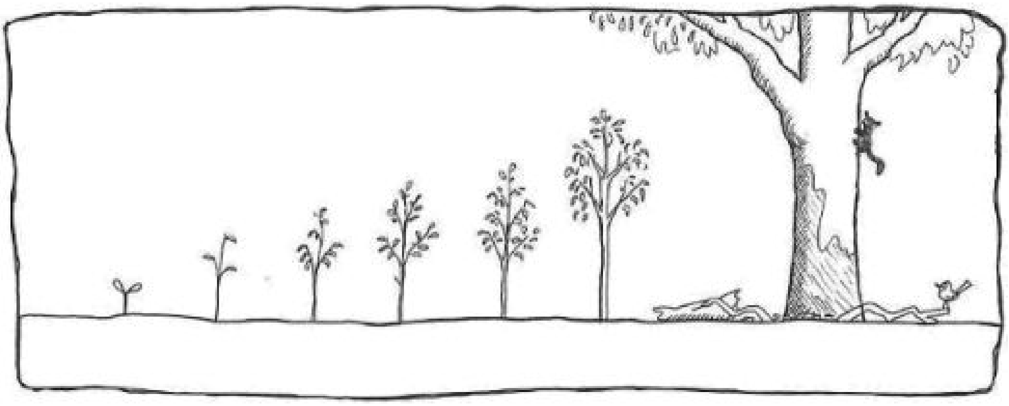
Department of Science, Information Technology, Innovation and the Arts



Carbon farming and natural resource management in eastern Australia

Great state. Great opportunity.

Queensland Government

Prepared by

Don Butler, Jason Halford and Megan Evans

Queensland Herbarium
Science Delivery Division
Department of Science, Information Technology, Innovation and the Arts
PO Box 5078
Brisbane QLD 4001

© The State of Queensland (Department of Science, Information Technology, Innovation and the Arts) 2014

The Queensland Government supports and encourages the dissemination and exchange of its information. The copyright in this publication is licensed under a Creative Commons Attribution 3.0 Australia (CC BY) licence

Under this licence you are free, without having to seek permission from DSITIA, to use this publication in accordance with the licence terms.

You must keep intact the copyright notice and attribute the State of Queensland, Department of Science, Information Technology, Innovation and the Arts as the source of the publication.

For more information on this licence visit http://creativecommons.org/licenses/by/3.0/au/deed.en

Disclaimer

This document has been prepared with all due diligence and care, based on the best available information at the time of publication. The department holds no responsibility for any errors or omissions within this document. Any decisions made by other parties based on this document are solely the responsibility of those parties. Information contained in this document is from a number of sources and, as such, does not necessarily represent government or departmental policy.

If you need to access this document in a language other than English, please call the Translating and Interpreting Service (TIS National) on 131 450 and ask them to telephone Library Services on +61 7 3170 5725

Citation

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.

Acknowledgements

This report has been prepared by the Department of Science, Information Technology, Innovation and the Arts. Acknowledgement is made of Ramona Maggini for providing data on modelled current threatened species richness from the NCCARF project 'Protecting and restoring habitat to help Australia's threatened species adapt to climate change' (Maggini et al. 2013). Thanks also to John Neldner, Megan Surawski, Keryn Oude-Egberink, Paula Peeters and Melinda Laidlaw for comments on draft manuscripts.

Cover illustration by Paula Peeters

December 2014

Executive summary

This report reviews some of the potential landscape changes that carbon farming may bring to eastern Australia[1].

Particular attention is given to vegetation based activities, such as reforestation using regrowth or plantings, because they are relatively high profile “headline” activities that may cause landscape-scale changes in natural resources and their management in eastern Australia.

Carbon farming in Australia revolves around the Carbon Farming Initiative/Emissions Reduction Fund (CFI/ERF) regulatory framework administered by the Commonwealth Government. There are also several international ‘voluntary’ markets.

The core model for carbon farming involves making changes to land management practices, beyond business as usual, to avoid greenhouse gas emissions or to achieve permanent removal of atmospheric greenhouse gas.

At the time of writing, nine CFI methodologies were available for activities with potential to significantly affect natural resource management at landscape scale. Activities covered by these methodologies include establishment of new forests (by planting, seeding or enhancing natural regeneration), native forest protection, savanna burning and soil carbon accumulation in grazing systems.

There is substantial scope for growth in carbon farming activities. Uptake to date has been very limited compared to published estimates of ‘potential’. Future uptake is expected to depend on complex interactions between social, economic, biophysical and policy factors. Historic policy shifts and associated uncertainty may explain some of the slow uptake to date. The need for land to be ‘permanently’ committed for carbon farming by biomass accumulation may be contributing to particularly low uptake of sequestration activities such as tree planting and regrowth retention and management.

Substantially more land could support profitable carbon farming based on native forest regrowth (assisted natural regeneration) than environmental plantings, primarily because regrowth has lower upfront costs than plantings. Up-front costs and complexity are also issues for the current avoided deforestation methodology, though it does show stronger economic prospects than environmental plantings. Opportunities for avoided deforestation in Queensland are currently strongly constrained by a mismatch between the entry requirements for the available methodology and the regulatory framework for native vegetation management in that State. That is, the method requires a permit but permits are not necessary for most ongoing types of clearing in Queensland.

Our analysis suggests that carbon prices above $20-$30 per tonne of carbon dioxide equivalent abatement are required before substantial portions of the east Australian landscape are likely to have potential for profitable carbon farming with regrowth or plantings. This is consistent with other economic analyses and must be evaluated in light of the well-established sensitivity of such analyses to assumptions about discount rates, opportunity costs and management costs.

[1] This report is focussed on six regional natural resource management regions; three are Queensland NRM bodies (Fitzroy Basin Association, Burnett Mary Regional Body and South-east Queensland Catchments) and three are Local Land Service regions in New South Wales (North Coast, Hunter and Greater Sydney). However, the study areas for economic and biodiversity analyses presented in Chapters 2 and 3 are broader than these regions to provide context and maximise utility.

Even with a moderate five per cent discount rate, activities with lower on-ground establishment costs show far more potential for economies of scale to provide benefit by reducing ongoing costs in large projects than activities with high up-front costs do. Profitability is a medium to long term prospect for carbon farming through vegetation sequestration, with time horizons longer than 10 years likely to be necessary before projects in many potentially profitable areas will break-even.

If carbon farming involving protection or establishment of new native forests does achieve widespread uptake there are likely to be significant co-benefits to natural resource management from changes to hydrology, climate (from micro to regional scales) and biodiversity. There is also potential for dis-benefits, especially from hydrological changes in highly regulated catchments or where in-stream salinity is already an issue. Other potential dis-benefits from carbon farming such as risks of increased native vegetation clearing for plantation establishment are currently effectively managed by restrictions within the regulatory framework for carbon farming in Australia.

Weather is the major driver of fire risk, and forecast hotter climates with greater potential evapotranspiration suggest that the incidence of extreme fire danger may increase in many parts of eastern Australia, particularly in the south. In this context the idea of increased forest extent may raise concerns about exacerbating future fire risk. Forests do increase risk from fire to infrastructure in close proximity to forest areas (within about 200 m), so new forests should not be located near fire-sensitive infrastructure. But at regional scales the cooler and generally moister environment within forests can reduce the rate of fire spread and modest additions to forest extent likely to occur under carbon farming may cause reductions in area burnt rather than adding to risk.

New forests are likely to benefit biodiversity by increasing habitat resources for native biota. There are myriad techniques that can be used to identify locations where revegetation would be most beneficial for biodiversity.

General principles that can guide the identification of beneficial locations for revegetation include complementarity for remaining vegetation, locations subject to extinction debt, connectivity to remaining natural habitat, and potential future habitat for threatened species.

Consideration of pre-clearing vegetation types and complementarity will continue to be useful indicators of priority for revegetation even though revegetation should not necessarily be expected to produce ecosystems analogous to pre-clearing vegetation. The aim should be to make a range of habitat types available to biodiversity, and land types heavily impacted by past clearing are generally indicators of soil and other conditions that are relatively uncommon among areas of remnant vegetation. Opportunities to revegetate or otherwise reduce land-use intensity in such places should be sought and supported to conserve biodiversity.

Contents

List of tables

List of figures

List of maps

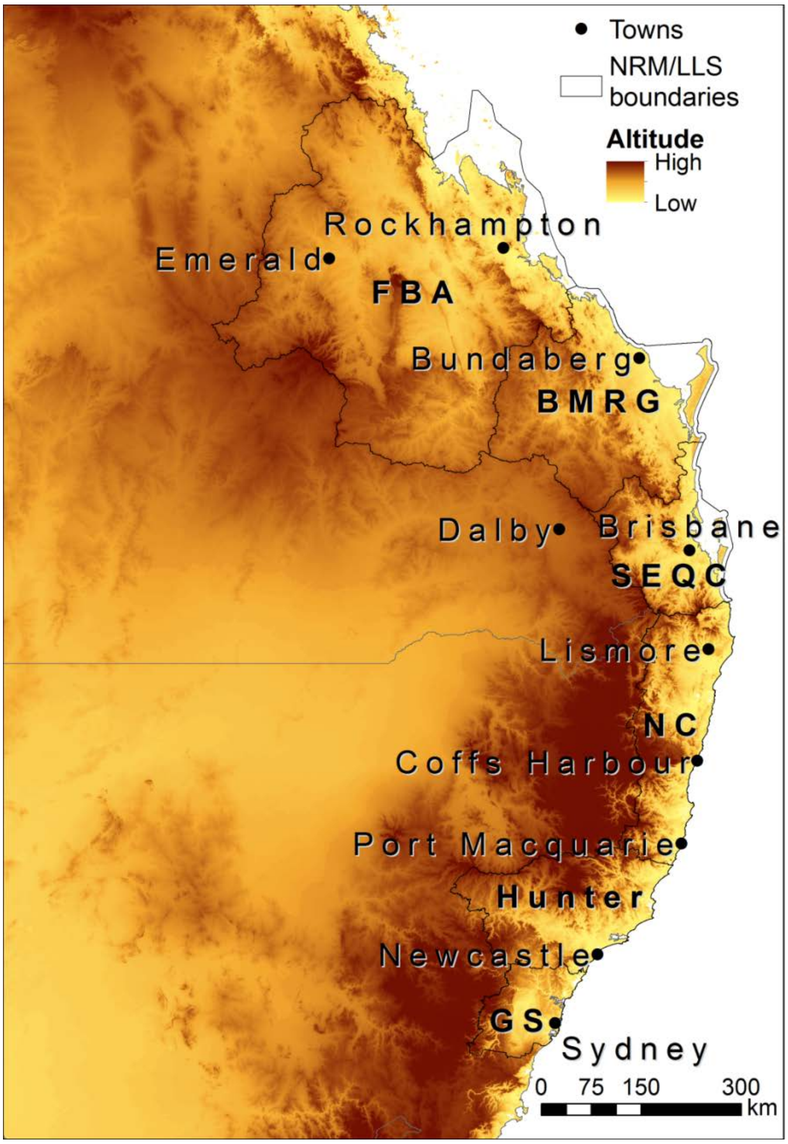
Introduction

This report focuses on six natural resource management areas in eastern Australia (Map 1). The theme is carbon farming. At its core, carbon farming is about changes to land-management that result in the deliberate and additional capture and storage of greenhouse gases, or deliberate and additional reductions in emissions of greenhouse gases. In this context, something is additional if it wouldn't have happened under plausible business as usual scenarios, without a deliberate change in land-management activity.

Particular attention is given to vegetation-based carbon farming activities, such as reforestation using regrowth or plantings, because potential for changes to landscape-scale natural resource management is concentrated in these activities in eastern Australia.

There are three sections to this report. They address overarching issues of policy (section 1), economics (section 2), and indirect effects on natural resources, particularly effects from changes to hydrology, fire and biodiversity (section 3).

The maps presented in the main body of the report provide broad-scale overviews. Larger-scale maps of key results for each of the six NRM regions are provided in Appendix 1. The datasets developed through this work are available as GIS files.



Map 1. East Australian elevation with boundaries of focal natural resource management areas.

1 Overview

1.1 Carbon farming as a regulatory framework

Carbon farming is an emerging and complex economic sector. The Commonwealth Government regulates Australia's highest-profile carbon crediting scheme. This scheme was established by the *Carbon Credits (Carbon Farming Initiative) Act 2011*, the CFI, which the Commonwealth is transitioning into the Emissions Reduction Fund (ERF) (*Carbon Farming Initiative Amendment Bill 2014*), with legislation that passed the Senate while this report was being prepared in November 2014.

The ERF is expanding rather than terminating the CFI. Australia will continue to use Australian Carbon Credit Units (ACCUs), which were established as the national carbon currency under the CFI. The central role of methodologies under the CFI in defining when, where and how ACCUs can be earned will also continue under the ERF, and methodologies available under the CFI are being transitioned to the ERF.

The CFI is not the only regulatory framework for carbon abatement activities in Australia, there is also a significant 'voluntary market', including the National Carbon Offset Standard (NCOS). NCOS includes trade in several eligible offset units, which include ACCUs as well as credits issued under international standards (Gold Standard and Verified Carbon Standard). This report focuses on the CFI/ERF because it has greatest potential for growth and is more likely to yield carbon credit prices high enough to effect landscape-scale change. Many of the general issues traversed in this report are also relevant to the domestic or voluntary market.

Tonnes of carbon dioxide equivalent (t CO_2-e) are the units for ACCUs, with one tonne per credit, which is common to all carbon markets. The CFI covers greenhouse gases, including carbon dioxide (CO_2), methane (NH4) and nitrous oxide (N2O), and the effect of an activity on each particular gas is converted to the common base of CO_2-e by simple multipliers.

For a project to earn ACCUs it must follow an approved methodology. Approved methodologies are listed and described on the Commonwealth Government's web site (http://www.cleanenergyregulator.gov.au/Carbon-Farming-Initiative/methodology-determinations/Pages/default.aspx). The CFI only allows methodologies related to land-based activities. In October 2014 there were 26 methodology determinations available as a basis for CFI projects (Appendix 2).

A register of CFI projects is also available online www.cleanenergyregulator.gov.au/Carbon-Farming-Initiative/Register-of-Offsets-Projects/Documents/Register of Offsets Projects.xlsx. This register shows that the majority of ACCUs generated to date were from emission reduction projects, particularly avoidance of emissions from waste, deforestation and savanna fires (Table 1). Dominance of the register by emission avoidance projects may be partly due to the absence of long term obligations for such projects to store captured greenhouse gases permanently, as well as the relatively rapid establishment of high volumes of abatement that can be achieved in emissions reduction projects.

Table 1. Summary of registered CFI projects for methodologies (data from www.cleanenergyregulator.gov.au/Carbon-Farming-Initiative/Register-of-Offsets-Projects/Documents/Register of Offsets Projects.xlsx accessed 29th October 2014).

Methodology	Number of projects	ACCUs issued
Capture and Combustion of Methane in Landfill Gas from Legacy Waste	68	5604912
Native Forest Protection (Avoided Deforestation)	27	2180281
Savanna Burning	22	467542
Reforestation and Afforestation	11	350963
Diversion of Legacy Waste to an Alternative Waste Treatment Facility	3	148369
Capture and Combustion of Methane in Landfill Gas from Legacy Waste: Upgrade projects	4	131978
Avoided Emissions from Diverting Legacy Waste through a Composting Alternative Waste Technology	4	85555
Avoided Emissions from Diverting Legacy Waste from Landfill for Process Engineered Fuel Manufacture	1	64103
Destruction of Methane Generated from Manure in Piggeries	7	47237
Reforestation by Environmental or Mallee Plantings—FullCAM	1	23443
Quantifying Carbon Sequestration by Permanent Mallee Plantings using the Reforestation Modelling Tool	3	22573
Enclosed Mechanical Processing and Composting Alternative Waste Treatment	2	21291
Permanent environmental plantings of native species using the CFI reforestation modelling tool	14	430
Human-Induced Regeneration of a Permanent Even-Aged Native Forest	3	0
Native Forest from Managed Regrowth	1	0
Total	**171**	**9148677**

To be effective, projects capturing and/or storing carbon, by growing vegetation for example, must retain stored carbon for a considerable period. The CFI requires permanence of carbon stores for 100 years, but the ERF also provides the option of 25 year permanence. This change may result in greater uptake of sequestration project types. Permanence obligations may also become less off-putting as the regulatory framework for carbon farming matures and becomes more stable.

Note that permanence requirements in the CFI exist alongside the general facility for project owners to terminate projects at any time, provided credits issued are reinstated to the regulator. Objective risks to carbon stocks are also shared between project owners and the regulator. For example, owners of sequestration projects would not be penalised for losing carbon through no fault of their own. The regulator withholds five per cent of credits from sequestration projects as a 'risk of reversal buffer' to protect the whole scheme against project-scale mishaps. A project that loses carbon to a naturally occurring event such as bushfire or pest outbreak must take reasonable action to re-establish carbon stores, but is not required to relinquish credits.

As well as transitioning existing CFI methodologies, the ERF will expand the scope of activities to include additional economic sectors, not just land-based activities. For example, titles of ERF methodology proposals that have been posted for public comment include "Coal mining", "Commercial building energy efficiency", "Transport", "Alternative wastewater treatment", "Industrial fuel and energy efficiency", and "Facilities". These are all based on emissions avoidance, which may indicate that the relatively small contribution made by vegetation-based sequestration projects to Australia's current register of carbon projects is unlikely to grow markedly. Despite their relatively small proportional contribution to ACCUs, land-based projects, and especially vegetation-based projects, do have a very high profile in the Commonwealth's information on carbon farming policy and the Direct Action Plan, and also have significant potential to impact Australia's landscapes and rural culture if they take hold.

The relatively high profile of vegetation-based activities under carbon farming may partly reflect their potential to affect land-use, and especially to yield environmental benefits. Emissions avoidance by changing waste management procedures for land-fill or piggery effluent involves a technological shift but not necessarily any substantial change in land-use. Whereas avoided deforestation or establishment of large-scale sequestration projects using mixed species environmental plantings or regrowth trees do imply changed land-use, and significant uptake could change some landscapes too.

1.2 Methodologies and landscape change in eastern Australia

Opportunities for carbon farming differ markedly between regions depending on land-use. In regions predominantly reliant on extensive and relatively natural environments, such as Australia's northern rangelands, the most suitable carbon farming approaches are also extensive and involve relatively subtle changes over large areas. This is because carbon stocks in all major natural carbon pools (vegetation above and below ground, woody debris, and soil) are often close to their potential in these landscapes. carbon farming is also extensive and involves relatively subtle changes over large areas. Examples of carbon farming activities well suited to extensive land-uses in intact natural systems include methodologies for avoidance of emissions from savanna burning and potential methodologies based on biomass increases through fire or grazing change.

In more modified landscapes, clearing of native forests and woodlands has reduced carbon stocks in vegetation and woody debris to levels often far below their potential, and the cultivation of soils has greatly reduced soil carbon. Large differences between current and potential carbon stocks, created by a history of more intensive land management, indicate greater potential for readily verifiable increases in carbon stocks to earn carbon credits in these systems.

Opportunities for land-based carbon farming in eastern Australia are concentrated in the more modified parts of the landscape, where current, or likely future management can change and cause increases in carbon stored in vegetation, woody debris and soils. Eastern Australia does support some extensive areas of relatively natural environments. However, in most such cases there is limited risk to natural carbon stocks from deliberate and legal actions (e.g. deforestation), or sustained emissions related to local management (e.g. extensive annual wildfires). Remnant areas of native vegetation are typically protected from clearing in eastern Australia and extensive wildfires are usually prevented as far as possible.

If an activity is largely prescribed by law, such as not clearing remnant forest, then logically there should not be scope to earn carbon credits from that activity[2]. The activity would not yield additional benefit. So flexibility of future management is another reason that opportunities for carbon farming are focused on the more modified parts of east Australia landscapes and on activities that increase carbon stocks in vegetation, woody debris and soils.

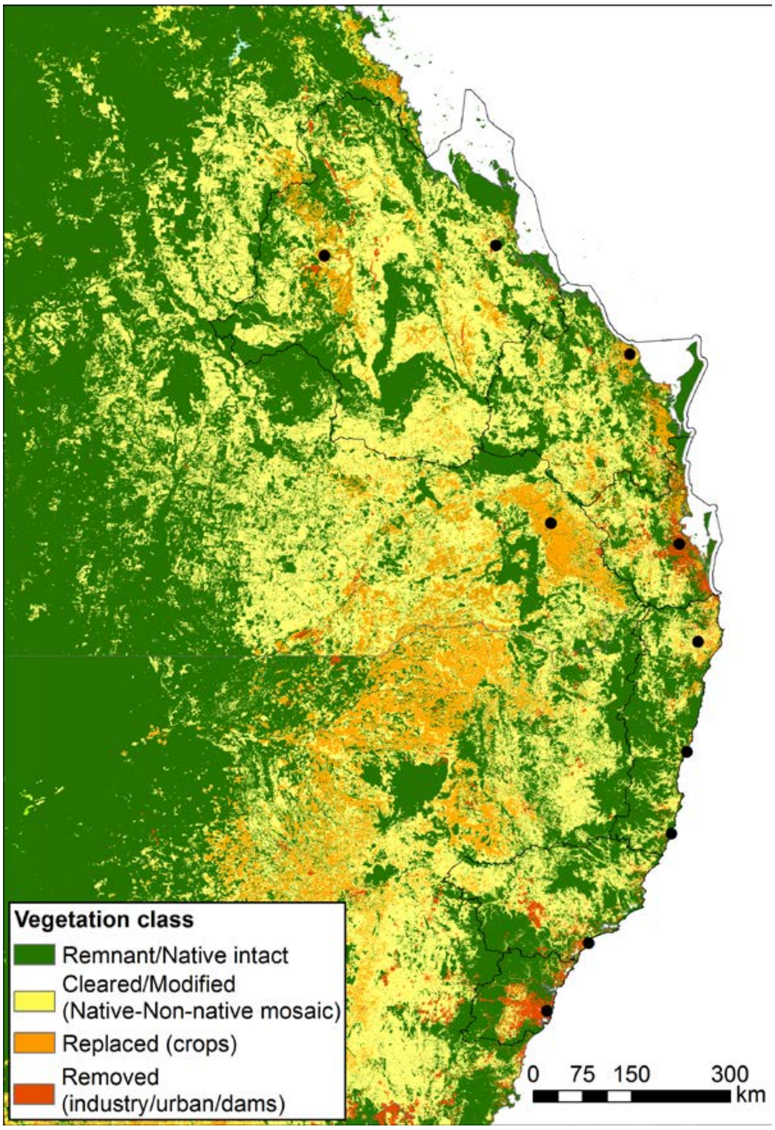
There are advocates for stewardship payments or other incentives for management to help maintain current carbon stocks in landscapes. Other schemes may offer financial support for stewardship, but stewardship is not carbon farming. Economic incentives from carbon farming are focussed on additional abatement, so there is currently no prospect of income from carbon farming by continuing business as usual.

Where clearing or cultivation pose an imminent threat to carbon stock in vegetation and soil, avoidance of clearing is a well-established activity to earn carbon credits. Clearing in eastern Australia is also concentrated in relatively modified landscapes. For example, in Queensland more than 70 per cent of clearing in recent years has been of regrowth vegetation (Queensland Department of Science, Information Technology, Innovation and the Arts, 2014). The three Queensland NRM regions in our study area amount to about 15 per cent of Queensland but contributed over a third of the extent of vegetation clearing in the State in 2011-12 (Queensland Department of Science, Information Technology, Innovation and the Arts. 2014). There is clearly scope for carbon farming based on avoidance of clearing in Queensland (Native forest protection), but administrative requirements prescribed for avoided deforestation projects under the CFI do not align well with Queensland's regulatory framework for vegetation management (Table 2).

Each methodology under the CFI contains specific eligibility requirements. There are also general eligibility requirements under the CFI, including the need for consent from anyone with an interest in the land on which a project will occur. Also, reforestation projects cannot be established on land that has been illegally cleared or legally cleared in the past seven years (five years if ownership has changed). Some general requirements under the CFI are

[2] Under the CFI, if an activity was required by law in March 2011 then subsequent changes withdrawing that legal requirement, such as new permissions to clear, do not make the activity additional (e.g. not clearing) so it can't be used to earn credits. More permissive regulations don't increase opportunities to earn carbon credits.

changing in the transition to the ERF, for authoritative advice consult the Clean Energy Regulator (http://www.cleanenergyregulator.gov.au/Pages/default.aspx).



Map 2. Vegetation extent across the study area.

New methodologies will continue to be approved for use under the ERF. For example, a methodology proposal for avoided clearing of regrowth was published for public comment in October 2014. Existing methodologies will also continue to be modified. For example, none of the current reforestation/regeneration methodologies account for expected increases in soil carbon. In principle, one or more vegetation-based sequestration methodologies could be modified to include soil carbon. Likewise, vegetation carbon pools are outside the scope of the soil carbon methodology for grazing systems.

The expansion in scope to industrial sectors may also see a change in the restrictions on industrial types of silviculture. The CFI did not incorporate exotic or monoculture plantation forestry, however the wider scope of the ERF, encompassing industry in the broadest sense, does seem likely to be a better fit for industrial approaches to land management.

Table 2. Methodologies for land-based activities approved for carbon farming with potential to affect regional-scale natural resource management (October 2014).

Methodology	Activity description	Brief eligibility	Abatement estimation	Restrictions	Complexity	Applicability
Sequestering carbon in soil in grazing systems	Management changes to increase soil carbon in grazing systems.	Land either permanent pasture or has been cropped for 5 yrs (with grazing) and will be converted to pasture by project.	Inventory to estimate stock change.	Few	Medium	Very broad applicability. Activities include range of grazing changes, pasture cropping, pasture rejuvenation, and conversion of cropping land to permanent pasture.
Environmental plantings	Planting or seeding native species on cleared land to establish forest.	5 years agricultural use. No forest without project.	Model (RMT)	Limited harvest, grazing limitation in early years.	Low	Very broad applicability to cleared land with no regeneration/ regrowth.
Human induced regeneration of a permanent even-aged native forest (2 versions)	Management change allowing native forest establishment and growth from seed and small supressed plants.	"Immaterial" regeneration in 10 years before project because land used for cropping or ongoing grazing; land would remain un-forested in absence of project.	Model (RMT)	Limited harvest, grazing limitation in early years.	Low	Broad applicability limited by reliance on suppressed regeneration potential.

Methodology	Activity description	Brief eligibility	Abatement estimation	Restrictions	Complexity	Applicability
Measurement based methods for new farm forestry plantations	Planting of permanent or perpetual 'for harvest' plantations	5 years agricultural use. No forest without project.	Inventory with allometric validation and model (FullCAM).	Harvest optional	High	Very broad applicability to cleared land with no regeneration/ regrowth
Native forest from managed regrowth	Cessation of cyclic re-clearing of native forest regrowth.	Grazing land with young regrowth (forest potential but not yet >2m tall and >20% cover)	Model (FullCAM)	Limited harvest, light grazing ok	Medium	Broad applicability for young regrowth or suppressed regeneration.
Native Forest Protection	Avoidance of permitted deforestation planned to convert native forest to cropland or grassland.	Permit, issued before 1 July 2010, to clear native forest (forest since pre-1990) to establish permanent grassland or cropland.	Inventory with allometric validation/ development, modelled baseline (FullCAM).	Limited harvest	High	Applicability limited by requirement for an existing permit. Very limited applicability in Queensland where clearing to establish grassland or cropland is regulated but does not involve consent in the form of a permit. Wider applicability in NSW for land subject to Property Vegetation Plans (PVPs).

Methodology	Activity description	Brief eligibility	Abatement estimation	Restrictions	Complexity	Applicability
Quantifying carbon sequestered by permanent plantings of native mallee eucalypt species using the CFI reforestation modelling tool	Establish mallee forest by direct seeding or planting-	Grazing or cropping land with average annual rainfall <600mm.	Model (RMT)	Limited harvest	Low	Not applicable within study area. Broad applicability in southern semi-arid regions.
Reforestation and Afforestation (3 versions)	Establish and maintain trees on agricultural land.	5 years agricultural use. No forest without project.	Full inventory with allometric validation or development.	Limited harvest	High	Very broad applicability to cleared land with no regeneration/regrowth
Reforestation by Environmental or Mallee plantings - FullCAM	Planting or seeding native species on cleared land to establish forest.	Within spatial domain. 5 years agricultural use. No forest without project.	Model (FullCAM) - new parameterisation yielding higher rate than RMT within domain	Limited harvest, grazing limitation in early years.	Medium	Broad applicability within defined spatial domain. Study area intersected but not comprehensively covered by 'temperate' environmental plantings domain.
Savanna burning (2 versions)	Reduction of greenhouse gas emissions through early dry season burning in humid tropics.	Annual rainfall >1000mm in tropical savannah.	Models including SavBAT and maps of vegetation and fire scars.	Area burnt cannot be reduced by increased grazing or extra late-season fires outside project.	Medium	Not applicable within study area. Applies to expansive tropical savannas subject to extensive late dry season fires.

1.3 Prospects

Uptake of new economic opportunities for carbon farming may be influenced by a diverse suite of factors spanning sociology, economics, ecology, climate science, and national and international politics. At the level of individual projects, decisions to start or stop carbon farming will undoubtedly also hinge on personal circumstances and values. The 26 methodologies and 171 registered projects under the CFI indicate considerable industry interest, yet carbon farming is in its infancy in Australia. The CFI was enacted in 2011 following a decade of turbulent policy around carbon offsets. Uptake can be expected to grow, provided economic and ecological barriers are not prohibitive. Even if it is economically rational, adoption of carbon farming by a majority of potential participants would likely require decades of sustained positive performance.

If economic benefits are forthcoming, land-use changes from carbon farming could include a range of potential co-benefits and dis-benefits, including:

- socioeconomic change
- altered hydrology
- changed fuel loads for fire
- biodiversity change (local additions and removals of species and habitats)
- altered connectivity for biodiversity, pests and fire
- changed regional and micro climates

Broad applicability has been a policy goal for methodology development under the CFI. With the exception of the methodology for native forest protection, the extent of potential application each of the methodologies in Table 2 runs to tens of millions of hectares. Applicability of the methodology for native forest protection is strongly constrained by its requirement for a permit pre-dating April 2010. Even with this restriction the extent of land meeting requirements in NSW alone may run to millions of hectares. There is also a new methodology proposal for native forest protection that may be applicable to Queensland. The total area with potential for native forest protection activities in Queensland could also run to millions of hectares (Queensland Department of Science, Information Technology, Innovation and the Arts, 2014).

Potentially attainable abatement from changes to forest management in Queensland was estimated at around 105 million t CO_2-e per year (Eady et al. 2009). This estimate of attainable annual credit generation is ten times the total credits generated under the CFI in its first three years. The 105 Mt estimate was based on economic modelling to identify eligible areas (Polglase 2009).

Conceptually, the economic prospects for carbon farming methodologies can be modelled as the net outcome of returns from carbon credits against losses from foregone opportunities and project costs. Estimation of returns from carbon credits involves both economic and biological inputs and is the topic of the second section in this report.

1.4 Conclusion

Carbon farming in Australia revolves around a regulatory and policy framework administered by the Commonwealth Government.

At the time of writing, the framework was transitioning from the Carbon Farming Initiative (CFI) to the Emissions Reduction Fund (ERF). That change will cause some variation in process but the over-arching principles of the CFI, and the activities available under the CFI, are expected to be maintained into the ERF.

Carbon credits can also be generated and traded under voluntary markets subject to other national and international guidelines.

The CFI/ERF, following established convention, only yields carbon credits to projects that follow an approved methodology.

After three years, the CFI has 26 methodology determinations available for use, and 171 registered projects have generated nine million Australian Carbon Credit Units (ACCUs, with 1 ACCU representing abatement of 1 tonne of CO_2-equivalent abatement). For comparison, Australia's emissions in 2012 were 559 Mt. CO_2-e, including 15 Mt from land use change and forestry (Department of the Environment 2014).

Most ACCUs generated to date have been via methodologies for avoiding emissions, particularly from landfill methane and native forest clearing.

Nine existing CFI methodologies cover activities with potential to impact significantly on landscapes. Activities covered by these methodologies include establishment of new forests (by planting, seeding or enhancing natural regeneration), native forest protection, savanna burning and soil carbon accumulation in grazing systems.

Methodologies to cover more activities are proposed, including expanded scope for protection of regrowth forests.

There is substantial scope for growth in carbon farming activities, but uptake may be influenced by complex interactions between social, economic and ecological factors.

2 Economics of carbon farming projects

Carbon farming projects earn credits for the difference between the greenhouse gas outcomes from the registered project and the relevant baseline. Baselines are essentially projections of greenhouse gas changes (emissions and sequestration) under business as usual. For example, a baseline for a forest protection project would reflect the emissions from clearing that would have occurred had the carbon farming project not gone ahead.

Carbon stocks in live plants and debris in mature native forests in eastern Australia are typically around 200-700 t.CO_2-e/ha. So clearing one hectare of forest to derive permanent grassland for pasture would emit around 200-700 t.CO_2-e. This emission would occur over years to decades depending on how quickly residual woody debris from clearing decayed or burned. The CFI requires credits from avoided deforestation projects to be spread over 20 years, so an east Australian forest protection project might report abatement in the order of 10 to 35 t. CO_2-e per hectare per year.

Determining whether 10 or 35 carbon credits per hectare per year is sufficient to warrant a shift from grazing to tree farming (for example), requires consideration of numerous factors including:

- likely value of carbon credits
- likely value of opportunities and cost foregone by not clearing
  - opportunity for revenue forgone from ongoing use and not increasing pasture productivity
  - cost-saving from not clearing
  - cost-saving from not needing ongoing tree suppression
- discount rate for value of future income in today's dollars
- costs of project establishment
  - legal support
  - project registration requirements
  - technical support (surveys, maps, forest inventory)
  - additional fencing, water infrastructure etc.
- running costs for project
  - additional fire management
  - additional weed management
  - monitoring
- reporting costs
  - data collection and processing
  - report writing
  - report audit requirements
- brokerage and accounting cost for credit disposal

Some of these factors are strongly dependent upon the methodology being applied, such as costs to establish, manage and report on the project. Others are highly specific to the landholder's business and values, particularly costs and benefits foregone, inherent value of project co-benefits such as biodiversity maintenance and salinity risk avoidance, and even the discount rate most appropriate for future income. Similarly, landholder values are pivotal to the impact of permanence requirements on willingness to participate. Perhaps most importantly the likely value of carbon credits is currently highly uncertain.

This chapter reports on an economic analysis of two types of activity that generate credits by establishing native forests: environmental plantings, and; reforestation based on changed management of natural regeneration (i.e. regrowth). Assessment is also made of potential for profit from avoided deforestation.

Polglase *et al.* (2011; 2013) analysed prospects for carbon forestry plantations and very clearly demonstrated the sensitivity of model results to variation within the plausible ranges of discount rates, carbon prices, establishment costs, and methods of forest carbon modelling. Their analysis considered a suite of scenarios, which were arguably all plausible, and suggested that the extent of cleared land where carbon plantations might be profitable in Australia ranged from zero to nearly 100 million hectares, depending on assumptions. Their modelling was spatially explicit, based on a 1km grid across Australia. The relative regional distribution of prospective areas was more stable across the various scenarios than the total extent that appeared potentially profitable. Eastern Australia contained several hotspots in many scenarios; particularly through the belt of subcoastal country from the Fitzroy Basin in the north and running south through the Dawson and upper Burnett, to the western slopes of the Great Dividing Range in northern NSW (Polglase *et al.* 2011).

Similarly, a recent assessment of prospects for reforestation projects using regrowth or plantings in Queensland (Evans et al. submitted) identified a concentration of land with relatively low carbon credit prices required to enable viable reforestation projects in east-central Queensland.

For this report we applied a methodology established by Evans *et al.* (submitted) to calculate costs and benefits, in terms of present-day dollar values, associated with carbon farming into the future. The study area in this section goes beyond the six natural resource management regions in eastern Australia to cover all of Queensland and New South Wales. The main aim of the analysis was to provide some guidance about the extent and location of land on which it may be economically positive to develop carbon farming projects under a range of prices for carbon credits, and how those variables depend on other economic factors such as management costs.

2.1 Method for economic analysis

The aim of this economic analysis is to estimate the order of magnitude of vegetation-based carbon farming activity that may potentially occur, and to consider where, at regional scale, the most prospective locations for the various carbon farming activities may be assessed.

The analysis involved three dimensions; suitability, costs and benefits. These were quantified for each cell in a 0.01 degree grid (~1km^2) covering New South Wales and Queensland using spatial data and attributes listed in Table 3. Costs and benefits were used to calculate the ACCU price at which each type of carbon farming would break-even after 10, 25 and 100 years (i.e. price at which net present value = zero at those time horizons) for each cell of suitable land. Suitability was assessed against spatial data for land-use and vegetation extent and used to estimate the area over which particular activities may be profitable, and to identify regions in which particular carbon farming activities tend to be profitable too.

Costs

Costs included estimates for: on-ground management, plantation establishment (environmental plantings only), project administration, reporting and compliance. These costs were estimated based on literature searches and expert consultation, particularly the work by Evans et al. (submitted). Cost also included opportunity costs, from foregone agricultural activity, based on a spatial dataset developed for profit at full equity for Australian agricultural enterprises (Marinoni et al. 2012).

