Balancing production and environmental outcomes in Australia's tropical savanna under global change

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Key words: climate change, ecosystem services, food security, global change, carbon, rangelands, tropical savanna, northern Australia, grazing, beef, fire

Paper submitted to *Environmental Research Letters*.

Abstract

Livestock production is an integral part of the global food system and the livelihoods of local people, but it also raises issues of environmental sustainability due to issues such as greenhouse gas (GHG) emissions, biodiversity decline, land degradation, and water use. Further challenges to the social and environmental sustainability of extensive livestock systems may arise from changes in climate and the global economy (e.g., changing livestock demand and carbon prices). However, significant potential exists for both mitigating these impacts and adapting to change via altering stocking rates, managing fire, improving pastures, and supplementing cattle to reduce methane emissions. We developed an integrated, spatio-temporal modelling approach to assess the effectiveness of these different options for land management in Australia's tropical savanna under different global change scenarios. Performance was measured against a range of sustainability indicators, including environmental outcomes (GHG emissions, biodiversity, water intake, and land condition) and production (profit, beef production). We find that maintaining baseline stocking rates is not environmentally sustainable due to the accelerated land degradation exacerbated by a changing climate. Alternatively, planned early dry season burning resulted in substantial emissions reductions, and in our simulations became profitable under all global change scenarios that included a carbon price. Although there were no perfect win-wins, the balance between production and environmental outcomes could be improved by stocking at modelled carrying capacity and implementing fire management. This scenario was the most profitable (with a four-fold increase from the historic baseline), prevented land degradation, and reduced GHG emissions by 15%. As climate change is likely to reduce the potential for cattle production in Australia and elsewhere, the opportunity to diversify income streams may prove vital in a changing climate.

Introduction

Livestock production, particularly beef cattle, is an important source of human nutrition and employs over 1.3 billion people worldwide (Herrero *et al* 2009), but grazing has a range of environmental impacts including biodiversity decline (Alkemade *et al* 2013), land degradation, and contributions to climate change. Globally, livestock emits 14.5% of anthropogenic greenhouse gas emissions, with cattle comprising 62% of these emissions (Cheng *et al* 2022). Extensive grazing systems cover almost half of the world's tropical savanna ecosystems (9.48 M km2) (Asner et al 2004), and cattle in these ecosystems have a particularly high methane intensity, due to poor quality pasture and limited options for intensification (Tomkins and Charmley 2015). Future environmental and socio-economic changes are likely to affect livestock production and livelihoods, and exacerbate environmental pressures. However, changes in land management have the potential to reduce these impacts and contribute to several UN Sustainable

Development Goals (e.g. SDGs 'No Poverty', 'Zero Hunger', 'Climate Action', and 'Life on Land') as small changes over such large areas can amount to large aggregate impacts (Steinfeld et al 2006, Thornton 2010, Witt et al 2011, Holechek 2013). Therefore, management interventions are urgently required to promote the sustainability of rangeland systems under rapid but highly uncertain socio-economic and environmental change.

In extensive grazing systems, management interventions for improving sustainability can include conservative stocking rates, dietary supplementation, and fire management, amongst others (O'Reagain *et al* 2014, Walton *et al* 2014). Stocking at, or just below, the carrying capacity of the land not only has environmental benefits (i.e. climate change, biodiversity, and land condition), but can also be profitable for the landholder in the long run (O'Reagain *et al* 2011). This is because higher stocking rates can cause environmental degradation, especially during low rainfall years, resulting in animals in poor condition (O'Reagain and Scanlan 2013) and reduced capacity of the land to respond to rainfall. Modified pastures can increase the rate of liveweight gain (Hunt *et al* 2013), but can destroy ecosystems with profound impacts on native species (Rhodes *et al* 2021). Supplementation to reduce enteric methane production shows promise (Kinley *et al* 2020), but is likely to come with a high economic cost (Callaghan *et al* 2014) especially in extensive areas. Prescribed burning of tropical savanna ecosystems early in the dry season can also help to mitigate climate change by preventing more intense wildfire late in the dry season (Lipsett-Moore *et al* 2018) as well as providing biodiversity benefits. While these management actions appear promising, their future performance under global change has not been evaluated.

Climate change will challenge the future economic and environmental sustainability of rangeland systems and the effectiveness of management interventions. Increasing temperatures and changes in rainfall will have direct effects and also influence fire regimes, potentially leading to more intense and more frequent fires (Jones *et al* 2022, Boer *et al* 2016). Climate change impacts biodiversity and ecosystem services both directly (e.g., by shifting habitat suitability) and via interactions with other key drivers (Williams *et al* 2022). These changes will also have complex implications for cattle grazing, primarily via their effects on pasture production (McKeon *et al* 2009), which can influence productivity, profitability, and the potential for land degradation. .

Changing global economic conditions add further uncertainties surrounding the viability of management actions. Changes in the price for beef cattle and the cost of farm inputs alter the profitability of livestock production (Thornton 2010). Growing global demand for beef is likely to increase livestock sale prices and revenues, however, the costs of production are also likely to increase (Hatfield-Dodds *et al* 2015a). These changes may create opportunities for emissions reduction (if livestock production becomes less profitable), or alternatively intensify the trade-off (if livestock production increases to meet global demand). On the other hand, a higher carbon price is likely to make emissions abatement efforts more profitable, but has complex interactions with other economic and environmental drivers. As profitability is likely to be a key factor in the level of uptake of any management interventions, their impact on production and environmental outcomes will ultimately depend on the future trajectories of multiple socio-economic and environmental drivers.

This paper is a significant advance on previous studies in tropical savanna that have looked at the relationship between livestock production and GHG sequestration (e.g., McDonald et al (2023) and Castonguay et al (2023)), as we have considered the combined effects of global climate and economic change and multiple sustainability indicators. Such work is urgently needed as savannas are globally important for both biodiversity and people, but are being degraded faster than most other ecosystems (Williams *et al* 2022). In particular, Australia's tropical savanna has been recurrently proposed as a location to intensify agricultural production to supply Australia and Asia (Ash and Watson 2018), yet a strong focus on production risks the degradation of other ecosystem services and loss of globally unique species.

Here we developed an integrated spatio-temporal model of Australia's savanna rangelands to assess the impact of management actions on socio-economic and environmental sustainability under global change. The model links economic and biophysical sub-models to estimate each outcome for each year to 2050. We ran the model under four future global outlooks which combine different internally consistent assumptions for climate, global emissions abatement, population, livestock demand, and GDP (Table 1). We developed four broad management scenarios (Baseline, Conservation, Balanced (+), and Production), which included plausible combinations of stocking rate changes, supplementation, prescribed burning, and modified pastures (Table 2). We explored how these management scenarios performed in terms of key SDG indicators including livestock production, GHG emissions, livelihoods, water use, land degradation, and biodiversity under different scenarios of climate change and global economic drivers. We show that continuing historic grazing management is not environmentally sustainable, but combinations of management actions can improve the balance between production and environmental outcomes, even under changing climatic and economic conditions.

Methods

Study Area

Northern Australia has a largely semi-arid tropical climate and highly seasonal rainfall, with 85% falling between November and April (Watson *et al* 2021) (Figure 1c). Soils are typically old, weathered, and nutrient poor, producing relatively sparse pasture (O'Reagain and Scanlan 2013). These conditions support large tracts of savanna grasslands and open woodlands, covering ~2 million km^2, forming one of the largest areas of mostly intact ecosystems in the world (Woinarski *et al* 2007, Beyer *et al* 2020). Fires are frequent and often extensive, with many areas experiencing fires every 1 – 2 years on average. The region's remoteness has posed major challenges for biodiversity research, but species richness generally increases with rainfall (Mokany *et al* 2022) and on sandstone escarpments (Oliver *et al* 2017) and there is a steady rate of discovery of new species (Tingley *et al* 2019). Beef production from rainfed native pastures is the dominant agricultural land use in the region (Figure 1), occupying ~60% of the land area, and grazing enterprises tend to be large (a number of properties greater than 1 million ha with some aggregations even larger), with sparse cattle grazing unimproved native pastures. Grazing has been implicated in the widespread declines of many birds, mammals, and reptiles across northern Australia, through alterations of the vegetation composition, ground cover and grass seed availability (Kutt *et al* 2012, Neilly *et al* 2021). Given large land areas and low productivity, management strategies must be relatively low cost and easy to implement, which typically excludes many more intensive management systems (e.g., cell grazing). Landholders' ability to impose management solutions can be constrained by land tenure arrangements. With the exception of small areas of freehold in the south-east, most of the study area is pastoral leasehold land (the land is owned by the Crown) and certain conditions of the lease need to be met (such as grazing cattle). The study area includes three Australian jurisdictions (Western Australia, the Northern Territory, Queensland) and lease conditions differ in each jurisdiction. Climate change is likely to bring higher temperatures and potentially more variable rainfall, making sustainable land management in northern Australia's rangelands even more challenging (McKeon *et al* 2009).

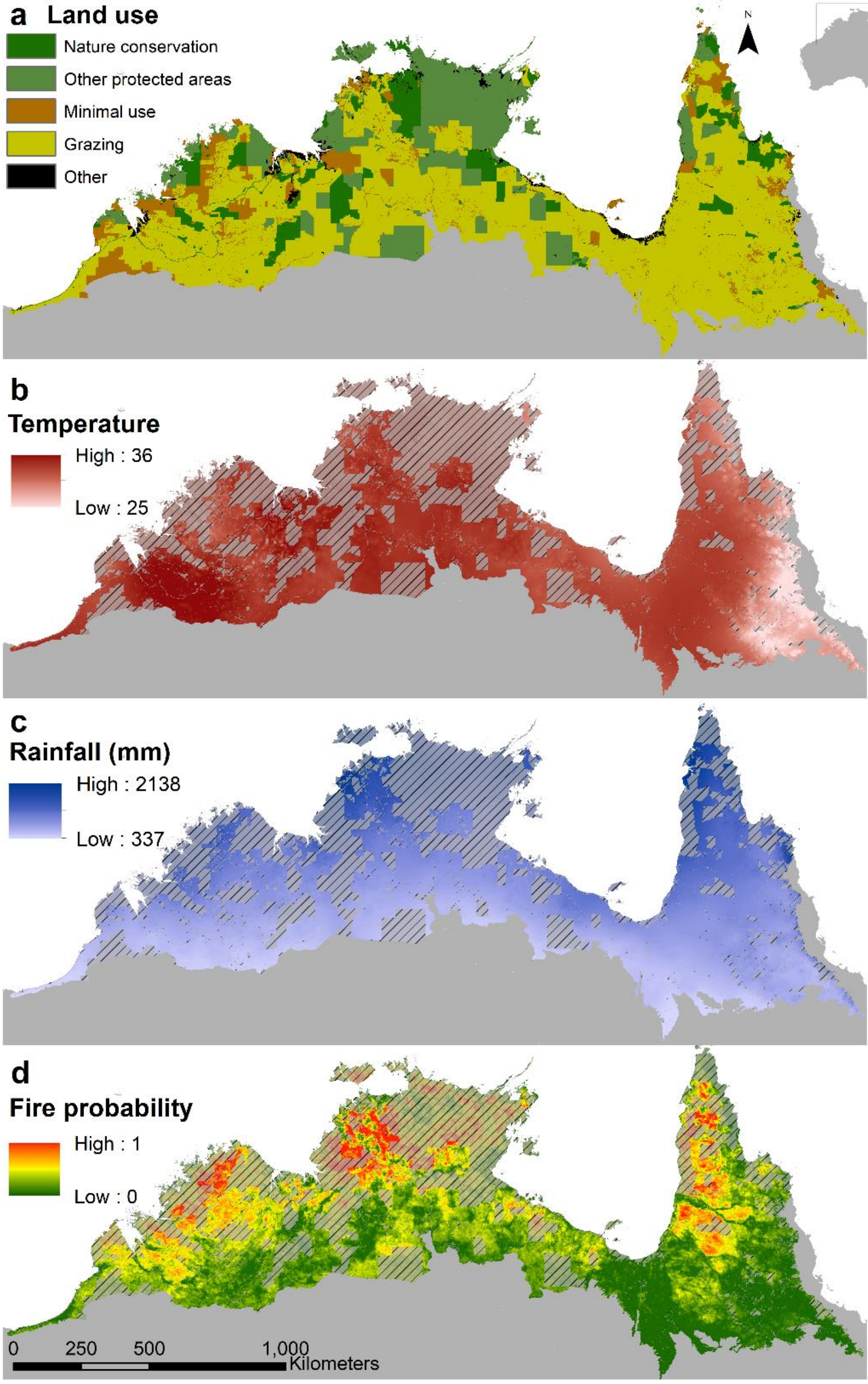

Figure 1 | The northern Australian study region. The area depicted was defined by the Interim Biogeographic Regionalisation for Australia (IBRA) (Australian Government 2012) at 0.01 decimal degrees (~1 km^2). Panel **(a)** shows the dominant land uses of the region from (ABARES 2016). “Other” includes water, forestry, and intensive uses; “minimal use” includes defence land (natural areas), stock routes, and residual native cover; and “other protected areas” includes Indigenous Protected Areas and managed resource protected areas (IUCN category VI). This study focuses on land managed for grazing (non-hatched areas in b–d). Panels **(b)** and **(c)** show the average daily maximum temperature (°C) and average annual rainfall (respectively) for grazing lands across 1987-2010 using data from Australian Government Bureau of Meteorology (Jeffrey *et al* 2001). Panel **(d)** shows the mean fire frequency (likelihood of vegetation burnt in a given year from 1988 – 2014) for grazing lands, as described in the Supplementary Information.

Integrated model

We developed an integrated, spatio-temporal model of land managed for cattle grazing across northern Australia's savannas (Fig 2). Simulation modelling offers a useful approach to assess the impact of global change, allowing the integration of economic and biophysical models. We used a combination of scenario analysis and sensitivity analysis to incorporate uncertainties in global change and local management strategies to 2050. In total we simulated 12 scenarios. This included 4 'global outlooks' from the Australian National Outlook (Hatfield-Dodds *et al* 2015a), which are linked to Representative Concentration Pathways (RCP) from the IPCC CMIP5 (van Vuuren *et al* 2011) and provide quantitative, internally consistent, projections of key economic parameters influencing livestock systems, including demand for livestock, and prices for oil and carbon (Bryan *et al* 2016) (Table 1). Within these outlooks projections of climate change parameters (e.g., temperature and rainfall change) were derived from 3 different GCMs to encompass the range of climate outcomes (Hatfield-Dodds *et al* 2015a, 2015b). Specifically, the GCM's used were: the Canadian Earth System Model (CanESM) (Chylek *et al* 2011); Max Planck Institute – Earth System Model – Low Resolution (MPI-ESM-LR) (Giorgetta *et al* 2013); and the Model for Interdisciplinary Research on Climate version 5 (MIROC5) (Watanabe *et al* 2010). To determine the impacts of management on sustainability outcomes under these different scenarios, the following sub-models were built and combined to form the integrated systems model (Fig. 2). Full details for each sub-model are provided in the Supplementary Information (SI).

