discount> (accessed 17 September 2014).
- Hatanaka N, Wright W, Loyn RH, et al. (2011) 'Ecologically complex carbon' – linking biodiversity values, carbon storage and habitat structure in some austral temperate forests. *Global Ecology and Biogeography* 20(2): 260–271.
- Hay I (2010) *Qualitative Research Methods in Human Geography*. 3 edition. Oxford; New York: Oxford University Press.

- Hayes N and Morrison-Saunders A (2007) Effectiveness of environmental offsets in environmental impact assessment: practitioner perspectives from Western Australia. *Impact Assessment and Project Appraisal* 25(3): 209–218.
- Henderson N and Sutherland W (1996) Two truths about discounting and their environmental consequences. *Trends in Ecology & Evolution* 11(12): 527.
- Hijmans RJ and van Etten J (2012) raster: Geographic analysis and modeling with raster data. R package version 2.0-12. Available from: <http://CRAN.R-project.org/package=raster>.
- Hockenstein J, Stavins R, and Whitehead B (1997) Crafting the next generation of market-based environmental tools. *Environment: Science and Policy for Sustainable Development* 39: 12–33.
- Hockings, M., Leverington, F., Cook, 2015. Protected Area Management Effectiveness, in: Worboys, G.L., Lockwood, M., Kothari, A., Feary, S., Pulsford, I. (Eds.), *Protected Area Governance and Management*. ANU Press.
- Hoffmann M, Hilton-Taylor C, Angulo A, et al. (2010) The impact of conservation on the status of the world's vertebrates. *Science* 330(6010): 1503–1509.
- Holmes G, Sandbrook C, and Fisher JA (2016) Understanding conservationists' perspectives on the new-conservation debate. *Conservation Biology* 31(2): 353–363.
- Hood C (1986) *The Tools of Government*. London: Chatham House Publishers.
- Hood C (2007) Intellectual obsolescence and intellectual makeovers: Reflections on the tools of government after two decades. *Governance* 20(1): 127–144.
- Hooper DU, Adair EC, Cardinale BJ, et al. (2012) A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature* 486(7401): 105–108.
- Hornyak MM and Halvorsen KE (2003) Wetland mitigation compliance in the Western Upper Peninsula of Michigan. *Environmental Management* 32(5): 535–540.
- Howe C, Suich H, Vira B, et al. (2014) Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change* 28: 263–275.
- Howlett M (1991) Policy instruments, policy styles, and policy implementation. *Policy Studies Journal* 19(2): 1–21.
- Howlett M, Ramesh M, and Perl A (2009) *Studying Public Policy: Policy Cycles and Policy Subsystems*. 3 edition. Ont.; New York: Oxford University Press.

-
- Hutton J, Adams WM, and Murombedzi JC (2005) Back to the barriers? Changing narratives in biodiversity conservation. *Forum for Development Studies* 32(2): 341–370.
- IFC (2012) *Performance Standard 6: Biodiversity Conservation and Sustainable Management of Living Natural Resources*. Washington, D.C.: International Finance Corporation.
- IUCN (2001) *IUCN Red List Categories and Criteria: Version 3.1*. Gland, Switzerland and Cambridge, UK: IUCN. Available from: http://www.iucnredlist.org/documents/redlist_cats_crit_en.pdf.
- IUCN (2016) *IUCN Policy on Biodiversity Offsets*. Gland, Switzerland: International Union for the Conservation of Nature.
- Ives CD and Bekessy SA (2015) The ethics of offsetting nature. *Frontiers in Ecology and the Environment* 13(10): 568–573.
- Jack BK, Kousky C, and Sims KRE (2008) Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms. *Proceedings of the National Academy of Sciences of the United States of America* 105(28): 9465–9470.
- Jha S and Bawa KS (2006) Population growth, human development, and deforestation in biodiversity hotspots. *Conservation Biology* 20(3): 906–912.
- Johnson RB and Onwuegbuzie AJ (2004) Mixed methods research: A research paradigm whose time has come. *Educational Researcher* 33(7): 14–26.
- Jones DA, Wang W, and Fawcett R (2009) High-quality spatial climate data-sets for Australia. *Australian Meteorological and Oceanographic Journal* 58(4): 233.
- Joppa LN and Pfaff A (2011) Global protected area impacts. *Proceedings of the Royal Society B: Biological Sciences* 278(1712): 1633–1638.
- Jordan A, Wurzel RKW, and Zito AR (2003a) Comparative conclusions - ‘new’ environmental policy instruments: An evolution or a revolution in environmental policy? *Environmental Politics* 12(1): 201–224.
- Jordan A, Wurzel RKW, and Zito AR (2003b) ‘New’ instruments of environmental governance: patterns and pathways of change. *Environmental Politics* 12(1): 1–24.
- Jordan A, Wurzel RKW, and Zito A (2005) The rise of ‘new’ policy instruments in comparative perspective: Has governance eclipsed government? *Political Studies* 53(3): 477–496.

- Jordan A, Wurzel RKW, and Zito AR (2013) Still the century of ‘new’ environmental policy instruments? Exploring patterns of innovation and continuity. *Environmental Politics* 22(1): 155–173.
- Kagan RA, Gunningham N, and Thornton D (2003) Explaining corporate environmental performance: How does regulation matter? *Law & Society Review* 37(1): 51–90.
- Kaplowitz MD, Lupi F, and Bailey D (2008) Wetland mitigation banking: The bankers’ perspective. *Journal of Soil and Water Conservation* 63(3): 162–172.
- Kareiva P and Fuller E (2016) Beyond resilience: How to better prepare for the profound disruption of the Anthropocene. *Global Policy* 7: 107–118.
- Keenan RJ, Reams GA, Achard F, et al. (2015) Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management* 352: 9–20.
- Keene M and Pullin AS (2011) Realizing an effectiveness revolution in environmental management. *Journal of Environmental Management* 92(9): 2130–5.
- Kehoe J (2009) Environmental law making in Queensland: The Vegetation Management Act 1999 (Qld). *Environmental and planning law journal* 26(5): 392–410.
- Keith DA, Rodríguez JP, Rodríguez-Clark KM, et al. (2013) Scientific Foundations for an IUCN Red List of Ecosystems. *PLoS ONE* 8(5): e62111.
- Kentula ME (2000) Perspectives on setting success criteria for wetland restoration. *Ecological Engineering* 15(3–4): 199–209.
- Kery M (2010) *Introduction to WinBUGS for Ecologists: Bayesian Approach to Regression, ANOVA, Mixed Models and Related Analyses*. Academic Press.
- Kesteven J and Landsberg J (2004) *National Carbon Accounting System Technical Report No. 23: Forest Productivity Model*. Australia: Commonwealth of Australia/Australian Greenhouse Office.
- Khan SA, Chiu G and Dubin JA (2009) Atmospheric concentration of chlorofluorocarbons: Addressing the global concern with the longitudinal bent-cable model. *CHANCE* 22(3): 8–17.
- Khan SA, Rana M, Li L, and Dubin, JA (2012) A statistical investigation to monitor and understand atmospheric CFC decline with the spatial-longitudinal bent-cable model. *International Journal of Statistics and Probability* 1(2): 56–68. Available from: <http://ccsenet.org/journal/index.php/ijsp/article/view/17253> (accessed 7 April 2014).
- Kingdon JW (1984) *Agendas, alternatives, and public policies*. Boston: Little, Brown & Co.

- Kingsford RT, Watson JEM, Lundquist CJ, et al. (2009) Major conservation policy issues for biodiversity in Oceania. *Conservation Biology* 23(4): 834–840.
- Kirby KR, Laurance WF, Albernaz AK, et al. (2006) The future of deforestation in the Brazilian Amazon. *Futures* 38(4): 432–453.
- Kirkpatrick JB (1991) The magnitude and significance of land clearance in Tasmania in the 1980s. *Tasforests* 3: 11–14.
- Kirkpatrick JB (1994) *A continent transformed: human impact on the natural vegetation of Australia*. Melbourne: Oxford University Press.
- Kirkpatrick JB (1998) Nature conservation and the Regional Forest Agreement Process. *Australian Journal of Environmental Management* 5(1): 31–37.
- Kiser E (1999) Comparing varieties of agency theory in economics, political science, and sociology: An illustration from state policy implementation. *Sociological Theory* 17(2): 146–170.
- Kleiman DG, Reading RP, Miller BJ, et al. (2000) Improving the evaluation of conservation programs. *Conservation Biology* 14(2): 356–365.
- Knill C and Liefnerink D (2013) The establishment of EU environmental policy. In: Adelle C and Jordan A (eds), *Environmental Policy in the EU: Actors, Institutions and Processes*, London: Routledge. Available from: <http://search.ebscohost.com/login.aspx?direct=true&db=nlebk&AN=504215&site=ehost-live>.
- Kragt ME, Dumbrell NP, and Blackmore L (2017) Motivations and barriers for Western Australian broad-acre farmers to adopt carbon farming. *Environmental Science & Policy* 73: 115–123.
- Laffont J-J and Martimort D (2009) *The Theory of Incentives: The Principal-Agent Model*. Princeton, NJ: Princeton University Press.
- Lambin EF and Meyfroidt P (2011) Global land use change, economic globalization, and the looming land scarcity. *Proceedings of the National Academy of Sciences of the United States of America* 108(9): 3465–3472.
- Lambin EF, Turner BL, Geist HJ, et al. (2001) The causes of land-use and land-cover change: moving beyond the myths. *Global Environmental Change* 11(4): 261–269.
- Lane J (2013) *Native Vegetation Regulation Review: Facilitator's Final Report*. Report to the Minister for the Environment, Sydney: Government of New South Wales. Available from: <http://www.environment.nsw.gov.au/vegetation/ReviewofNVRegulations.htm>.

- Lane MB (1999) Regional forest agreements: Resolving resource conflicts or managing resource politics? *Australian Geographical Studies* 37(2): 142–153.
- Lasswell HD (1971) *A pre-view of policy sciences*. New York: American Elsevier Pub. Co.
- Laurance WF (1999) Reflections on the tropical deforestation crisis. *Biological Conservation* 91(2): 109–117.
- Laurance WF (2001) The future of the Brazilian Amazon. *Science* 291(5503): 438–439.
- Laurance WF, Useche DC, Rendeiro J, et al. (2012) Averting biodiversity collapse in tropical forest protected areas. *Nature* 489(7415): 290–4.
- Laurance WF, Sayer J, and Cassman KG (2014) Agricultural expansion and its impacts on tropical nature. *Trends in Ecology & Evolution* 29(2): 107–116.
- Law BE, Thornton PE, Irvine J, et al. (2001) Carbon storage and fluxes in ponderosa pine forests at different developmental stages. *Global Change Biology* 7(7): 755–777.
- le Polain de Waroux Y, Garrett RD, Heilmayr R, et al. (2016) Land-use policies and corporate investments in agriculture in the Gran Chaco and Chiquitano. *Proceedings of the National Academy of Sciences of the United States of America* 113(15): 4021–4026.
- Lehmann EA, Wallace JF, Caccetta PA, et al. (2013) Forest cover trends from time series Landsat data for the Australian continent. *International Journal of Applied Earth Observation and Geoinformation* 21: 453–462.
- Lemos MC and Agrawal A (2006) Environmental Governance. *Annual Review of Environment and Resources* 31(1): 297–325.
- Lewis SL and Maslin MA (2015) Defining the Anthropocene. *Nature* 519(7542): 171–180.
- Lin BB, Macfadyen S, Renwick AR, et al. (2013) Maximizing the environmental benefits of carbon farming through ecosystem service delivery. *BioScience* 63(10): 793–803.
- Lindenmayer D (2005) Chapter 9: Vegetation loss and degradation. In: Lindenmayer D and Burgman M (eds), *Practical Conservation Biology*, Collingwood, Victoria: CSIRO Publishing. Available from: <http://www.publish.csiro.au/pid/5034.htm> (accessed 21 November 2015).
- Lindenmayer D (2014) Forests, forestry and forest management. In: Lindenmayer D, Morton S, and Dovers S (eds), *Ten Commitments Revisited: Securing Australia's Future Environment*, Collingwood, Victoria: CSIRO Publishing.
- Lindenmayer DB (2015) Continental-level biodiversity collapse. *Proceedings of the National Academy of Sciences of the United States of America*: 201502766.

-
- Lindenmayer DB, Hulvey KB, Hobbs RJ, et al. (2012) Avoiding bio-perversity from carbon sequestration solutions. *Conservation Letters* 5: 28–36.
- Lindenmayer DB, Crane M, Evans MC, et al. (2017) The anatomy of a failed offset. *Biological Conservation* 210, Part A: 286–292.
- Lindquist EJ, D’Annunzio R, Gerrand A, et al. (2012) *Global forest land-use change 1990–2005*. Rome: Food and Agriculture Organization of the United Nations.
- Liu J, Li S, Ouyang Z, et al. (2008) Ecological and socioeconomic effects of China’s policies for ecosystem services. *Proceedings of the National Academy of Sciences of the United States of America* 105(28): 9477–9482.
- Liverman D (2004) Who governs, at what scale and at what price? Geography, environmental governance, and the commodification of nature. *Annals of the Association of American Geographers* 94(4): 734–738.
- Lockie S (2013) Market instruments, ecosystem services, and property rights: Assumptions and conditions for sustained social and ecological benefits. *Land Use Policy* 31: 90–98.
- Lockie S and Higgins V (2007) Roll-out neoliberalism and hybrid practices of regulation in Australian agri-environmental governance. *Journal of Rural Studies* 23(1): 1–11.
- Lockwood M and Davidson J (2010) Environmental governance and the hybrid regime of Australian natural resource management. *Geoforum* 41(3): 388–398.
- Lockwood M, Davidson J, Curtis A, et al. (2010) Governance principles for natural resource management. *Society & Natural Resources* 23(10): 986–1001.
- Lubell M (2013) Governing institutional complexity: The ecology of games framework. *Policy Studies Journal* 41(3): 537–559.
- Lubell M (2015) Collaborative partnerships in complex institutional systems. *Current Opinion in Environmental Sustainability* 12: 41–47.
- Lukey P, Cumming T, Paras S, et al. (In press) Making biodiversity offsets work in South Africa – A governance perspective. *Ecosystem Services*. Available from: <http://www.sciencedirect.com/science/article/pii/S2212041617303017>.
- Lunn DJ, Thomas A, Best N, et al. (2000) WinBUGS - A Bayesian modelling framework: Concepts, structure, and extensibility. *Statistics and Computing* 10(4): 325–337.
- Ma M, Haapanen T, Singh RB, et al. (2014) Integrating ecological restoration into CDM forestry projects. *Environmental Science & Policy* 38: 143–153.

- Macedo MN, DeFries RS, Morton DC, et al. (2012) Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *Proceedings of the National Academy of Sciences of the United States of America* 109(4): 1341–1346.
- Macintosh A (2004) Why the Environment Protection and Biodiversity Conservation Act's referral, assessment and approval process is failing to achieve its environmental objectives. *Environmental and Planning Law Journal* 21(4): 288–311.
- Macintosh A (2009a) *Australia's National Environmental Legislation: A Response to Early*. SSRN Scholarly Paper, Rochester, NY: Social Science Research Network. Available from: <http://papers.ssrn.com/abstract=1546134> (accessed 27 October 2013).
- Macintosh A (2009b) The Environment Protection and Biodiversity Conservation Act 1999 (Cth): An evaluation of its cost-effectiveness. *Environmental and Planning Law Journal and Planning Law Journal* 26(5): 337–362.
- Macintosh A (2012) The Australia clause and REDD: a cautionary tale. *Climatic Change* 112(2): 169–188.
- Macintosh A (2013) The Carbon Farming Initiative: removing the obstacles to its success. *Carbon Management* 4(2): 185–202.
- Macintosh A (2015) The impact of ESD on Australia's environmental institutions. *Australasian Journal of Environmental Management* 22(1): 33–45.
- Macintosh A and Waugh L (2012) An introduction to the Carbon Farming Initiative : Key principles and concepts. *Environmental and Planning Law Journal* 2: 439–461.
- Macintosh A and Waugh L (2014) Compensatory mitigation and screening rules in environmental impact assessment. *Environmental Impact Assessment Review* 49: 1–12.
- Macintosh AK (2010a) Best practice environmental impact assessment: A model framework for Australia. *Australian Journal of Public Administration* 69(4): 401–417.
- Macintosh AK (2010b) The Australian Government's environmental impact assessment (EIA) regime: using surveys to identify proponent views on cost-effectiveness. *Impact Assessment and Project Appraisal* 28(3): 175–188.
- Madsen B, Carroll N, and Moore Brands K (2010) *State of Biodiversity Markets Report: Offset and Compensation Programs Worldwide*. Available from: Available at: <http://www.ecosystemmarketplace.com/documents/acrobat/sbdlmr.pdf>.
- Marcos-Martinez R, Bryan BA, Connor JD, et al. (2017) Agricultural land-use dynamics: Assessing the relative importance of socioeconomic and biophysical drivers for more targeted policy. *Land Use Policy* 63: 53–66.

-
- Margoluis R, Stem C, Salafsky N, et al. (2009) Design alternatives for evaluating the impact of conservation projects. *New Directions for Evaluation* 2009(122): 85–96.
- Margules CR and Pressey RL (2000) Systematic conservation planning. *Nature* 405(6783): 243–253.
- Marinoni O, Navarro Garcia J, Marvanek S, et al. (2012) Development of a system to produce maps of agricultural profit on a continental scale: An example for Australia. *Agricultural Systems* 105(1): 33–45.
- Maron M and Fitzsimons JA (2007) Agricultural intensification and loss of matrix habitat over 23 years in the West Wimmera, south-eastern Australia. *Biological Conservation* 135(4): 587–593.
- Maron M, Dunn PK, McAlpine CA, et al. (2010) Can offsets really compensate for habitat removal? The case of the endangered red-tailed black-cockatoo. *Journal of Applied Ecology* 47(2): 348–355.
- Maron M, Hobbs RJ, Moilanen A, et al. (2012) Faustian bargains? Restoration realities in the context of biodiversity offset policies. *Biological Conservation* 155: 141–148.
- Maron M, Rhodes JR, and Gibbons P (2013) Calculating the benefit of conservation actions. *Conservation Letters* 6(5): 359–367.
- Maron M, Catterall C, Possingham HP, et al. (2013) Public Statement of Concern from Queensland Scientists. *Public Statement of Concern from Queensland Scientists*. Available from: <https://concernedqldscientists.wordpress.com/> (accessed 30 November 2015).
- Maron M, Bull JW, Evans MC, et al. (2015) Locking in loss: Baselines of decline in Australian biodiversity offset policies. *Biological Conservation* 192: 504–512.
- Maron M, Ives CD, Kujala H, et al. (2016) Taming a wicked problem: Resolving controversies in biodiversity offsetting. *BioScience* 66(6): 489–498.
- Martin N, Evans M, Rice J, et al. (2016) Using offsets to mitigate environmental impacts of major projects: A stakeholder analysis. *Journal of Environmental Management* 179: 58–65.
- Martin TG (2010) Grazing away our woodland birds. In: Lindenmayer D, Bennett AF, and Hobbs RJ (eds), *Temperate Woodland Conservation and Management* Place: Collingwood, Victoria: CSIRO Publishing, 249–258.
- Martin TG and McIntyre S (2007) Impacts of livestock grazing and tree clearing on birds of woodland and riparian habitats. *Conservation biology : the journal of the Society for Conservation Biology* 21(2): 504–14.

-
- Martin TG, Catterall CP, Manning A, Szabo, J (2012) Australian Birds in a Changing Landscape: 220 years of European colonization. In: Fuller RJ (ed.), *Birds and Their Habitats*. Cambridge, United Kingdom: Cambridge University Press, 453–480.
- Maseyk FJF, Evans MC and Maron M (2017) *Guidance for deriving ‘Risk of Loss’ estimates when evaluating biodiversity offset proposals under the EPBC Act*. Report to the National Environmental Science Programme Department of the Environment and Energy.
- MATLAB version 7.10.0.499 (2010) Natick, Massachusetts: The Mathworks.
- May J, Hobbs RJ, and Valentine LE (2017) Are offsets effective? An evaluation of recent environmental offsets in Western Australia. *Biological Conservation* 206: 249–257. Available from: <http://linkinghub.elsevier.com/retrieve/pii/S0006320716309363> (accessed 19 December 2016).
- McAlpine CA, Fensham RJ, and Temple-Smith DE (2002) Biodiversity conservation and vegetation clearing in Queensland: principles and thresholds. *The Rangeland Journal* 24(1): 36–36.
- McCarthy DP, Donald PF, Scharlemann JPW, et al. (2012) Financial costs of meeting global biodiversity conservation targets: Current spending and unmet needs. *Science* 338(6109): 946–949.
- McCarthy MA, Andelman SJ, and Possingham HP (2003) Reliability of relative predictions in population viability analysis. *Conservation Biology* 17(4): 982–989.
- McDonald JA, Carwardine J, Joseph LN, et al. (2015) Improving policy efficiency and effectiveness to save more species: A case study of the megadiverse country Australia. *Biological Conservation* 182: 102–108.
- McDonald-Madden E, Gordon A, Wintle BA, et al. (2009) ‘True’ Conservation Progress. *Science* 323(5910): 43–44.
- McGrath CJ (2007) End of broadscale clearing in Queensland. *Environment and Planning Law Journal* 24: 5–13.
- McIntyre S and Martin TG (2002) Managing intensive and extensive land uses to conserve grassland plants in sub-tropical eucalypt woodlands. *Biological Conservation* 107(2): 241–252.
- McKenney B and Kiesecker JM (2010) Policy development for biodiversity offsets: a review of offset frameworks. *Environmental Management* 45(1): 165–76.
- Meadows DH, Meadows DL, Randers J, et al. (1972) *Limits to Growth: A Report for the Club of Rome’s Project on the Predicament of Mankind*. New York: Universal Books.