The costs outlined in Table 3 reflect a project scale that may be similar to a small to medium sized environmental planting project, with annual costs for maintenance and paperwork running to $65/ha, and $100/ha for paperwork at project establishment. Larger scale projects are expected to have lower costs per unit area. To assess the potential increase in economic viability that may arise from economies of scale two other costs scenarios were tested for plantings and regrowth:

1) a moderate cost scenario with annual fees of $5/ha/year for maintenance and ongoing paperwork, plus the same $100/ha establishment fee, and;
2) a low cost scenario with $5/ha/year running costs and $10/ha for administrative costs at project establishment.

Table 3. Parameters for economic analysis. See Table 4 for more detail on data sources.

Suitability	Cost	Benefit
S1. Land-use = grazing or dryland cropping or dryland horticulture (cropping only suited to environmental plantings), i.e. Land use not irrigated cropping/horticulture, forestry, conservation, industrial, mining, urban, or water. S2. Native vegetation extent map: remnant/intact native vegetation (Avoided deforestation), or not (Regrowth and Environmental plantings).	C1. Yearly Opportunity cost = Profit at full equity 2005/6 data from Marinoni et al. (2012) adjusted to current value assuming 2.7% annual inflation C2. Yearly management cost (includes on-ground and administration) a) $65/ha/yr default (both plantings and regrowth) b) $5/ha/yr for moderate and low cost scenarios C3. One-off $2000/ha establishment cost for plantings and avoided deforestation (involves detailed carbon inventory and allometric equation development) C5. One-off project establishment cost of (contracts etc.) a) $100/ha default and moderate cost scenario b) $10/ha low cost scenario	B1. Carbon sequestration: rate estimated by emulation of FullCAM (assumed slightly slower growth for regrowth). B2. Future benefits (carbon credits x carbon price) discounted at 5% to estimate present value.

Economic analysis requires numerous assumptions about costs and benefits. Previous economic research has demonstrated substantial sensitivity in the outcomes of economic models of carbon farming to such assumptions about costs, carbon prices and discount rates. We focussed our relatively limited assessment of model sensitivity on variation in costs. This is justified because: sensitivity to price is directly addressed by evaluating price-to-break-even; sensitivity to discount rate is already well established by previous work (Polglase et al. 2011; Evans et al. submitted), and; cost is sensitive to project scale which, in turn, is highly relevant to the potential to influence natural resource management at landscape scale.

As well as fixed project costs, the key cost variable is opportunity cost. For this analysis an estimate of profit at full equity for agricultural enterprises was used as opportunity cost (Marinoni et al. 2012, Map 3). Marinoni et al. (2012) derived their estimate from a range of data, primarily from ABARE, for the 2005/2006 financial year.

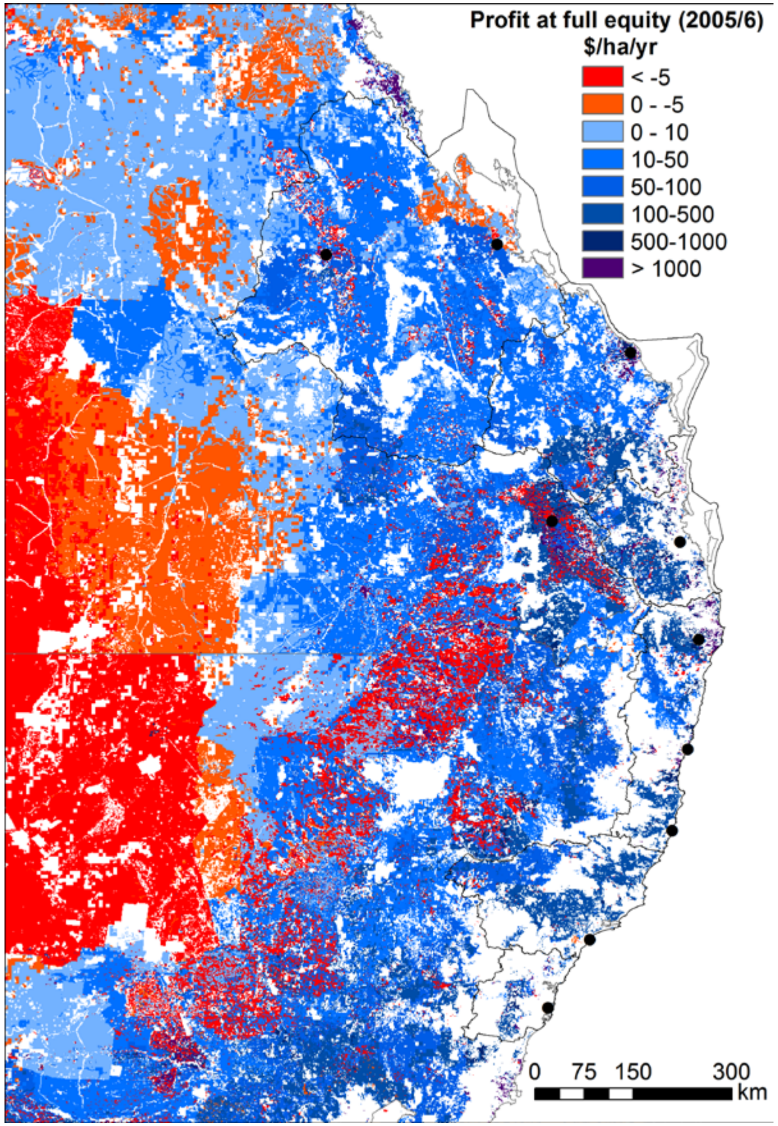
The 2005/6 financial year was not a particularly good year for agriculture. Across eastern Australia a dry year in 2004 was followed by average to slightly below average rainfall in 2005 and below average rainfall in 2006 (http://www.bom.gov.au/climate/annual_sum/2005/page12.pdf & http://www.bom.gov.au/climate/annual_sum/2006/page12.pdf). This weather scenario implies that the estimate may be relatively low, which would tend to make carbon farming look more profitable than it should. The concentration of negative profit at full equity in major grain growing regions across eastern Australia in Map 3 (including the Darling Downs, Queensland's central highlands, the Moree Plain, and the Liverpool Plain) certainly suggests that the profit at full equity estimate may reflect austere times. However, even with this limitation, an estimate of profit at full equity is conceptually more suitable as an estimate of opportunity cost than the value of land as has been used in other studies.

Suitability

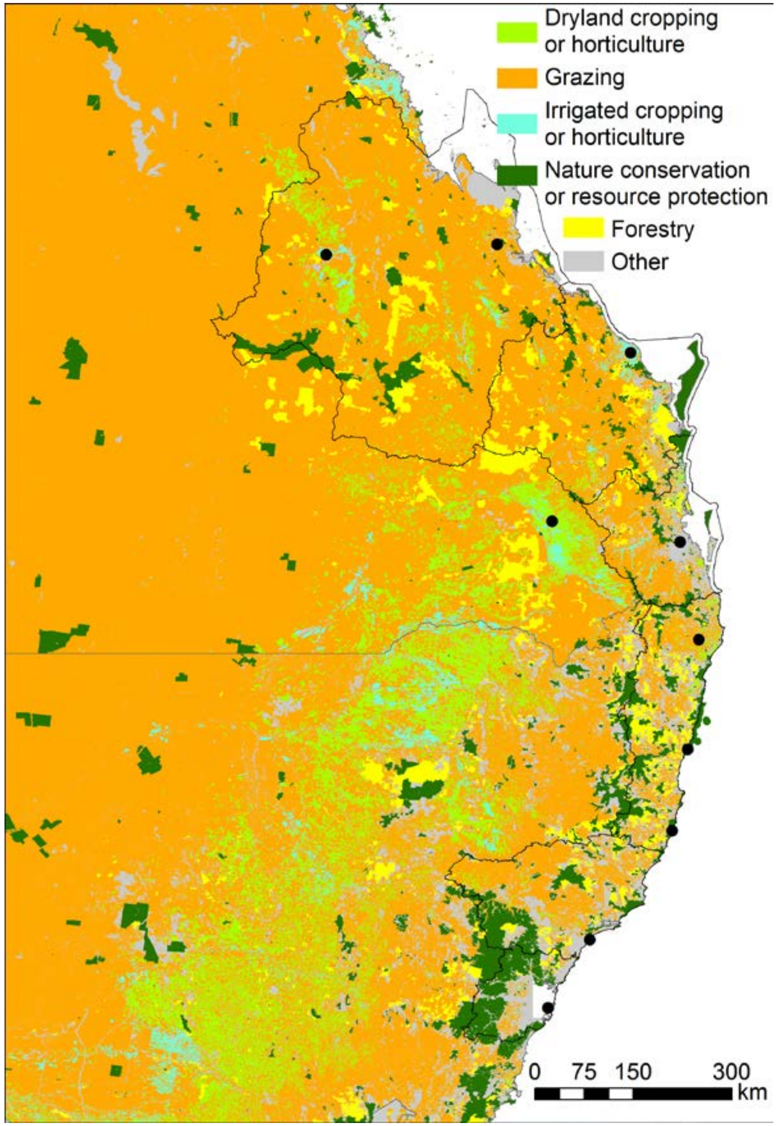
Land potentially suited to carbon farming activities was identified using spatial data on vegetation condition and land-use. Principal sources are described in Table 4.

Table 4. Key data sources for identification of land suited to carbon farming using regrowth, environmental plantings or avoided deforestation.

	Queensland	New South Wales
Vegetation condition	Regional Ecosystem mapping (v8.0)	State of the catchments vegetation extent (2008)
Land-use	Queensland land use (QLUMP) 2009	Land Use: New South Wales (v1.1) 2007
Other	Analysis also constrained by extent of data for profit at full equity (limited to agricultural lands).	



Map 3. Profit at full equity for agricultural lands in 2005/6 across central-eastern Australia. Estimated by Marinoni et al. (2012).



Map 4. Generalised land-use in central-eastern Australia.

The extent of potentially suitable land in Queensland and New South Wales varied between activities. Grazing is the most common land use in the study area (Map 4) and is broadly compatible with regrowth plantings or avoided deforestation. Land with intact or remnant native vegetation was required for avoided deforestation but excluded for regrowth and plantings. All activities were also excluded from irrigated land for cropping or horticulture, and mining, urban and other intensive land uses. Dryland cropping and horticulture was included in analysis of plantings but not regrowth. These exclusions are consistent with the requirements of the applicable methodologies, but the regional scale of the analysis undoubtedly glosses over local complexities in landuse and the options for carbon farming that may be suited.

Note that no attempt was made to identify land with current regrowth, within the broad extent of lands with appropriate land-use and cleared native vegetation. This is primarily because available methodologies for regrowth require relatively young regenerating forest, which in the authors experience is extremely difficult to map with any reliability. Similarly local constraints on suitability for plantings, such as site access and water availability, were not considered. The results are not recommended for property scale use.

Perhaps the most significant obvious disconnect between this analysis and reality was for the extent of land potentially suited to avoided deforestation projects. Prospects for application of the current methodology for avoided deforestation to Queensland are extremely restricted by the requirement for a permit to clear vegetation, issued before July 2010, which specifies that the clearing must result in permanent pasture or crop land. This requirement has been achievable under the regulatory framework in New South Wales. However, in Queensland the regulatory framework for most agricultural vegetation-clearing is applied through maps, identifying go and no-go zones, but rarely involves issuing permits to landholders. Without an historical document indicating intent to clear, it is hard to argue that the right to clear vegetation equals the intent to clear that vegetation. Therefore, either methodologies must be narrowly constrained to situations where clearing is clearly business as usual (e.g. regrowth from clearing in recent decade), or different approaches need to be applied to estimate the fraction of potential abatement that is actually additional when protecting a native forest where clearing is allowed but an historical permit (pre-dating carbon farming uptake) does not exist.

Biomass and carbon

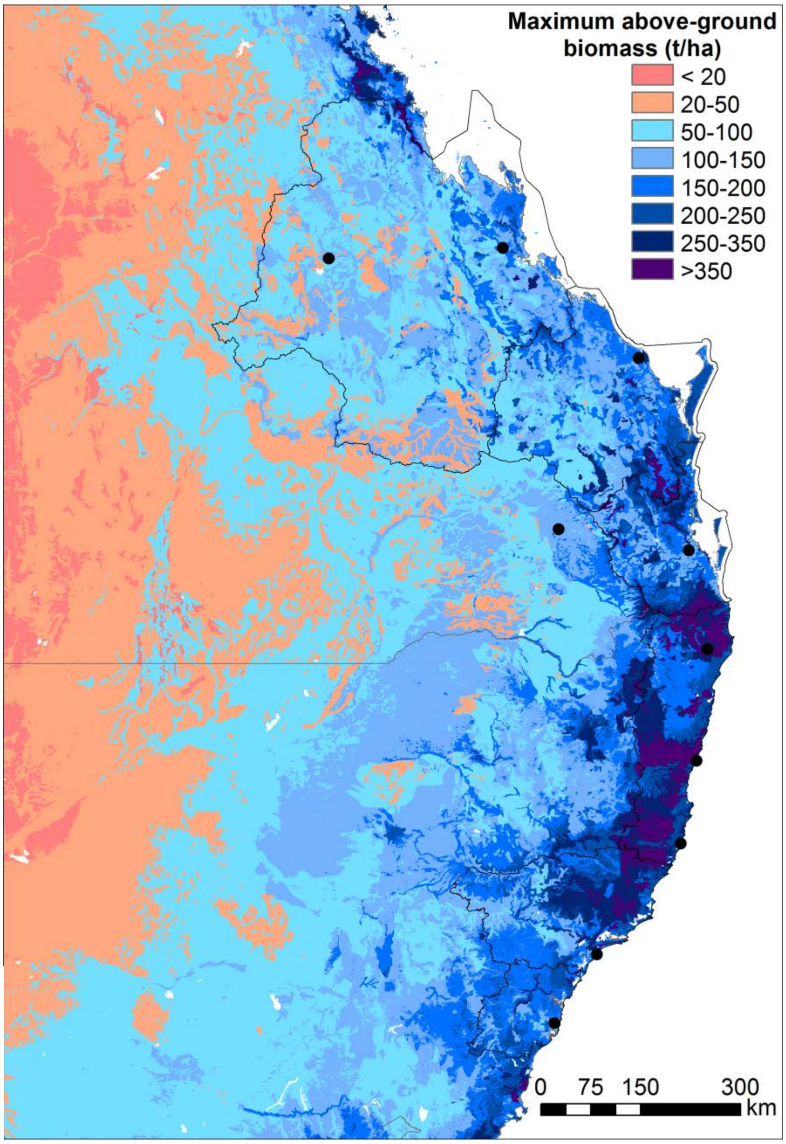
The data and techniques used to model carbon benefits from carbon farming activities in this report are consistent with approaches under Australia's National Greenhouse Gas Inventory (Department of the Environment 2014). This is most appropriate because carbon farming methodologies are required to be consistent with the National Greenhouse Gas Inventory. The spatial patterns in carbon dynamics for native vegetation in the National Greenhouse Gas Inventory are driven by a spatial model of forest productivity (Kesteven et al. 2004). The same spatial model of forest productivity was used for this analysis but the model of temporal changes in carbon stock was a slightly simplified version of the method used for the National Greenhouse Gas Inventory and for carbon farming projects (i.e. FullCAM).

Forest carbon stocks were modelled for each cell in the one kilometre grid as a function of time since hypothetical forest establishment and the maximum above-ground biomass in native forests for each cell (i.e. from the national forest productivity model, Map 5, Eq. 1). This approach was developed by Richards and Brack (2004) and remains the basis of modelling carbon dynamics involving native forests in Australia's National Greenhouse Gas Inventory.

The model calculates above-ground biomass (*B*) a number of years (*a*) after forest establishment:

$$B_a = M.e^{(-k/a)} \quad \text{Eq. 1}$$

Where *M* is maximum above-ground biomass and *k* is a parameter that determines the age of maximum biomass increment and *e* is Euler's number (2.7183). *k* was set at 20 for plantations and 24 for regrowth, corresponding to maximum biomass 10 and 12 years after establishment (Richards and Brack 2004). We added biomass for roots, equal to 25 per cent of above-ground biomass, and converted biomass to carbon dioxide equivalent units by multiplying by 0.5 (a standard ratio for converting biomass to carbon) and then by 3.67 (the ratio of the molecular mass of CO_2 to the atomic mass of carbon is 44:12). This formulation ignores debris, and is therefore somewhat conservative.



Map 5. Model of potential above ground native forest biomass model for central-east Australia, as used in calculation of Australia's National Carbon Account for land-use, land-use change and forestry.

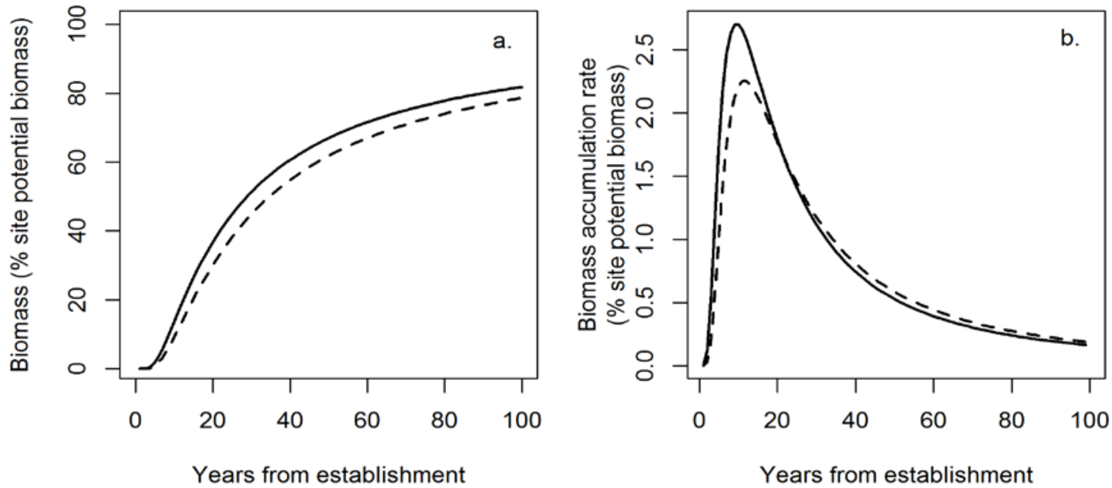


Figure 1. Model of biomass dynamics in developing regrowth (dashed lines) and planting (solid lines): a. Total biomass, and; b. the rate of biomass accumulation.

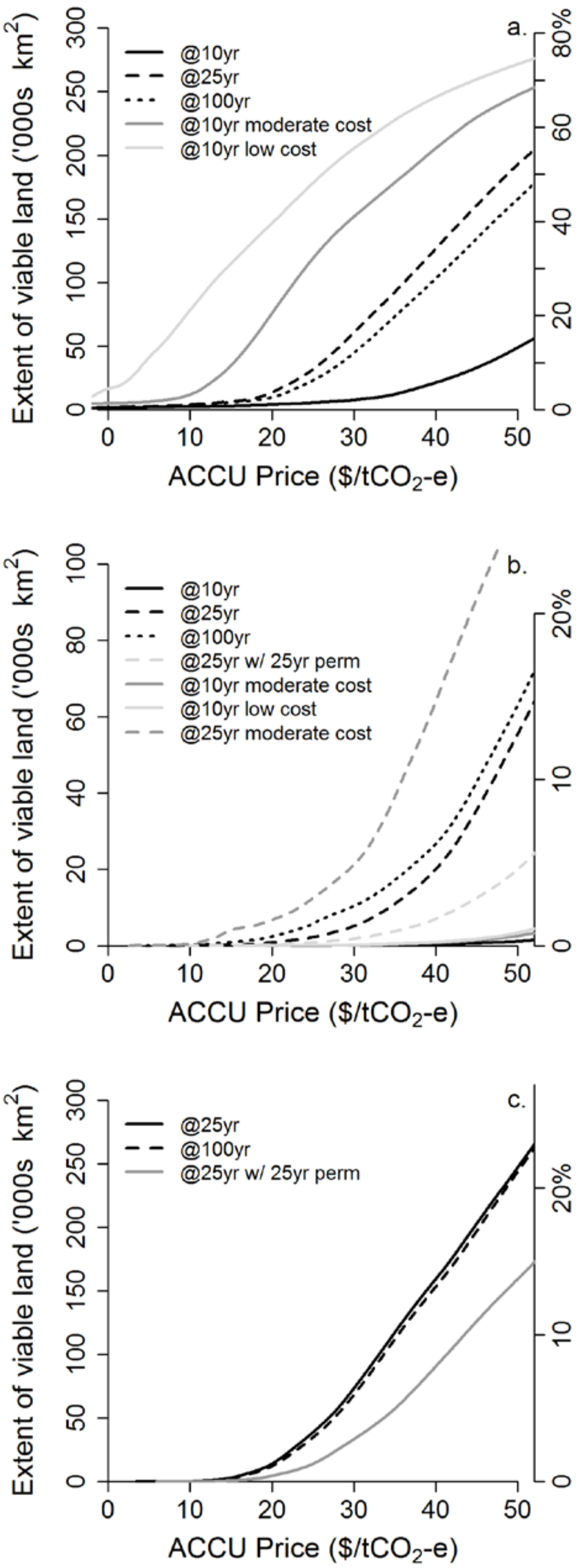


Figure 2 Models of relationships between credit price and the extent of viable land for different carbon farming scenarios in Queensland and New South Wales. a. managed regrowth; b. environmental plantings, and; c. avoided deforestation. Note variation in scales of y-axes. Secondary y-axis shows percentage of land with suitable landuse and vegetation status.

A key point about the rate of carbon accumulation into regrowth or planted forests is that it changes as the forest develops (Fig. 1). The rate of forest growth typically increases with age in very young forests, peaks in the second decade or so, and subsequently declines with age (Fig. 1b).

Note that the models used to predict carbon benefits from new forest in this analysis assume ongoing average forest productivity reflected in the model of potential biomass (Map 5). Actual accounting for carbon sequestration in carbon farming projects (and the National Greenhouse Accounts) will produce rate of carbon gain that reflect variation in rainfall.

Regrowth biomass gains were modelled as if the regrowth only commenced at project commencement, as it does for plantings. However, this is a somewhat conservative assumption because project establishment under available methodologies requires regrowth to be sufficiently advanced to have potential to form native forest. It is likely that most regrowth projects will begin with a young forest around five to ten years old.

Carbon abatement estimates for avoided deforestation were also based on the potential biomass dataset. Annual carbon abatement for avoided deforestation was equal to five per cent of the land's potential biomass, because the methodology for avoided deforestation spreads carbon credits evenly over a 20 year crediting period.

2.2 Results and Discussion

The prices at which large areas of Queensland and New South Wales might support profitable carbon farming differs between activities (managed regrowth, environmental plantings and avoided deforestation), all show rapid increases in viable area for vegetation-based carbon farming as credit prices rise above \$20-\$30, and all show limited prospects at prices less than ten dollars per tonne of abatement (Fig. 2).

The extent of land where carbon farming projects were likely to break-even within ten years of project establishment was low for regrowth, even up to $40 per tonne CO_2-e. But it was far lower for plantings (note the difference in y-axis scale between plots in Fig. 2). This directly reflects the assumed difference in on-ground establishment costs for plantings. All other establishment and ongoing costs were the same for regrowth and plantings, and the models for carbon yields from plantings gave slightly higher sequestration rates for plantings than regrowth. Changes in investment timeframe influenced the extent of land on which each of the carbon farming projects might be economically viable.

The proportion of generally suitable land on which projects might break-even over ten years was low under the planting and regrowth scenarios assessed, but much more land was viable over 25 or 100 year timeframes (Fig. 2a & 2b, Maps 6-9).

Reducing annual costs per hectare, as may be achieved with large-scale projects or by project aggregation, greatly increases the extent of land potentially viable within ten years for regrowth projects (Fig. 2a). Lower ongoing costs had a similarly positive relative-impact on viable area for planting. However, in absolute terms, the curves for break-even price after ten years in plantings, including low and moderate cost scenarios, are barely distinguishable from the x-axis in Fig. 2b. Reductions in ongoing costs for plantings have a more noticeable impact on viable area if longer investment timeframes are considered (Table 5).

The strong influence of timeframes on the price needed for projects to break-even highlights the sensitivity inherent in these results to the discount rate applied (Polglase et al. 2011; Evans et al. submitted). The rate of five percent applied here is moderate, comparable to returns that might be expected from relatively low risk investments. Higher discount rates (indicative of higher expected returns on capital invested) would therefore result in fewer areas appearing viable for carbon farming with environmental plantings unless the establishment cost could be brought well below the $2000/ha assumed here.

Opting for 25 year permanence, with the resulting 20 per cent deduction of credits, increased the price required to make environmental plantings viable over a given area by about 50 per cent (Fig. 2b). Avoided deforestation models show lower sensitivity. For a given carbon price the potentially viable area under 25 year permanence is equal to roughly 70 per cent of viable area with 100 year permanence.

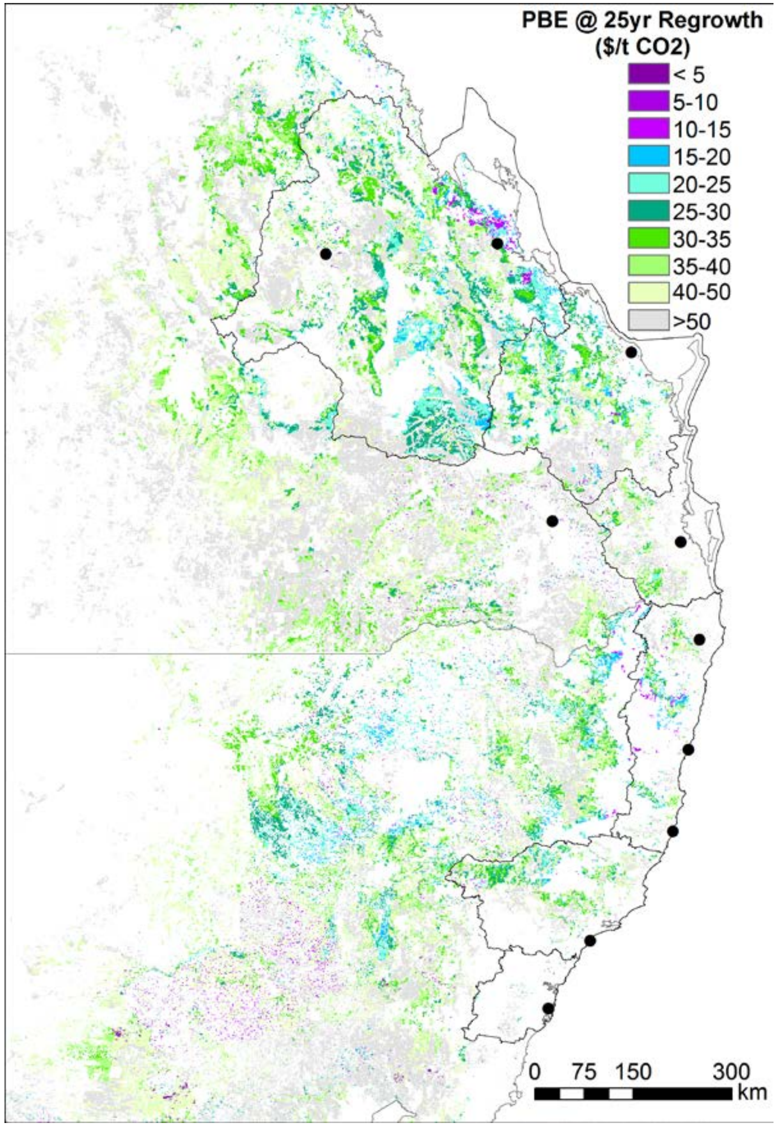
In spatial terms, the six natural resource management regions targeted in this study represent a significant fraction of the land in eastern Australia with potential for viable carbon farming under the scenarios assessed.

Spatial pattern in economic prospectivity is broadly similar for regrowth and planting activities because they use the same underlying spatial models of native forest biomass potential and agricultural profit at full equity. Avoided deforestation differs from regrowth and plantings because it requires more or less intact forest managed for grazing, rather than cleared land.

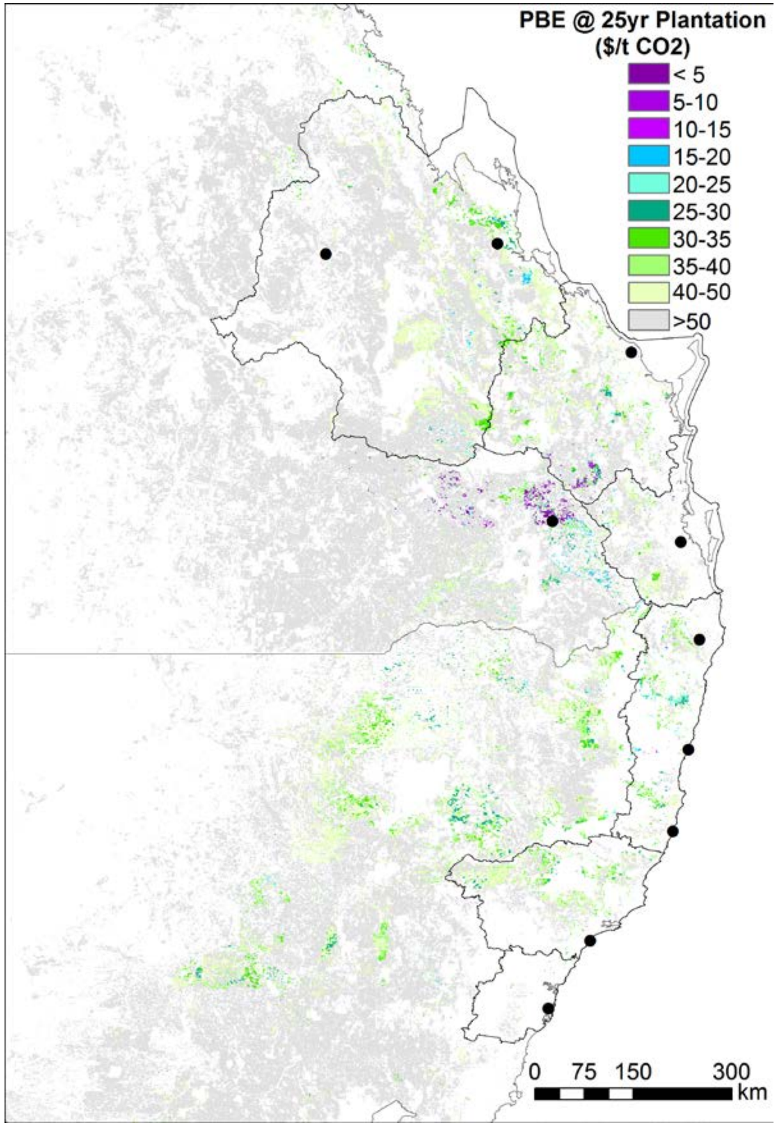
Table 5 provides indicative data on the potential extent of land in New South Wales and Queensland that may be economically viable after 25 years for various carbon farming scenarios and prices from $15 to $30 per ACCU. The Table also contains estimates of the amount of sequestration that could theoretically be achieved over 10 and 20 years if all viable land was used for carbon farming. Of-course given the complexity, uncertainty and slow returns on investment in carbon farming as it is currently structured, uptake is likely to be a small fraction of potential at any price. The data provide an upper limit for likely potential.

Table 5.Indicative figures for potential magnitude of sequestration for land in New South Wales and Queensland during the first ten years of carbon farming where current analysis suggests economic viability over a 25 year investment horizon given credit prices of $15, $20 or $30 per ACCU (i.e. price to break-even < ACCU price).

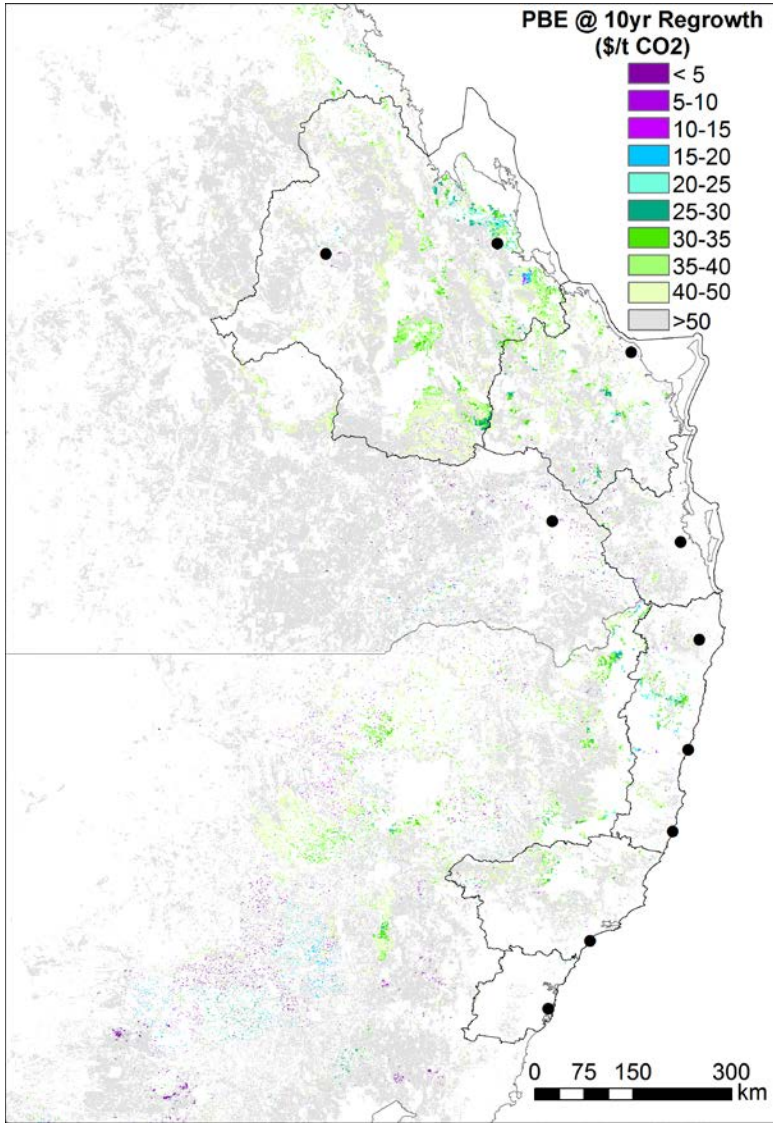
Scenario	Extent viable ('000s km^2)			Potential CO_2-e sequestered over 10 years (Mt CO_2-e)			Potential rate of ongoing sequestration after 10 years (Mt CO_2-e /year)			Potential CO2-e sequestered over 20 years (Mt CO_2-e)			Potential rate of ongoing sequestration after 20 years (Mt CO_2-e /year)		
	$15	$20	$30	$15	$20	$30	$15	$20	$30	$15	$20	$30	$15	$20	$30
Regrowth	6	14	61	19	45	184	4	11	43	62	149	609	4	9	37
Regrowth – moderate ongoing costs	151	199	262	280	375	505	66	88	118	930	1245	1678	57	76	103
Regrowth – low ongoing costs	183	224	274	327	412	523	76	96	123	1085	1368	1738	66	84	106
Planting	0.2	0.8	5	2	6	33	0.3	1	7	4	17	90	0.2	1	5
Planting with 25 yr permanence	0.1	0.3	2	0.8	2	13	0.2	0.4	3	2	6	34	0.1	0.3	2
Planting – moderate ongoing costs	4	7	21	14	27	104	3	5	21	37	73	283	2	4	14
Avoided deforestation	3	14	74	86	316	1153	9	32	115	173	632	2306	9	32	115
Avoided deforestation – 25 year permanence	0.9	4.6	33	29	124	633	3	12	63	58	248	1267	3	12	63



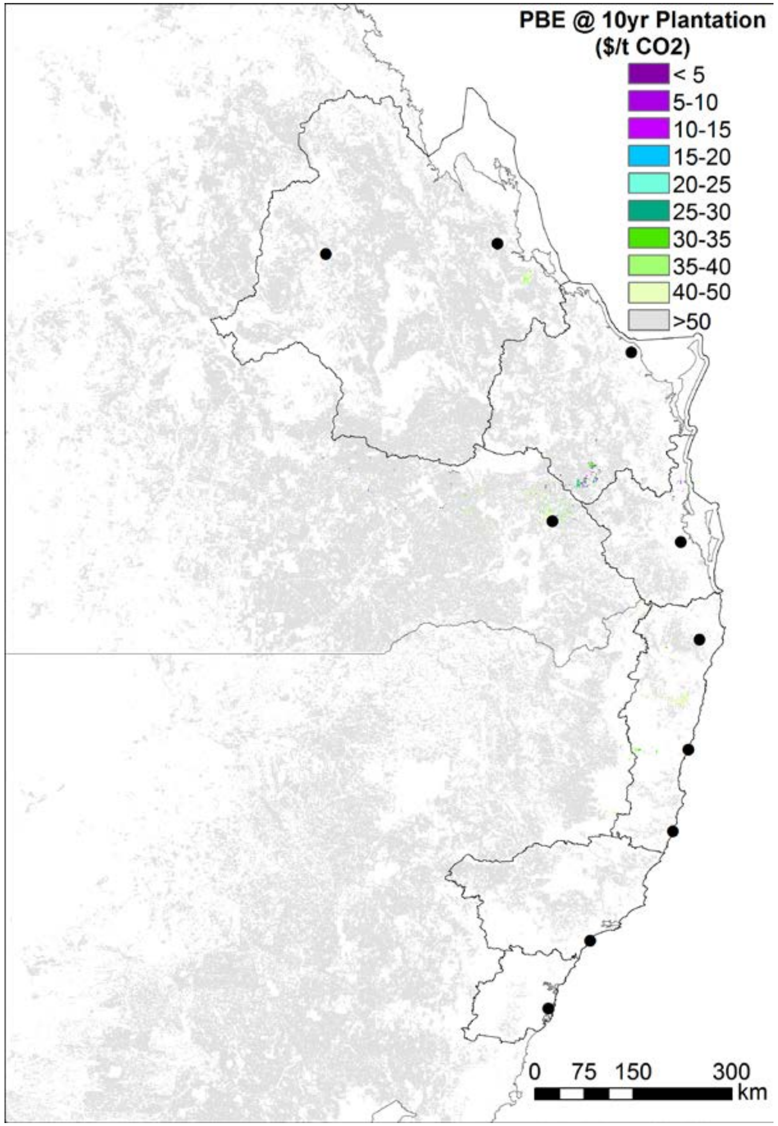
Map 6. ACCU price at which carbon farming projects may break-even over a 25 year investment period for native forests from managed regrowth.