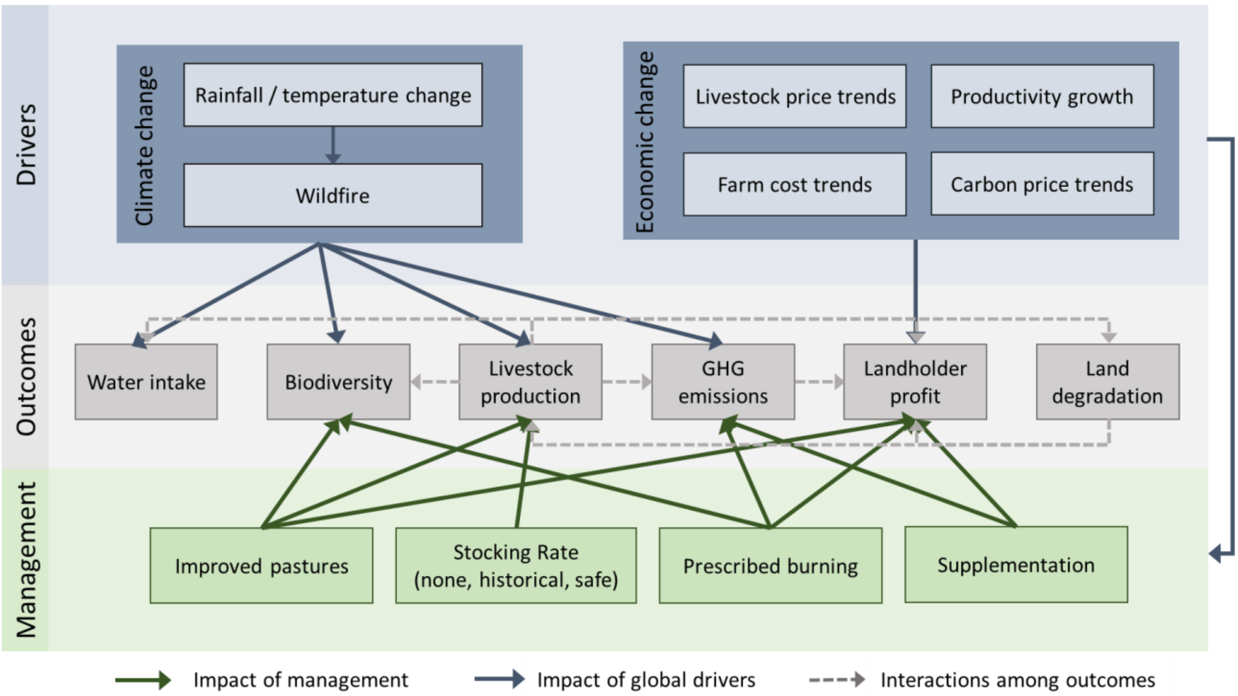

Figure 2 | A simplified conceptual model of the integrated assessment of sustainable management for grazing land under global change in northern Australia.

Table 1 | Key components of the global change scenarios used in this study (Hatfield-Dodds *et al* 2015a, Bryan *et al* 2014).

Parameter	**Units**	**Global Outlook**			
		L1	**M3**	**M2**	**H3**
Representative Concentration Pathway		2.6	4.5	4.5	8.5
Temperature increase in 2100	°C	1.3 – 1.9	2.0 – 3.0	2.0 – 3.0	4.0 – 6.1
Population	billion people	8.1	10.6	9.3	10.6
Abatement effort		Very strong	Strong	Moderate	None
Cumulative emissions (2007 – 2050)	Gt CO_2^{e}	1437	2091	2091	2823
Emissions per capita	$t\ CO_2^{-e}\ yr^{-1}$	2.2	4.7	5.4	8.7
Size of the global economy (GDP)	US$ trillion	161.6	197.0	179.1	197.8
Carbon price (in 2050)	$A\$\ tCO_2^{-1}$	199.74	118.73	59.31	0
Livestock price	% change 2007 – 2050	147	112	22	61
Oil price	% change 2007 – 2050	42	44	45	43

Table 2 | Different management scenarios, formed by combinations of stocking, dietary supplementation, prescribed burning, and pasture. "Safe" stocking rates refer to the number of livestock that could be supported by the amount of pasture growth in each year without adversely impacting land condition.

Management scenario	***Stock***	***Supplementation***	***Prescribed burning***	***Pasture***
Baseline	Historical	Urea	-	Native
Conservation	-	-	Yes	Native
Balanced	Safe	-	Yes	Native
Balanced +	Safe	Macroalgae	Yes	Native
Production	Safe	Urea	-	Modified

<u>Livestock production</u>. A regression model was developed to predict pasture growth, with annual rainfall and average maximum daily temperature as the explanatory variables, and was used to project pasture growth to 2050 under the 12 scenarios (SI section 2). We then calculated the number of cattle (adult equivalents, AE) that could be supported by the amount of simulated pasture growth in each year without adversely impacting land condition (i.e., the modelled *'safe' stocking rate* (Scanlan *et al* 1994)) by combining pasture growth, safe utilisation rates for different pasture types, and animal intake (SI section 3.1). *Modifying pastures* could increase the safe stocking level and revenue while also reducing the

methane produced per head (due to faster liveweight gain), so we simulated a management action of aerial sowing of legumes (e.g., stylo (Stylosanthes spp.) by helicopter or light aircraft (SI section 3.6). To simulate a continuation of the *baseline stocking level*, we also included a spatial approximation of historical stocking rates by updating livestock density maps from Navarro et al (2016) (SI section 3.2).

Land condition. In some cases, the stocking rate could result in more pasture being consumed than could recover in each year, causing land degradation. This was modelled using a threshold function with different forms (linear, concave, convex) where the level of stocking exceeds the carrying capacity of the pasture (SI section 5). In addition, we also accounted for the impacts of overgrazing on liveweight gain and profits using a (thresholded) linear function (Fig. S23).

Landholder profit. We calculated the profitability (measured as profit at full equity) of the baseline and simulated safe stocking rates from historic time series data for each Australian broadacre region in our study area (Navarro *et al* 2016, ABARES 2015). We then calculated the change in profit under each global outlook by varing livestock price trends, oil price trends, and future efficiency gains from technological innovation in line with scenario assumptions (Table 1).

GHG emissions. Quantifying emissions involved two sub-models: one accounted for fire risk reduction from prescribed burning, and the second accounted for methane emission reductions (from reduced stocking rates and/or supplementation).

- Future fire frequency and severity was modelled using stochastic simulations, determined by the instantaneous hazard for each year (calculated using recurrent-event regression analysis with shared frailty (Munda *et al* 2012) from historic burn scar data and future climatic conditions). Fuel load was increased where previously grazed land was destocked (and vice versa). GHG emissions from wildfire, and the emissions abated via prescribed burning, were calculated using methods adapted from the Australian Government GHG accounting methodology (DEE 2015) using plausible ranges for emission reductions for prescribed burning (Russell-Smith *et al* 2013, 2009b, Heckbert *et al* 2010).
- GHG emissions per head of cattle were calculated for each broadacre region (adjusting for herd structure) (Navarro *et al* 2016). Supplementation (with macroalgae) has the potential to reduce biogenic emissions from cattle without impacting livestock production) (Kinley *et al* 2016, 2020), but this comes with additional costs and uncertain outcomes in extensive grazing systems (Callaghan *et al* 2021). We therefore included a large range in potential methane reduction (and costs) from macroalgae supplementation via lick blocks.

Biodiversity under climate change was modelled using a combination of existing species distribution models for 609 vertebrates (43 amphibians, 286 birds, 93 mammals and 187 reptiles (Table S12)) (Graham *et al* 2019) in conjunction with taxa-specific dispersal kernels and expert elicitation of management impacts for each functional group (Alvarez-Romero *et al* 2021). This gives a 'biodiversity index' based on probability-adjusted species richness for each pixel in each year.

Water intake by cattle will increase with the higher temperatures that come with climate change. We modified the equation linking water intake and temperature for *Bos indicus* cattle (Watts *et al* 1994) to simulate water intake over the study region under climate change and for different stocking levels.

Sensitivity analysis

We conducted a global sensitivity analysis using elementary effects parameter sampling for 23 parameters (Table 3) (Gao and Bryan 2016). A triangular distribution for each parameter was produced based on the lower, mid, and upper values for each parameter (Table 3). In the cases where the input parameters were spatial, different values were used for each pixel. The elementary effects parameter sampling produced 250 parameter combinations (with 0-1 for each parameter) which were used to return the corresponding value from the triangular distribution. This analysis allowed us to determine the uncertainty for each management scenario and outcome, along with the model parameter sensitivity.

Table 3 | Parameters varied in the global sensitivity analysis. This does not include global outlooks or GCMs. Code corresponds to the X-axis in Fig. S25.

Parameter (code)	**Units**	**Lower**	**Mid**	**Upper**	**Detail**
Historical rainfall baseline (RainBase)	Percentile	10	50	90	Baseline for historical rainfall. Percentiles calculated over the range of years used to generate the historical climate (1987-2010).
Historical temperature baseline (TempBase)	Percentile	10	50	90	Baseline for historical temperature. Percentiles calculated over the range of years used to generate the historical climate (1987-2010).
Wildfire frequency and severity (Fire)	Spatial simulations	Lowest 20%	Mean	Highest 20%	Lower: mean of lowest 20% of fire simulations for each pixel. Mid: mean of all fire simulations for each pixel. Upper: mean of highest 20% of fire simulations for each pixel.
Safe pasture utilisation rate (Utilise)	Proportion (spatial)	Low	Mid	Upper	Safe pasture utilisation rates for each pasture community (from Table S7). The range varied per community.
Dry matter intake (IntakeAE)	kg day^{-1}	8	9	10	Cattle dry matter intake per AE per day.
Cattle increase from modified pastures (AEincrImprov)	Percentage (spatial)	Low	Mid	Upper	Increase in adult equivalents from modified pastures. The values (and range) varied by broadacre region (Table S9)

Land condition functional form (DegFunction)	z value	-2.5	0	2.5	Land condition function z value (0 gives a linear function) (Supplementary Information). Negative or positive values give convex and concave functional forms. All functions have a threshold at the safe utilisation rate (Table S7).
Prescribed burning emissions reductions (ERBurn)	Proportion	0.25	0.34	0.48	Emissions reduction from wildfire by undertaking prescribed burning. This was set at 0.34 for the main analysis (Russell-Smith *et al* 2013, 2009b) and varied between 0.25 (a conservative estimate of management effectiveness (Heckbert *et al* 2010)) and 0.48 (the upper potential of management (Russell-Smith *et al* 2009a)).
Change in fuel load (FuelChange)	Percentage	0.077	0.11	0.143	The percent (0.11%) increase in biomass each year following stock removal, or decrease if grazing ungrazed land. Upper and lower ± 30%
Macroalgae supplementation cost (SeaweedCost)	\$ per Adult Equivalent (AE) year^{-1}	62.05	93.08	124.1	The additional cost of using macroalgae lick blocks instead of urea. Low, mid and upper = 1, 1.5, and 2 times cost of molasses nitrate supplementation respectively.
Macroalgae supplementation emissions reduction (SeaweedGHG)	Percent reduction per AE	0	18.14	36.28	The GHG emissions reduction (per animal) of using macroalgae lick blocks instead of urea. Informed by Roque et al (2021) and Callaghan et al. (2021).
Cattle revenue (AERevenue)	\$ per AE per year	-SD	Mean	+SD	Baseline revenue per AE (without pasture improvement). Used the mean and standard deviation of time series farm survey data (1997-2013) for each broadacre region (Navarro *et al* 2016) (Table S10).
Cattle costs (AECost)	\$ per AE per year	-SD	Mean	+SD	Baseline costs per AE (without pasture improvement) calculated as per cattle revenue.
Cattle GHG emissions (AECO2e)	Mg CO_2e per AE per year	-SD	Mean	+SD	Biogenic GHG emissions per AE (without pasture improvement), using the mean and standard deviation for the historic baseline (Navarro *et al* 2016). Modified according to the total head and herd structure per broadacre region (Table S10).
Gross margin increase from modified pastures (ImpAERev)	% gross margin increase	Lower	Mid	Upper	Increase in gross margin per AE from modified pastures. The main value and range varied by broadacre region (Table S9).
GHG emissions reductions from modified pastures (ImpAECO2e)	% decrease in CO2e per AE	Lower	Mid	Upper	The reduction in biogenic GHG emissions per AE from modified pastures. The main value and range varied by broadacre region (Table S9).
Modified pasture cost (ImpAEcost)	\$ per km^2	150	270	720	Cost per km^2 for modified pastures. The main value and range varied by broadacre region (Table S9).
Prescribed burning cost (BurnCost)	\$ per km^2	32.795	46.85	60.905	Cost per km^2 for prescribed burning. Upper and lower = ± 30%.
TFP increase (TFP)	TFP increase per year	0%	1%	2%	Future annual increases in total factor productivity (TFP).
Fire impact on biodiversity (FireThreat)	Percentile /best guess	5th	Best	95th	'Best guess', 5th and 95th percentiles from the expert elicitation of fire impact on biodiversity.
Grazing impact on biodiversity (Grazthreat)	Percentile /best guess	5th	Best	95th	'Best guess', 5th and 95th percentiles from the expert elicitation for grazing impact on biodiversity.
Modified pastures impact on biodiversity (ShrubThreat)	Percentile /best guess	5th	Best	95th	'Best guess', 5th and 95th percentiles from the expert elicitation for introduced species impact on biodiversity.
Overgrazing impact (LWGImpact)	x	0.85	1	1.15	Overgrazing impact x value (see supplementary information for function). This would lessen (lower) or increase (upper) the impact of overgrazing on liveweight gain and profit.

Results

Continuing with the historical level of grazing, in the absence of any emissions abatement actions (baseline) performs poorly across all outcomes by 2050 (Fig. 3). When historical stocking rates were left unchanged (baseline), climate change accelerated land degradation, which ultimately tempered profits from the increasing livestock prices that occurred under all global outlooks (Fig. 4, Table 1). Further, GHG emissions continued to rise to 9.3 million Mg CO_2e yr^{-1} in 2050 (M3, MPI, unless otherwise stated), varying from 8.66 to 9.67 million Mg CO_2e yr^{-1} over the different GCM's and outlooks. The total water intake of cattle increased by 21.96 ML day^{-1} in 2050 (ranging from 9.83 ML day^{-1} (L1, MR5) to 27.83 ML day^{-1} (H3, MPI)) (Fig. 4), which represented a moderate increase (13%, Table 4). These results are clear that maintaining the historical rate and pattern of grazing pressure is not environmentally sustainable.