-
- Messina JP, Walsh SJ, Mena CF, et al. (2006) Land tenure and deforestation patterns in the Ecuadorian Amazon: Conflicts in land conservation in frontier settings. *Applied Geography*, 26(2): 113–128.
- Meyfroidt P and Lambin EF (2009) Forest transition in Vietnam and displacement of deforestation abroad. *Proceedings of the National Academy of Sciences of the United States of America* 106(38): 16139–16144.
- Mickwitz P (2003) A framework for evaluating environmental policy instruments: Context and key concepts. *Evaluation* 9(4): 415–436.
- Miles MB, Huberman AM and Saldaña J (2013) *Qualitative Data Analysis*. Thousand Oaks, CA: SAGE.
- Millennium Ecosystem Assessment (2005) *Ecosystems and human well-being: Policy Responses: Findings of the Responses Working Group of the Millennium Ecosystem Assessment*. Washington, DC.: Island Press.
- Miller GH, Magee JW, Johnson BJ, et al. (1999) Pleistocene extinction of *Genyornis newtoni*: Human impact on Australian megafauna. *Science* 283(5399): 205–208.
- Miller KL, Trezise JA, Kraus S, et al. (2015) The development of the Australian environmental offsets policy: from theory to practice. *Environmental Conservation* 42(4): 306–314.
- Mitchell CD, Harper RJ, and Keenan RJ (2012) Current status and future prospects for carbon forestry in Australia. *Australian Forestry* 75(3): 200–212.
- Miteva DA, Pattanayak SK, and Ferraro PJ (2012) Evaluation of biodiversity policy instruments: what works and what doesn't? *Oxford Review of Economic Policy* 28(1): 69–92.
- Mittermeier RA and Mittermeier CG (eds) (1997) *Megadiversity: Earth's Biologically Wealthiest Nations*. 1st edition. México, D.F.: CEMEX.
- Mittermeier RA, Myers N, Mittermeier CG, Robles Gil, P. (1999) *Hotspots: Earth's biologically richest and most endangered terrestrial ecoregions*. Mexico City, Mexico: CEMEX, S.A., Agrupación Sierra Madre, S.C.
- Moilanen A, van Teeffelen AJA, Ben-Haim Y, et al. (2009) How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratios for impacted habitat. *Restoration Ecology* 17(4): 470–478.
- Moon K and Blackman D (2014) A guide to understanding social science research for natural scientists: Social science for natural scientists. *Conservation Biology* 28(5): 1167–1177.

- Moreno-Mateos D, Maris V, Béchet A, et al. (2015) The true loss caused by biodiversity offsets. *Biological Conservation* 192: 552–559.
- Munro NT, Lindenmayer DB, and Fischer J (2007) Faunal response to revegetation in agricultural areas of Australia: A review. *Ecological Management & Restoration* 8(3): 199–207.
- Munro NT, Fischer J, Wood J, et al. (2009) Revegetation in agricultural areas: the development of structural complexity and floristic diversity. *Ecological Applications* 19(5): 1197–1210.
- Muradian R and Gómez-Baggethun E (2013) The institutional dimension of ‘Market-Based Instruments’ for governing ecosystem services: Introduction to the Special Issue. *Society & Natural Resources* 26(10): 1113–1121.
- Muradian R, Arsel M, Pellegrini L, et al. (2013) Payments for ecosystem services and the fatal attraction of win-win solutions. *Conservation Letters* 6(4): 274–279.
- Myers N, Mittermeier RA, Mittermeier CG, et al. (2000) Biodiversity hotspots for conservation priorities. *Nature* 403(6772): 853–858.
- National Research Council (U.S.) and Committee on Mitigating Wetland Losses (2001) *Compensating for wetland losses under the Clean Water Act*. Washington, D.C.: National Academy Press. Available from: <http://search.ebscohost.com/login.aspx?direct=true&scope=site&db=nlebk&db=nlabk&AN=86906> (accessed 18 June 2013).
- Natural Resource Management Ministerial Council (2001) *National Framework for the Management and Monitoring of Australia’s Native Vegetation*. Canberra: Department of Environment and Heritage.
- Natural Resource Management Ministerial Council (NRMCC) (2009) *Australia’s Strategy for the National Reserve System 2009 - 2030*. Canberra: Commonwealth of Australia.
- Nature Conservation Council of NSW (2016) *Paradise Lost - The weakening and widening of NSW biodiversity offsetting schemes, 2005-2016*.
- Nelder VJ, Niehus RE, Wilson BA, et al. (2017) *The Vegetation of Queensland. Descriptions of Broad Vegetation Groups*. Brisbane: Queensland Herbarium, Department of Science, Information Technology and Innovation.
- Neldner VJ, Wilson BA, Thompson EJ, et al. (2005) *Methodology for Survey and Mapping of Regional Ecosystems and Vegetation Communities in Queensland, Version 3.1*. Brisbane:

-
- Nelson E, Polasky S, Lewis DJ, et al. (2008) Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. *Proceedings of the National Academy of Sciences of the United States of America* 105(28): 9471–9476.
- Nemes V, Plott CR and Stoneham G (2008) Electronic BushBroker exchange: Designing a combinatorial double auction for native vegetation offsets. *SSRN Electronic Journal*: 1–36.
- Nepstad D, McGrath D, Stickler C, et al. (2014) Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science* 344(6188): 1118–1123.
- Newell P (2008) The political economy of global environmental governance. *Review of International Studies* 34(03): 507–529.
- Newell P, Pattberg P and Schroeder H (2012) Multiactor governance and the environment. *Annual Review of Environment and Resources* 37(1): 365–387.
- Nicholson E, Keith DA and Wilcove DS (2009) Assessing the threat status of ecological communities. *Conservation Biology* 23(2): 259–274.
- Nicol S, Evans MC, Chadès I, Whitten S, Wolfe N, and Rough T (2014) *Developing a risk-based approach to environmental compliance and monitoring*. Australia: CSIRO.
- Niles JO, Brown S, Pretty J, et al. (2002) Potential carbon mitigation and income in developing countries from changes in use and management of agricultural and forest lands. *Philosophical Transactions of the Royal Society of London. Series A: Mathematical, Physical and Engineering Sciences* 360(1797): 1621–1639.
- Nolte C, Agrawal A, Silvius KM, et al. (2013) Governance regime and location influence avoided deforestation success of protected areas in the Brazilian Amazon. *Proceedings of the National Academy of Sciences of the United States of America* 110(13): 4956–4961.
- Nolte C, le Polain de Waroux Y, Munger J, et al. (2017) Conditions influencing the adoption of effective anti-deforestation policies in South America’s commodity frontiers. *Global Environmental Change* 43: 1–14.
- North DC (1991) Institutions. *The Journal of Economic Perspectives* 5(1): 97–112.
- Norton T and Mitchell N (1993) Towards the sustainable management of southern temperate forest ecosystems: lessons from Australia and New Zealand. *Pacific Conservation Biology* 1(4): 293–300.
- Norton TW (1996) Conserving biological diversity in Australia’s temperate eucalypt forests. *Forest Ecology and Management* 85(1–3): 21–33.

-
- Novacek MJ and Cleland EE (2001) The current biodiversity extinction event: Scenarios for mitigation and recovery. *Proceedings of the National Academy of Sciences of the United States of America* 98(10): 5466–5470.
- O'Connor P (2009) *BushBroker Implementation: Evaluation after two years of operations*. Report prepared for the Department of Sustainability and Environment, Victoria.
- Ostrom E, Janssen MA, and Anderies JM (2007) Going beyond panaceas. *Proceedings of the National Academy of Sciences of the United States of America* 104(39): 15176–15178.
- Overton JM, Stephens RTT, and Ferrier S (2013) Net present biodiversity value and the design of biodiversity offsets. *Ambio* 42(1): 100–10.
- Paterson S and Bryan BA (2012) Food-carbon trade-offs between agriculture and reforestation land uses under alternate market-based policies. *Ecology and Society* 17(3). Available from: <http://www.ecologyandsociety.org/vol17/iss3/art21/> (accessed 9 December 2013).
- Pattanayak SK, Wunder S, and Ferraro PJ (2010) Show me the money: Do payments supply environmental services in developing countries? *Review of Environmental Economics and Policy* 4(2): 254–274.
- Paul KI, Reeson A, Polglase P, et al. (2013) Economic and employment implications of a carbon market for integrated farm forestry and biodiverse environmental plantings. *Land Use Policy* 30: 496–506.
- Pawliczek J and Sullivan S (2011) Conservation and concealment in SpeciesBanking.com, USA: an analysis of neoliberal performance in the species offsetting industry. *Environmental Conservation* 38(04): 435–444.
- Peel J and Godden L (2005) Australian environmental management: A ‘Dams’ Story. *The University of New South Wales Law Journal* 28(3): 668.
- Penca J (2013) Marketing the market: The ideology of market mechanisms for biodiversity conservation. *Transnational Environmental Law* 2(02): 235–257.
- Penca J (2015) Biodiversity offsetting in transnational governance: Biodiversity offsetting. *Review of European, Comparative & International Environmental Law* 24(1): 93–102.
- Perrings C and Pearce D (1994) Threshold effects and incentives for the conservation of biodiversity. *Environmental and Resource Economics* 4(1): 13–28.
- Perry M (2015) *rasterstats 0.7.0*. Available from: <https://github.com/perrygeo/python-rasterstats>.

-
- Peters-Stanley M, Gonzalez G, and Yin D (2013) *Covering New Ground: State of the Forest Carbon Markets 2013*.
- Pezzey JCV and Jotzo F (2013) Carbon tax needs thresholds to reach its full potential. *Nature Climate Change* 3(12): 1008–1011.
- Phelps J., Webb EL, and Adams WM (2012a) Biodiversity co-benefits of policies to reduce forest-carbon emissions. *Nature Climate Change* 2: 497–503.
- Phelps J., Friess DA, and Webb EL (2012b) Win–win REDD+ approaches belie carbon–biodiversity trade-offs. *Biological Conservation* 154: 53–60.
- Pichancourt J-B, Firn J, Chadès I, et al. (2014) Growing biodiverse carbon-rich forests. *Global Change Biology* 20(2): 382–393.
- Pickett EJ, Stockwell MP, Bower DS, et al. (2013) Achieving no net loss in habitat offset of a threatened frog required high offset ratio and intensive monitoring. *Biological Conservation* 157: 156–162.
- Pilgrim JD, Brownlie S, Ekstrom JMM, et al. (2013) A process for assessing the offsetability of biodiversity impacts. *Conservation Letters* 6(5): 376–384.
- Pimm SL and Raven P (2000) Biodiversity: Extinction by numbers. *Nature* 403(6772): 843–845.
- Pimm SL, Russell GJ, Gittleman JL, et al. (1995) The Future of Biodiversity. *Science* 269(5222): 347–350.
- Pimm SL, Jenkins CN, Abell R, et al. (2014) The biodiversity of species and their rates of extinction, distribution, and protection. *Science* 344(6187): 1246752.
- Pirard R (2012) Market-based instruments for biodiversity and ecosystem services: A lexicon. *Environmental Science & Policy* 19–20: 59–68.
- Pittock J, Hussey K and McGlennon S (2013) Australian climate, energy and water policies: conflicts and synergies. *Australian Geographer* 44: 3–22.
- Pittock J, Hussey K and Dovers S (2015) Ecologically sustainable development in broader retrospect and prospect: evaluating national framework policies against climate adaptation imperatives. *Australasian Journal of Environmental Management* 22(1): 62–76.
- Polglase P, Paul K, Hawkins C, et al. (2008) *Regional Opportunities for Agroforestry Systems in Australia*. RIRDC Publication No. 08/176
- Polglase PJ, Reeson A, Hawkins CS, et al. (2013) Potential for forest carbon plantings to offset greenhouse emissions in Australia: economics and constraints to implementation. *Climatic Change* 121(2): 161–175.

-
- Pressey RL, Ferrier S, Hager TC, et al. (1996) How well protected are the forests of north-eastern New South Wales? -- Analyses of forest environments in relation to formal protection measures, land tenure, and vulnerability to clearing. *Forest Ecology and Management* 85(1–3): 311–333.
- Pressey RL, Whish GL, Barrett TW, et al. (2002) Effectiveness of protected areas in north-eastern New South Wales: recent trends in six measures. *Biological Conservation* 106(1): 57–69.
- Pressey RL, Visconti P, and Ferraro PJ (2015) Making parks make a difference: poor alignment of policy, planning and management with protected-area impact, and ways forward. *Philosophical Transactions of the Royal Society B* 370(1681): 20140280.
- Primmer E (2017) Institutional constraints on conservation auction: Organizational mandate, competencies and practices. *Land Use Policy* 63: 621–631.
- Productivity Commission (1999) *Implementation of Ecologically Sustainable Development by Commonwealth Departments and Agencies*. Inquiry Report, Canberra: Commonwealth of Australia. Available from: <http://www.pc.gov.au/inquiries/completed/ecologically-sustainable-development/report> (accessed 23 January 2017).
- Productivity Commission (2004) *Impacts of native vegetation and biodiversity regulations*. Report no. 29. Melbourne: Productivity Commission.
- Pullin AS and Knight TM (2009) Doing more good than harm - Building an evidence-base for conservation and environmental management. *Biological Conservation* 142(5): 931–934.
- Pullin AS, Knight TM, Stone DA, et al. (2004) Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation* 119(2): 245–252.
- Purdy R (2009) Using earth observation technologies for better regulatory compliance and enforcement of environmental laws. *Journal of Environmental Law* 22(1): 59–87.
- Queensland Audit Office (2015) *Managing water quality in Great Barrier Reef catchments*. Brisbane: State of Queensland ??.
- Quétier F and Lavorel S (2011) Assessing ecological equivalence in biodiversity offset schemes: Key issues and solutions. *Biological Conservation* 144(12): 2991–2999.
- Quigley JT and Harper DJ (2006) Effectiveness of fish habitat compensation in Canada in achieving no net loss. *Environmental Management* 37(3): 351–66.

-
- R Development Core Team (2012) *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. Available from: <http://www.R-project.org>.
- R Development Core Team (2017) *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. Available from: <http://www.R-project.org>.
- Rands MRW, Adams WM, Bennun L, et al. (2010) Biodiversity conservation: Challenges beyond 2010. *Science* 329(5997): 1298–1303.
- Redford KH and Taber A (2000) Writing the wrongs: Developing a safe-fail culture in conservation. *Conservation Biology* 14(6): 1567–1568.
- Regan HM, Colyvan M and Burgman MA (2002) A taxonomy and treatment of uncertainty for ecology and conservation biology. *Ecological Applications* 12(2): 618–628.
- Regan TJ, Taylor BL, Thompson GG, et al. (2013) Testing decision rules for categorizing species' extinction risk to help develop quantitative listing criteria for the U.S. Endangered Species Act. *Conservation Biology* 27(4): 821–831.
- Reiss KC, Hernandez E, Brown MT, et al. (2009) Evaluation of permit success in wetland banking: a Florida case study. *Wetlands* 29(3): 907–918.
- Renwick AR, Robinson CJ, Martin TG, et al. (2014) Biodiverse planting for carbon and biodiversity on Indigenous land. *PLoS ONE* 9(3): e91281.
- Reserve Bank of Australia (2014). *Inflation Calculator*. Available from: <http://www.rba.gov.au/calculator/> (accessed 21 September 2014).
- Reside AE, Beher J, Cosgrove AJ, et al. (2017) Ecological consequences of land clearing and policy reform in Queensland. *Pacific Conservation Biology*. 00. Available from: <http://www.publish.csiro.au/PC/PC17001> (accessed 19 June 2017).
- Rey Benayas J (2007) Abandonment of agricultural land: an overview of drivers and consequences. *CAB Reviews: Perspectives in Agriculture, Veterinary Science, Nutrition and Natural Resources* 2(057). Available from: <http://www.cabi.org/cabreviews/?loadmodule=review&page=4051&reviewid=33805&Site=167> (accessed 16 December 2013).
- Rey Benayas JM, Newton AC, Diaz A, et al. (2009) Enhancement of biodiversity and ecosystem services by ecological restoration: A Meta-Analysis. *Science* 325(5944): 1121–1124.
- Rhodes J, Beyer H, Preece H, et al. (2015) *South East Queensland Koala Population Modelling Study*. Brisbane, Australia: UniQuest.

- Rhodes JR, Wiegand T, McAlpine CA, et al. (2006) Modeling species' distributions to improve conservation in semiurban landscapes: Koala case study. *Conservation Biology* 20(2): 449–459.
- Richards G. and Brack C. (2004) A continental biomass stock and stock change estimation approach for Australia. *Australian Forestry* 67: 284–288.
- Richards KR and Stokes C (2004) A review of forest carbon sequestration cost studies: A dozen years of research. *Climatic Change* 63(1/2): 1–48.
- Richards P, Arima E, VanWey L, et al. (2017) Are Brazil's deforesters avoiding detection? *Conservation Letters*. Available from: <http://onlinelibrary.wiley.com/doi/10.1111/conl.12310/abstract>.
- Richards PD, Myers RJ, Swinton SM, et al. (2012) Exchange rates, soybean supply response, and deforestation in South America. *Global Environmental Change*, 22(2): 454–462.
- Ring I, Drechsler M, van Teeffelen AJ, et al. (2010) Biodiversity conservation and climate mitigation: what role can economic instruments play? *Current Opinion in Environmental Sustainability* 2(1–2): 50–58.
- Ring I, Hansjürgens B, Elmqvist T, et al. (2010) Challenges in framing the economics of ecosystems and biodiversity: the TEEB initiative. *Current Opinion in Environmental Sustainability* 2(1–2): 15–26.
- Ritchie EG, Bradshaw CJA, Dickman CR, et al. (2013) Continental-scale governance and the hastening of loss of Australia's biodiversity. *Conservation Biology* 27(6): 1133–1135.
- Robertson M and Hayden N (2008) Evaluation of a market in wetland credits: Entrepreneurial wetland banking in Chicago. *Conservation Biology* 22(3): 636–646.
- Robertson MM (2000) No net loss: Wetland restoration and the incomplete capitalization of nature. *Antipode* 32(4): 463–493.
- Robins L and Kanowski P (2011) 'Crying for our Country': eight ways in which 'Caring for our Country' has undermined Australia's regional model for natural resource management. *Australasian Journal of Environmental Management* 18(2): 88–108.
- Robinson BE, Holland MB and Naughton-Treves L (2014) Does secure land tenure save forests? A meta-analysis of the relationship between land tenure and tropical deforestation. *Global Environmental Change* 29: 281–293.
- Robinson CJ, Renwick AR, May T, et al. (2016) Indigenous benefits and carbon offset schemes: An Australian case study. *Environmental Science & Policy* 56: 129–134.

-
- Rockstrom J, Steffen W, Noone K, et al. (2009) A safe operating space for humanity. *Nature* 461(7263): 472–475.
- Rolfe J (2000) Broadscale tree clearing in Queensland. *Agenda* 7(3): 219–236.
- Rolfe J (2002) Economics of vegetation clearing in Queensland. *The Rangeland Journal* 24(1): 152–169.
- Rossi PH, Lipsey MW and Freeman HE (2004) *Evaluation: A Systematic Approach*. Thousand Oaks, CA: SAGE.
- Rudel TK, Coomes OT, Moran E, et al. (2005) Forest transitions: towards a global understanding of land use change. *Global Environmental Change* 15(1): 23–31.
- Rule S, Brook BW, Haberle SG, et al. (2012) The aftermath of megafaunal extinction: Ecosystem transformation in pleistocene Australia. *Science* 335(6075): 1483–1486.
- Salzman J and Ruhl JB (2000) Currencies and the commodification of environmental law. *Stanford Law Review* 53(3): 607–694.
- Salzman J and Ruhl JB (2010) Gaming the past: The theory and practice of historic baselines in the administrative state. *Vanderbilt Law Review*: 1–57.
- Sampaio AB, Holl KD and Scariot A (2007) Regeneration of seasonal deciduous forest tree species in long-used pastures in Central Brazil. *Biotropica* 39: 655–659.
- Sánchez-Azofeifa GA, Pfaff A, Robalino JA, et al. (2007) Costa Rica’s payment for environmental services program: Intention, implementation, and impact. *Conservation Biology* 21(5): 1165–1173.
- Sandbrook CG, Fisher JA and Vira B (2013) What do conservationists think about markets? *Geoforum* 50: 232–240.
- Saunders DA (1989) Changes in the Avifauna of a region, district and remnant as a result of fragmentation of native vegetation: the wheatbelt of western Australia. A case study. *Biological Conservation*, 50(1–4): 99–135.
- Scanlan JC (1991) Woody overstorey and herbaceous understorey biomass in *Acacia harpophylla* (brigalow) woodlands. *Austral Ecology* 16: 521–529.
- Schirmer J and Field J (2000) *The Cost of Revegetation*. Canberra: ANU Forestry and FORTECH.
- Scriven M (1991) *Evaluation Thesaurus*. Thousand Oaks, CA: SAGE Publications.
- Seabrook L, McAlpine C, and Fensham R (2006) Cattle, crops and clearing: Regional drivers of landscape change in the Brigalow Belt, Queensland, Australia, 1840–2004. *Landscape and Urban Planning* 78: 373–385.