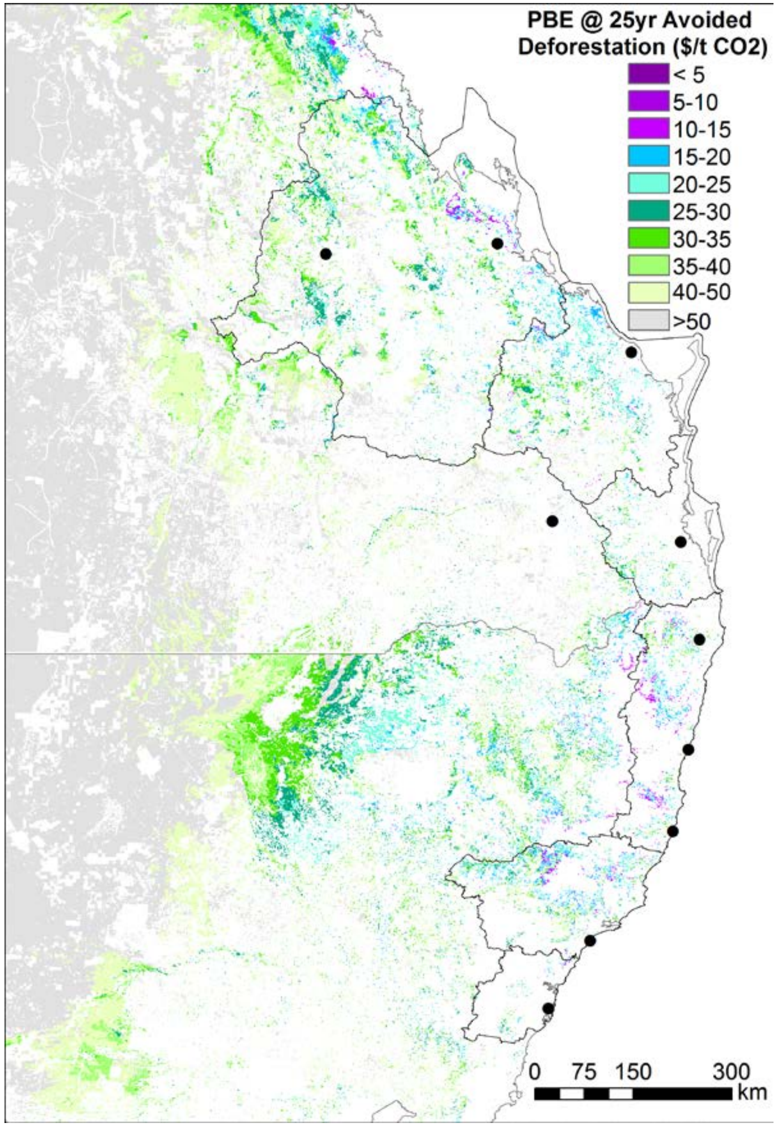
Map 7. ACCU price at which carbon farming projects may break-even over a 25 year investment period for environmental planting.



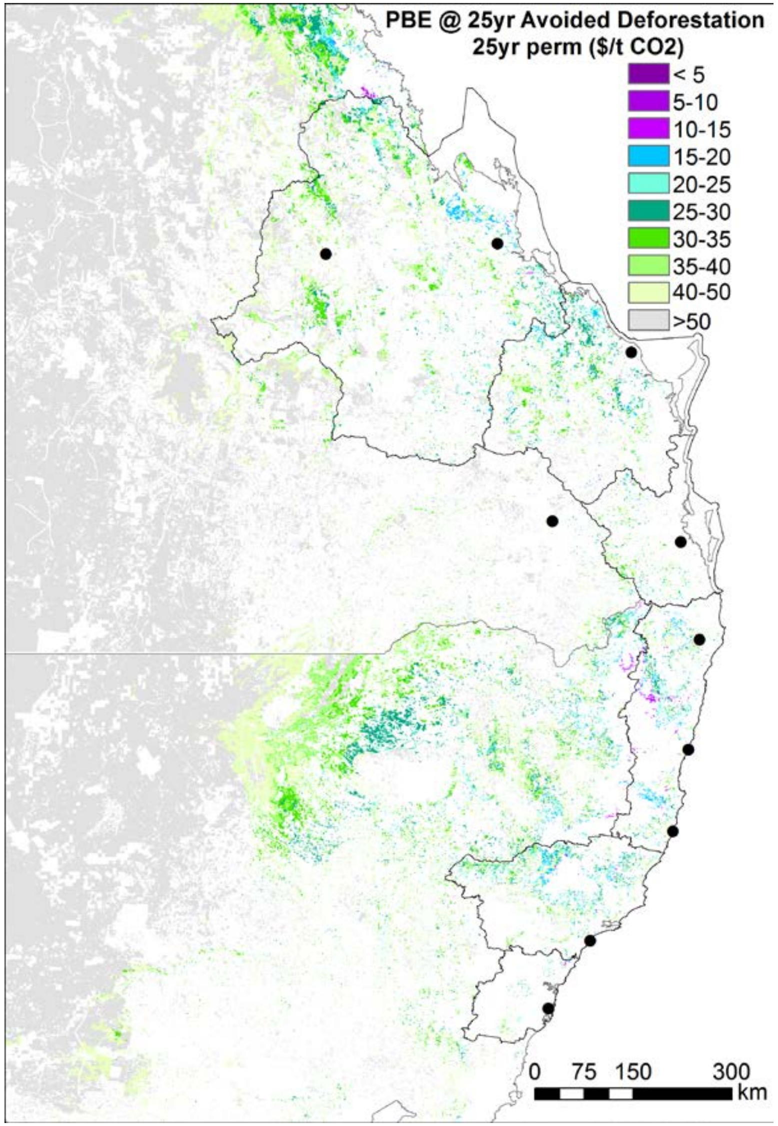
Map 8. ACCU price at which carbon farming projects may break-even over a 10 year investment period for regrowth.



Map 9. ACCU price at which carbon farming projects may break-even over a 10 year investment period for environmental plantings.



Map 10. ACCU price at which carbon farming projects may break-even over a 25 year investment period for avoided deforestation with 100 year permanence.



Map 11. ACCU price at which carbon farming projects may break-even over a 25 year investment period for avoided deforestation with 25 year permanence.

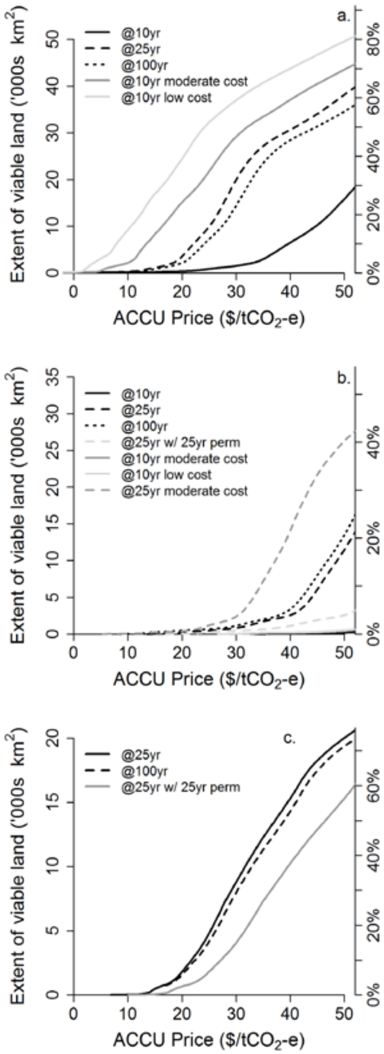


Figure 3. Fitzroy Basin - land with potential for profitable carbon farming. a. managed regrowth; b. environmental plantings, and; c. avoided deforestation. Secondary y-axis shows percentage of land with suitable landuse and vegetation status.

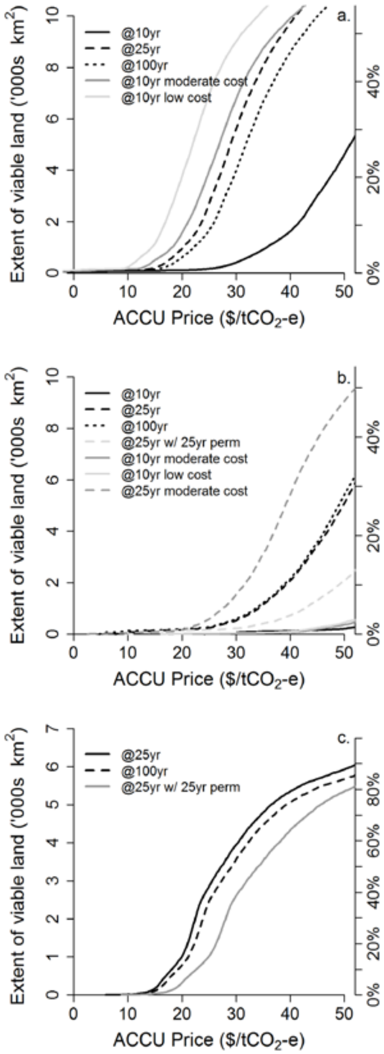


Figure 4. Burnett-Mary - land with potential for profitable carbon farming. a. managed regrowth; b. environmental plantings, and; c. avoided deforestation. Secondary y-axis shows percentage of land with suitable landuse and vegetation status.

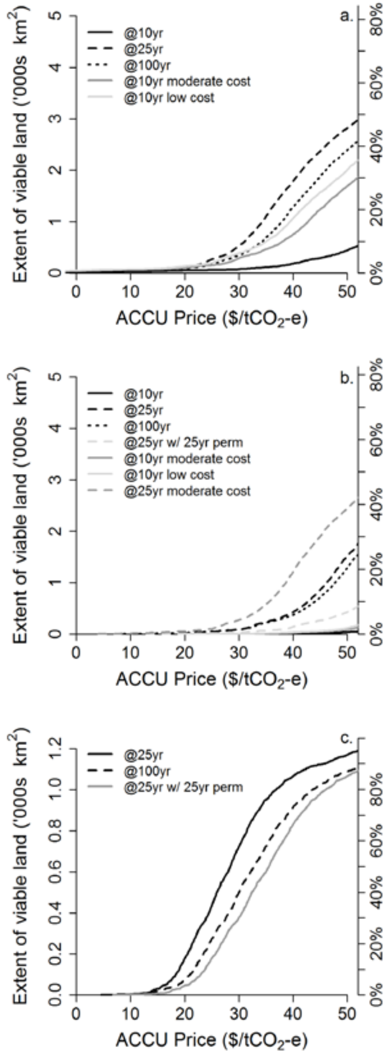


Figure 5. South-east Queensland - land with potential for profitable carbon farming. a. managed regrowth; b. environmental plantings, and; c. avoided deforestation. Secondary y-axis shows percentage of land with suitable landuse and vegetation status.

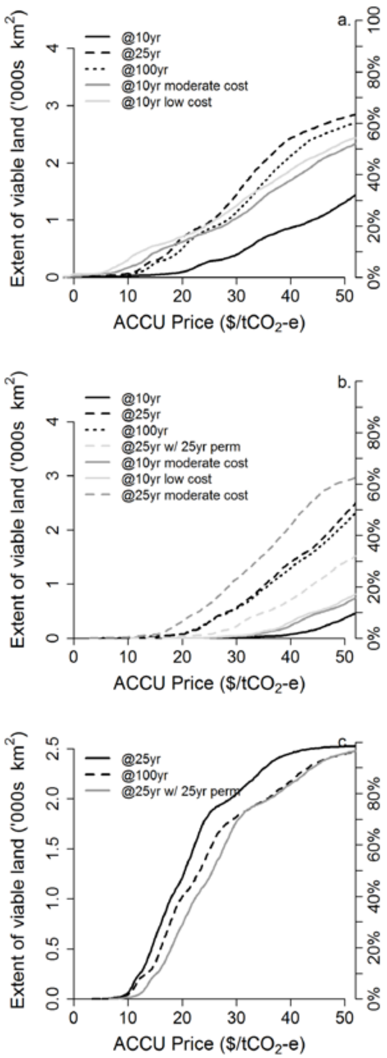


Figure 6. North-coast - land with potential for profitable carbon farming. a. managed regrowth; b. environmental plantings, and; c. avoided deforestation. Secondary y-axis shows percentage of land with suitable landuse and vegetation status.

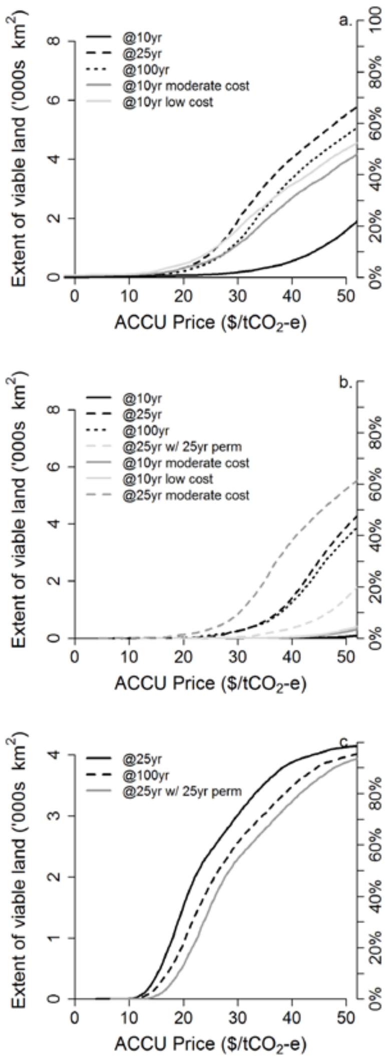


Figure 7. Hunter - land with potential for profitable carbon farming. a. managed regrowth; b. environmental plantings, and; c. avoided deforestation. Secondary y-axis shows percentage of land with suitable landuse and vegetation status.

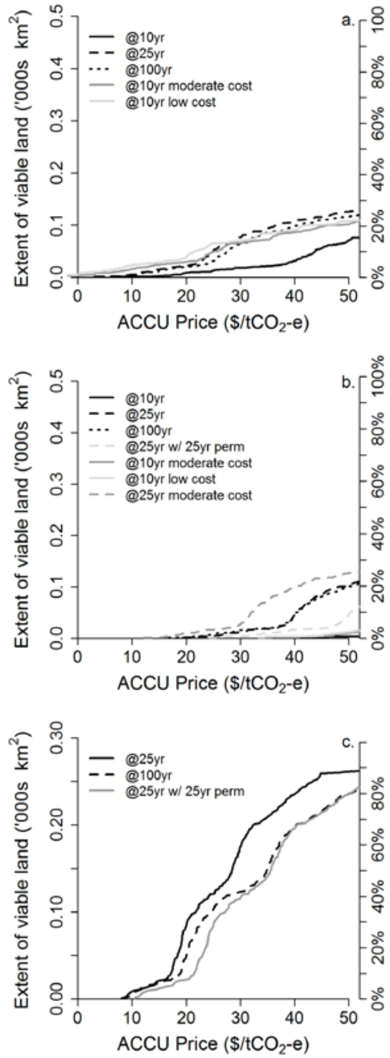


Figure 8. Greater Sydney - land with potential for profitable carbon farming. a. managed regrowth; b. environmental plantings, and; c. avoided deforestation. Secondary y-axis shows percentage of land with suitable landuse and vegetation status.

Table 6. Indicative figures for each NRM region's potential magnitude of sequestration during first ten years of carbon farming where current analysis suggests economic viability over 25 year investment timeframe given $15, $20 or $30 per ACCU (i.e. price to break-even < $15-$30)

Scenario	Extent viable ('000s km^2)			Potential CO_2-e sequestered over 10 years (Mt CO_2-e)			Potential rate of ongoing sequestration 10-20 years (Mt CO_2-e /year)		
	$15	$20	$30	$15	$20	$30	$15	$20	$30
Fitzroy Basin									
Regrowth	1	4	20	5	13	58	1	3	14
Regrowth – moderate ongoing costs	27	36	47	70	88	105	16	21	25
Planting	0.01	0.2	0.8	0.04	2	5	0.01	0.3	1.0
Planting – moderate ongoing costs	0.2	0.4	2	2	3	14	0.3	1	3
Burnett-Mary									
Regrowth	0.2	1	6	1	5	21	0.2	1	5
Regrowth – moderate ongoing costs	5	9	12	19	29	37	4	7	9
Planting	0.1	0.2	1	1	1	4	0.2	0.2	1
Planting – moderate ongoing costs	0.1	0.2	2	0.5	1	11	0.1	0.2	2
South-east Queensland									
Regrowth	0.1	0.1	0.5	0.3	0.6	2.7	0.1	0.1	0.6
Regrowth – moderate ongoing costs	0.2	0.5	1.9	0.8	2.3	8.5	0.2	0.5	2.0
Planting	0.004	0.02	0.1	0.04	0.1	0.7	0.01	0.02	0.1
Planting – moderate ongoing costs	0.03	0.1	0.3	0.2	0.4	2.3	0.03	0.1	0.5

Scenario	Extent viable ('000s km^2)			Potential CO_2-e sequestered over 10 years (Mt CO_2-e)			Potential rate of ongoing sequestration 10-20 years (Mt CO_2-e /year)		
	$15	$20	$30	$15	$20	$30	$15	$20	$30
North Coast									
Regrowth	0.3	1	1	2	4	8	0.5	1	2
Regrowth – moderate ongoing costs	1	1	2	5	7	13	1	2	3
Planting	0.0	0.1	1	0.2	1	6	0.04	0.2	1
Planting – moderate ongoing costs	0.1	0.3	1	0.7	3	10	0.1	0.6	2
Hunter									
Regrowth	0.1	0.3	2	0.5	2	9	0.1	0.4	2
Regrowth – moderate ongoing costs	1	2	4	3	9	19	1	2	4
Planting	0.003	0.02	0.25	0.03	0.1	2	0.01	0.03	0.40
Planting – moderate ongoing costs	0.03	0.1	1	0.2	1	7	0.04	0.2	1
Greater Sydney									
Regrowth	0.02	0.02	0.1	0.1	0.1	0.3	0.02	0.02	0.1
Regrowth – moderate ongoing costs	0.06	0.07	0.1	0.2	0.3	0.4	0.1	0.1	0.1
Planting	0	0.001	0.01	-	0.01	0.1	-	0.001	0.02
Planting – moderate ongoing costs	0.001	0.01	0.03	0.01	0.07	0.18	0.001	0.01	0.04

The estimates for potential abatement calculated in this analysis are broadly consistent with estimates of potential abatement published elsewhere. Evans et al. (submitted) estimated potential area with profitable carbon farming by either regrowth or environmental plantings to be around for 1.2M ha in Queensland alone with a price for ACCUs of $20, which compares well with our estimate of around 1.4M ha for regrowth alone in Queensland and New South Wales.

The estimates provided here for potential abatement are slightly lower than those calculated by Polglase et al. (2013) for comparable scenarios. This is primarily because Polglase et al. (2013) used a different model for forest growth, which increased the carbon benefits per unit area. Despite this difference in magnitude of potential, the spatial locations identified by Polglase et al. (2013) as having the greatest potential for profitable farming are generally consistent with our analysis. Areas with relatively high prospectivity are in the northern Brigalow Belt (Fitzroy Basin), northern Burnett-

Mary NRM regions, a belt down the western slopes of the Great Dividing Range in northern New South Wales, the upper Clarence Catchment (North Coast LLS region) and north-western end of the Hunter LLS region.

2.3 Conclusions

Economic analysis of carbon farming at regional scale, such as presented above and developed by other authors (Burns et al. 2011; Comerford et al. 2012; Polglase et al. 2013; Evans et al. submitted) highlights the complexity of decisions facing land managers and their financial partners.

There are clearly opportunities for profitable carbon farming using available methodologies under the current regulatory regime.

High up-front costs, such as those associated with establishing environmental plantings or conducting complex forest-carbon inventory, place serious constraint on profitable carbon farming. Higher prices, longer investment horizons and lower discount rates can counter-balance such up-front costs.

Economies of scale, providing larger or aggregated projects with low costs per unit area/abatement, may greatly increase the extent of land potentially profitable for carbon farming.

Low ongoing costs provide greater benefit in scenarios with lower up-front costs, such as the regrowth scenarios assessed here.

The proportion of land likely to support profitable carbon farming varies greatly between regions in eastern Australia. Regions with large cities show lower potential, because of the limited extent of agricultural land and high opportunity costs associated with land-uses that would be displaced by carbon farming.

It is important to note that this analysis is structured to err on the side of caution, and provides fairly conservative assessment of potential for profit.

One significant way that the economics of real projects may differ from the structure of this analysis is that real carbon farming projects may not completely displace existing land use. For example, conservatively managed grazing is compatible with some planting methodologies and with regrowth projects too. As such, profit from existing grazing enterprises may not all be lost by establishment of a new forest. Over the medium term, there will of-course be a reduction in grazing capacity because of forest development, but this will not necessarily be as comprehensive as it has been modelled here.

3 NRM risks and co-benefits

As well as having a pivotal role in carbon storage in landscapes, trees affect local microclimate, hydrology and biodiversity, amongst other things. These local effects can scale up to regional effects on climate, hydrology, soil movement, salinity, fire and biodiversity. So there is an expectation that carbon farming by sequestering carbon into new forests, or increasing carbon stores in existing forests, or even reducing the share of pasture-growth that goes into cattle and out the farm gate (rather than into soil), might have knock-on effects. These could be economic and social as well as biophysical, but here we focus on the biophysical.

Much of what we know about biophysical effects of vegetation comes from studies of deforestation. We assume that establishment of forests might reverse the many documented effects of deforestation. There is ample evidence that this is often a sound assumption. Reforestation has documented efficacy in reversing issues with salinity, soil movement and local biodiversity loss that emerge after deforestation (Lamb et al. 2005). These positive effects do not require that new forests perfectly reproduce the pre-clearing ecosystem (Chazdon 2008).

Some of the most threatening potential negative impacts (dis-benefits) of carbon farming are manageable with appropriate policy (Lin et al. 2013). For example, risks of increased land clearing, monoculture plantations replacing diverse remnants, and reduced water availability in regulated catchments are all addressed to a large extent by restrictions in the current CFI/ERF framework.

3.1 Vegetation and hydrology

Vegetation is a natural regulator of hydrological processes within catchments. Changes to woody vegetation cover through different land management practices alter hydrological processes. Landscape deforestation increases catchment water yield through increased runoff, whereas reforestation decreases runoff through rainfall interception by tree canopies and by increased evapotranspiration. The difference in mean annual evapotranspiration between forested and non-forested landscapes is greatest in high rainfall areas (Zhang et al. 2001). Investigations in eastern Australia have shown that deforestation can result in an increase in runoff by as much as 58 per cent (Siriwardena et al. 2006). For example, runoff in the Comet River in the Fitzroy Basin NRM region has increased by more than 40 per cent since extensive land clearing has occurred (Siriwardena et al. 2006). By contrast, runoff and soil erosion is significantly reduced under increasing vegetation cover on rehabilitated land due to increased infiltration of water into the soil under tree canopies (Loch 2000).

Increased runoff through land clearing can degrade land and reduce water quality by increasing levels of salinity, soil erosion and sediment accumulation in waterways. So it is often true that reductions in run-off following revegetation can positively affect landscape sustainability and improve water quality. However, there are situations were reduced run-off might have negative impacts. These include highly-committed water resource areas, where water use by new vegetation might exacerbate an already limited flow regime. Reduced run-off may also exacerbate issues with in-stream salinity, by reducing dilution. In long-cleared ecosystems, changes to hydrology from clearing eventually lead to a shift in species composition and vegetation structure (Bren 1992; Whited et al. 2007; Tockner et al. 2010; Whalley et al. 2011). So reforestation in long-cleared landscapes may, in some circumstance, disrupt a newly established equilibrium and may therefore threaten some biophysical values. Awareness of the hydrological effects of vegetation will help identify such potential negative impacts. Reductions in run-off caused by new forests may also be a problem for water storage infrastructure, but in this case costly reductions in flow should be traded-off against beneficial reductions in nutrient, sediment and salt transport.

Vegetation and secondary salinity

Secondary salinity is one of the best known effects of altered hydrology often caused by vegetation clearing and/or irrigation. Irrigation with saline water can directly lead to salt concentration in upper soil layers. Secondary salinity is also commonly caused by an increase in deep drainage of water to the water table. This additional water may come from irrigation or may occur in dryland situations after forest clearing reduces water use by trees. Secondary salinity arises if the increase in deep drainage is sufficient to raise the water table and carry soil salt up toward the soil surface. The risk of secondary salinity is greatest where clearing occurs in recharge or catchment intake areas that have a naturally shallow depth (< 6m) to ground water and contain soils naturally high in salt. Salinity often occurs at lower discharge areas of the landscape such as hill toes, valley floors and deeply incised creeks and gullies where saline groundwater seeps across the land surface or into freshwater systems.

Secondary salinity is recognised as a serious concern in at least four of the six NRM regions studied here, including the Fitzroy Basin, Burnett Mary, SEQ and Hunter regions.

In the Fitzroy Basin, soils with high salt content have been identified in catchment runoff areas under extensively cleared, former brigalow (*Acacia harpophylla*) lands (Bui and Henderson 2003). The clearing of extensive areas of brigalow vegetation potentially exposes large areas of the Fitzroy Basin to secondary salinity. The semi-arid climate in which brigalow grows has higher potential transpiration rates than rainfall most of the time and thus may limit deep drainage to groundwater. This groundwater limitation combined with low soil permeability could reduce the likelihood of salinity from cleared brigalow sites in the Fitzroy Basin (Thorburn et al. 1991) and the relatively few sites in the basin where salinity is currently expressed (Forster 2007) are consistent with this suggestion. However, more recent investigations of soils under cleared and remnant brigalow at locations throughout the Fitzroy Basin, revealed significant soil salt mobilisation in areas that have been cleared and cropped (Radford et al. 2009; Silburn et al. 2009). By contrast, soil mobilisation was less pronounced in cleared brigalow soils supporting pasture, once pasture was established, and salts under remnant vegetation remained relatively stable. These mobile salts caused from landscape alteration increase groundwater salinity but might not be evident as surface salinity in discharge areas for decades to centuries (Silburn et al. 2009).

Land clearing and irrigation in the Burnett Mary has left areas containing high chloride salt content vulnerable to salinity (Ridge 2005; Cresswell 2006). Salinity affected landscapes in both coastal and inland parts of the Burnett Mary represent some of the more problematic areas for dryland salinity in Queensland (Biggs 2007). The main areas exposed to secondary salinity are the Burnett Valley, Bundaberg and Kingaroy (Biggs and Mottram 2008), the Isis area (Ridge 2005) and the Mary Valley (Wylie et al. 1993; Ridge 2005).

In inland areas of the Burnett-Mary and South-east Queensland NRM regions, soils with high salt content have been identified in catchment runoff areas under brigalow plant communities including those with *Casuarina cristata* (belah) (see Bui and Henderson 2003), and intake areas including softwood scrubs and *Eucalyptus melanophloia* (silver-leaved ironbark) woodlands. *Melaleuca bracteata* (black tea tree) can also be indicative of soils with naturally high salt content (Hughes 1984). In coastal areas, vegetation indicating naturally high saline soils includes *Melaleuca nodosa* (prickly-leaved paperbark) (Evans 1967; Hughes 1984), as well as *Allocasuarina luehmannii* (bull oak) (Evans 1967), which occurs in association with the vulnerable *Eucalyptus hallii* (Goodwood gum) in the Goodwood-Woodgate area. In riparian zones along waterways of the Mary River catchment, a direct relationship was found between water salinity and dieback in *Casuarina cunninghamiana* and *Eucalyptus* species (Wylie et al. 1993). This investigation also found a link between water salinity and tree dieback with localised tree clearing activities.

In SEQ, the main area exposed to secondary salinity is in the Lockyer Valley, with expressions also evident at Crows Nest and the Beaudesert – Boonah area (Biggs and Mottram 2008). Much of the salinity in the Lockyer Valley results from irrigation using groundwater from small alluvial aquifers that become highly salt concentrated during droughts (Dixon and Chiswell 1992).

Land clearing, mining and water use by irrigators and heavy industry in the Hunter region have left areas containing high chloride salt content vulnerable to salinity (Connor et al. 2004) with the lower Hunter Valley considered to be one of the main areas of dryland salinity in New South Wales (Charman and Junor 1989). Current salinity levels in the Hunter are higher than catchments of similar size in the Murray-Darling Basin (Beale et al. 2001).Whilst there have been no significant changes to stream salinity, groundwater salinity in the basin is generally rising except in areas with high levels of groundwater extraction. Predicted increases in salinity are expected in the Hunter region from groundwater expression. Salinity discharge into streams is also likely to increase with an increase in mining activity (Beale et al. 2001).

Future effects of climate change on secondary salinity are difficult to determine due to the predicted shift to highly variable rainfall patterns. Salinity in some areas may be alleviated because higher temperatures increase evaporation rates, or if rainfall is reduced, but these effects may be offset by increased irrigation for food production to feed a growing population (Yeo 1999). Also, even though extended dry periods can reduce groundwater levels, flood events can result in rapid recharge that leads to the development of salinity expressions that may take several years to decline (Biggs and Mottram 2008). Increasing variability and extreme high rainfall events under future climate may therefore increase salinity risk even if conditions are drier on average.

Reforestation is an effective method of preventing or even reversing secondary salinity. Reforestation of recharge areas in salinity affected systems can rapidly lower the saline groundwater table (within ten years) and reduce salinity in the groundwater (Bell et al. 1990; Bari and Schofield 1992). If species are being selected for plantings in discharge areas they should ideally be vigorous, deep rooted and have high water usage requirements.

Carbon farming seems most likely to have positive effects on salinity issues. Exceptions, to this general statement include salinity affected water-bodies, where reduction in run-off may decrease dilution before any beneficial changes to saline discharge to the system take effect. Reduction in run-off from catchments with limited salinity issues could also increase the relative contribution of stream flow coming from salty catchments and therefore exacerbate in-stream salinity issues in some situations.

Carbon farming will also often improve water quality, by reducing sediment and nutrient transport into aquatic systems. In places like Sydney and the North Coast LLS region, with little secondary salinity in agricultural land, this water quality benefit from carbon farming is likely to be the main potential hydrological benefit from new forests. However, all six of the NRM regions in this study contain examples of floodplains or drainage lines with seriously depleted riparian vegetation. New forests in riparian settings have potential to improve water quality in all regions. Riparian vegetation extent and condition are already a strong focus in the northern regions because of well documented negative impacts of agricultural runoff and sedimentation on the biota of the Great Barrier Reef (e.g. Fabricius et al. 2005; Packett et al. 2009; Kroon et al. 2012). Carbon farming may increase the cost effectiveness of riparian forest restoration.

Hydrological changes are also a path by which changes to vegetation cover can alter local and regional climate, which is discussed in the following section.

3.2 Vegetation, climate and fire

While increasing water retention and use in landscapes, vegetation also shades the soil surface, and changes the absorption of incident solar radiation and movement of air, which in turn affect the atmosphere and regional climate (McAlpine et al. 2009). There are numerous well established examples of deforestation leading to higher temperatures and reductions in rainfall at regional scale, and simulation modelling suggests that land cover change from past clearing may already be exacerbating Australia's renowned climate extremes (McAlpine et al. 2007; Deo et al. 2009).

At more local scales trees produce shade, reduce wind speed and increase woody debris, which can reduce temperature and increase humidity, and often reduces grass biomass. This combination of local scale effects has a substantial impact on fire.

Carbon farming involving sequestration to vegetation and soils will alter loads, connectivity and characteristics of fuels for wildfire. The question that naturally arises is how much might these changes increase the risk from fire to people and infrastructure in eastern Australia?

Fire and weather

Weather has the greatest effect on fire behaviour and intensity. Historically, property loss from wildfire in Australia has occurred during rare fire weather conditions. The majority of house loss has occurred when the Forest Fire Danger Index (FFDI) has exceeded 100 (Catastrophic) and minimal house loss has occurred on days when the FFDI has failed to reach 50 (no higher than Very High) (Blanchi et al. 2010).

Seasonal weather variation has an over-riding influence over fire timing and intensity (Murphy et al. 2013). Fires occur in tropical Australia predominantly from winter to spring, in subtropical eastern Australia during spring, and in south eastern Australia from spring to summer. Fire return intervals also vary across the continent. In the north, fires can be almost annual. The carbon farming methodology for savanna burning is about abatement achieved by a shift from extensive fires in the late dry season to early dry season fire which burns less area.

Longer weather cycles, such as El Niño-Southern Oscillation (ENSO), also have an influence on fire, particularly in the temperate and arid parts of the continent (Hennessy et al. 2005; Bradstock 2010; Sullivan et al. 2012; Murphy et al. 2013). It is often observed that increases in fuel loads through wet years lead to an increase in fire incidence, especially when followed by drought (Hennessy et al. 2005; Russell-Smith et al. 2007; Bradstock 2010; Gill et al. 2010). Extended drought can increase fire spread in east Australian landscapes by reducing the fuel moisture in creeks and gullies that might otherwise act as a barrier to fire (Sullivan et al. 2012).

Carbon farming and wildfire

The amount of flammable fuel in landscapes depends on the rate fuel of accumulation (litterfall input), fuel moisture levels and fuel composition (ratio of grass to woody plant materials) (Bradstock 2010; Sullivan et al. 2012; Collins et al. 2013), all of which are influenced by a change from pasture to forest. Generally speaking, forests fires are often limited by fuel moisture whereas grass fires are most often limited by the amount of biomass present (Bradstock 2010). In grassy woodlands, tree cover reduces grass biomass and also changes its moisture levels. As a result, trees in grassy woodlands are likely to be reducing the period within each year that an area may burn, compared to pastures without trees in a similar location.

Fuel composition is influenced by vegetation type and structure. Open woodlands typically have grass and herbs in the understorey, whereas forests contain more shrub species. Vegetation structure is also controlled by fire frequency, creating feedback between vegetation structure, fuel and fire (Bradstock 2010; Sullivan et al. 2012). For example, repeated burning can replace a shrubby forest understorey with a grassy one (e.g. Birk and Bridges 1989). Fuels containing grass are highly flammable because they are arranged in an open, vertical structure that allows greater airflow to fire and because they dry quickly. So fire spreads at faster rates in grass than when burning more woody fuels (Cheney and Sullivan 2008; Sullivan et al. 2012; Collins et al. 2013).

The quick growth and dessication of grass in dry seasons also allows more frequent fire in grassy systems than forests (Russell-Smith et al. 2007; Murphy et al. 2013) However, shrubby forests often have more fuel and greater connectivity of fuels from the ground layer to the canopy, which means that when they do burn under extreme conditions, they have fires of greater intensity than grassy systems (Bradstock 2010; Collins et al. 2013).

A concern surrounding reforestation in the landscape is the potential for increased fire threat to built assets such as houses, due to increasing fuel loads. However, modelling suggests that modest levels of reforestation are unlikely to result in an increase in fire size or fire intensity in many east Australian landscapes (Collins et al. 2013).

Burning woody vegetation throws many more embers into immediately adjacent areas than does burning pasture, but the rate of spread of fire in pasture is reduced where the pasture is interrupted by woody vegetation, especially in undulating terrain (Cheney et al. 1998). Indeed, threats to assets may be reduced by the presence of reforested areas that are sufficiently wide (>540 m), at least during moderate fire-weather and low pasture fuel loads (Collins et al. 2013).

To ensure built asset protection against wildfire it is recommended that fuel be managed within a 200 m area surrounding the asset, with fuel reduction being the most effective strategy for asset protection from wildfire (Gibbons et al. 2012; Collins et al. 2013). Prescribed burns are most effective if they are conducted within close proximity to the property they intend to protect (i.e. to within 500 m of built assets) and have occurred within the previous five years (Price and Bradstock 2010; Gibbons et al. 2012).