Removing cattle and managing the land through prescribed burning ("Conservation" management scenario) delivered the best outcomes for the environment of all the potential management scenarios (Fig. 3). GHG emissions were reduced to 2.69 (2.23 to 2.93) million Mg CO_2e yr^{-1} in 2050 (Fig. 4), which were solely comprised of GHG emissions from fire (Fig. S25). Additionally, there was no land degradation nor water intake from cattle, and biodiversity outcomes were improved (Figs. 3 and 4). This came at the expense of beef production outcomes. Although the only profit to the landholder was via carbon payments, this delivered robust profits, and became more profitable than the "Production" scenario in global outlooks L1 and M2 (Figs. 4 and S25). In contrast, in H3 (the global outlook without a carbon price) the landholder faced a loss, which suggests a conflict between environmental and economic objectives (Figs. 5 and 6a).

Our "Balanced" scenario evaluated a range of management options to achieve a balance between competing production and environmental outcomes. This scenario set stocking rates in accordance with simulated pasture growth and therefore eliminated land degradation but reduced food production by 18% relative to the historical stocking level (Table 4). This scenario reduced GHG emissions to 6.84 (6.80-6.92) million Mg CO_2e yr^{-1} (Fig. 4), was the most profitable (except in H1), and had the second-best outcome for biodiversity (though substantially lower than the "Conservation" scenario) (Fig. 3). The "Balanced +" scenario, which included the additional emissions abatement action of dietary supplementation, reduced GHG emissions even further (to 5.68 (5.62-5.77) million Mg CO_2e yr^{-1}), but supplementation on its own never became profitable, even with a high carbon price (Fig. S25).

Integrating exotic legumes into native pastures, evaluated in the "Production" scenario, maintained a high level of food production (-4% relative to the historical stocking level) and profit (the most profitable management without a carbon price, H3), and did not cause land degradation by pasture over-use (Fig 4). Here, the GHG emissions per animal were lower than the baseline (due to faster liveweight gain) which led to lower overall emissions. However, the absence of additional abatement actions (such as prescribed burning or supplementation) meant overall emissions were still high (8.37 million Mg CO_2e yr^{-1}, ranging from 8.27-8.52 million over GCMs and outlooks). Unfortunately, the introduction of exotic plants can be damaging to species in northern Australia, which also gives this management scenario the worst biodiversity outcomes (Figs. 3 and 4).

All outcomes and management scenarios showed substantial variation across northern Australia to 2050 (Figure 5 and 6). Cattle production was generally higher in the east (in the state of Queensland), and particularly the south-east, due to better conditions for grazing (e.g., less extreme temperatures). However, the decline in livestock production brought about by climate change were also more intense in this area (Figure 5). Species richness was generally higher in the East, and climate change brought increases in richness in the south, due to a slightly wetter (on average) climate (Fig. 5, column 4). Without management, GHG emissions are likely to increase in the north of the study area, although much of this can be abated with prescribed burning in the early dry season (which is a component of the Conservation, Balanced, and Balanced + management scenarios) (Figure 5, column 3). These spatial patterns were similar under the different GCMs and global outlooks (Figures S26-S36). Aside from the spatial patterns, there was also considerable uncertainty across all scenarios and objectives from variations in key parameters (Table 3), but general trends were still identifiable (Fig. 6). The parameters that contributed the most to this variation was the frequency and severity of fire (for GHG emissions and biodiversity), the safe pasture utilisation rate (for beef production) and future increases in technological innovation (for profit) (Fig. S25).

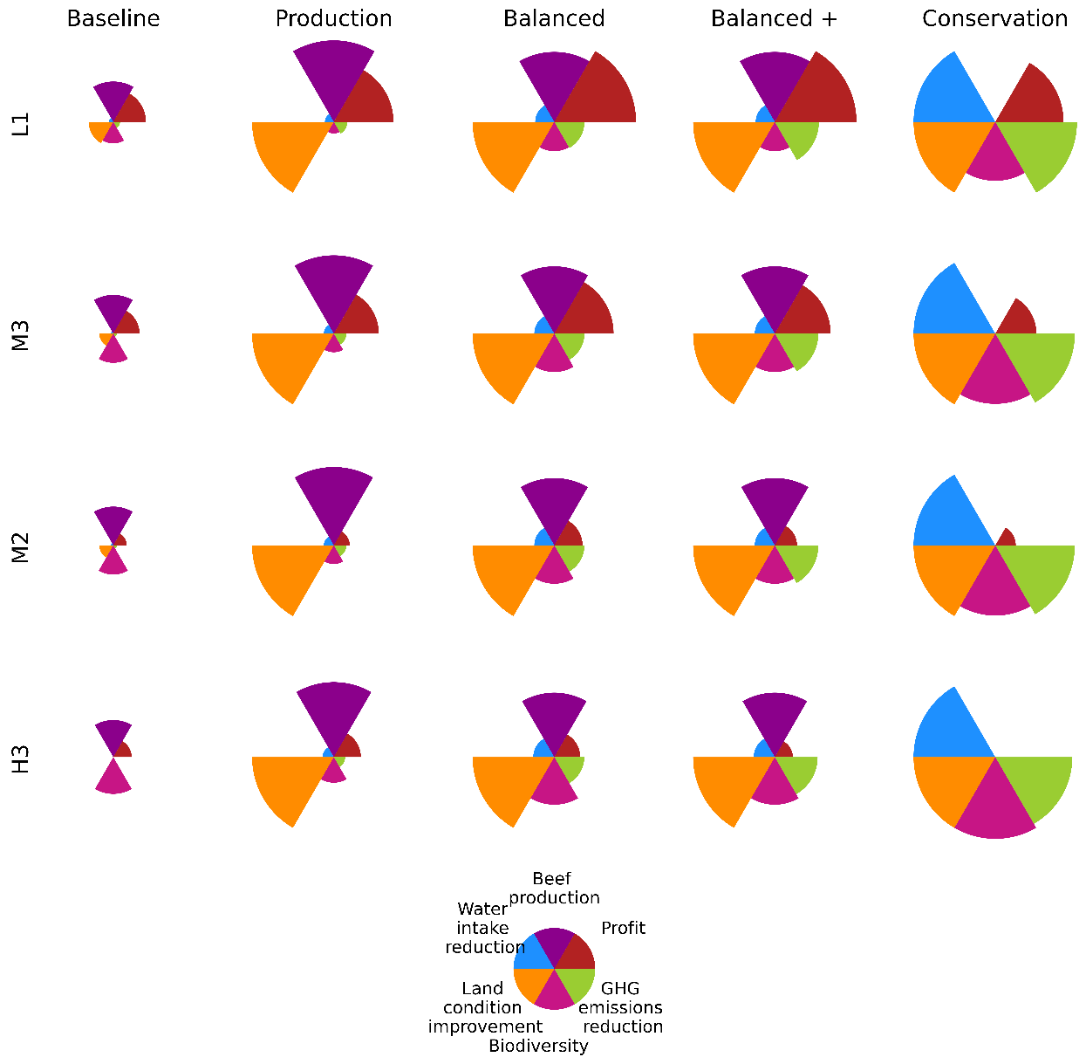

Figure 3 | Sustainability of different future management scenarios for northern Australia in 2050 under different global outlooks from L1 (strong global emissions abatement) to H3 (global business as usual), based on the means across the three GCMs used. Each outcome (beef production, landholder profit, GHG emissions reduction, biodiversity, land condition improvement and water intake reduction) is range-normalised on a scale of 0-1 (0 at the centre, 1 on the edge). Therefore, 0 refers to the minimum value across all scenarios (rather than the complete absence of that outcome).

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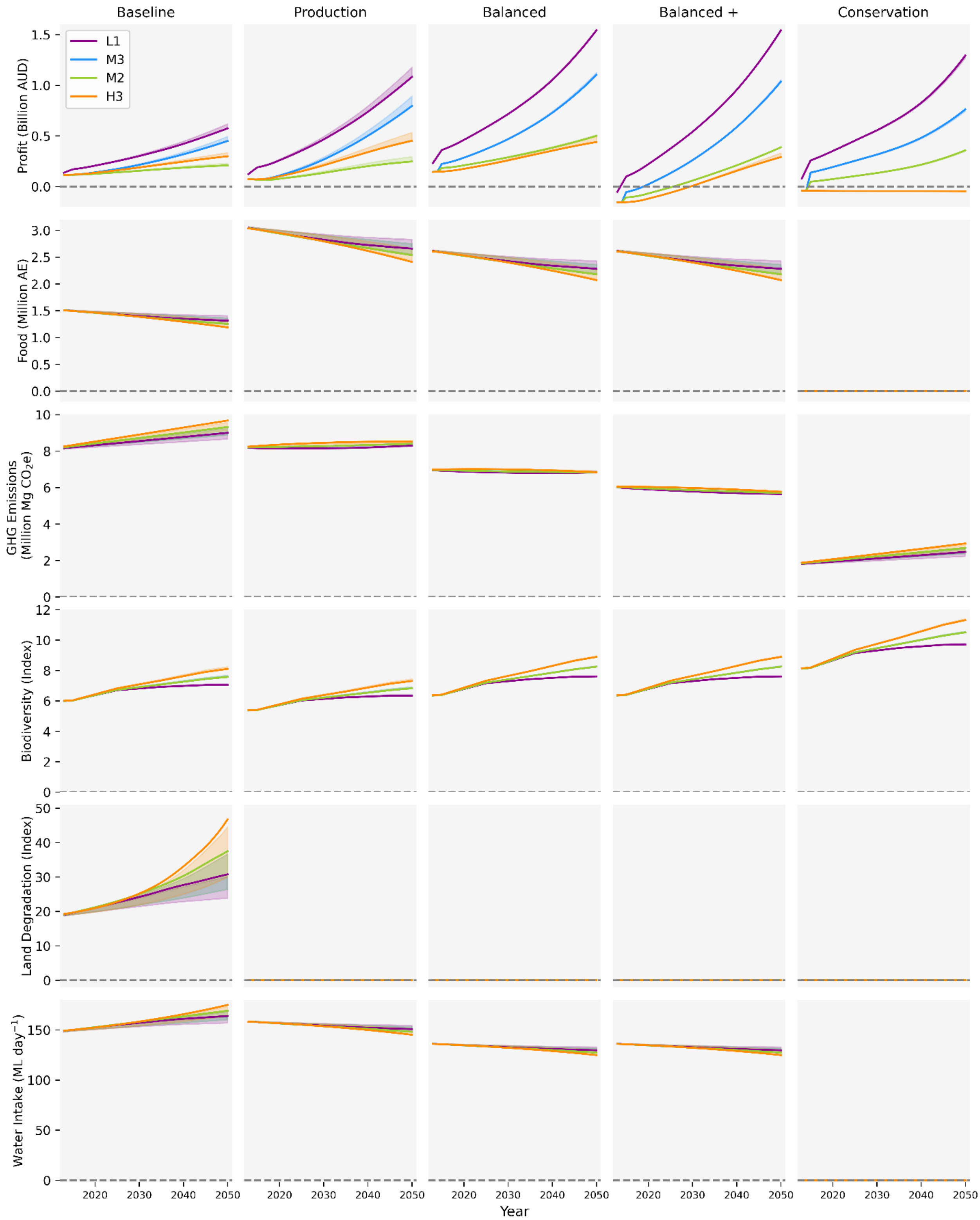

341 **Figure 4 |** Change over time for each outcome under the different future management scenarios for
342 northern Australia. Solid line shows GCM MPI, with the variation from GCMs CE2 and MR5 as shading.

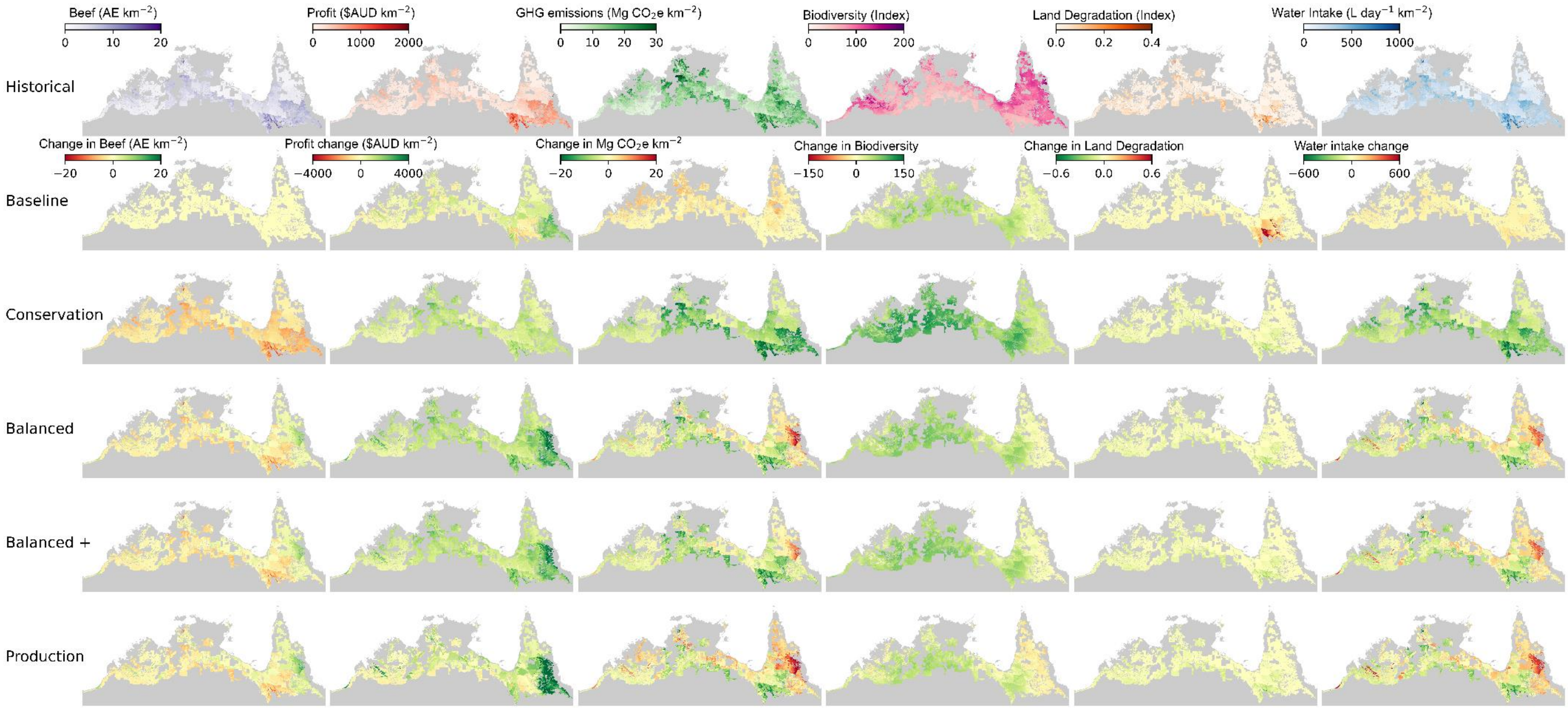

Figure 5 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050. Results are depicted for GCM MPI and global outlook M3 (spatial outcomes for the other GCMs and global outlooks are given in the Supplementary information). “Historical” represents stocking rates and climatic conditions representative of the period from 1987-2010. The remaining rows show the change from historical conditions to 2050 for each outcome under each management scenario.

