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Land use efficiency: anticipating future demand for land-sector greenhouse gas emissions abatement and managing trade-offs with agriculture, water, and biodiversity

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Abstract

Competition for land is increasing, and policy needs to ensure the efficient supply of multiple ecosystem services from land systems. We modelled the spatially explicit potential future supply of ecosystem services in Australia's intensive agricultural land in response to carbon markets under four global outlooks from 2013 to 2050. We assessed the productive efficiency of greenhouse gas emissions abatement, agricultural production, water resources, and biodiversity services and compared these to production possibility frontiers (PPFs). While interacting commodity markets and carbon markets produced efficient outcomes for agricultural production and emissions abatement, more efficient outcomes were possible for water resources and biodiversity services due to weak price signals. However, when only two objectives were considered as per typical efficiency assessments, efficiency improvements involved significant unintended trade-offs for the other objectives and incurred substantial opportunity costs. Considering multiple objectives simultaneously enabled the identification of land use arrangements that were efficient over multiple ecosystem services. Efficient land use arrangements could be selected that meet society's preferences for ecosystem service provision from land by adjusting the metric used to combine multiple services. To effectively manage competition for land via land use efficiency, market incentives are needed that effectively price multiple ecosystem services.

Keywords: carbon sequestration, climate change, ecosystem services, food security, land use change, scenarios, trade-offs, water resources

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Introduction

Competition for land is increasing as demand for multiple land uses and ecosystem services grows (Power, 2010; Smith *et al.*, 2010; Harvey & Pilgrim, 2011). Emerging carbon markets (Newell *et al.*, 2014) are creating price signals for the conversion of agricultural land to other uses such as reforestation (Polglase *et al.*, 2013). This is occurring in parallel with other growing demands from land systems for urbanization and amenity, mining, and energy (Foley *et al.*, 2005), and other ecosystem services such as food production (Paterson & Bryan, 2012), water resources (Jackson *et al.*, 2005), and biodiversity (Lindenmayer *et al.*, 2012). While land use change may increase the supply of some ecosystem services, trade-offs may occur with other services (Bennett *et al.*, 2009; Gordon *et al.*, 2010; Raudsepp-Hearne *et al.*, 2010; Reyers *et al.*, 2013). Managing increasing competition for supply of these services requires efficient land allocation (Johnson *et al.*, 2014). Hence, carbon markets need to achieve emissions abatement with minimal trade-offs with other services (Smith *et al.*, 2013). More generally, market policy for achieving efficient provision of ecosystem services from land systems requires detailed, quantitative, integrated analyses of land use responses and resultant ecosystem service impacts at scale (Crossman & Bryan, 2009; George *et al.*, 2012; Reyers *et al.*, 2013; Rounsevell *et al.*, 2014).

The potential of market-based incentives for motivating reforestation for carbon sequestration has been recognized internationally (Strengers *et al.*, 2008; Golub *et al.*, 2009; Jackson & Baker, 2010; Torres *et al.*, 2010; Nijnik *et al.*, 2013). Extensive research has found reforestation to be competitive in Australia's agricultural land, even under modest carbon prices (Flugge & Schilizzi, 2005; Flugge & Abadi, 2006; Harper *et al.*, 2007; Hunt, 2008; Maraseni & Cockfield, 2011; Paul *et al.*, 2013a,b; Polglase *et al.*, 2013; Bryan *et al.*, 2014; Longmire *et al.*, 2015). An earlier review (Richards & Stokes, 2004) found economic potential to sequester up to 500 MtC yr^{-1} in the USA, and over 2000 MtC yr^{-1} globally,

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at carbon prices ranging from 10 to 150 \$ tC^{-1}. At the higher carbon prices (>120 AU \$ tCO_2^{-1}) necessary to limit global warming to <3 °C (Newth *et al.*, 2015), the potential for reforestation-based carbon sequestration is even more pronounced (Canadell & Raupach, 2008; Strengers *et al.*, 2008; Bryan *et al.*, 2014). However, carbon markets have a complex influence on land use and ecosystem services, potentially resulting in co-benefits and trade-offs for other ecosystem services (Bryan, 2013; Lin *et al.*, 2013; Bustamante *et al.*, 2014).

Reforestation typically involves a trade-off with food production through the displacement of agriculture (Nelson *et al.*, 2010; Ausseil *et al.*, 2013; Smith *et al.*, 2013; Lawler *et al.*, 2014). In an integrated assessment of multiple incentive interactions in South Australia's agricultural land, Bryan & Crossman (2013) estimated that with current agricultural commodity prices, a 30 \$ tCO_2^{-1} carbon price could reduce agricultural production by 225 \$M yr^{-1} through conversion of agricultural land use to reforestation. Paterson & Bryan (2012) found that efficient and targeted reforestation policy could realize one-third of the total carbon sequestration possible for a loss of just one-tenth of the agricultural production in Australia's Lower Murray region.

Water resource impacts from reforestation have also been demonstrated (Farley *et al.*, 2005; Jackson *et al.*, 2005; Chisholm, 2010). While reforestation may improve water quality (Townsend *et al.*, 2012; Martinuzzi *et al.*, 2014), it may also reduce catchment water yields as trees intercept and evapotranspire more water than grassland/cropland (Zhang *et al.*, 2001; van Dijk & Renzullo, 2011). Estimates from forest carbon plantations in New Zealand indicate water yield reductions of 30–50% in some catchments (Dymond *et al.*, 2012). Schrobback *et al.* (2011) found that a carbon price exceeding 100 \$ tCO_2^{-1} could motivate large-scale plantations in the high water-yielding south-eastern Murray–Darling Basin catchments.

The potential impact of carbon-motivated reforestation on biodiversity depends on policy design (Deal *et al.*, 2012; Robertson *et al.*, 2014; B.A. Bryan *et al.*, in review). Carbon markets alone are unlikely to supply biodiversity co-benefits (Thomas *et al.*, 2013; Bryan *et al.*, 2014; B.A. Bryan *et al.*, in review) and may have negative effects (Lindenmayer *et al.*, 2012; Bradshaw *et al.*, 2013). Monocultures are likely to economically outcompete biodiverse plantings due to their higher rates of carbon sequestration and/or lower costs (Kanowski & Catterall, 2010; Bryan *et al.*, 2014), but have limited benefit for biodiversity (Smith, 2009; Hall *et al.*, 2012). They also preclude future opportunities for conservation as long-lived monoculture plantations may be established in areas important for the restoration of threatened ecosystems, or that may become important under climate change (Crossman *et al.*, 2012; Summers *et al.*, 2012; T.D. Harwood *et al.*, unpublished data).

Thus, like many changes in land use and management, the reforestation of agricultural land can influence multiple ecosystem services. Many studies have quantified trade-offs across multiple ecosystem services (Howe *et al.*, 2014). Assessments of the impacts of changes in land use and management have been most commonly undertaken in resource accounting terms comparing the relative effects across multiple ecosystem services (Rodriguez *et al.*, 2006; Nelson *et al.*, 2009, 2010; Gordon *et al.*, 2010; Raudsepp-Hearne *et al.*, 2010; Bryan & Crossman, 2013; Butler *et al.*, 2013; Lawler *et al.*, 2014). However, this approach provides little information on the efficiency of land use configurations. In a significant advance, and although not framed in efficiency terms, Moilanen *et al.* (2011) used a conservation planning approach to minimize trade-offs between environment (i.e. biodiversity conservation, carbon storage) and development (i.e. agricultural production, urban development) needs via optimizing the spatial allocation of land use.

Production possibility frontiers (PPFs), also called *Pareto* or *efficiency* frontiers, can define productive efficiency across multiple outcomes and delineate the trade-off space (Smith *et al.*, 2012). Each point on the frontier represents the maximum production of one objective for a given level of another such that it is impossible to increase production of one without decreasing the other. Polasky *et al.* (2008) used PPFs to identify land use policies which maximized joint production of biodiversity and economic returns, and Nelson *et al.* (2008) used a similar approach to assess the efficiency of the joint production of carbon and biodiversity. While a few studies have assessed the productive efficiency of land use and management options across multiple objectives (Higgins *et al.*, 2008; White *et al.*, 2012; Lautenbach *et al.*, 2013; Kragt & Robertson, 2014), these are case studies undertaken in small areas. Integrated assessment of land use efficiency for supplying multiple ecosystem services has not been undertaken at the landscape-scale resolution required to capture spatial heterogeneity in these services over national/continental extents which are meaningful for supporting evidence-based policy under global change (Falloon & Betts, 2010).

We thus modelled the potential competition for land and the efficiency in supplying multiple ecosystem services in Australia's agricultural land under global change from 2013 to 2050. We considered competition between agriculture, carbon plantings (fast-growing *Eucalyptus* monocultures; CP), and environmental plantings (mix of local native tree species; EP) and the impact on emissions abatement, agricultural production, water resources, and biodiversity services. The impact of uncertainty in global drivers was assessed through four future scenarios (termed *global outlooks*). Sensitivity to key domestic uncertainties in the rate of agricultural productivity increase and in land use change adoption behaviour was also assessed. We calculated ecosystem service supply in response to a carbon price from 0 to 300 \$ tCO_2^{-1}. We calculated PPFs to identify efficient levels of emissions abatement paired with agricultural production, water resources, and biodiversity services, and used these to evaluate the efficiency of a carbon market under the global outlooks. We calculated the costs and benefits associated with increasing productive efficiency using PPFs based first on two objectives, then on all four objectives. We discuss the implications for achieving the efficient supply of multiple ecosystem services from land systems under global change, and for integrated land use, agriculture, emissions abatement, water, and biodiversity policy at multiple scales.