Climate change will reduce the number of low risk fire weather days suitable for prescribed burning (Hennessy et al. 2005). Increases in the extent of land managed for carbon storage may also make prescribed burns more difficult to apply. Constraint on fire-use could be a serious potential negative impact of carbon farming on fire management. This risk could be reduced by facilitating effective fire management planning for carbon projects, including adequate fire-breaks and other infrastructure, and the application of prescribed burning were appropriate. Strategic reforestation of gullies and other situations where forest might slow or interrupt fire spread may be a useful strategy for landscape-scale fire management and planning.

Fuel loads in fire prone areas can be managed with frequent, low intensity prescribed burns. Prescribed burns are effective at reducing the intensity of wildfires in recently burnt areas. However fuel loads in woody vegetation can accumulate rapidly, reaching pre-burn levels within three to four years (Birk and Bridges 1989; Penman and York 2010) and in extreme fire weather, wildfires can burn through vegetation burnt as recently as one year prior (Chafer et al. 2004; Price and Bradstock 2010).

To minimise fire risk, plantings and regrowth carbon farming areas should be at least 100 m from fire sensitive assets. This distance should be more like 200 m on highly fire-exposed aspects. Any woody vegetation within 40 m of a built asset increases the chance of asset loss if that woody vegetation burns. It is essential to ensure access is made available between the planting and the asset for fire fighting vehicles and equipment.

Fire and climate change

Climate change is predicted to have substantial impacts on fire weather and fire behaviour in eastern Australia. In winter rainfall areas in the south east of Australia, fire weather risk is predicted to increase significantly with an increase in the frequency of days with very high and extreme FFDI ratings. The fire season in the south east of the continent is also likely to start earlier, leading to a longer season (Hennessy et al. 2005; Clarke et al. 2011). A prolonged season with increased fire risk will reduce the number of days suitable for prescribed burning and will likely force a shift in these activities to winter (Hennessy et al. 2005). An increase in fire weather risk in south eastern Australia is already apparent, with changes evident over the past few decades (Clarke et al. 2013). By contrast, fire risk levels in coastal tropical and subtropical areas dominated by summer rainfall are likely to remain at or near current levels (Clarke et al. 2011).

Fire regimes in the southern parts of our study region, the Hunter and Greater Sydney regions, are influenced by uniform seasonal rainfall. A bias towards summer rainfall in the north and inland parts of the study region results in the fire season occurring in spring and summer (see Hennessy et al. 2005; Clarke et al. 2011). Climate change modelling based on weather data from Williamtown (near Newcastle) and Richmond (west of Sydney) suggests that fire season length in the Hunter and Greater Sydney regions could increase significantly either side of the current season (late August – late January), to early August – early February by 2050 (Hennessy et al. 2005). This modelling also predicted that the average FFDI could increase by nearly 10% by 2020 and around 20% by 2050 with an increase in the number of days when the FFDI is very high to extreme.

Fire regimes in subtropical Australia are influenced by monsoonal summer rainfall and to a lesser extent by rainfall events from temperate influences during early and mid-winter. Hence, the fire season in the subtropics predominately occurs in spring, after fuels dry through winter and temperatures begin to increase. Based on climate change modelling in coastal (Coffs Harbour, Hennessy et al. 2005) and inland (Miles, Williams et al. 2001) subtropical regional centres, it is reasonable to assume that fire season length in the north of the study area, from the Fitzroy Basin to the North Coast in NSW could increase by one to two weeks either side of the current season, if at all. However fire danger is likely to increase with more severe fires occurring earlier in the fire season (Williams et al. 2001). In coastal and sub-coastal areas, fire risk under climate change is expected to remain close to current levels (marginal increase) due to an increase in humidity and summer rainfall (Clarke et al. 2011). However, weather data from between 1973 and 2010 from both Amberley (west of Brisbane) and Rockhampton in coastal central Queensland, reveal significant increases in the sum of annual FFDI values, the likelihood of intense fires; as well as autumn and winter median FFDI values. By comparison, weather data from the Brisbane Airport revealed no significant changes to fire weather during this time period (Clarke et al. 2013).

Highly variable changes to rainfall patterns predicted under climate change scenarios make the future impacts on fuel loads, fire intensity and fire spread difficult to determine. Modelling of forage production for the grazing industry in northern Australia suggests that an increase in mean temperatures of 3 ^{o}C may reduce pasture production by about a quarter, but that increased water-use efficiency from higher CO_2 may increase production by about the same amount (McKeon et al. 2009). Further south, where the trend for drying is clearer, an increasingly warm and dry climate may result in reduced fuel loads but may still increase fuel availability for fire due to a reduction in moisture content (Matthews et al. 2012). However, in tropical and subtropical areas predicted to receive an increase in humidity and summer rainfall, fire spread may be suppressed as wetter forests advance and fuel moisture levels remain high. In areas predicted to become drier, available fuel loads in forests will likely increase as more litter is able to be cured, increasing fire intensity. In woodlands, vegetation may become more open and discontinuous with a reduction in fuel load, reducing fire intensity and spread (Bradstock 2010). Natural ignition of fuels caused by lightning may increase if the climate becomes warmer under climate change (Price and Rind 1994).

Large and catastrophic fires can cause carbon leakage from accumulated forest carbon stocks and can therefore be counteractive to carbon sequestration projects (Murdiyarso et al. 2002). Appropriate fire regimes in some vegetation may limit carbon leakage from wildfires (Hurteau et al. 2008). However, if fire frequency and intensity increases under climate change as predicted, the rate of prescribed fires in fire prone vegetation such as eucalypt woodlands is likely to increase to reduce the threat of wildfires. Increased rates of prescribed burning will increase the rate of carbon leakage into the atmosphere reducing the overall effectiveness of some vegetation types as carbon sinks (Bradstock et al. 2012).

Fire and landscape change

Climate change is one of many changes occurring in east Australian landscapes. Urban expansion into fire prone environments, and changes to the biophysical environment are two other important themes.

Increasing the extent of interface between urban and natural environments is stretching fire management resources and limiting capacity for ecological burn. It is also increasing the risk of property loss to fire. It is recognised that less reliance needs to be placed on emergency services to protect property from wildfire, with a greater emphasis of property protection placed on the landholder. However, there is some evidence to suggest that people who are inexperienced with wildfire can be more often less aware of the risks than people more regularly exposed (Rhodes 2003; Childs et al. 2006; Bushnell et al. 2007; Gill 2009), although regular bushfire exposure does not necessarily lead to an adoption of protective measures. Communities that are more regularly exposed to bushfires are also more likely to develop greater social networks and an enhanced local bushfire knowledge that leads to greater awareness, preparedness, and ultimately greater self-reliance (Rhodes 2003; Childs et al. 2006; Paton et al. 2006; Gill 2009).

With an increase in urban expansion and semi-rural settlement, the issues surrounding fire management are likely to become more problematic as increasingly more inexperienced residents become exposed to bushfire threats. Greater community consultation on bushfire management and property protection will be required for those living at the urban-bushland interface, particularly as fire frequency and fire intensity increases with the effects of climate change.

On the biophysical side, bulky exotic grasses are transforming the fuel characteristics in many regions. A commonly cited example is particularly relevant in the Fitzroy Basin NRM region where large exotic grasses such as buffel grass (*Cenchrus ciliaris*) and green panic (*Megathyrsus maximus*) have increased fuel loads under remnant and regrowth areas of the nationally threatened brigalow ecological community and semi-evergreen vine thicket ecological community, particularly if cattle are excluded. The increased fuel loads generated by these exotic grasses can cause intense fires that can destroy vegetation in such fire sensitive plant communities (see Butler and Fairfax 2003), with devastating effect on biodiversity.

Whilst the use of fire should be discouraged from fire sensitive vegetation such as brigalow and semi evergreen vine thickets, fire at appropriate frequencies should continue to be used as a management tool in open forests and woodlands for maintaining ecological integrity and as a cost effective form of weed management. However in open forest and woodlands invaded by large exotic grasses, careful management is required to ensure fires are burnt at low intensities to minimise loss of habitat.

Gamba grass (*Andropogon gayanus*) is one such grass that requires careful consideration when present in a fire-managed landscape (e.g. Setterfield et al. 2010). Whilst still mostly confined to more tropical environments, this species was recently recorded north of Rockhampton (Queensland Herbarium 2014). Control of gamba grass in the Fitzroy Basin NRM needs to be a priority in order to limit further spread of this ecosystem transformer species.

Large, exotic ponded pasture species, para grass (*Brachiaria mutica*), olive hymenachne (*Hymenachne amplexicaulis*) and aleman grass (*Echinochloa polystachya*) pose a similar fire threat in floodplain communities due to their ability to produce substantially high amounts of biomass. If not consumed through grazing, this biomass has the potential to cause extremely intense fires once cured in the dry season. Controlling biomass production in all large, invasive, exotic grass species will be an ongoing problem that will require long-term maintenance. Localised strategic removal can be carried out using chemical and mechanical treatments. However at the landscape scale it is likely that this will be best achieved by grazing with livestock. In this situation, grazing should be done on high rotation to minimise impacts to native species and cattle should be excluded from adjacent areas under rehabilitation in which these weeds are not present.

Grazing is not always a viable management option for the control of some large invasive grasses. Thatching grass (*Hyparrhenia rufa*) and grader grass (*Themeda quadrivalvis*) are becoming increasingly more invasive in some coastal and sub-coastal open forest and woodland communities. Like introduced Parramatta and rat's tail grasses (introduced *Sporobolus* spp.) and African lovegrass (*Eragrostis curvula*), their control is even more complex than it is for invasive pasture grasses more palatable to livestock. One management option to reduce biomass in these species is controlled burns earlier in the dry season when curing percentages are lower. The management of these species is particularly important following wet seasons with greater than average rainfall in which large volumes of biomass are generated.

3.3 Carbon farming and biodiversity co-benefits

The potential for carbon farming to benefit biodiversity is widely recognised (Fensham and Guymer 2009; Crossman et al. 2011; Bradshaw et al. 2013). The types of carbon farming considered in this report are particularly obvious candidates for biodiversity co-benefits because they all provide increased habitat for native species either by increasing native forest extent, or by avoiding decreases in native forest extent.

This section describes a spatial analysis to identify parts of eastern Australia with the greatest potential for co-benefits to biodiversity from new native forests, such as would be delivered by carbon farming with environmental plantings or native forest regrowth. We apply four pragmatic principles to develop a rule-of-thumb for where revegetation using local native species could provide the most benefit to native biodiversity:

1. Revegetation is more beneficial on land that historically supported ecosystems heavily impacted by habitat destruction than on land types that are still largely intact
2. Revegetation is more beneficial in landscapes that have been heavily impacted by habitat destruction than in landscapes that are still largely intact

3. Revegetation is more beneficial to biodiversity if it occurs in places well connected to existing natural habitats than in places that are remote from natural habitats, and
4. Revegetation is more beneficial where it may provide additional habitat for a large number of threatened species than in places likely to support few threatened species.

These four principles identify places where revegetation might mitigate habitat destruction by building on the remaining network of habitats within the range of biodiversity most impacted by historic habitat destruction. They are not an exhaustive list of the types of rules that might be used to identify places where new native forests might benefit biodiversity, and were selected because of their clear priority and because they are amenable to assessment with data available for many regions. Far more complex approaches to scheduling restoration priority can be developed (see Wilson et al. 2011 for example). However, the four principles are based on well-established scientific understanding of the effects of habitat destruction, and they offer a high degree of flexibility in their application to natural resource management planning.

It is also possible to apply established techniques for conservation reserve selection to identify the most efficient sets of priority locations for ecological restoration (Crossman and Bryan 2006; Noss et al. 2006; Thompson et al. 2009;). However, there is also evidence that using simple rules of thumb, such as “protect the available site with the highest richness”, to choose sound incremental investments among small sets of currently available candidates can provide outcomes comparable to approaches based on efficient sets (Moilanen 2008; Butler 2009); especially when site availability is unpredictable (Meir et al. 2004).

Decisions based on rules of thumb (heuristics) may incur a risk of inefficiency but they can also have strong advantages in terms of transparency and how well they identify the conservation problem under consideration (Moilanen 2008). Heuristics can also avoid arbitrary decisions about targets, reducing planner stress and avoiding some negative connotations of target-setting as a policy expression (Soule and Sanjayan 1998). Wilson et al. (2011) found that the efficiency of restoration strategies based on well-chosen heuristics can be almost as effective as more complex numerical techniques. The four principles outlined above are justifiable from ecological theory and also make intuitive sense. They’re easy to explain.

The first two principles prioritise revegetation of locations that would complement remaining patches of native vegetation, to better reflect the full diversity of habitat types a given landscape displayed prior to clearing. The first applies a ‘land-type’ lens to identification of habitat extent, and the second is perhaps a more intuitive ‘landscape’ view of habitat extent.

Complementarity is a fundamental consideration for efficiency in conservation planning (Pressey and Nicholls 1989; Margules et al 2002; Ferrier and Wintle 2009). It directs conservation reserve acquisition toward land types that are not already represented in the network. In reserve acquisition, the aim is often to efficiently achieve a set target for representation of each component of biodiversity. Classically, to find the cheapest set of properties that will satisfy an established goal, so that they can be acquired and protected.

Regional NRM organisations tend to operate in partnership with regional communities. They tend to provide a positive input or influence rather than directing action from the top down. The conceptual and mapping method for the metric described in this analysis was developed in consultation with the Fitzroy Basin NRM association and is intended to lend itself to bottom-up approaches to action that align with practice in regional NRM.

As well as complementarity, the first two principles also help assess extinction debt at a range of scales. There is a very well-established connection between habitat destruction and species loss

(Tilman et al. 1994). The concept of extinction debt follows from the observation that habitat loss leads to the slow decline of species populations over decades, ending with the loss of species from landscapes (Hanski 2011). That is, clearing induces an extinction debt in a landscape until the new and lower equilibrium number of species that can be supported by the reduced habitat is reached.

This period of 'relaxation' in biodiversity to new lows following clearing can take many decades. One well documented example is the avifauna of the Mt Lofty Ranges near Adelaide. The birds of the Adelaide Hills are still slowly dwindling many decades after most clearing occurred (Szabo et al. 2011). Revegetation may increase the number of species that can be sustained in the Adelaide Hills and in other heavily cleared landscapes, and that is why both of the first two principles are about the extent of past clearing. A fundamental benefit of revegetation for biodiversity is that increased habitat can soften the impact of past clearing on the number of species supported.

Principle 3 is about spatial complementarity for existing native vegetation, not pre-clearing vegetation. It prioritises restoration where it will be accessible from, and may support species within, remaining relatively-natural ecosystems. Connectivity also has implications for the likelihood of colonisation of potential restoration sites by native biota, i.e. successful forest restoration, as well as the likely utility of a new forest as habitat for native biota.

Principle 4 adds a dimension to the index correlated with species-scale threats from habitat destruction. It calls for information about species richness and vulnerability.

Although these principles are simplistic rules-of-thumb, they are based on some of the strongest foundations of ecology and in combination they can provide a relatively transparent way in which two or more potential restoration sites in broadly similar condition might be evaluated against each other. The process used here to apply these principles is to map them more-or-less independently on a 250 m grid across a large section of eastern Australia encompassing the six target NRM regions. As individual grids they could be incorporated as separate factors along with other data into a multi-criteria analysis. However, we also present a compiled product that is simply the sum of values for each principle, scaled to range from zero to one, with equal weight on each.

Mapping the four principles for revegetation benefit across eastern Australia

This section briefly describes data used to map each of the four principles outlined above.

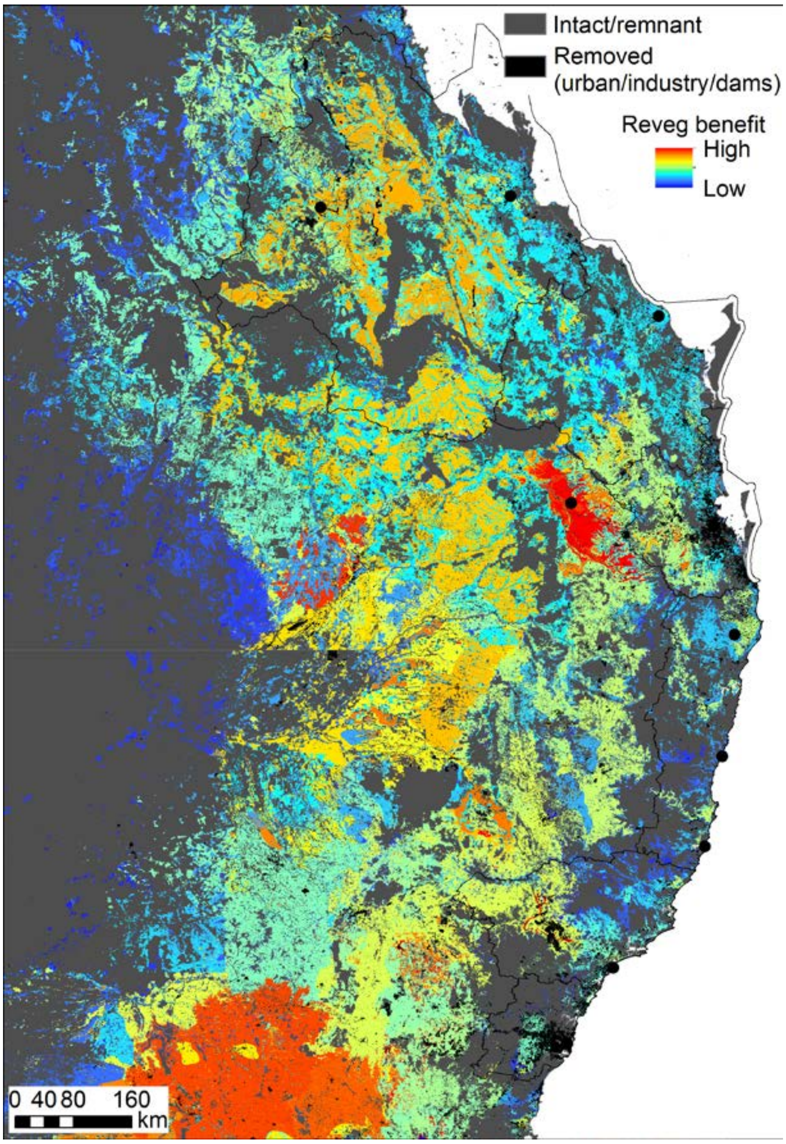
1. **Revegetation is more beneficial on land that historically supported ecosystems heavily impacted by habitat destruction than on land types that are still largely intact**

Mapping this principle requires data on the pre-clearing and current distribution of ecosystem types or other land classes. When working only in Queensland we use 1:100 000 scale spatial data for pre-clearing and remnant Regional Ecosystems for this purpose (see Appendix 3 for maps showing the difference between approaches based on QLD species data and this study). However, for consistency across the study area, we intersected spatial data defining 100 biotypes across the study region (Drielsma et al. in prep), with spatial data on pre-clearing extent of major vegetation classes from Australia's National Vegetation Information System (Version 4.1, http://www.environment.gov.au/topics/science-and-research/databases-and-maps/national-vegetation-information-system). The resulting dataset, which has an effective scale around 1:500 000 and included over 1000 classes, was intersected with data on the distribution of intact or remnant vegetation[3] across the study area (Map 2) to provide an estimate of the proportion of pre-clearing extent still intact (*i*) for each class.

[3] 2008 vegetation extent data for NSW and 2011 remnant vegetation extent for QLD

The relationship between species and area follows an exponential form and this knowledge was used to transform the proportion remaining (*i*) for each class to give an index of extinction debt reflecting principle 1 (P1).

$$P1 = 1 - i^{0.25} \qquad \text{Eq. 2}$$



Map 12. Index of restoration priority based on remaining extent of pre-clearing ecological classes in eastern Australia.

The 0.25 exponent in Eq. 2 reflects common species richness to habitat area relationships for islands and is suitable as a general conservative guide to levels of likely extinction from habitat destruction (Brooks 2011; Rybicki and Hanski 2013). Map 12 shows the resultant index across the study area, which identifies lands that supported ecological classes most heavily impacted by past habitat destruction as priorities for revegetation.

It must be stressed that identifying lands that historically supported extensively cleared vegetation types as priorities for revegetation does not imply a static view of landscapes or biodiversity. It does not require that revegetation be directed solely toward restoration of the pre-clearing vegetation type. This is particularly true in the context of climate change, where a general expectation of change must be taken as a starting point (Dunlop et al. 2013).

Clearing for agriculture tends to target particular components within landscapes. Vegetation on clay soils is often targeted, based on soil capability and the difference in productivity between cleared and uncleared states (Fensham and Fairfax 2003).

Land types (P1) and landscapes (P2) that have been heavily impacted by clearing should remain priorities for revegetation as the climate changes because they complement (i.e. provide something different than) the remaining relatively intact parts of the landscape, and because they indicate concentrations of extinction debt at various scales. These observations are arguably robust despite biotic changes that are likely to arise from changing climate. For example, if clay soils have been targeted for clearing and sandy soils are largely intact, as is often the case. Revegetation of habitats on clay soils will assist conservation of biodiversity even if biodiversity is changing, because it will increase the extent of habitats on clay soils that may be available to immigrant taxa, and it may aid the persistence of remaining populations of locally native biota associated with clay soils.

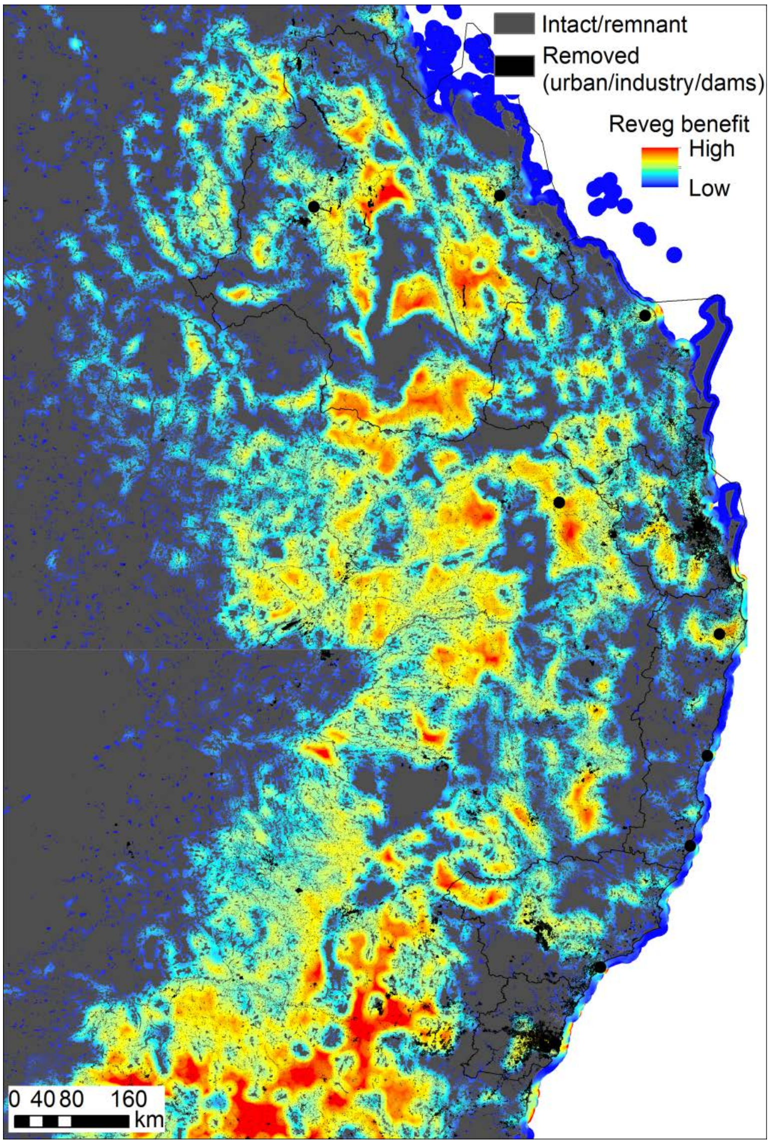
2. **Revegetation is more beneficial in landscapes that have been heavily impacted by habitat destruction than in landscapes that are still largely intact**

Principle 2 differs from Principle 1 in the scale and classification of 'habitat' addressed. Principle 1 is about ecological classes while Principle 2 is about landscape context and location.

Principle 2 is assessed here as the proportion of land in 'remnant' or 'intact' condition (*r*) within a 10km radius around each location. This was calculated from the vegetation extent data mentioned previously (Map 2) and once again an exponential transformation based on species-area relations was applied.

$$P2=1-r^{0.25} \qquad \text{Eq. 3}$$

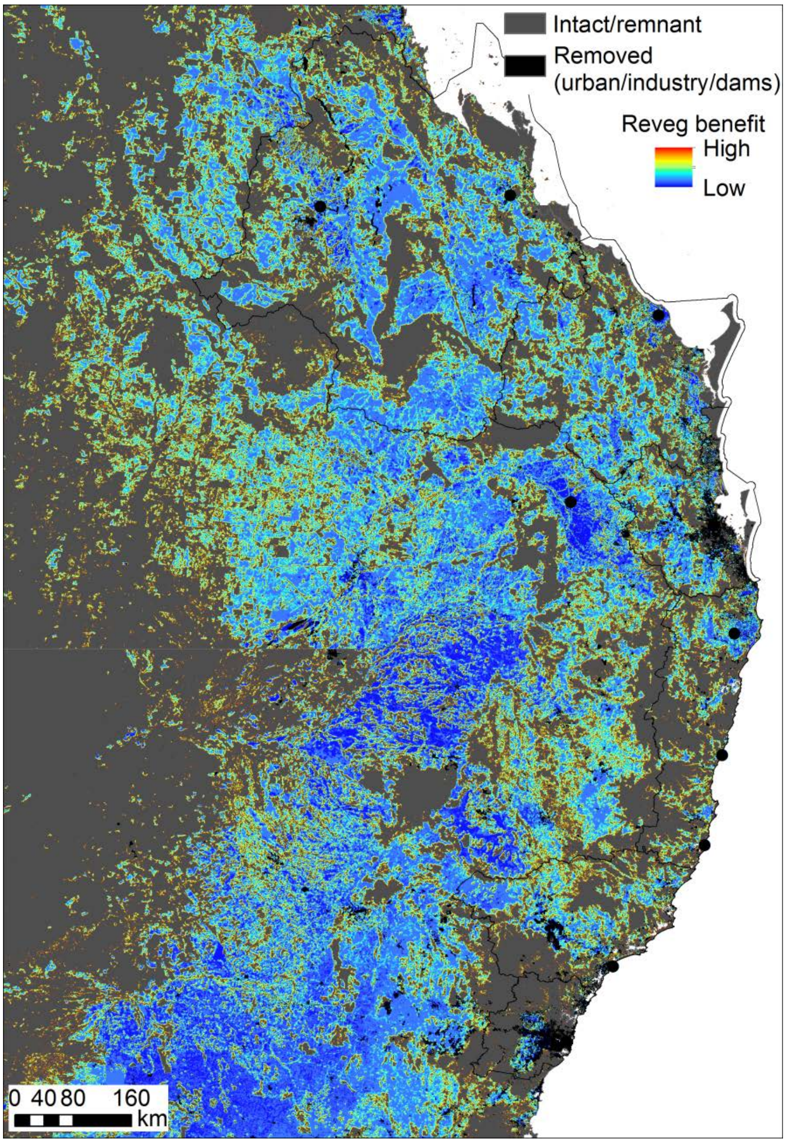
The resulting dataset highlights subregions most impacted by past habitat destruction as priorities for revegetation (Map 13).



Map 13. Index of priority for revegetation across eastern Australia based on extent of remnant intact vegetation within 10km.

3. **Revegetation is more beneficial to biodiversity if it occurs in places well connected to existing natural habitats than in places that are remote from natural habitats**

To mitigate extinction debt incurred by recent habitat destruction, revegetation should provide new habitat for biota temporarily persisting in fragmented landscapes. Connectivity between remaining native habitats and revegetation areas is therefore an important consideration. We use a measure of neighbourhood habitat value (NHV) described by Drielsma et al. (2007) to map priorities based on this principle.



Map 14. . Index of connectivity to natural habitats (neighbourhood habitat value) across eastern Australia.

NHV is calculated by coupling data on the distribution of habitat value across the landscape, with data on variation in landscape permeability (in terms of habitat value) using geometry based on least cost paths (Drielsma et al. 2007). There are clearly some complex generalisations to navigate in this approach. However, the principle of connectivity to habitat value is an important one and the approach described by Drielsma et al. (2007) is well-grounded in a substantial body of metapopulation theory (Hanski 1999).

Data on land-use was compiled and used to map a 'VAST' type classification based on 'naturalness' (Thackway & Lesslie 2008) which was associated with scores for habitat value and landscape permeability as indicated in Table 7. Basically, land uses were arrayed in increasing order of 'naturalness' and each step received a habitat value score one order of magnitude greater than the last.

Permeability was also assigned to each cell in the grid based on naturalness. Permeability is a measure of how quickly the presence of a patch of native forest or other habitat ceases to affect habitat available at a given point. A given habitat resource is assumed to be more relevant to areas closer to it, and its relevance also depends on the character of the intervening space. As an extreme example, habitat value extends further through native vegetation than across a parking lot. In effect, benefit to wildlife from proximity to native vegetation and other habitat was modelled as declining much more rapidly with distance across industrial land or intensive cropping than through cleared grazing land (Table 7).

Neighbourhood habitat value was first calculated on a 25 m grid (which is available but extremely large). The 25 m grid was used to maintain as much information as possible about narrow habitat corridors that are highly important to connectivity in fragmented landscapes. The output at 25 m was generalised (by averaging) to the 250 m grid size of the first two principles for integration with the other datasets.

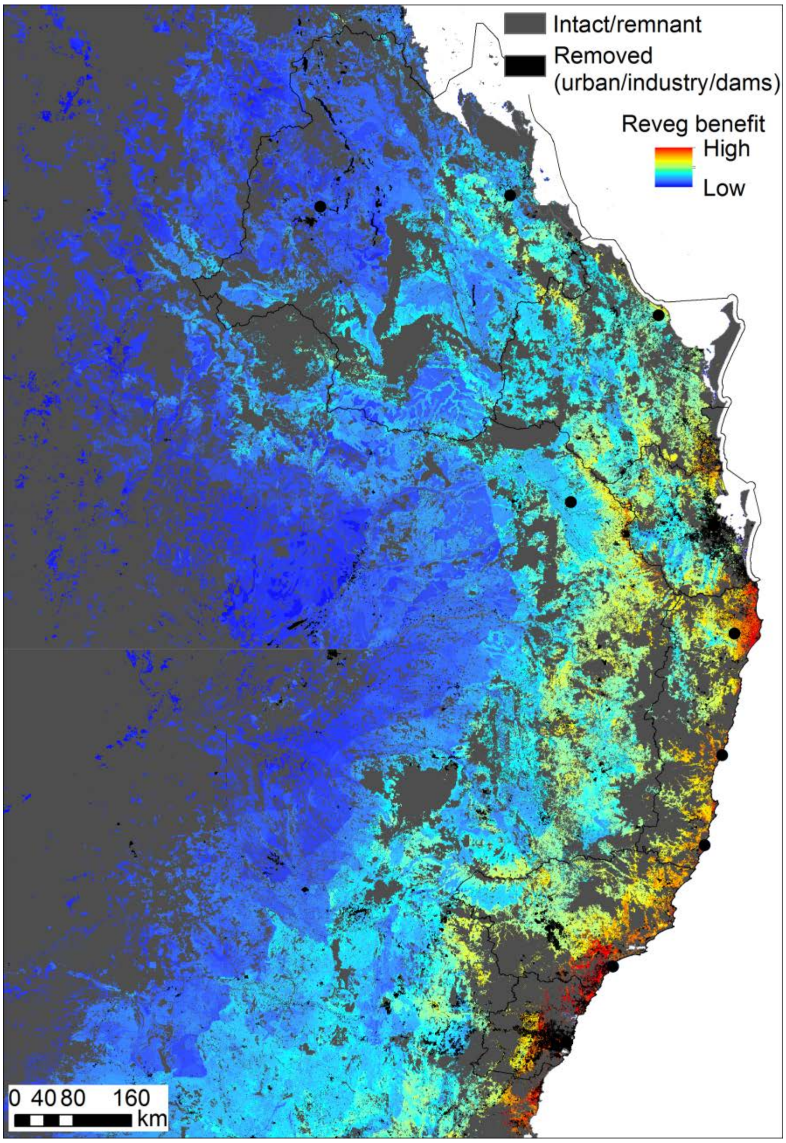
Table 7. Lookup table for habitat values and permeability of broad land condition classes used to map "neighbourhood habitat value" as a measure of connectivity to natural habitats.

Land class	Habitat value	Permeability (25m grid)[1]
Remnant or intact native vegetation (residual/modified)	100	1000
Cleared grazing and other extensive uses (transformed)	10	250
Cropping (replaced)	1	100
Industrial/urban/mining (removed)	0.1	50

[1] Values are for α, which is the distance over which habitat value declines to about half of its source value.

4. **Revegetation is more beneficial where it may provide additional habitat for a large number of threatened species than in places likely to support few threatened species.**

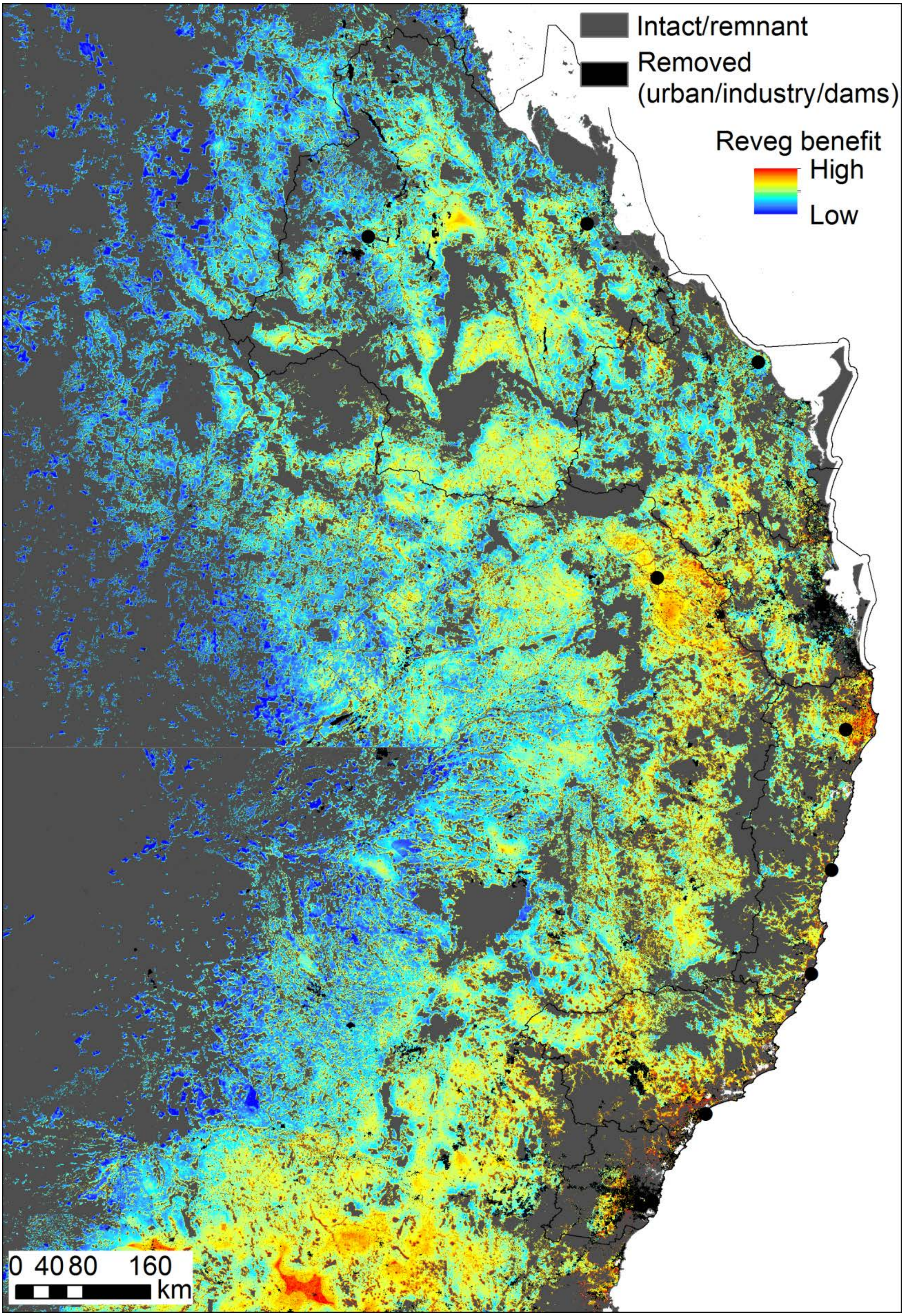
The forth rule of thumb injects a dimension correlated with species richness into the analysis. For this we used a dataset combining models of 504 threatened species at the continental scale: 355 plants, 48 birds, 22 amphibians, 30 reptiles and 49 mammals (Maggini et al 2013). The data were developed to assess potential impact of climate change on threatened species, and were combined elegantly with numerous other data sets to identify over 800000 km^2 of optimal locations for habitat protection and restoration. The authors' assessment of impact suggested that under a business as usual emissions scenario 60 percent of threatened species would no longer be suited to conditions across more than half of their current modelled ranges by 2085.