- Senate Environment and Communications References Committee (2015) *Report - National Landcare Program*. Canberra: Commonwealth of Australia. Available from: http://www.aph.gov.au/About_Parliament/Parliamentary_Departments/Parliamentary_Library/pubs/rp/BudgetReview201415/Environment (accessed 6 December 2015).
- Senate Legislation Committee Environment and Communications (2015) *Environment Protection and Biodiversity Conservation Amendment (Standing) Bill 2015*. Canberra: Commonwealth of Australia. Available from: http://www.aph.gov.au/Parliamentary_Business/Committees/Senate/Environment_and_Communications/EPBC_Standing_Bill/Report (accessed 5 December 2015).
- Senate Standing Committee on Environment and Communications Legislation Committee (2015) *Answers to questions on notice: Environment portfolio, Question No. 150*.
- Senate Standing Committee on Environment and Communications Legislation Committee (2016) *Answers to questions on notice: Environment portfolio, Question No. 06*.
- Shapiro SP (2005) Agency Theory. *Annual Review of Sociology* 31(1): 263–284.
- Shono K, Cadaweng EA and Durst PB (2007) Application of assisted natural regeneration to restore degraded tropical forestlands. *Restoration Ecology* 15(4): 620–626.
- Shoo LP, Storlie C, Vanderwal J, et al. (2011) Targeted protection and restoration to conserve tropical biodiversity in a warming world. *Global Change Biology* 17(1): 186–193.
- Short J and Smith A (1994) Mammal decline and recovery in Australia. *Journal of Mammalogy* 75(2): 288–297.
- Sills EO, Herrera D, Kirkpatrick AJ, et al. (2015) Estimating the impacts of local policy innovation: The synthetic control method applied to tropical deforestation. *PLOS ONE* 10(7): e0132590.
- Silver WL, Ostertag R and Lugo AE (2000) The potential for carbon sequestration through reforestation of abandoned tropical agricultural and pasture lands. *Restoration Ecology* 8(4): 394–407.
- Slee D (1998) *Remnant native vegetation – perceptions and policies: A review of legislation and incentive programs*. Research Report 2/98, Canberra: Environment Australia.
- Smith J (2002) Afforestation and reforestation in the clean development mechanism of the Kyoto Protocol: implications for forests and forest people. *International Journal of Global Environmental Issues* 2(3): 322–343.
- Smith J and Scherr SJ (2003) Capturing the value of forest carbon for local livelihoods. *World Development* 31(12): 2143–2160.

-
- Snowdon P, Eamus D, Gibbons P, et al. (2000) *National Carbon Accounting System, Technical Report No . 17 Synthesis of Allometrics, Review of Root Biomass and Design of Strategies*. Canberra: Commonwealth of Australia/Australian Greenhouse Office ???
- Soares-Filho B, Rajão R, Macedo M, et al. (2014) Cracking Brazil's forest code. *Science* 344(6182): 363–364.
- Sodhi NS, Koh LP, Brook BW, et al. (2004) Southeast Asian biodiversity: an impending disaster. *Trends in ecology & evolution* 19(12): 654–60.
- Spash CL (2015) Bulldozing biodiversity: The economics of offsets and trading-in nature. *Biological Conservation* 192: 541–551.
- Spash CL and Aslaksen I (2015) Re-establishing an ecological discourse in the policy debate over how to value ecosystems and biodiversity. *Journal of Environmental Management* 159: 245–253.
- Squelch J (2007) Land clearing laws in Western Australia. *The Agricultural Industry* 9. Available from: http://cbsntweb.curtin.edu.au/files/Land_Clearing_Laws_in_Western_Australia.pdf.
- St. John FAV, Keane AM, Jones JPG, et al. (2014) Robust study design is as important on the social as it is on the ecological side of applied ecological research. *Journal of Applied Ecology* 51(6): 1479–1485.
- Stafford Smith M, Horrocks L, Harvey A, et al. (2011) Rethinking adaptation for a 4°C world. *Philosophical transactions. Series A, Mathematical, physical, and engineering sciences* 369(1934): 196–216.
- Standing Committee on the Environment (2014) *Streamlining environmental legislation: Inquiry into streamlining environmental regulation, 'green tape', and one stop shops*. Canberra: Commonwealth of Australia. Available from: <http://www.aph.gov.au/greentapereport> (accessed 5 December 2015).
- Standing Committee on the Environment and Public Affairs (2015) *Petition No. 42 – Request to repeal the Environmental Protection (Environmentally Sensitive Areas) Notice 2005*. Perth: Government of Western Australia.
- State of Queensland (1999) *Vegetation Management Act 1999*. Brisbane: State of Queensland. Available from: http://www.austlii.edu.au/au/legis/qld/consol_act/vma1999212/ (accessed 9 June 2017).
- State of Queensland (2016) *Vegetation Management (Reinstatement) and Other Legislation Amendment Bill 2016*. Brisbane: State of Queensland.

-
- State of the Environment 2011 Committee (2011) *Australia State of the Environment 2011*. Independent report to the Australian Government Minister for Sustainability, Environment, Water, Population and Communities. Canberra: ??
- Stavins RN (2003) Chapter 9 Experience with market-based environmental policy instruments. In: Karl-Göran Mäler and Jeffrey R. Vincent (ed.), *Handbook of Environmental Economics*, Amsterdam; Boston: Elsevier, 355–435. Available from: <http://www.sciencedirect.com/science/article/pii/S1574009903010143> (accessed 29 July 2013).
- Stavins RN, Keohane N, and Revesz R (1998) The choice of regulatory instruments in environmental policy—~~Harvard—Belfer Center for Science and International Affairs~~. *Harvard Environmental Law Review* 22(2): 313–367.
- Steffen W, Crutzen PJ, and McNeill JR (2007) The Anthropocene: Are humans now overwhelming the great forces of nature. *AMBIO: A Journal of the Human Environment* 36(8): 614–621.
- Steffen W, Grinevald J, Crutzen P, et al. (2011) The Anthropocene: conceptual and historical perspectives. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences* 369(1938): 842–867.
- Steffen W, Richardson K, Rockström J, et al. (2015) Planetary boundaries: Guiding human development on a changing planet. *Science* 347(6223): 1259855.
- Stephenson NL, Das AJ, Condit R, et al. (2014) Rate of tree carbon accumulation increases continuously with tree size. *Nature* 507: 90–93.
- Sterner TP (2002) *Policy Instruments for Environmental and Natural Resource Management*. Washington, DC; Stockholm, Sweden: Routledge.
- Stoneham G, Chaudhri V, Ha A, et al. (2003) Auctions for conservation contracts: an empirical examination of Victoria’s BushTender trial. *The Australian Journal of Agricultural and Resource Economics* 47(4): 477–500.
- Stoner A and Parker R (2013) *NSW Government announces changes to Native Vegetation Regulation*. Media release, Sydney: Office of Environment & Heritage.
- Suding KN (2011) Toward an era of restoration in ecology: Successes, failures, and opportunities ahead. *Annual Review of Ecology, Evolution, and Systematics* 42(1): 465–487.
- Sudol MF and Ambrose RF (2002) The US Clean Water Act and habitat replacement: evaluation of mitigation sites in Orange County, California, USA. *Environmental management* 30(5): 727–734.

- Sullivan S and Hannis M (2015) Nets and frames, losses and gains: Value struggles in engagements with biodiversity offsetting policy in England. *Ecosystem Services* 15: 162–173.
- Sutherland WJ, Pullin AS, Dolman PM, et al. (2004) The need for evidence-based conservation. *Trends in Ecology & Evolution* 19(6): 305–308.
- Sutherland WJ, Dicks, LV, Ockendon N, and Smith RK (2015) *What Works in Conservation 2015*. Cambridge: Open Book Publishers. - need to change in text citation, because Dicks was left out.
- Taherzadeh O and Howley P (2017) No net loss of what, for whom?: Stakeholder perspectives to biodiversity offsetting in England. *Environment, Development and Sustainability* 00. Available from: <http://link.springer.com/10.1007/s10668-017-9967-z> (accessed 17 June 2017).
- Tashakkori A and Teddlie C (2003) *Handbook of Mixed Methods in Social & Behavioral Research*. Thousand Oaks, CA: SAGE.
- Taylor MFJ (2013) *Bushland at risk of renewed clearing in Queensland*. Sydney: WWF-Australia.
- Taylor MFJ (2015) *Bushland destruction rapidly increasing in Queensland*. Sydney: WWF-Australia.
- Taylor MFJ and Dickman CR (2014) *NSW Native Vegetation Act saves Australian wildlife*. Sydney: WWF-Australia. Available from: http://www.researchgate.net/profile/Martin_Taylor4/publication/267155925_NSW_Native_Vegetation_Act_saves_Australian_wildlife/links/544ec1c50cf2bca5ce90bd1b.pdf (accessed 11 November 2015).
- Taylor MFJ, Sattler PS, Evans M, et al. (2011) What works for threatened species recovery? An empirical evaluation for Australia. *Biodiversity and Conservation* 20(4): 767–777.
- ten Kate K, Bishop J, and Bayon R (2004) *Biodiversity offsets: views, experience, and the business case*. Gland, Switzerland: IUCN--The World Conservation Union.
- The Senate Environment and Communications References Committee (2014) *Environmental offsets*. Canberra: Commonwealth of Australia.
- The Wilderness Society (2007) *Land Clearing in Queensland*. The Wilderness Society. Available from: <https://www.wilderness.org.au/land-clearing-queensland> (accessed 5 December 2015).
- The World Bank (2015) *World Development Indicators*. Available from: <http://data.worldbank.org/data-catalog/world-development-indicators> (accessed 30 September 2015).

- Thomas SC and Martin AR (2012) Carbon content of tree tissues: A synthesis. *Forests* 3(2): 332–352.
- Tietenberg TH (1990) Economic instruments for environmental regulation. *Oxford Review of Economic Policy* 6(1): 17–33.
- Tischew S, Baasch A, Conrad MK, et al. (2010) Evaluating restoration success of frequently implemented compensation measures: Results and demands for control procedures. *Restoration Ecology* 18(4): 467–480.
- Tlozek E (2015) Cattle grazing leases in Qld national parks unlikely to be renewed. *ABC News*, 26th April. Available from: <http://www.abc.net.au/news/2015-04-26/cattle-grazing-leases-in-qld-national-parks-no-renewal/6422408> (accessed 5 December 2015).
- Toms JD and Lesperance ML (2003) Piecewise regression: a tool for identifying ecological thresholds. *Ecology* 84(8): 2034–2041.
- Torabi N, Mata L, Gordon A, et al. (2016) The money or the trees: What drives landholders' participation in biodiverse carbon plantings? *Global Ecology and Conservation* 7: 1–11.
- Treweek J and ten Kate K (2014) Biodiversity Offsetting Outside the UK: an Overview. Paper Presented to the CIEEM Conference Biodiversity Offsetting: from Theory to Practice; March 18th, Birmingham, UK
- Trotter C, Tate K, Scott N, et al. (2005) Afforestation/reforestation of New Zealand marginal pasture lands by indigenous shrublands: the potential for Kyoto forest sinks. *Annals of Forest Science* 62(8): 865–871.
- Tucker CJ, Pinzon JE and Brown ME (2014) *Global Inventory Modeling and Mapping Studies*. College Park, Maryland: Global Land Cover Facility, University of Maryland. Available from: <https://nex.nasa.gov/nex/projects/1349/> (accessed 28 June 2017).
- Tulloch AIT, Barnes MD, Ringma J, et al. (2016) Understanding the importance of small patches of habitat for conservation. *Journal of Applied Ecology* 53(2): 418–429.
- UNEP-WCMC and IUCN, 2016. Protected Planet Report 2016. UNEP-WCMC and IUCN, Cambridge UK and Gland, Switzerland.
- U. S. Government Accountability Office (2005) Wetlands protection: Corps of engineers does not have an effective oversight approach to ensure that compensatory mitigation is occurring. (GAO-05-898). Available from: <http://www.gao.gov/products/GAO-05-898> (accessed 12 April 2017).

- U.S. Energy Information Administration (2015) *EIA Short-Term Energy Outlook*, November 2015.
- U.S. Fish & Wildlife Service (2003) *Guidance for the Establishment, Use, and Operation of Conservation Banks*. United States Department of the Interior.
- van Kooten GC (2000) Economic dynamics of tree planting for carbon uptake on marginal agricultural lands. *Canadian Journal of Agricultural Economics/Revue canadienne d'agroéconomie* 48(1): 51–65.
- van Oosterzee P, Blignaut J and Bradshaw CJ a. (2012) iREDD hedges against avoided deforestation's unholy trinity of leakage, permanence and additionality. *Conservation Letters* 5(4): 266–273.
- van Oosterzee P, Dale A and Preece ND (2014) Integrating agriculture and climate change mitigation at landscape scale: Implications from an Australian case study. *Global Environmental Change* 29: 306–317.
- Vedung E (2000) *Public policy and program evaluation*. New Brunswick, N.J.: Transaction Publishers.
- Venter O and Koh LP (2012) Reducing emissions from deforestation and forest degradation (REDD+): game changer or just another quick fix? *Annals of the New York Academy of Sciences*. 1249: 137-150. Available from: <http://www.ncbi.nlm.nih.gov/pubmed/22168380>.
- Venter O, Fuller RA, Segan DB, et al. (2014) Targeting global protected area expansion for imperiled biodiversity. *PLoS Biology* 12(6): e1001891.
- Vié J-C, Hilton-Taylor C, and Stuart SN (2009) *Wildlife in a changing world: an analysis of the 2008 IUCN Red List of Threatened Species*. Gland; Barcelona: IUCN; Lynx Edicions.
- Vitousek PM, Mooney HA, Lubchenco J, et al. (1997) Human domination of Earth's ecosystems. *Science* 277(5325): 494–499.
- Waldron A, Mooers AO, Miller DC, et al. (2013) Targeting global conservation funding to limit immediate biodiversity declines. *Proceedings of the National Academy of Sciences of the United States of America* 110(29): 12144–12148.
- Walker S, Brower AL, Stephens RTT, et al. (2009) Why bartering biodiversity fails. *Conservation Letters* 2(4): 149–157.
- Watson JEM, Evans MC, Carwardine J, et al. (2011) The capacity of Australia's protected-area system to represent threatened species. *Conservation Biology* 25(2): 324–332.
- Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D.J., Dokken, D.J. (Eds.), 2000. 2.2.3.3. Deforestation, in: *Land Use, Land-Use Change, and Forestry*:

-
- A Special Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK.
- Weimer DL and Vining AR (2010) *Policy Analysis: Concepts and Practice*. 5 edition. Boston: Routledge.
- Weir JT and Schluter D (2007) The latitudinal gradient in recent speciation and extinction rates of birds and mammals. *Science* 315(5818): 1574–1576.
- Weitzman ML (2001) Gamma discounting. *American Economic Review* 91(1): 260–271.
- White H (2009) Theory-based impact evaluation: principles and practice. *Journal of Development Effectiveness* 1(3): 271–284.
- Whitten SM, Coggan A, and Martin T (2012) What really matters in market design for biodiversity offset markets. In: *Conference on the Regulatory and Institutional Frameworks for Markets for Ecosystem Services*, University of Surrey, UK.
- Whitten SM, Wünscher T, and Shogren JF (2017) Conservation tenders in developed and developing countries - status quo, challenges and prospects. *Land Use Policy* 63: 552–560.
- Williams KJ, Ford A, Rosauer DF, et al. (2011) Forests of east Australia: The 35th biodiversity hotspot. In: Zachos FE and Habel JC (eds), *Biodiversity Hotspots*, Springer Berlin Heidelberg, 295–310. Available from: http://link.springer.com/chapter/10.1007/978-3-642-20992-5_16 (accessed 20 November 2015).
- Wilson KA, Meijaard E, Drummond S, et al. (2010) Conserving biodiversity in production landscapes. *Ecological Applications* 20(6): 1721–1732.
- Woinarski JCZ, Rankmore B, Hill B, et al. (2009) Fauna assemblages in regrowth vegetation in tropical open forests of the Northern Territory, Australia. *Wildlife Research* 36(8): 675–690.
- Woinarski JCZ, Armstrong M, Brennan K, et al. (2010) Monitoring indicates rapid and severe decline of native small mammals in Kakadu National Park, northern Australia. *Wildlife Research* 37: 116–126.
- Woinarski JCZ, Legge S, Fitzsimons JA, et al. (2011) The disappearing mammal fauna of northern Australia: context, cause, and response: Disappearing mammal fauna of north Australia. *Conservation Letters* 4(3): 192–201.
- Woinarski JCZ, Burbidge AA and Harrison PL (2015) Ongoing unraveling of a continental fauna: Decline and extinction of Australian mammals since European settlement. *Proceedings of the National Academy of Sciences of the United States of America* 112(15): 4531–4540.

- Woinarski JCZ, Garnett ST, Legge SM, et al. (2017) The contribution of policy, law, management, research, and advocacy failings to the recent extinctions of three Australian vertebrate species. *Conservation Biology* 31(1): 13–23.
- Wunder S (2007) The efficiency of payments for environmental services in tropical conservation. *Conservation Biology* 21(1): 48–58.
- Wunder S (2013) When payments for environmental services will work for conservation. *Conservation Letters* 6(4): 230–237.
- Wurzel, R., Jordan, A., Zito, A.R., 2013. *Environmental governance in Europe : a comparative analysis of new environmental policy instruments*. Edward Elgar Pub, Cheltenham, UK ; Northampton, MA.
- Wyborn, C., 2015. Co-productive governance: A relational framework for adaptive governance. *Global Environmental Change* 30, 56–67.
doi:10.1016/j.gloenvcha.2014.10.009
- WWF International (2015) Chapter 5: Saving Forests at Risk. In: *WWF Living Forests Report*, Geneva, Switzerland: WWF International.
- Xepapadeas AP (1991) Environmental policy under imperfect information: Incentives and moral hazard. *Journal of Environmental Economics and Management* 20(2): 113–126.
- Yin RK (2009) *Case Study Research: Design and Methods*. Thousand Oaks, CA: SAGE.
- Young MD and Gunningham N (1997) Mixing instruments and institutional arrangements for optimal biodiversity conservation. In: Hale P and Lamb (eds), *Conservation Outside Nature Reserves*, Centre for Conservation Biology, University of Queensland, 123–135. Available from:
<https://publications.csiro.au/rpr/pub?list=BRO&pid=procite:8ed8bc73-fa85-44c0-a6ca-b54bd5df7d7f> (accessed 24 March 2017).
- Young MD, Gunningham N, Elix J, et al. (1996) *Reimbursing the future: An evaluation of motivational, voluntary, price-based, property-right, and regulatory incentives for the conservation of biodiversity*. Biodiversity series, Paper no. 8, Canberra: Department of the Environment, Sport and Territories. Available from:
<http://www.environment.gov.au/archive/biodiversity/publications/series/paper9/index.html> (accessed 23 January 2017).
- Zak MR, Cabido M and Hodgson JG (2004) Do subtropical seasonal forests in the Gran Chaco, Argentina, have a future? *Biological Conservation* 120: 589–598.
- Zedler JB (1996) Ecological issues in wetland mitigation: An introduction to the forum. *Ecological Applications* 6(1): 33–37.

APPENDICES

Appendix One

Chapter Two Supplementary Material

Forest extent and change spatial data were derived from Landsat MSS, TM & ETM+ satellite imagery (Australian Department of the Environment, 2015) from 23 epochs as described in Table A.1 below.

Table A.1. Conversion deforestation and forest extent data from epochs to annual values

Epoch	Time between epochs (yrs)	Assigned to years:
1972	5	1972, 1973, 1974, 1975, 1976
1977	3	1977, 1978, 1979
1980	5	1980, 1981, 1982, 1983, 1984
1985	3	1985, 1986, 1987
1988 (early)	2	1988, 1989
1989 (end)	1	1990, 1991 (3 months)
1991 (early)	2	1991 (3 months), 1992 (3 months)
1992	3	1992 (9 months), 1993, 1994
1995	3	1995, 1996, 1997
1998	2	1998, 1999
2000	2	2000, 2001
2002	2	2002, 2003
2004	1	2004
2005	1	2005
2006	1	2006
2007	1	2007
2008	1	2008
2009	1	2009
2010	1	2010
2011	1	2011
2012	1	2012
2013	1	2013
2014	1	2014

Annual values from 1972 to 2004 were derived by averaging the deforestation estimate from an epoch over the number of years between it and the following epoch, as per advice from the Australian Department of the Environment (S Reddy, pers comm). Note that due to change in sensor from Landsat MSS and Landsat TM in 1988/89, satellite imagery was taken earlier than usual in 1988 and 1991, and later than usual in 1989 – hence this data had to be proportionally assigned to years 1990, 1991 and 1992. Further details are provided in Furby (2002).

Literature cited

Australian Department of the Environment. (2015). Human Induced Forest Extent & Change (version 11). Canberra.

Furby, S. (2002). Land cover change: specifications for remote sensing analysis. National Carbon Accounting System Technical Report No. 9. Commonwealth of Australia.

