Review

Brave new green world – Consequences of a carbon economy for the conservation of Australian biodiversity

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ABSTRACT

Pricing greenhouse gas emissions is a burgeoning and possibly lucrative financial means for climate change mitigation. Emissions pricing is being used to fund emissions-abatement technologies and to modify land management to improve carbon sequestration and retention. Here we discuss the principal land-management options under existing and realistic future emissions-price legislation in Australia, and examine them with respect to their anticipated direct and indirect effects on biodiversity. The main ways in which emissions price-driven changes to land management can affect biodiversity are through policies and practices for (1) environmental plantings for carbon sequestration, (2) native regrowth, (3) fire management, (4) forestry, (5) agricultural practices (including cropping and grazing), and (6) feral animal control. While most land-management options available to reduce net greenhouse gas emissions offer clear advantages to increase the viability of native biodiversity, we describe several caveats regarding potentially negative outcomes, and outline components that need to be considered if biodiversity is also to benefit from the new carbon economy. Carbon plantings will only have real biodiversity value if they comprise appropriate native tree species and provide suitable habitats and resources for valued fauna. Such plantings also risk severely altering local hydrology and reducing water availability. Management

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Forestry
Greenhouse gases
Invasive species
Livestock
Plantings
Regrowth
Stocking

of regrowth post-agricultural abandonment requires setting appropriate baselines and allowing for thinning in certain circumstances, and improvements to forestry rotation lengths would likely increase carbon-retention capacity and biodiversity value. Prescribed burning to reduce the frequency of high-intensity wildfires in northern Australia is being used as a tool to increase carbon retention. Fire management in southern Australia is not readily amenable for maximising carbon storage potential, but will become increasingly important for biodiversity conservation as the climate warms. Carbon price-based modifications to agriculture that would benefit biodiversity include reductions in tillage frequency and livestock densities, reductions in fertiliser use, and retention and regeneration of native shrubs; however, anticipated shifts to exotic perennial grass species such as buffel grass and kikuyu could have net negative implications for native biodiversity. Finally, it is unlikely that major reductions in greenhouse gas emissions arising from feral animal control are possible, even though reduced densities of feral herbivores will benefit Australian biodiversity greatly.

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1. Introduction

As world greenhouse gas emissions (see glossary: Table 1) continue to track worst-case projections (Intergovernmental Panel on Climate Change, 2007), humanity is beginning to implement workable financial mechanisms to abate them. The basic rationale for such mechanisms is to provide industry with incentives via a financial penalty ('carbon pricing') or offset scheme ('carbon credits'), thereby promoting investment practices that reduce emissions, produce 'clean' energy, or increase energy efficiency. A key inclusion within such programs is the recognition for the potential to sequester carbon in soils and vegetation.

Deforestation, particularly the destruction of biodiverse tropical rainforests, is thought to have contributed between 10 and 20% of the anthropogenic CO_2 emissions since the Industrial Revolution (van der Werf et al., 2009). Thus, there should be a good fit between conservation of biodiversity outcomes and carbon storage given that forests are the most carbon-dense ecosystems on Earth (Luyssaert et al., 2008). Indeed, this is the underlying logic of schemes such as Reduced Emissions from Forest Deforestation and Degradation (REDD) in tropical forests (Phelps et al., 2010), which hold well over 60% of the world's species (Bradshaw et al., 2009). However, schemes such as REDD (and its variants, including REDD+; van Oosterzee et al., 2012) are extremely complex to manage and in some geo-political settings, are vulnerable to perverse outcomes, such as clearing of high-diversity native forests to establish forestry plantations (Venter et al., 2010).

While the implications for biodiversity of greenhouse gas-abatement schemes have been highlighted generally (Phelps et al., 2010), there have not been any reviews dedicated to the implications for biodiversity in an Australian context. Our overall objective here is to determine to what extent proposed changes to land use and management in Australia following the implementation of a carbon price might affect biodiversity. While our focus is mainly on those landscape transformations that affect forest[1] cover directly, these are not of course the only types of landscape processes that will be affected. The direct effect on forest cover is obvious, but there are many complex interactions and synergies with related components of the landscape that will affect biodiversity persistence, both positively and negatively. We focus on six main areas of policy and management intervention: (1) environmental plantings, (2) policies and practices to deal with native regrowth, (3) forestry management, (4) fire management, (5) agricultural practices (including cropping and grazing), and (6) feral animal control. For each of these aspects, we give a brief background, outline the main changes foreseen following the implementation of Australian emissions-price legislation, and discuss the anticipated positive and negative effects on biodiversity. We identify possible unanticipated negative impacts (Lindenmayer et al., 2012), and how these can be avoided. While our examples are predominantly Australian, the key concepts and recommendations transcend regional focus and are relevant to emissions-mitigation schemes globally.

2. Policy setting

Many countries and regions have already adopted mandatory or voluntary emissions-reduction mechanisms (Fahey et al., 2009), including New Zealand's Emissions Trading Scheme (http://www.climatechange.govt.nz/emissions-trading-scheme), the European Commission's cap-and-trade Emissions Trading Scheme (ec.europa.eu/clima/policies/ets), California's cap-and-trade (www.arb.ca.gov/cc/capandtrade/capandtrade.htm) scheme, the Chicago Climate Exchange (www.theice.com/ccx.jhtml), the U.S. Regional Greenhouse Gas Initiative (www.rggi.org), the Greenhouse Gas Reduction Program and Offset Credit System in Alberta, Canada (carbonoffsetsolutions.climatechangecentral.com), and of course, the Clean Development Mechanism under the Kyoto Protocol (Box 1). Under the Kyoto Protocol (Box 1), the Australian Government has committed to reducing Australia's greenhouse gas emissions by 5–25% from 2000 levels by 2020, depending on the scale of international action, and carbon pricing has recently been imposed under the 'Clean Energy Future' legislation (Commonwealth of Australia, 2011) (www.cleanenergyfuture.gov.au). The legislation incorporates four elements: a carbon price from which agriculture is exempt, promotion of renewable energy, encouragement of energy efficiency and action on the land facilitated through a Land Sector Package. The Land Sector Package includes six measures: Carbon Farming Futures, the Indigenous Carbon Farming Fund, Carbon Farming Skills, the Carbon Farming Initiative (Box 2) Non-Kyoto Carbon Fund, the Biodiversity Fund, and Regional Natural Resource Management (NRM) Planning for Climate Change. As a whole-of-government initiative, the Land Sector package spans multiple government agencies, which creates additional challenges when trying to integrate issues such as biodiversity conservation. A Land Sector Carbon and Biodiversity Board has also been established to advise on implementation of the various measures.

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[1] We define 'forest' very broadly as "an area that is dominated by trees having usually a single stem and a mature or potentially mature stand height >2 m and with existing or potential crown cover of overstorey strata >20%" (see Bradshaw, 2012). Major forest vegetation classes in Australia are shown in Fig. 1.

Box 1 The Kyoto Protocol.

The Kyoto Protocol (http://unfccc.int/kyoto_protocol/items/2830.php) is an international agreement established in 1997 under the United Nations Framework Convention on Climate Change (UNFCCC) in Kyoto, Japan. The Protocol sets binding targets for signatory industrialised countries and the European community to reduce greenhouse gas emissions by an average of 5% relative to 1990 output from 2008-2012 (first commitment period).
Australia had refused to be a signatory of the initial agreement, but ratified the Protocol in 2007 upon a change of government, coming into effect in Australia in 2008. During negotiations in Durban, South Africa, in 2011, Parties to the Kyoto Protocol established a second commitment period from 2013. Australia agreed to join the second commitment period covering 2013 to 2020. Forest management will form part of Australia's national accounts in this second commitment period.
Signatory countries must meet their emissions-reduction targets mainly through national measures, but the Protocol offers additional market-based mechanisms to assist in meeting those targets:

1. **Emissions trading** (the 'carbon market'): Commitment targets are expressed as levels of allowed emissions (assigned amounts) that are divided into 'assigned amount units' (AAUs). Emissions trading allows countries with spare AAUs (emissions permitted but not used) to sell this excess capacity to countries that are over their targets.

2. **Clean development mechanism** (CDM): This allows countries to implement an emissions-reduction project in developing countries. Projects can earn saleable certified emission reduction credits (=1 tonne CO_2-e) that can be counted towards Kyoto targets. The CDM is intended to stimulate sustainable development and emission reductions while simultaneously allowing industrialised countries some flexibility in meeting emissions targets.

3. **Joint implementation** (JI): JI allows countries to earn emission reduction units (ERUs) from an emission-reduction or emission-removal project in another participating country (=1 tonne CO_2-e) that can be counted towards Kyoto targets.

Box 2 Australia's Carbon Farming Initiative (CFI).

Passed by Parliament on 23 August 2011, the Carbon Farming Initiative (www.climatechange.gov.au/cfi) provides a financial incentive to land managers and farmers to reduce greenhouse gas emissions from business-as-usual activity and/or sequester (store) carbon on land. The CFI is essentially an 'offset' scheme and is part of Australia's emissions trading market. It can be summarised as follows:

- Despite its name, the Carbon Farming Initiative applies both to sequestration of carbon and to avoidance of emissions of carbon dioxide, methane and nitrous oxide.

- Land managers can earn Australian Carbon Credit Units (ACCU) from modified activities that reduce/store carbon, and these can be sold to organisations that are required or choose to offset their emissions. Participation in the CFI is entirely voluntary, but several hundred Australian companies are required to pay to offset their direct emissions.
- 1 ACCU = 1 tonne CO_2-e = AU\$23 (as of 1 July 2012). This price will rise at 2.5% $year^{-1}$ until 2014/2015 and will be set by the market thereafter.
- Nationally accounted ACCU that are compliant with the Kyoto Protocol (see Box 1) include reforestation, avoided deforestation, reductions in emissions from livestock and manure/fertiliser/waste deposited in landfills before 1 July 2012. Kyoto-compliant ACCU can be traded in international markets.
- Under the Kyoto Protocol (Box 1), certain activities are not included in the national accounting, such as soil carbon, feral animal management, improved forest management and non-forest revegetation – through the CFI, these activities can earn 'non-Kyoto' ACCU.
- The CFI is administered by the Clean Energy Regulator, which is responsible for approving CFI projects, issuing credits and managing the holding, transfer, retirement, relinquishment and cancellation of units through the Registry (an electronic system used to track the issue, trade and retirement of emissions units under the carbon price mechanism).
- Emissions-avoidance projects covered by the CFI include: agricultural emissions, introduced animal emissions and legacy landfill emissions.
- Sequestration-offsets projects covered by the CFI include: sequestering carbon in plants as they grow, increasing soil organic matter, avoided vegetation loss, afforestation, reforestation, revegetation, rangeland restoration and native forest protection.
- To be credited with ACCU, a project must use vetted 'methodologies' to measure and/or predict carbon balance. This includes:
    a. ensuring abatement is measureable and verifiable,
    b. supporting measurement methods by peer-reviewed science that are consistent with Australia's international accounts,
    c. accounting for leakage and variability,
    d. using conservative assumptions,
    e. ensuring additionality,
    f. establishing permanence, and
    g. adhering to specific monitoring and reporting requirements (including audits).
- Carbon sequestration or emissions abatement delivered by CFI projects must be additional to what would have been achieved in the absence of the project. ACCU generated by CFI projects must genuinely offset the emissions produced by the buyer.
- Many forestry, agricultural and landfill/waste-treatment methodologies have already been approved under the CFI. As of February 2013, these include: (1) savanna burning, (2) environmental plantings of native species, (3) human-induced regeneration of a permanent, even-aged native forest, (4) reforestation and afforestation, (5) destruction of methane from manure in piggeries, (6) destruction of methane from piggeries using engineered bio-digesters, (7) destruction of methane generated from dairy manure in