Materials and methods

This analysis used a simplified version of the Land Use Trade-Offs (LUTO) model – an integrated environmental-economic systems model of Australian land use futures (Bryan *et al.*, 2014; Connor *et al.*, 2015). The model, written in Python (van Rossum & the Python community, 2013), calculates the potential future impact of changes in global and domestic drivers on land use and ecosystem services at a 1.1-km grid cell resolution, on an annual time step. Economic calculations were undertaken in real terms in 2010 Australian dollars.

Study area

The study area is the 85.3 million ha intensive agricultural land of Australia (Fig. 1). Excluded are remnant native ecosystems, water bodies, urban areas, and other nonagricultural land

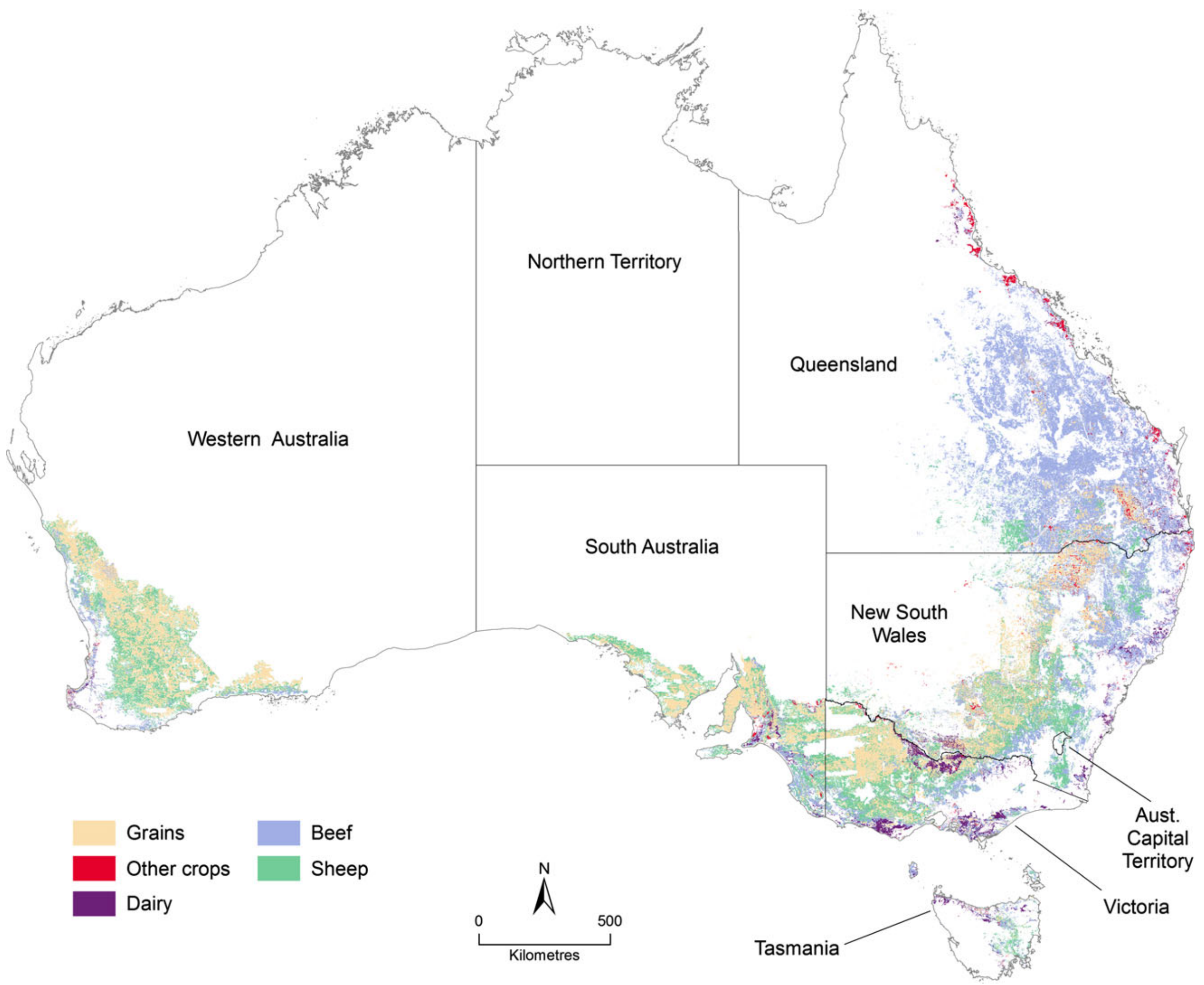

Fig. 1 Broad agricultural land use in the study area. Land use mapping is a snapshot for 2005/06.

uses. Agriculture is dominated by grains, sheep, beef, dairy, and irrigated crops (Marinoni *et al.*, 2012) with mixed-farming rotations of crops and livestock common. About 60% of annual agricultural production is exported (Australian Bureau of Statistics, 2012).

Global and domestic uncertainties

Four plausible global outlooks which cover uncertainties in key global drivers were specified as part of CSIRO's Australian National Outlook initiative (Hatfield-Dodds *et al.*, 2015) and mapped to representative concentration pathways (RCPs) (van Vuuren *et al.*, 2011). Global outlooks (Table 1) specified a range of emissions trajectories (van Vuuren *et al.*, 2011), climatic warming projections (Harman, 2013), and assumptions about the size of the global population and economy. Integrated assessment of the global outlooks (Newth *et al.*, 2015) provided trajectories for carbon price and oil price, and demand for crops and livestock from 2013 to 2050 which were used as inputs into land use modelling.

Sensitivity of the results to two influential domestic uncertainties identified through global sensitivity analysis (L. Gao, B.A. Bryan, M. Nolan & J.D. Connor, in review) was also assessed. These uncertainties included the rate of increase in agricultural productivity, and land use change adoption behaviour. Three annual increases in agricultural productivity rate from 2013 to 2050, resulting from improvements in agricultural management and technology, were considered – low (L, 0.0% p.a.), medium (M, 1.5% p.a.), and high (H, 3.0% p.a.). Three adoption rates were also assessed to capture the behavioural inertia common in empirical land change studies (Dilling & Failey, 2013; Bustamante *et al.*, 2014; Upton *et al.*, 2014). These were implemented as adoption hurdle rates of 1×, 2×, and 5×, which indicate how many times as profitable as agriculture reforestation needed to be for land use change to occur (e.g. under 2×, reforestation needed to be twice as profitable as agriculture).

Modelling ecosystem services

Carbon sequestration and GHG emissions. For carbon plantings CP and environmental plantings EP, we calculated 100-year carbon accumulation layers (tCO_2^{-e} ha^{-1}) by modifying the Polglase *et al.* (2008) 3PG2-modelled spatial layers of 20-year carbon accumulation. Regression was used to adjust these layers according to the potential impacts of climate change under each global outlook (Bryan *et al.*, 2014). We reduced sequestration rates by 20% as a buffer to capture the risk of modelling uncertainty, as well as natural and moral hazard. We estimated the risk- and climate-adjusted annual carbon sequestration $\mathbf{c}(y)^f_{t,o}$ for $f \in$ {CP, EP} (bold notation indicates spatial layer) of a t-year old forest stand planted in calendar year y via a von Bertalanffy–Chapman–Richards (vBCR) growth curve (Zhao-gang & Feng-ri, 2003) for use in economic calculations. Annual average climate and risk-adjusted carbon sequestration $\bar{\mathbf{c}}(y)^f_o$ was calculated over 100 years for use in estimating supply of emissions abatement.

For agriculture, we considered the total *cradle to farm gate* greenhouse gas (GHG) emissions $\mathbf{c}^{AG}$ (tCO_2^{-e} ha^{-1}) for the 23 major irrigated and dryland commodity classes produced in the study area (Marinoni *et al.*, 2012). Spatially explicit

Table 1 Description of the four global outlooks in 2050 modelled by integrated assessment (Hatfield-Dodds *et al.*, 2015; Newth *et al.*, 2015) including the inputs to land use modelling

Characteristics	Units	Value in 2010	Global outlook L1	M3	M2	H3
Representative concentration pathway	–	–	2.6	4.5	4.5	8.5
Global abatement effort	–	–	Very strong	Strong	Modest	No action
Coverage of abatement policy	–	–	All sources	All sources except livestock		No sources
Cumulative emissions	$GtCO_2$	1089	3 134	3 769	3 769	4 588
Atmospheric concentration in 2100	ppm CO_2	390	445 (declining)	650 (stable)	650 (stable)	1360 (rising)
Radiative forcing in 2100	Wm^{-2}	–	Peak at 3.0 then decline to 2.6	4.5	4.5	8.5
Climatic warming by 2100	°C	–	1.3–1.9	2.0–3.0	2.0–3.0	4.0–6.1
Global population	Billion people	6.9	8.1	10.6	9.3	10.6
Emissions per capita	tCO_2 cap^{-1} yr^{-1}	7.0	2.2	4.7	5.4	8.7
World GDP	US$ Trillion	61.0	161.6	197	179.1	197.8
World GDP per capita	US$ 2010 Thousands cap^{-1}	8.8	20.0	18.6	19.3	18.6
Carbon price	A$ tCO_2^{-1}	–	199.74	118.73	59.31	0
Grains demand	% change from 2007	–	75	118	11	61
Livestock demand	% change from 2007	–	147	112	22	61
Oil price	% change from 2007	–	42	44	45	43

agricultural GHG emissions were calculated by life cycle assessment combining agricultural commodity mapping, farm management information, and a life cycle inventory (J. Navarro, B.A. Bryan, O. Marinoni, S. Eady & A. Halog, in review). *Cradle to farm gate* agricultural emissions accounted for the production of inputs (e.g. chemicals, fodder, seed, fuel) and their transport to the farm, the operation of machinery on-farm up to and including harvest, but excluded machinery and infrastructure manufacturing processes.