Map 15. Numbers of modelled threatened species for locations across eastern Australia (Maggini et al. 2013) used here as index of priority for revegetation under principle 4.

The analysis by Maggini et al. (2013) identifies the east coast of Australia as a most important current habitat for the threatened species in their analysis, and predicts it will be even more important in a warmer future (see Maggini et al. 2013 fig 19). However, the 'optimal' locations they identified for actions were all inland, away from places with the highest ecological values because of the strong effect of land cost on the optimal allocation of their three billion dollar budget. Despite this seemingly large budget their analysis suggested that further investment, beyond three billion dollars, would have continued to add value with no signs of diminishing returns. The dataset is a 1km grid with counts for numbers of species with modelled current habitat overlapping each cell.

Map 16 shows the sum of the spatial data representing the four principles described above.



Map 16. Index of biodiversity benefits from revegetation compiled from the four principles described above.

In combination, the four principles tend to highlight areas around the remaining patches and corridors in extensively cleared landscapes as offering the greatest benefit from restoration, especially in higher rainfall, eastern areas where threatened species are concentrated.

Riparian habitat networks in extensively modified landscapes feature quite clearly as areas offering significant benefit to biodiversity if revegetated. Floodplain restoration can be planned in ways that increase connectivity throughout the landscape by identifying remnants that can be linked along primary and secondary drainage lines. These linkages are particularly important in areas of high rainfall and soil fertility that have been heavily cleared of native vegetation, such as those that contain threatened ecological communities (e.g. central Hunter Valley eucalypt forest and woodland complex and lower Hunter Valley dry rainforest, Darling Downs grasslands and Brigalow in eastern areas).

3.4 Conclusions

Adding forest to landscapes will affect hydrology, climate and biodiversity at a range of spatial scales. These changes will often be beneficial, particularly where new forests reduce run-off and deep drainage to groundwater, and thereby reduce resultant transport of salt, soil and nutrients into aquatic systems.

New forests in cleared grassy landscapes will typically increase the mass of fuel available to burn and therefore increase the intensity of fire within the newly forested area. This general increase in likely fire intensity would increase risk to nearby fire-sensitive infrastructure and may therefore increase fire risk. However, forests in grassy landscapes also tend to slow the rate of spread of fire and reduce fire frequency, because they shift fuel characteristic away from fast drying and fast burning grass fuels to heavier and slower burning woody fuels. So the effect of carbon farming on fire risk is extremely complex to predict.

New forests should not be located close to fire-sensitive infrastructure (e.g. within 40-200m depending on characteristics of location and infrastructure). Beyond such local effects the literature offers little evidence for a general increase in fire risk across landscapes from carbon farming.

Rather than increasing fire risk, carbon farming may alter regional fire regimes by expanding the extent of land managed for fire suppression. Fires can generally be considered a threat to carbon stocks in vegetation-based carbon farming projects. So it is likely that an expansion of carbon farming might be a poor match for areas that are currently highly fire prone.

New forests offer beneficial habitat resources for native biota. There are myriad techniques that can be used to identify locations where revegetation would be most beneficial for biodiversity.

General principles that can guide the identification of beneficial locations for revegetation include complementarity for remaining vegetation, locations subject to extinction debt, connectivity to remaining natural habitat, and potential future habitat for threatened species.

Consideration of pre-clearing vegetation types and complementarity will continue to be useful indicators of priority for revegetation even though revegetation should not necessarily be expected to produce ecosystems analogous to pre-clearing vegetation. The aim should be to make a range of habitat types available to biodiversity, and land types heavily impacted by past clearing are generally indicators of soil and other conditions that are relatively uncommon among areas of remnant vegetation. Opportunities to revegetate or otherwise reduce landuse intensity in such places should be sought and supported to conserve biodiversity.

References

Bari, N.A., and Schofield, N.J. (1992) Lowering of a shallow, saline water table by extensive eucalypt reforestation. *Journal of Hydrology* 133: 273-291.

Beale, G., Gilmore R., Simons M., Realica, S., and Nanadakumar N. (2001) NSW coastal rivers salinity audit: predictions for the Hunter Valley – issue 1: December 2000. NSW Department of Land and Water Conservation, Wagga Wagga, Australia.

Bell, R.W., Schofield, N.J., Loh, I.C., and Bari, M.A. (1990) Groundwater response to reforestation in the Darling Range of Western Australia. *Journal of Hydrology* 119: 179-200.

Biggs, A. (2007) The landscape and marginal cropping lands – inherent and introduced. *Tropical Grasslands* 41: 133-138.

Biggs, A.J.W., and Mottram, P. (2008) Links between dryland salinity, mosquito vectors, and Ross River Virus disease in southern inland Queensland – an example and potential implications. *Australian Journal of Soil Research* 46: 62-66.

Birk, E.M., and Bridges, R.G. (1989) Recurrent fires and fuel accumulation in even-aged blackbutt (*Eucalyptus pilularis*) forests. *Forest Ecology and Management* 29: 59-79.

Blanchi, R., Lucas, C., Leonard, J., and Finkele, K. (2010) Meteorological conditions and wildfire-related house loss in Australia. *International Journal of Wildland Fire* 19: 914-926.

Bradshaw, C.J.A., Bowman, D.M.J.S., Bond, N.R., Murphy, B.P., Moore, A.D., Fordham, D.A., Thackway, R., Lawes, M.J., McCallum, H., Gregory, S.D., Dalal, R.C., Boer, M.M., Lynch, A.J.J., Bradstock, R.A., Brook, B.W., Henry, B.K., Hunt, L.P., Fisher, D.O., Hunter, D., Johnson, C.N., Keith, D.A., Lefroy, E.c., Penman, T.D., Meyer, W.S., Thompson, J.R., Thornton, C.M., VanDerWal, J., Williams, R.J., Keniger, L., Specht, A. (2013) Brave new world – consequences of a carbon economy for the conservation of Australian biodiversity. *Biological Conservation* 161: 71-90.

Bradstock, R.A. (2010) A biogeographic model of fire regimes in Australia: current and future implications. *Global Ecology and Biogeography* 19: 145-158.

Bradstock, R.A., Boer, M.M., Cary, G.J., Price, O.F., Williams, R.J., Barrett, D., Cook, G., Gill, A.M., Hutley, L.B.W., Keith, H., Maier, S.W., Meyer, M., Roxberg, S.H., and Russell-Smith, J. (2012) Modelling the potential for prescribed burning to mitigate carbon emissions from wildfires in fire prone forests of Australia. *International Journal of Wildland Fire* 21: 629-639.

Bren, L.J. (1992) Tree invasion of an intermittent wetland in relation to changes in the flooding frequency of the River Murray, Australia. *Australian Journal of Ecology* 17: 395-408.

Brooks (2011) Extinction: consider all species. *Nature* 474: 284 (16 June 2011) doi:10.1038/474284b

Bui, E.N., and Henderson, B.L. (2003) Vegetation indicators of salinity in northern Queensland. *Austral Ecology* 28: 539-552.

Burns, K., Hug, B., Lawson, K., Ahammad, H., and Zhang, K. (2011) *Abatement potential from reforestation under selected carbon price scenarios,* ABARES Special Report, Canberra, July.

Bushnell, S., Balcombe, L., and Cottrell, A. (2007) Community and fire service perceptions of bushfire issues in Tamborine Mountain: what's the difference? *The Australian Journal of Emergency Management* 22: 3-9.

Butler, D.W. (2009) Planning iterative investment for landscape restoration: choice of biodiversity indicator makes a difference. *Biological Conservation* 142: 2202-2216.

Butler, D.W., and Fairfax, R.J. (2003) Buffel Grass and fire in a Gidgee and Brigalow woodland: a case study from central Queensland. *Ecological Management and Restoration* 4: 120-125.

Chafer, C.J., Noonan, M., and Macnaught, E. (2004) The post-fire measurement of fire severity and intensity in the Christmas 2001 Sydney wildfires. *International Journal of Wildland Fire* 13: 227-240.

Charman, P.E.V., and Junor, R.S. (1989) Saline seepage and land degradation – a New South Wales perspective. *Journal of Australian Geology and Geophysics* 11: 195-203.

Chazdon, R.L. (2008) Beyond deforestation: Restoring forests and ecosystem services on degraded lands. *Science* 320: 1458–1460.

Cheney, N.P., Gould, J.S., and Catchpole, W.R. (1998) Prediction of fire spread in grasslands. *International Journal of Wildland Fire* 8: 1-13.

Cheney, P., and Sullivan, A. (2008) Grassfires: fuel, weather and fire behaviour. Second edition. CSIRO Publishing, Collinwood, Australia.

Childs, I., Pritchard, F., Gow, K., and Hastings, P. (2006) Bushfire preparedness of rural-residents in selected Southeast Queensland Shires. Proceedings of the Social Change in the 21st Century Conference 2006, Queensland University of Technology, Carseldine, Australia. Available at: http://eprints.qut.edu.au/6084/1/6084.pdf Accessed: August 8th, 2014.

Clarke, H., Lucas, C., and Smith, P. (2013) Changes in Australian fire weather between 1973 and 2010. *International Journal of Climatology* 33: 931-944.

Clarke, H.G., Smith, P.L., and Pitman, A.J. (2011) Regional signatures of future fire weather over eastern Australia from global climate models. *International Journal of Wildland Fire* 20: 550-562.

Collins, L., Penman, T., Price, O., and Bradstock, R. (2013) *Assessment of the potential effect of revegetation on fire regimes and fire risk within agricultural landscapes of the Hawkesbury-Nepean catchment area*. Report to Hawkesbury-Nepean Catchment Management Authority, Goulburn, Australia. University of Wollongong. Comerford, E., Norman, P., Le Grand, J. (2011) *Report on the project: economics of carbon forestry in Queensland*. Queensland Department of Environment and Resource Management, Brisbane.

Department of the Environment (2014) *National Inventory Report 2012 Volume 2.* Commonwealth of Australia. Canberra. Available at: http://ageis.climatechange.gov.au/

Cresswell, R.G. (2006) Dryland salinity in Queensland: how do we compare? In "Regolith 2006 - Consolidation and dispersion of ideas: proceedings of the CRC LEME Regional Regolith Symposia, Hahndorf Resort, South Australia", eds. R.W. Fitzpatrick, & P. Shand, pp. 35-41. Cooperative Research Centre for Landscape Environments and Mineral Exploration, Canberra, Australia.

Connor, L., Albrecht, G., Higginbotham, N., Freeman, S., and Smith, W. (2004) Environmental change and human health in upper Hunter communities of New South Wales, Australia. *Eco Health* 1: 47-58.

Crossman, N.D., and Bryan, B.A. (2006) Systematic landscape restoration using integer programming. *Biological Conservation* 128: 369-383.

Crossman, N.D., Bryan, B.A., and Summers, B.N. (2011) Carbon payments and low cost conservation. *Conservation Biology* 225: 835–845.

Department of the Environment (2014) *National Inventory Report 2012 Volume 2.* Commonwealth of Australia, Canberra.

Deo, R.C., Syktus, J.I., McAlpine, C.I., Lawrence, P.J., McGowan, H.A., and Phinn, S.R. (2009) Impact of historical land cover change on daily indices of climate extremes including droughts in eastern Australia. *Geophysical Research Letters* 36: L08705, doi:10.1029/2009GL037666

Dixon, W., and Chiswell, B. 1992. The use of hydrochemical sections to identify recharge areas and saline intrusions in alluvial aquifers, southeast Queensland, Australia. *Journal of Hydrology* 135: 259-274.

Drielsma, M., Ferrier, S., and Manion, G. (2007) A raster-based technique for analysing habitat configuration: The cost–benefit approach. *Ecological Modelling* 202: 324–332.

Michael Drielsma, M., Manion, G., Love, J., Williams, K.J., Harwood, T. and Saremi, H. (2014) *Draft 3C Modelling; for biodiversity management under future climate*. Office Of Environment

And Heritage NSW. Available at https://www.terranova.org.au/repository/3c-modelling-east-coast-central-slopes-and-murray-basin-nrm-collection/draft-report-3c-modelling-for-biodiversity-management-under-future-climate/3c-draft-201114.pdf/viewDunlop, M., Parris, H., Ryan, P., and Kroon, F. (2013) Climate-ready conservation objectives: a scoping study, National Climate Change Adaptation Research Facility, Gold Coast, pp.102.

Eady, S.J., Grundy, M., Battaglia, M., Keating, B. (2009) *An analysis of greenhouse gas mitigation and carbon biosequestration opportunities from rural land use.* CSIRO Sustainable Agriculture Flagship, Brisbane. Available at http://www.csiro.au/resources/carbon-and-rural-land-use-report.html

Evans, M.C., Cardwine, J., Fensham, R.J., Butler, D.W., Wilson, K.A., Possingham, H.P., and Martin, T.G. (submitted). Carbon farming via assisted natural regeneration as a cost-effective mechanism for restoring biodiversity in agricultural landscapes. *Environmental Science and Policy*.

Evans, T.R. (1967) A salinity problem in pasture establishment in the coastal lowlands of southern Queensland. *Journal of the Australian Institute of Agricultural Sciences* 33: 216-218.

Fabricius, K., De'ath, G., McCook, L., Turak, E., and Williams, D.McB. (2005) Changes in algal, coral and fish assemblages along water quality gradients on the inshore Great Barrier Reef. *Marine Pollutants Bulletin* 51: 384-398.

Fensham, R.J., and Fairfax, R.J. (2003) A land management history for central Queensland, Australia as determined from land-holder questionnaire and aerial photography. *Journal of Environmental Management* 68: 409-420.

Fensham, R.J., and Guymer, G.P. (2009) Carbon accumulation through ecosystem recovery. *Environmental Science and Policy* 12: 367-372.

Ferrier, S., and Wintle, B.A. (2009) Quantitative approaches to spatial conservation prioritisation: matching the solution to the need, eds. A. Moilanen, K.A. Wilson & H.P. Possingham pp.1-13. *Spatial Conservation Prioritisation: quantitative methods & computational tools*. Oxford University Press, Oxford, U.K.

Forster, B.A. (2007) Review of salinity occurrences in the Fitzroy Basin. Department of Natural Resources and Water, Indooroopilly, Australia.

Gibbons, P., van Bommel, L., Gill, A.M., Cary, G.J., Driscoll, D.A., Bradstock, R.A., Knight, E., Moritz, M.A., Stephens, S.L., Lindenmayer, D.B. (2012) Land management practices associated with house loss in wildfires. *Plos One* 7: e29212.

Gill, A.M. (2009) Fire, science and society at the urban-rural interface. *Proceedings of the Royal Society of Queensland* 115: 153-160.

Gill, A.M., King, K.J., and Moore, A.D. (2010) Australian grassland fire danger using inputs from the GRAZPLAN grassland simulation model. *International Journal of Wildland Fire* 19: 338-345.

Hanski I. (1999) *Metapopulation Ecology*. Oxford University Press, Oxford.

Hanski, I. (2011) Habitat loss, the dynamics of biodiversity, and a perspective on conservation. *Ambio* 40: 248-255.

Hennessy, K., Lucas, C., Nicholls, N., Bathols, J., Suppiah, R., and Ricketts, J. (2005) Climate change impacts on fire weather in south-east Australia. Commonwealth Scientific and Industrial Research Organisation, Melbourne, Australia.

Hughes, K.K. (1984) Trees and Salinity. *Queensland Agricultural Journal* 110: 13-14.

Hurteau, M.D., Koch, G.W., and Hungate, B.A. (2008) Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. *Frontiers in Ecology and the Environment* 6: 493-498.

Kesteven, J., Landsberg, J., and URS Australia (2004) National carbon accounting system: developing a national forest productivity model. Technical Report No. 23. Australian Greenhouse Office, Canberra.

Kroon, F.J., Kuhnert, P.M., Henderson, B.C., Wilkinson, S.N., Kinsey-Henderson, A., Abbott, B., Brodie, J.E., and Turner, R.D.R. (2012) River loads of suspended solids, nitrogen, phosphorous, and herbicides delivered to the Great Barrier Reef lagoon. *Marine Pollution Bulletin* 65: 167-181.

Lamb, D., Erskine, P.D., and Parrotta, J.A. (2005) Restoration of degraded tropical forest landscapes. *Science* 310: 1628-1632.

Lin, B.B., Macfadyen, S., Renwick, A.R., Cunningham, S.A., and Schellhorn, N.A. (2013) Maximizing the environmental benefits of carbon farming through ecosystem service delivery. *Bioscience* 63: 793-803.

Loch, R.J. (2000) Effects of vegetation cover on runoff and erosion under simulated rain and overland flow on a rehabilitated site on the Meandu Mine, Tarong, Queensland. *Australian Journal of Soil Research* 38: 299-312.

Maggini, R., Kujala, H., Taylor, M.F.J., Lee, J.R., Possingham, H.P., Wintle, B.A. & Fuller, R.A. (2013) Protecting and restoring habitat to help Australia's threatened species adapt to climate change, National Climate Change Adaptation Research Facility, Gold Coast, 54pp.

Margules, C.R., Pressey, R.L., and Williams, P.H. (2002) Representing biodiversity: data and procedures for identifying priority areas for conservation. *Journal of Biosciences* 27: 309-326.

Marinoni, O., Navarro Garcia, J., Marvanek, S., Prestwidge, D., Clifford., D., and Laredo, L.A. (2012) Development of a system to produce maps of agricultural profit on a continental scale: An example for Australia. *Agricultural Systems* 105: 33-45.

Matthews, S., Sullivan, A.L., Watson, P., and Williams, R.J. (2012) Climate change, fuel and fire behaviour in a eucalypt forest. *Global Change Biology* 18: 3212-3223.

McAlpine, C.A., Syktus, J., Deo, R.C., Lawrence, P.J., McGowan, H.A., Watterson, I.G., and Phinn, S.R. (2007) Modeling the impact of historical land cover change on Australia's regional climate. Geophysical Research Letters 34: L22711.

McAlpine, C.A., Syktus, J., Ryan, J.G., Deo, R.C., McKeon, G.M., McGowan, H.A., and Phinn, S.R. (2009) A continent under stress: interactions, feedbacks and risks associated with impact of modified land cover on Australia's climate. *Global Change Biology* 15: 1365-2486.

McKeon, G.M., Stone, G.S., Syktus, J.I., Carter, J.O., Flood, N.R., Ahrens, D.G., Bruget, D.N., Chilcott, C.R., Cobon, D.H., Cowley, R.A., Crimp, S.J., Fraser, G.W., Howden, S.M., Johnston, P.W., Ryan, J.G., Stokes, C.J., and Day, K.A. (2009) Climate change impacts on northern Australian rangeland livestock carrying capacity: a review of issues. *The Rangeland Journal* 31: 1–29.

Meir, E., Andelman, S., Possingham, H.P. (2004)Does conservation planning matter in an uncertain world? *Ecology Letters* 7: 615-612.

Moilanen, A. (2008) Two paths to a suboptimal solution – once more about optimality in reserve selection. Biological Conservation 141: 1919-1923.

Murdiyarso, D., Widodo, M., and Suyamto, D. (2002) Fire risks in forest carbon projects in Indonesia. *Science in China (Series C)* 45: 65-74.

Murphy, B.P., Bradstock, R.A., Boer, M.M., Carter, J., Cary, G.J., Cochrane, M.A., Fensham, R.J., Russell-Smith, J., Williamson, G.J., and Bowman, D.M.J.S. (2013) Fire regimes of Australia: a pyrogeographic model system. *Journal of Biogeography* 40: 1048-1058.

Noss, R.F., Beier, P., Covington, W.W., Grumbine, R.E., Lindenmayer, D.B., Prather, J.W., Schmiegelow, F., Sisk, T.D. and Vosick, D.J. (2006) Recommendations for integrating restoration ecology and conservation biology in ponderosa pine forests of the southwestern United States. *Restoration Ecology* 14: 4–10.

Packett, R., Dougall, C., Rohde, K., and Noble, R. (2009) Agricultural lands are hotspots for annual runoff polluting the southern Great Barrier Reef lagoon. *Marine Pollution Bulletin* 58: 976-986.

Paton, D., Kelly, G., Burgelt, P.T., and Doherty, M. (2006) Preparing for bushfires: understanding intentions. *Disaster Prevention and Management* 15: 566-575.

Penman, T.D., and York, A. (2010) Climate and recent fire history affect fuel loads in *Eucalyptus* forests: implications for fire management in a changing climate. *Forest Ecology and Management* 260: 1791-1797.

Polglase, P., Reeson, A., Hawkins, C., Paul, K., Siggins, A., Turner, J., Crawford, D., Jovanovic, T., Hobbs, T., Opie, K., Carwardine, J., and Almeida, A. (2011) *Opportunities for carbon forestry in Australia: Economic assessment and constraints to implementation*. CSIRO, Canberra.

Polglase, P.J., Reeson, A., Hawkins, C.S., Paul, K.I., Siggins, A.W., Turner, J., Crawford, D.F., Jovanovic, T., Hobbs, T.J., Opie, K., Carwardine, J., and Almeida, A. (2013) Potenital for forest carbon plantings to offset greenhouse emissions in Australia:economics and constraints to implementation. *Climatic Change* 121: 161-175.

Pressey, R.L. and Nicholls, A.O. (1989) Efficiency in conservation evaluation: Scoring versus iterative approaches. *Biological Conservation* 50: 199-218.

Price, O.F., and Bradstock, R.A. (2010) The effect of fuel age on the spread of fire in sclerophyll forest in the Sydney region of Australia. *International Journal of Wildland Fire* 19: 35-45.

Price, C., and Rind, D. (1994) Possible implications of global climate change on global lightning distributions and frequencies. *Journal of Geophysical Research* 99: 10823-10831.

Queensland Department of Science, Information Technology, Innovation and the Arts. (2014) Land cover change in Queensland 2011–12: a Statewide Landcover and Trees Study (SLATS) report. DSITIA, Brisbane.

Queensland Herbarium (2014) Queensland Herbarium specimen label database (HERBRECS).

Radford, B.J., Silburn D.M., and Forster, B.A. (2009) Soil chloride and deep drainage responses to land clearing for cropping at seven sites in central Queensland, northern Australia. *Journal of Hydrology* 379: 20-29.

Rhodes, A. (2003) Understanding community preparedness and response to wildfire. Safer Sustainable Communities 2003 Australian Disaster Conference, Canberra, Australia. Emergency Management Australia. Available at: http://unpan1.un.org/intradoc/groups/public/documents/APCITY/UNPAN019478.pdf Accessed: August 8th, 2014.

Richards, G., and Brack, C. (2004) A continental biomass stock and stock exchange estimation approach for Australia. *Australian Forestry* 67: 284-288.

Ridge, R. (2005) Salinity in the Burnett- Mary Region. In: Country to coast – a healthy sustainable future, vol 5J- priority action plan 3.2 farming land systems. Burnett Mary Regional Group. Available at: http://www.bmrg.org.au/files/4513/9407/3659/Volume_5J_PAP3.2Oct_2005_Paper_Only_for_Website_Low_Res.pdf Accessed: November 2nd, 2014.

Russell-Smith, J., Yates, C.P., Whitehead, P.J., Smith, R., Craig, R., Allan, G.E., Thackway, R., Frakes, I., Cridland, S., Meyer, M.C.P., and Gill, A.M. (2007) Bushfires 'down under': patterns and implications of contemporary Australian landscape burning. *International Journal of Wildland Fire* 16: 361-371.

Rybicki, J., and Hanski, I. (2013) Species-area relationships and extinctions caused by habitat loss and fragmentation. *Ecology Letters* 16: 27-38.

Setterfield, S.A., Rossiter-Rachor, N.A., Hutley, L.B., Douglas, M.M., and Williams, R.J. (2010) Turning up the heat: the impacts of *Andropogon gayanus* (gamba grass) invasion on fire behaviour in northern Australian savannas. *Diversity and Distributions* 16: 854-861.

Silburn, D.M., Cowie, B.A., and Thornton, C.M. (2009) The Brigalow Catchment Study revisited: effects of land development on deep drainage determined from non-steady chloride profiles. *Journal of Hydrology* 373: 487-498.

Siriwardena, L., Finlayson, B.L., and McMahon, T.A. (2006) The impact of land use change on catchment hydrology in large catchments: the Comet River, Central Queensland, Australia. *Journal of Hydrology* 326: 199-214.

Soule, M.E., and Sanjayan, M.A. (1998) Conservation targets: do they help? *Science* 279: 2060-2061.

Sullivan, A.L., McCaw, W.L., Cruz, M.G., Matthews, S., and Ellis, P.F. (2012) Fuel, fire weather and fire behaviour in Australian ecosystems. In “Flammable Australia: fire regimes, biodiversity and ecosystems in a changing world” eds. R. A. Bradstock, A. M. Gill and R. J. Williams, pp. 51-77. CSIRO Publishing, Collingwood, Australia.

Szabo, J.K., Vesk, P.A., Baxter, P.W.J., and Possingham, H.P. (2011) Paying the extinction debt: woodland birds in the Mount Lofty Ranges, South Australia. *Emu* 111: 59-70.

Thackway, R., and Lesslie, R. (2008) Describing and mapping human-induced vegetation change in the Australian landscape. *Environmental Management* 42: 572-590.

Thorburn, P.J., Cowie, B.A., and Lawrence, P.A. (1991) Effect of land development on groundwater recharge determined from non-steady chloride profiles. *Journal of Hydrology* 124: 43-58.

Tilman, D., May, R.M., Lehman, C.L., and Nowak, M.A. (1994) Habitat destruction and the extinction debt. *Nature* 371: 65-66.

Tockner, K., Pusch, M., Borchardt, D., and Lorang, M.S. (2010) Multiple stressors in coupled river-floodplain ecosystems. *Freshwater Biology* 55: 135-151.

Whalley, R.D.B., Price, J.N., MacDonald, M.J., and Berney, P.J. (2011) Drivers of change in the social-ecological systems of the Gwydir Wetlands and Macquarie Marshes in northern New South Wales, Australia. *The Rangeland Journal* 33: 109-119.

Whited, D.C., Lorang, M.S., Harner, M.J., Hauer, F.R., Kimball, J.S., and Stanford, J.A. (2007) Climate, hydrological disturbance, and succession: drivers of floodplain pattern. *Ecology* 88: 940-953.

Williams, A.A.J., Karoly, D.J., and Tapper, N. (2001) The sensitivity of Australian fire danger to climate change. *Climate Change* 49: 171-191.

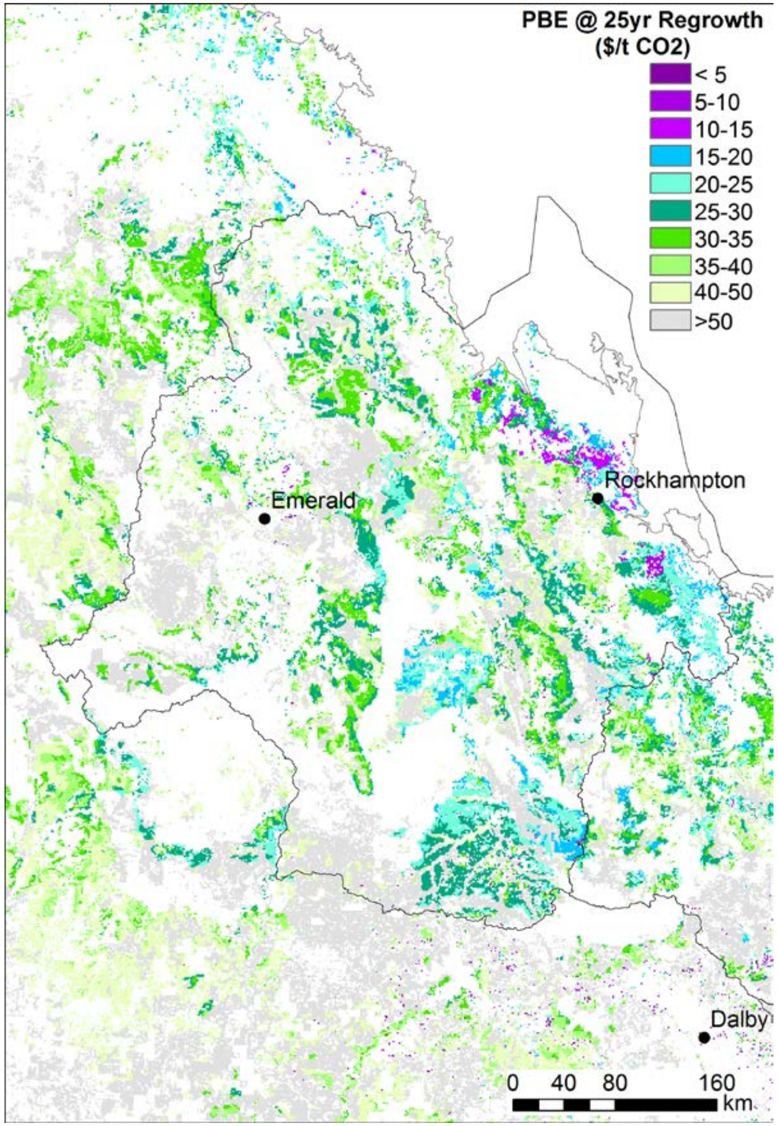
Wilson, K.A., Lulow, M., Burger, J., Fang, Y.-C., Andersen, C., Olsen, D., O’Connell, M., and McBride, M.F. (2011) Optimal restoration: accounting for space, time and uncertainty. Journal of Applied Ecology 48: 715-725.

Wylie, F.R., Johnston, P.J.M., and Forster, B.A. (1993) Decline of *Casuarina* and *Eucalyptus* in the Mary River catchment. Forest Research Institute Research Paper No. 17. Queensland Department of Primary Industries, Brisbane, Australia.

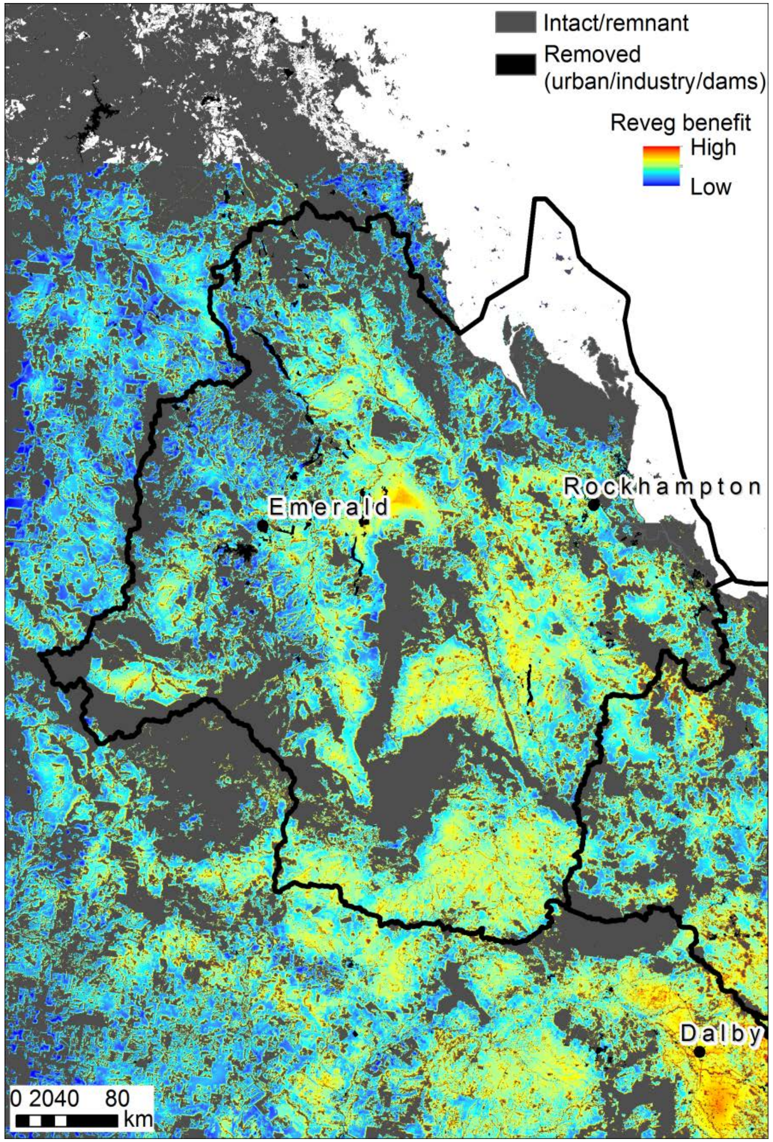
Yeo, A. (1999) Predicting the interaction between the effects of salinity and climate change on crop plants. *Scientia Hoticulturae* 78: 159-174.

Zhang, L., Dawes, W.R., and Walker, G.R. (2001) Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research* 37: 701-708.