Table 4 | Percentage change in outcomes from historical conditions. Results are shown for the mean across GCMs for global outlook M3 in 2050. The values in parenthesis show the variation across all global outlooks and GCMs. If there are no values in parenthesis there was no variation. Shading represents changes in the sustainability indicators as improvements (green) or deterioration (blue).

Management Scenario	Profit	Beef production	GHG emissions	Biodiversity index	Land degradation	Water Intake
Baseline	129% (3-204)	-53% (-56- -48)	14% (7.9-20.6)	38% (27-49)	84% (31-157)	13% (7-19)
Production	308% (21-478)	-4% (-12-4)	4% (3-6)	24% (14-34)	-100%	2% (-1-5)
Balanced	445% (116-663)	-18% (-24- -11)	-15% (-15- -14)	49% (37-61)	-100%	-12% (-15- -10)
Balanced +	412% (43-663)	-18% (-24- -11)	-29% (-30- -28)	49% (37-62)	-100%	-12% (-15- -10)
Conservation	271% (-123-535)	-100%	-68% (-72- -63)	90% (75-106)	-100%	-100%

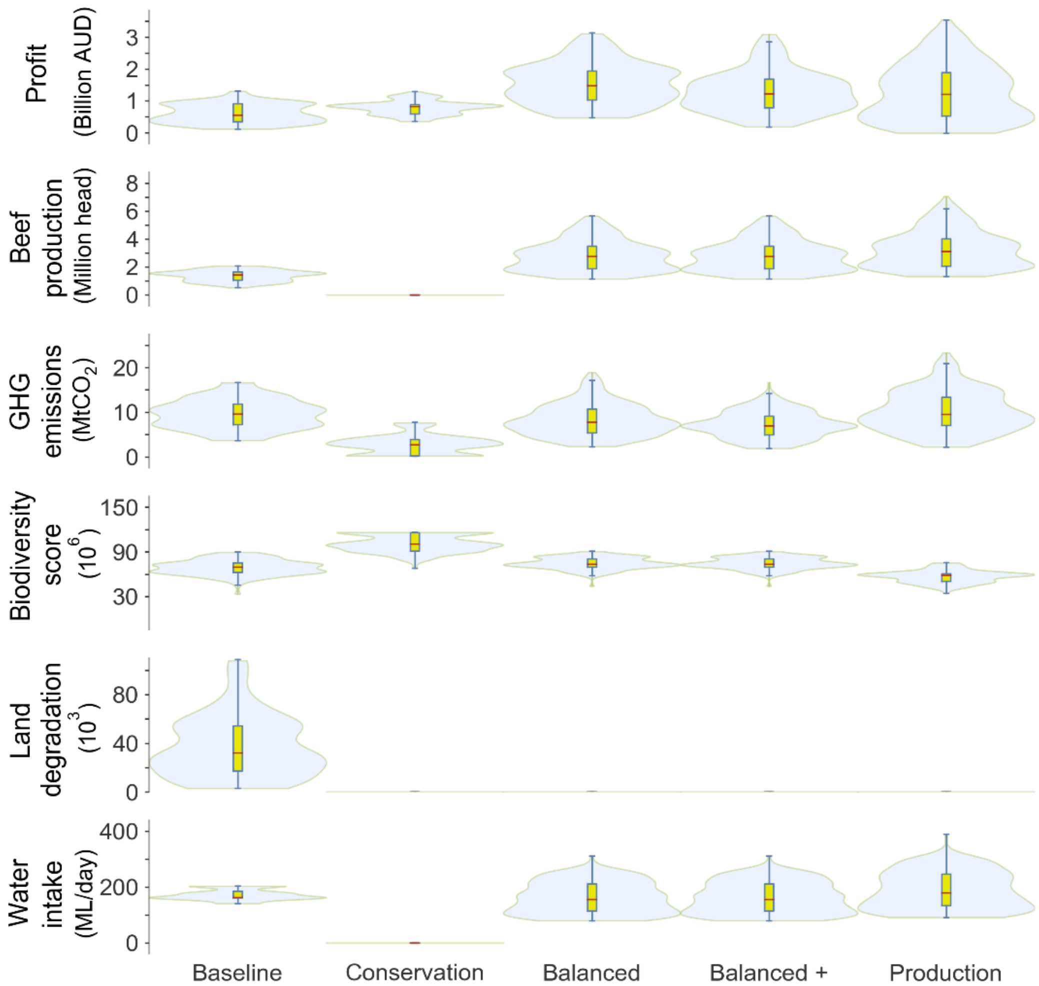

Figure 6 | The variation in outcomes for each management scenario based on a global sensitivity analysis of all 23 parameters in Table 4. All outcomes are for global outlook M3, GCM MPI, and year 2050.

Discussion

Cumulative impacts on sustainability indicators

By integrating the cumulative impacts of climate change, external economic drivers, and management actions on a range of sustainability indicators, we showed that the future of rangelands in Australia's savannas has the potential to balance production and environmental outcomes (Fig. 3). In the "Balanced" management scenario, combining prescribed burning with stocking at the carrying capacity of pastures prevents land degradation, reduces GHG emissions by 15%, supports higher species richness (increases the biodiversity index by 49%), and more than doubles baseline profits (compared to the baseline in M3, Table 4). In fact, this was the most profitable management scenario across all global outlooks that included a carbon price (L1, M3, M2). However, this scenario still represents a significant compromise, as compared the "Conservation" scenario, the biodiversity index was reduced by 22% and emissions were 166% higher (Fig.4). Overall, our findings are in line with other studies that have found significant emissions abatement potential from managed fire across the region (Adams and Setterfield 2013, Heckbert *et al* 2012), and these emissions reductions (and profits) could be further increased if the maximum (rather than average) potential for emissions reduction is achieved (Russell-Smith *et al* 2009a).

However, we found that climate change will likely reduce the capacity of northern Australia to support livestock, with the number of cattle that could be safely stocked declining over time and especially under more severe projections of climate change. This finding is supported by other studies, with a review by McKeon et al. (2009) finding that safe stocking rates were strongly dependent on climate. Yet, profits increased under all scenarios due to rising livestock and carbon prices (Table 1), with strong global emissions abatement (L1) delivering the highest profits (Fig. 4). Additional climatic factors not included here may reduce the modelled safe stocking rates and profitability. This includes, extreme events such as droughts and floods (Murray-Tortarolo and Jaramillo 2019, Harrison *et al* 2016) and elevated atmospheric CO_2 which may lead to woody thickening and reduced pasture quality (Raubenheimer *et al* 2022, Chilcott *et al* 2020). Ultimately, fewer cattle resulted in lower total GHG emissions from livestock, and we found these emissions could be further reduced by supplementing cattle with macroalgae (to reduce enteric methane emissions, "Balanced +" scenario). Replacing urea with alternatives to reduce GHG emissions is not yet proven for extensive grazing systems, and the cost may be prohibitive. However, this may become feasible in some markets, particularly if low carbon (or carbon neutral) beef can be sold at a premium (Kilders and Caputo 2023).

Livestock grazing has largely negative impacts on biodiversity in northern Australia by degrading habitat, altering ecological communities and facilitating the spread of invasive species (Woinarski *et al* 2011, Garnett *et al* 2010). Biodiversity outcomes are somewhat improved with lower stocking rates and are significantly improved with destocking and fire management (Legge *et al* 2011a, Lunt *et al* 2007, Legge *et al* 2019). Our results also showed that species richness may increase over time in northern Australian rangelands under climate change. Australia's savannas have evolved with wide climatic tolerances, including adaption to drought and high temperatures. The projected increases in species richness correspond with projected increases in annual precipitation within the savannas, particularly increased in bird species richness in southern part of the savanna (Reside *et al* 2012). However, the positive trend in total species richness is far from certain, and including climate extremes (rather than averages) in species distribution models may restrict future species ranges (Morán-Ordóñez *et al* 2018). Similarly, other threats (such as invasive species) show large impacts on the savanna species (especially small mammals), and these threats are likely to be exacerbated by climate change (Dunlop *et al* 2012).

Influencing land management change

Our results can inform future modelling of land use change in the region under different global change scenarios, but these results need to be combined with realistic models of human behaviour (Rounsevell *et al* 2014). Although actions to mitigate greenhouse gas emissions become more profitable under most global outlooks, landholders have a wide range of risk aversion behaviours and attitudes towards adopting new practices (Rolfe and Gregg 2015). Data from cattle graziers in northern Australia's rangelands found that 85% of sampled pastoralists had low interest in adapting to climate change (Stokes *et al* 2012, Marshall and Stokes 2014, Marshall *et al* 2014). Land tenure may also constrain options for conservation land management, particularly pastoral leasehold which has a requirement to run cattle, although these conditions are not always enforced and diversification leases are emerging (DPLH 2023). Further, Indigenous lands cover large areas in northern Australia (ABARES 2016) and Indigenous peoples' attitudes towards different types of grazing land management has not yet been explored in the region. Accordingly, the potential increase in profitability of GHG emissions abatement actions is unlikely to directly translate into management change, so risk aversion and barriers to adoption should also be considered (Bryan *et al* 2016).

Additionally, it may not be possible to achieve these multiple objectives through financial incentives alone, and a more strategic planning approach may be required (Morán-Ordóñez *et al* 2016). For instance, while planned early dry season burning is likely to have positive impacts on biodiversity (Woinarski and Legge 2013) and carbon (Russell-Smith *et al* 2013), having a diversity of time-since-burnt patches across the landscape (pyrodiversity) is hypothesised to be optimal for biodiversity to accommodate the different responses of various taxa to fire (Martin and Sapsis 1992, Griffiths *et al* 2015, Perry *et al* 2016). Achieving such pyrodiversity would require a more strategic design of prescribed fires across the landscape (Legge *et al* 2011b), including the involvement of, and benefits to, Indigenous people (Perry *et al* 2018). Strategic planning may also be needed to ensure the landscape is robust to uncertainty (Runting *et al* 2018, Polasky *et al* 2011, Reside *et al* 2017). By conducting a global sensitivity analysis, we were able to establish that there is substantial spatial and temporal variation in all sustainability outcomes to 2050. This uncertainty stems not only from the different trajectories of global climate and economic change, but also the full range of model parameters. Ultimately, any spatial plan or policy needs to be robust to these uncertainties to ensure a sustainable future is not solely dependent on a particular set of parameters.

Future directions

Our model was necessarily general to encompass the broad scale of Australia's northern rangelands, so some details and dynamics were omitted that may be relevant at finer scales. Our estimates of safe stocking numbers were primarily determined by pasture growth and type (Scanlan *et al* 1994). Whilst this relationship is broadly representative, other factors can also influence the safe stocking rate at finer scales, particularly topography, location of water bodies, and the spatial distribution of grazing pressure within a property (Orr and O'Reagain 2011). Additionally, landholders do not have perfect information about future pasture growth, and herd management has many complexities not included here, so stock numbers may be unintentionally set above or below the carrying capacity of the property in a given year, with subsequent implications for land condition (O'Reagain *et al* 2014). Dynamic simulations that more closely resemble grazier actions exist at smaller spatial scales (Scanlan *et al* 2013, Ash *et al* 2015), but scaling this up to larger regions is an area for future research.

Although our study included multiple indicators (food production, landholder profit, GHG emissions, land degradation, water intake, and biodiversity), the management strategies could have further environmental impacts not considered here. While extensive livestock grazing has lower environmental impacts (per unit area) than other more intensive land use options, local and cumulative impacts can still be significant (Halpern *et al* 2022, Eldridge *et al* 2022). For example, grazing is likely to influence hydrological ecosystem services in the region, especially as grazing pressure tends to be concentrated around water points and water courses (O'Reagain and Scanlan 2013), leading to heterogenous impacts on vegetation, soils, and water, along with the potential for gully erosion (Wilkinson *et al* 2018). Management of stocking rates and fine-scale grazing pressure is particularly challenging in the region, due to low overall densities of cattle and relatively high costs of fencing or adding water points to alter grazing patterns (O'Reagain *et al* 2014). Stocking at safe levels can reduce, but not eliminate, hydrological impacts, and recovery from past grazing can take many years (Koci *et al* 2020) and involve rehabilitation measures (Bartley *et al* 2020). Ideally, future studies should consider the impacts of grazing land management on the full suite of ecosystem services.

Conclusions

Integrating multiple climate and economic drivers is often overlooked in assessments of ecosystem services, which can create misleading results and limit their utility for decision making (Runting *et al* 2017). Here we incorporated multiple drivers (i.e., temperature increase, rainfall change, fire, productivity growth, and price trajectories for livestock, farm inputs, and carbon) to assess multiple sustainability indicators to 2050. Although there were no perfect win-wins, and compromises are required under all scenarios, it is clear that the balance between production and environmental outcomes could be substantially improved by combining safe stocking rates and emission abatement action. Although our modelling is based on northern Australia, our findings are likely to be relevant to other savanna rangelands facing similar climatic and economic changes. The low input and low productivity cattle grazing systems in northern Australia are fairly typical of grazing enterprises throughout the globe's tropical savannas, which all face a likely increase in temperatures and uncertain changes in rainfall with climate change (Williams *et al* 2022). Rising cattle prices, driven by a growing demand for beef, is also a global phenomenon that influences markets beyond northern Australia (Turk 2016). Constraining climate change to the less severe scenarios will require strong global action, producing substantial incentives for emissions abatement (Hatfield-Dodds *et al* 2015a). As the grazing lands in northern Australia and elsewhere become less suitable for livestock production, the opportunity to diversify income streams may prove vital in a changing climate (Russell-Smith and Sangha 2018).