Agricultural production. Agricultural production data were sourced from Australian Bureau of Statistics' agricultural census data for the census years 1996, 2001, and 2006. Average yields were calculated for each agricultural commodity and adjusted for climate change impacts under each global outlook by regressing climate and modelled yield data, and projecting based on estimates of climate change (Bryan *et al.*, 2014). Economic value, calculated as the sum of yield × price in 2010, was used as an integrated metric of agricultural production.

Water resources. The impact of reforestation on water resources $\mathbf{W}^f$ was modelled as the difference in annual water use (ML ha^{-1} yr^{-1}) between shallow-rooted and deep-rooted vegetation. We used the Australian Water Resources Assessment – Landscape model (AWRA-L) (van Dijk, 2010; van Dijk & Warren, 2010) to calculate water impacts. AWRA-L is a 0.05° grid-based model of groundwater and surface water dynamics combining climate, remote sensing, metering, and monitoring information with groundwater and river models. Water use by agricultural commodities $\mathbf{W}^{AG}$ at the local government area level was sourced from Australian Bureau of Statistics' agricultural water use data (Marinoni *et al.*, 2012). While we analysed spatially heterogeneous impacts of reforestation on water resources, we did not attempt to disaggregate these effects on surface water run-off, stream flow, or aquifer recharge effects.

Biodiversity services. *Biodiversity services* are defined here in a general sense to cover the diverse ways in which biodiversity may contribute to human well-being (i.e. regulator of ecosystem processes, ecosystem good, or ecosystem service) (Mace *et al.*, 2012). A relative spatial indicator of biodiversity services resulting from the ecological restoration of areas of cleared agricultural land via environmental plantings under climate change was calculated based on a metric of biodiversity priority incorporating concepts of complementarity, representativeness, area, and landscape connectivity was quantified using a generalized dissimilarity model (Ferrier *et al.*, 2007). The approach related compositional turnover of vascular plant species between 325 459 site pairs to 21 soil and climate variables. Compositional turnover was then predicted for each grid cell in response to six 2050 climate futures – combinations of two climate scenarios (RCP 4.5 and RCP 8.5) and three general circulation models (GCMs; Can ESM2, MPI ESM2, and MIROC5).

Biodiversity priorities were calculated for each of the six climate futures as a weighted combination of the contribution the future environment of each grid cell to the representativeness of vascular plant species, and their connectedness to remnant habitat through the species–area relationship. A single biodiversity priority layer was then calculated by integrating priority layers for the six climate futures using the limited degree of confidence approach (McInerney *et al.*, 2012). This layer was constant over time and robust to future climate change. Cells with a higher priority for restoration are those that increase species representation, area, and connectedness of plant communities given uncertainty in future climate (Bryan *et al.*, 2014; T.D. Harwood *et al.*, unpublished data). Biodiversity services **Bs** were calculated as the biodiversity priority score for each cell calculated as a percentage of the total biodiversity priority score summed across all cells.

Economic returns

Economic returns to land use were calculated in net present value (NPV) terms discounted at a commercial rate of 10% p.a. over a 100-year rolling time horizon using a profit function approach (Hajkowicz & Young, 2005; Bryan *et al.*, 2009, 2011b, 2014; Marinoni *et al.*, 2012). The NPV of agriculture was calculated for each commodity, for each calendar year y, as revenue (price × yield) less all fixed and variable costs of production. Under each global outlook, for each year, returns were influenced by the impact of climate change and productivity increases on yield, the impact of global food demand on commodity price, and the impact of global oil price (Table 1) on production costs.

Similarly, economic returns to carbon plantings and environmental plantings were calculated for each year y and global outlook o as revenue (carbon price × carbon sequestration) minus upfront and ongoing costs discounted to NPV terms over 100 years. We used the climate-adjusted and risk-adjusted annual increment $\mathbf{c}(y)^f_{t,o}$ (i.e. following the growth curve) to estimate carbon sequestration. Upfront costs included the cost of establishment which varied spatially based on biophysical characteristics (Summers *et al.*, 2015), and the cost of general security water entitlements which varied by catchment (Burns *et al.*, 2011) and in response to changes in water availability associated with climate change (Harman, 2013). Ongoing costs included annual maintenance and transaction costs of 120 \$ ha^{-1} yr^{-1} (Bryan *et al.*, 2014). Water costs varied over time with climate change-induced water scarcity, and other costs varied over time with changing oil price.

Potential land use change and supply of ecosystem services

In estimating supply of emissions abatement, agricultural production, water resources, and biodiversity services, potential land use change was identified under carbon prices p, for each year y, for the four global outlooks o, three productivity rates u, and three adoption hurdle rates h. Land use in each cell was allocated to carbon plantings $\mathbf{x}(y)^{CP}_{p,h,u,o}$, environmental plantings $\mathbf{x}(y)^{EP}_{p,h,u,o}$ and agriculture $\mathbf{x}(y)^{AG}_{p,h,u,o}$ based on maximum economic potential:

$$\mathbf{x}(y)^{\mathrm{CP}}_{p,h,u,o} = \begin{cases} 1 \text{ if } \mathbf{NPV}(y)^{\mathrm{CP}}_{p,o} = \max(\mathbf{NPV}(y)^{\mathrm{AG}}_{u,o} \\ \qquad \times h,\ \mathbf{NPV}(y)^{\mathrm{EP}}_{p,o},\ \mathbf{NPV}(y)^{\mathrm{CP}}_{p,o}) \\ 0 \text{ otherwise} \end{cases}$$
$$\mathbf{x}(y)^{\mathrm{EP}}_{p,h,u,o} = \begin{cases} 1 \text{ if } \mathbf{NPV}(y)^{\mathrm{EP}}_{p,o} = \max(\mathbf{NPV}(y)^{\mathrm{AG}}_{u,o} \\ \qquad \times h,\ \mathbf{NPV}(y)^{\mathrm{EP}}_{p,o},\ \mathbf{NPV}(y)^{\mathrm{CP}}_{p,o}) \\ 0 \text{ otherwise} \end{cases}$$
$$\mathbf{x}(y)^{\mathrm{AG}}_{p,h,u,o} = \begin{cases} 1 \text{ if } \mathbf{NPV}(y)^{\mathrm{AG}}_{p,o} = \max(\mathbf{NPV}(y)^{\mathrm{AG}}_{u,o} \\ \qquad \times h,\ \mathbf{NPV}(y)^{\mathrm{EP}}_{p,o},\ \mathbf{NPV}(y)^{\mathrm{CP}}_{p,o}) \\ 0 \text{ otherwise} \end{cases} \tag{1}$$

The sum of potential land use allocation in each cell must equal 1:

$$\mathbf{x}(y)^{\mathrm{CP}}_{p,h,u,o} + \mathbf{x}(y)^{\mathrm{EP}}_{p,h,u,o} + \mathbf{x}(y)^{\mathrm{AG}}_{p,h,u,o} = 1 \tag{2}$$

and in the rare case of a tie for most profitable land use, we assigned agriculture highest priority, then EP, with CP lowest. GHG emissions abatement $\mathrm{EA}(y)_{p,h,u,o}$ was calculated as the annual average carbon sequestered by CP or EP plus the GHG emissions abatement achieved through the cessation of agriculture summed over cells with economic potential for reforestation $f \in F$ {CP, EP} (operations over grid cells are represented with square brackets, i.e. sum[...]). This was done for each carbon price p, hurdle rate h, productivity rate u, and global outlook o:

$$\mathrm{EA}(y)_{p,h,u,o} = \mathrm{sum}\left[\sum_{f \in F} (\bar{\mathbf{c}}(y)^{f}_{o} + \mathbf{c}^{\mathrm{AG}}) \times \mathbf{a} \times \mathbf{x}(y)^{f}_{p,h,u,o}\right] \tag{3}$$

Agricultural production $\mathrm{AP}(y)_{p,h,u,o}$ was calculated as the value of annual production of agricultural commodities for each cell ($\mathbf{V}(y)^{\mathrm{AG}}_{u,o} \times \mathbf{a}$), summed over all cells with economic potential for agriculture:

$$\mathrm{AP}(y)_{p,h,u,o} = \mathrm{sum}\left[\boldsymbol{V}(y)^{\mathrm{AG}}_{u,o} \times \mathbf{a} \times \mathbf{x}(y)^{AG}_{p,h,u,o}\right] \tag{4}$$

Water resources $\mathrm{WR}(y)_{p,h,u,o}$ were calculated as the sum of the annual water use by irrigated agriculture $\mathbf{W}^{\mathrm{AG}}$ and by reforestation $\mathbf{W}^{f}$ via increased interception and evapotranspiration, and expressed as a negative number to provide a more intuitive indicator of water resource use:

$$\mathrm{WR}(y)_{p,h,u,o} = -\left(\mathrm{sum}\left[\mathbf{W}^{\mathrm{AG}} \times \mathbf{a} \times \mathbf{x}(y)^{\mathrm{AG}}_{p,h,u,o}\right] + \mathrm{sum}\left[\sum_{f \in F} \mathbf{W}^{f} \times \mathbf{a} \times \mathbf{x}(y)^{f}_{p,h,u,o}\right]\right) \tag{5}$$

Biodiversity services $\mathrm{BS}(y)_{p,h,u,o}$ were calculated as the sum of biodiversity services **Bs** achieved by EP minus that the conservation opportunity foregone by establishing CP as it offers little biodiversity benefit and precludes future ecological restoration:

$$\mathrm{BS}(y)_{p,h,u,o} = \mathrm{sum}\left[\mathbf{Bs} \times \mathbf{a} \times \mathbf{x}(y)^{\mathrm{EP}}_{p,h,u,o}\right] - \mathrm{sum}\left[\mathbf{Bs} \times \mathbf{a} \times \mathbf{x}(y)^{\mathrm{CP}}_{p,h,u,o}\right] \tag{6}$$

Supply of GHG emissions abatement, agricultural production, water resources, and biodiversity services was graphed against carbon prices from 0 to 300 \$ tCO_2^{-1}. Supply under the specific carbon price modelled for each global outlook (Table 1) was also plotted.