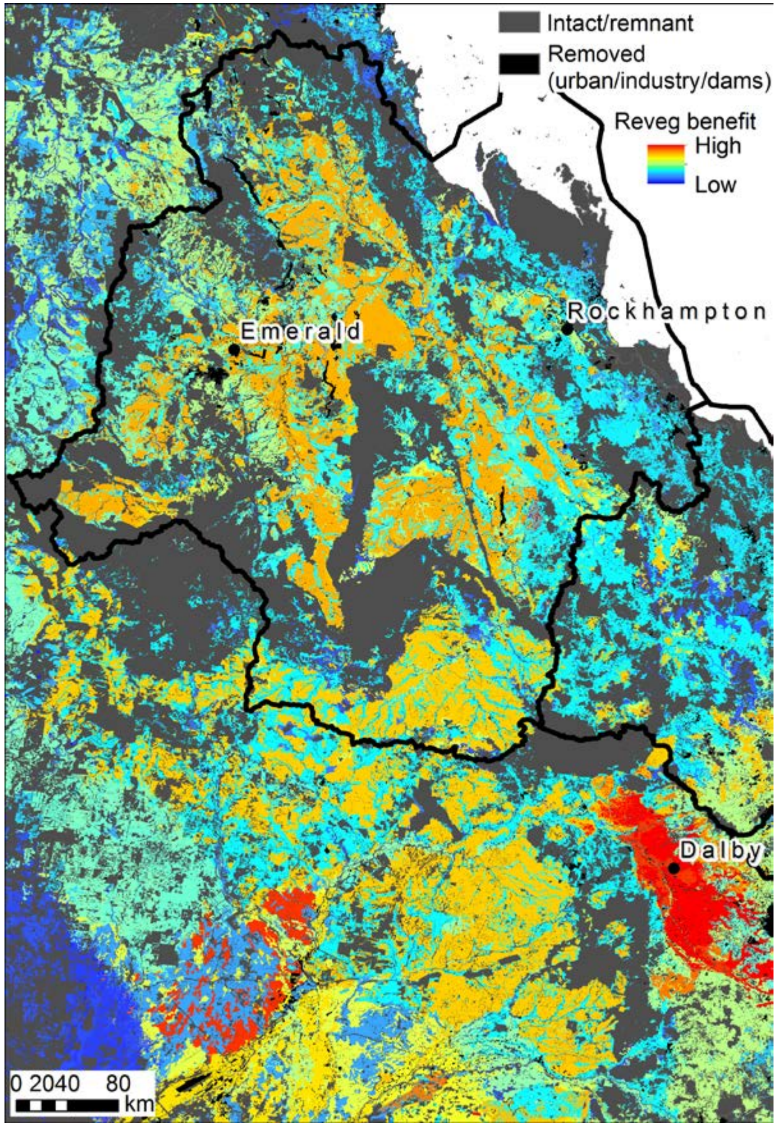
Appendix 1. Maps for each NRM region



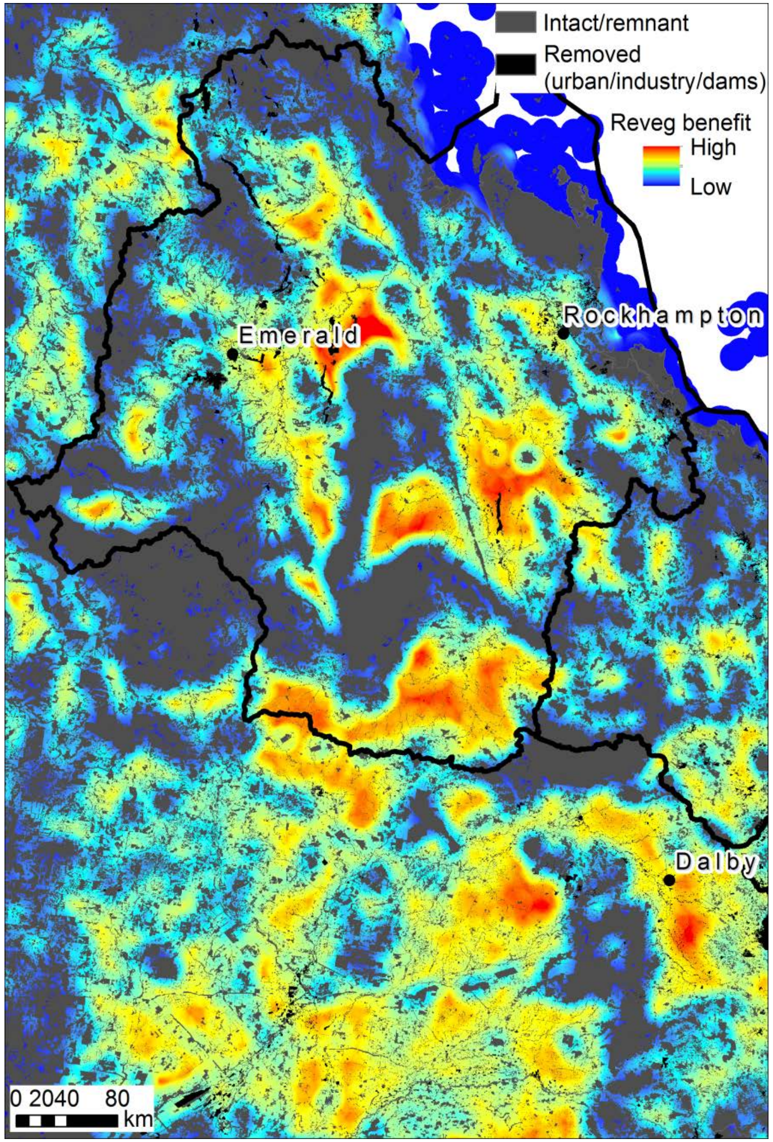
Map 17. FBA - ACCU price to break-even over 25 year investment period for regrowth projects with 100 year permanence



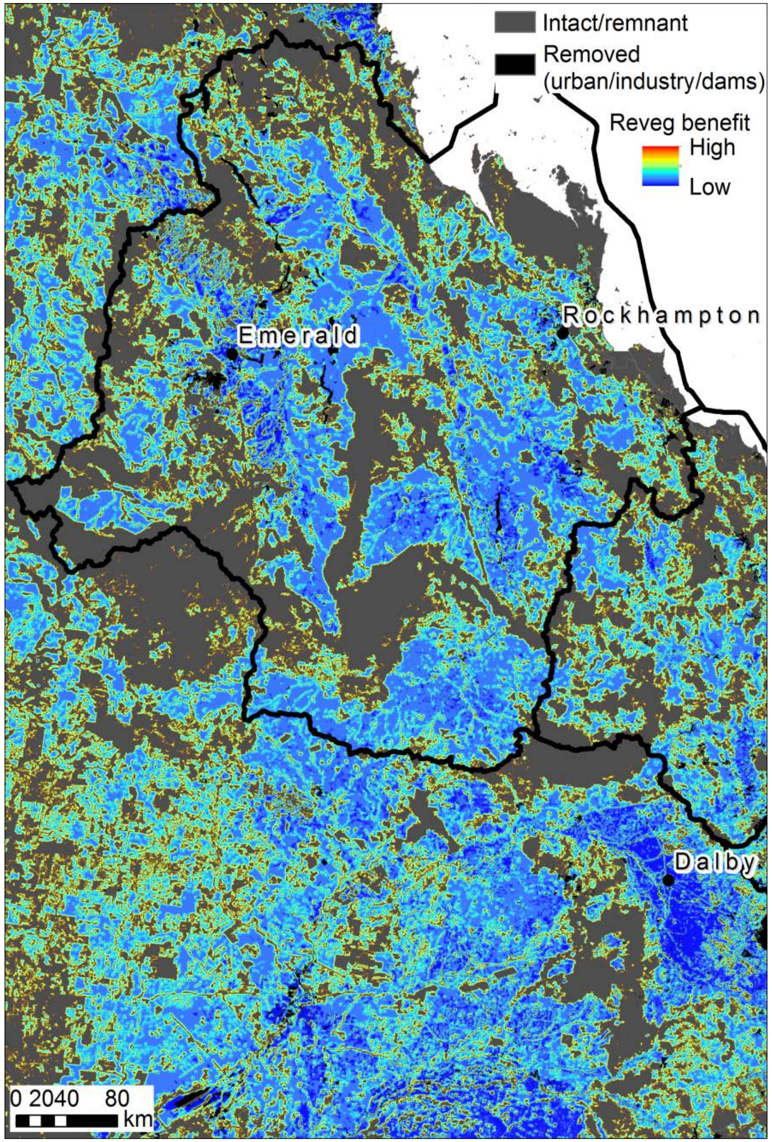
Map 18. FBA - Revegetation benefit metric (this study)



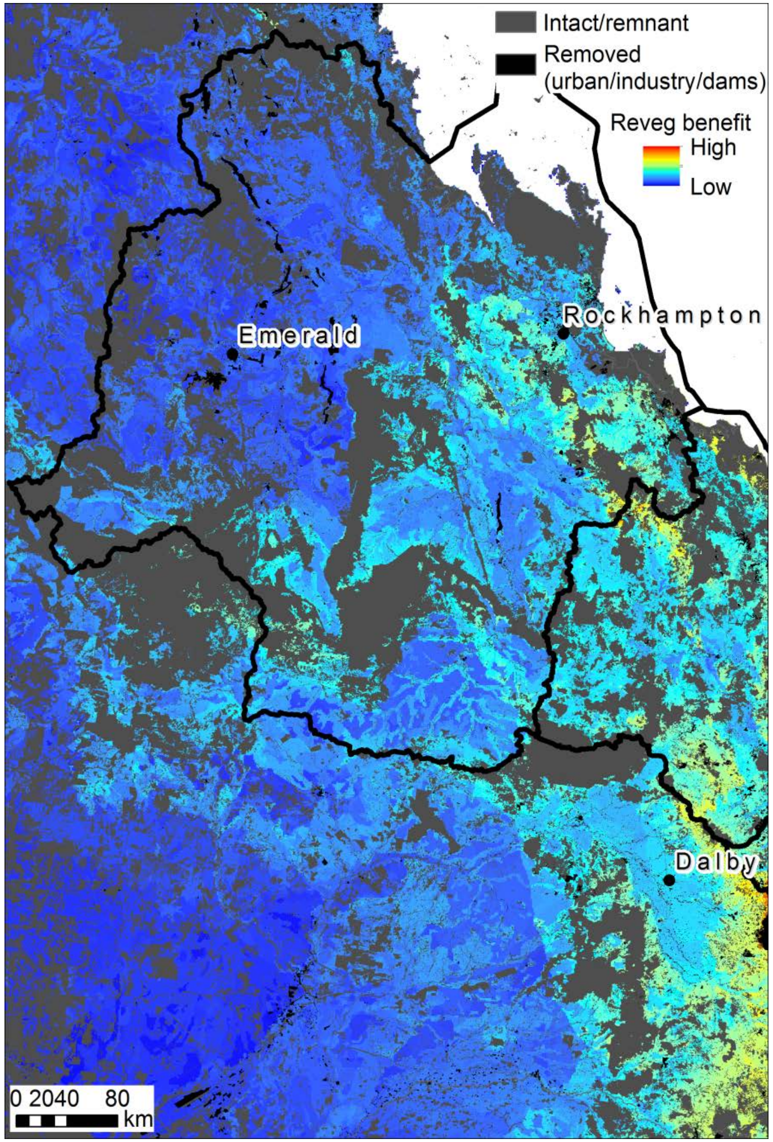
Map 19. FBA - Land type/pre-clearing veg benefit



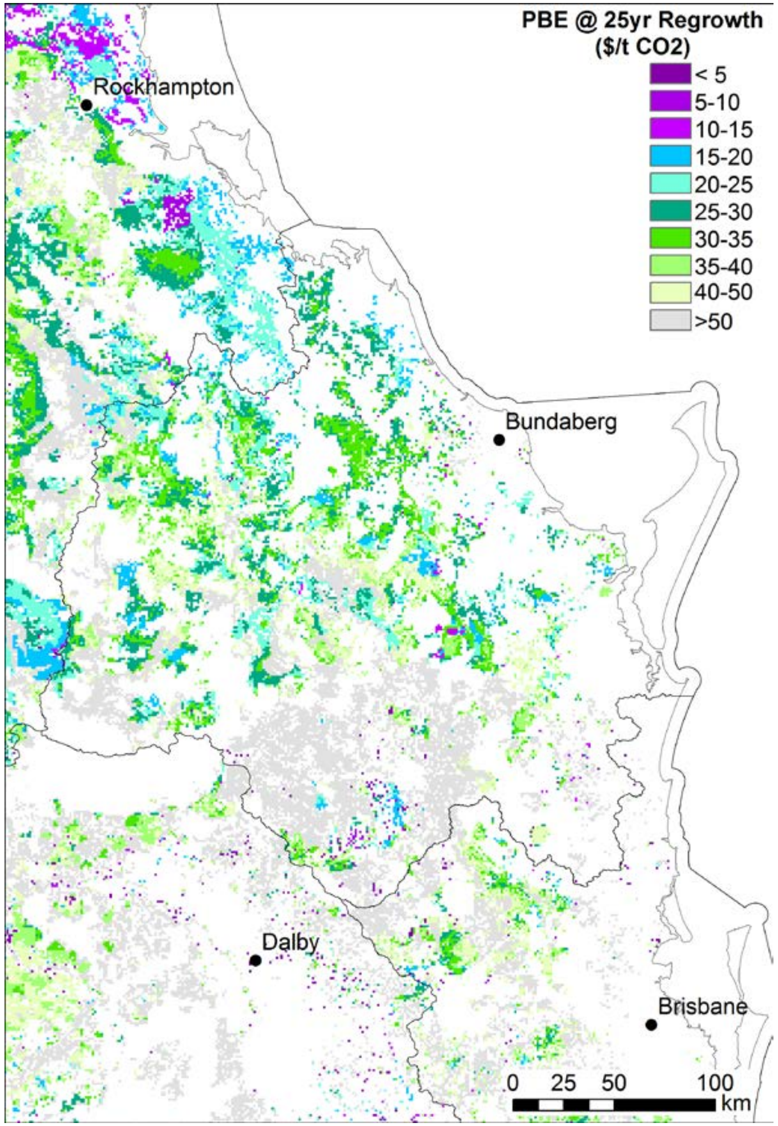
Map 20. FBA - Remnant in 10km benefit



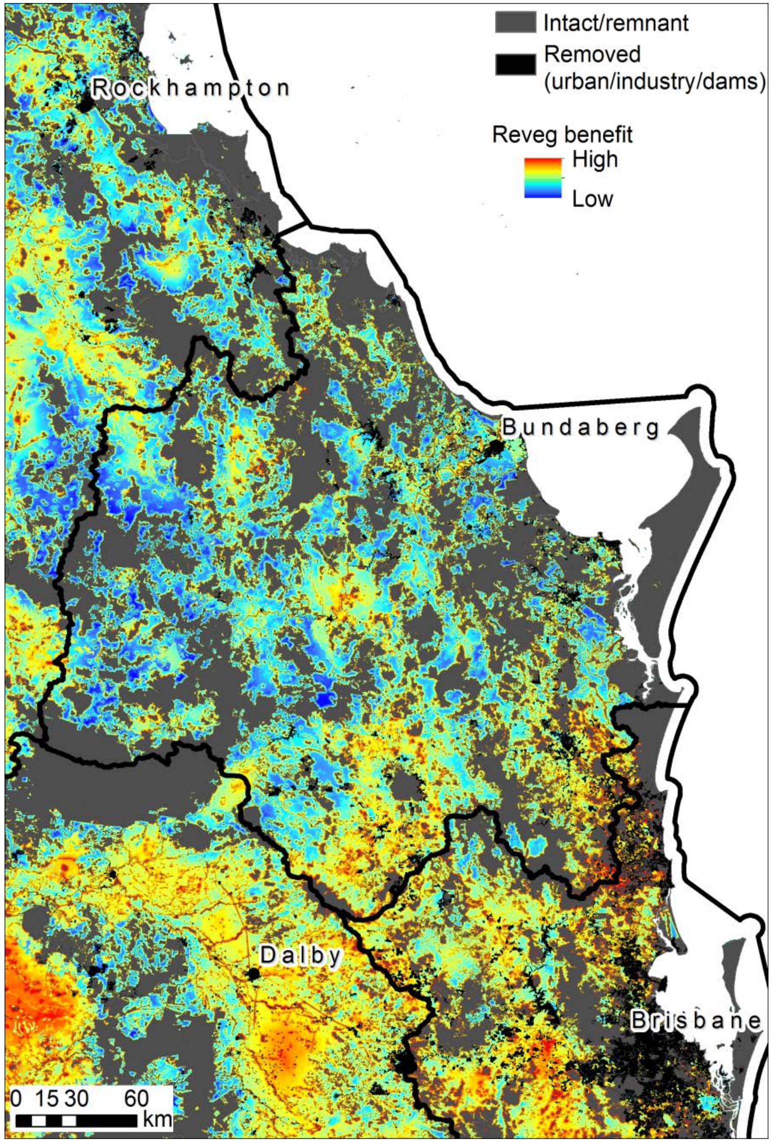
Map 21. FBA - Neighbourhood habitat benefit



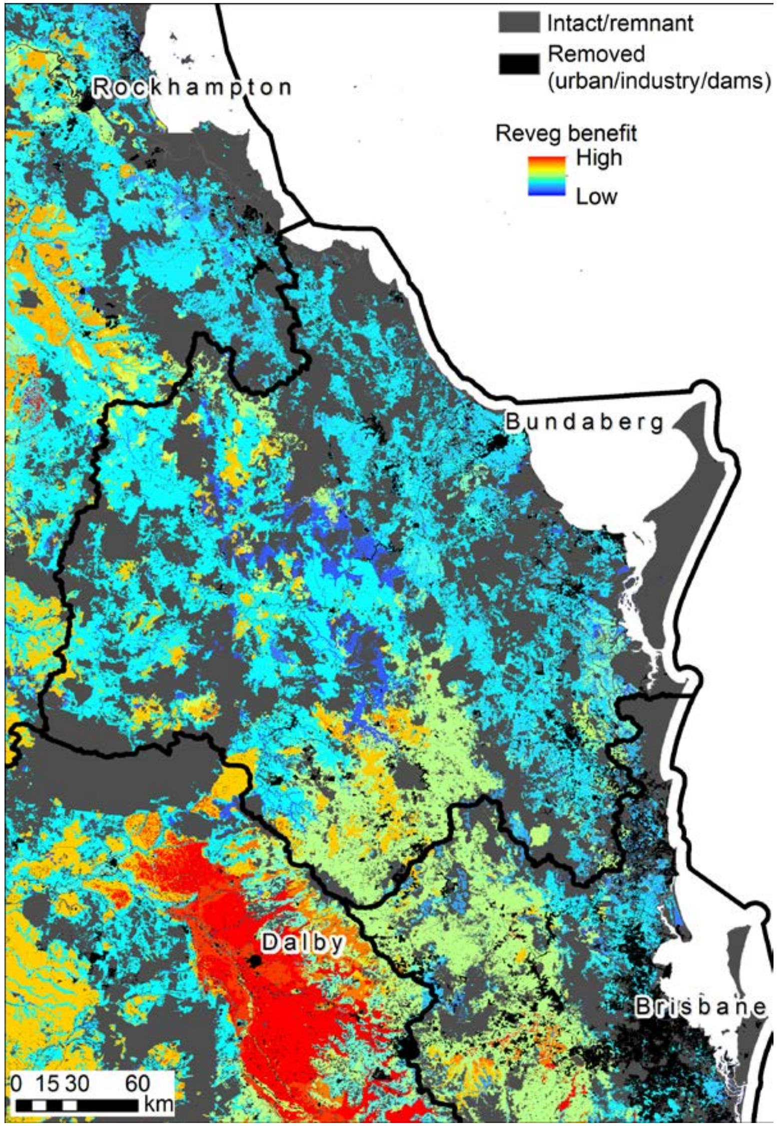
Map 22. FBA - Threatened species benefit



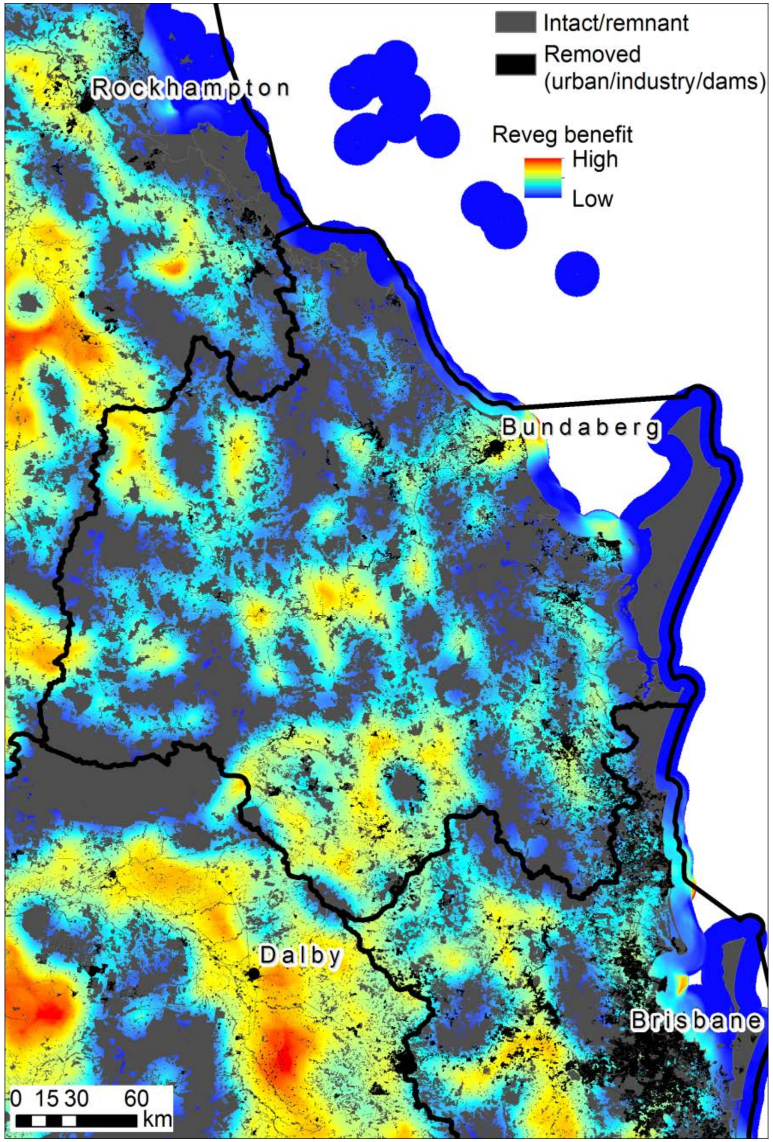
Map 23. BMRG - ACCU price to break-even over 25 year investment period for regrowth projects with 100 year permanence



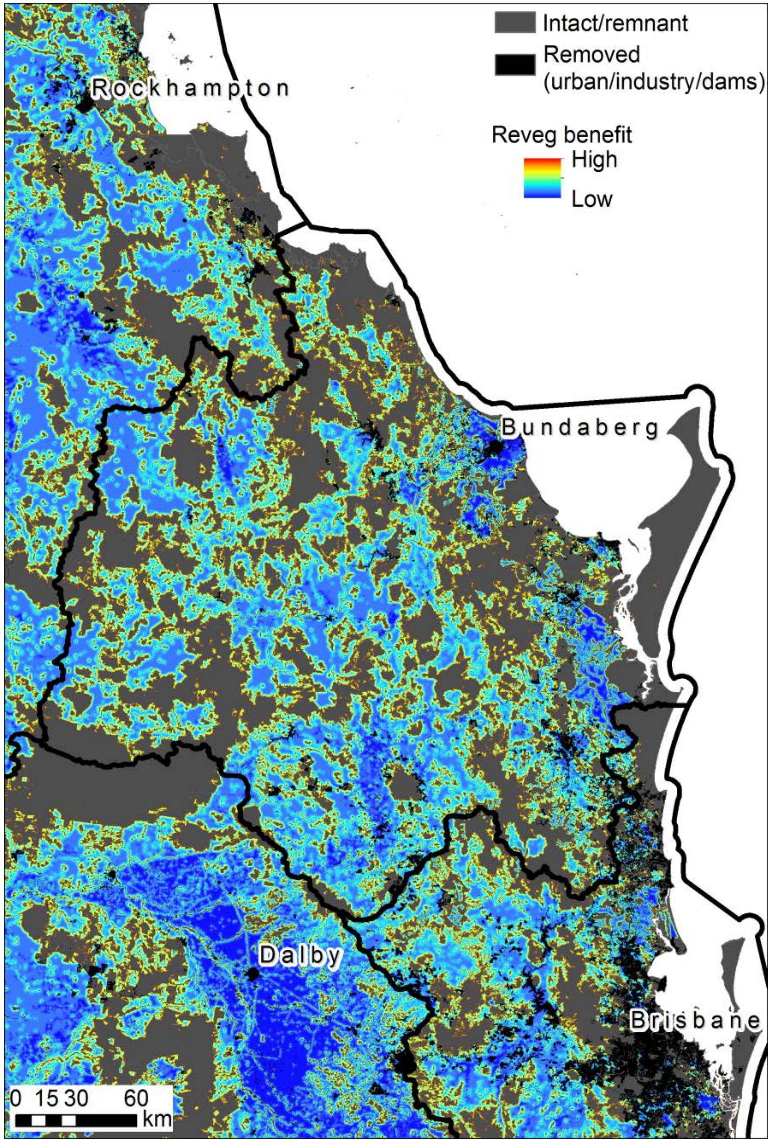
Map 24. BMRG - Revegetation benefit metric



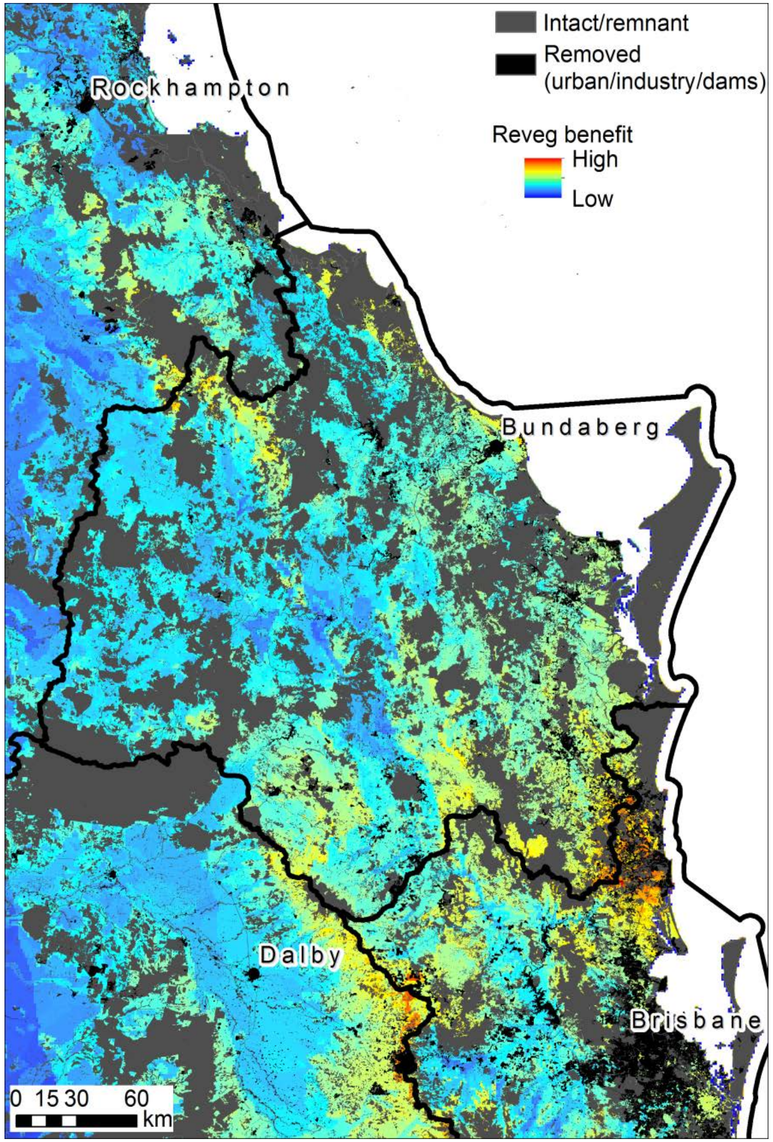
Map 25. BRMG - Land type/pre-clearing veg benefit



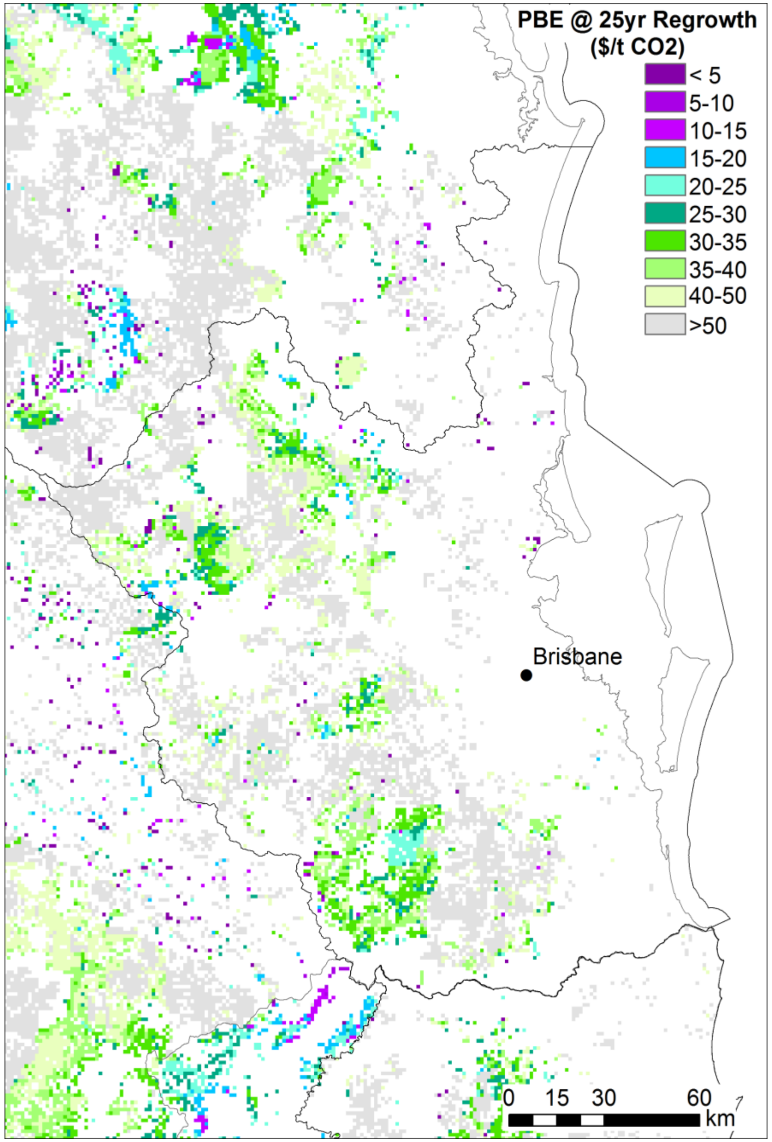
Map 26. BMRG - Remnant in 10km benefit



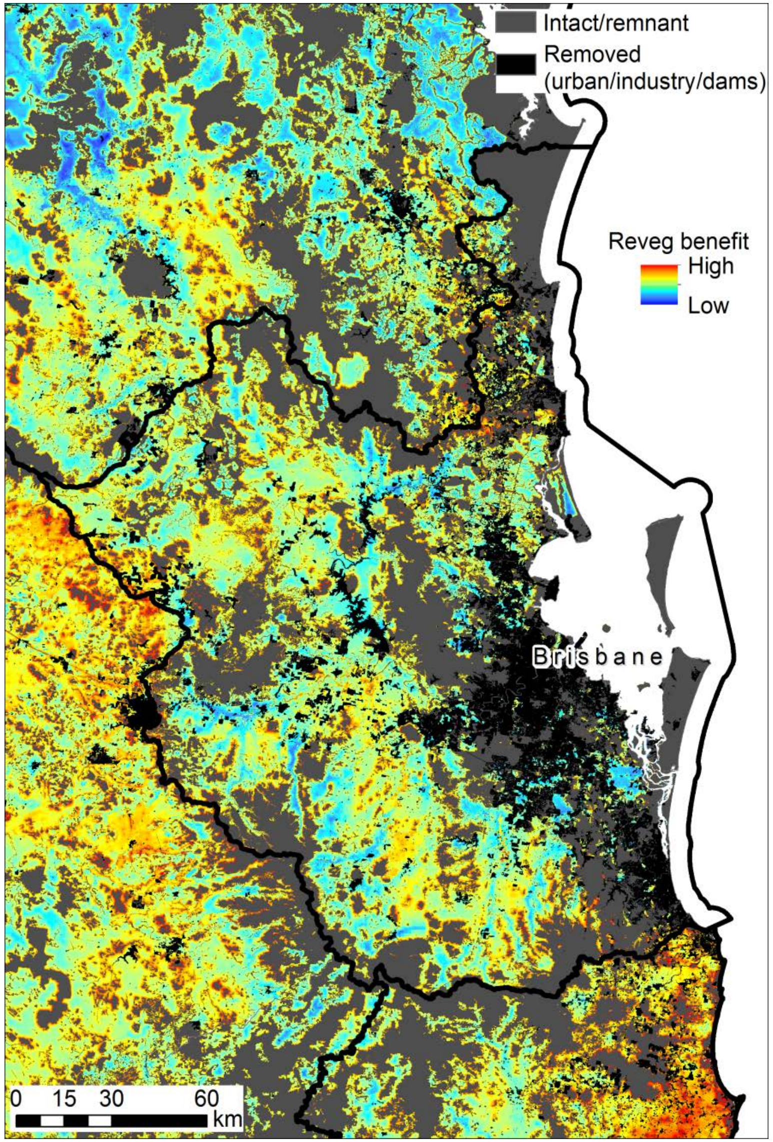
Map 27. BMRG - Neighbourhood habitat benefit



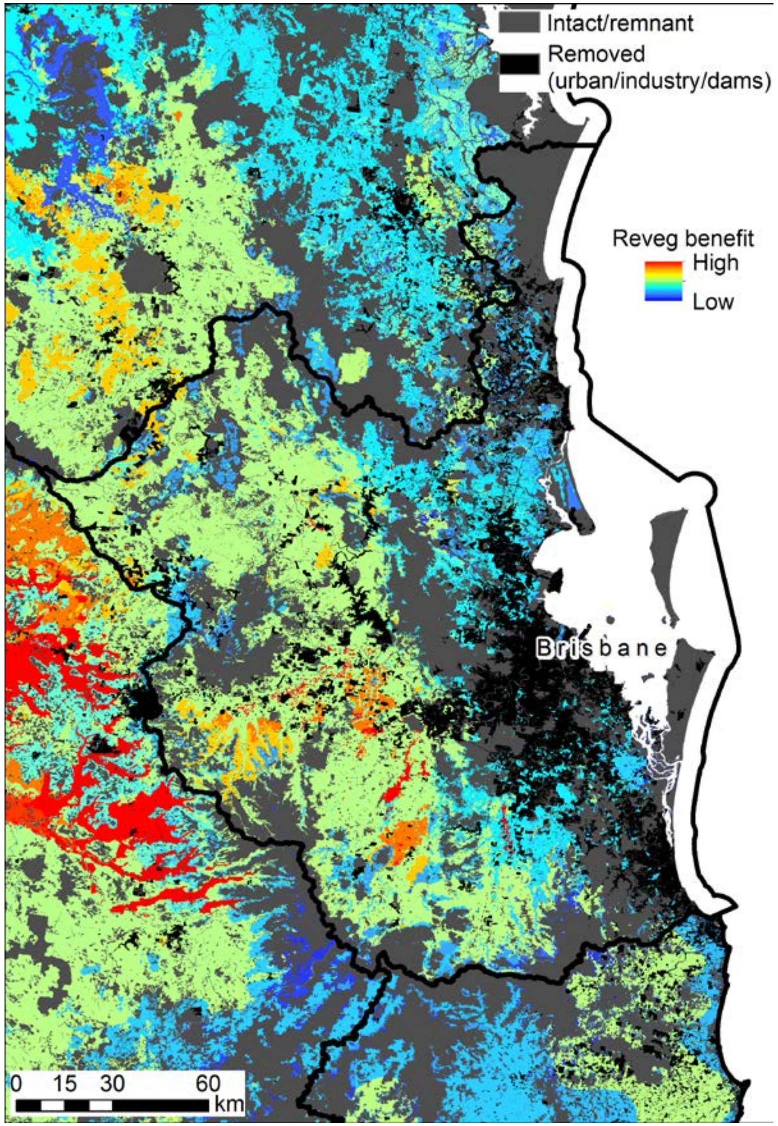
Map 28. BMRG - Threatened species benefit



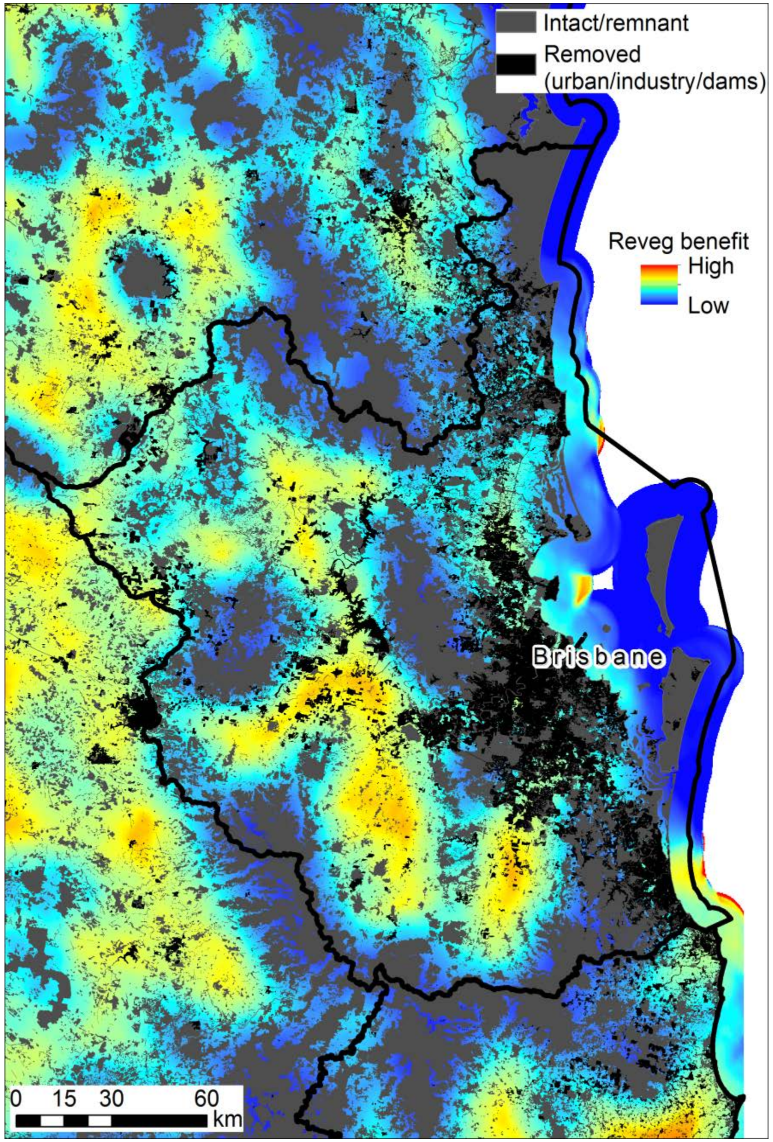
Map 29. SEQ - ACCU price to break-even over 25 year investment period for regrowth projects with 100 year permanence



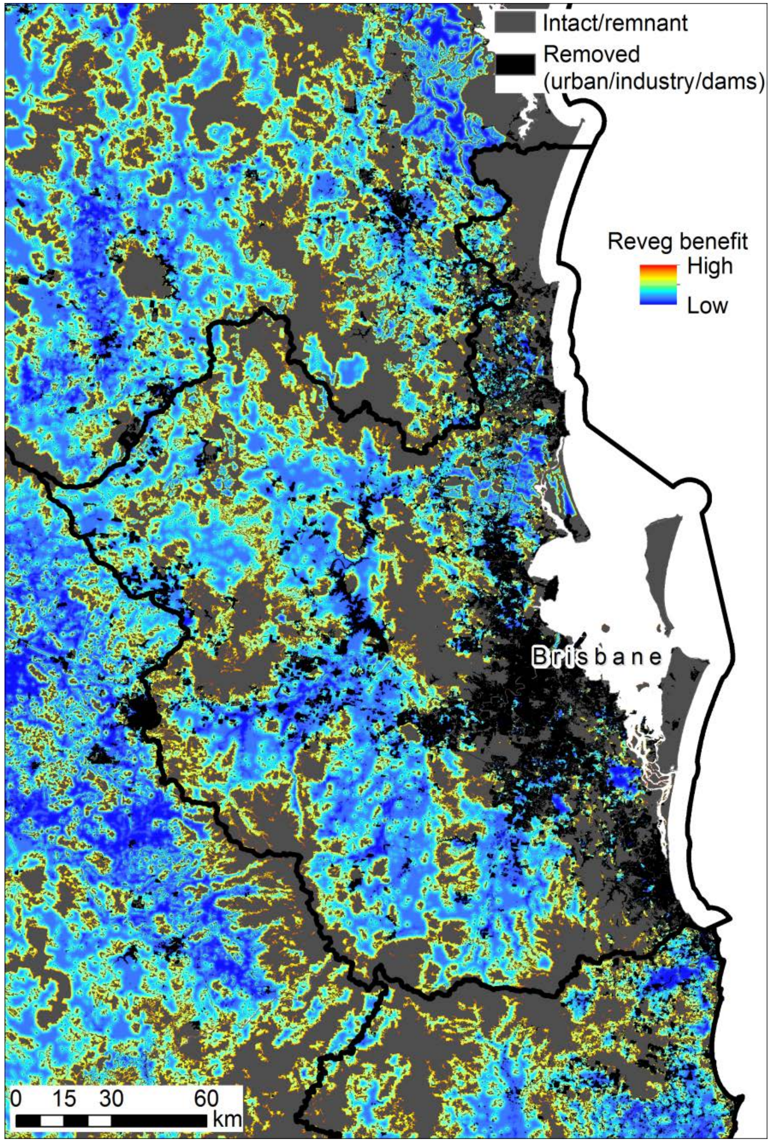
Map 30. SEQ - Revegetation benefit metric