Data Availability

Data and code will be made available in the University of Melbourne repository upon acceptance.

Acknowledgements

RKR was supported by an Australian Research Council Discovery Early Career Research Award (DE210100492). JRR was supported by an Australian Research Council Future Fellowship (FTFT200100096). We would like to thank Dr. Tim Doherty for ecological advice.

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Supplementary Information

Balancing production and environmental outcomes in Australia's tropical savanna under global change

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1 Fire Modelling

Wildfire impacts greenhouse gas emissions through the combustion of vegetation, with hotter and more frequent fires generally having a greater impact (Hunt et al 2014). We calculated fire frequency and severity for each pixel in the study region using recurrent-event regression analysis with shared frailty (Munda et al 2012) based on 27 years of burn scar data (1988 – 2014) and simulations based on Relative Difference Normalised Burn Ratio (DEE 2015) calculated from time-series satellite imagery. The key output of this modelling was the fire risk (occurrence and severity) in each pixel, which can be interpreted as the proportion of vegetation burned, for the historic baseline and the year 2050. High fire risk is characterised by warm temperatures, a lack of temperature seasonality, and high (but seasonal) rainfall, with much of the northern savanna having a high chance of experiencing fire. This model found that climate change increased fire frequency and intensity, primarily through higher temperatures, although there was some variation across space and GCMs. Consequently, there was a general increase in fire risk across the area currently managed for grazing (e.g., under RCP M3 and GCM MPI, fire increased by 50.7% by 2050). To calculate the change in the proportion of vegetation burnt over time, we assumed a linear change in fire risk from the historic baseline to 2050. The central setting of the integrated simulation was based on the mean fire risk, with the mean of the lowest and highest 20% of simulations used to bound the sensitivity analysis (main text).

1.1 Fire hazard

Fire hazard in the north of Australia was modelled using survival analysis in the *R* statistical software environment (R Core Team 2015). Modelling the relationship of both temperature and rainfall to fire events for each location in the study area enabled the simulation of fire hazard to be extended to consider the effects of climate change.

Fire frequency data for Australia from 1988 – 2014 was obtained from WA Firewatch, Landgate (*www.firewatch.landgate.wa.gov.au*). This 1 km spatial resolution data was resampled to 2 km and combined with resampled 3″ANUCLIM outputs of mean annual temperature, mean annual rainfall (Hutchinson *et al* 2008) and resampled 100 m NVIS 3.1 vegetation presence (0, 1) (Department of the Environment and Water Resources (DEWR) 2007). This data was reformatted into a survival dataset, and parametric frailty modelling (PFM) was undertaken for vegetated locations using the R package *parfm 2.5.15* (Munda *et al* 2012). The *select.parfm* function was used to compute Akaike and Bayesian information criterion (AIC and BIC) values of parametric frailty models with different baseline hazards and different frailty distributions (Table S1). Although the lognormal and loglogistic distributions had a lower AIC and BIC we used the Weibull distribution to represent baseline hazard with a gamma distribution for frailty because of its flexibility and interpretability (Eqn S1 – R code).

```
parFrail <- parfm(Surv(Time, Status) ~ meanrain + meantemp,   cluster="ID", data=survDS,
dist="weibull", frailty="gamma", method="Nelder-Mead", maxit=50000, showtime=TRUE)
```

(S1)

Table S1 | AIC and BIC results.

Baseline hazard distribution	Frailty distribution					
	AIC			BIC		
	gamma	inverse Gaussian	positive stable	gamma	inverse Gaussian	positive stable
exponential	851.907	848.529	873.069	865.625	862.246	886.787
weibull	811.113	811.565	846.897	828.26	828.712	864.044
gompertz	843.624	----	874.806	860.771	----	891.953
loglogistic	760.35	----	790.104	777.497	----	807.251
lognormal	756.629	757.692	----	773.776	774.839	----

Frailty for each vegetated location was then calculated from the PFM output parameters (Table S2) (Munda *et al* 2012). Results were then imported into a GIS and a mean focal statistics method was used to provide frailty measures for (currently) non-vegetated areas. The frailty was then used in *R* to calculate and export instantaneous hazard (Eqn S2 – R code) for each year (t) in a 100 year period for each location under mean annual rainfall and temperature:

```
hz <- rho * lambda * t^(rho-1)  * frailModXY_full$frailMod * exp(meanraincoeff *
    dFXYPCs$meanrain + meantempcoeff * dFXYPCs$meantemp)
```

(S2)

Table S2 | Parametric frailty modelling results

	Estimate	**Standard error**	**p-value**
theta	1.320	0.004	
rho	1.564	0.001	
lambda	7.891316e-07	4.097809e-08	
meanrain	0.002	8.006945e-06	0 ***
meantemp	0.388	0.002	0 ***

Loglikelihood: -3992791.98
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
Kendall's Tau: 0.398

Changes in rainfall and temperature for 2050, modelled under three climate scenarios (RCP2.6, RCP4.5 and RCP8.5) (Figures S1 and S2), were then applied to the mean annual rainfall and temperature and instantaneous hazard for a 100 year period again calculated (Eqn S3 – R code). Figure S3 provides examples of instantaneous hazard for three locations.

```
hz <- rho * lambda * t^(rho-1) * frailModXY_full$frailMod * exp(meanraincoeff *
        precipDelta + meantempcoeff * tempDelta)                    (S3)
```

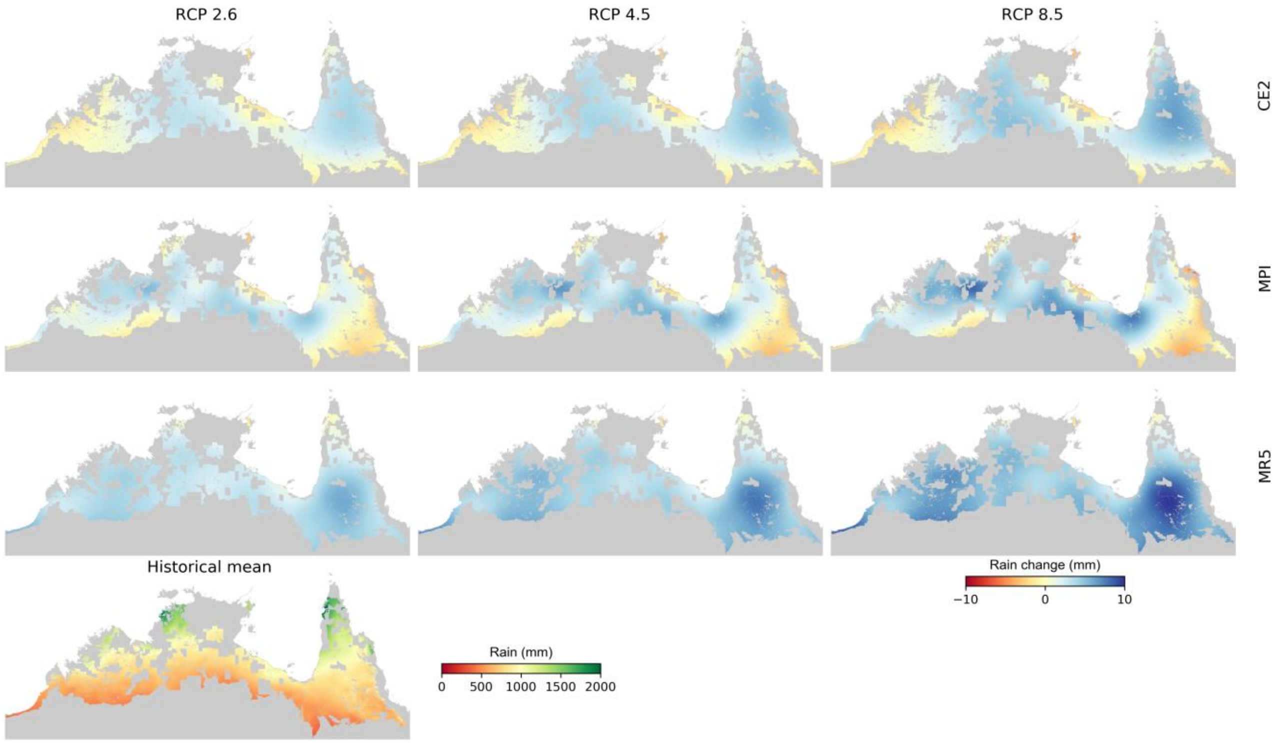


Figure S2 | Rainfall in 2050 across scenarios compared with the ANUCLIM historical mean.

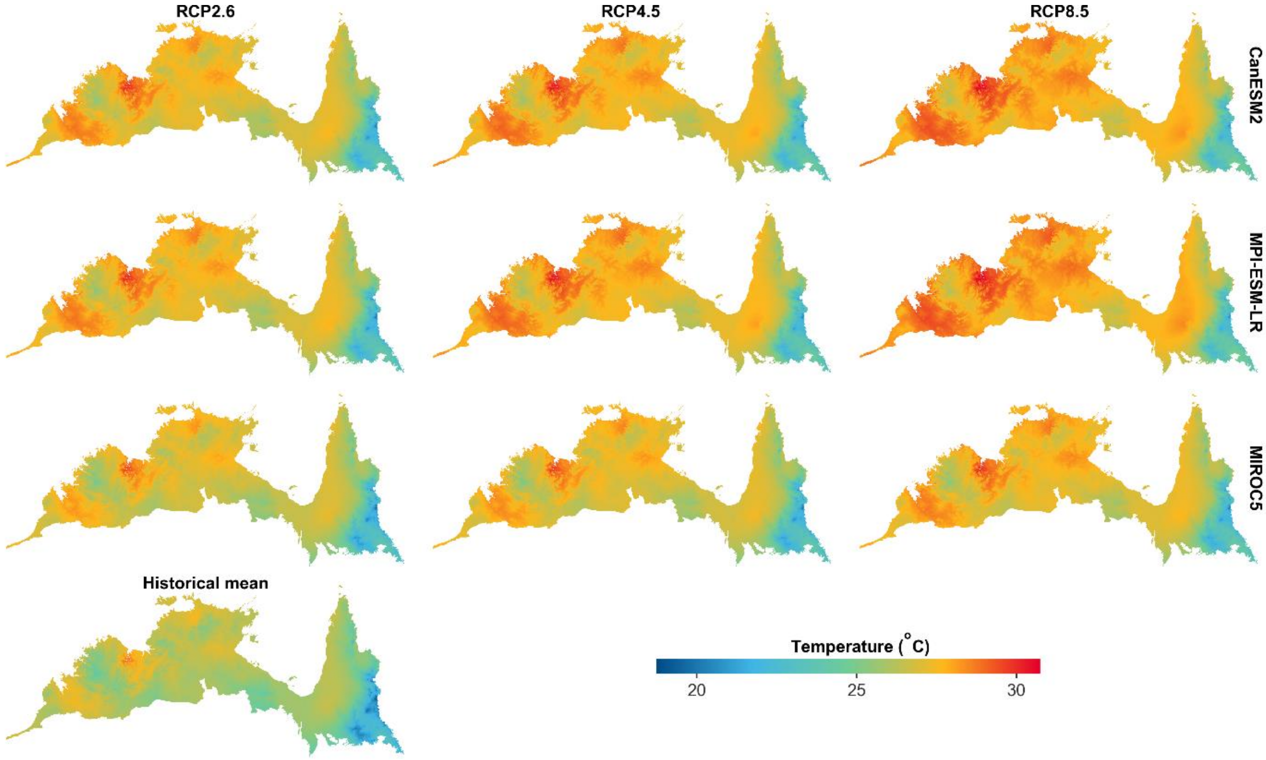


Figure S3 | Mean annual temperature in 2050 across scenarios compared with the ANUCLIM historical mean.

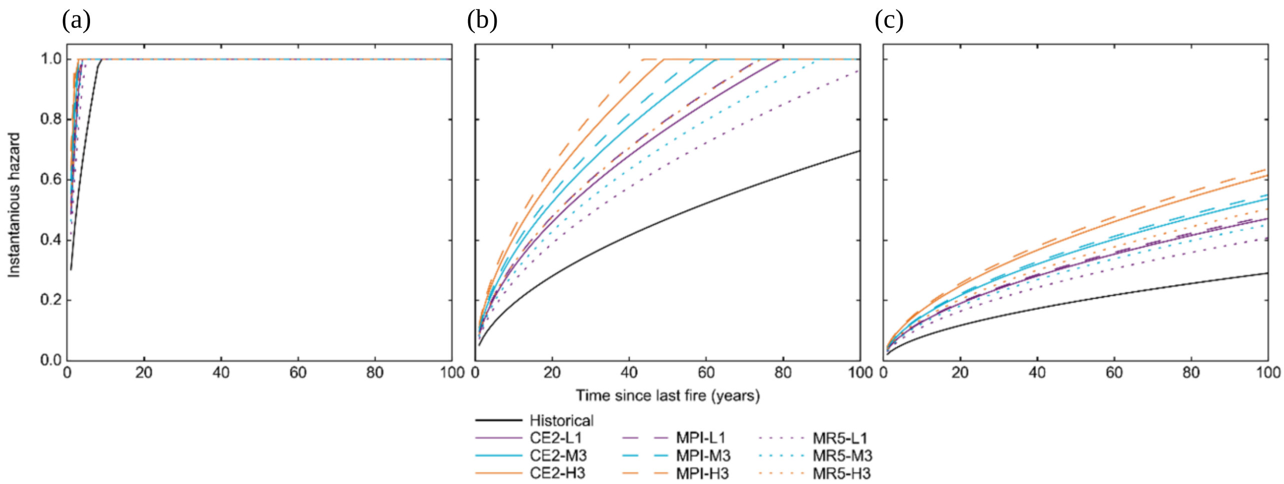


Figure S4 | Examples of calculated instantaneous hazard from 3 different locations (a-c). Here, global outlook M3 represents both M3 and M2, as these were both based on RCP 4.5.