covered anaerobic ponds, (8) avoided emissions from diverting waste from landfill for process-engineered fuel manufacture, (9) capture and combustion of landfill gas, (10) diverting waste to an alternative waste treatment facility and (11) destruction of methane from legacy waste.
- For the methodologies relevant to this review, emissions can be reduced by (1) shifting burning from late to early dry season and reducing the area burnt each year, (2) establishing an environmental planting by seeding or planting native species on cleared land, or permanently maintaining the planting, (3) reforesting cleared land and afforesting land where no forests previously existed, and (4) establishing a forest through cessation of the activities causing suppression or destruction of vegetation regrowth.
- For savanna burning, project owners must (i) develop a vegetation map, (ii) determine the fire history for their area from ten years before project commencement to calculate baselines, (iii) apply the fire management in an area >1 km^2 with >1000 mm yr^{-1} rainfall that contains a class of vegetation specified in the methodology, and (iv) not use cattle to control fire or increase fire outside the project area.
- For native vegetation planting, project owners must (i) plant species native to the local area (mixed or monoculture), (ii) establish permanent plantings on land that was wholly or partially cleared for five years prior to planting (excludes natural forest regeneration/regrowth), (iii) ensure that there is potential for the project to attain at least 20% crown cover and height of 2 m, (iv) not harvest wood products (excluding 10% debris removal per year for personal use), (v) prevent livestock grazing for the first three years after planting, and (vi) avoid 'ripping and mounding' in high rainfall areas in at least 90% of the area.

Agriculture and forestry contribute approximately 18% of Australia's emissions, so the potential for the land sector to contribute to emissions abatement is high. Another government program: the 'Carbon Farming Initiative' (www.climatechange.gov.au/cfi; Box 2) has thus also been established with the aim of reducing greenhouse gas emissions and increasing the sequestration of carbon in biomass and soils. It has been argued that the Initiative does not specifically integrate biodiversity considerations; rather, it employs safeguards to reduce the probability of further damage to biodiversity values (van Oosterzee, 2012). Nonetheless, the potential modifications to the Australian landscape under the Initiative are substantial, with associated broad-scale consequences for biodiversity persistence.

So-called mitigation 'methodologies' – the processes through which carbon credits are created, quantified, bought and sold – are numerous and the subject of ongoing research, validation and implementation, but we do not debate the efficacy of these methodologies here. We therefore assume that any methodologies resulting in landscape transformation conform to existing legislative requirements. These requirements include *inter alia* that sequestration is *permanent* (more specifically, that the carbon-related investment results in sequestration for a minimum of 100 years and this obligation is recorded on the land title), is *additional* (i.e., that the intervention ends up with more carbon sequestered than would have occurred without intervention), and that any unanticipated increase in emissions outside a project's accounting boundary, called *leakage*, is avoided or accounted for (van Oosterzee et al., 2012) (Table 1). To date, only two of the four

Table 1
Glossary of terms.

| Term/expression	Definition
Additionality	For a carbon-sequestration or -abatement project, ensuring that the intervention ends up with more carbon than would have been sequestered without intervention
Afforestation	Planting trees in a naturally treeless area
Baseline	According to Carbon Farming Initiative methodologies (Box 2), a baseline is a temporal or spatial reference point from which additionality is assessed
Bioenergy	Burning forestry by-products and woody wastes to generate electricity
Biosequestration	Process of storing carbon from greenhouse gases by photosynthesis
CO_2-e	The global warming potential that a given type and amount of greenhouse gas causes relative to one unit of carbon dioxide
Enteric fermentation	In herbivorous animals, the digestive process involving microorganisms to break down carbohydrates into simple molecules for absorption into the bloodstream; produces methane
Environmental plantings	Sequestration of carbon through planting endemic plant species
Greenhouse gas	An atmospheric gas that absorbs and emits radiation within the infra-red range, principally water vapour, carbon dioxide, methane, nitrous oxide and ozone
Kyoto protocol	An international agreement setting binding targets for 37 industrialised countries and the European community for reducing greenhouse gas emissions (see Box 1)
Leakage	For a carbon-sequestration project, the unanticipated increase in emissions outside a project's accounting boundary (in this case, domestic leakage only)
Carbon offset	A reduction in the emissions of greenhouse gases to compensate for emissions made elsewhere
Offsets integrity standards	Requirements regarding leakage and permanence with which methodologies must conform under the Carbon Farming Initiative Act
Permanence	For a carbon-sequestration project, the carbon-related investment must result in sequestration for a sufficient period into the future to account for the carbon being offset by a (carbon-credit) buyer
Reforestation	Planting trees in human-cleared treeless areas
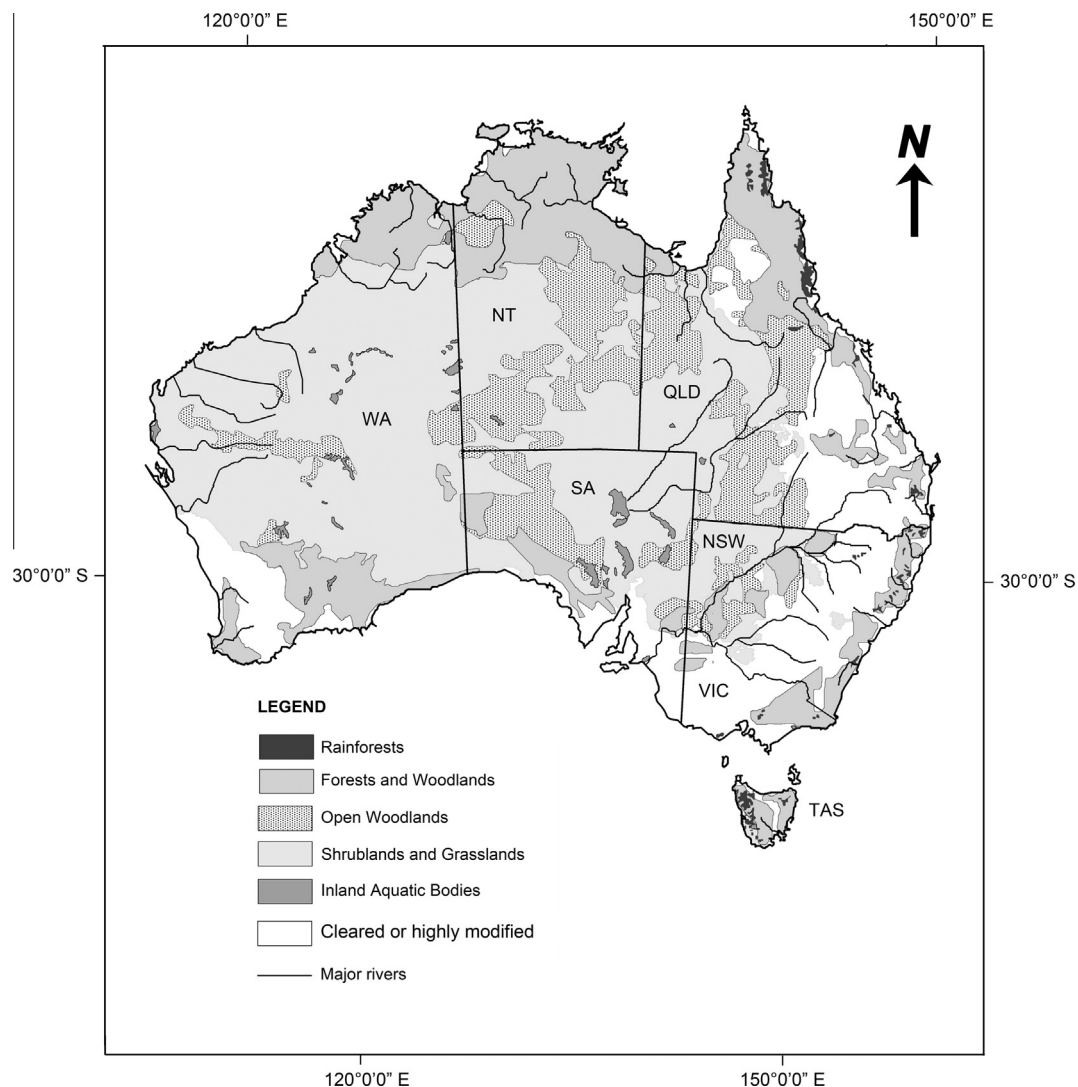
Fig. 1. Vegetation coverage of Australia derived from the NVIS Major Vegetation Groups identified in the Australian State of the Environment Report 2011, Commonwealth of Australia (2011). States/Territories are shown as follows: New South Wales (NSW), the Northern Territory (NT), Queensland (QLD), South Australia (SA), Tasmania (TAS), Victoria (VIC) and Western Australia (WA).

methodologies conforming to these requirements and approved under the Carbon Farming Initiative (Box 2) are relevant to this review: (1) savanna burning and (2) environmental plantings of native species (see Box 2 for more details on these methodologies); other methodologies discussed here are at various stages of development.

3. Environmental plantings

Recently, many regions of the world have prohibited broad-scale forest clearing and are now implementing reforestation initiatives to compensate for historical losses. Ecosystem restoration has traditionally been focused on issues such as biodiversity protection, improvement of soil health and salinity management, but there is an increasing interest in using land and vegetation for carbon sequestration (Hatton et al., 2011).

3.1. Anticipated changes under carbon-price legislation

One of the uses of the funds generated from the Australian carbon price is investment in the sequestration of carbon through tree plantings, variously referred to as 'carbon plantings', 'biodiversity plantings', 'environmental plantings', 'enrichment plantings' or 'carbon forestry' (Eady et al., 2009; Crossman et al., 2011). Here we use the term 'environmental plantings' to include the aforementioned definitions, and note that this excludes commercial plantations. Environmental plantings can include afforestation (planting naturally treeless areas) or reforestation (planting human-cleared areas). Both produce carbon credits tradeable under the Kyoto Protocol (Box 1; Table 1). Carbon sequestered by activities that do not comply with the Kyoto Protocol (Box 1), such as non-forest revegetation with native species, can still be traded in Australia through voluntary carbon markets and will be subsidised through a Carbon Farming Initiative Non-Kyoto Carbon Fund.