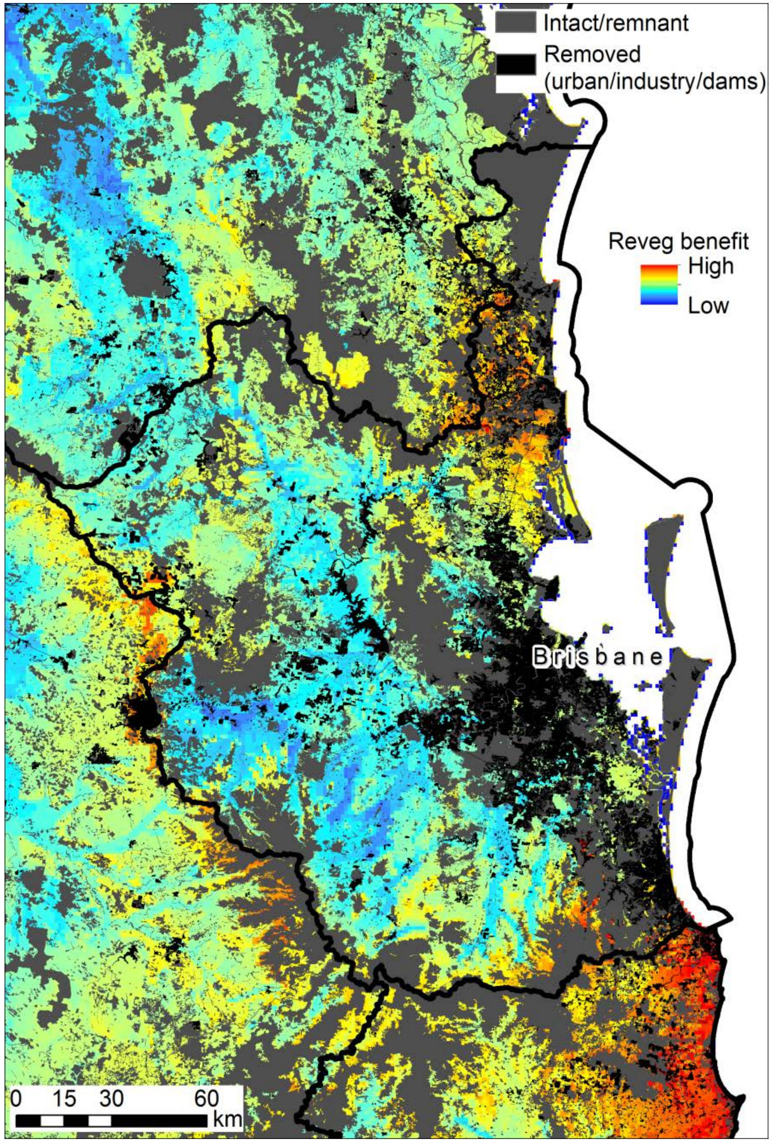
Map 31. SEQ - Land type/pre-clearing veg benefit



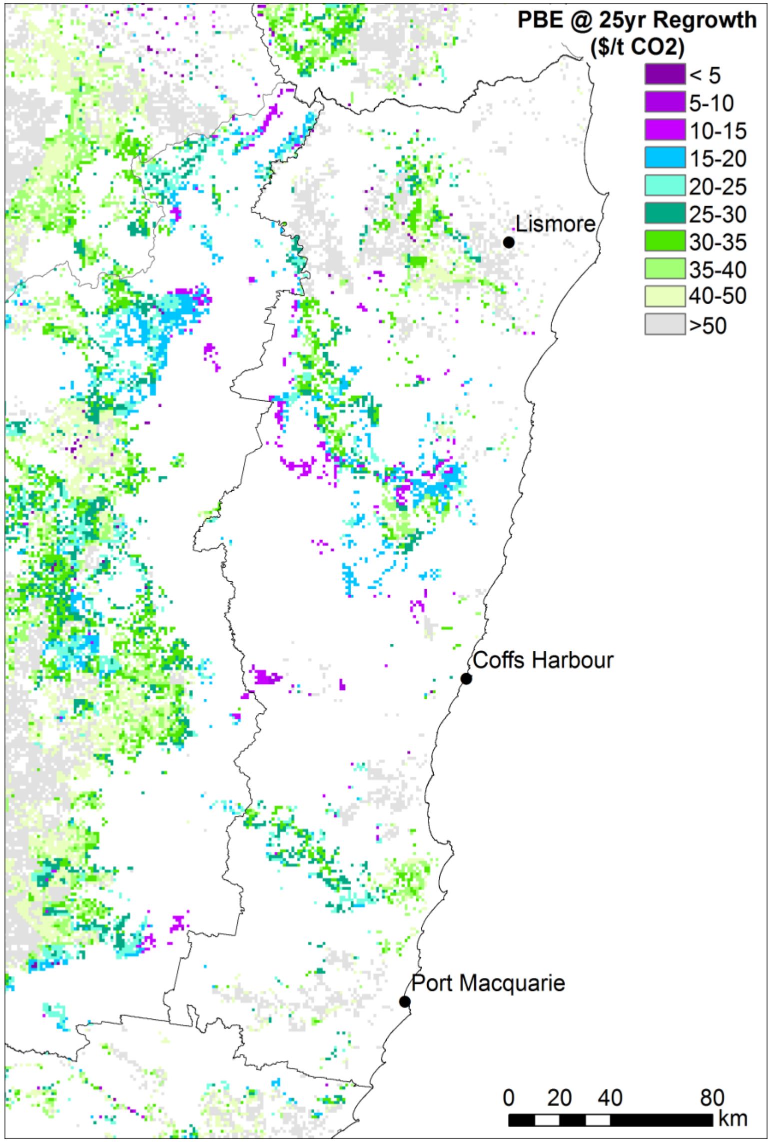
Map 32. SEQ - Remnant in 10km benefit



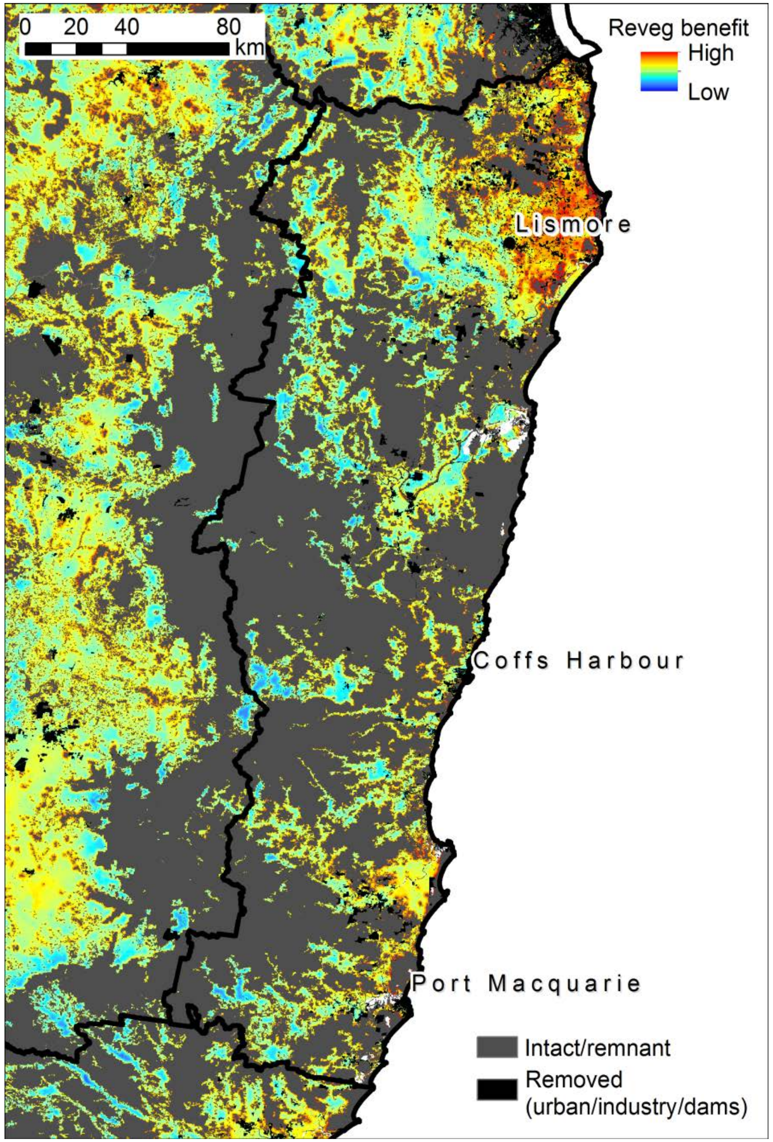
Map 33. SEQ - Neighbourhood habitat benefit



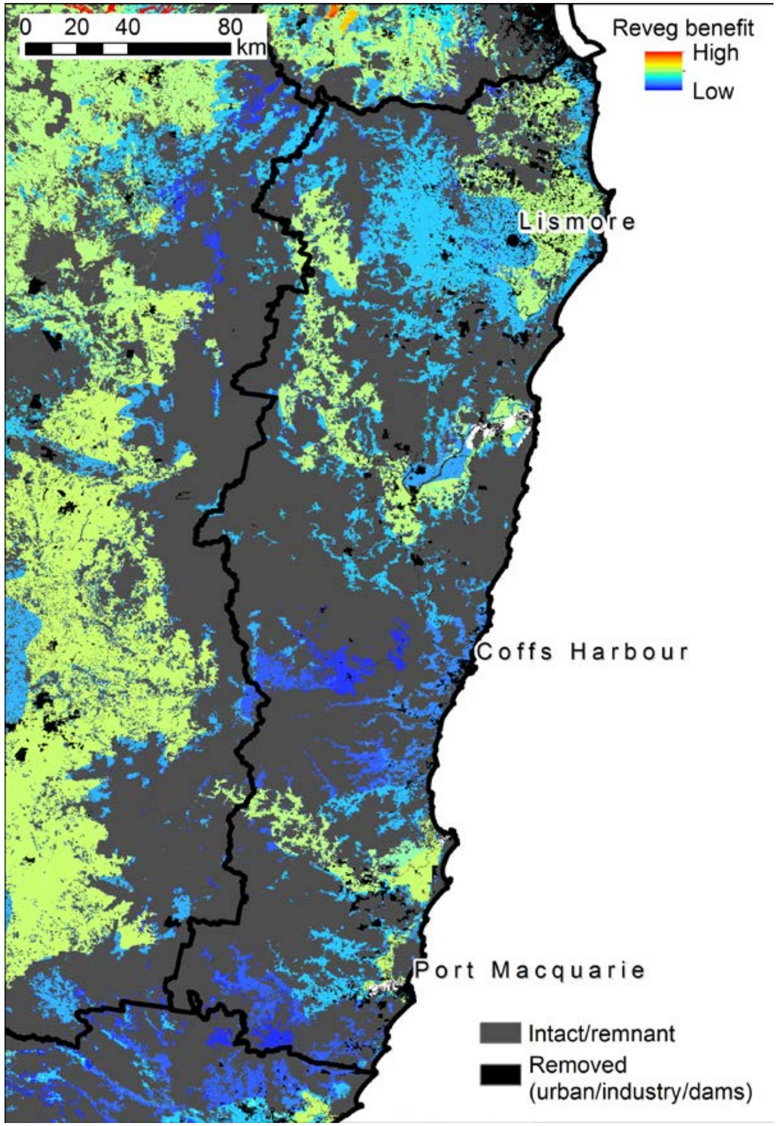
Map 34. SEQ - Threatened species benefit



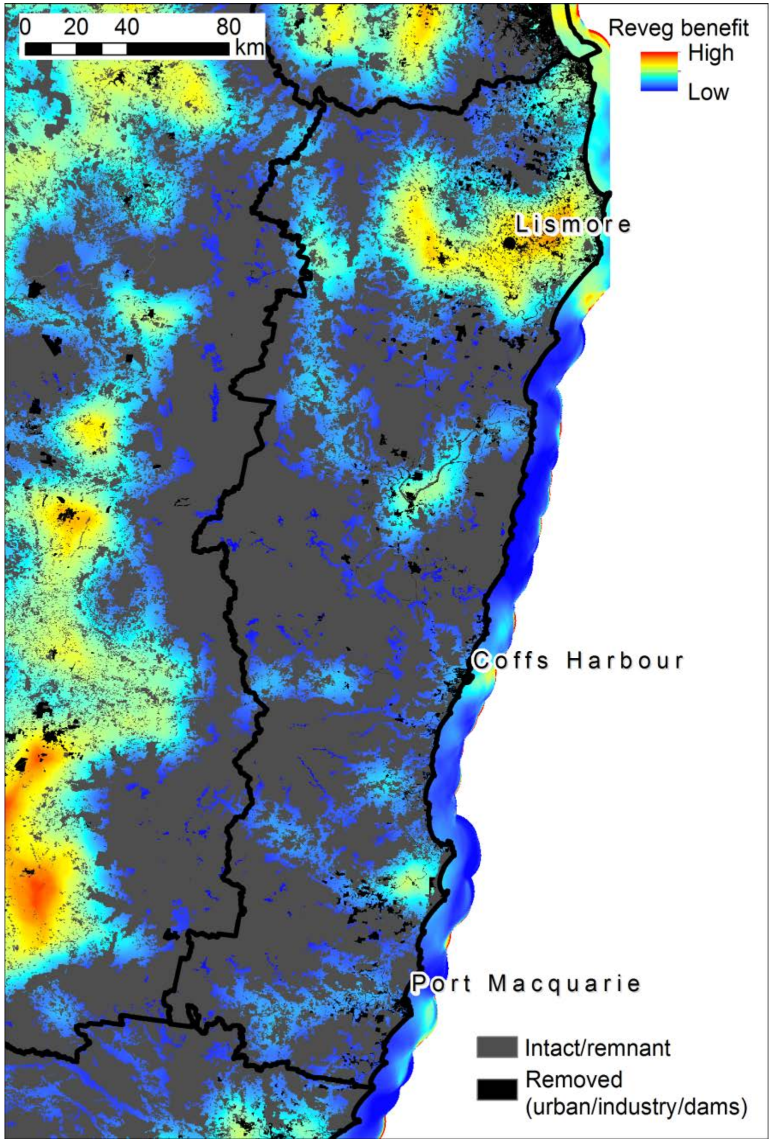
Map 35. NC - ACCU price to break-even over 25 year investment period for regrowth projects with 100 year permanence



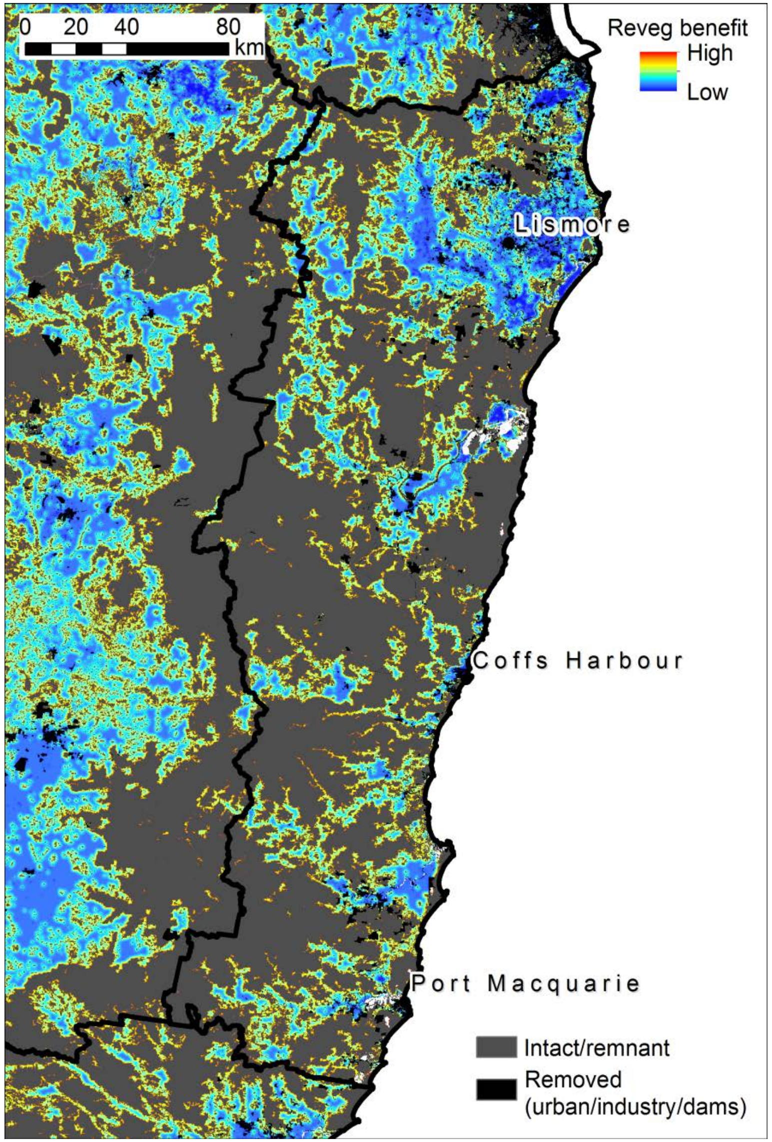
Map 36. NC - Revegetation benefit metric



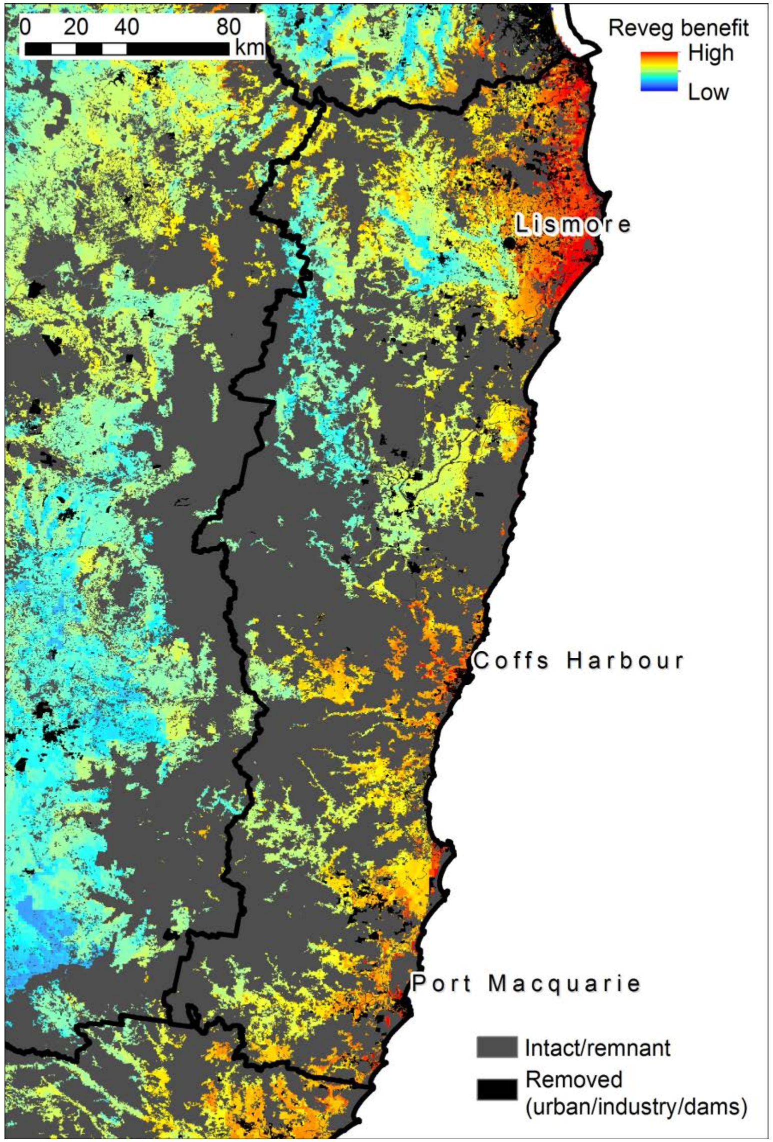
Map 37. NC - Land type/pre-clearing veg benefit



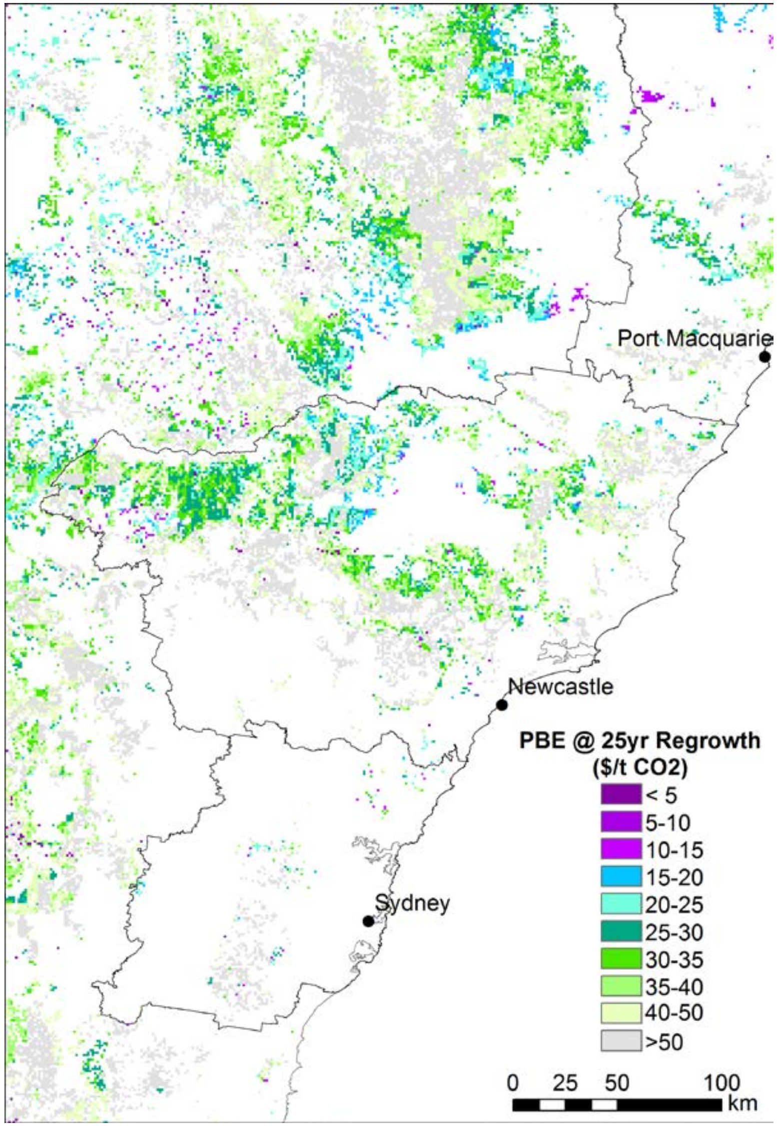
Map 38. NC - Remnant in 10km benefit



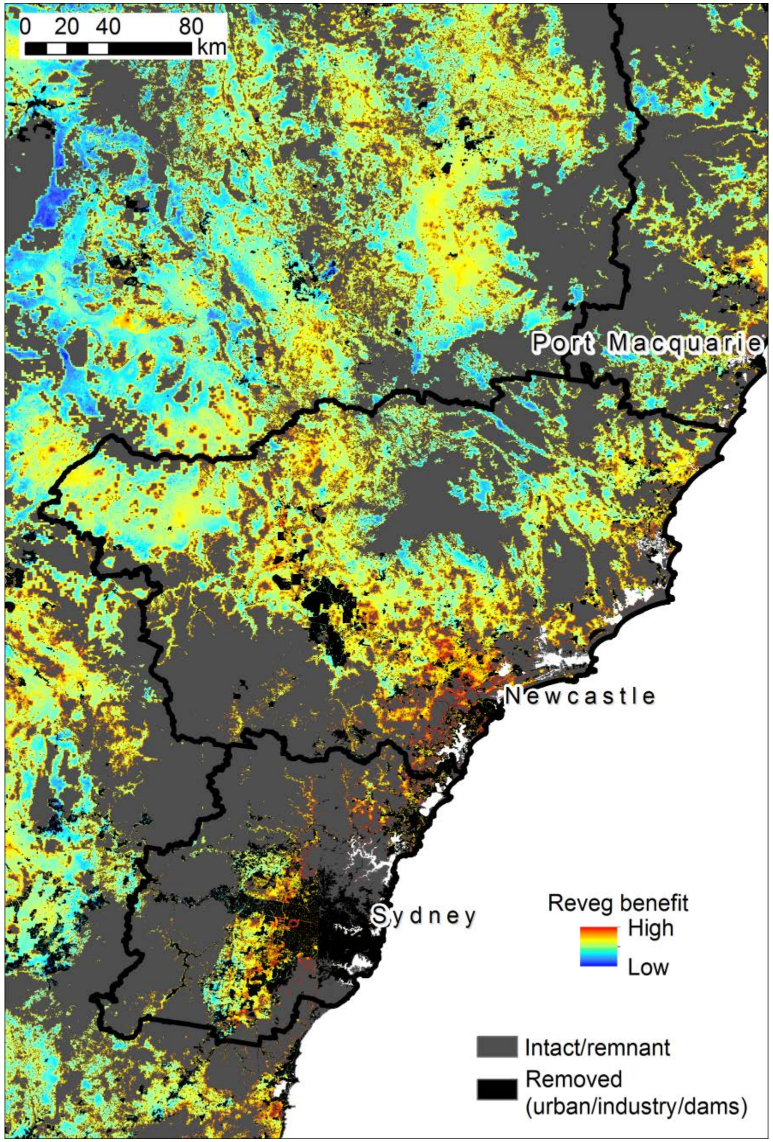
Map 39. NC - Neighbourhood habitat benefit



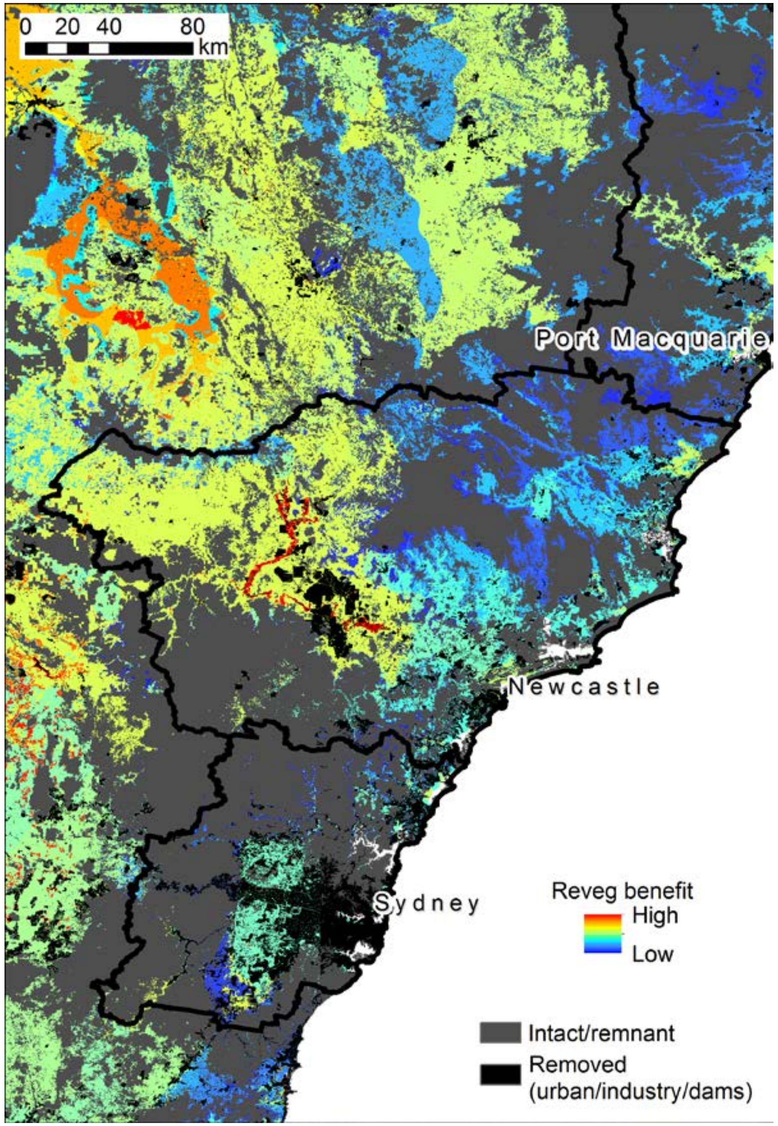
Map 40. NC - Threatened species benefit



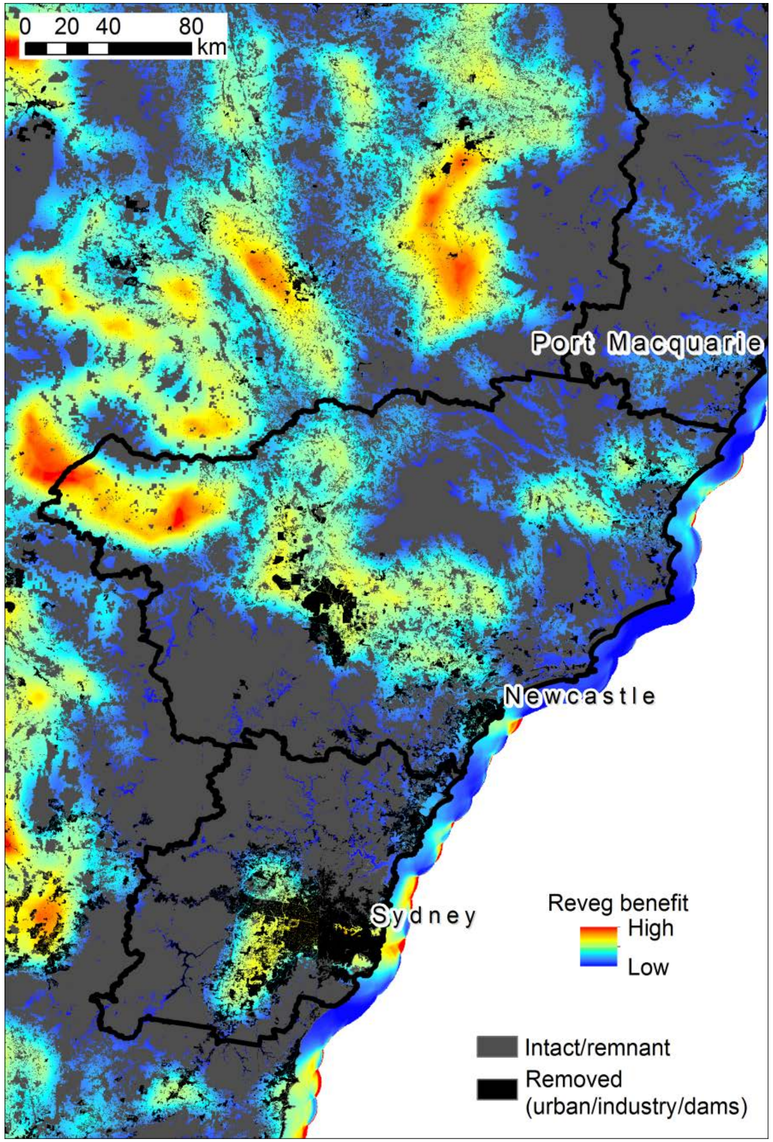
Map 41. Hunter - ACCU price to break-even over 25 year investment period for regrowth projects with 100 year permanence



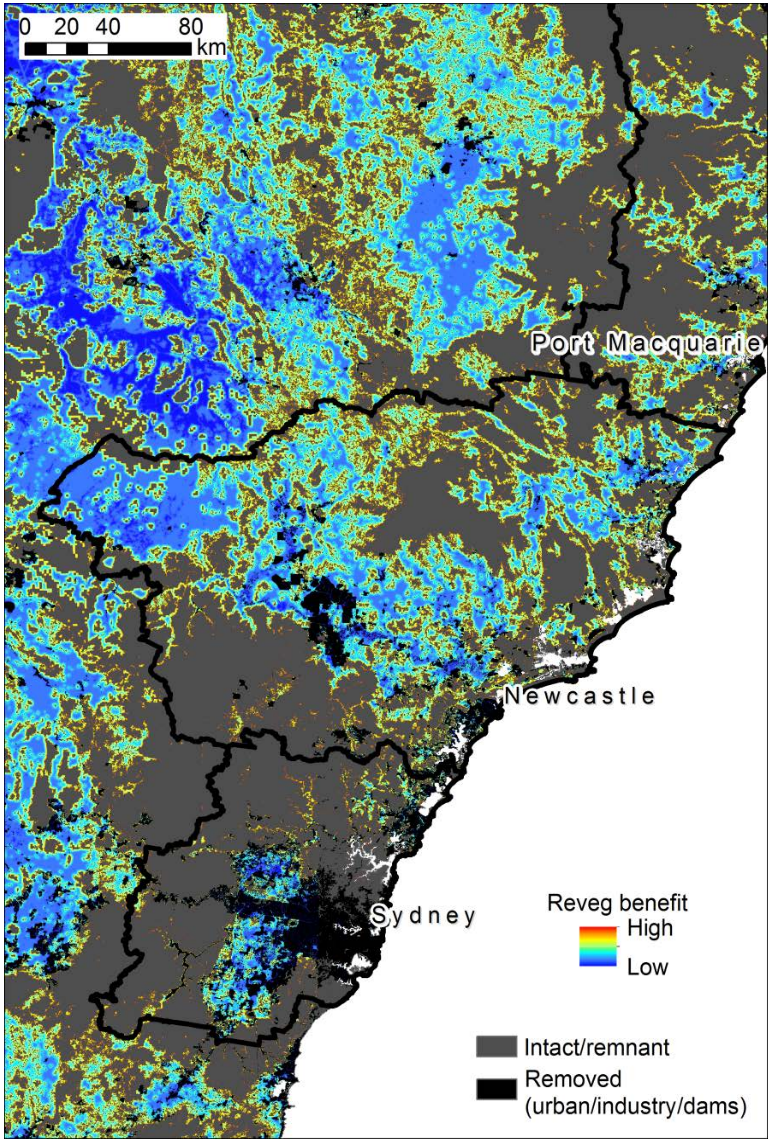
Map 42. Hunter - Revegetation benefit metric



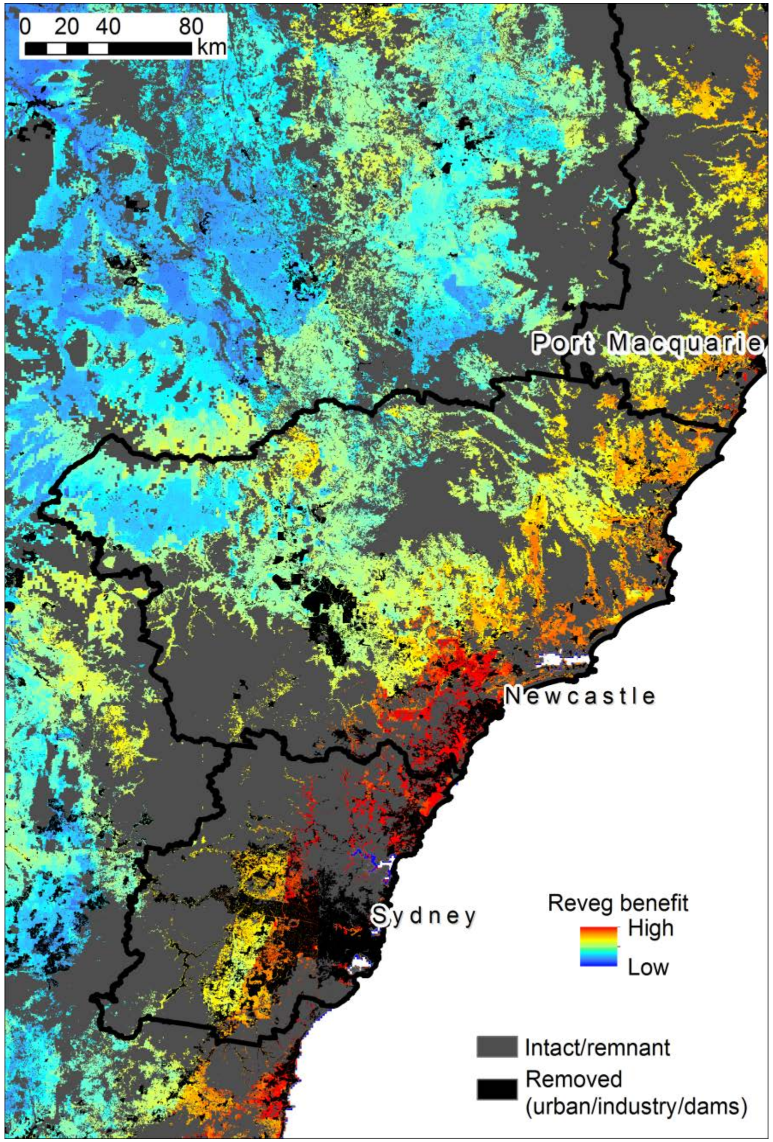
Map 43. Hunter - Land type/pre-clearing veg benefit