1.2 Fire severity

Fire severity, as the percentage of biomass lost to fire, was modelled using the MODIS Nadir BRDF-Adjusted Reflectance 16-Day L3 Global 500m data for years from 2002 – 2014 (Nasa Lp Daac 2015). The Normalised Burn Ratio (NBR - Eqn S4) was originally developed with Landsat satellite data using the near infra-red band 4 and mid infra-red band 7 (Lopez Garcia and Caselles 1991).

$$NBR = \frac{iR_n - iR_m}{iR_n + iR_m} \qquad \text{(S4)1}$$

Where iR_n is near infra-red and iR_m is mid infra-red. The differencing of MODIS derived pre-fire NBR and post-fire NBR has been used in burned area mapping (Loboda *et al* 2007). A relative differencing of the NBR (RdNBR - Eqn S5) using Landsat satellite data has been found to allow a more direct comparison of severity between fires across space and time (Miller and Thode 2007).

$$RdNBR = \frac{NBR_{Pre-fire} - NBR_{Post-fire}}{\sqrt{|NBR_{Pre-fire}|}} \qquad \text{(S5)}$$

MODIS Band 2 (near infra-red) and Band 7 (mid infra-red) were used to calculate the relative differenced normalised burn ratio (RdNBR) for burn areas defined by the Landgate dataset. The 5th, 50th (median) and 95th percentile of RdNBR for Interim Biogeographic Regionalisation for Australia (Australian Government 2012) regions was calculated (Figure S4).

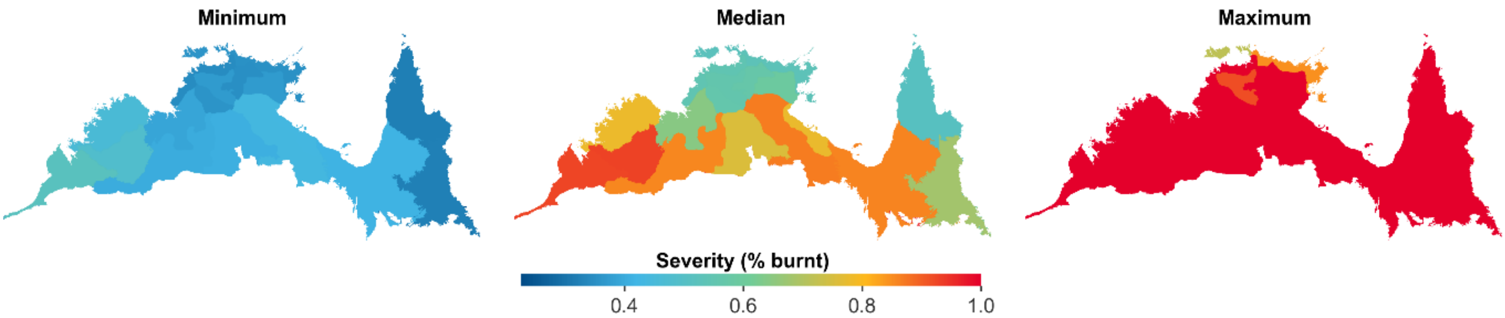


Figure S5 | Range of severity by IBRA regions.

1.3 Fire simulation

The fire simulations were produced using Python (van Rossum and the Python Community 2012) and Numpy (Jones *et al* 2001). For each location, over a one hundred year period, fire events and their severity were simulated under mean conditions and for 2050 under the three climate scenarios. The fire simulations modelled at the 2 km spatial resolution was resampled to 0.01 degree spatial resolution for use in the integrated simulation model. Fire events at each location were simulated using a random draw from a binomial distribution determined by the instantaneous hazard with time since last fire event determining the level of hazard. Severity of fire events was drawn from a triangular distribution using the range of RdNBR for each location.

1.4 Results (fire modelling)

The simulations of fire events under historical mean conditions were used to assess model accuracy (Figure S5). A mean absolute error of 4.07% and a standard error of 5.72% indicates a good fit with mapped historical fire events. A bias, mean difference between historical fire frequency and simulated fire frequency, of -0.34% shows a slight overall overestimate of fire frequency. Figure 5 provides a comparison of actual versus modelled fire frequency for simulations resampled to 0.01 degree spatial resolution. Although some spatial accuracy is lost in the resampling of results a visual comparison of mapped actual and simulated percentage frequency of fire events at the 0.01 degree resolution shows the overall pattern of fire frequency is reproduced by the simulations (Figure S6).

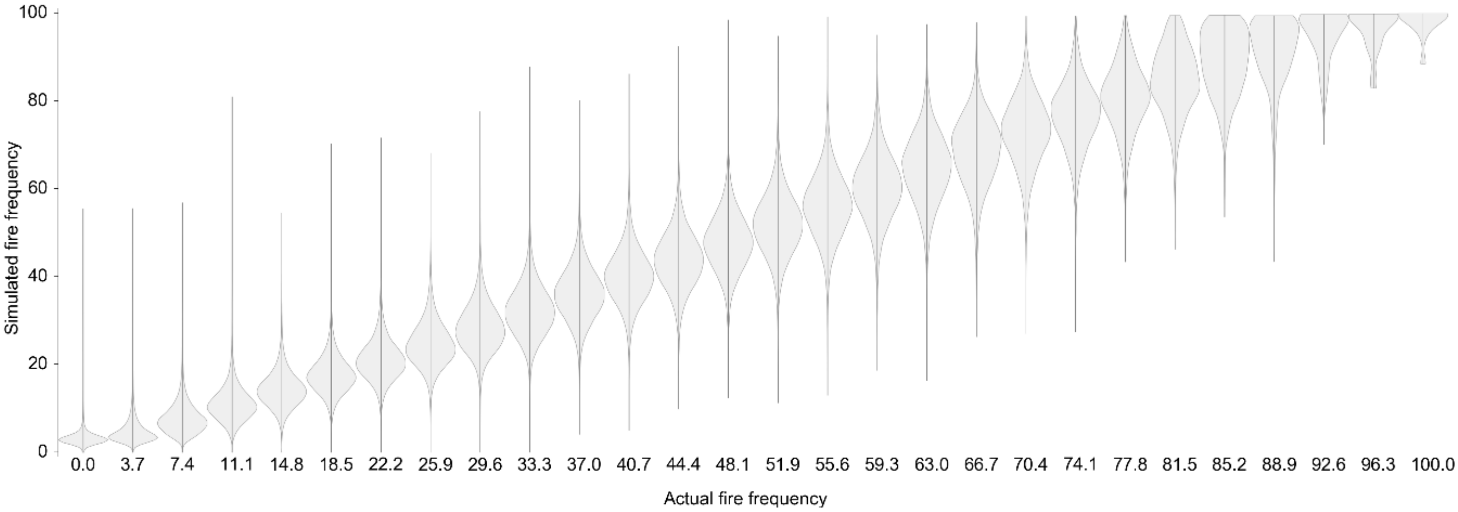


Figure S6 | Violin plot of actual versus simulated fire frequency. Actual fire frequency was calculated as the number of years burnt within the 27 years of burn area data.

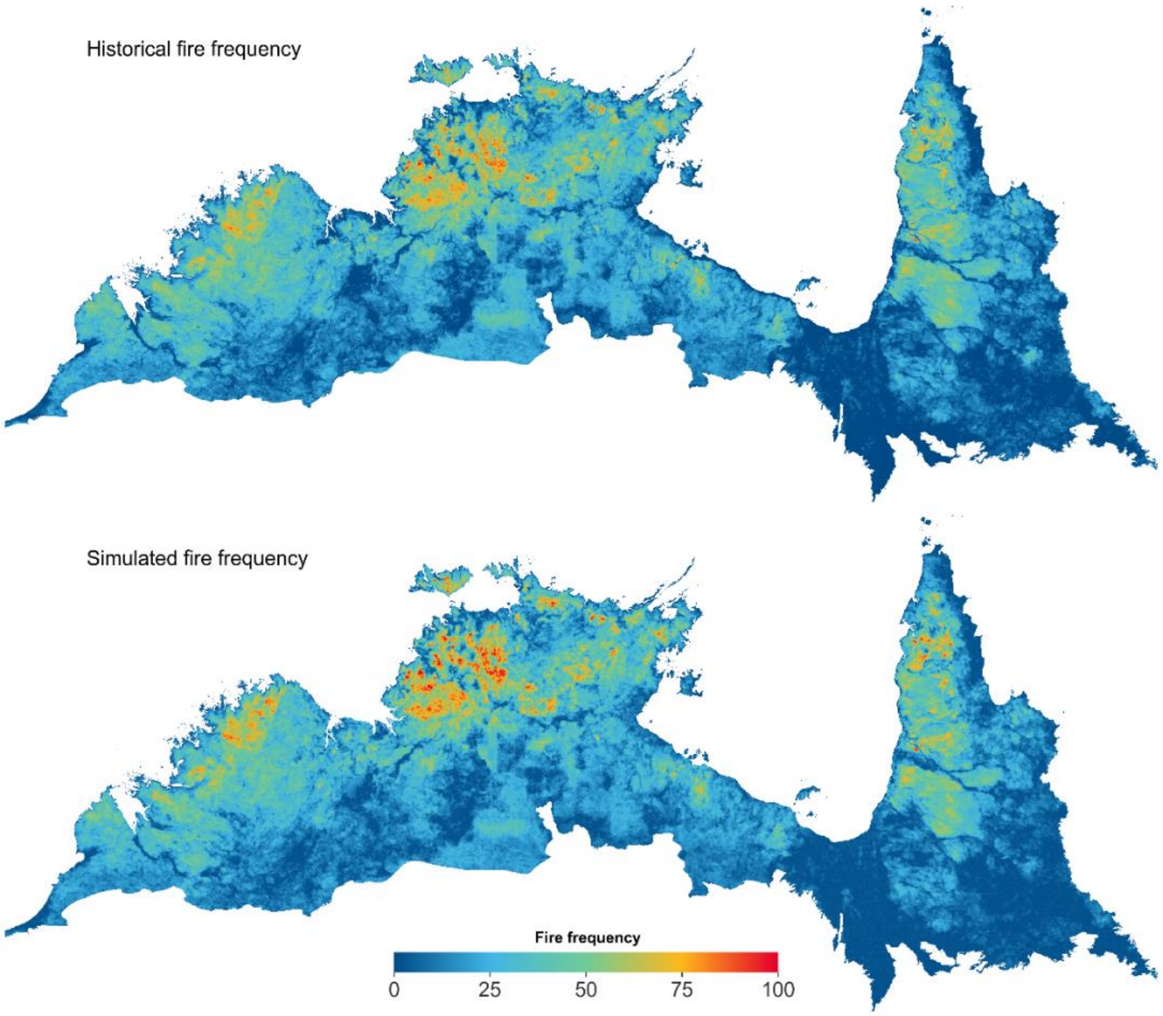


Figure S7 | Comparison of fire frequency (top) with fire event simulations modelled on historical mean climate (bottom).

Temperature increases vary between all climate scenarios with this variation reflected in the fire event simulations (Figure S7) as expected with the positive relationship between fire events and temperature indicated by the PFM temperature coefficient. Mean frequency of simulations match actual, and increase with increasing temperature in the 2050 simulations (Table S3). The MIROC5 global climate modelling having the smallest increase followed by CanESM2 with the MPI-ESM-LR modelling having the highest. Area of low frequency fires reduces, and areas of higher frequency fires increases as temperatures increases (Table S4). The median percentage biomass lost (Figure S8) increases as with fire events by climate scenario however, the spatial pattern of increase reflects variations in severity by IBRA regions.

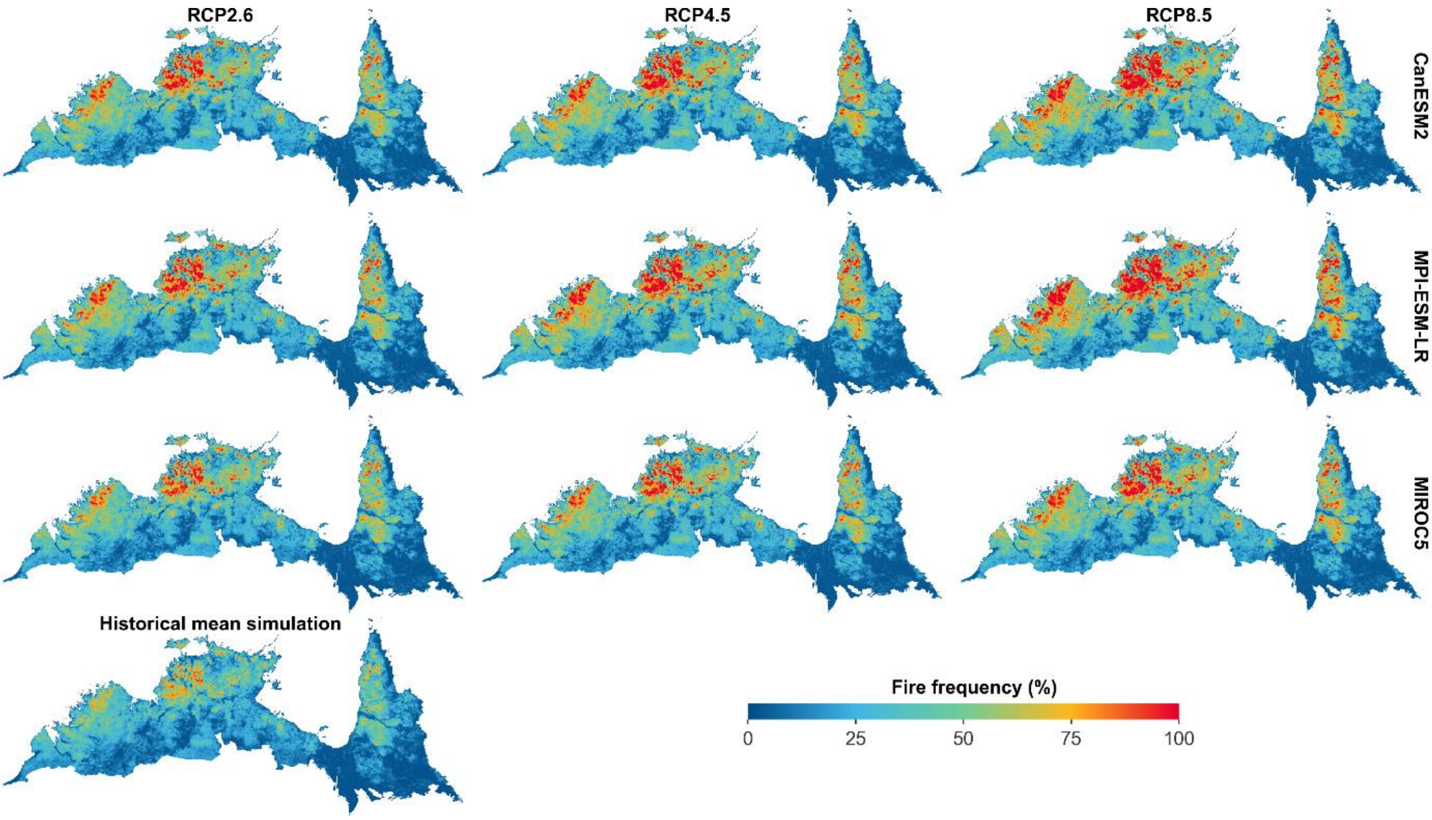


Figure S8 | Fire event simulations in 2050 for RCP's 2.6, 4.5, and 8.5, for 3 different GCM's, compared to the historical mean.