Appendix Two

Chapter Three Supplementary Material

Appendix 2.1 Additional details on regulatory policies for controlling deforestation in Queensland, 1988-2014

Table A2.1: Policy changes corresponding to Figure 3.2

Year	Policy name	Details
(a) 1995	<i>Land Act 1994</i>	Introduced to control native vegetation clearing on leasehold and State lands. Clearing on freehold still regulated by local governments under the <i>Local Government Act (LGA) 1993</i> and the <i>Planning and Environment Act 1990</i>
(b) 2000	<i>Vegetation Management Act 1999 (VMA)</i>	Enabled state-level regulation of native vegetation clearing on freehold land. Clearing of regrowth vegetation still allowed.
(c) 2003		A moratorium on tree clearing applications imposed by the Queensland Government in May 2003
(d) 2004	<i>Vegetation Management and Other Legislation Bill 2004</i>	Regulation of vegetation clearing on leasehold and State land removed from the <i>Land Act 1994</i> and placed under the VMA. A ballot for clearing permits totalling 500,000 hectares was held in September 2004. Provided \$150 million of financial assistance over five years
(e) 2006		All clearing permits issued under the ballot held in 2004 expired on 31 December 2006.
(f) 2009	<i>Vegetation Management and Other Legislation Amendment Act 2009</i>	VMA amended to protect 'high value regrowth' regrowth (vegetation not cleared since 31 December 1989)
(g) 2012		Review ordered into the enforcement of the VMA; all investigations into noncompliance suspended
(h) 2013	<i>Vegetation Management Regulation 2012 under the Vegetation Management Framework Amendment Act 2013</i>	Introduced a series of self-assessable codes for vegetation clearing, removed regulations on 'high value' regrowth clearing, introduced permitted clearing for necessary environmental clearing, high and irrigated high value agricultural clearing

Appendix 2.2. R code and output for covariate selection process

```

#final df
newdf<-read.csv("newdf4_new_Evans13June_qld.csv")
newdf <- data.frame(newdf[, -1])
newdf_ayy <- data.frame(newdf[, c(1,2,6)])

#qld Lgas
qldlgas<-read.csv("qlflgasforanalysis_13June.csv")

#####Climate covariates

lgaclimate<-read.csv("lgaClimate.csv")
lgadem<-read.csv("lga_dem_ndvi.csv")

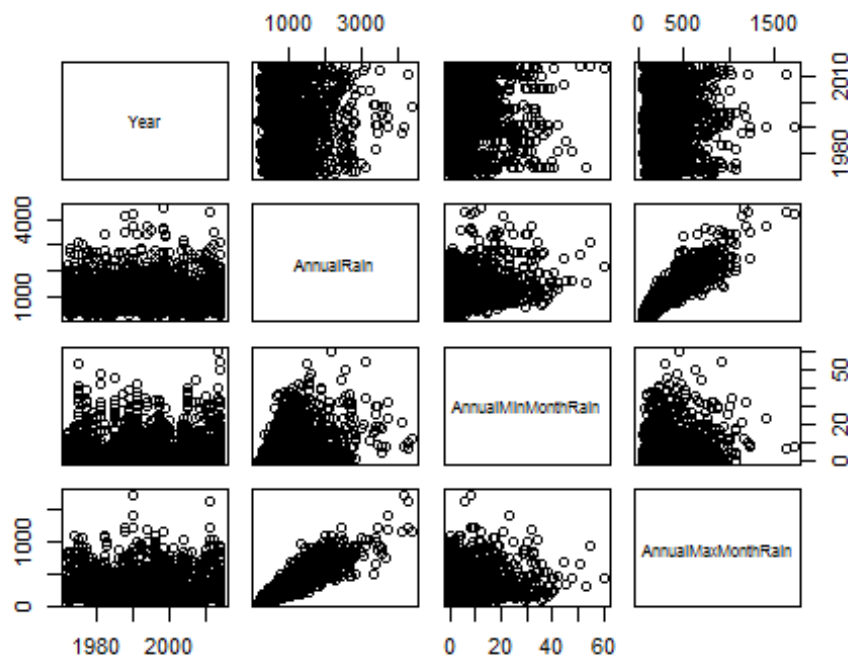
#select out the LGAs we're analysing
qldlgclimate<-merge(lgaclimate, qldlgas, by=intersect(names(lgaclimate),
names(qldlgas)))
qldlgadem<-merge(lgadem, qldlgas, by.x="LGA_CODE11", by.y="LGA_CODE")

#add ayy (response variable y[it])
ayyqldlgclimate<-merge(qldlgclimate, newdf_ayy, by=intersect(names(lgaclimate),
names(newdf_ayy)))
ayyqldlgadem<-merge(qldlgadem, newdf_ayy, by.x="LGA_CODE11", by.y="LGA_CODE")

#Log transform climate variables
ayyqldlgclimate$logAnnualMinMonthRain<-log10(ayyqldlgclimate$AnnualMinMonthRain+1)
ayyqldlgclimate$logAnnualMaxMonthRain<-log10(ayyqldlgclimate$AnnualMaxMonthRain+1)
ayyqldlgclimate$logAnnualRain<-log10(ayyqldlgclimate$AnnualRain+1)
ayyqldlgclimate$logVap<-log10(ayyqldlgclimate$AnnualMeanVpd09)

####Scatterplots
## Untransformed rainfall and vapour pressure
plot(qldlgclimate[, c(2:5)])

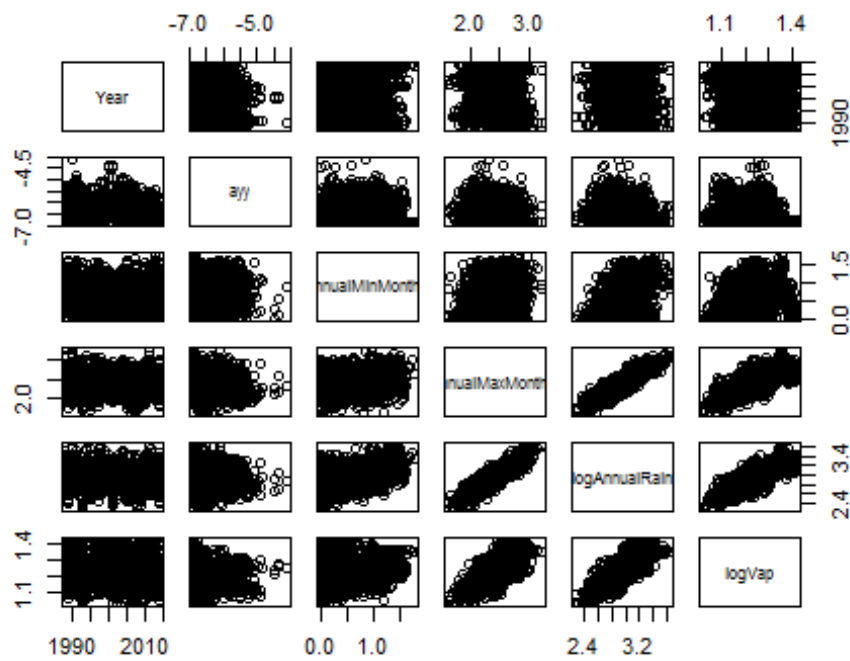
```



```
cor(qldlgaclimate[, c(2:5)])
```

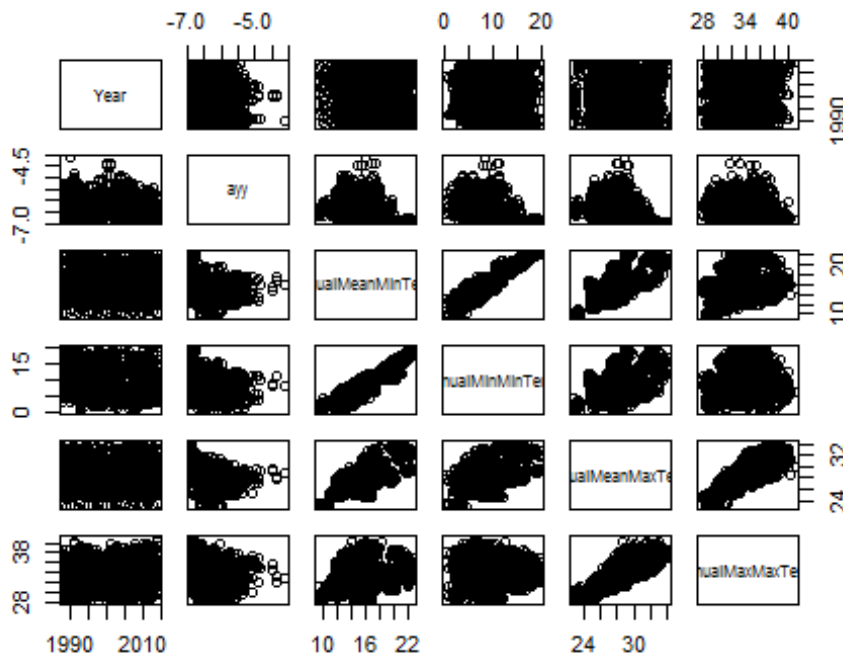
```
##              Year   AnnualRain AnnualMinMonthRain
## Year          1.000000000 -0.004828387      0.09340903
## AnnualRain    -0.004828387  1.000000000      0.3668961
## AnnualMinMonthRain 0.093409030 0.36689608      1.00000000
## AnnualMaxMonthRain -0.041190610 0.889617485      0.18610639
##              AnnualMaxMonthRain
## Year          -0.04119061
## AnnualRain      0.88961748
## AnnualMinMonthRain 0.18610639
## AnnualMaxMonthRain 1.00000000
```

```
#ayy, year and log transformed rainfall and vapour pressure
plot(ayyqldlgaclimate[, c(2,13:17)])
```



#All rain and vapour variables pertain to moisture, and none stands out
#as an obvious choice. WE CHOOSE VAPOUR PRESSURE.

```
##Temperature
## Untransformed temperature
plot(ayyqldlgaclimate[, c(2,13, 7:10)])
```



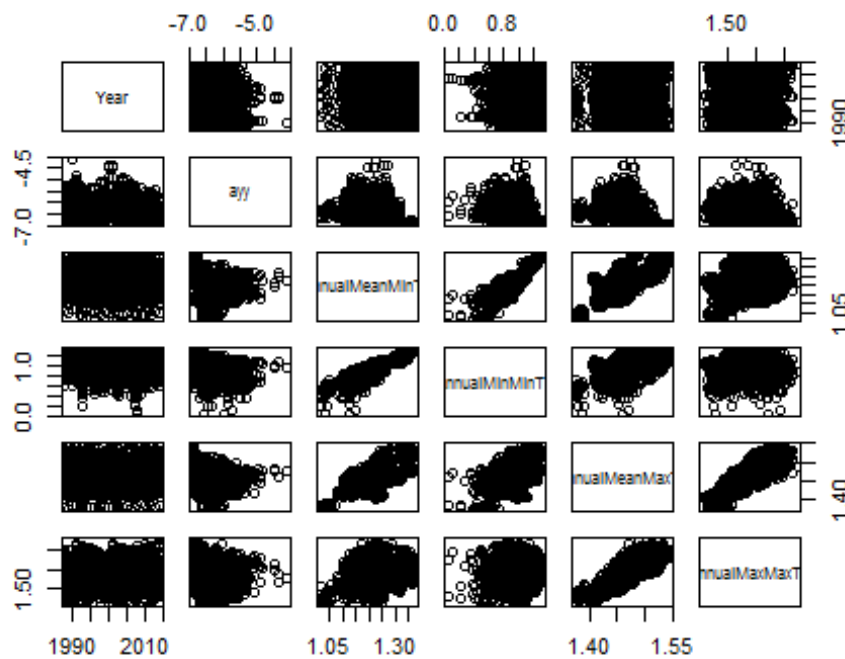
```
cor(ayyqldlgacclimate[, c(2,13, 7:10)])
```

```
##              Year          ayy AnnualMeanMinTemp
## Year          1.00000000 -0.1841749      -0.02188517
## ayy           -0.18417486  1.00000000      -0.40106774
## AnnualMeanMinTemp -0.02188517 -0.4010677      1.00000000
## AnnualMinMinTemp  -0.04240353 -0.3730308      0.94286401
## AnnualMeanMaxTemp  0.02034789 -0.3887173      0.70818730
## AnnualMaxMaxTemp   0.07129730 -0.2006276      0.28036365
##
## AnnualMinMinTemp AnnualMeanMaxTemp AnnualMaxMaxTemp
## Year          -0.04240353      0.02034789      0.07129730
## ayy           -0.37303079     -0.38871727     -0.20062755
## AnnualMeanMinTemp  0.94286401      0.70818730      0.28036365
## AnnualMinMinTemp   1.00000000      0.53479279      0.07170248
## AnnualMeanMaxTemp   0.53479279      1.00000000      0.82098644
## AnnualMaxMaxTemp    0.07170248      0.82098644      1.00000000
```

```
#Log transform
```

```
ayyqldlgacclimate$logAnnualMeanMinTemp<-log10(ayyqldlgacclimate$AnnualMeanMinTemp+1)
ayyqldlgacclimate$logAnnualMinMinTemp<-log10(ayyqldlgacclimate$AnnualMinMinTemp+1)
ayyqldlgacclimate$logAnnualMeanMaxTemp <-log10(ayyqldlgacclimate$AnnualMeanMaxTemp+1)
ayyqldlgacclimate$logAnnualMaxMaxTemp<-log10(ayyqldlgacclimate$AnnualMaxMaxTemp+1)
```

```
##ayy, year and log transformed temperature
plot(ayyqldlgacclimate[, c(2,13, 18:21)])
```



```
cor(ayyqldlgaclimate[, c(2,13, 18:21)])
```

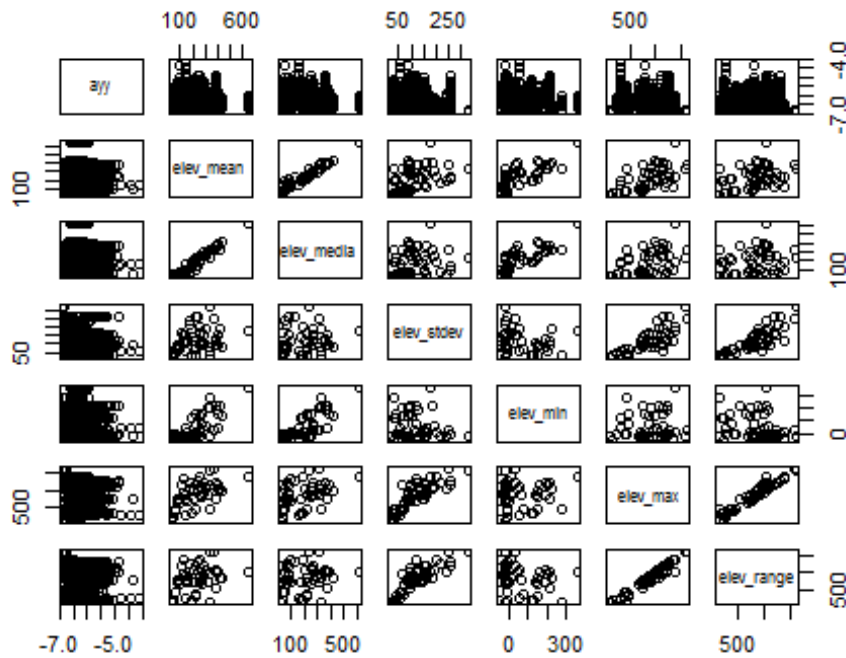
```
##              Year          ayy logAnnualMeanMinTemp
## Year          1.00000000 -0.1841749        -0.02394022
## ayy          -0.18417486  1.00000000        -0.36793368
## logAnnualMeanMinTemp -0.02394022 -0.3679337         1.00000000
## logAnnualMinMinTemp -0.05636583 -0.3033580         0.90757342
## logAnnualMeanMaxTemp  0.02121855 -0.3730941         0.70875948
## logAnnualMaxMaxTemp  0.07112957 -0.1945425         0.29200122
##              logAnnualMinMinTemp logAnnualMeanMaxTemp
## Year          -0.05636583         0.02121855
## ayy          -0.30335801        -0.37309411
## logAnnualMeanMinTemp  0.90757342         0.70875948
## logAnnualMinMinTemp  1.00000000         0.49771370
## logAnnualMeanMaxTemp  0.49771370         1.00000000
## logAnnualMaxMaxTemp  0.04923780         0.83031241
##              logAnnualMaxMaxTemp
## Year          0.07112957
## ayy          -0.19454247
## logAnnualMeanMinTemp  0.29200122
## logAnnualMinMinTemp  0.04923780
## logAnnualMeanMaxTemp  0.83031241
## logAnnualMaxMaxTemp  1.00000000
```

#None of temp variables stands out as an obvious choice.

#WE CHOOSE MEANMAXTEMP (highest corr with response ayy).

#####Biophysical covariates

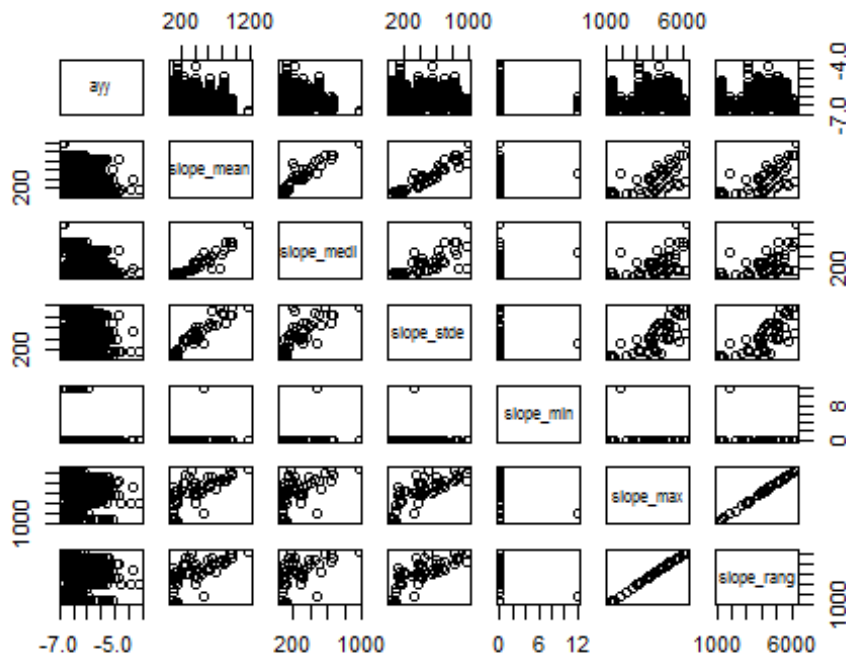
```
##Untransformed elevation
plot(ayyqldlgadem[,c(21,5:10)])
```



```
cor(ayyqldlgadem[,c(21,5:10)])
```

```
##          ayy elev_mean elev_media elev_stdev elev_min
## ayy      1.00000000 0.02855343 0.05356424 -0.07107317 0.07944516
## elev_mean 0.02855343 1.00000000 0.96746196 0.35055171 0.77298668
## elev_media 0.05356424 0.96746196 1.00000000 0.12652407 0.82481262
## elev_stdev -0.07107317 0.35055171 0.12652407 1.00000000 -0.16233502
## elev_min   0.07944516 0.77298668 0.82481262 -0.16233502 1.00000000
## elev_max   0.07153512 0.59647558 0.44577600 0.80609237 0.11493058
## elev_range 0.05087606 0.39582254 0.23235147 0.84490271 -0.14230368
##          elev_max elev_range
## ayy      0.07153512 0.05087606
## elev_mean 0.59647558 0.39582254
## elev_media 0.44577600 0.23235147
## elev_stdev 0.80609237 0.84490271
## elev_min   0.11493058 -0.14230368
## elev_max   1.00000000 0.96690896
## elev_range 0.96690896 1.00000000
```

```
#Untransformed slope
plot(ayyqldlgadem[,c(21,11:16)])
```



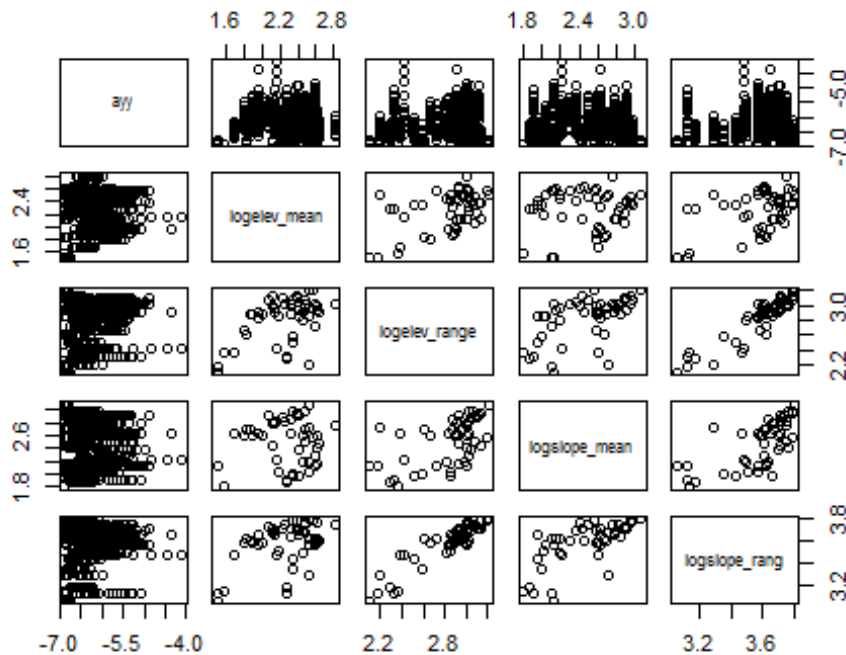
```
cor(ayyqldlgadem[,c(21,11:16)])
```

```
##              ayy  slope_mean  slope_medi  slope_stde  slope_min
## ayy          1.000000000 -0.03058673 -0.04887301 -0.001940041 -0.0768446
## slope_mean -0.030586728  1.000000000  0.92959860  0.949089978  0.0587080
## slope_medi -0.048873013  0.92959860  1.000000000  0.776482499  0.1558698
## slope_stde -0.001940041  0.94908998  0.77648250  1.000000000 -0.0586083
## slope_min  -0.076844608  0.05870802  0.15586982 -0.058608371  1.0000000
## slope_max   0.034588532  0.66269723  0.52211208  0.772201198 -0.2319268
## slope_rang  0.034672973  0.66243607  0.52177180  0.772052776 -0.2330879
##              slope_max  slope_rang
## ayy          0.03458853  0.03467297
## slope_mean   0.66269723  0.66243607
## slope_medi   0.52211208  0.52177180
## slope_stde   0.77220120  0.77205278
## slope_min   -0.23192687 -0.23308799
## slope_max    1.00000000  0.99999929
## slope_rang   0.99999929  1.00000000
```

##All elevation variables are highly correlated with each other,
##and same with slope variables, so select mean and range for now

```
#Log transform slope and elev mean and range
ayyqldlgadem$logelev_mean<-log10(ayyqldlgadem$elev_mean+1)
ayyqldlgadem$logelev_range<-log10(ayyqldlgadem$elev_range+1)
ayyqldlgadem$logslope_mean <-log10(ayyqldlgadem$slope_mean+1)
ayyqldlgadem$logslope_rang<-log10(ayyqldlgadem$slope_rang+1)

plot(ayyqldlgadem[,c(21:25)])
```