In Australia, ~40% of total forest cover has been lost in the last 200 years (Bradshaw, 2012), so environmental plantings have considerable potential to contribute both to forest biodiversity enhancement and carbon sequestration. Total annual sequestration potential in soils and vegetation in Australia is estimated to exceed 1000 Mt CO2-e year-1 for the period 2010–2050 (Eady et al.,2009). Estimates of potential storage in environmental plantings range from ~350 Mt CO2-e year-1 (Eady, 2009; Wentworth Group of Concerned Scientists, 2009) up to ~600 Mt CO2-e year-1 (Burns et al., 2011), while those for forestry (timber plantations) range from ~265 Mt CO2-e year-1 (Burns et al., 2011) up to ~400 Mt CO2-e year-1 (Wentworth Group of Concerned Scientists, 2009). Capturing 15% of this capacity would offset the equivalent of ~25% of Australia's current greenhouse gas emissions over that 40-year period (Wentworth Group of Concerned Scientists, 2009).

Carbon sequestration via environmental plantings is intended to increase net forest cover with native tree species, either as environmental 'woodland' plantings or as native hardwood plantations (see also *Forestry* section below). 'Woodland' plantings involve establishing multiple strata of trees and shrubs, usually of mixed species, but they can also be monocultures of local provenances (Eady et al., 2009). Establishment can be from seedlings, direct seeding, or regeneration from remnant vegetation (see also *Regrowth* section). Because carbon sequestered in vegetation and soils can be released into the atmosphere through fire[2] or logging, sequestration projects covered by the Carbon Farming Initiative are subject to permanence obligations (Department of Climate Change and Energy Efficiency, 2010b); environmental plantings (apart from woodlots) cannot be harvested for timber.

The carbon price-driven changes to forest cover and composition will have major implications for hydrology in previously forested areas. Rainfall is one of the most important factors limiting plant growth in Australia (Raupach et al., 2001; Roxburgh et al., 2004), and prior to the establishment of a carbon market, timber plantations were considered economically viable only in areas where rainfall exceeds 600 mm year^{-1} (Zhang et al., 2007). Crossman et al. (2011) have demonstrated the sensitivity of carbon farming profitability (relative to existing land use) to a carbon price in the agricultural areas of southern Australia. Environmental plantings on more agriculturally marginal land became financially viable only at relatively high carbon prices; the percentage of agricultural land area shifting to environmental plantings ranged from <20% at AU$10 t^{-1} CO$_2$-e to ~90% at AU$45 t^{-1} CO$_2$-e. Monoculture plantations are predicted to be far more profitable than environmental plantings at intermediate carbon prices (Kapambwe and Keenan, 2009; Crossman et al., 2011). Thus, as the price of carbon increases, carbon farming will potentially expand into more marginal (both in terms of agricultural potential and native forest productivity) and lower-rainfall areas. It is in these areas where the impacts on water yield will be greatest (Zhang et al., 2001).

3.2. Making sure plantings work for biodiversity

3.2.1. Potential benefits

The excessive fragmentation of forested habitats in many parts of Australia is a considerable threat to native biodiversity (Bradshaw, 2012); therefore, increases in the extent of forest cover in human-modified landscapes can be expected to prolong the persistence of forest-dependent species by (1) increasing the area of habitat (Koh et al., 2010), (2) increasing connectivity (Watling et al., 2011), and (3) minimising the edge:core habitat ratio (Laurance and Curran, 2008) via increasing average patch size. The potential benefits of reforestation extend well beyond biodiversity preservation *per se* – many valuable ecosystem services (sensu Costanza et al., 1997) also can be enhanced, including increasing pollination efficiency for higher agricultural yields (Hoehn et al., 2008; Carvalheiro et al., 2011), freshwater purification (Daily, 1997) and flood regulation (Bradshaw et al., 2007b).

As forest habitats become fragmented by human activities, the populations within them are exposed to different threats due to their reduced range and small population size. They are more likely to be driven extinct through stochastic events, such as wildfire or extreme weather (Lande, 1993). Environmental plantings can increase the long-term persistence of populations in fragmented habitats by creating new suitable habitat patches. Whether they are specifically designed as habitat corridors connecting isolated patches, or 'stepping stones' through which mobile species can disperse, environmental plantings can help sustain a species. Just how many populations or species benefit depends on the composition and structure of the species used in the plantings. There is also growing evidence that the composition of intervening ('matrix') habitat separating habitat fragments is an important determinant of the composition and persistence of biodiversity within habitat fragments (Watling et al., 2011). For forest-dependent species, we can efficiently maximise the co-benefits of environmental plantings for biodiversity by locating plantings to reconnect existing patches (Venter et al., 2009).

Benefits are also enhanced by integrating environmental management programs and objectives and increasing their spatial overlap (Fahey et al., 2009). For example, focusing plantings and revegetation efforts around some aquatic systems could enhance both water quality and ecosystem connectivity; simultaneously

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[2] According to the Australian Carbon Farming Initiative: "If carbon is lost because of bushfire, drought, disease or requirements to establish firebreaks, landholders are not required to return credits"; however, "landholders must take reasonable action to ensure that carbon stores are re-established following natural disturbances. Project proponents will not receive credits while the carbon stores are recovering."

aligning efforts to expand habitats and thence viability of threatened species and ecological communities would address another core Australian Government commitment (Wentworth Group of Concerned Scientists, 2008). However, deciding which habitat fragments to reconnect and where to initiate environmental plantings for maximum biodiversity benefit is still a developing field (Beier et al., 2008).

Plantings for carbon sequestration can potentially alter catchment water balances, thereby offsetting the effects of past deforestation such as dryland salinity and high rates of sediment and nutrient export (Jackson et al., 2005; George et al., 2012; Sochacki et al., 2012). Even away from the stream margins, forested areas can enhance aquatic diversity by buffering streams from the effects of climatic variability (Thomson et al., 2012). In their natural condition, riparian habitats are structurally and functionally diverse (Naiman and Decamps, 1997), so the extent of riparian buffers required to restore biodiversity values from replantings is highly variable and the subject of considerable ongoing research (Hansen et al., 2010).

Research on the biodiversity value of commercial and environmental plantings suggests that, in general, the habitat value of revegetation increases with floristic diversity, structural complexity, age of planting, patch area, and proximity to remnant native vegetation (Hartley, 2002; Martínez-Garza and Howe, 2003; Munro et al., 2007; Brockerhoff et al., 2008; Bremer and Farley, 2010). However, there is also considerable variation in benefits between faunal groups in revegetated areas. For example, while revegetated areas provide habitat for many species of bird and some arboreal marsupials, there is little evidence that bats, small terrestrial mammals, reptiles or amphibians benefit from revegetation in the short term (Kavanagh et al., 2005; Munro et al., 2007; Kavanagh et al., 2010; Law et al., 2011). Even among birds, environmental plantings are often used mainly by common, generalist species, which are usually of least conservation concern (e.g., Selwood et al., 2009). Indeed, revegetation might promote invasion by non-native species that often dominate disturbed forest edges (Sizer and Tanner, 1999).

The longer-term benefits of environmental plantings, both in terms of their biodiversity value and eventual sequestration potential, are difficult to assess because most plantings are <30 years old. Active management of young plantings will be required to supplement habitat resources, manage feral animals and weeds (Ferretti and de Britez, 2006; Munro et al., 2011), and maintain natural processes such as fire (Hartley, 2002). For example, tree hollows and coarse woody debris can take many decades to develop naturally (Vesk et al., 2008). However, the addition of coarse woody debris and artificial hollows to facilitate colonisation by some dependent species (Hartley, 2002; Munro et al., 2007) is unlikely to occur solely through carbon farming. Although environmental plantings might increase habitat area or quality in the ways mentioned above, decisions that prioritise sequestration alone are likely to encourage monocultures, or fast-growing, low-diversity timber stands that maximise carbon gains (Bekessy and Wintle, 2008; Munro et al., 2009). Indeed, the currently approved Carbon Farming Initiative methodology for native species planting permits native monocultures (Box 2). The biodiversity values of such plantations are likely to be low (Forrester et al., 2006; Kirby and Potvin, 2007). Environmental plantings will only have real biodiversity value if they comprise appropriate native tree and shrub species and provide suitable habitats and resources for valued fauna, which has been recognised through carbon-farming initiatives (Department of Climate Change and Energy Efficiency, 2011) such as the Australian Clean Energy Future Biodiversity Fund (www.cleanenergyfuture.gov.au/biodiversity-fund).

The carbon stock of undisturbed native forests is considerably greater on average than forests subject to commercial logging and monoculture plantations (Moroni et al., 2010; Preece et al., 2012), because the high rate of carbon fixation in younger, regenerating commercial plantations does not compensate for their smaller carbon stock (Mackey et al., 2008). The sequestration value of commercial plantations increases when substitution benefits are considered. Commercial plantings with greater genetic, taxonomic and functional diversity of natural forests are likely to be more resilient to disturbance (e.g., fire, pests, diseases) and resistant to climate change (Mackey et al., 2008). While substantial areas have already been revegetated across Australia using environmental plantings, no national protocols are in place to track, monitor or evaluate their outcomes (Atyeo and Thackway, 2009).

3.2.2. Potential negative effects

The first requirement to avoid perverse environmental outcomes from environmental plantings is that they should not replace existing native vegetation with smaller carbon stocks, such as grasslands and shrublands (Kapambwe and Keenan, 2009; Bond and Parr, 2010; Lindenmayer et al., 2012). While this will not be permitted in most regions of Australia given strong anti-clearing legislation (Bradshaw, 2012; Supplementary Table S1), it remains a considerable danger in many other countries (e.g., Koh and Wilcove, 2009).

Forest regrowth and large-scale plantings also might have strong negative effects on catchment hydrology in agricultural areas (Vertessy et al., 2003), with lowered water tables having profound consequences for existing vegetation (Struhsaker et al., 1989; Jayasuriya et al., 1993). Also, when forest cover exceeds 15–20%, the reduction in runoff is typically proportional to the percentage of catchment that is forested (Zhang et al., 2007). In low-rainfall areas, this results in seasonal or permanent loss of stream flow (Lane et al., 2003). In a global analysis of runoff changes in 504 catchments, Jackson et al. (2005) reported that commercial plantations (>20% catchment area; median 80%) caused streams to dry completely for at least a year in 13% of the catchments studied. The loss of perennial flow and extension of cease-to-flow periods can have substantial, negative ecological impacts (Boulton and Hancock, 2006; Lake, 2011), but might often be overlooked from a hydrological perspective because of the minor contribution of dry season flows to total water yields.