Table S3 | Historical and simulated fire frequency mean and standard deviation.

Scenario	Mean	STD
Actual 1988-2014	22.31	17.66
Historical mean climate	22.65	17.80
MIROC5 RCP2.6 - 2050	28.06	21.82
MIROC5 RCP4.5 - 2050	29.89	23.09
MIROC5 RCP8.5 - 2050	31.97	24.44
CanESM2 RCP2.6 - 2050	30.36	23.19
CanESM2 RCP4.5 - 2050	32.78	24.68
CanESM2 RCP8.5 - 2050	35.42	26.20
MPI-ESM-LR RCP2.6 - 2050	30.98	23.82
MPI-ESM-LR RCP4.5 - 2050	33.65	25.52
MPI-ESM-LR RCP8.5 - 2050	36.61	27.23

Table S4 | Areas of fire frequency ranges

Scenario	Area (Mha)			
	0-25	25-50	50-75	75-100
Actual 1988-2014	78.867	42.909	10.049	0.691
Historical mean climate	83.593	37.357	9.988	1.568
MIROC5 RCP2.6 - 2050	70.298	40.211	16.619	5.009
MIROC5 RCP4.5 - 2050	66.382	40.474	18.490	6.437
MIROC5 RCP8.5 - 2050	62.292	40.610	20.147	8.195
CanESM2 RCP2.6 - 2050	64.864	41.332	18.965	6.525
CanESM2 RCP4.5 - 2050	60.160	41.467	20.934	8.491
CanESM2 RCP8.5 - 2050	55.633	41.215	22.301	10.963
MPI-ESM-LR RCP2.6 - 2050	64.167	40.401	19.714	7.245
MPI-ESM-LR RCP4.5 - 2050	59.285	40.359	21.384	9.668
MPI-ESM-LR RCP8.5 - 2050	54.569	39.868	22.377	12.577

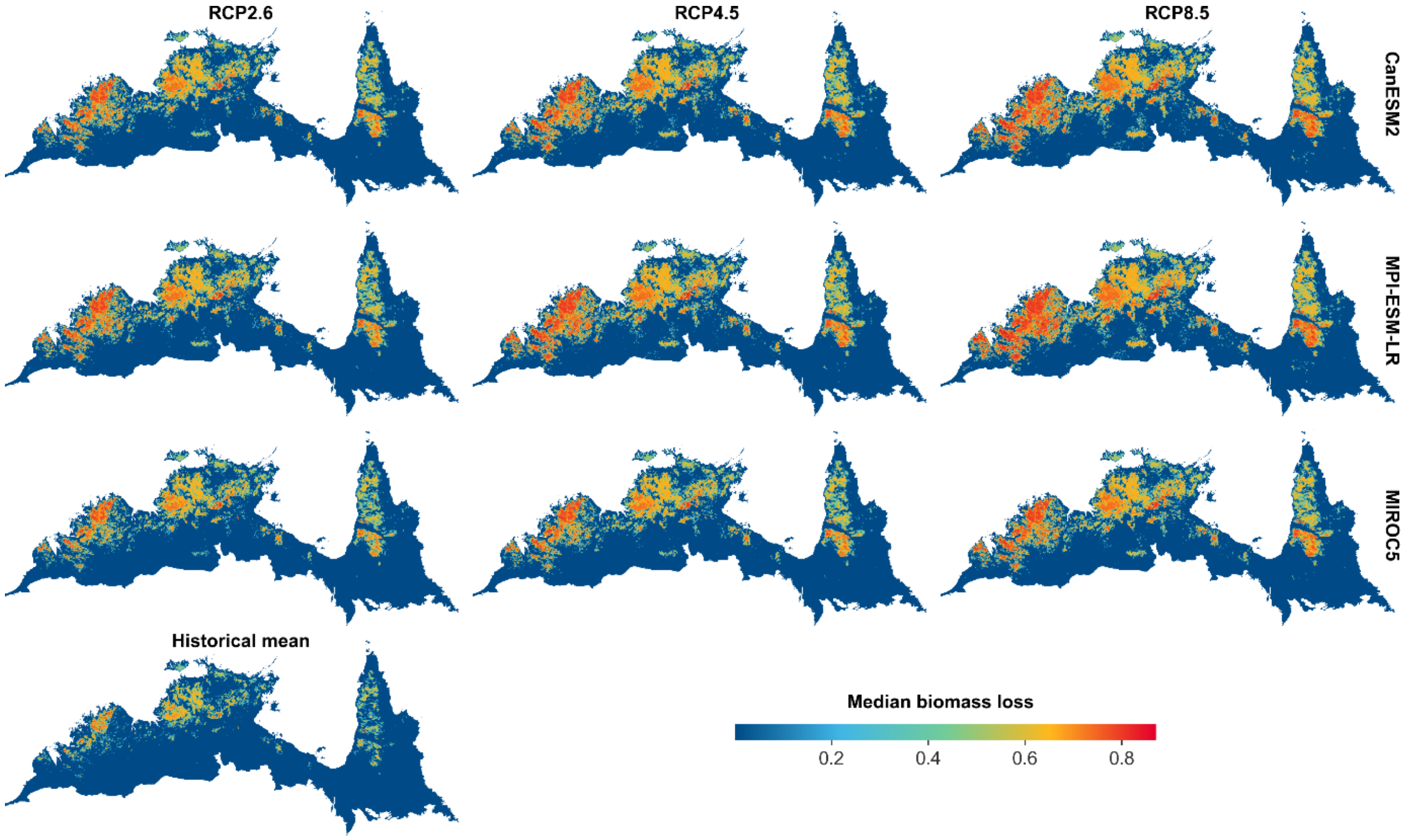


Figure S9 | Median percentage of biomass lost in 2050 for RCP’s 2.6, 4.5, and 8.5, for 3 different GCM’s, compared to the historical mean.

The key output of from this modelling was the fire risk (occurrence and severity) in each pixel, which can be interpreted as the proportion of vegetation burned, for the historic baseline and the year 2050. High fire risk is characterised by warm temperatures, a lack of temperature seasonality, and high (but seasonal) rainfall, with much of the northern savanna having a high chance of experiencing fire. This model found that climate change increased fire frequency and intensity, primarily through higher temperatures, although there was some variation across space and GCMs. Consequently, there was a general increase in fire risk across the area currently managed for grazing (e.g., under RCP M3 and GCM MPI, fire increased by 50.7% by 2050). To calculate the change in the proportion of vegetation burnt over time, we assumed a linear change in fire risk from the historic baseline to 2050. The central setting of the integrated simulation was based on the mean fire risk, with the mean of the lowest and highest 20% of simulations used to bound the sensitivity analysis.

1.5 GHG Emissions Calculations

We calculated the GHG emissions from wildfire, and the emissions abated via prescribed burning, using methods adapted from the official greenhouse gas accounting methodology of the Australian Government (DEE 2015). Prescribed burns are typically undertaken early in the dry season, with the aim of preventing the extent and severity of wildfires late in the dry season by reducing the fuel load (Russell-Smith *et al* 2013). The official methodology was designed to apply to the property scale, so modifications were necessary to be suitable for a broad scale assessment (akin to Heckbert et al. (2012) and Adams and Setterfield (2013)). Burnable fuel was calculated by reclassifying vegetation data from the National Vegetation Information System (NVIS 2016) and applying the corresponding value for burnable fuel given in Heckbert et al. (2012). The fuel load was increased by 5.6% over the modelling period where destocking was allocated on previously grazed land (i.e., increased by 0.11% of initial value per year; derived from the figures in (Bray and Golden 2009)). Accordingly, the fuel load was decreased by 5.6% over the modelling period if grazing occurred on previously ungrazed pixels (i.e., -0.11% of initial value per year). As the study focused on land currently allocated for grazing, this only occurred on 0.25% of pixels. Oversowing with legumes was assumed to have a negligible effect on fire and did not impact fuel loads. The mass of fuel burnt (in Gg) in each year from 2013 – 2050 was calculated by:

$$M_i = BF_i \times FR_i \times (1 - ER) \qquad \text{(S6)}$$

Where M_i is the mass of fuel burnt in each pixel, BF_i is the burnable fuel in each pixel, FR_i is the simulated fire risk (occurrence and severity) for each pixel, and ER is the reduction in fire risk from management (i.e. prescribed burns). *ER* was set to either 0 (to represent no management), or a proportion to represent the emissions reduced by management. This was set at 0.34 for the main analysis (Russell-Smith *et al* 2013, 2009b) and varied between 0.25 (a conservative estimate of management effectiveness (Heckbert *et al* 2010)) and 0.48 (the upper potential of management (Russell-Smith *et al* 2009a)) in the sensitivity analysis.

Only methane and nitrous oxide emissions are accounted for in the Australian GHG accounting methodology, as it is assumed that any CO_2 released is eventually re-absorbed as the vegetation regrows (DEE 2015). Therefore, to convert the mass of fuel burnt into greenhouse gas emissions, the following equations were applied:

$$EM_i = M_i \times CC \times EF_{CH_4} \times G_{CH_4} \qquad \text{(S7)}$$

$$EN_i = M_i \times CC \times EF_{N_sO} \times G_{N_sO} \times NC \qquad \text{(S8)}$$

$$GHG_i = MP_{CH_4} EM_i + MP_{N_2O} EN_i \qquad \text{(S9)}$$

Where EM_i and EN_i are the annual emissions of methane and nitrous oxide respectively for each pixel *i*, *CC* is the carbon content of fuels (0.46 (DEE 2015, Heckbert *et al* 2012)), EF_{CH_4} and EF_{N_sO} are the emission factors for methane (0.00455) and nitrous oxide (0.00784) (DEE 2015), G_{CH_4} and G_{N_sO} are the elemental to molecular mass fractions for methane (1.33) and nitrous oxide (1.57) (DEE 2015, Heckbert *et al* 2012), *NC* is the nitrogen to carbon ratio (0.00857) (DEE 2015), MP_{CH_4} and MP_{N_sO} are the multipliers to convert methane (25) and nitrous oxide (298) to CO_2 equivalents (CO_2e) (DEE 2016), and GHG_i is the Mg of CO_2e in each pixel *i*.

2 Pasture production model

2.1 Climate

Historical climate data used in the model was derived from the Bureau of Meteorology’s 5 km gridded Australia daily datasets (Jeffrey *et al* 2001) (Figure S9 and S10). Daily data was aggregated to monthly, seasonal or annual data for analysis and resampled to 1 km grid cells. Additional summary layers were calculated to use as the historical baseline from which estimates of future climate could be derived. Within the northern Australian study area rainfall across the region is subject to monsoonal patterns of wet and dry with the higher rainfall wet season typically occurring between September and March while the period between April and October is generally dry (Gleeson *et al* 2012).

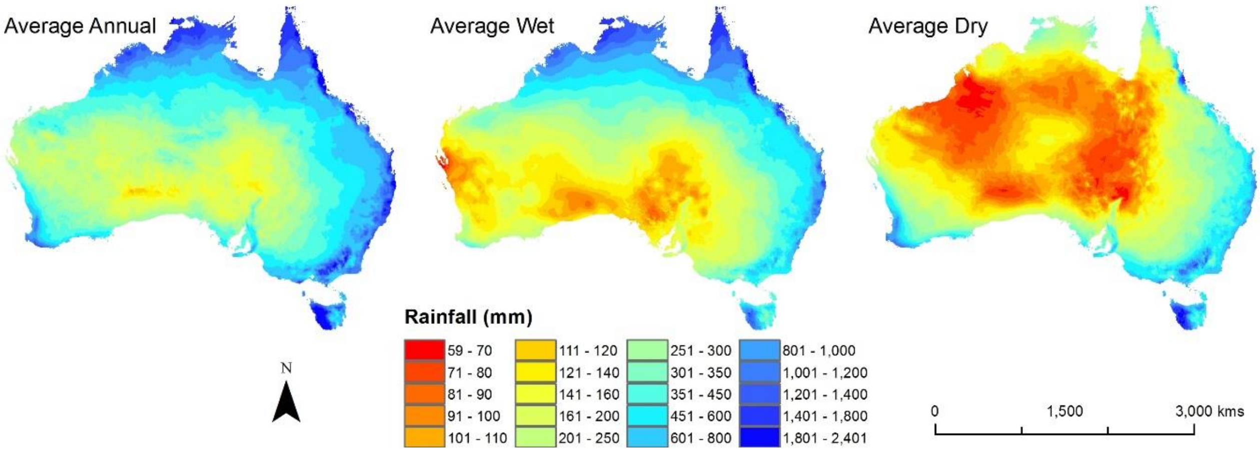


Figure S10 | Average annual, wet season, and dry season rainfall for Australia (Jeffrey *et al* 2001).

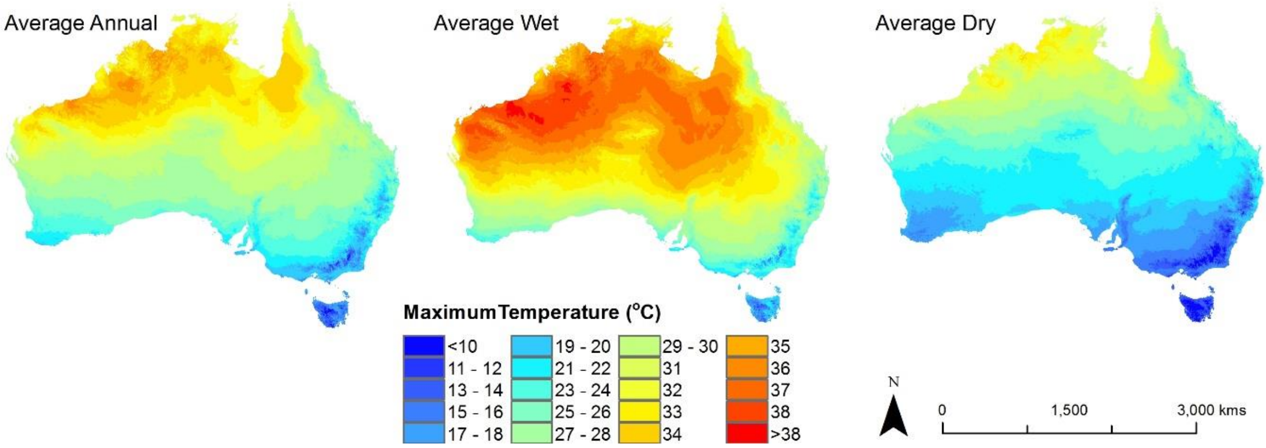


Figure S11 | Average annual, wet season, and dry season maximum temperature for Australia (Jeffrey *et al* 2001).