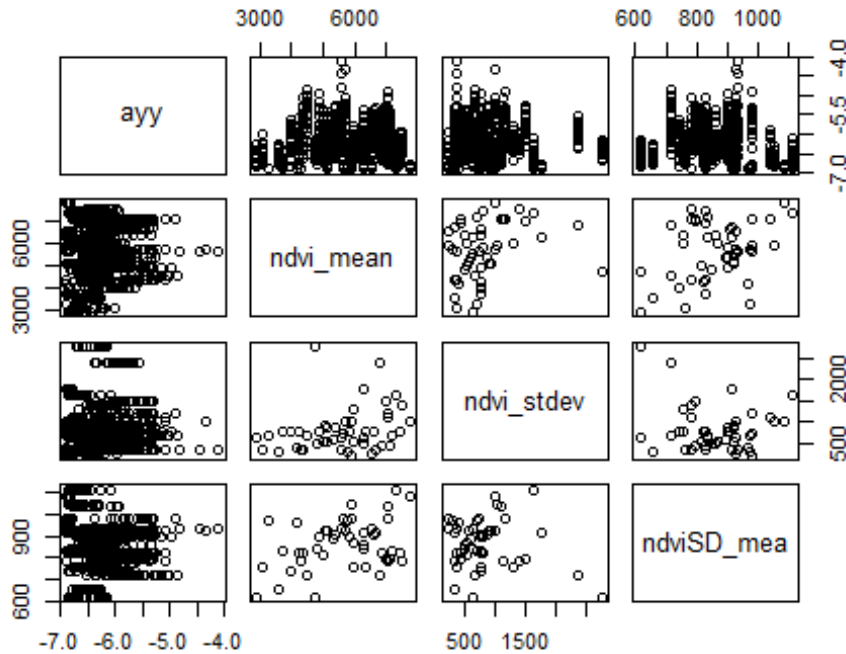
```
cor(ayyqldlgadem[,c(21:25)])

##               ayy logelev_mean logelev_range logslope_mean
## ayy           1.00000000  0.09355166  0.1390674  0.06358031
## logelev_mean  0.09355166  1.00000000  0.5245611  0.07242372
## logelev_range 0.13906743  0.52456115  1.0000000  0.54859437
## logslope_mean 0.06358031  0.07242372  0.5485944  1.00000000
## logslope_rang 0.11090982  0.48167175  0.9197467  0.67160320
##               logslope_rang
## ayy           0.1109098
## logelev_mean  0.4816717
## logelev_range 0.9197467
## logslope_mean 0.6716032
## logslope_rang 1.0000000
```

```
##None stands out as obvious choice, so
##CHOOSE JUST ELEV_MEAN for interpretability
```



```
##NDVI
#Look at ayy vs ndvi
plot(ayyqldlgadem[, c(21, 17:19)])
```

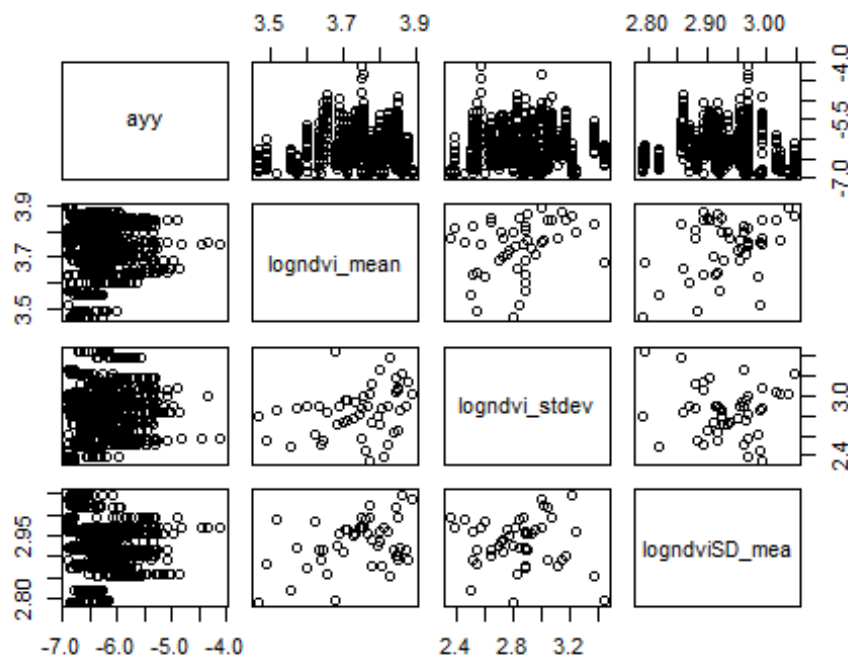


```
cor(ayyqldlgadem[, c(21, 17:19)])

##              ayy ndvi_mean ndvi_stdev ndviSD_mea
## ayy          1.00000000 0.1098183  0.06964855 -0.1898965
## ndvi_mean    0.10981828 1.00000000  0.27468511  0.3399635
## ndvi_stdev   0.06964855 0.2746851  1.00000000 -0.1384215
## ndviSD_mea  -0.18989645 0.3399635 -0.13842150  1.0000000

#Log transform
ayyqldlgadem$logndvi_mean<-log10(ayyqldlgadem$ndvi_mean+1)
ayyqldlgadem$logndvi_stdev <-log10(ayyqldlgadem$ndvi_stdev+1)
ayyqldlgadem$logndviSD_mea<-log10(ayyqldlgadem$ndviSD_mea+1)

##ayy, year and log transformed NDVI
plot(ayyqldlgadem[, c(21, 26:28)])
```



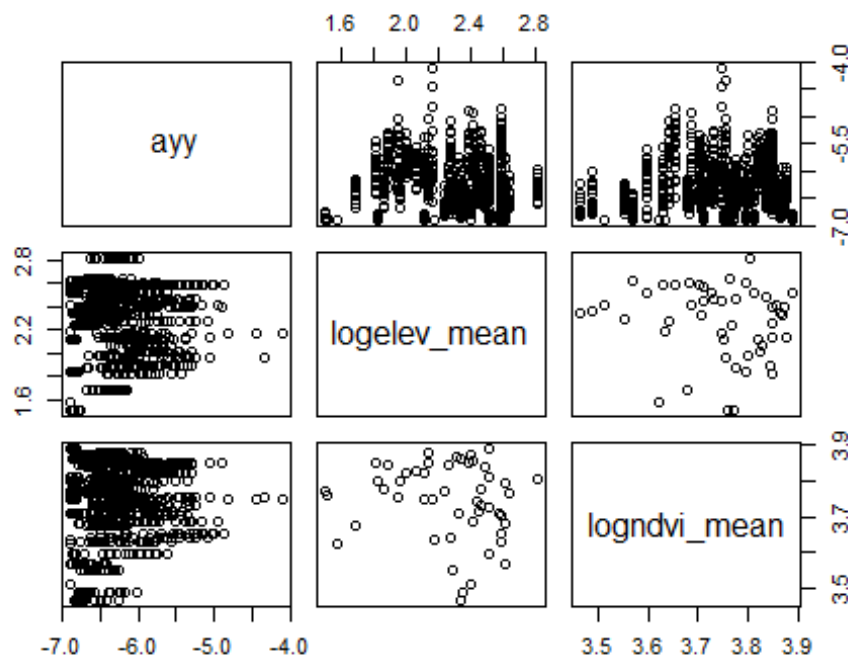
```
cor(ayyqldlgadem[, c(21, 26:28)])

##              ayy logndvi_mean logndvi_stdev logndviSD_mea
## ayy           1.0000000    0.1363419    0.1059620   -0.1691899
## logndvi_mean  0.1363419    1.0000000    0.2923990    0.3763235
## logndvi_stdev 0.1059620    0.2923990    1.0000000   -0.0794819
## logndviSD_mea -0.1691899    0.3763235   -0.0794819    1.0000000

##CHOOSE NDIV_MEAN for interpretability

#Examine selected climate and biophysical variables together
select_dem<-ayyqldlgadem[, c(1,21,22,26)]
select_clim<-ayyqldlgacclimate[, c(1,2,13,17,20)]

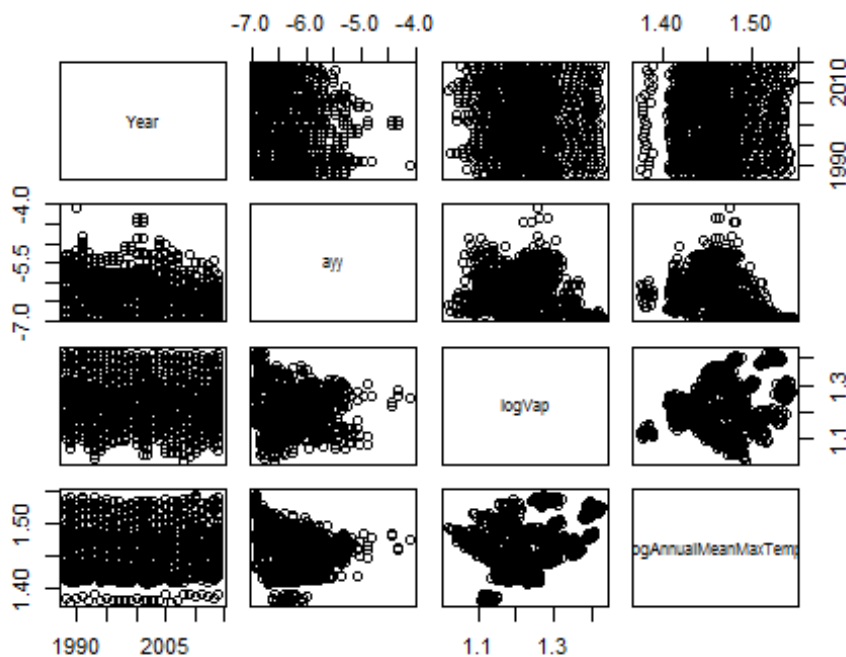
plot(select_dem[-1])
```



```
cor(select_dem[-1])

##               ayy logelev_mean logndvi_mean
## ayy           1.00000000  0.09355166   0.1363419
## logelev_mean  0.09355166  1.00000000  -0.1421020
## logndvi_mean  0.13634188 -0.14210203  1.0000000

plot(select_clim[ ,2:5])
```



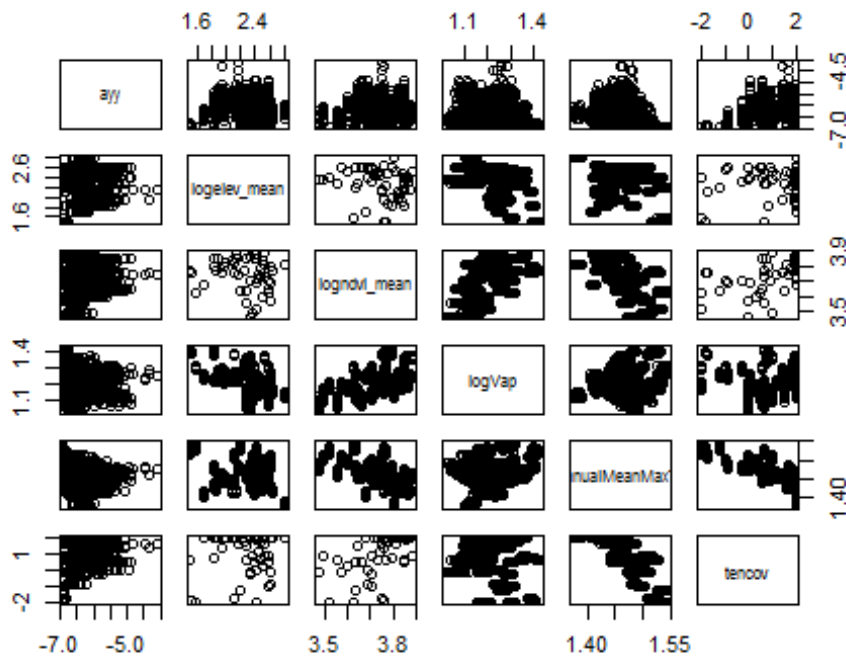
```
cor(select_clim[,2:5])

##              Year      ayy      logVap
## Year          1.00000000 -0.1841749 -0.03429284
## ayy          -0.18417486  1.00000000 -0.29181861
## logVap       -0.03429284 -0.2918186  1.00000000
## logAnnualMeanMaxTemp  0.02121855 -0.3730941  0.29744465
##              logAnnualMeanMaxTemp
## Year              0.02121855
## ayy              -0.37309411
## logVap           0.29744465
## logAnnualMeanMaxTemp  1.00000000

#merge
select_newdf<-merge(select_dem[-2], select_clim, by.x="LGA_CODE11", by.y="
LGA_CODE")

###Land tenure
tenure<-read.csv("proptenure_tencov_14-6-17_qldonly.csv")
tenure<-tenure[,c(2,6)]
select_newdf<-merge(select_newdf, tenure, by.x="LGA_CODE11", by.y="LGA_COD
E")

#Plot with Land tenure
plot(select_newdf[,c(5,2,3,6,7,8)])
```



```
cor(select_newdf[,c(5,2,3,6,7,8)])

##              ayy logelev_mean logndvi_mean      logVap
## ayy              1.00000000  0.09355166   0.1363419 -0.2918186
## logelev_mean      0.09355166  1.00000000  -0.1421020 -0.5563906
## logndvi_mean      0.13634188 -0.14210203   1.0000000  0.4686681
## logVap            -0.29181861 -0.55639061   0.4686681  1.0000000
## logAnnualMeanMaxTemp -0.37309411 -0.20994323 -0.6007638  0.2974447
## tencov             0.40014492  0.09059084   0.5522605 -0.2487662
##              logAnnualMeanMaxTemp      tencov
## ayy              -0.3730941  0.40014492
## logelev_mean      -0.2099432  0.09059084
## logndvi_mean      -0.6007638  0.55226053
## logVap             0.2974447 -0.24876616
## logAnnualMeanMaxTemp  1.0000000 -0.84803929
## tencov            -0.8480393  1.00000000
```

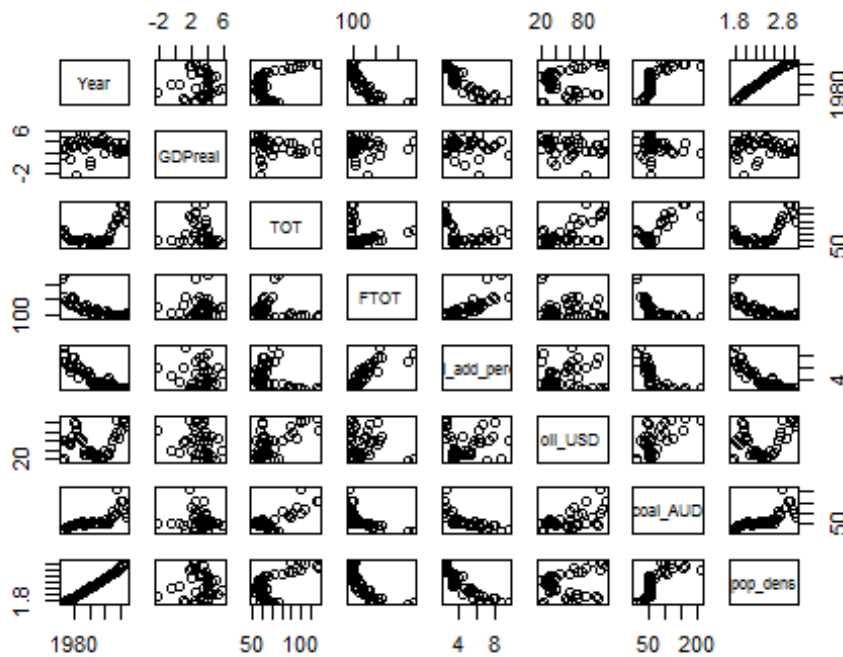
##with VAPOUR in the mix, NDVI doesn't provide additional information

##Macroeconomic covariates

```
macrodf<-read.csv("macro_df_march2016.csv")
```

##Broad macro variables

```
plot(macrodf[,c(1,3,5,6,17,19,20,21)])
```



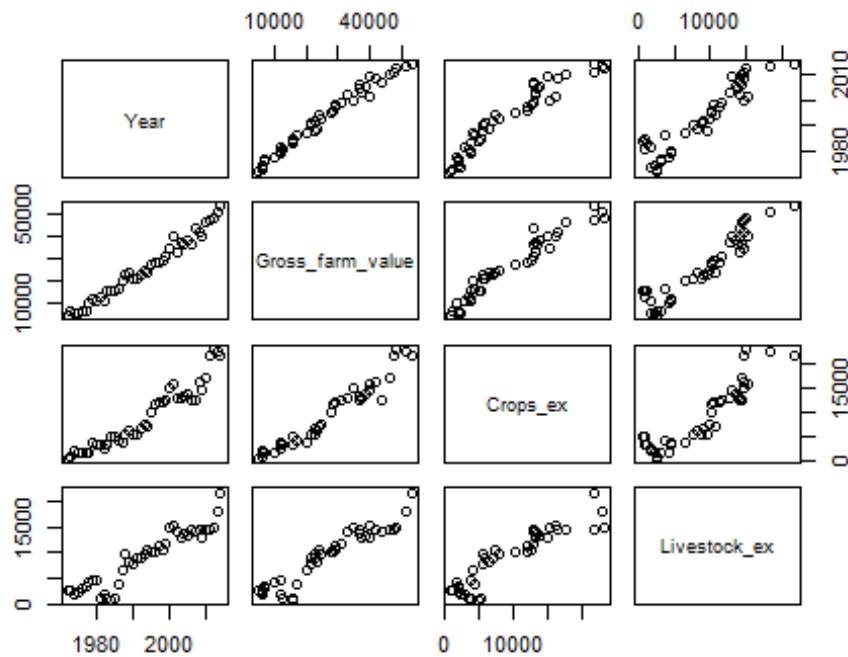
```
cor(macrodF[,c(1,3,5,6,17,19,20,21)])
```

	Year	GDPreal	TOT	FTOT	Ag_val_add_percGDP	oil_USD	coal_AUD	pop_dens
Year	1.0000000	0.022366065	0.6231801	-0.80163985	-0.90649169	0.2702941	0.7618196	0.99617526
GDPreal	0.02236607	1.000000000	-0.1514287	0.02221356	-0.07994097	-0.2687156	-0.1359170	-0.00418417
TOT	0.62318014	-0.151428718	1.0000000	-0.23000222	-0.42941453	0.6456314	0.7863692	0.66865623
FTOT	-0.80163985	0.022213560	-0.2300022	1.00000000	0.83579318	-0.1660492	-0.6139455	-0.77176732
Ag_val_add_percGDP	-0.90649169	-0.079940970	-0.4294145	0.83579318	1.00000000	-0.0768725	-0.6750377	-0.89001390
oil_USD	0.27029412	-0.268715619	0.6456314	-0.16604924	-0.07687250	1.0000000	0.5844126	0.31272952
coal_AUD	0.76181960	-0.135916955	0.7863692	-0.61394550	-0.67503773	0.5844126	1.0000000	0.77074410
pop_dens	0.99617527	-0.004184176	0.6686562	-0.77176732	-0.89001390	0.3127295	0.7707441	1.00000000

```
#CHOOSE GDPREAL, OILUSD, % AG to GDP TO MINIMIZE CONFOUNDING WITH YEAR
```

```
##Farm variables
```

```
plot(macrodf[,c(1,8,10,11)])
```



```
cor(macrodf[,c(1,8,10,11)])
```

```
##
## Year      1.0000000  0.9891603  0.9566678  0.9335221
## Gross_farm_value 0.9891603  1.0000000  0.9705191  0.9436573
## Crops_ex    0.9566678  0.9705191  1.0000000  0.9146534
## Livestock_ex 0.9335221  0.9436573  0.9146534  1.0000000
```

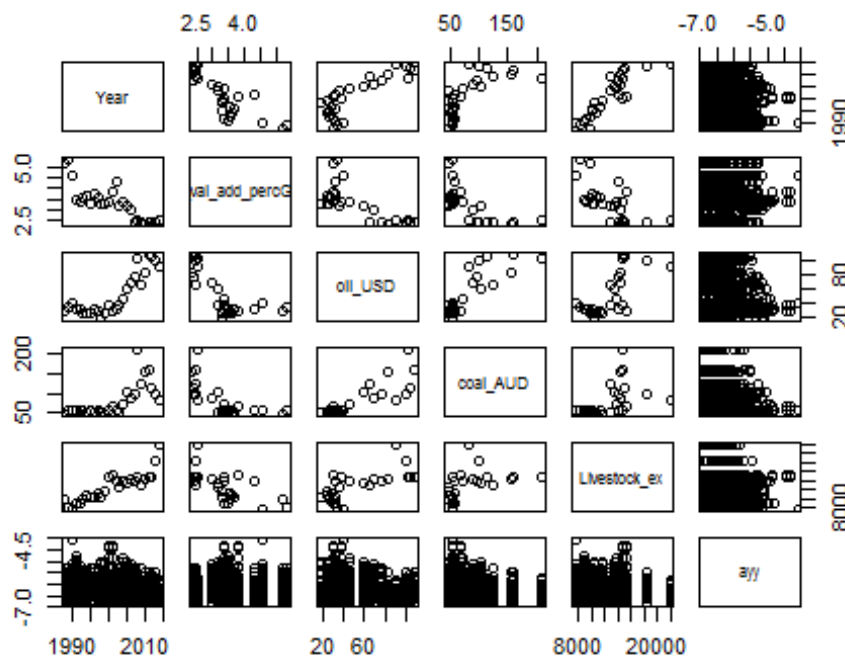
```
#Check Livestock exports
```

```
select_macro<-macrodf[,c(1,17,19,20,11)]
```

```
##add ayy
```

```
select_macro_ayy<-merge(select_macro, newdf_ayy , by=intersect(names(select_macro), names(newdf_ayy)))
```

```
plot(select_macro_ayy[-6])
```