The ecological effect of environmental plantings in reducing runoff is further exacerbated by pre-existing channelisation and erosion of riverbanks, which have caused river incision and sediment deposition in the bed, greatly reducing in-stream hydraulic retention (Page and Carden, 1998; Mactaggart et al., 2006). This results in reduced hydrological permanence of small streams and rivers (Mactaggart et al., 2006). There is a consequent risk that riverine, lacustrine and palustrine dry-season refuges will be lost, even in areas of low pre-European tree densities (Bond and Lake, 2004). These issues will need to be carefully considered against the benefits of broad-scale, high-density plantings.

4. Regrowth

Although much of the developing world continues to clear its forests at alarming rates (Bradshaw et al., 2009), broad-scale vegetation clearing in Australia has effectively ceased (Bradshaw, 2012). Small-scale clearing continues (see also Supplementary Table S1) and southern and eastern landscapes in Australia that were formerly treed still suffer from extensive fragmentation and from extinction debts (McIntyre and Hobbs, 1999; Bradshaw, 2012). In addition, rural die-back (Landsberg and Wylie, 1988) will ensure that many modified, former wooded landscapes continue to lose tree cover and alter in composition over coming decades. Nonetheless, regrowth of native vegetation in many formerly treed, shrubby or perennial grassy landscapes is becoming more
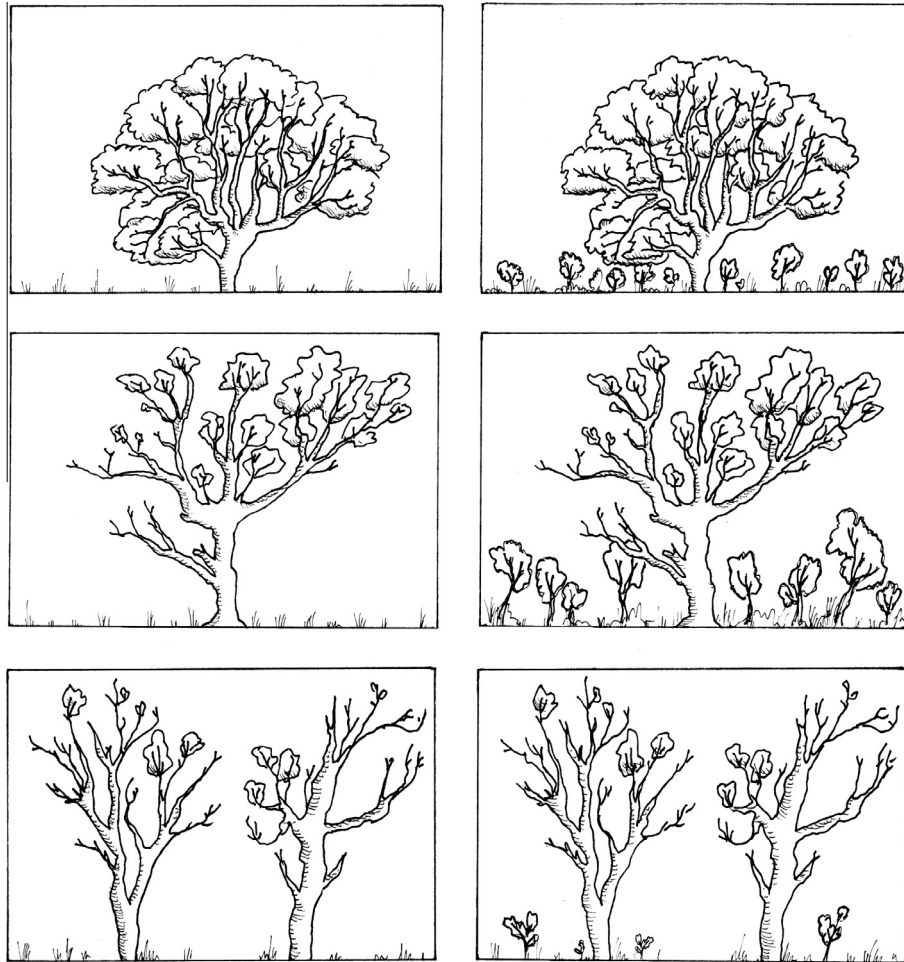
Fig. 2. The processes of regrowth illustrating various syndromes of modified grassy woodlands. A lack of regrowth (left panels) caused by intensive grazing management can generally be addressed by changing management practices to achieve a more desirable plant community structure (right panels). Where there are problems with too much regrowth (right panels), this can generally be addressed by changing management practices to achieve a more acceptable plant community structure (i.e., the reference condition of less regrowth) (left panels). By 'acceptable', we refer either to a baseline from which additionality can be assessed after the introduction of regrowth management. The sketches also show the issue of how tree health and vigour can mitigate regrowth (left panels, top to bottom) and the result (right panels, top to bottom). The conclusion is that if more vigorous regrowth is desired, management should focus on enhancing younger, more fertile trees.

prevalent (Dorrough and Moxham, 2005), much as it is in the United Kingdom and Europe following agricultural abandonment (Stoate et al., 2009). Depending on the severity and intensity of management practices over time, native vegetation systems retain varying degrees of regrowth, often described as 'regenerative capacity' (Thackway and Lesslie, 2008) (Fig. 2).

Regrowth can have both desirable and undesirable outcomes according to the environmental, economic or social contexts, including public–private land conservation management partnerships (Thackway and Olsson, 1999). However, the appropriate management of regrowth could offer a range of benefits for both biodiversity (Woinarski et al., 2009) and carbon sequestration.

4.1. How regrowth management will change under carbon pricing

By *regrowth* management, we mean the action of keeping (i.e., not clearing) existing human-modified native vegetation of varying ages and condition states on agricultural land, or avoiding cropping and continuous grazing in areas previously under agriculture (Thackway and Lesslie, 2008). Remnant (unmodified) vegetation is largely protected in Australia, and cannot be included in carbon-trading schemes due to additionality restrictions. Therefore, vegetation retention under Australian carbon legislation defaults

to regrowth. Since most young regrowth can be legally re-cleared for agriculture (Supplementary Table S1), there is an opportunity to protect it for its carbon-sequestration role. Regrowth falls within the definition of environmental 'woodland' planting (Eady et al., 2009) and potentially also within the definition of reforestation under the Kyoto Protocol.

There are, however, substantial differences in the protection offered to regrowth in different jurisdictions (see Supplementary Table S1 for state-specific legislation). For example, regrowth is protected if: >10 years since last clearing or >50% foliage cover of vegetation present prior to European settlement in New South Wales; if >70% of the pre-European canopy height in Queensland; and >50% of the pre-European species composition in Victoria. Other legislative complications mean that in some states (New South Wales and Queensland), native regrowth in over-cleared landscapes might already be protected for biodiversity as the only remaining habitat or as an endangered ecological community (Supplementary Table S1). Lastly, some regrowth vegetation is declared to be 'invasive native species' (e.g., in New South Wales; see Supplementary Table S1) and so may be replaced with more productive vegetation cover, or re-cleared for agricultural production.

In some states, legislation prevents thinning of dense regrowth stands. In the Brigalow Belt of Queensland (Fig. 1), for example,

dense *Callitris* regrowth cannot be easily or cheaply removed to promote rehabilitation of native grasses and a productive grazing system, but offers the potential to sequester large amounts of carbon (McAlpine et al., 2011). A compromise may be to 'thin' the *Callitris* regrowth to create grassy woodlands more typical of the area prior to the introduction of domestic stock (i.e., pre-European conditions).

4.2. Regrowth benefits for biodiversity

Regrowth vegetation potentially has high biodiversity value (Woinarski et al., 2009), can sequester large amounts of carbon (Moroni et al., 2010), and when appropriately managed, can be restored to the vegetation state present at time of European colonisation. Management of regrowth for this purpose has other advantages over environmental plantings because intensive ecosystem restoration is not required, the species are locally adapted and therefore potentially more resilient, and the stands contribute to restoration of mature, local ecosystems within fragmented landscapes (Fensham and Guymer, 2009). In particular, *Acacia*-dominated regrowth (e.g., brigalow *A. harpophylla* and mulga *A. aneura*) not only has high biosequestration potential because such ecosystems are extensive, including on fallow agricultural and pastoral lands (Dwyer et al., 2009; Witt et al., 2009), but can benefit a wide range of native flora and fauna (Dwyer et al., 2009; McAlpine et al., 2011).

There is increasing awareness and knowledge of how to apply appropriate land-management practices to native vegetation to change ecosystem function (Thackway and Lesslie, 2008). For example, exclusion of grazing in and burning of brigalow regrowth ecosystems enhance carbon sequestration (Howden et al., 2001; Witt et al., 2009). However, some grazing combined with thinning of regrowth (e.g., to 4000–6000 stems ha^{-1} of brigalow) can also increase carbon retention without increasing fire risk (McAlpine et al., 2011), provided the net carbon benefits are positive after accounting for any emissions increases associated with livestock grazing. Integrating environmental plantings within a mixed agricultural enterprise can also reduce the loss of economic productivity to private landowners from reduced grazing or cropping opportunities.

Many of the same negative outcomes for biodiversity described for environmental plantings could arise from unmanaged regrowth, including increased fire risk as the wood biomass increases, inappropriate floristic composition, a reduced opportunity to manage feral animals within dense regrowth, and a lower availability of surface water. A potential method to reduce these risks is the high stocking density, fast-rotation (e.g., less than or equal to 90 days/year) grazing method (Fischer et al., 2009). This reduces the effects of continuous stocking while minimising vegetation damage and promoting regeneration. While complete exclusion of livestock grazing is commonly used to enhance natural regeneration, it imposes high opportunity costs, whereas fast-rotational grazing allows the land to remain economically productive. This provides a win–win opportunity for tree regeneration and commercial livestock grazing (Fischer et al., 2009). With the use of electric fencing, the increased infrastructure needed for fast-rotation grazing can be provided relatively cheaply. Another way to reduce fire risk in either revegetation or regrowth-management areas is to coordinate property-specific fire planning with regional fire management plans applied to neighbouring areas of native vegetation.

Setting clear definitions for 'baseline' vegetation targets (see also Table 1) – and well-planned management interventions that might include soil scarification, thinning, weed and feral animal removal and fertiliser restrictions – can also assist in planning and measuring the success of interventions to manage regrowth to meet multiple outcomes (Thackway, 2012). Regrowth management should include the requirement to pass an additionality test, such that improvement can be measured after the cessation of suppression of regrowth or the exclusion of livestock. Ensuring additionality might also be feasible in situations where vegetation is not protected and deforestation is avoided through the surrender of a permit or approval to clear.