Dual-objective efficiency and trade-off analysis

We then used PPFs to explore the demand for emissions abatement, and the trade-offs with agricultural production, water resources, and biodiversity services. These illustrate the maximum levels of ecosystem services that can be jointly produced by altering the spatial arrangement of land use within the study area. PPFs were used to define dual-objective productive efficiency using an integer programming formulation specifying the three binary land use allocation terms $\mathbf{x}(y)^{\{\mathrm{AG,CP,EP}\}}_{p,h,u,o}$ in Eqns 3–6 as variables. First, we normalized the supply of each ecosystem service (denoted by ′) by linearly rescaling ecosystem service layers to the range [0,1] with 0 being the minimum and 1 the maximum possible supply aggregated over all grid cells. A weighted, normalized, dual-objective indicator of joint production of the two ecosystem services was then developed. PPFs were calculated for the year y = 2050 at the specific carbon price p of each global outlook o (Table 1) by arranging land use to maximize joint production subject to the constraint in Eqn 2. To create each frontier, we solved the land use allocation problem for all weights w in the set $W\{0.000, 0.005, 0.010, \ldots, 1.000\}$. Thus, to create the emissions abatement and agricultural production frontier, land use was allocated to maximize:

$$\mathrm{EA}'(y)_{p,h,u,o} \times w + \mathrm{AP}'(y)_{p,h,u,o} \times (1-w) \text{ for } \forall w \text{ in } W \tag{7}$$

to create the emissions abatement and water resources frontier, land use was allocated to maximize:

$$\mathrm{EA}'(y)_{p,h,u,o} \times w + \mathrm{WR}'(y)_{p,h,u,o} \times (1-w) \text{ for } \forall w \text{ in } W \tag{8}$$

and to create the emissions abatement and biodiversity services frontier, land use was allocated to maximize:

$$\mathrm{EA}'(y)_{p,h,u,o} \times w + \mathrm{BS}'(y)_{p,h,u,o} \times (1-w) \text{ for } \forall w \text{ in } W \tag{9}$$

Within the PPF-defined trade-off space, we plotted the supply of ecosystem services for 2050 under each global outlook o given its specific carbon price p (Table 1), for each hurdle rate h and agricultural productivity rate u. Locating the supply of ecosystem services resulting from the potential carbon price-driven land use responses in the context of the PPF enabled the quantification of the productive efficiency of land use allocations for ecosystem services under global outlooks. Inefficient land use responses to global outlook carbon market policy were further explored.The costs and benefits of increasing dual-objective productive efficiency of land use configurations were illustrated using carbon sequestration and water resources in M3. Two alternative land use arrangements (M3-Carbon and M3-Water) were identified which both lie on the PPF for emissions abatement and water resources. M3-Carbon is the land use arrangement which maximised emissions abatement for the same impact on water resources as the M3 land use response. M3-Water minimised the impact on water resources for the same emissions abatement as the M3 land

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use response. The land use responses were mapped, and the cost and impact of improving dual-objective productive efficiency was quantified across all objectives.

Multi-objective efficiency and trade-off analysis

To provide an integrated perspective on the productive efficiency of ecosystem services, we calculated multi-objective PPFs for each global outlook o, and assessed sensitivity to agricultural productivity rate u and adoption hurdle rate h. Multi-objective frontiers extended the dual-objective PPFs, delineating the efficient supply of all ecosystem services such that increasing one service on the frontier leads to an aggregate decline in the other three. In quantifying these frontiers, a set J was created containing all 1 373 701 combinations of four discrete weight parameters $w_{j,\mathrm{EA}}$, $w_{j,\mathrm{AP}}$, $w_{j,\mathrm{WR}}$, $w_{j,\mathrm{BS}} \in W$ which sum to 1 (i.e. $w_{j,\mathrm{EA}} + w_{j,\mathrm{AP}} + w_{j,\mathrm{WR}} + w_{j,\mathrm{BS}} = 1$). Land use responses and ecosystem service outcomes were calculated for all combinations of weighting parameters j in J to create each multi-objective PPF hypersurface by maximizing:

$$\mathrm{EA}'(y)_{p,h,u,o} \times w_{j,\mathrm{EA}} + \mathrm{AP}'(y)_{p,h,u,o} \times w_{j,\mathrm{AP}} + \mathrm{WR}'(y)_{p,h,u,o} \times w_{j,\mathrm{WR}} + \mathrm{BS}'(y)_{p,h,u,o} \times w_{j,\mathrm{BS}} \tag{10}$$

For the M3 outlook, we then identified three illustrative land use arrangements located on the multi-dimensional efficiency frontier: M3-Maximum, M3-Balanced, and M3-AgCentric. To do this, we first calculated the supply of each ecosystem service from the land use arrangement (Eqns 3–6) resulting from each weighting combination j in J as a proportion of the range of possible supply over all weight combinations j in J. Proportional supply calculation (denoted *) is illustrated for emissions abatement:

$$\mathrm{EA}^*(y)_{j,p,h,u,o} = \frac{\mathrm{EA}(y)_{j,p,h,u,o} - \min_{j \in J} \mathrm{EA}(y)_{j,p,h,u,o}}{\max_{j \in J} \mathrm{EA}(y)_{j,p,h,u,o} - \min_{j \in J} \mathrm{EA}(y)_{j,p,h,u,o}} \tag{11}$$

M3-Maximum was the land use arrangement that maximized aggregate proportional supply over all ecosystem services:

$$\max_{j \in J} \left(\mathrm{EA}^*(y)_{j,p,h,u,o} + \mathrm{AP}^*(y)_{j,p,h,u,o} + \mathrm{WR}^*(y)_{j,p,h,u,o} + \mathrm{BS}^*(y)_{j,p,h,u,o} \right) \tag{12}$$

M3-Balanced was that closest to the ideal point – an infeasible solution where all ecosystem services were maximized (i.e. equal 1) simultaneously. This was identified using a goal programming approach:

$$\min_{j \in J} \sqrt{(1 - \mathrm{EA}^*(y)_{j,p,h,u,o})^2 + (1 - \mathrm{AP}^*(y)_{j,p,h,u,o})^2 + (1 - \mathrm{WR}^*(y)_{j,p,h,u,o})^2 + (1 - \mathrm{BS}^*(y)_{j,p,h,u,o})^2} \tag{13}$$

M3-AgCentric was calculated in the same way as M3-Maximum (Eqn 12), but with agricultural production having twice the influence of the other three services.

These illustrative, efficient land use arrangements were plotted in the context of the multi-objective PPFs, and the costs and benefits of increasing the productive efficiency of land use arrangements were quantified, mapped, and compared.

Results

Outputs for the underpinning environmental and economic modelling are presented in the Supporting Information. These include spatial layers describing the carbon sequestration by CP (Fig. S1) and EP (Fig. S2), GHG emissions from agriculture (Fig. S3), value of agricultural production (Fig. S4), water use by reforestation (Fig. S5) and agriculture (Fig. S6), biodiversity priority score (Fig. S7), and economic returns to land use (Fig. S8). Below, we present illustrative results for the central settings for agricultural productivity rate increase (medium) and adoption hurdle rate (2×), denoted M 2× with full sensitivity analysis presented in the Supporting Information. We also focus the results on the calendar year 2050 to emphasize the difference between global outlooks.

Potential land use change

Little potential for land use change existed in the study area at carbon prices below 65 \$ tCO_2^{-1}. Beyond this price, potential for carbon plantings increased, particularly in the north-east of the study area which currently produces modest returns from beef cattle grazing. Significant potential existed for CP to outcompete both agriculture and EP due to the ability of fast-growing monocultures to sequester carbon in this area. Difference in carbon price drove very different land use change outcomes across global outlooks (Fig. 2). The area of potential land use change was sensitive to key assumptions with more change likely at lower rates of agricultural productivity improvement and lower adoption hurdle rate (Table S1).