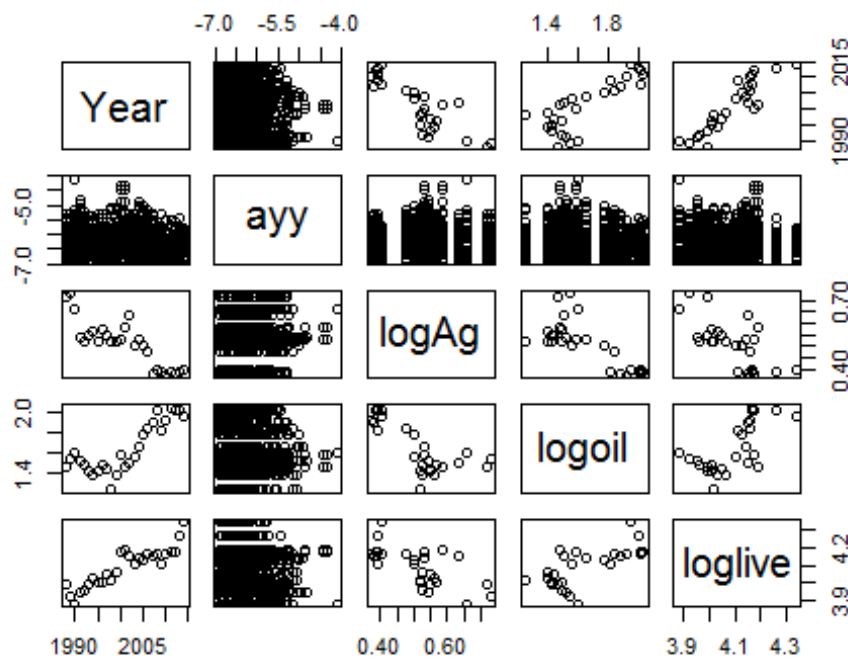


```
cor(select_macro_ayy[-6])
```

```
##              Year Ag_val_add_percGDP  oil_USD  coal_AUD
## Year          1.0000000      -0.8293058  0.8369721  0.6962969
## Ag_val_add_percGDP -0.8293058        1.0000000 -0.7003019 -0.6440887
## oil_USD           0.8369721      -0.7003019  1.0000000  0.8435394
## coal_AUD          0.6962969      -0.6440887  0.8435394  1.0000000
## Livestock_ex       0.8820347      -0.6339391  0.6918659  0.4672437
## ayy              -0.1841749       0.1624686 -0.1601019 -0.1589144
##              Livestock_ex      ayy
## Year          0.8820347 -0.1841749
## Ag_val_add_percGDP -0.6339391  0.1624686
## oil_USD          0.6918659 -0.1601019
## coal_AUD          0.4672437 -0.1589144
## Livestock_ex      1.0000000 -0.1468161
## ayy              -0.1468161  1.0000000
```

```
select_macro_ayy$logAg<-log10(select_macro_ayy$Ag_val_add_percGDP)
select_macro_ayy$logoil<-log10(select_macro_ayy$oil_USD)
select_macro_ayy$loglive<-log10(select_macro_ayy$Livestock_ex)
```

```
plot(select_macro_ayy[c(1,7:10)])
```

```
cor(select_macro_ayy[c(1,7:10)])
```

	Year	ayy	logAg	logoil	loglive
Year	1.0000000	-0.1841749	-0.8596448	0.8230612	0.9061581
ayy	-0.1841749	1.0000000	0.1740008	-0.1386798	-0.1520259
logAg	-0.8596448	0.1740008	1.0000000	-0.7395789	-0.6865213
logoil	0.8230612	-0.1386798	-0.7395789	1.0000000	0.6733363
loglive	0.9061581	-0.1520259	-0.6865213	0.6733363	1.0000000

```
##remove coal (too similar to oil) and livestock (too similar to year)

select_macro_ayy<-select_macro_ayy[,c(-6,-7)]

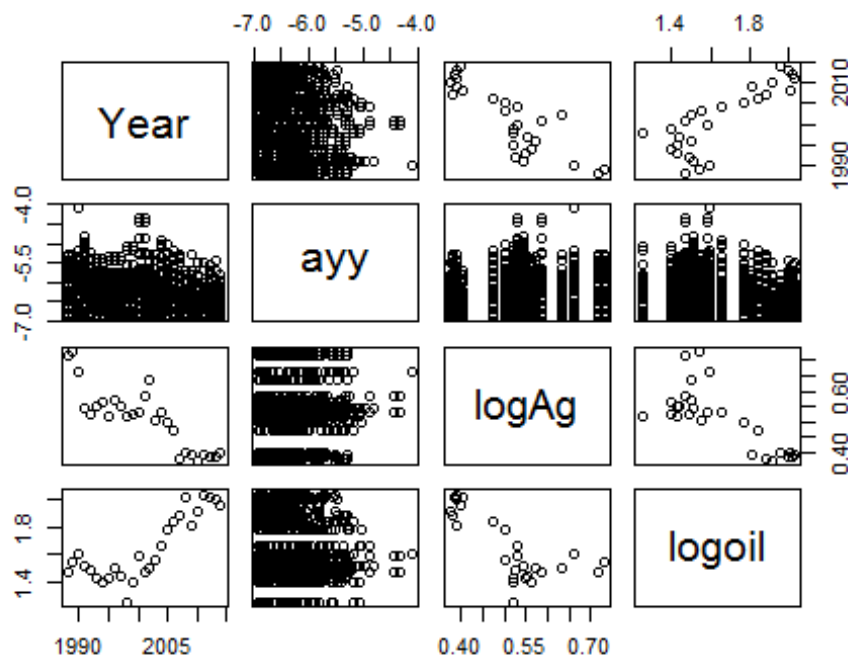
#####Combine final covariates

select_macro$logAg<-log10(select_macro$Ag_val_add_percGDP)
select_macro$logoil<-log10(select_macro$oil_USD)
select_macro$loglive<-log10(select_macro$Livestock_ex)

finaldf_ayy<-merge(select_macro, select_newdf,
                    by=intersect(names(select_macro), names(select_newdf)))

finaldf_ayy<-finaldf_ayy[,c(-2,-3,-4,-5)]

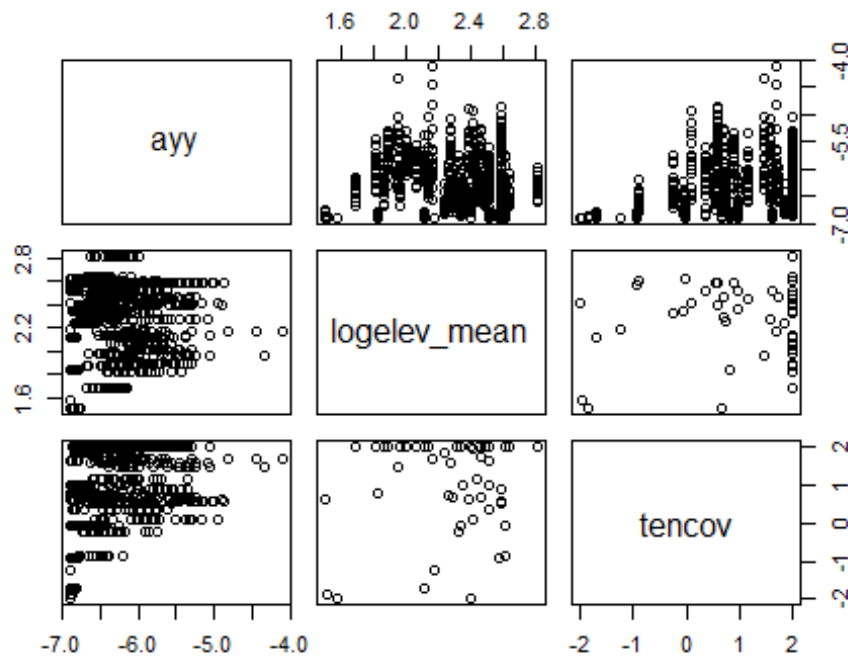
#Macro
plot(finaldf_ayy[,c(1,8,2,3)])
```



```
cor(finaldf_ayy[,c(1,8,2,3)])

##           Year           ayy          logAg          logoil
## Year      1.0000000 -0.1841749 -0.8596448  0.8230612
## ayy      -0.1841749  1.0000000  0.1740008 -0.1386798
## logAg    -0.8596448  0.1740008  1.0000000 -0.7395789
## logoil    0.8230612 -0.1386798 -0.7395789  1.0000000
```

```
#Geographical (biophysical and institutional)
plot(finaldf_ayy[,c(8,6,11)])
```

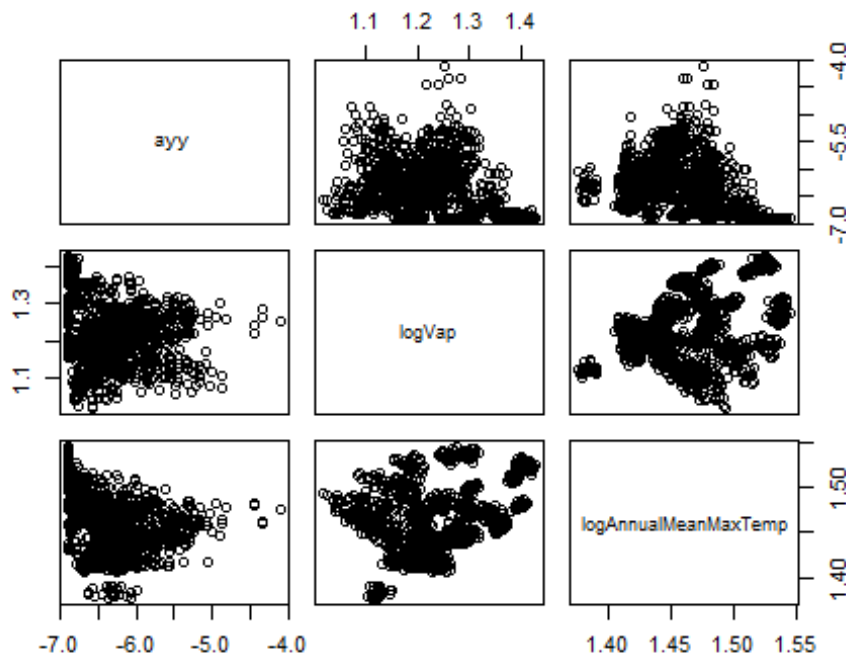


```
cor(finaldf_ayy[,c(1,8,6,11)])
```

```
##           Year           ayy logelev_mean           tencov
## Year      1.000000e+00 -0.18417486  0.00000000 -3.449718e-21
## ayy      -1.841749e-01  1.00000000  0.09355166  4.001449e-01
## logelev_mean 0.000000e+00  0.09355166  1.00000000  9.059084e-02
## tencov    -3.449718e-21  0.40014492  0.09059084  1.000000e+00
```

```
#Climate
```

```
plot(finaldf_ayy[,c(8,9,10)])
```



```
cor(finaldf_ayy[,c(1,8,9,10)])
```

```
##           Year      ayy      logVap
## Year      1.00000000 -0.1841749 -0.03429284
## ayy      -0.18417486  1.00000000 -0.29181861
## logVap   -0.03429284 -0.2918186  1.00000000
## logAnnualMeanMaxTemp 0.02121855 -0.3730941 0.29744465
##
## logAnnualMeanMaxTemp
## Year      0.02121855
## ayy      -0.37309411
## logVap    0.29744465
## logAnnualMeanMaxTemp 1.00000000
```

Appendix 2.3: Local government areas

Table A2.2: Local government areas in Queensland

LGA Code	LGA Name	Area of LGA (ha)	% forest in 1972	Included in analysis	Li
30250	Aurukun	734,721	63.3	y	1.85
30300	Balonne	3,110,618	27.7	y	1.56
30370	Banana	2,854,631	38.4	y	1.14
30410	Barcaldine	5,352,067	14.1	y	0.34
30450	Barcoo	6,182,501	0.5	-	-
30760	Blackall Tambo	3,038,906	28.7	y	0.53
30900	Boulia	6,095,581	-	-	-
31000	Brisbane	133,809	45.8	y	2
31750	Bulloo	7,376,280	0.5	-	-
31820	Bundaberg	643,564	53.7	y	1.99
31900	Burdekin	504,342	36.8	-	-
31950	Burke	4,003,921	4	-	-
32070	Cairns	411,511	70	y	0.97
32250	Carpentaria	6,412,490	4.8	y	1.97
32260	Cassowary Coast	468,499	62.2	y	1.99
32270	Central Highlands	5,983,489	41.9	y	0.68
32310	Charters Towers	6,837,355	48.5	y	0.89
32330	Cherbourg	3,160	65.9	y	1.63
32450	Cloncurry	4,798,313	0.6	-	-
32500	Cook	#####	61.6	y	1.71
32600	Croydon	2,948,710	6.3	y	1.24
32750	Diamantina	9,466,685	-	-	-
32770	Doomadgee	183,516	2.5	-	-
33100	Etheridge	3,920,102	28.1	-	-
33200	Flinders	4,119,266	14.9	y	0.04
33220	Fraser Coast	710,250	64.7	y	1.99
33360	Gladstone	1,046,579	47.4	y	2
33430	Gold Coast	133,168	47	y	2
33610	Goondiwindi	1,925,549	24.4	y	2
33620	Gympie	688,454	51	y	1.99
33800	Hinchinbrook	280,135	52.9	y	0.72
33830	Hope Vale	110,475	58.5	y	0.81
33960	Ipswich	108,849	30	y	2
33980	Isaac	5,871,984	37.8	y	0.07
34420	Kowanyama	254,317	14.1	-	-
34570	Lockhart River	357,806	75.5	-	-
34580	Lockyer Valley	226,874	44.7	y	2
34590	Logan	95,810	51.2	y	2
34710	Longreach	4,057,171	3.6	y	0.97

34770	Mackay	760,120	56.7	y	0.6
34800	McKinlay	4,073,371	0.1	-	-
34830	Mapoon	54,799	61.3	-	-
34860	Maranoa	5,871,136	42.4	y	0.89
35010	Moreton Bay	203,330	58.5	y	2
35250	Mornington	124,421	36.7	-	-
35300	Mount Isa	4,318,804	0.9	-	-
35600	Murweh	4,069,849	51	y	0.59
35670	Napranum	199,791	62.4	y	0.64
35760	North Burnett	1,966,677	41.6	y	1.74
35780	Northern Peninsula Area	105,691	59.3	-	-
35790	Palm Island	7,063	43.3	-	-
35800	Paroo	4,761,641	17.3	y	0.69
36070	Pormpuraaw	442,884	27.7	-	-
36150	Quilpie	6,742,331	11.3	y	0.06
36250	Redland	53,625	51.1	y	2
36300	Richmond	2,658,022	3	y	-2
36360	Rockhampton	1,831,174	44.6	y	1.47
36510	Scenic Rim	424,807	26.9	y	2
36580	Somerset	537,328	40.7	y	1.98
36630	South Burnett	838,170	28.8	y	2
36660	Southern Downs	711,172	31.4	y	2
36710	Sunshine Coast	312,071	56.2	y	2
36810	Tablelands	6,479,381	36.4	y	0.92
36910	Toowoomba	1,295,773	23	y	2
36950	Torres	88,271	38.9	-	-
36960	Torres Strait Island	48,924	29.7	-	-
37010	Townsville	372,738	60.7	y	1.85
37300	Weipa	1,082	11.2	-	-
37310	Western Downs	3,793,849	30.5	y	2
37340	Whitsunday	2,380,438	37.2	y	0.25
37400	Winton	5,381,439	-	-	-
37550	Woorabinda	39,028	50	y	1.69
37570	Wujal Wujal	1,118	80.8	-	-
37600	Yarrabah	15,884	78.7	-	-

Appendix 2.4: Prior distributions and model variations

We specify the following Bayesian prior distributions:

$$\begin{aligned} b_0, b_{1k}, b_{2k}, b_3, a_k &\sim N(\text{mean} = m_1, \text{var} = u) \text{ for all } k \\ \mathcal{T} &\sim N(\text{mean} = m_2, \text{var} = 100) \\ \ell_\gamma &\sim N(\text{mean} = \log 5.5, \text{var} = [\log 10]^2) \\ v^{-2}, \sigma_\tau^{-2}, \sigma_\gamma^{-2}, \sigma_k^2, \sigma_{k0}^{-2} &\sim \text{Gamma}(1, 0.01) \text{ for all } k \end{aligned}$$

where $m_1 = 16$ and $u = 100$ are the hyperparameter values for b_0 , but $m_1 = 0$ and $u = 10$ for the other slope parameters; and m_2 corresponds to the model variations below. We chose our hyperparameter values above based on preliminary (exploratory) model fits, some of which appear in Appendix 2.5.

Key variation (a): Timing of “bend”

To test whether the “bend” occurs around the year 2000 at the introduction of the original VMA, or around 2007 after the VMA amendments came into effect, we specify the hyperparameter values as $m_2 = 2000$ and $m_2 = 2007$, respectively.

Key variation (b): Spatial adjacency weighting

To test if weighting spatial adjacency by land tenure similarity affects model fit, as compared to unweighted spatial adjacency, we specify the CAR structure in Level 2 of the modeling framework as:

$$\left\{ \begin{array}{l} \beta_{10i} \mid \{\beta_{10j} : i \neq j\} \sim N\left(\text{mean} = \frac{\sum_{j \neq i} w_{ij} \beta_{10j}}{\sum_{j \neq i} w_{ij}}, \text{var} = \frac{\sigma_{10}^2}{\sum_{j \neq i} w_{ij}}\right) \\ \text{unweighted case: } w_{ij} = \mathbf{1}\{i, j \text{ share border}\} \text{ for all } i \neq j \\ \\ \text{weighted case: } w_{ij} = \frac{\mathbf{1}\{i, j \text{ share border}\}}{|L_i - L_j| + 0.00001} \text{ for all } i \neq j \end{array} \right.$$

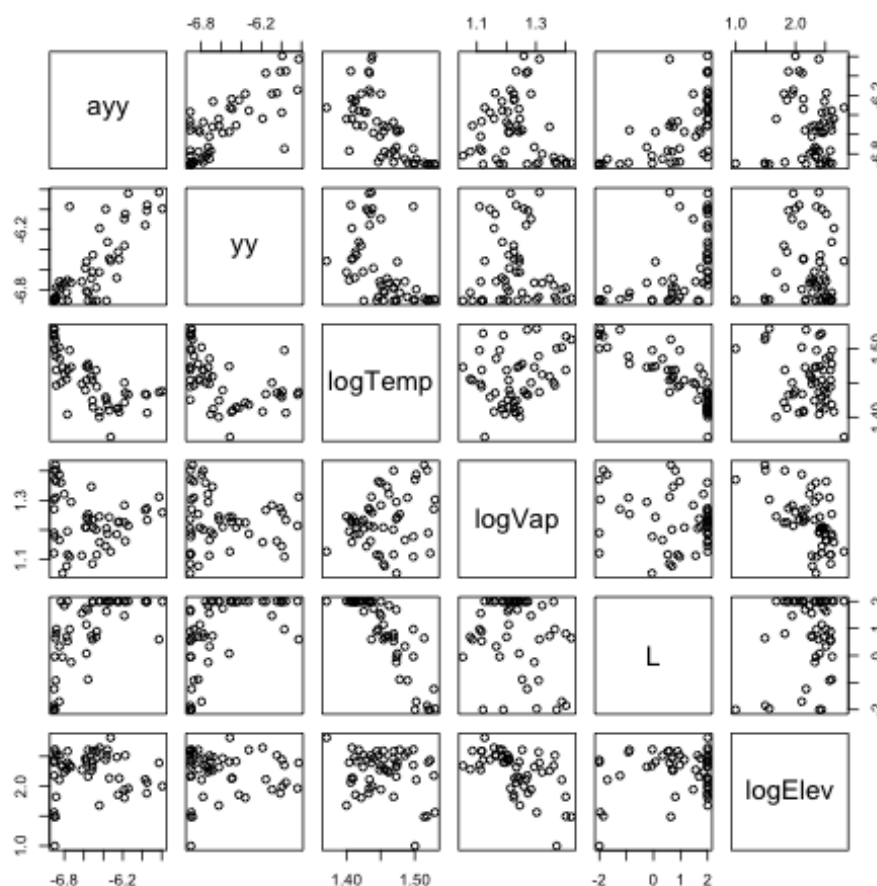
Other variations

Minor variations were considered and adopted for our final models if goodness-of-fit could be improved:

- alternative specification of prior distributions, e.g. bivariate joint prior for τ_i and $\log \gamma_i$
 - not adopted
- alternative hyperparameter values
 - not adopted
- alternative model parametrization in the WinBUGS implementation
 - not adopted
- reduced model complexity by taking $\log \gamma_i = \ell_\gamma$ for all i
 - adopted
- reduced model complexity by removing redundant driver variables
 - see Appendix 2.5

Appendix 2.5: Justification for discarding model covariates

In addition to state-level scatterplots in Appendix 2.2, LGA-specific as well as year-specific scatterplots (e.g. year 2014 shown below) were additionally inspected.



From such inspection, land tenure was subsequently removed from consideration as a driver variable due to its redundancy with temperature throughout 1988 to 2014.

The remaining drivers were more formally assessed through preliminary model fits using WinBUGS or R-INLA (Rue et al., 2009). Preliminary WinBUGS results suggested that Ag and Oil were highly redundant. Oil was kept for further assessment by R-INLA. Sample R-INLA results appear below. A smaller DIC value and effective number of parameters together suggest better goodness-of-fit. Thus, GDP was kept as the sole driver variable in our final models that are reported in Chapter 3 and summarised in Appendix 2.6.