5. Fire management

Deliberate application of fire under mild fire-weather conditions (prescribed burning) can influence the carbon cycle by: (1) indirectly increasing carbon storage and (2) abating non-CO$_2$
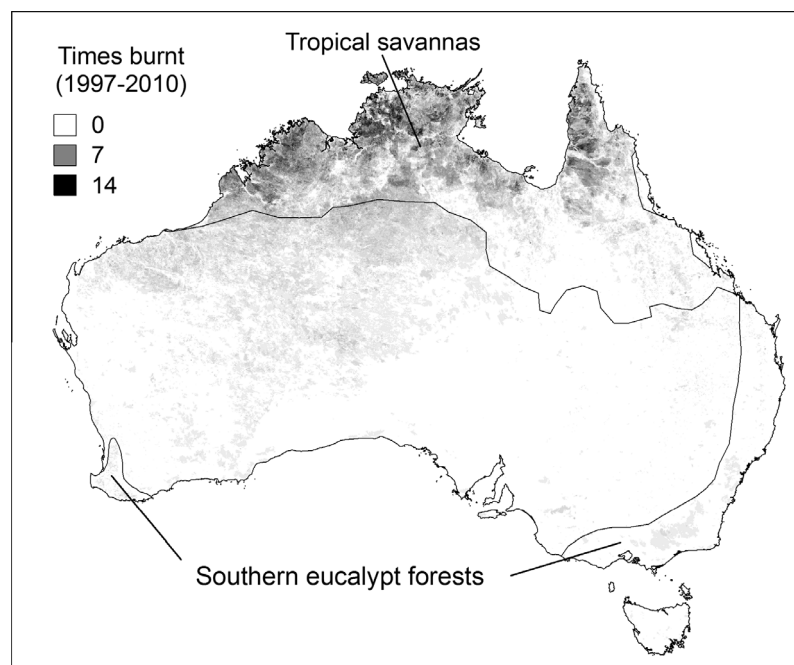

Fig. 3. Fire frequency throughout Australia from 1997 to 2010, inclusive, derived from AVHRR satellite imagery (Russell-Smith et al., 2007). The approximate domains of the tropical savannas in the north, and the eucalypt forests in the south are shown.

greenhouse gas emissions. There is clear evidence that certain fire regimes can increase relative carbon storage (Williams et al., 2012). High-intensity wildfires consume large amounts of biomass, leading to large losses of carbon to the atmosphere. Reducing the frequency of high-intensity wildfires should therefore increase biomass retention and carbon storage. Secondly, burning biomass produces large quantities of potent greenhouse gases such as methane and nitrous oxides (with global warming potentials ~21–25 and ~298 times that of CO_2 over 100 years, respectively) (Forster et al., 2007). Hence, minimising the amount of fuel burnt by landscape fires can generate a substantial carbon benefit, irrespective of any increases in carbon storage. Critically, such abatement of non-CO_2 greenhouse gas emissions is not subject to permanency requirements because it does not rely on increasing, or even maintaining, the amount of carbon stored in the system (Cook and Meyer, 2009).

5.1. Anticipated fire regime changes under carbon-price legislation

The emerging carbon economy is likely to affect fire management in north Australian savannas more than any other region, because this is where the largest areas of burning (Russell-Smith et al., 2007) (Fig. 3) and biomass consumed by fires (Department of Climate Change and Energy Efficiency, 2010a) are located. The Australian savannas are home to the world's first landscape-scale carbon and fire-management scheme: the 23,000-km^2 West Arnhem Land Fire Abatement (WALFA) project that generates a carbon credit of 100,000 t CO_2-e year^{-1} (based on abatement of non-CO_2 greenhouse gas emissions only) as a result of prescribed burning under a voluntary offset arrangement (Russell-Smith et al., 2009). A key objective of WALFA is to provide training and employment for local indigenous people, in a region where such opportunities are otherwise scarce, and to re-establish customary fire management practices. A recent cost−benefit analysis (Heckbert et al., 2012) suggests that WALFA-style fire projects are likely to be viable across 50 million ha of north Australian savannas, assuming a carbon price of $23 t CO_2-e^{-1}. Complementing voluntary offset projects such as WALFA, the Australian Government has recently approved a carbon–offset methodology using savanna fire management (Box 2) to abate non-CO_2 greenhouse gas emissions (Australian Government, 2012a) and a methodology for increasing carbon storage in savanna biomass is already under development (J. Russell-Smith, Charles Darwin University, Darwin, pers. comm.). Under both approaches, managers aim to reduce the amount of fuel consumed by fires by (1) burning under mild fire-weather conditions, thereby reducing fire severity, or (2) reducing the overall extent of burning.

The potential for fire management to generate substantial carbon credits in less fire-prone parts of Australia is limited. For example, modelling by Bradstock et al. (2012) suggests that prescribed burning in southern Australia's eucalypt forests is ineffective for generating carbon credits. This is because of a combination of factors: prescribed burning itself generates substantial greenhouse gas emissions, and wildfires are unlikely to encounter recently treated areas unless large areas of forest are treated, thereby negating the benefits of reducing wildfire frequency. A similar conclusion was reached for forested regions of the western USA (Campbell et al., 2011).

Fire is considered mandatory for the regeneration of Australian old-growth and regrowth eucalypt forests (McCarthy et al., 1999), but forest systems that require periodic disturbance for regeneration are particularly complex to manage. For example, some argue that in temperate flammable forests such as in the western USA, thinning of dense tree regeneration (that has developed due to decades of fire suppression and logging), followed by frequent, low-intensity fires, can make these systems more stable and stronger

carbon sinks by reducing the risks of severe stand-replacing fires or conversion to treeless vegetation (Hurteau and Brooks, 2011). However, others dispute this claim, arguing that carbon losses from thinning of forests will be greater than carbon loss from wildfires in untreated forests, and that large areas of fuel treatment are required to influence wildfire activity at the landscape scale (Campbell et al., 2011).

The use of fire in the management of Australian temperate tall, wet eucalypt forests is controversial and illustrates the complexities of managing flammable forests for biodiversity, wood production and ecosystem services. These forests require fire for regeneration (Ashton, 1981) but in the rare event that stands remain unburnt beyond the lifespan of the dominant eucalypts (>500 years), temperate rainforest can form a climax community that excludes eucalypts (Jackson, 1968). Conversely, high fire frequencies can favour multi-aged forests or non-forest vegetation with much lower biomass (Jackson, 1968).

5.2. Implications for biodiversity

Andersen et al. (2012) recently noted that, while carbon and biodiversity values are currently aligned in north Australian savannas, they might diverge if long-term fire exclusion was implemented because this could eventually reduce biodiversity values. Savanna fire managers aim to prevent large, high-intensity fires that typically occur late in the dry season (Murphy et al., 2009; Russell-Smith et al., 2009); there is ample evidence that this strategy will be beneficial to many components of biodiversity, since frequent, high-intensity fires cause the decline of rainforests (Russell-Smith and Bowman, 1992) and stands of the fire-sensitive conifer Callitris intratropica (Trauernicht et al., 2012), sandstone heaths (Russell-Smith et al., 2002) and probably also small mammals (Pardon et al., 2003; Woinarski et al., 2010, 2011). Indeed, the West Arnhem Land Fire Abatement project was established with the primary aim of re-imposing fire regimes conducive to the maintenance of biodiversity, subsidised by the generation of carbon credits (Russell-Smith et al., 2009). In the first six years of the project's operation, there has been a strong shift from a late dry season- (August–November) to an early dry season- (April–July) dominated fire regime, including a slight reduction in the area burnt annually (Price et al., 2012).

5.3. Improving fire-managed carbon for biodiversity

As described in Section 4.1, there appears to be little scope to use fire to manage carbon in southern Australia's forests because of the infrequent occurrence of landscape fire and the carbon costs of interventions. However in northern Australia, fire management projects are rapidly emerging with multiple objectives: carbon credits, biodiversity conservation and maintaining indigenous cultural values while providing socio-economic benefits for local indigenous people. A greater focus on reducing fire frequencies could greatly benefit savanna biodiversity (Woinarski et al., 2010, 2011). For instance, several researchers have recommended an urgent reduction in overall fire frequencies and the extent of area burnt to address the declines of fire-sensitive taxa such as small mammals (Woinarski et al., 2010; Andersen et al., 2012). However, implementing biodiversity-friendly savanna fire regimes remains an enormous management challenge. We cannot yet identify fire regimes that are optimal for maintaining biodiversity. Current management paradigms, such as 'patch mosaic burning', tend to be based on vague heuristic models for which there is little theoretical or empirical support (Parr and Andersen, 2006). Our capacity to impose optimal fire regimes is also limited, with some researchers questioning whether prescribed burning – the key fire management tool in the savannas – can actually deliver a

reduction in overall fire frequencies (Archibald, 2011). This conundrum is exemplified by a rapid decline of small mammals in Kakadu National Park over the last three decades, suspected to be partly linked to inappropriate fire regimes (Pardon et al., 2003; Woinarski et al., 2010, 2011), despite a well-resourced fire-management program in the park. Systematic research into the identification and implementation of optimal fire regimes for biodiversity conservation in Australia's fire-prone ecosystems is therefore urgently needed. There is also a need to evaluate different management scenarios within a formal decision analysis framework to ensure that multiple objectives are achieved.

6. Forestry

Forestry in Australia is in the final stages of a historic transition from past heavy reliance on old-growth forest for harvest (*Eucalyptus* and tropical and subtropical rainforests) to increasing harvests of plantations (Bradshaw, 2012), mainly of *Eucalyptus* and *Pinus*, as well as of regrowth eucalypt forests. Plantations have been established on native grasslands, agricultural lands, converted native forest and woodland and native grasslands (Grimbacher, 2011). Fire is often, but not always, used in the establishment of plantations, as it is a cheap and effective tool for removing logging debris. However, a price on emissions, concerns about smoke pollution on human health (Johnston et al., 2002), and runaway fires could make this practice less attractive.

Australian foresters have developed a silvicultural system to mimic the natural fire regimes of tall eucalypt forests (Mount, 1979). Key elements of the system are clear-felling large areas (c. 50 ha), burning the logging debris in intense fires, and sowing eucalypt seeds on the ashbed (Hickey et al., 2001). The primary objective is to produce cohorts of young eucalypts with a shorter rotation (c. 100 years) that grow more rapidly than trees in old-growth stands. However, Lindenmayer et al. (2011) argue that an unappreciated aspect of forestry in tall eucalypt forests has been the development of a 'landscape trap'. Here, landscapes are shifted into a highly compromised structural and functional state that is maintained as the result of multiple temporal and spatial feedbacks between human and natural disturbance regimes. However, Ferguson and Cheney (2011) reject this view, claiming that frequent wildfires create 'landscape traps' whereas logged areas contain lower fuel mass, thereby reducing wildfire intensity and risk. The issue becomes more problematic given that climate change will likely increase the frequency and intensity of dry periods, and thereby increase the risk of fire in southern Australia's forests while reducing their capacity to recover from disturbance (Caccamo et al., 2012).