Supply of ecosystem services

Ecosystem service supply followed the same general response to increasing carbon price as did potential land use change (Fig. 3). Emissions abatement increased with land conversion to reforestation. Agricultural production declined only slightly as it was the least productive

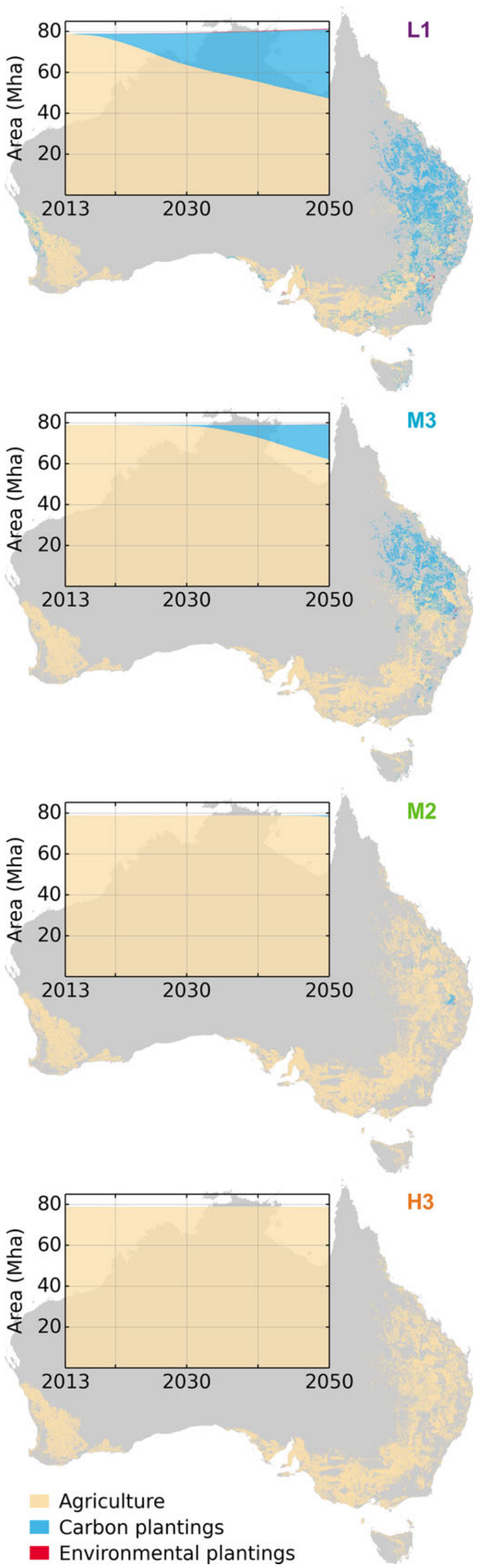

Fig. 2 Timing and location of potential land use change under the four global outlooks from 2013 to 2050.

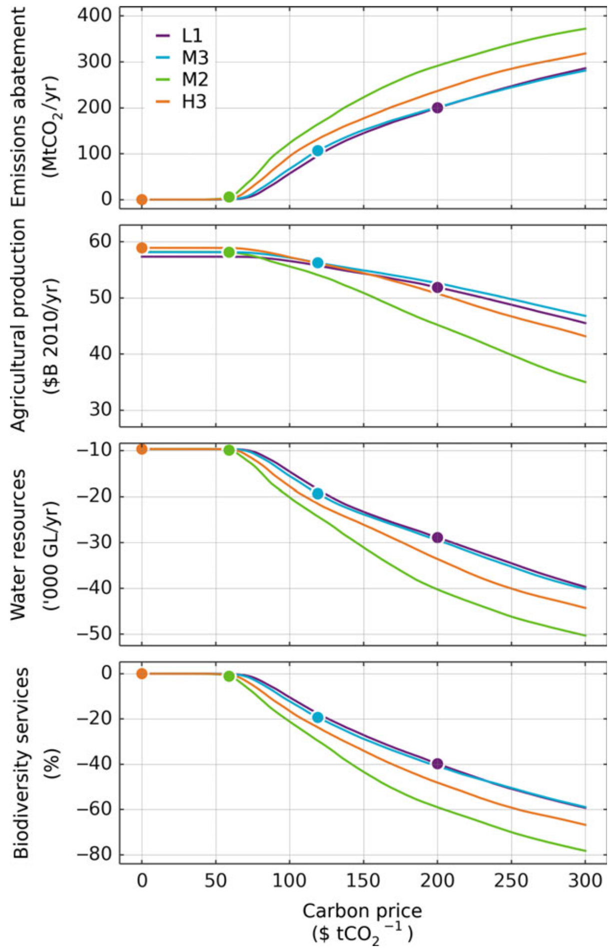

Fig. 3 Supply of emissions abatement, agricultural production, water resources, and biodiversity services under the four global outlooks in response to carbon price. The dots indicate supply under outlook-specific carbon prices (Table 1).

and thereby least profitable areas that tended to be converted first. Water resources declined with the increased interception and evapotranspiration from reforestation. Biodiversity services also declined with the foregone opportunity for restoring high-priority biodiversity areas associated with conversion to monoculture carbon plantings. Potential supply of emissions abatement and concomitant decrease in agricultural production, water resources, and biodiversity services in response to carbon price was greatest in M2 then H3 driven by the lower agricultural commodity prices in these two global outlooks (Fig. 3). Adoption hurdle rates and agricultural productivity rates had a substantial influence on ecosystem services supply (Fig. S9–S12).

In the central settings for productivity and adoption hurdle rates, H3 with a carbon price of 0 \$ tCO_2^{-1} and M2 with a 2050 carbon price of 59.31 \$ tCO_2^{-1}, which saw little potential for land use change and emissions abatement, agricultural production was greater than in other global outlooks, and there was little impact on water resources or biodiversity services. M3, with a carbon price of 118.73 \$ tCO_2^{-1}, saw emissions abatement of 105 $MtCO_2$ yr^{-1} (24% of maximum possible), agricultural production fell 4.5% to 56 \$B yr^{-1}, water resources decreased from −9692 GL yr^{-1} to −19 335 GL yr^{-1}, and biodiversity services decreased to −19%. L1, with a carbon price of 199.74 \$ tCO_2^{-1}, saw emissions abatement of 196 $MtCO_2^{-1}$ yr^{-1}, agricultural production fell by 12% to 52 \$B yr^{-1}, water resources decreased to −28 890 GL yr^{-1}, and biodiversity services decreased to −40% (Fig. 3). These results were sensitive to uncertainty in adoption hurdle rate and agricultural productivity assumptions (Table S2).

Productive efficiency and trade-offs between ecosystem services

The three dual-objective PPFs illustrating trade-offs between emissions abatement (EA) and agricultural production (AP), water resources (WR), and biodiversity services (BS) resulting from alternative land use arrangements in the study area (Fig. 4) were very similar across global outlooks. PPFs were convex in shape demonstrating that any change in a land use arrangement sitting on the frontier to increase one service came only at the expense of the other.

Land use at point A on all three PPFs (Fig. 4) was entirely reforested – dominated by high-sequestering CP, with pockets of EP where it sequestered more carbon. Along the EA–AP frontier (i.e. emissions abatement and agricultural production), land use graded to agriculture at the other end of the frontier (i.e. coinciding with the M2 and H3 land use responses). Along the EA–WR frontier to point B, land use graded to a combination of dryland agriculture with little impact on water resources and reforestation replacing irrigated agriculture in areas where trees used less water. At point B, water resources increased from −9692 to −1830 GL yr^{-1} but agricultural production was also reduced by 31% (41 \$B yr^{-1}). From point A along the EA–BS frontier, land use graded from nearly all CP with its associated foregone opportunity for biodiversity services, to all EP which maximized biodiversity services at point C. Here, EP sequestered carbon but generally less so than did CP which is reflected in the emissions abatement and biodiversity trade-off. In replacing GHG-emitting agriculture, reforestation achieved substantial emissions abatement but eliminated all agricultural production at all points along the EA–BS frontier.

Global outlooks, with their specific carbon prices taken into account (Table 1), varied in their productive efficiency (Fig. 4). Land use arrangements under all outlooks lied close to the EA–AP efficiency frontier. Productive efficiency was lower when compared against the EA–WR frontier, and lower still when compared against the EA–BS frontier, with global outlooks positioned well inside the latter two frontiers. Key sensitivities of agricultural productivity and adoption behaviour assumptions had a significant influence on the magnitude of ecosystem service impacts but did not substantially alter the productive efficiency (Fig. S13). This suggests significant potential exists for improving the productive efficiency of emissions abatement with regard to both water resources and biodiversity services through alternative land use allocation.

Increasing dual-objective productive efficiency

To illustrate the implications of increasing productive efficiency, consider the EA–WR trade-offs under M3 which had an emissions for abatement of 105 Mt CO_2 yr^{-1} and water resource use of −19 335 GL yr^{-1}. With the alternative land use arrangements of M3-Carbon, a 181% increase in emissions abatement (295 Mt

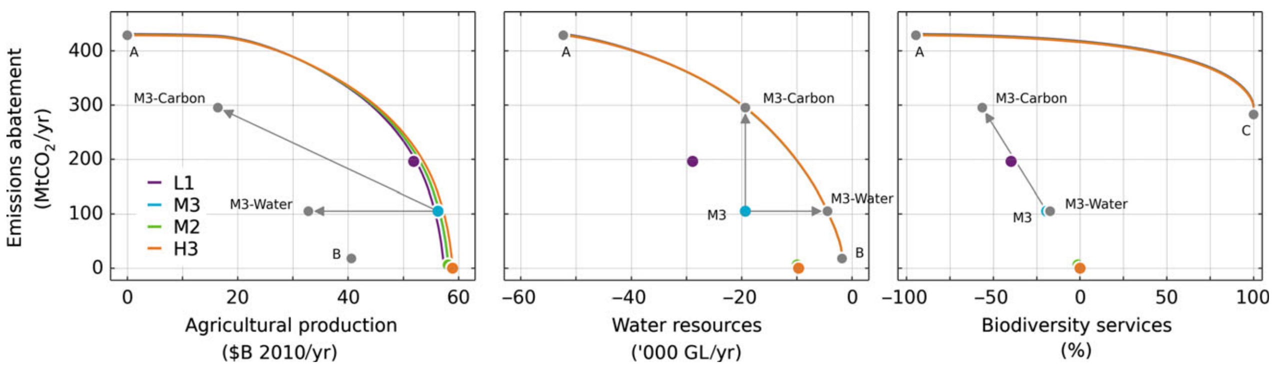

Fig. 4 Production possibility frontiers (from left to right EA–AP, EA–WR, and EA–BS) illustrating dual-objective trade-offs under the four global outlooks, the performance of land use responses under each outlook, and the impact of increasing carbon–water productive efficiency under M3 (M3-Carbon and M3-Water).