Unweighted CAR and iid year effect, multiple drivers and year:

```
summary(result495_21)
```

```
##
```

```
## Fixed effects:
```

##		mean	sd	0.025quant	0.5quant	0.975quant	mode	kld
##	(Intercept)	13.5073	8.3423	-2.9157	13.5077	29.9085	13.5091	0
##	GDP	-0.0278	0.0203	-0.0677	-0.0278	0.0121	-0.0278	0
##	oil	-0.0097	0.0351	-0.0788	-0.0097	0.0594	-0.0097	0
##	temp	-0.0635	0.0629	-0.1863	-0.0637	0.0605	-0.0641	0
##	vap	0.0813	0.0569	-0.0304	0.0812	0.1932	0.0810	0
##	elev	-0.0555	0.0541	-0.1617	-0.0556	0.0513	-0.0558	0
##	Year	-0.0099	0.0042	-0.0181	-0.0099	-0.0017	-0.0099	0

```
## Deviance Information Criterion (DIC) ....: -4.243
```

```
## Effective number of parameters .....: 73.84
```

Unweighted CAR and iid year effect, single driver and year:

```
summary(result495_2)
```

```
##
```

```
## Fixed effects:
```

```
##           mean      sd 0.025quant 0.5quant 0.975quant      mode kld
## (Intercept) 16.5864 3.9808      8.7707  16.5862      24.3954 16.5864    0
## GDP          -0.0268 0.0155     -0.0572  -0.0268       0.0036 -0.0268    0
## Year         -0.0115 0.0020     -0.0154  -0.0115     -0.0076 -0.0115    0
```

```
## Deviance Information Criterion (DIC) ....: -9.363
```

```
## Effective number of parameters .....: 71.45
```

Appendix 2.6: Goodness-of-fit values for final models

Table A2.3. Model fits³⁶ for identically parametrized models: GDP and year are only covariates, with LGA-specific incoming and outgoing slopes, LGA-specific τ_i , but common γ . Note that small deviance together with small p_V imply good fit. As all four models involve the same number of parameters, the deviance can be used to compare all four against each other. In contrast, we do not use p_V to compare between a Simple and a Weighted model, because weighting appears to cause a drastic increase in the instability³⁷ of the WinBUGS numerical approximation algorithms.

Model variation	hyperparameter m_2	Spatial weight w_{ij}	Posterior median deviance	p_V estimate ³⁸
Simple CAR	2000	$\mathbf{1}\{i, j \text{ share border}\}$	-929.5	369616.0
Simple CAR	2007		-926.45	350471.5
Weighted CAR	2000	$\frac{\mathbf{1}\{i, j \text{ share border}\}}{ L_i - L_j + 0.00001}$	-1487.0	556042.4
Weighted CAR	2007		-1516.0	535166.6

³⁶ The results reported in Chapter 3 and these Supplementary Materials are based on locally converged models, which may differ from the global best fit. Having a mere 50 LGAs in the dataset was the likely reason for WinBUGS to exhibit (a) unstable estimates of p_V and (b) difficulty in locating the global best fit; indeed multiple local minima are inherent in the bent-cable model deviance (Chiu et al., 2006; Chiu & Lockhart, 2010).

³⁷ As suggested by the unusually large p_V estimates.

³⁸ Computed by dividing the posterior variance of the deviance by two (Gelman et al., 2013).

Appendix 2.7: Influence of land tenure similarity weighting on spatial estimates

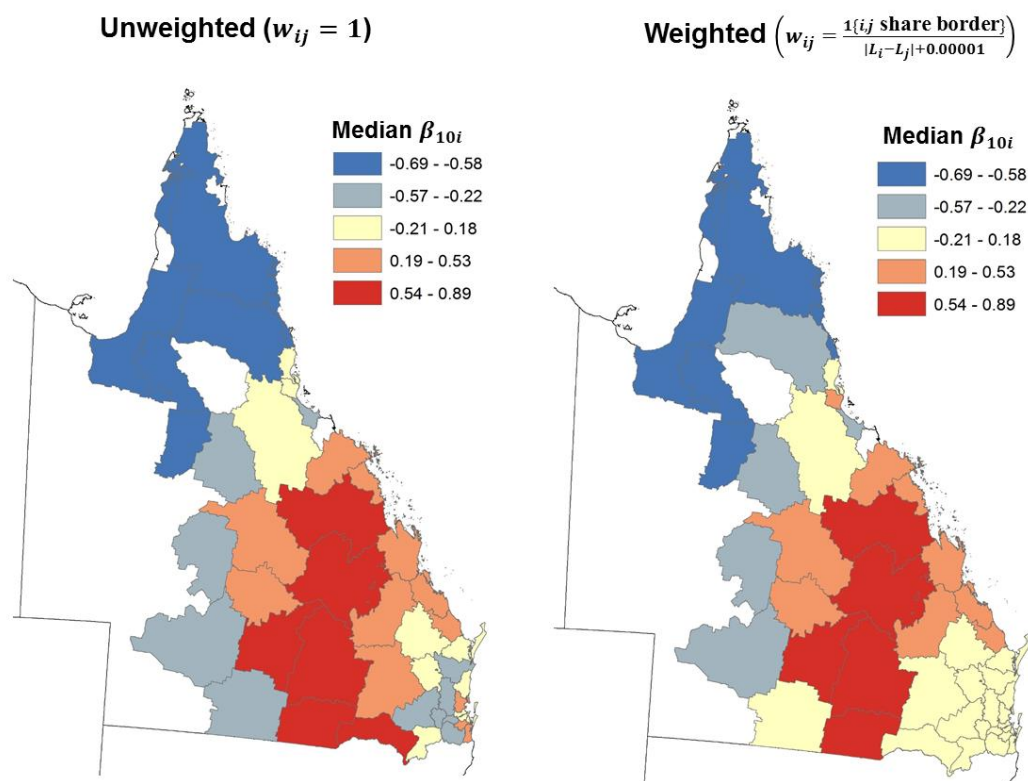
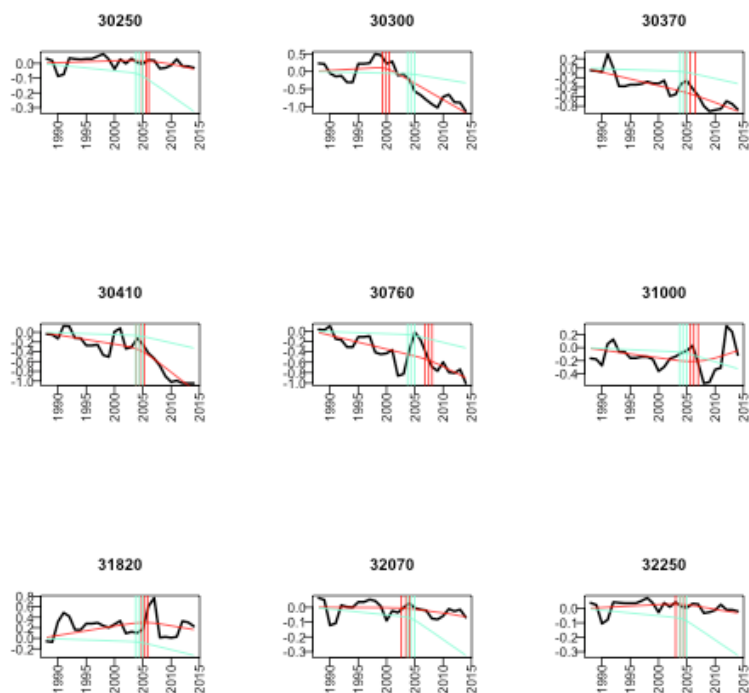
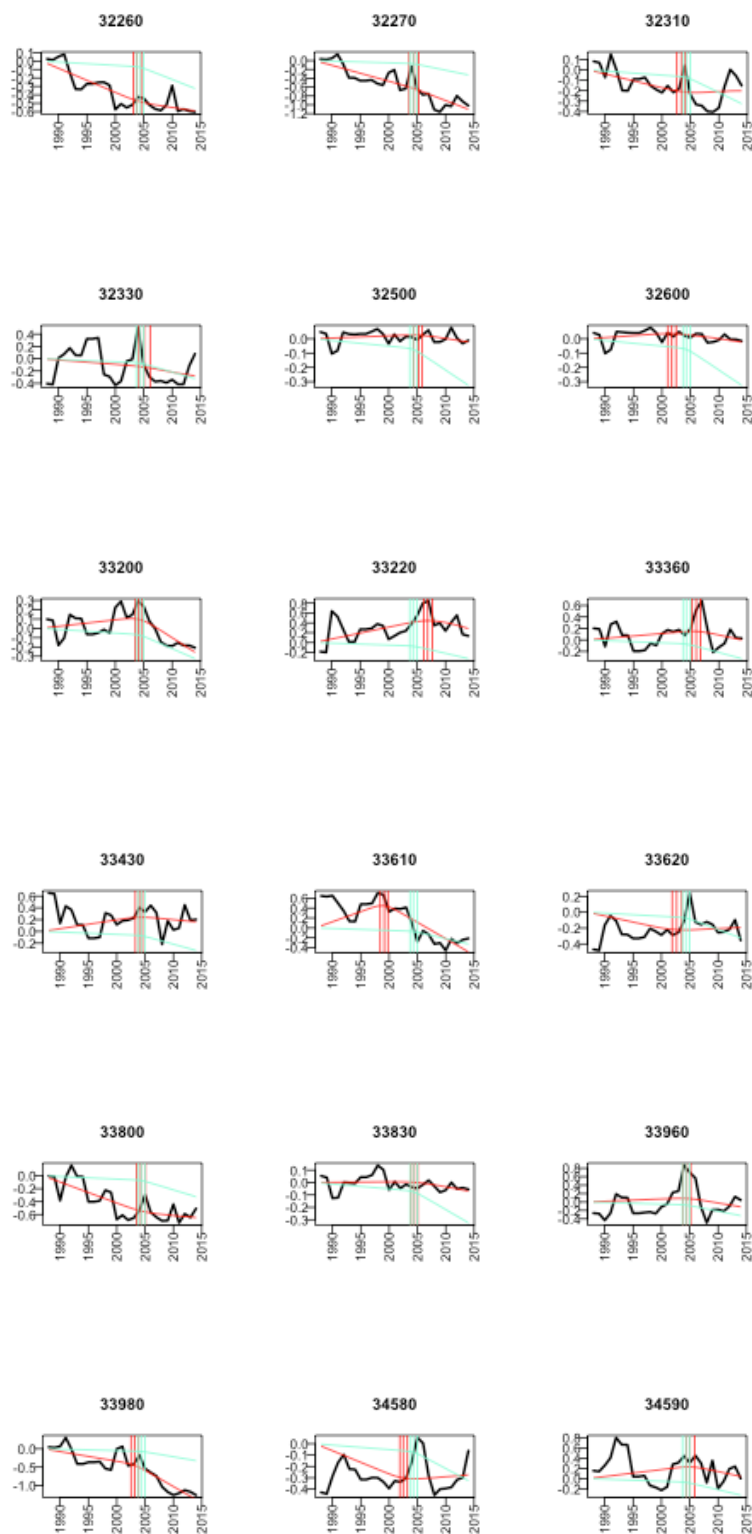


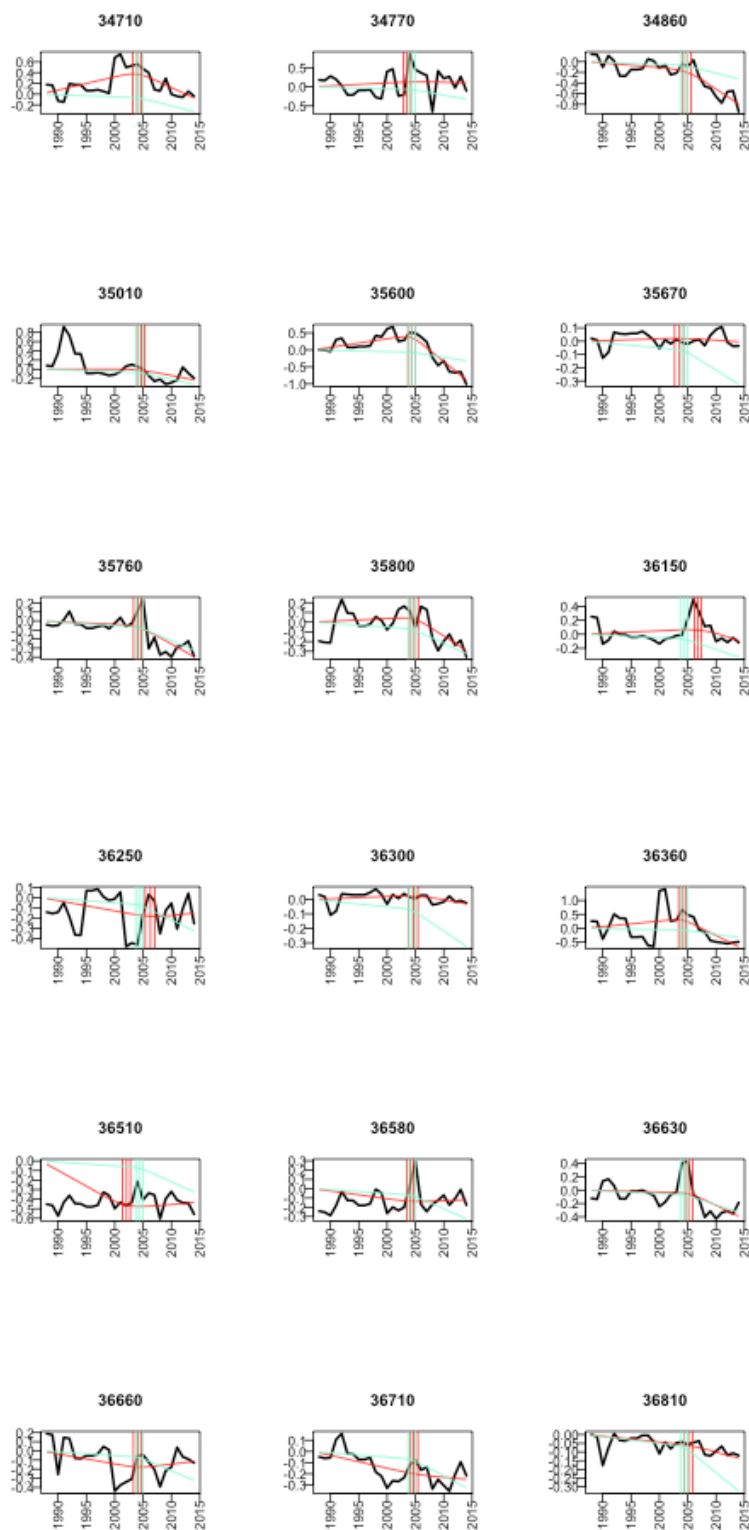
Figure A2.2. Estimates (posterior medians) of spatial random effects β_{10i} for unweighted model (left, median deviance ~ -930) and weighted model (right, median deviance ~ -1500). Smaller deviance indicates better goodness-of-fit.

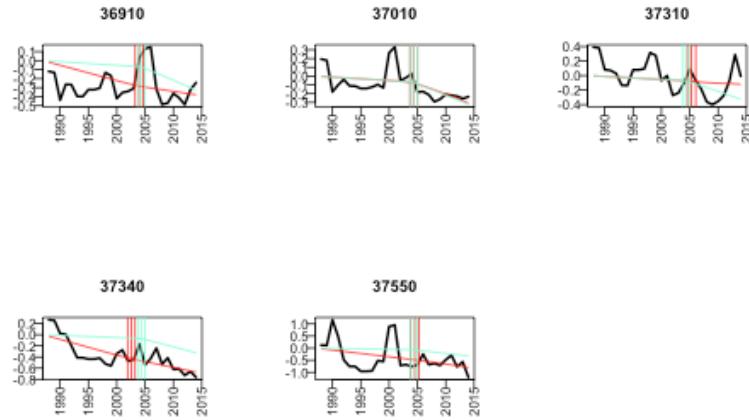
Appendix 2.8: LGA-level time series ($n = 50$)

Figure A2.3: LGA-level time series y_{it} ($n = 50$) after detrending (black), with LGA-level bent-cable estimate (red) and fitted population-level bent cable (green).









]

Literature cited

Rue H, Martino S and Chopin N (2009) Approximate Bayesian inference for latent Gaussian models using integrated nested Laplace approximations (with discussion). *Journal of the Royal Statistical Society, Series B* 71(2): 319–392.

Appendix Three

Chapter Four Supplementary Material

Appendix 3.1: Methodologies and carbon accounting under the Carbon Farming Initiative (CFI)

Two offset methodologies for assisted natural regeneration (ANR, also known as managed regrowth) are available under the Australian Carbon Farming Initiative (Commonwealth of Australia, 2013a, 2013b). Generation of carbon credits via ANR requires one or more of the following activities to be undertaken: (a) the exclusion of livestock, (b) the management of the timing and the extent of grazing, (c) the management, in a humane manner, of feral animals, (d) the management of plants that are not native to the project area, (e) the cessation of mechanical or chemical destruction, or suppression, of regrowth vegetation. A project which employs the Native Forest from Managed Regrowth methodology (Commonwealth of Australia, 2013a) must include (e), and may also involve one or more of (a) to (d). The Human-Induced Regeneration of a Permanent Even-Aged Native Forest methodology (Commonwealth of Australia, 2013b) differs in that any one or more of (a) to (e) may be undertaken.

The two ANR methodologies currently available under the CFI differ in how carbon abatement is calculated. The Human-Induced Regeneration of a Permanent Even-Aged Native Forest methodology assumes a zero baseline for carbon stocks, whereas the Native Forest from Managed Regrowth methodology accommodates baseline scenarios that allow for fluctuations in carbon stocks. The FullCAM model (Richards and Evans, 2004; FullCAM, Richards, 2001), which forms the basis of the RMT, is required to calculate abatement where a non-zero, dynamic baseline for carbon stocks is assumed. Despite this difference, our findings are still broadly applicable to the two ANR methodologies recognized under the CFI. Dynamic baselines will generally be small compared to sequestration potential, and will almost always be lower than the carbon stock at project commencement. Estimation of

biomass accumulation as modeled by FullCAM has recently been updated (Paul et al., 2013), so the carbon sequestration rates we use here should be regarded as conservative.

Australia's national greenhouse gas accounting system and the CFI are imperfectly aligned, which means that methodologies cannot be wholly classified as Kyoto-compliant or non-compliant. For a CFI project to be considered for Kyoto compliance, the project proponent must submit an application to be assessed by the Australian Clean Energy Regulator (Commonwealth of Australia, 2012a, 2011). Activities listed under the two ANR methodologies under the CFI fall under the Kyoto definition of reforestation (Commonwealth of Australia, 2011); however, the Clean Energy Regulator maintains authority over decisions pertaining to the interpretation of the CFI Act (2011), regulations and methodology determinations in relation to the crediting of projects.

Appendix 3.2: Deriving an estimate of the establishment cost for ANR

Schirmer and Field (2000) estimated costs of establishing an assisted natural regeneration project, including fencing materials, fencing labour and transport of fencing materials for 1 ha, 10 ha and 50 ha projects. We ignore the project management and monitoring costs considered by Schirmer and Field (2000) in deriving an initial establishment cost at $t=0$ as these annual costs are considered elsewhere in our calculations (see Equation 6 in manuscript).

Table A3.1. Costs of an assisted natural regeneration (ANR) project. Adapted from Schirmer and Field (2000)

Cost description	Cost/ha for 1ha project (\$)	Cost/ha for 10 ha project (\$)	Cost/ha for 50 ha project (\$)
Fencing materials @ \$2550/km*	1020	357	153
Fencing labour @ 44 hours/km = \$660/km	264	92	40
Spot spraying for weeds 3 times @ \$75/ha/application	225	225	225
Transport @ \$0.52/km for 500 km = \$572 total**	260	26	5
Total cost per hectare	1769	700	423

* Project areas are assumed to be square in shape and to require fencing on all sides, for the purposes of providing a consistent comparison

** Site is assumed to be 50 kilometres from nearest fencing supplier and the office location of the nearest revegetation professional. Farmer transports fencing materials at \$0.52/kilometre; revegetation professional makes 5 site trips at \$0.52 per kilometer.

We derived an estimated establishment cost for ANR for a 100ha project to suit the spatial scale of our analysis (1km²) by fitting a power relationship (Figure A3.1, $R^2 = 0.996$) to the cost estimates for a 1ha, 10ha and 50ha project as provided by Schirmer and Field (2000).

Table A3.2. Derived estimate of an ANR establishment cost for 100 ha project area.