6.1. Anticipated forestry changes under carbon-price legislation

In addition to established concerns about the aesthetic, biodiversity and soil erosion impacts of industrial forestry, traditional approaches to forestry management have been criticised because of the mounting need to minimise greenhouse gas emissions and maximise carbon storage. Given the high carbon density of old-growth, tall eucalypt forests (Keith et al., 2009), some advocate that these forests can become substantial carbon stores if the remaining old growth stands are conserved and logging rotation periods are increased (thereby realising the coupe's 'carbon carrying capacity' and providing more habitat area for old-growth-specialist species) (Mackey et al., 2008). Others argue instead that the concept of 'carbon carrying capacity' is unrealistic for tall eucalypt forests because of their inherent dependence on wildfires (Moroni et al., 2010), prompting a debate about the measurement and interpretation of estimated forest carbon stocks in these systems

(Dean, 2011; Moroni et al., 2012). For example, it is unrealistic to assume that all forest stands can simultaneously support maximum carbon stocks, which raises the question of what is an appropriate mix of stand ages in a landscape. Dean et al. (2012) argue that lengthening the harvest rotation from 80 to 200 years would substantially reduce long-term emissions from these forests, which would benefit a wider array of forest-dependent species (Lindenmayer and McCarthy, 2002). Modifications to forest management practices, such as lengthening rotation, are not covered by the current version of the Carbon Farming Initiative.

Mitchell et al. (2012) note four major ways that forestry can contribute to carbon mitigation under the Kyoto Protocol: (1) afforestation or reforestation; (2) avoidance of deforestation; (3) forest management, and (4) bioenergy. They note that forest management was not included in Australia's targets for the Kyoto Protocol because of "difficulties in establishing a baseline estimate and potentially large emissions from fire and drought". Evaluation of the impact of logging on the carbon balance must consider the life cycle of the wood products, emissions associated with harvesting, and the carbon embedded in products used as substitutes if eucalypt forests are not harvested (i.e., leakage) (Harmon et al., 1990; Lippke et al., 2011; Dean et al., 2012; Moroni, 2012; Ximenes et al., 2012). Currently there is a paucity of data to derive a holistic appreciation of the carbon budget of forestry in Australia.

Forestry wastes remain a peripheral energy source: only 1% of Australia's bioenergy (bioenergy itself representing only 1% of the nation's electrical supply) is provided by these wastes (Mitchell et al., 2012). Yet, forest management and bioenergy are important and controversial for forest biodiversity conservation. Improved forest management could reduce carbon emissions, and of particular importance are the greenhouse gas emissions resulting from the silvicultural practice of burning logging debris in intense fires. Avoiding post-logging burning would lead to a substantial reduction in greenhouse gas and protect coarse woody debris and organic soil horizons (Slijepcevic, 2001). Yet leaving heavy loads of logging debris after logging is unrealistic given the high fire risk posed by these fuels. One mooted alternative is to burn logging debris in furnaces to generate electricity (bioenergy). This was classified as an eligible renewable energy source under the *Commonwealth Renewable Energy (Electricity) Act 2000* until a recent controversial amendment to the Act.

6.2. Improving forestry for biodiversity in a carbon economy

The optimal way to manage flammable forests to maintain biodiversity values and carbon storage remains elusive. At one pole of the debate is an attempt to log forests 'sustainably', while the other aims to use native forests to sequester carbon while maintaining old-growth forest biodiversity values. Both approaches are vulnerable to perverse outcomes. For example, exclusive conservation of Australian forests could increase destruction of tropical forests to provide sawn timber supplies (an example of international 'leakage'), increase greenhouse gas pollution from wood substitutes such as steel, and increase competition for land and water between plantations and agriculture (Mitchell et al., 2012). Further, Ximenes et al. (2012) argue that increasing the reservation status of existing Australian production forests could instead lead to an increase in greenhouse gas pollution because of forgoing the opportunity to generate renewable energy from forests, which will become increasingly profitable as the carbon price rises, especially if the conversion of biomass to liquid fuels becomes commercially feasible (Mitchell et al., 2012). Thinning forests could provide a way of reducing destructive fires, thereby protecting biodiversity, increasing carbon stocks and producing bioenergy (Lippke et al., 2011; Bowman et al., 2013). Conversely, unsustainable forestry can result in ecologically degraded forests that store a small

fraction of their carbon potential, have reduced biodiversity due to loss of tree hollows and coarse woody debris, and are vulnerable to recurrent landscape fires (Lindenmayer and McCarthy, 2002). Another negative implication for Australian biodiversity is that the recent application of a price on carbon emissions in Australia potentially makes woody material a more economically attractive product. This is because forest products have become low-emissions goods given that the forest industry will not have to pay for emissions from fertilisers, timber harvesting or off-road vehicles and machinery (Australian Government, 2012b). Thus, under current policy settings and directions, woody biomass management could face an increasing tension between harvesting to supply society with low-emission products, and storing carbon in landscapes by avoiding harvesting.

There is therefore a profound philosophical tension between the harvest of wood products and preservation of forests for biodiversity. In the future this debate will no doubt be sharpened greatly by the urgent need to find sustainable energy supplies to replace fossil fuels and the recognition of the limits to forest sequestration shaped by the saturation of forests managed for carbon, demands for agricultural lands and climate change effects on the biosphere. Active forest management to store carbon and conserve biodiversity is essential, especially in Australia where landscape fire is an essential feature of most ecosystems. Such management should be evidence-based and adaptive, an approach that is at odds with the current common simplistic and binary attitudes about forestry.

7. Agriculture

In the 60% of Australia that is devoted to cropping and grazing, numerous interventions have been proffered to sequester carbon or otherwise reduce net greenhouse gas emissions. At the time of writing, only a small number of these interventions have reached the 'positive list' of activities within the Carbon Farming Initiative (Box 2). In this section therefore, we will mainly consider activities that are serious candidates for future inclusion on the positive list and that will affect ecosystem function (either on- or off-site) sufficiently to have substantial effects on biodiversity. These interventions include changes to the management of soils, increases in the perenniality and structural complexity of plant communities in grazing lands, and reductions in the numbers of grazing ruminants.

7.1. Anticipated changes under carbon-price legislation

7.1.1. Soil management

The application of nitrogenous (N) fertilisers under unfavourable conditions results in the release of nitrous oxides, which are powerful greenhouse gases (Dalal et al., 2003). These N losses represent environmental pollution and a direct economic cost to agriculture. In addition, considerable CO_2 is emitted from energy sources during the manufacture and transport of N fertilisers. Interventions to make N fertiliser applications more efficient or to replace them with biologically fixed N are therefore likely to be encouraged by a price on greenhouse gas emissions.

Amendment of soil with biochar (the residue from pyrolysis of plant biomass) is a means of sequestering carbon in the soil on millennial time scales (Sohi et al., 2010). Biochar addition appears to improve agricultural productivity in many situations; while the mechanisms are unclear (Sohi et al., 2010), this increases its attractiveness as a potential carbon–offset methodology. Reductions in tillage frequency increase ground cover and improve soil structure, thus increasing soil water storage and reducing runoff and erosion. While reduced tillage assists in maintaining soil carbon and potentially reduces greenhouse gas emissions (Radford and Thornton, 2011; Wang et al., 2011), it is too common a practice to meet the

additionality requirements for carbon credits; instead, a specific subsidy for equipment purchases to encourage further adoption of reduced tillage has been included in the overall Clean Energy Futures program.

7.1.2. Changes to agricultural vegetation

Introducing, or managing to increase, the perennial grass content of pastures and rangelands is likely to enhance carbon sequestration, either through higher overall net primary production or greater allocation of below-ground resources (Sanderman et al., 2010). In a tropical savanna, Ash et al. (1995) showed that pastures dominated by annual grasses had 42% less soil carbon than those dominated by perennial grasses. In annual-dominated agricultural systems in Mediterranean environments, kikuyu (Pennisetum clandestinum) is a favoured C_4 perennial (e.g., McDowall et al., 2003); there was a substantial soil carbon increase when annual pastures were converted to kikuyu in south-western Western Australia that was larger than the estimated leakage through higher methane emissions from livestock (Thomas et al., 2012). In temperate grasslands based on C_3 species, however, the difference in soil carbon accumulation under similarly managed perennial and annual pastures appears to be small (Chan et al., 2010). Sequestration of soil carbon requires a corresponding sequestration of soil nutrients (Kirkby et al., 2011), in particular phosphorus. Carbon-sequestration activities in using exotic perennial grasses temperate areas are therefore likely to require ongoing fertiliser inputs.

Increased retention and regrowth of native shrubs in arid and semi-arid agricultural landscapes are likely outcomes of the new carbon economy. Native shrubs have long been used as forage in the pastoral zone, especially during periods of drought. There has been interest over the last 20 years in integrating shrubs into low-rainfall farming landscapes, where many would have grown prior to clearing for agriculture (Lefroy et al., 1992). These shrubs have an economic value as a predictable source of feed in the dry season and in unfavourable years, and are expected to increase soil carbon stocks and to reduce soil erosion with its associated carbon losses (Monjardino et al., 2009). More recently, some shrubs such as Eremophila spp. (Vercoe et al., 2009) have been identified as containing bioactive substances that might mitigate enteric methane emissions from ruminants (Table 1).

Government agencies across Australia are now changing ground-cover management approaches to increase the vegetative ground cover protecting the soil surface from wind and water erosion in agricultural and rangeland areas. These changes will potentially lead to increased carbon sequestration and improved maintenance of biodiversity (Leys et al., 2009).

7.1.3. Reductions in ruminant numbers

Approximately 10% of Australia's total greenhouse gas emissions are methane gas produced by livestock enteric fermentation, with about half of these emissions derived from the northern Australia beef industry (Henry et al., 2012). Destocking areas of the Australian rangelands has therefore been proposed as a greenhouse gas-mitigation strategy with co-benefits for rehabilitation of over-grazed ecosystems (Eady et al., 2009). The potential for additional greenhouse gas mitigation via soil carbon sequestration following destocking is uncertain, as are the timeframe and extent of recovery in landscape function. Incremental changes, such as reduction in stocking rate, pasture resting and rotational grazing are more likely to be adopted by graziers aiming to increase soil carbon stores under native pastures. However, to be eligible under the Carbon Farming Initiative, for example, any shifts in grazing regime would have to meet additionality requirements. These options will be particularly relevant in the rangelands where grazing is almost

exclusively on native pastures, and other management options are limited.

An alternative to reducing grazing pressure in the rangelands is to shift the economic basis of the rangelands away from ruminants and toward kangaroos, which are foregut fermenters and produce much less methane (Wilson and Edwards, 2008; Madsen and Bertelsen, 2012) (Table 2). This proposed shift in rangeland use would be transformative; as Wilson and Edwards (2008) acknowledge, it would involve both the development of a completely new value chain and major social shifts amongst both producers and consumers of rangeland meat.