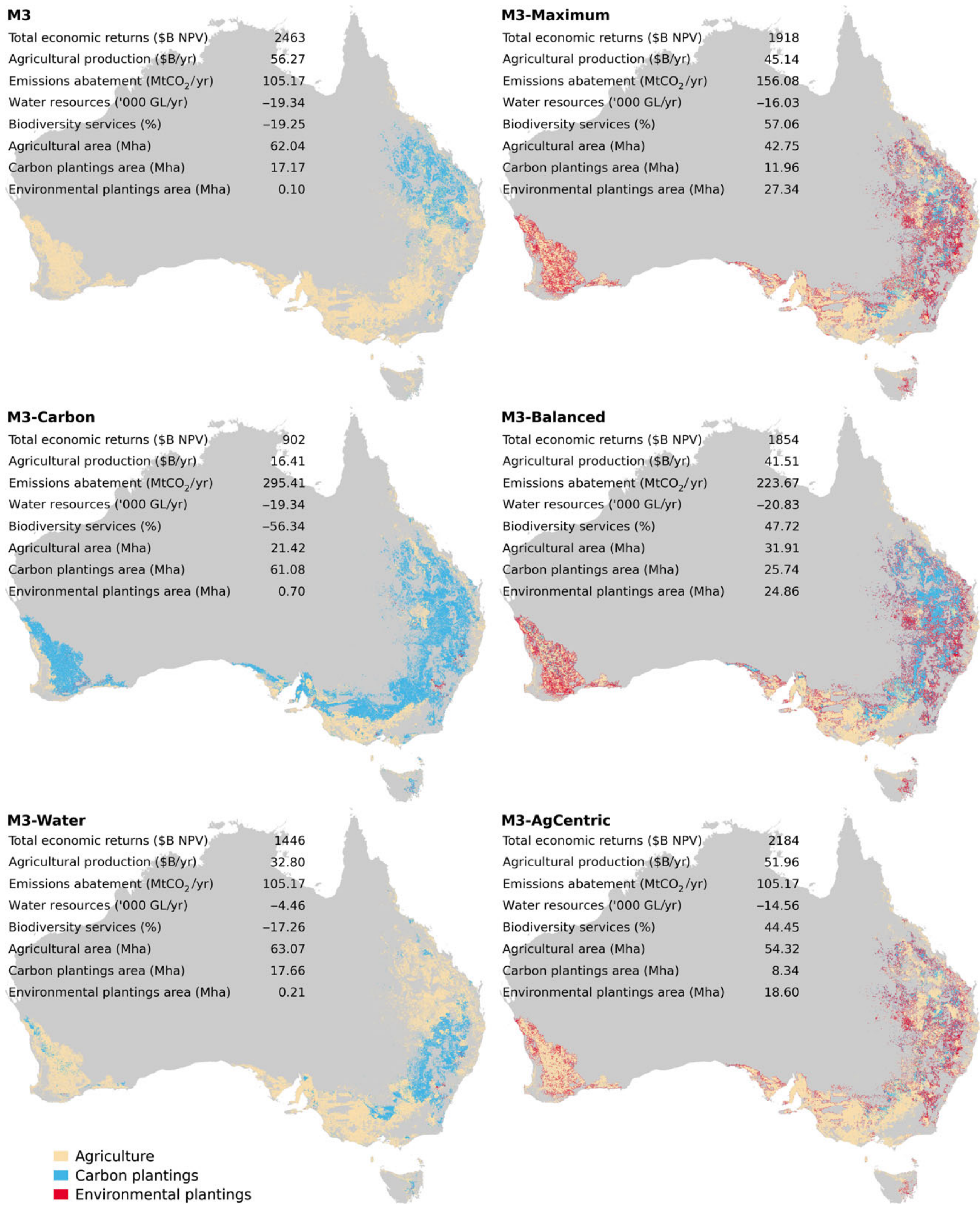

Fig. 5 Impact on land use, ecosystem services, and economic cost of increasing the productive efficiency of ecosystem services under the M3 global outlook. Note that the total area of land use may vary slightly as new land uses are assumed to occur throughout each grid cell whereas agriculture may only occur in a share of each cell.

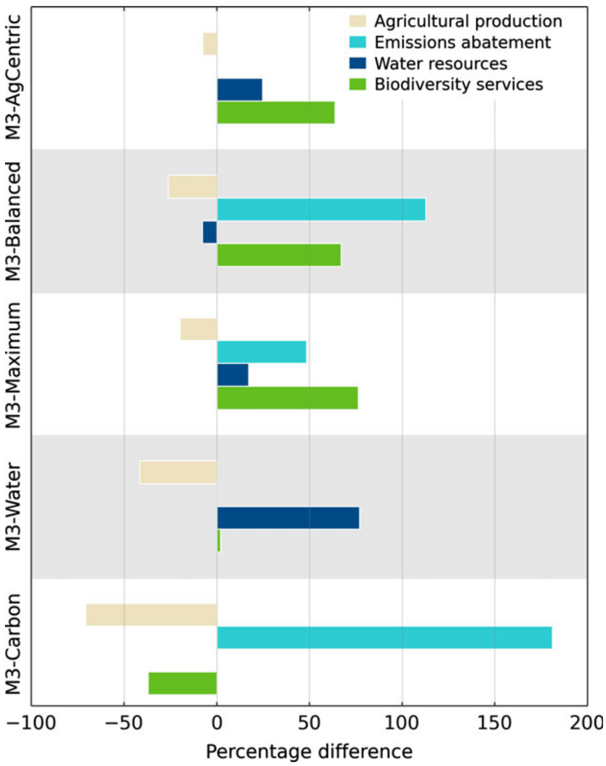

Fig. 6 Impact of land use arrangements on ecosystem service supply (% change from M3).

CO_2 yr^{-1}) was achieved for no change in water resource use. Conversely, M3-Water saw a 77% increase (−14 877 GL yr^{-1}) in water resources with no change in emissions abatement (Fig. 4).

While both of these land use pattern changes were productively efficient for emissions abatement and water resources, they substantially reduced the efficiency of other ecosystem services and incurred significant opportunity costs (Figs 5 and 6). M3-Carbon reduced agricultural production by 71% and biodiversity by 37% compared to M3, and had an opportunity cost of 1561 $B NPV. This involved a large expansion in CP throughout the drier parts of the study area typically used for dryland cropping and grazing (Fig. 5). Similarly, although M3-Water had a negligible impact on biodiversity compared to M3, it reduced agricultural production by 42%, and had an opportunity cost of 1017 $B NPV. This involved a large-scale shift in CP from the wetter south-east Queensland region to the drier areas of central New South Wales (Fig. 5).

Increasing multi-objective productive efficiency

Multi-dimensional PPFs indicate possibilities for the most efficient production of multiple ecosystem services. PPFs form convex hypersurfaces in multi-dimensional trade-off space (Fig. 7, Video S1–S6). Taking a two-dimensional cross-section, the PPFs were projected as an efficiency area (Fig. 8). The three illustrative efficient land use arrangements (M3-Maximum, M3-Balanced, and M3-AgCentric) improved the productive efficiency of ecosystem services under the M3 global outlook, but showed substantial differences in ecosystem service supply, cost, and land use arrangement. Compared to M3, M3-Maximum increased biodiversity services by 76%, emissions abatement by 48%, and water resources by 17%, with a negative impact on agricultural production (−20%) and had an opportunity cost of 545 $B NPV (Fig. 6). M3-Balanced increased emissions abatement by 113% and biodiversity services by 67%, but decreased water resources (−8%) and agricultural production (−26%), and had an opportunity cost of 609 $B NPV. M3-AgCentric had less of an impact on agricultural production (−8%), no appreciable impact on emissions abatement, and increased water resources by 25% and biodiversity services by 64%, with an opportunity cost of 279 $B NPV (Fig. 6). These costs, benefits, and the spatial arrangement of land use are summarized in Fig. 5.