Project area (ha)	Cost/ha	Predicted cost/ha	Adjusted for inflation (2.9% pa between 2000 and 2013)
1	1769	1728.7	
10	700	740.8	
50	423	409.7	
100		317.5	459.8

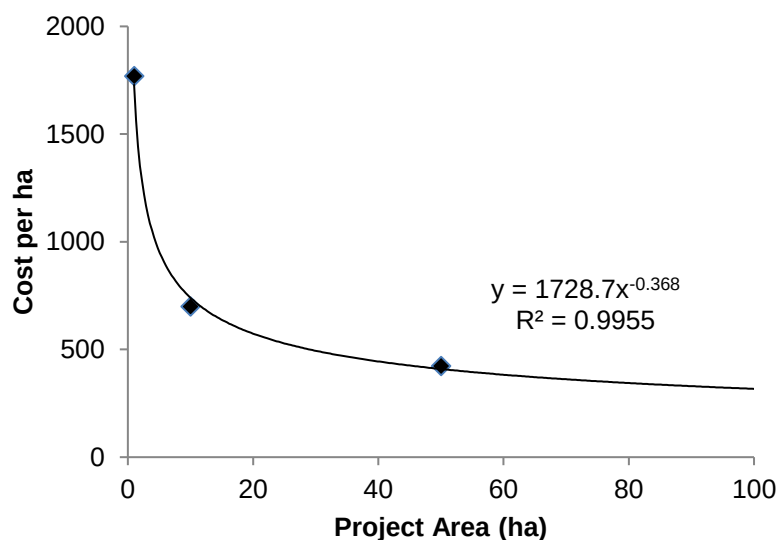


Figure A3.1. Fitted relationship used to derive establishment cost estimate for ANR

Appendix 3.3: Supplementary results

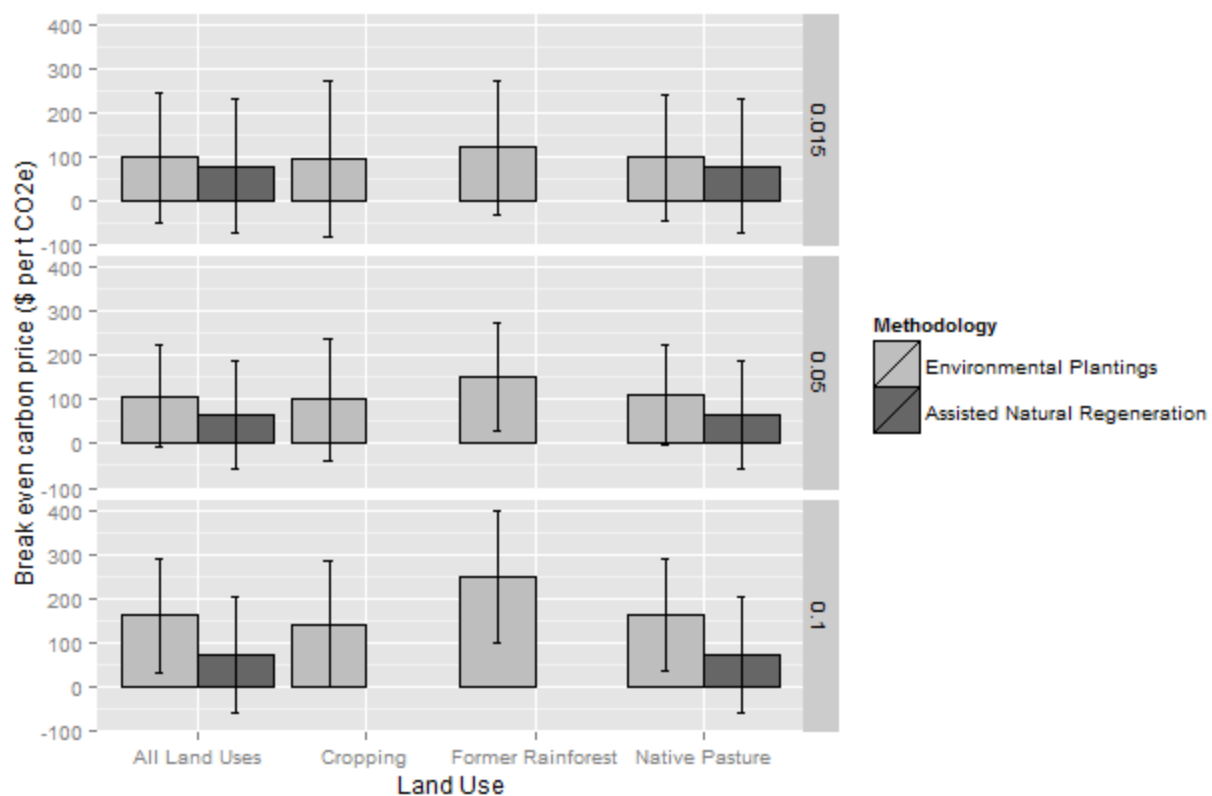


Figure A3.2. Break-even carbon prices for each land use averaged over all eligible sites. Discount rates (from top panel) are 1.5%, 5% and 10%. Project duration is 100 years.

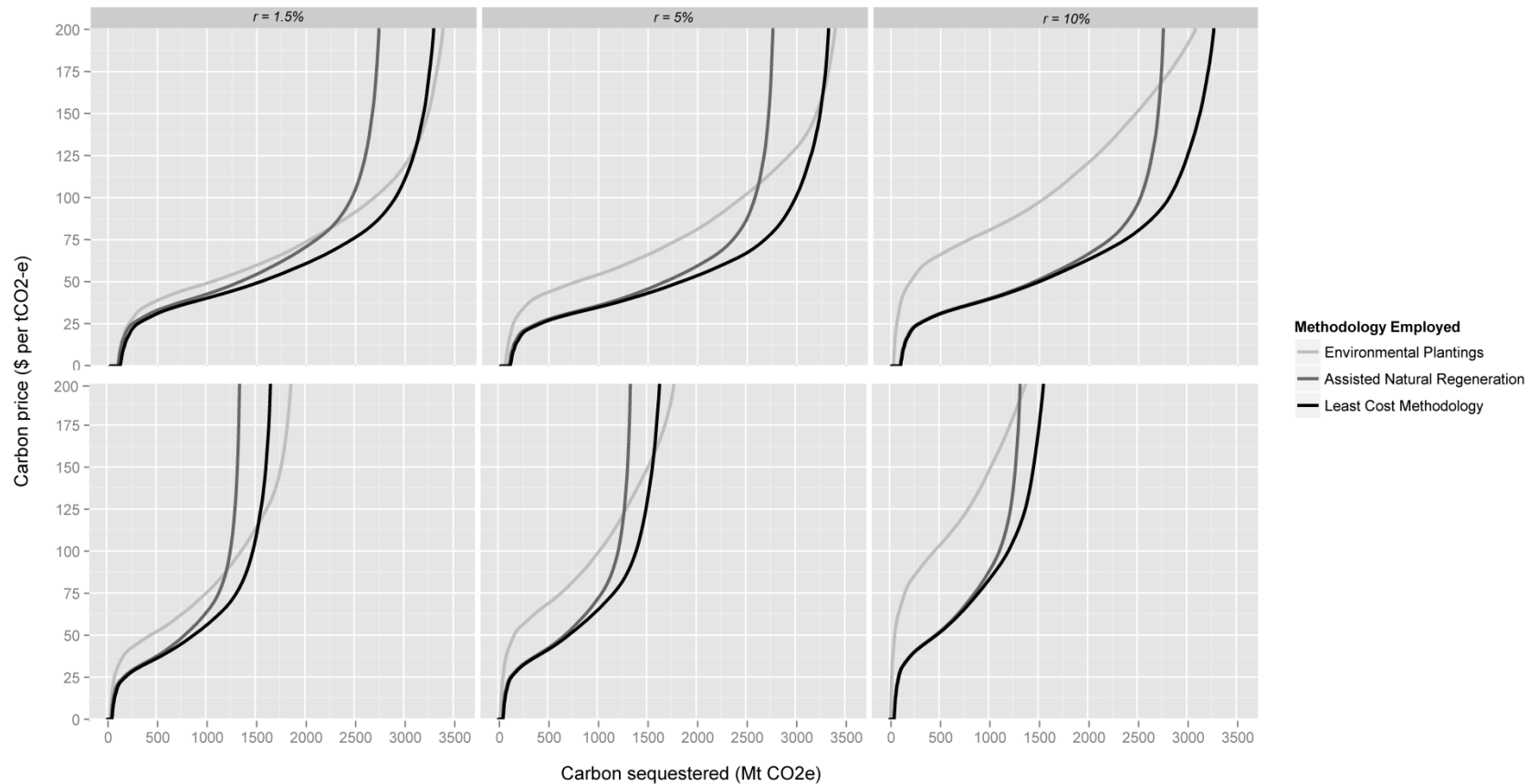


Figure A3.3. Effects of discounting and project duration on carbon sequestration supply (top panel, $T=100$; bottom panel, $T=25$). The y-axis is restricted to \$200 per t CO₂-e or less for ease of interpretation.

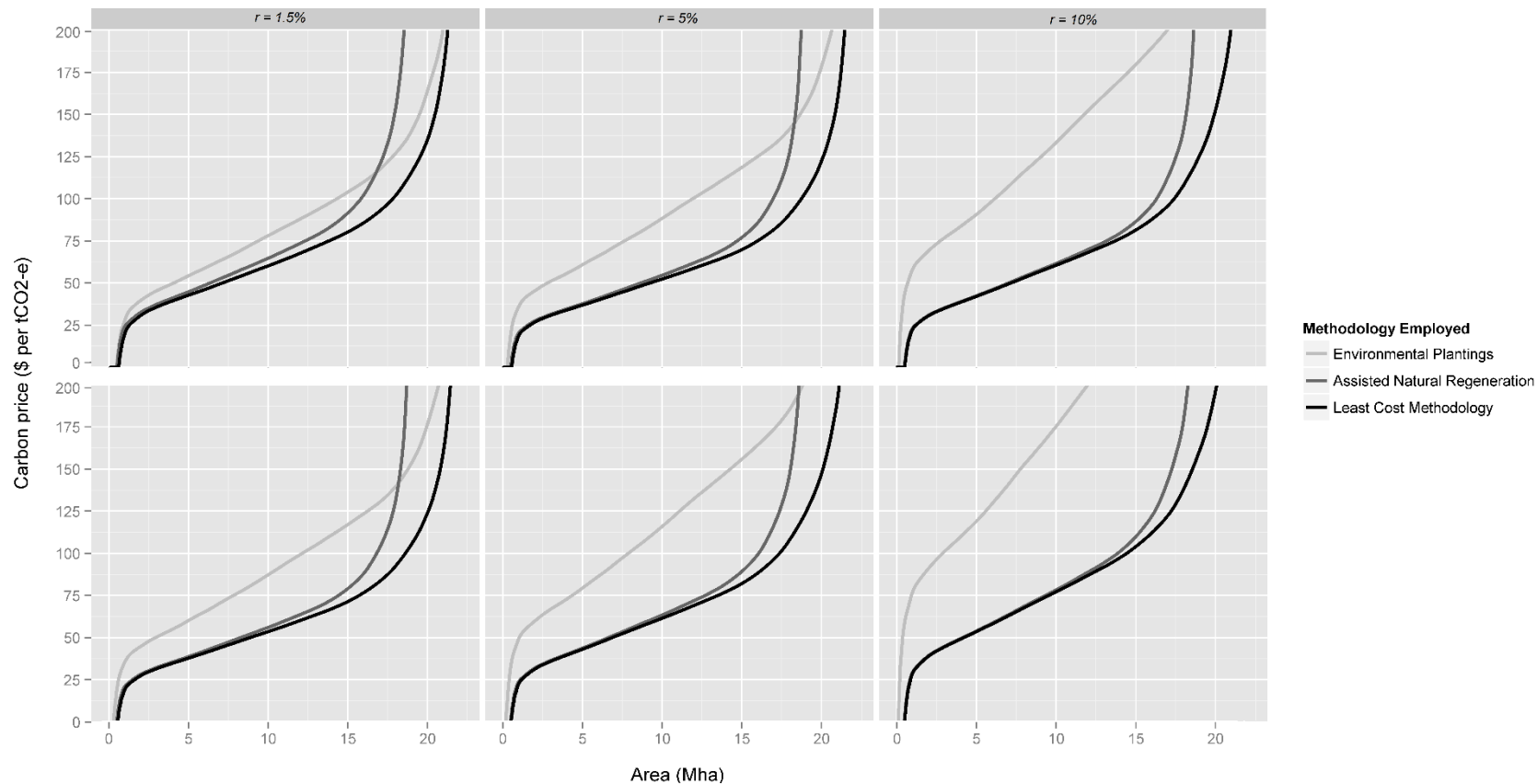


Figure A3.4. Effects of discounting and project duration on area viable for carbon farming (top panel, T=100; bottom panel, T=25). The y-axis is restricted to \$200 per t CO₂-e or less for ease of interpretation.

Literature cited

- Commonwealth of Australia (2011) Carbon Credits (Carbon Farming Initiative) Act 2011. Available from: <http://www.comlaw.gov.au/Details/F2013L02036> (accessed 16 December 2013).
- Commonwealth of Australia (2012) The Carbon Farming Initiative Handbook. Canberra.
- Commonwealth of Australia (2013a) Carbon Credits (Carbon Farming Initiative) (Human Induced Regeneration of a Permanent Even-Aged Native Forest—1.1) Methodology Determination 2013 - F2013L01189. Canberra, Australia: Department of the Environment. Available from: <http://www.comlaw.gov.au/Details/F2013L01189>.
- Commonwealth of Australia (2013b) Carbon Credits (Carbon Farming Initiative) (Native Forest from Managed Regrowth) Methodology Determination 2013 - F2013L02036. Canberra, Australia: Department of the Environment. Available from: <http://www.comlaw.gov.au/Details/F2013L02036>.
- Paul KI, Roxburgh S, Raison J, et al. (2013) Improved estimation of biomass accumulation by environmental planting and mallee plantings using FullCAM. Report for The Department of the Environment. Canberra, Australia: CSIRO Sustainable Agriculture Flagship.
- Richards GP (2001) The FullCAM carbon accounting model: development, calibration and implementation for the National Carbon Accounting System / Gary P. Richards. National Carbon Accounting System technical report ; no. 28., PANDORA electronic collection., Canberra: Australian Greenhouse Office. Available from: <http://www.greenhouse.gov.au/ncas/reports/tr28final.html>, <http://nla.gov.au/nla.arc-23322>, <http://www.greenhouse.gov.au/ncas/reports/pubs/tr28final.pdf> (accessed 23 January 2014).
- Richards GP and Evans DMW (2004) Development of a carbon accounting model (FullCAM Vers. 1.0) for the Australian continent. *Australian Forestry* 67(4): 277–283.

Appendix Four

Chapter Seven Supplementary Material



Appendix 4.1: Sample interview questions

1. Could you describe your current role, and contact with offsets in this role?
2. From your perspective, what do you consider to be a “good” or “successful” biodiversity offset outcome?
3. Do you think that the offsetting arrangements you’re involved would meet the definition of a “suitable offset” as defined by the EPBC Act Environmental Offsets Policy? Why/why not?

“Suitable offsets must deliver an overall conservation outcome that improves or maintains the viability of the protected matter”

Australian Government (2012) Environment Protection and Biodiversity Conservation Act 1999 Environmental Offsets Policy

4. Thinking now about the implementation of biodiversity offsetting: Which parts of the offset process are you involved in, which parts are conducted by other parts of the Department?
5. From your perspective, what are some of the key things needed for the biodiversity offsetting process to go smoothly?
6. What are some reasons that process might not go smoothly?
7. Thinking now about monitoring and evaluation of biodiversity offsetting: What role does your branch/section play in monitoring, reporting and evaluation of biodiversity offsetting?
8. Do you think there is adequate monitoring and evaluation of biodiversity offsetting? Why/why not?
9. In your view, what are 3 key things needed to improve biodiversity offsetting in Australia/under the EPBC Act?
10. Is there anything else you would like to add?

If you have any questions or would like any further information on this research project, please contact either myself or one of my supervisors, Karen Hussey or Stuart Whitten:

Megan Evans
PhD Candidate
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Mobile: +61 418 984 248

Professor Karen Hussey
Primary PhD Supervisor
Email: k.hussey@uq.edu.au
Phone: +61 7 3443 3112

Dr Stuart Whitten
PhD supervisor
Email: stuart.whitten@csiro.au
Phone: +61 2 6246 4359

Ethics Committee Clearance:

The ethical aspects of this research have been approved by the ANU Human Research Ethics Committee (Human Ethics Protocol 2015/274). If you have any concerns or complaints about how this research has been conducted, please contact:

Ethics Manager
The ANU Human Research Ethics Committee
The Australian National University
Telephone: +61 2 6125 3427
Email: Human.Ethics.Officer@anu.edu.au

Appendix 4.2: Participant Information Sheet

Project Title: Stakeholder perceptions on the efficacy of biodiversity offset policy in Australia

Researcher:

My name is Megan Evans, and I am a PhD candidate student from the Fenner School of Environment and Society within the College of Medicine, Biology and Environment at the Australian National University. I am undertaking a PhD research project on the role of key actors within the processes of biodiversity offset policy implementation. This research is funded by an Australian Postgraduate Award and a CSIRO top-up scholarship (Grant ID: 27226).

General Outline of the Project:

Description and Methodology: The overall aim of the research is to gain an understanding of what the key barriers and enablers are to achieving effective, efficient and equitable biodiversity offset policy. This research will explore the views of different policy actors involved in the implementation of biodiversity offset policy, and seek to understand the extent to which existing institutional structures are aligned in favour of achieving positive environmental outcomes through offsetting. Perceptions of key policy actors in Australia will be elicited through semi-structured interviews.

Participants Approximately 25 semi-structured interviews will be conducted with key policy actors, including policy administrators, project proponents, market intermediaries and third-party offset suppliers. Participants will be identified using a snowball sampling approach.

Use of Data and Feedback: This research will be presented as an ANU PhD thesis for the Fenner School of Environment and Society. It is the intention that at least two peer-reviewed publications will be prepared once the project is completed. As a participant in this research, you will be offered an electronic version of the publications once they are published, as well as a shorter project summary. You will be able to contact me at any time to discuss the research.

Participant Involvement:

Voluntary Participation & Withdrawal: Your participation in this research project is **voluntary** and you may, without any penalty, decline to take part or withdraw from the research at any time until the work is prepared for publication without providing an explanation. You are free to refuse to answer any questions. If you do choose to withdraw, the data from your interview will be destroyed and not used.

What does participation in the research request of you? Your participation in this research would be in the form of a semi-structured interview. This will involve me (Megan Evans) asking you a series of questions, but in a flexible manner – for example, we may not go through all of the questions I have prepared, or we may discuss some issues in more detail

than others, depending on where the interview takes us. With your consent, I will record the audio of our interview, and I may also take handwritten notes during the interview. Once the interview is completed, the audio recording will be transcribed by professional transcribing service. I may contact you again to cross-check the transcription and my notes with you, to make sure I have adequately and accurately captured your views as expressed in the interview. Information provided during our interview as well as interviews with other participants will be analysed to gain an understanding of what are the key issues important for different policy actors to enable the successful implementation of biodiversity offset policy. During the data analysis phase, I will cross-check and discuss the key themes arising from the interview data with my supervisors, Prof Karen Hussey (UQ) and Dr Stuart Whitten (CSIRO).

Location and Duration: The interview may take place either in person or over the phone if an in-person interview is not possible. The total length of the interview is expected to be about an hour, and the total time requested of you in this research is two hours (accounting for time taken to arrange the interview, and any follow-up correspondence).

Remuneration: Participation in this research is voluntary, and no incentives will be provided for participation.

Risks: The risk to you as a participant in this research is expected to be low, due to the nature of the research and the steps which will be taken to minimise risks. Your personal and financial circumstances (or of the organisations you represent) will not be discussed, unless such information is volunteered by you. Neither you nor the specific organisation you represent will be identified during the research, and instead will simply be classified by sector. Your name and the notes from the interview will be kept confidential to minimize identification risk.

Benefits: It is unlikely that you will personally benefit from participating in this research, but we anticipate that this research will provide practical insights to the policy community, particularly those groups interested in, or responsible for, the implementation of the EPBC Act Environmental Offsets Policy.

Confidentiality:

Confidentiality: The procedures for confidentiality will comply with the *Commonwealth Privacy Act of 1988* and the ANU Responsible Conduct of Research Policy. Your privacy will be protected as far as the law allows. Your name and the name of your organisation will not be disclosed. Collected data will be securely stored, and interview notes will not be kept in the same place as participant's details. Pseudonyms will be used in the reporting of research to ensure minimal risk of participant identification.

Data Storage:

Where: All interview notes and transcriptions will be stored in a locked cabinet at the ANU. An online backup using the secure Dropbox domain will be used. This online data will be copied to an external hard drive and deleted from Dropbox after data analysis. The data

collected will be stored for 5 years on a hard drive after the date of research publication in order to comply with ANU guidelines.

How long: Data will be stored for a period of at least five years from the date of research publication.

Destruction of Data: At the end of the storage period, the data will be destroyed.

Queries and Concerns:

Contact Details for More Information: If you have any questions or would like any further information on this research project, please contact either myself or one of my supervisors, Karen Hussey or Stuart Whitten:

Megan Evans
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Ethics Committee Clearance:

The ethical aspects of this research have been approved by the ANU Human Research Ethics Committee. If you have any concerns or complaints about how this research has been conducted, please contact:

Ethics Manager
The ANU Human Research Ethics Committee
The Australian National University
Telephone: +61 2 6125 3427
Email: Human.Ethics.Officer@anu.edu.au

Appendix Five

- Miller, K.L., Trezise, J.A., Kraus, S., Dripps, K., **Evans, M.C.**, Gibbons, P., Possingham, H.P. & Maron, M. (2015). The development of the Australian environmental offsets policy: from theory to practice. *Environmental Conservation*, 42(4): 306–314.
- Gibbons, P., **Evans, M.C.**, Maron, M., Gordon, A., Le Roux, D., von Hase, A., Lindenmayer, D.B. & Possingham, H.P. (2016). A Loss-Gain Calculator for Biodiversity Offsets and the Circumstances in Which No Net Loss Is Feasible. *Conservation Letters*, 9(4): 252–259
- Maron, M., Bull, J.W., **Evans, M.C.** & Gordon, A. (2015). Locking in loss: Baselines of decline in Australian biodiversity offset policies. *Biological Conservation*, 192: 504–512.
- Martin, N., **Evans, M.C.**, Rice, J., Lodhia, S. & Gibbons, P. (2016). Using offsets to mitigate environmental impacts of major projects: A stakeholder analysis. *Journal of Environmental Management*, 179: 58–65.
- Maron, M., Ives, C.D., Kujala, H., Bull, J.W., Maseyk, F.J.F., Bekessy, S., Gordon, A., Watson, J.E.M., Lentini, P.E., Gibbons, P., Possingham, H.P., Hobbs, R.J., Keith, D.A., Wintle, B.A. & **Evans, M.C.** (2016). Taming a Wicked Problem: Resolving Controversies in Biodiversity Offsetting. *BioScience*, 66 (6): 489–498.