7.2. Implications for biodiversity

7.2.1. Soil management

Where nitrogen moves from agricultural systems into wetlands and water bodies, the change in nutrient balances will alter biological dynamics. Many aquatic systems have large denitrification capacity, but in the process, degradation of some species and shifts in species composition occur (Harris, 2001). More efficient methods of supplying nitrogen to crops and pastures that also reduce the movement of nitrogen into water bodies can therefore be expected to benefit biodiversity. Reduced tillage should increase soil faunal abundance, but the effects on soil biodiversity are mixed: diversity of microorganisms increases, but effects on the mesofauna are inconsistent (Holland, 2004).

There is little evidence on which to base an assessment of the consequences of biochar application for biodiversity, although Khodadad et al. (2011) report losses in soil microbial diversity upon amendment with biochar. Current application methods typically involve incorporation into the soil (Sparkes and Stoutjesdijk, 2011), implying that biochar applications in Australia will be limited to highly transformed agricultural lands. Consequences of biochar *production* for biodiversity should be similar to those of plantation forestry where the char is produced for woody biomass.

7.2.2. Changes to agricultural vegetation

The biodiversity consequences of introducing or managing perennial grasses for carbon sequestration will depend on which vegetation types they replace. In the rangelands, introduced C_4 grasses (such as buffel grass *Cenchrus ciliaris*) are sown widely. Introducing buffel grass decreases native plant diversity and abundance, hampers tree recruitment (Fairfax and Fensham, 2000; Clarke et al., 2005; Jackson, 2005) and increases the intensity of fires that kill overstorey plants, changing woodland structure (Miller et al., 2010). These effects on vegetation composition and structure reduce the diversity of native fauna, including invertebrates and reptiles (Eyre et al., 2009; Smyth et al., 2009) and small, browsing native mammals such as the bridled nailtail wallaby *Onychogalea fraenata* (Green, 2010).

In mixed-farming areas, the biodiversity consequences of a shift to sown perennial pastures are likely to differ. Bridle et al. (2009) found that farm-scale species richness correlates with the amount of pasture (and remnant native vegetation), as well as site and landscape characteristics. The consequences for biodiversity of a land-use shift to kikuyu pastures are unknown, but kikuyu (like buffel grass) is strongly competitive and likely to exclude native plant species. On the other hand, the introduction of kikuyu pastures is likely to be associated with a lengthening of pasture phases in rotations, allowing a build-up in structural diversity – and hence array of micro-habitats – compared to the annual pastures that they would replace.

Persistence of native plant species in the managed grasslands of the high-rainfall zone is negatively correlated with both grazing intensity and soil phosphorus (Dorrough and Scroggie, 2008). Stock utilisation rate is expected to decline in pastures managed for carbon sequestration, thus enhancing diversity of native plant species. However, the phosphorus fertiliser inputs likely required to maintain soil carbon sequestration rates can be expected to reduce biodiversity. Nonetheless, there appear to be management strategies for at least some native grasslands that support relatively high net primary production while maintaining vegetation diversity (Kemp et al., 2003). Grasslands with a perennial grass component are likely to be structurally more diverse, which should enhance invertebrate diversity (Reid and Hochuli, 2007). Retention and regeneration of native shrubs such as *Eremophila* and *Atriplex* spp., has potential to improve biodiversity persistence and recovery in low-rainfall areas. Indeed, plantings of saltbush (*Atriplex*

Table 2
Annual methane (and CO_2 equivalent) production for herbivore species introduced into Australia. Cattle, sheep (livestock) and kangaroo emissions are shown for comparison.

Species	Estimated population	kg CH_4 indiv^{-1} year^{-1} (source: NGGI[a])	Total t CH_4 year^{-1}	Total t CO_2-e year^{-1} (1 t CH_4 = 21 t CO_2-e)
Cattle	>20 million	~90	>1.8×10^6	>3.78×10^7
Sheep	68 million	~7	4.76×10^5	1.0×10^7
Kangaroo[b]	33.8 million[c]	0.14[d]	4.83×10^3	1.01×10^5
Pigs	13.5 million[e,f]	3.5[a,g]	4.73×10^4	9.93×10^5
Camels	>1 million[h]	39.3	>3.93×10^4	>8.25×10^5
Buffalo	0.35 million[i]	53.8	1.88×10^4	3.95×10^5
Goats	2.6 million[j]	4.8	1.25×10^4	2.63×10^5
Rabbits	200 million	0.05[k]	1.0×10^4	2.1×10^5
Deer	0.2 million[l]	11.9	2.38×10^3	5.0×10^4
Total livestock (cattle and sheep)			>2.28×10^6	>4.79×10^7
Total feral herbivores			1.30×10^5	2.73×10^6

[a] National Greenhouse Gas Inventory (http://ageis.climatechange.gov.au/NGGI.aspx).
[b] Red (*Macropus rufus*), western grey (*M. fuliginosus*), eastern grey (*M. giganteus*), euro (*M. robustus*).
[c] Mean population estimate between 2001–2006 for commercial harvest areas (Wilson and Edwards, 2008); therefore, all estimates represent minimum values.
[d] Based on 0.003 t indiv^{-1} yr^{-1} greenhouse gas equivalents (Kempton et al., 1976).
[e] (Department of the Environment and Heritage, 2005; Department of Sustainability, 2011b).
[f] Hone (1990a).
[g] Most greenhouse gas emissions associated with domestic pigs originate from their manure, but feral pig faeces will decompose in a largely aerobic environment, thus producing a much lower quantity of methane or N_2O. Therefore, direct methane emissions for feral pigs will be a maximum of 3 kg CH_4 indiv^{-1} year^{-1} (based on the National Greenhouse Gas Inventory[a] 0.7% of gross energy intake and 3.5 kg dry matter intake day^{-1}).
[h] Saalfeld and Edwards (2008).
[i] Bradshaw et al. (2007a).
[j] Department of Sustainability (2011a).
[k] Franz et al. (2011).
[l] Queensland Government (2012).

spp.) have higher plant and/or bird diversity than the crops and pastures that they replace (Seddon et al., 2009; Collard et al., 2011).

7.2.3. Reductions in ruminant numbers

Pastoralism in southern Australia is associated with declines in around half of Australia's terrestrial mammals and the extinction of >20 species (Fisher et al., 2003; Johnson, 2006). Overgrazing is also blamed for the decline of tropical granivorous grassland birds (Franklin et al., 2005). Resultant loss of biodiversity in rangelands appears to have occurred because (1) sheep and cattle have removed native pasture plants that were palatable to native browsers and grazers, (2) overgrazing has created adverse soil conditions such as compaction and erosion which are not conducive to biodiversity retention, (3) damage to ground-level vegetation caused by hard hooves increases habitat suitability and hunting success for introduced red foxes (Vulpes vulpes) and feral cats (Felis catus), and (4) persecution of dingoes in the sheep rangelands advantages foxes in particular (Johnson, 2006; Johnson et al., 2007). Destocking of rangelands in key areas of habitat should abate these drivers (Legge et al., 2011), although recovery from adverse soil conditions could be slow. Replacement of hoofed ungulates by soft-footed kangaroos in semi-arid rangelands would benefit biodiversity in much the same way as for destocking, in particular by disadvantaging feral predators (Grigg, 1989).

In some cases, however, destocking can indirectly reduce biodiversity in grazed savannas given that grazing has been linked to woody vegetation thickening (Burrows et al., 2002). For example, vigorous regrowth and proliferation of mulga (Acacia aneura) in south-western Queensland can result in higher concentration of soil carbon and above-ground biomass, but lower species richness (Witt et al., 2011). In mixed-farming areas, greenhouse gas mitigation via the removal of ruminants would be accompanied by a land-use shift toward cropping with nitrogen fertilisation that would almost certainly have negative consequences for plant, animal and soil fauna biodiversity, and increased greenhouse gas emissions.

Where livestock numbers are reduced rather than removed, the consequences of changes in grazing management on carbon sequestration and biodiversity will be determined primarily by their effects on vegetation structure and composition (Pringle et al., 2011). The effectiveness of more intensive rotational grazing systems in improving land condition and carbon stocks is unclear despite frequent claims of beneficial effects (e.g., Alchin et al., 2010). In a recent study in Queensland's rangelands, Hall et al. (2011) found no consistent differences in vegetation or land condition between cell grazing and conventional continuous grazing, which suggests that soil carbon stocks would also have been similar between grazing systems. Likewise, Sanjari et al. (2008) found no change in soil carbon stocks after five years of cell grazing in south-eastern Queensland.

7.2.4. Wider considerations

The most important feature of carbon-biodiversity interactions in agricultural and rangeland landscapes is that they are only part of a larger picture. In particular, only interventions that are economically viable for landholders will make their way into practice. For example, while reductions in nitrogen losses from fertilisers are clearly desirable from greenhouse gas-mitigation and biodiversity perspectives, it is essential for enterprise sustainability that any reductions in production be more than offset by cost savings and carbon credits. At the other end of the scale, a shift from a ruminant-based to a kangaroo-based rangeland economy would clearly have major biodiversity and greenhouse gas-mitigation benefits (Table 2); but a carbon price of at least $40/tonne CO_2-e (Wilson and Edwards, 2008), well above prices expected in the next decade, is required to make this proposal competitive with

existing practices – even before applying a risk premium for the major social changes involved.

An important area of uncertainty, where careful analysis and research is needed, is to identify easily adoptable practices such as perennial plantings. For these interventions, greenhouse-gas mitigation or biodiversity benefits alone might be insufficient to motivate landholders to adopt them, but the combination could result in a viable case. In the specific case of perennial forage plantings, analysis will have to account for a trade-off between the greenhouse gas and biodiversity benefits obtainable from reduced stocking rates, and the associated loss of livestock production.

Evaluating the biodiversity consequences of greenhouse gas mitigations arising from changing agricultural practices will be complicated by the different temporal and spatial scales at which they operate. Some of the intermediate causes, such as changed fire regimes, will be relatively rapid. Others, such as provision of habitat for fauna, will take years or decades to develop, and other impacts such as changes in nitrogen fluxes in wetlands will affect entire catchments over time. In the rangelands in particular, our capacity to assess these consequences will be limited by inadequate long-term monitoring (Fitzhardinge, 2012) and the measurement problems caused by inherent spatial and temporal variability.

8. Feral animals

Methane is estimated to be responsible for about 20% of anthropogenic global warming and comes predominantly from fossil fuels (gas leakage), ruminant-animal digestion, and anaerobic vegetation decay (Forster et al., 2007). As a climate-forcing agent, methane is 25 times more powerful than carbon dioxide when its effect is integrated over a 100-year time horizon (Forster et al., 2007). Feral herbivores in Australia, like livestock, produce methane through fermentation and from the decomposition of manure. Large ruminant herbivores such as feral buffalo (Bubalus bubalis) and camels (Camelus dromedarius) produce the most methane per individual (up to ~50 kg CH_4 year^{-1}) (Table 2). Feral pigs (Sus scrofa), which are much more abundant than buffalo and camels (Hone, 1990a; Department of Sustainability, 2011b), produce up to 36% of all methane emissions from Australia's feral animals (Table 2). In total though, feral herbivores produce a little more than 5% of the methane produced from cattle and sheep in Australia (Table 2). In contrast, marsupial herbivores produce much less methane per unit body weight (Wilson and Edwards, 2008) (Table 2).