Discussion

Ecosystem service supply and trade-offs

Demand for GHG emissions abatement is a key element of global futures if climate change is to be attenuated – a service the land sector is well-placed to deliver. Strong future demand is also expected for agricultural production (Tilman & Clark, 2014), water resources (Falkenmark, 2013; Hejazi *et al.*, 2014), and biodiversity conservation (Cardinale *et al.*, 2012), amongst other services from land systems. We quantified the efficiency of future supply of emissions abatement in response to a carbon price and the impacts on other ecosystem services in the intensive agricultural land of Australia under four global outlooks to the year 2050. Little potential for land use change was apparent at carbon prices below 65 $ tCO_2^{-1}. But beyond that price, carbon plantings began to outcompete other land uses particularly in the north-east of the study area. This is consistent with other findings in Australia (Flugge & Schilizzi, 2005; Flugge & Abadi, 2006; Harper *et al.*, 2007; Bryan *et al.*, 2008, 2010, 2011a, 2014; Hunt, 2008; Crossman *et al.*, 2011b; Maraseni & Cockfield, 2011; Paterson & Bryan, 2012; Paul *et al.*, 2013a,b; Polglase *et al.*, 2013; Renwick *et al.*, 2014; Longmire *et al.*, 2015) and internationally (Richards & Stokes, 2004; Lubowski *et al.*, 2006; Benitez *et al.*, 2007; Strengers *et al.*, 2008; Golub *et al.*, 2009; Jackson & Baker, 2010; Torres *et al.*, 2010; Nijnik *et al.*, 2013). With their high carbon prices, the L1 and, to a lesser extent, M3 global outlooks

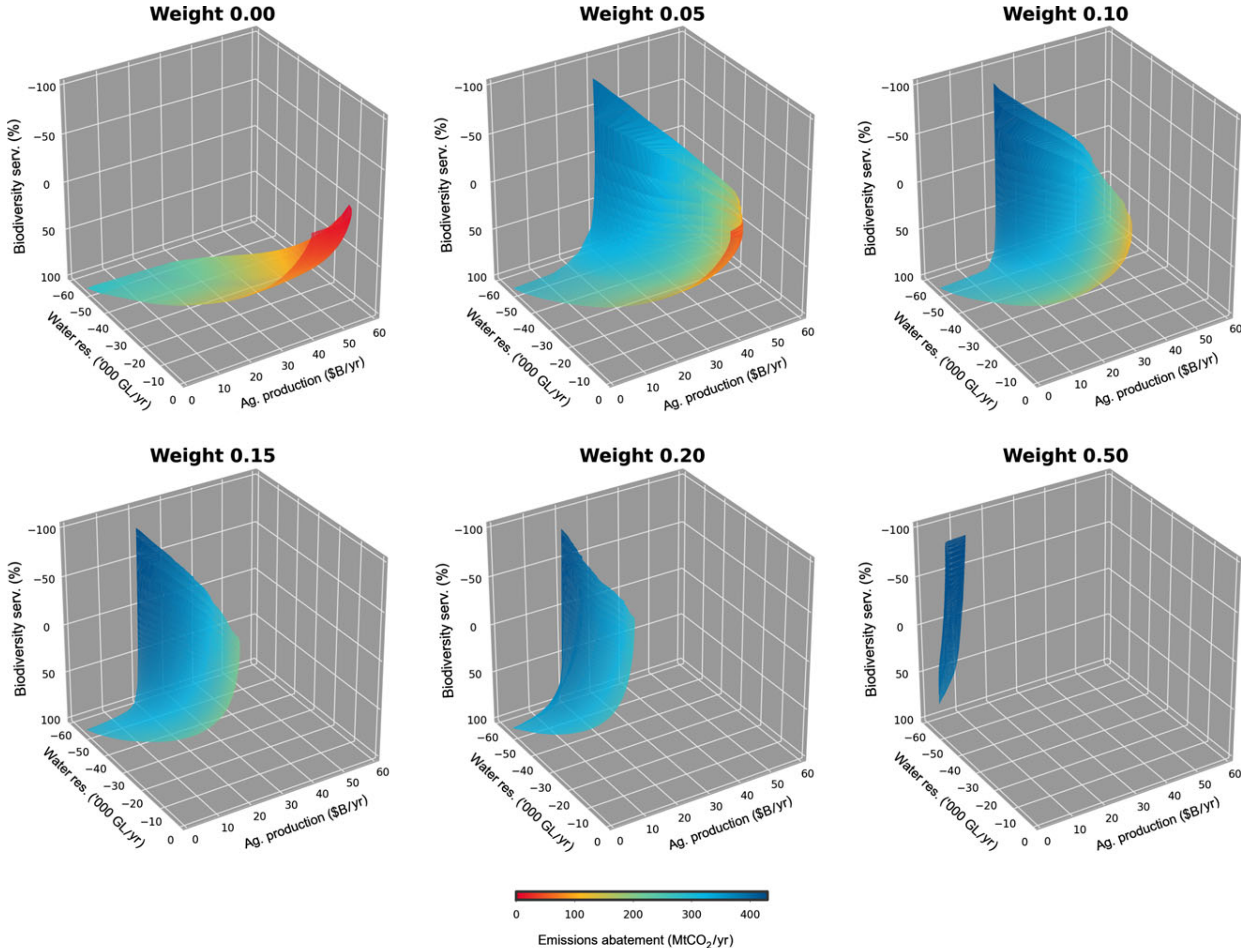

Fig. 7 Representation of the four-dimensional PPF under the M3 global outlook. Slices are presented at five weight levels on emissions abatement. See Video S1–S6 for an animation of these panels.

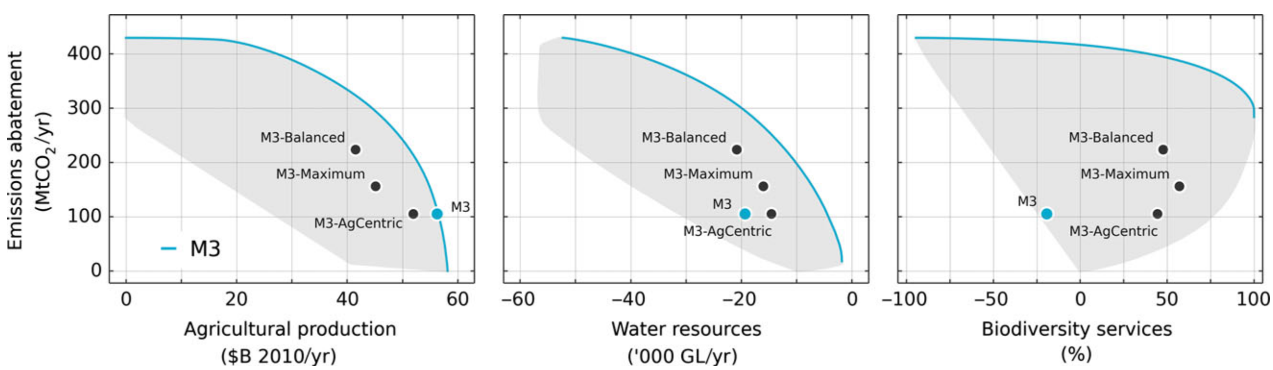

Fig. 8 Multi-dimensional production possibility frontiers projected onto dual-objective plots and the performance of the three illustrative land use arrangements under the M3 global outlook.

saw large potential for carbon plantings. This generated substantial emissions abatement, with minor consequences for agricultural production but more significant impacts on water supply and biodiversity services.

We analysed land use efficiency using three dual-objective production possibility frontiers, between emissions abatement, and agricultural production, water resources, and biodiversity services. Within these trade-off spaces, we located the supply of ecosystem services from potential land use responses under global outlooks which included a price on carbon. Future land use arrangements had a high productive efficiency for emissions abatement and agricultural production because both of these services had a market price and together, strongly influenced potential land use change. For emissions abatement and water resources, the productive efficiency was lower, with the supply of these services located well inside the efficiency frontier. In modelling potential land use change, water resource use was included as a cost via the compulsory upfront purchase of water entitlements for reforestation. However, this cost was not a strong driver of land use change (L. Gao, B.A. Bryan, M. Nolan & J.D. Connor, in review) and was not sufficient to achieve productive efficiency for water resources. Analogous effects have been found where charging irrigators a cost of water delivery, which is often far from the true scarcity value of water, often only results in small (suboptimal) reductions in water use (Iglesias & Blanco, 2008).

This effect was even more pronounced for biodiversity. Land use arrangements under global outlooks occurred well into the interior of the trade-off space indicating low productive efficiency. This was a result of biodiversity services having no market value under the global outlooks in this study and hence, lacking any influence on land use. This result gives effect to concerns of adverse biodiversity impacts of carbon markets (van Oosterzee *et al.*, 2010; Lindenmayer *et al.*, 2012; Bradshaw *et al.*, 2013). Our inclusion of the foregone conservation opportunity in areas of monoculture plantations presents a new approach and found decidedly greater impacts than have previous assessments (Crossman *et al.*, 2011b; Polglase *et al.*, 2013; Bryan *et al.*, 2014; Renwick *et al.*, 2014). Strongly negative impacts on biodiversity occurred from the preclusion of ecological restoration by long-lived but ecologically inferior monocultures in areas that are likely to become important for the representation of plant species and landscape connectivity given future geographic shifts in climate (Crossman *et al.*, 2012; Summers *et al.*, 2012; T.D. Harwood *et al.*, unpublished data).

Previous studies have established that land use allocations which are inefficient with respect to dual-objective trade-offs can be rearranged to increase the production of one ecosystem service without impacting on the other (Polasky *et al.*, 2008; Hurford *et al.*, 2014). We illustrated this by increasing the emissions abatement under M3, without impacting water resources (i.e. M3-Carbon), and vice versa (i.e. M3-Water). We found that while that aim was readily achieved, it had dramatic effects on the spatial arrangement of land use, decreased the supply and productive efficiency of other ecosystem services, and incurred very large costs. While we agree with Smith *et al.* (2012) that PPFs are a useful tool for quantifying trade-offs and evaluating policy, these results show that limiting these assessments to just two objectives can have serious unintended consequences. Multi-objective PPFs tempered this effect and were able to identify efficient land use arrangements considering all four ecosystem services. The M3-Maximum, M3-Balanced, and M3-AgCentric land use arrangements were all efficient and provided better compromises over multiple objectives than when only two objectives were considered. While we do not attempt to recommend the best option, our examples illustrate that choosing efficient outcomes implies very different land use arrangements, impacts for ecosystem services, and opportunity costs. It remains a decision for society about where on the efficiency frontier the preferred future lies – including the desirable level of benefits for ecosystem services and the costs it is prepared to bear to achieve them.