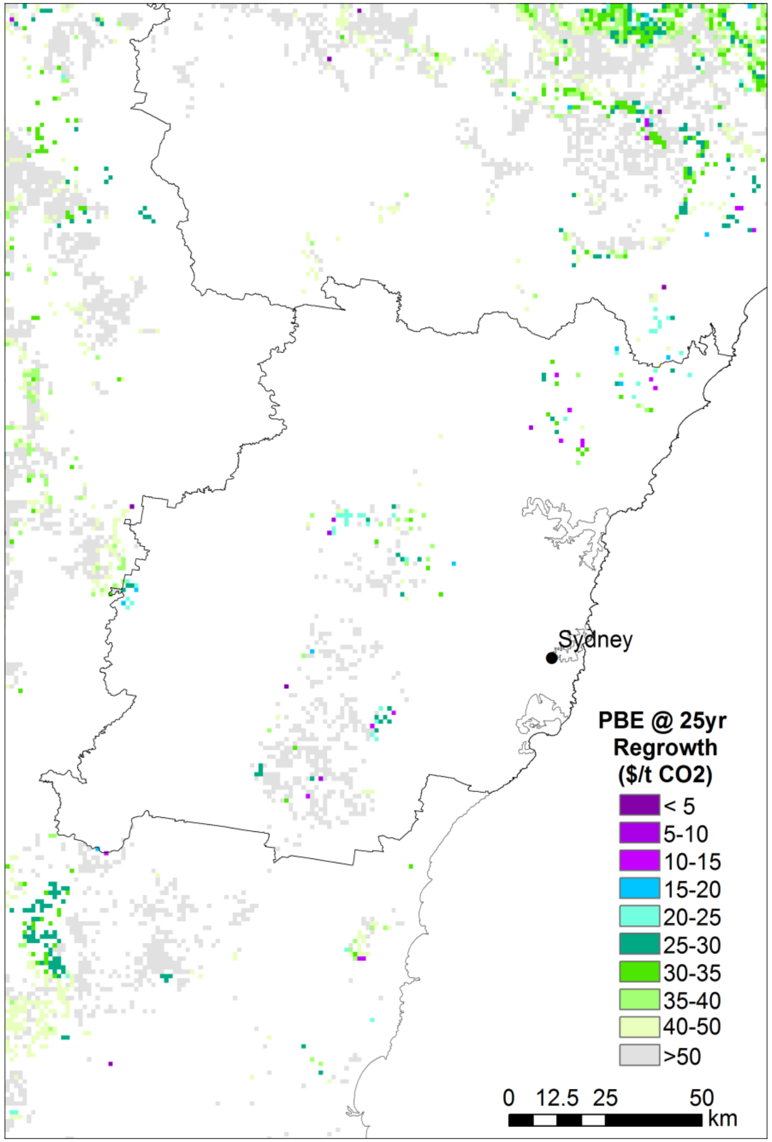
Map 44. Hunter - Remnant in 10km benefit



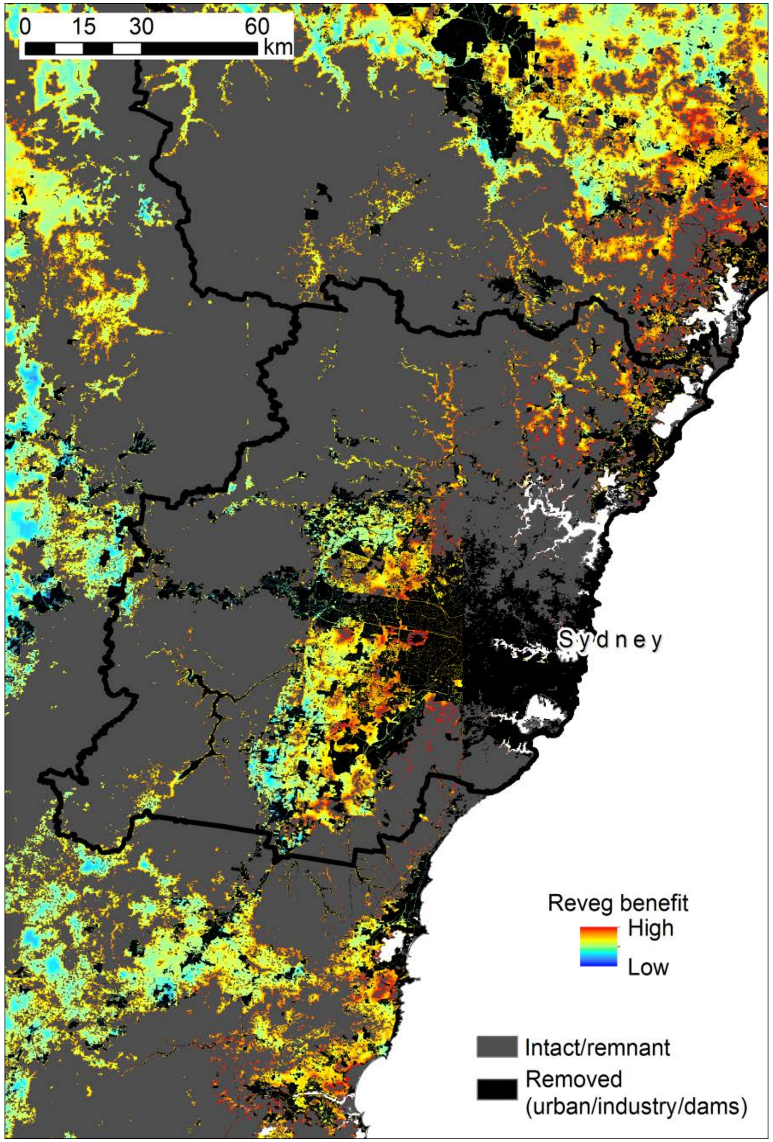
Map 45. Hunter - Neighbourhood habitat benefit



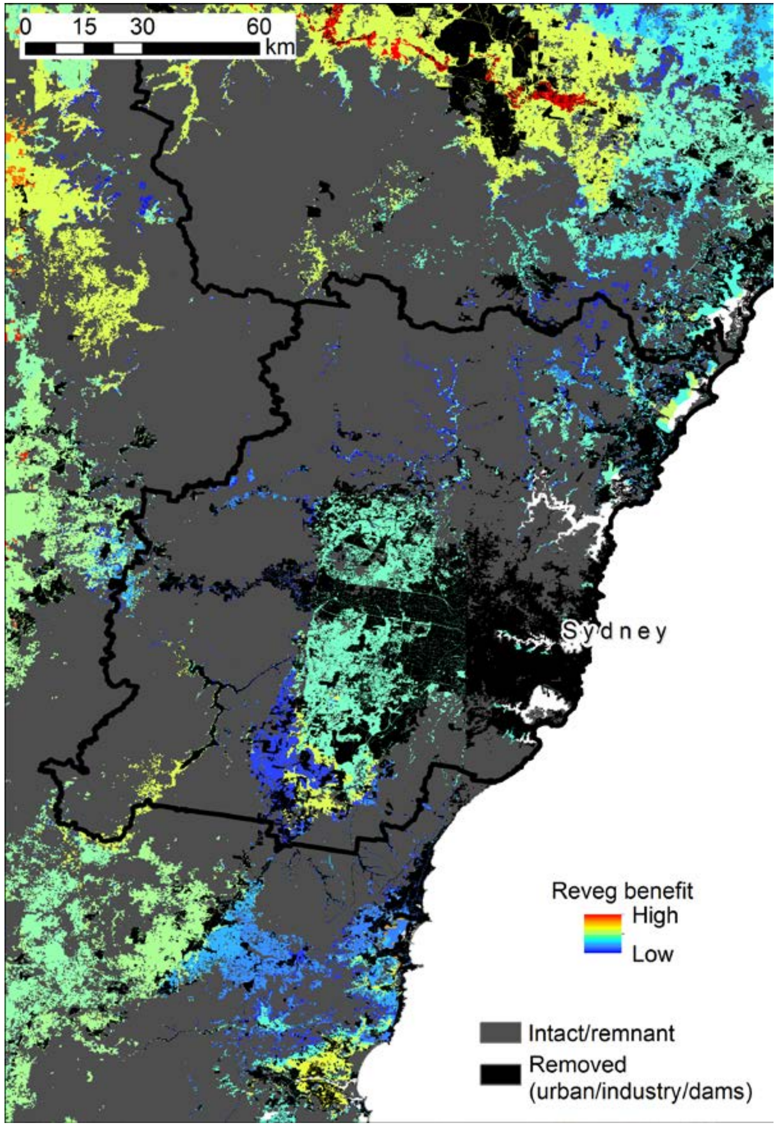
Map 46. Hunter - Threatened species benefit



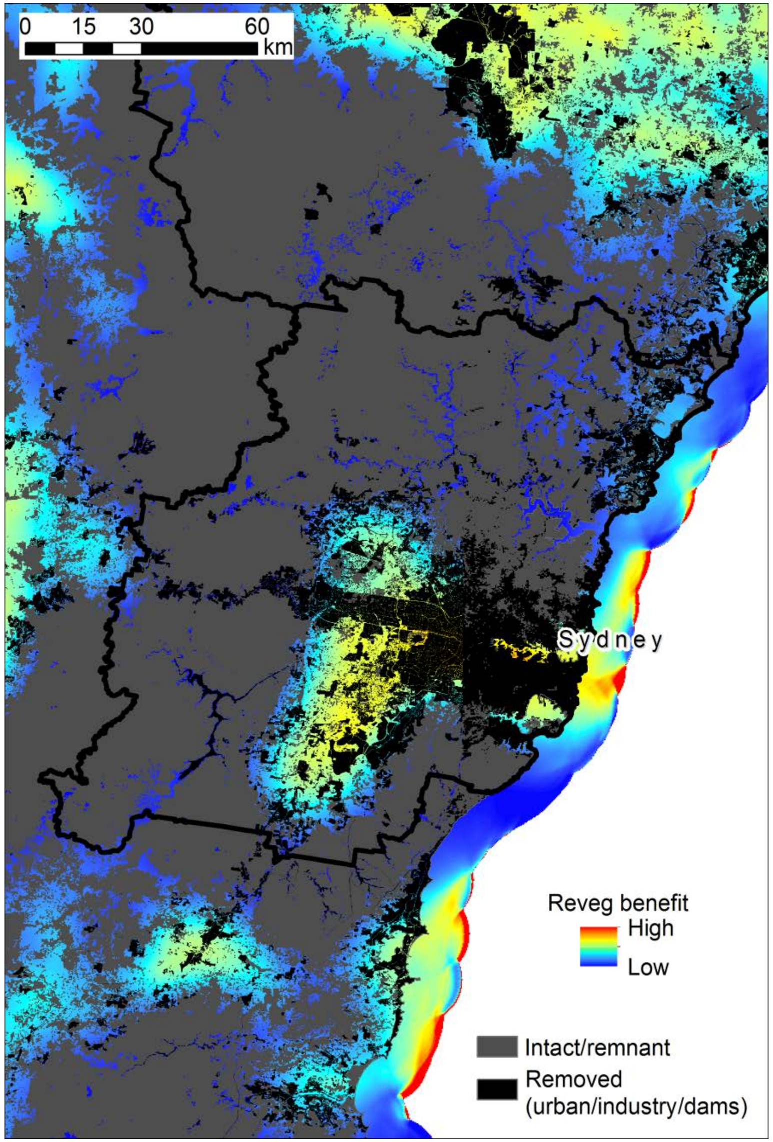
Map 47. GS - ACCU price to break-even over 25 year investment period for regrowth projects with 100 year permanence



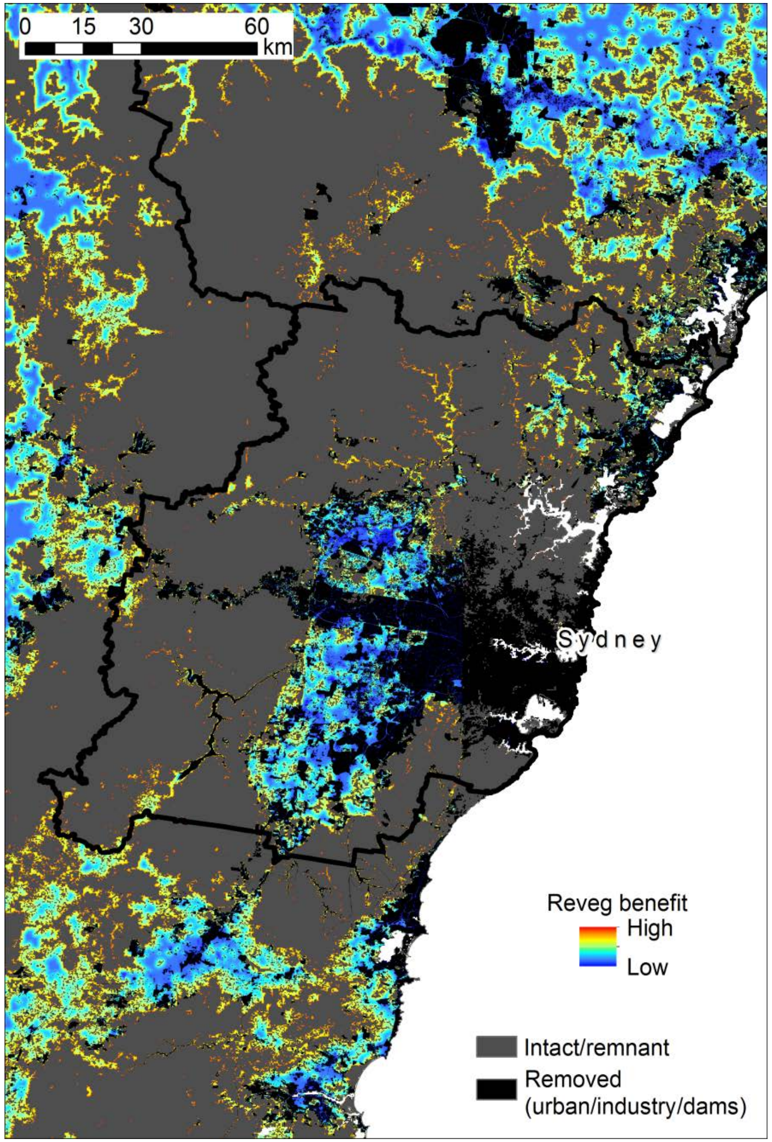
Map 48. GS - Revegetation benefit metric



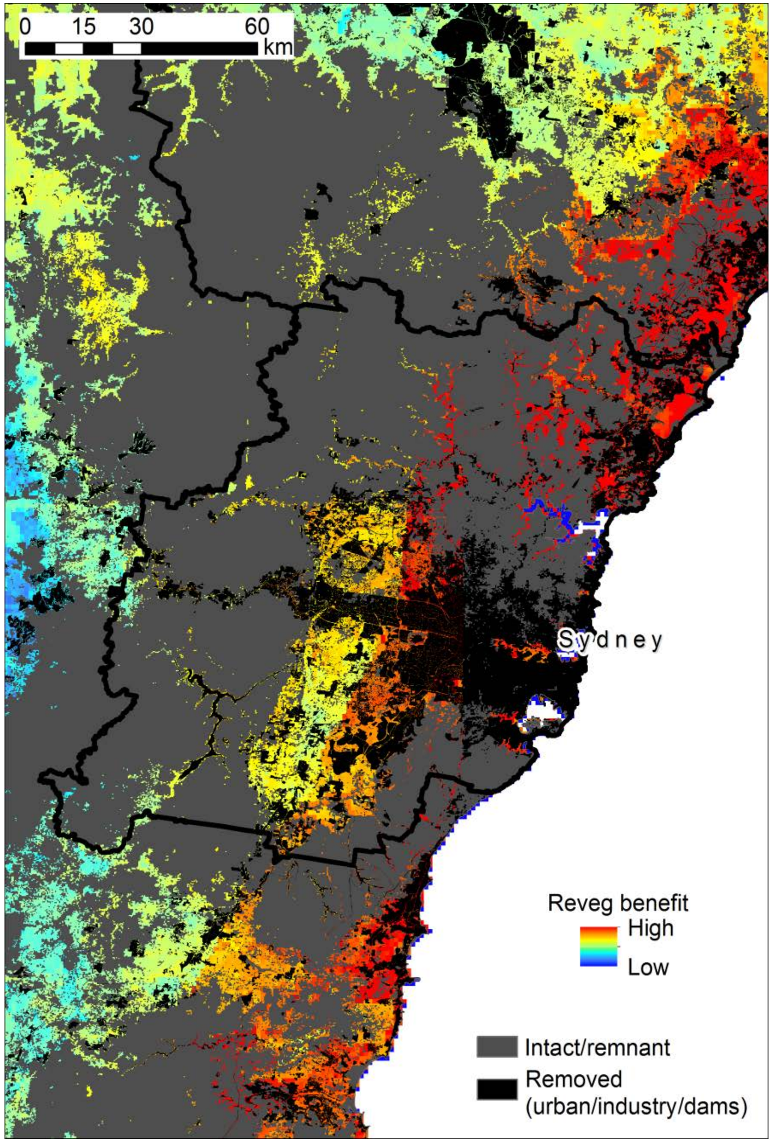
Map 49. GS - Land type/pre-clearing veg benefit



Map 50. GS - Remnant in 10km benefit



Map 51. GS - Neighbourhood habitat benefit



Map 52. GS - Threatened species benefit

Appendix 2. CFI Methodology Determinations available in October 2014 (www.climatechange.gov.au/reducing-carbon/carbon-farming - initiative/methodologies/methodology-determinations)

Agriculture (livestock, soil carbon, fertilisers, feral animals)

1. Destruction of methane generated from dairy manure in covered anaerobic ponds
2. Destruction of methane from piggeries using engineered biodigesters
3. Destruction of methane generated from manure in piggeries
4. Destruction of methane generated from manure in piggeries 1.1
5. Reducing greenhouse gas emissions in beef cattle through feeding nitrate containing supplements
6. Reducing greenhouse gas emissions in milking cows through feeding dietary additives
7. Sequestering carbon in soils in grazing systems

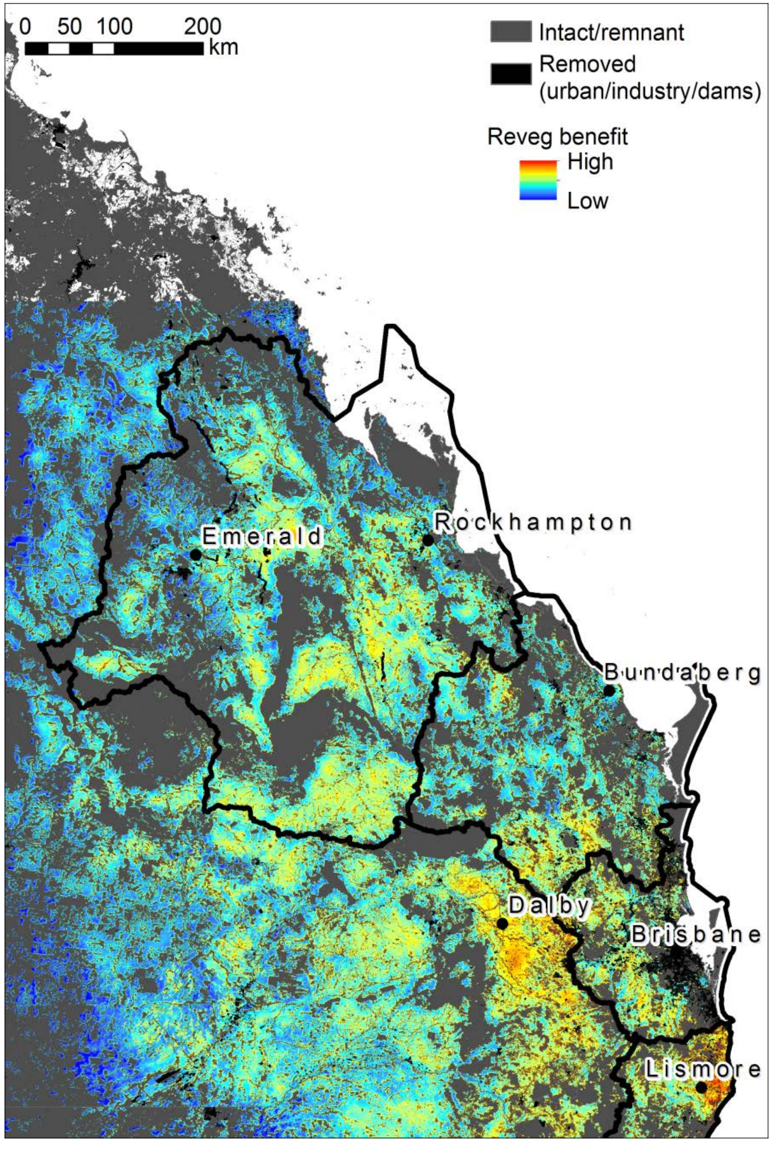
Vegetation (regrowth, reforestation, avoided clearing and avoided harvest)

8. Environmental Plantings
9. Human-Induced regeneration of a permanent even-aged native forest
10. Human-induced regeneration of a permanent even-aged native forest 1.1
11. Measurement based methods for new farm forestry plantations
12. Native forest from managed regrowth
13. Native forest protection (avoided deforestation)
14. Quantifying carbon sequestration by permanent plantings of native mallee eucalypt species using the CFI reforestation modelling tool
15. Reforestation and Afforestation
16. Reforestation and Afforestation 1.1
17. Reforestation and Afforestation 1.2
18. Reforestation by Environmental or Mallee Plantings - FullCAM
19. Savanna burning
20. Savanna burning 1.1

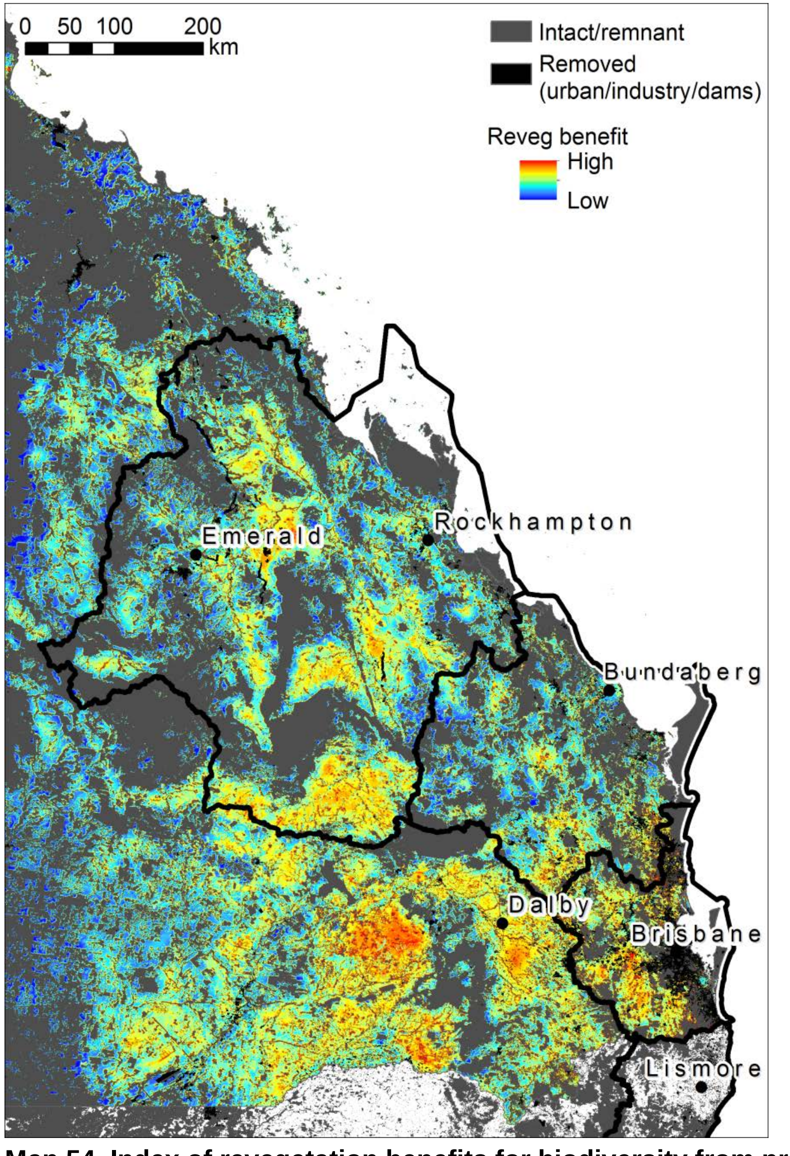
Landfill and alternative waste treatment (AWT)

21. Avoided emissions from diverting waste from landfill for process engineered fuel manufacture
22. Avoided emissions from diverting waste from landfill through a composting AWT technology
23. Capture and combustion of landfill gas
24. Capture and combustion of methane in landfill gas from legacy waste: upgraded projects
25. Diverting waste to an alternative waste treatment facility
26. Enclosed mechanical processing and composting alternative waste treatment

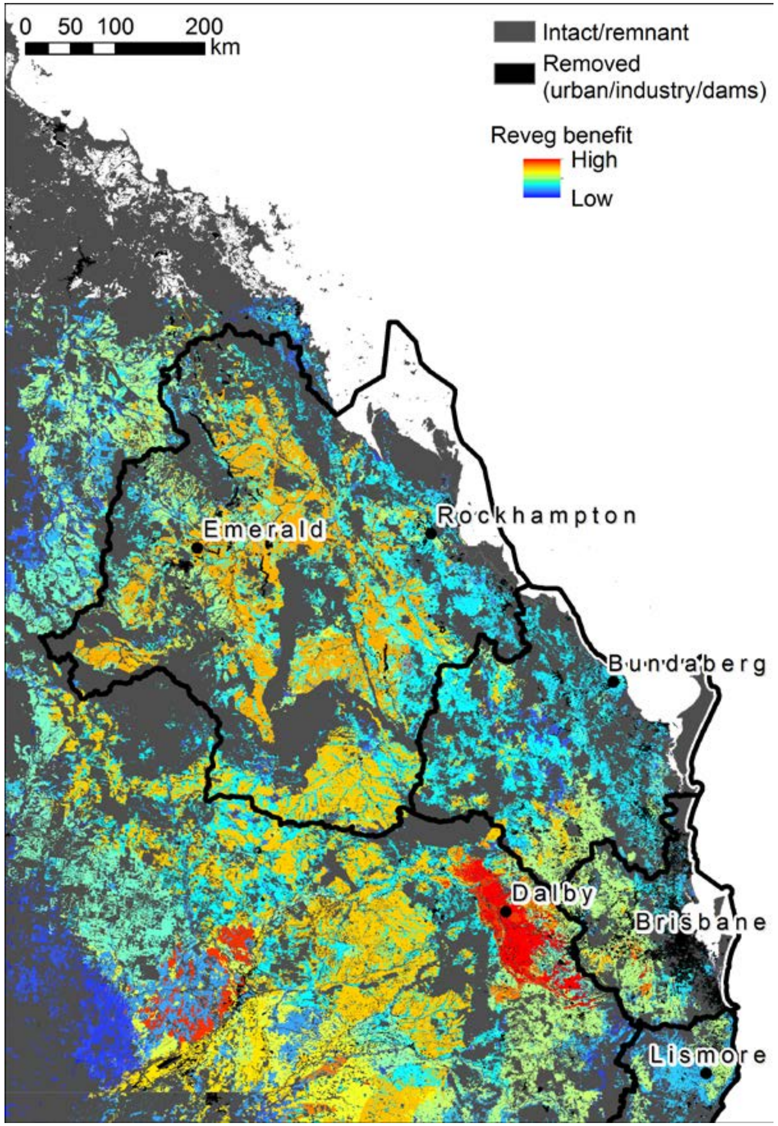
Appendix 3. Comparison between indices from this study and previous work with different inputs



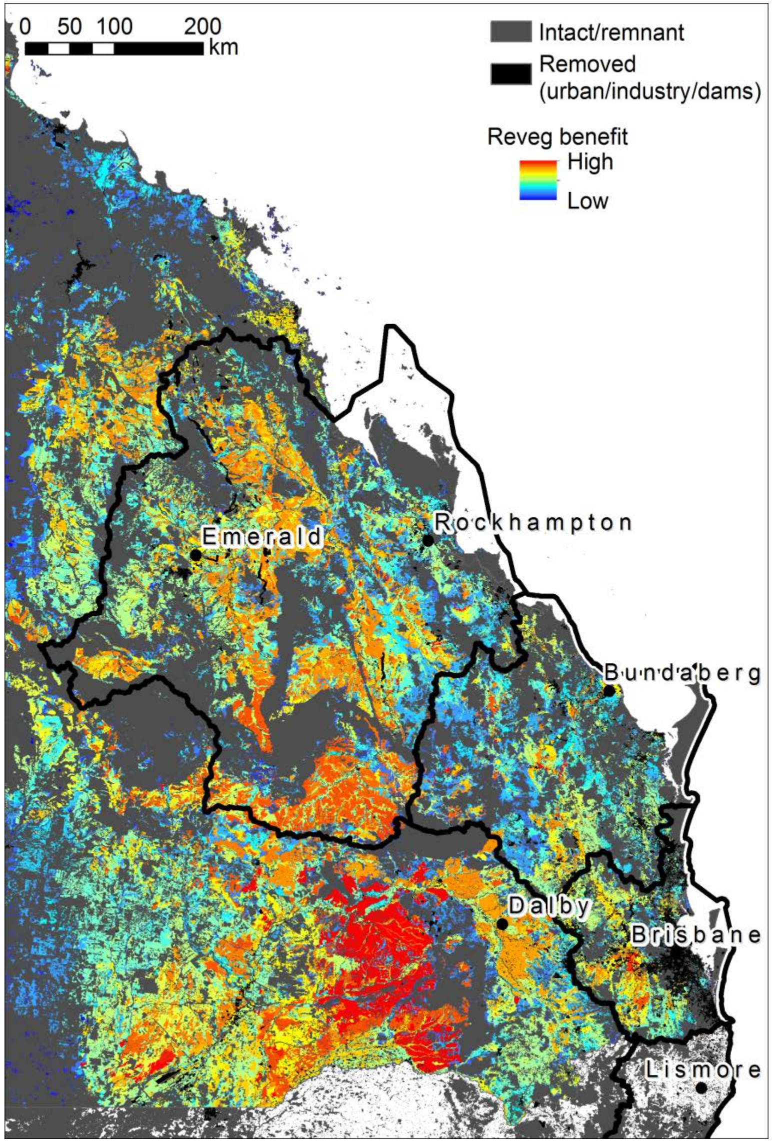
Map 53. Index of revegetation benefits for biodiversity from this study



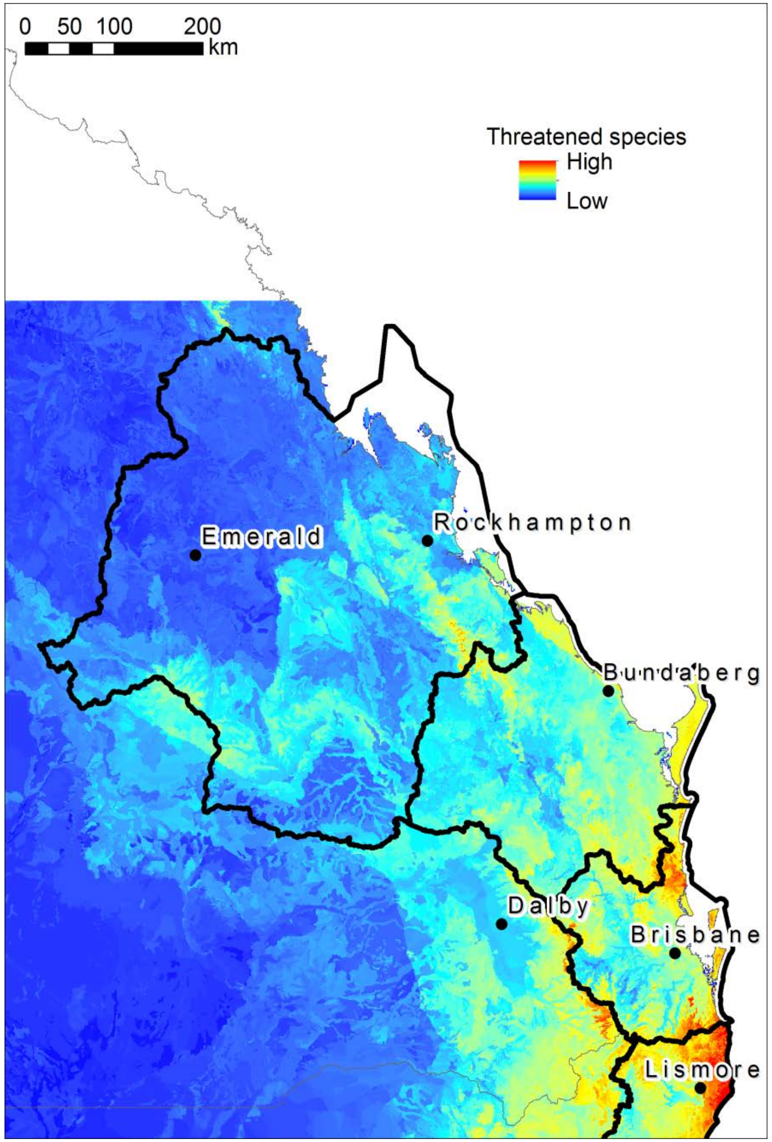
Map 54. Index of revegetation benefits for biodiversity from previous work with different input data



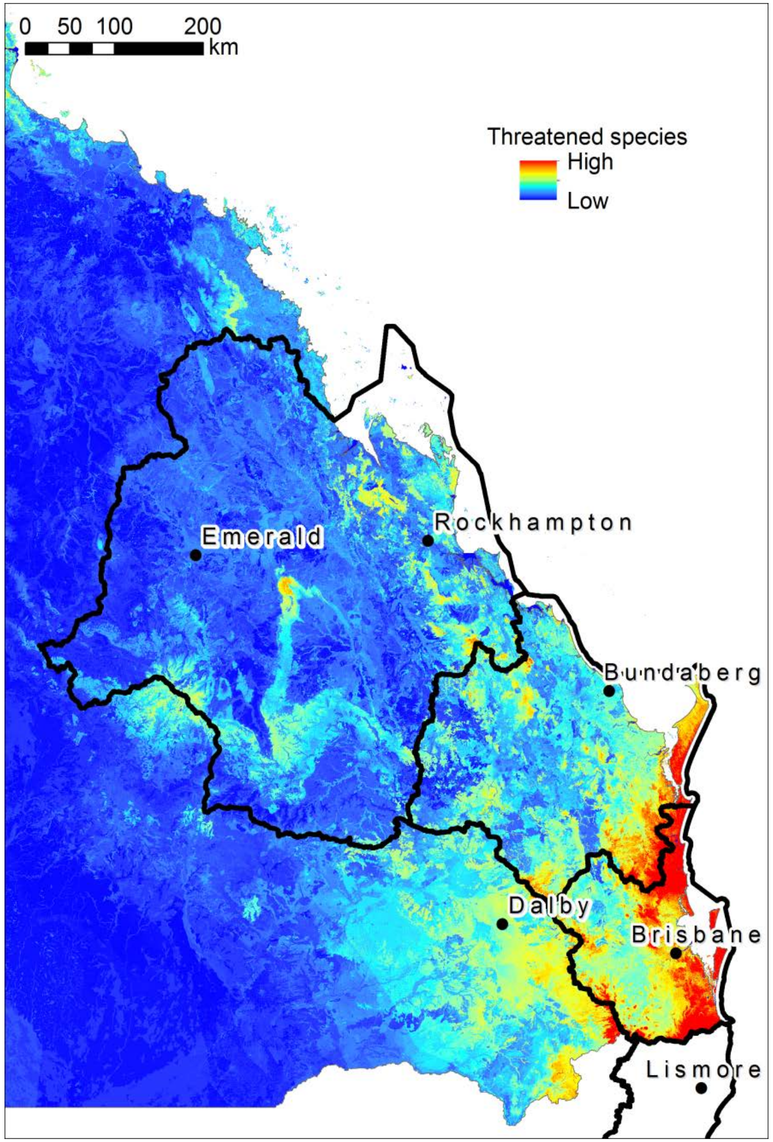
Map 55. Benefits based on land types derived for this study from OEH modelling and NVIS 4.1



Map 56. Benefits based on pre-clearing Regional Ecosystems



Map 57. Spatial patterns in richness of modelled threatened species from Maggini et al. (2013) used as index of benefits for threatened species in this study.



Map 58. Spatial patterns in richness of modelled threatened species from other modelling work completed for Queensland (DSITIA 2013).