8.1. Changing feral animal management with carbon pricing

Carbon price-driven incentives will conceivably be made available to offset the costs of feral animal control in Australia, but given the relatively small contribution of their emissions relative to livestock (Table 2), efforts to limit the emission of methane from feral animals are not expected to mitigate global climate change relative to agricultural shifts; the main benefits of reducing feral species will be to native biodiversity (Table 3).

Under the Australian Government's Carbon Farming Initiative (Commonwealth of Australia, 2011), carbon credits are potentially claimable for: "Introduced animal emissions avoidance projects; projects that avoid emissions of methane from the digestive tract of an introduced animal or emissions of methane or nitrous oxide from the decomposition of introduced animal urine or dung". Importantly, a project for which carbon credits can be claimed must be demonstrated to go "... beyond common practice in the relevant industry" (i.e., to be additional) and be on an approved list of activities, which currently includes the "... reduction of

Table 3

Potential benefits to biodiversity (both native plant and animal communities) from reductions in introduced herbivore populations based on their well-documented negative impacts.

Feral herbivore species	Negative impacts on native biodiversity
Camels (*Camelus dromedarius*)	• Damage to native vegetation through feeding and trampling • Suppression of recruitment in native plant species • Damage to waterholes, wetlands and ephemeral rivers • Competition with native fauna (Edwards et al., 2010)
Pigs (*Sus scrofa*)	• Negative population-level effects on many plant, invertebrate and vertebrate prey consumed (e.g., northern snake-necked turtle) (Fordham et al., 2006; Bradshaw et al., 2007a; Fordham et al., 2008) • Extensive digging causes erosion/siltation and decreases plant species richness (Hone, 2002; Bradshaw et al., 2007a) • Weed dispersal (Bradshaw et al., 2007a)
Deer (*Dama dama, Cervus elaphus, Axis axis, Hyelaphus porcinus, Cervus timorensis, Rusa unicolor*)	• Changes in vegetation composition and density through seed dispersal of native and exotic species (Davis et al., 2010; Forsyth and Davis, 2011)
Goats (*Capra hircus*)	• Browsing/grazing has negative impact on native vegetation composition and structure (Russell et al., 2011) • Feeding habits decrease overall vegetation cover and hooves cause soil surface damage resulting in increased erosion (Russell et al., 2011) • Competition with native fauna for food and habitat (Russell et al., 2011)
Buffalo (*Bubalus bubalis*)	• Intensive feeding alters ground cover and plant diversity • Damage water ways (erosion/saltwater intrusion) (Bradshaw et al., 2007a; Albrecht et al., 2009)

methane emissions through the management, in a humane manner, of feral goats, feral deer, feral pigs or feral camels".

The one proposal made so far to reduce methane emissions from the control of feral herbivores in Australia addresses one of the largest emitters – camels (Northwest Carbon Pty Ltd., 2011). The growing population of camels in Australia is estimated to exceed a million individuals (Table 2); modelling suggests the production of 1.9 million t CO_2-e year^{-1} from their methane emissions by 2020, after accounting for expected further population increase (Northwest Carbon Pty Ltd., 2011). Using a variety of control techniques and taking fuel, electricity and other sources of greenhouse gas emissions from the harvest operation into account, the proponents claim a net overall reduction in greenhouse gas emissions from camel culling, although the magnitude of the reduction to be achieved remains unclear. One concern is that low-level harvest might do little more than stimulate a density-dependent 'rebound', and not achieve real density reduction. For instance, Pople and McLeod (2010) estimated that >11,000 camels would have to be removed annually from the approximately 140,000 in the Northern Territory to begin reducing the population size.

8.2. Challenges to reducing emissions and advancing biodiversity conservation

Australian landscapes suffer from intense ecological damage from feral animals (Table 3), and large reductions in feral animal densities will benefit Australian ecosystems substantially (McLeod, 2004; Bradshaw et al., 2007a). However, to reduce greenhouse gas emissions simultaneously is a complex proposition that requires stringent cost-benefit analysis and assessment of logistical challenges prior to implementation. Given the relatively low contribution of methane from feral herbivores relative to cattle and sheep (Table 2), it is unlikely that the suppression of pig densities would greatly benefit Australia's ability to meet its greenhouse gas emission targets. Broad-scale control programs for feral pigs, particularly if they involve shooting from vehicles or aircraft (e.g., Hone, 1990b; Reddiex et al., 2006), can also result in substantial operational CO_2 emissions. Another issue not previously considered is the potential role large feral herbivores play in reducing fire severity via their consumption of vegetation 'fuel' loads (Bowman, 2012), even though the contributions to greenhouse gases via methane release is approximately ten times that occurring from

fires (Cook et al., 2010) – clearly the feedbacks are complex. Compensatory responses in recruitment following lethal control of pigs can mean that control has a limited effect on either population growth rate or population density (Hanson et al., 2009). Further research is needed to ensure that reductions in feral animal numbers result in simultaneous reductions in net greenhouse gas emissions.

9. Discussion

Australia's largest potential greenhouse gas-mitigation activity using ecological processes rests in landscape-management interventions that enhance woody biomass. However, our review has highlighted many ways in which managing for carbon sequestration and emissions reductions can result in negative biodiversity outcomes. It is not unreasonable to suggest that current and likely future Australian policies covering woody biomass do not adequately explore the synergies between landscape carbon storage, biodiversity and resource use. Increasing pressure to exploit wood as a building material and energy source (Lippke et al., 2011; Ximenes et al., 2012) means that reforestation and afforestation could become more financially viable for many landholders. To benefit biodiversity while simultaneously providing the appropriate financial incentives, landscape-level biodiversity goals and management plans (including regional Natural Resource Management objectives) must be set in place before plantings proceed, although estimating the scale of realistic uptake would also require substantial economic modelling. The current gap in our understanding of the trade-offs between carbon sequestration and biodiversity requirements means that mutually beneficial outcomes of afforestation/reforestation are not guaranteed. For example, the likely impact of changes to forestry management and uptake under the Carbon Farming Initiative requires dedicated economic analysis (Burns et al., 2011; Maraseni and Cockfield, 2011; Polglase et al., 2011). The Australian Government must therefore strive to support research that investigates these trade-offs for the maximum benefit to society, including ecological, sociological and economic analyses. This might include analyses based on decision theory to ensure integration of biodiversity conservation with emissions-abatement objectives.

Nonetheless, we conclude that most biodiversity-related enhancement schemes (including environmental plantings and invasive species reduction) can be compatible with carbon-sequestration initiatives, as long as biodiversity persistence is taken into

account at the planning and implementation stages. Indeed, future conservation planning approaches will need to incorporate carbon sequestration/emissions abatement into their algorithms to optimise the simultaneous goals of biodiversity conservation and carbon mitigation. It is our opinion that, in most circumstances, the two goals are not mutually exclusive and indeed, can enhance components of the other. The careful amalgamation of such carbon-mitigation approaches with other incentive schemes such as biodiversity offsets (i.e., ensuring no net biodiversity loss via the protection of habitats as offsets for other areas cleared for development) will be required to ensure related negative outcomes do not arise (Bekessy et al., 2010). For instance, appropriate site selection, long-term monitoring and enforcement will be essential to ensure biodiversity improvement, or indeed, to avoid net loss (Kiesecker et al., 2009; Bekessy et al., 2010). In principle, we therefore support the implementation of financial mechanisms for the mitigation of carbon and enhancement of biodiversity, but only with the caveat that many potential loop-holes in Australian legislation that could lead to extensive negative outcomes for biodiversity must be identified and rectified. These include restrictions in regrowth management, incentives for planting solely for carbon sequestration (to the exclusion of biodiversity values) and the incentive to replace annuals with perennials in pastures without full consideration of their implications for biodiversity.

In addition to the direct intervention of planting forests to increase biomass, management of regrowth, fire, agriculture and feral animals might all play roles in greenhouse gas abatement. The potential role of some of these interventions is perhaps not well appreciated in relation to carbon sequestration, biodiversity or ecosystem comparisons (e.g., what might work in a temperate forest might be sub-optimal in tropical savannas). Clearly, effective greenhouse gas mitigation will require a multi-faceted approach from all land-use sectors to be successful. Of course, many uncertainties persist, including how different sectors respond to changes in emissions prices. It is difficult to anticipate or even model the corollaries of shifting prices to each sector, or whether thresholds exist where emissions pricing becomes functionally redundant for biodiversity maintenance. Another major consideration for the success of landscape carbon management is that a failure to stem global emissions arising largely from society's dependence on fossil fuels will mean that few net gains for biodiversity will be possible. It will be ultimately pointless to store carbon in the landscape if we do not reduce or remove this dependence to avoid the worst ravages of climate disruption on the Earth's biodiversity.

We have identified several gaps in our understanding that require more research or validation. For example, the contribution of land management practices to carbon dynamics in most of Australia – the arid and semi-arid centre – is still largely unquantified (Franklin et al., 2008). Much more research is required on the biodiversity and carbon roles of arid vegetation, its interface with pastoralism, and the role of fire (Fisher and Harris, 1999). Certainly, the most intriguing and possibly least-understood component of landscape carbon dynamics is the role of climate change itself (Hulme, 2005). Predicting forest resilience, agricultural trends, fire regime shifts, and the response of feral animals to a warming climate is in its infancy, so anticipating the effects on coupled carbon flux-biodiversity impacts is even less certain (Hoffmann et al., 2012). Other impacts, such as the potential for plantings to alter runoff, are perhaps better understood, but we still require careful regional-scale planning to avoid negative outcomes for biodiversity.

Finally, the essential component of measuring biodiversity, and the associated ecosystem services, effectively using meaningful metrics to determine the responses to landscape-scale changes remains a major challenge. Remote sensing and other broad-scale approaches are constantly improving, but we are still a long way

from consensus and standardisation (Jones et al., 2011) despite recent progress in the measurement of ecosystem services (Kapambwe and Keenan, 2009; The Economics of Ecosystems and Biodiversity, 2010; Perrings et al., 2011). This inconsistency contrasts starkly with the elements of carbon-accounting and mitigation methodologies that require strict validation at the biological, physical, economic and political levels. We need to bring biodiversity and ecosystem services 'accounting' into a similar framework to avoid perversions in the name of emissions mitigation, even though it is generally agreed that measuring and valuing biodiversity is substantially more difficult, expensive and uncertain.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2013.02.012.

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