Policy implications

We have shown that the introduction of a carbon market in conjunction with established agricultural commodity markets under plausible global outlooks can lead to the efficient supply of emissions abatement and agricultural production. However, the supply of water resources and biodiversity services was inefficient. We modelled policy requiring new land use to account for water use through the purchase of entitlements – policy which currently operates in Australia and elsewhere (Connor & Kaczan, 2013). Following other findings (Bryan & Crossman, 2013; L. Gao, B.A. Bryan, M. Nolan & J.D. Connor, in review), this policy had some impact at recent historical prices, but was not a strong driver of land use. A fully functioning water market with a cap on resource use, where reforestation must compete with other water users (i.e. urban, industry, irrigated agriculture) for scarce water resources, is likely to be more efficient (Nordblom *et al.*, 2010). However, we did not model this policy as it is only operational in fully subscribed catchments in the study area such as parts of the Murray–Darling Basin. Similarly, the lack of a widespread, uniform market policy for biodiversity services greatly limited productive efficiency and drove strongly negative outcomes on the ground as high biodiversity priority areas were converted to monoculture plantations.

Alternative land use arrangements that increased productive efficiency over multiple objectives were possible, yet these had substantial costs in terms of foregone economic opportunity. A policy challenge then is how to meet these costs and increase productive efficiency over multiple objectives. The other challenge is in reducing the transactions costs associated with policy administration and targeting of multiple ecosystem services, and with the supply of ecosystem services by landholders. Two approaches are to either bundle or stack credits for ecosystem services and both involve the broadening of markets for ecosystem services (Raudsepp-Hearne *et al.*, 2010; Deal *et al.*, 2012; Robertson *et al.*, 2014). Credit bundling involves collating and selling all services from, in this case, a new land use such as environmental plantings, into a single market such as a payment for ecosystem services (Connor *et al.*, 2008; Crossman *et al.*, 2011a; B.A. Bryan *et al.*, in review). In this case, the benefits can be calculated using a multi-objective metric (e.g. Eqn 10). Our results (e.g. Figs 6 and 8) show how metric design can substantially influence the nature of the benefits to society. Stacking involves selling credits into individual markets for ecosystem services (Bryan & Crossman, 2013; Robertson *et al.*, 2014). Either bundled or stacked, careful design of markets and supporting institutional arrangements (e.g. multi-objective benefits metric, clear property rights, low transactions costs, etc.) is important for increasing productive efficiency across multiple ecosystem services (B.A. Bryan *et al.*, in review).

Innovation and limitations

Others have identified trade-offs between the supply of emissions abatement and food (West *et al.*, 2010; Paterson & Bryan, 2012; Smith *et al.*, 2013), biodiversity (Crossman *et al.*, 2011b; Hall *et al.*, 2012; Phelps *et al.*, 2012; Bryan *et al.*, 2014), and water (Jackson *et al.*, 2005; Chisholm, 2010; Schrobback *et al.*, 2011; Dymond *et al.*, 2012), and some have considered multiple services (Townsend *et al.*, 2012; Briner *et al.*, 2013; Bryan & Crossman, 2013; Petz *et al.*, 2014). But none have assessed productive efficiency. Studies that have assessed the influence of alternative land use arrangements on productive efficiency (Nelson *et al.*, 2008; Polasky *et al.*, 2008) have only considered dual objectives. Our results suggest that this may lead to substantial inefficiencies and costs for other ecosystem services. Only recently have studies begun to assess productive efficiency of land use and management options across multiple objectives (Higgins *et al.*, 2008; White *et al.*, 2012; Lautenbach *et al.*, 2013; Kragt & Robertson, 2014). Beyond quantifying multi-objective productive efficiency, we have made a significant advance in the assessment of policy for increasing it, and in doing this under global change. The calculation of land use efficiency under global change and policy scenarios is directly applicable to other regions and contexts to help manage emerging competition for land.

While we have paid special attention to uncertainty through the use of global outlooks, and sensitivity to the key variables of agricultural productivity increases and adoption behaviour (L. Gao, B.A. Bryan, M. Nolan & J.D. Connor, in review) – model uncertainty remains our chief limitation. In particular, uncertainty in the underlying spatial layers – including the spatial distribution of carbon sequestration rates, water interception, agricultural production, and economic returns – is a key determinant of the trade-offs and productive efficiency of land use in our study area. Despite the use of the state-of-the-art models and data, and our best efforts to understand (L. Gao, B.A. Bryan, M. Nolan & J.D. Connor, in review) and reduce (Summers *et al.*, 2015) uncertainty, some important model parameters, particularly socio-economic parameters (e.g. costs of agricultural production), remain highly uncertain. This may affect the precision of the results but is unlikely to change our conclusions. In addition, given that there are many more ecosystem services (TEEB, 2010) than the four we modelled, our analysis may be subject to the same criticism that we directed at two-dimensional analyses – that important impacts on unassessed services may be missed. In addition, we have not considered leakage or indirect land use change where a decrease in Australian agricultural production can lead to increased production elsewhere, with potential implications for emissions and other ecosystem services (Lambin & Meyfroidt, 2011). Future research could be broadened further to include the complex trade-offs with other services such as water quality, soil erosion mitigation, and aesthetic, recreation, and cultural values. While many dimensions could be assessed in theory, innovation in multi-dimensional visualization techniques will be required to effectively interpret the results.

We have presented an integrated assessment of the efficiency of land use in supplying emissions abatement, agricultural production, water resources, and biodiversity services in response to a carbon market in Australia's intensive agricultural land under four global outlooks. By 2050, substantial emissions abatement may be supplied under carbon prices exceeding 65 \$ tCO_2^{-1} seen in the L1 and M3 global outlooks with stronger action on climate. But this came at the cost of reduced agricultural production, water resources, and biodiversity opportunity. Productive efficiency of the carbon market in conjunction with existing markets for agricultural commodities was high for agricultural production and emissions abatement, but lower for water resources with its weak price signal, and lower still for biodiversity services with no price signal. Increasing productive efficiency of one service (e.g. water) led to substantial unintended consequences for other services and incurred high opportunity costs. Negative impacts were reduced by the consideration of productive efficiency over multiple ecosystem services, and this enabled the exploration of the trade-off space and the identification of land use arrangements that produced efficient outcomes across all four services. A national conversation about the relative levels of ecosystem services produced from Australia's agricultural land and the price we are willing to pay is required. Market policy capturing a broader array of ecosystem services can then be designed to achieve these desirable outcomes and efficiently manage future competing demands from land.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Figure S1. Total 100-year carbon sequestration by carbon plantings (derived from Polglase *et al*., 2008).
Figure S2. Total 100-year carbon sequestration by environmental plantings (derived from Polglase *et al*., 2008).
Figure S3. *Cradle to farm gate* greenhouse gas emissions from agriculture (J. Navarro, B.A. Bryan, O. Marinoni, S. Eady & A. Halog, in review).
Figure S4. Value of agricultural production in 2010 dollars (derived from Marinoni *et al*., 2012).
Figure S5. Water use (interception) by reforestation (derived from van Dijk & Renzullo, 2011).
Figure S6. Water use by irrigated agriculture (derived from Marinoni *et al*., 2012).
Figure S7. Biodiversity priority score as calculated by generalized dissimilarity modelling (derived from Bryan *et al*., 2014).
Figure S8. Economic returns to land use.
Figure S9. Emissions abatement supply curves for 2050 – sensitivity to agricultural productivity rate (L, M, H) and adoption hurdle rate (1×, 2×, 5×).
Figure S10. Agricultural production supply curves for 2050 – sensitivity to agricultural productivity rate (L, M, H) and adoption hurdle rate (1×, 2×, 5×).
Figure S11. Water resources supply curves for 2050 – sensitivity to agricultural productivity rate (L, M, H) and adoption hurdle rate (1×, 2×, 5×).
Figure S12. Biodiversity services supply curves for 2050 – sensitivity to agricultural productivity rate (L, M, H) and adoption hurdle rate (1×, 2×, 5×).
Figure S13. Dual-objective production possibility frontiers – sensitivity to agricultural productivity rate (L, M, H) and adoption hurdle rate (1×, 2×, 5×).
Table S1. Areas of potential land use change at 2050 – sensitivity to agricultural productivity rate (L, M, H) and adoption hurdle rate (1×, 2×, 5×).
Table S2. Ecosystem service supply under global outlooks – sensitivity to agricultural productivity rate (L, M, H) and adoption hurdle rate (1×, 2×, 5×).
Video S1. Multi-objective production possibility frontier, emissions abatement weight = 0.00.
Video S2. Multi-objective production possibility frontier, emissions abatement weight = 0.05.
Video S3. Multi-objective production possibility frontier, emissions abatement weight = 0.10.
Video S4. Multi-objective production possibility frontier, emissions abatement weight = 0.15.
Video S5. Multi-objective production possibility frontier, emissions abatement weight = 0.20.
Video S6. Multi-objective production possibility frontier, emissions abatement weight = 0.50.