2.2 Pasture Production Estimation

We used long run data outputs from the AussieGrass pasture production model. This model has been developed by Department of Environment and Resource Management in Queensland and represents the most complete model of pasture production in Australia. The AussieGrass model is based fundamentally on a point based soil-water balance pasture production model called GRASP. Much like APSIM the GRASP model uses soil and climatic parameters in a plant phenology model to estimate pasture production rates under specified conditions on a daily time step. Within AussieGrass, the GRASP model runs across a 5km by 5km grid covering all of Australia. Outputs are calibrated against values from NOAA's Normalized Difference Vegetation Index (NDVI) and ground-truthed through 600,000 field observations (Stone *et al* 2010). Long run and large scale datasets (as used in this model) are only available at more aggregated sub-IBRA region levels (Australian Government 2012) (Figure S11).

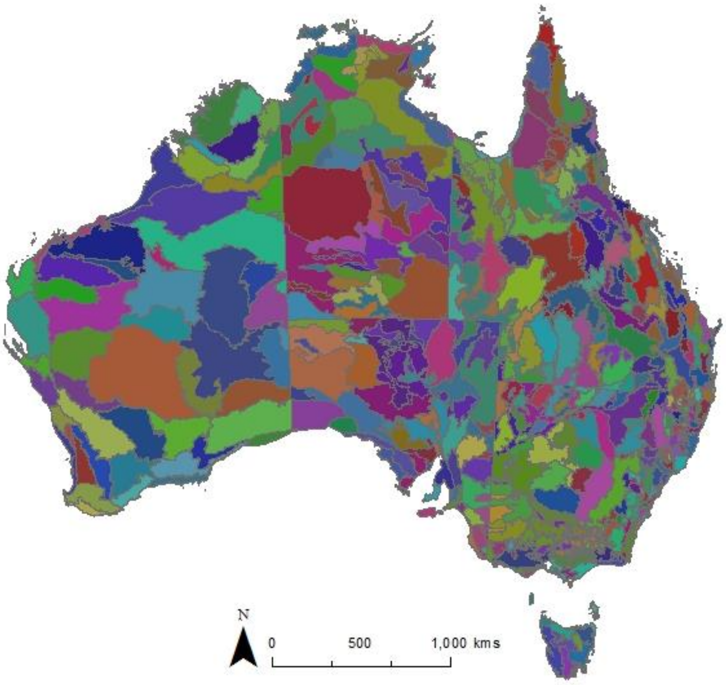


Figure S12 | Australian IBRA sub-regions (Australian Government 2012). Colours are randomly applied to facilitate the visualisation of IBRA sub-region boundaries.

In total 125 years of monthly pasture growth data based on the historical climate record 1890 to 2015 were obtained. AussieGrass model parameters and outputs were provided at the monthly time step and include rainfall, min and max temperatures, evaporation, pasture growth, total standing dry matter, and three safe stocking rate parameters (% utilization, total cover and eaten) (Table S5).

Table S5 | Example data from AussieGrass modelling.

Year	Month	rai	max	min	evap	growth	tsdm	utilization	totalcover	eaten
1890	1	267.3	29.6	20.7	5.1	1581.2	4264.9	1.1	89.3	16.7
	2	181.2	30	20.6	5	461.4	4525.8	1.7	91.4	15
	3	367.2	29.9	19.9	4.7	183.3	4481.1	2.2	91.7	13
	4	47.8	27	17.2	4.2	57.2	4308.5	2.9	91.5	14.1
	5	80.7	24.8	14.4	3.3	15	4070.9	3.6	91.4	14.5
	6	27.7	23.3	11.8	2.9	20	3842.5	4	91	9.9
	7	29.1	22.5	9	3.2	5.5	3574.3	4.5	90.8	10.3
	8	4.2	25.1	10	4.1	1.5	3281.6	5	90.5	10.3
	9	56.5	28	13.4	5.5	7.6	2941.9	5.6	90.2	14.5
	10	24.3	31.8	16.8	6.9	27.2	2618.5	92.8	89.5	15
	11	47.2	32	17.6	7.2	90.8	2387.5	43.2	88.8	14.5
	12	75.4	32.6	19.6	6.8	538.6	2592.4	11.9	88.4	18.7
1891	1	288	30.9	21.2	5.4	1526.6	3818.2	3.4	89.2	18.7
	2	223.5	29.2	20	4.7	1165.4	4728.5	2.6	91.4	16.9
...	...	...	...	...	...	...	...	...	...	...
2014	11	5	33.7	20.2	9.3	1.1	802.1	10.1	77.9	19.6
	12	66.4	34.2	21.8	8.3	43.7	678.1	78.7	75.4	23.4

2.3 Future climate modelling

Three possible future climate scenarios (RCP 2.6, RCP 4.5, and RCP 8.5) (Hatfield-Dodds *et al* 2015b, van Vuuren *et al* 2011) resulting from specified emissions trajectories were modelled through three General Circulation Models (GCM). Each GCM (CanESM2, MPI-ESM, and MIROC5) produced future climate deltas for rainfall and temperature for each year between 2013 and 2050 at ~1.88° resolution. The mid-points of these data were then interpolated to 1.1 km grid cell resolution using a regularized spline interpolation technique. This approach is an exact interpolator where interpolated values honour the original value at the data point, with a smooth surface in between (continuous first derivative) (Figure S12). It is important to note that the original climate deltas are an average value for the entire 295km^2 grid cell as modelled in the three climate models. Therefore, the interpolation approach has the potential to violate some of the original assumptions/processes used in the climate modelling. However, as high-resolution data is necessary to produce a smooth high-resolution surface (removing unrealistic sharp spatial edges between very coarse grid cells).

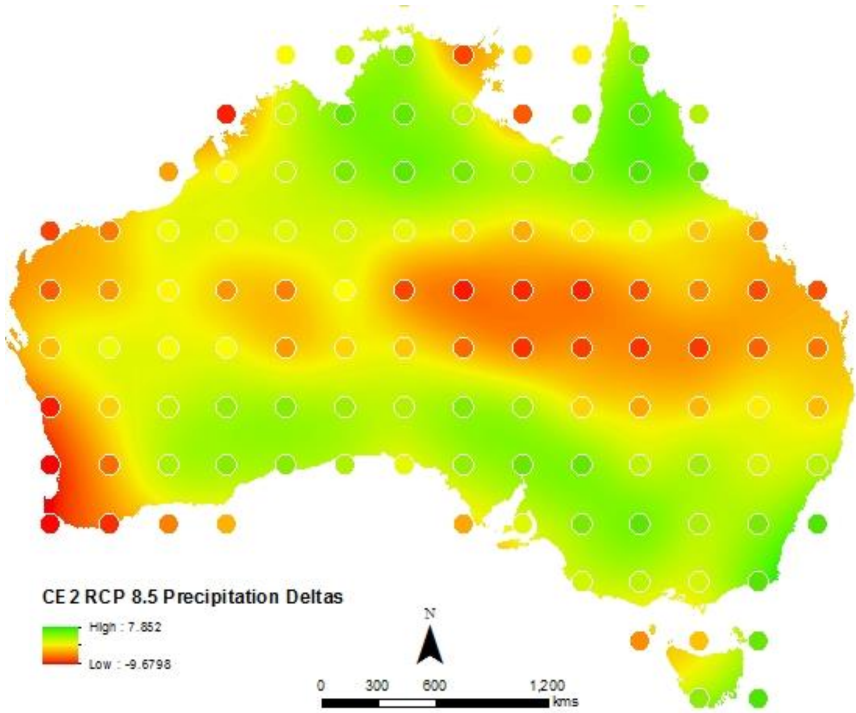


Figure S13 | An example output of the climate data interpolation technique.

The historical climate data series carries considerable variability over time and space and while we can generally reproduce the spatial variability there is uncertainty associated with predicting each future year. The climate deltas represent an expected average change for each given location. Future climate prediction in this model assumes average historical climate as a baseline and predicts forwards using the interpolated climate deltas. Each year generates a new mean climate layer for rainfall and temperature to which regression function applied and pasture predicted.

2.4 Regression

AussieGrass data from a set of randomly selected locations was examined to explore the relationship between climatic variables and pasture production. The three climate parameters produced in the AussieGrass outputs are rainfall, temperature and evapotranspiration. This means our analysis did not include the potential for elevated atmospheric CO_2 concentration to influence pasture growth via woody thickening, reducing available space for pasture (though note the high R-squared values of the selected model in table S6). Scatter plots of model variables for the randomly selected regions provide a first cut indication of any potential correlation between climate parameters and pasture growth (Figure S13). These scatter plots of indicated a likely relationship between rainfall and pasture and less of a relationship between temperature or evapotranspiration and pasture. In order to identify the drivers of pasture production we tested several regression equations on the sample locations. Three regression approaches (linear, quadratic, General Additive Model) were considered each with a variation of rainfall, temperature and evapotranspiration (Table S6). Analysis of the regressions returned R-squared values in the range of 0.6 to 0.98 with linear regression exhibiting the best fit using rainfall and maximum temperature as the independent variables (Table S6). Simulations using this model were closely aligned with actual data (Figure S14). A baseline of annual rainfall and maximum temperature was created by taking the mean from 1987 to 2010 from using data from Australian Government Bureau of Meteorology (Jeffrey *et al* 2001). We also created upper and lower bounds based on the 10^{th} and 90^{th} percentiles. These baselines were used to project the change in maximum temperature, rainfall, and subsequently pasture growth based on the projections for each global outlook and GCM.

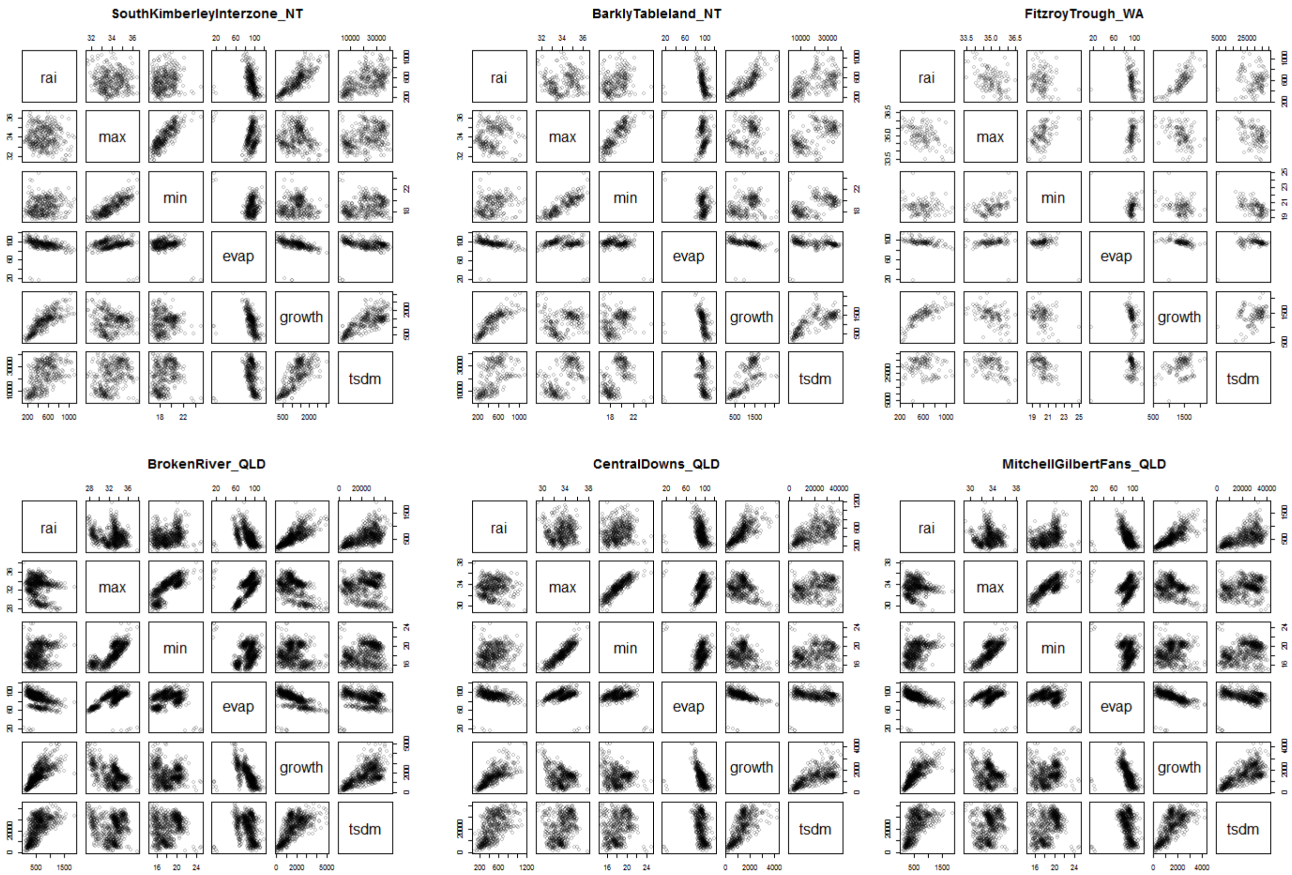


Figure S14 | Scatter plots of climate and pasture production in six selected sub regions.

Table S6 | Regression R-squared results for sample locations

WA	**NT**		**QLD**			
Fitzroy Trough	**Barkly Tableland**	**South Kimberley Interzone**	**Central Downs**	**Mitchell Gilbert Fans**	**Broken River**	**Model**
0.753302	0.766419	0.695247	0.568605	0.676282	0.633264	general additive model of growth and rainfall

0.763583	0.826339	0.695339	0.758941	0.78647	0.809728	general additive model of growth and rainfall + max temp
0.790674	0.793865	0.786059	0.779543	0.796874	0.822701	general additive model of growth and rainfall + evap
0.966094	0.952617	0.944574	0.901814	0.919118	0.906028	linear model of growth and rainfall (intercept removed)
0.98744	0.934126	0.957822	0.94912	0.980617	0.963415	linear model of growth and rainfall + max temp
0.985824	0.952847	0.944576	0.901897	0.921319	0.906029	linear model of growth and rainfall + evap
0.653982	0.716879	0.659891	0.54238	0.623092	0.581039	linear model of growth and rainfall + quadratic rainfall
0.661594	0.729609	0.660056	0.648834	0.67925	0.741891	linear model of growth and rainfall + quadratic max temp
0.654612	0.722806	0.689872	0.604389	0.667123	0.704259	linear model of growth and rainfall + quadratic evap

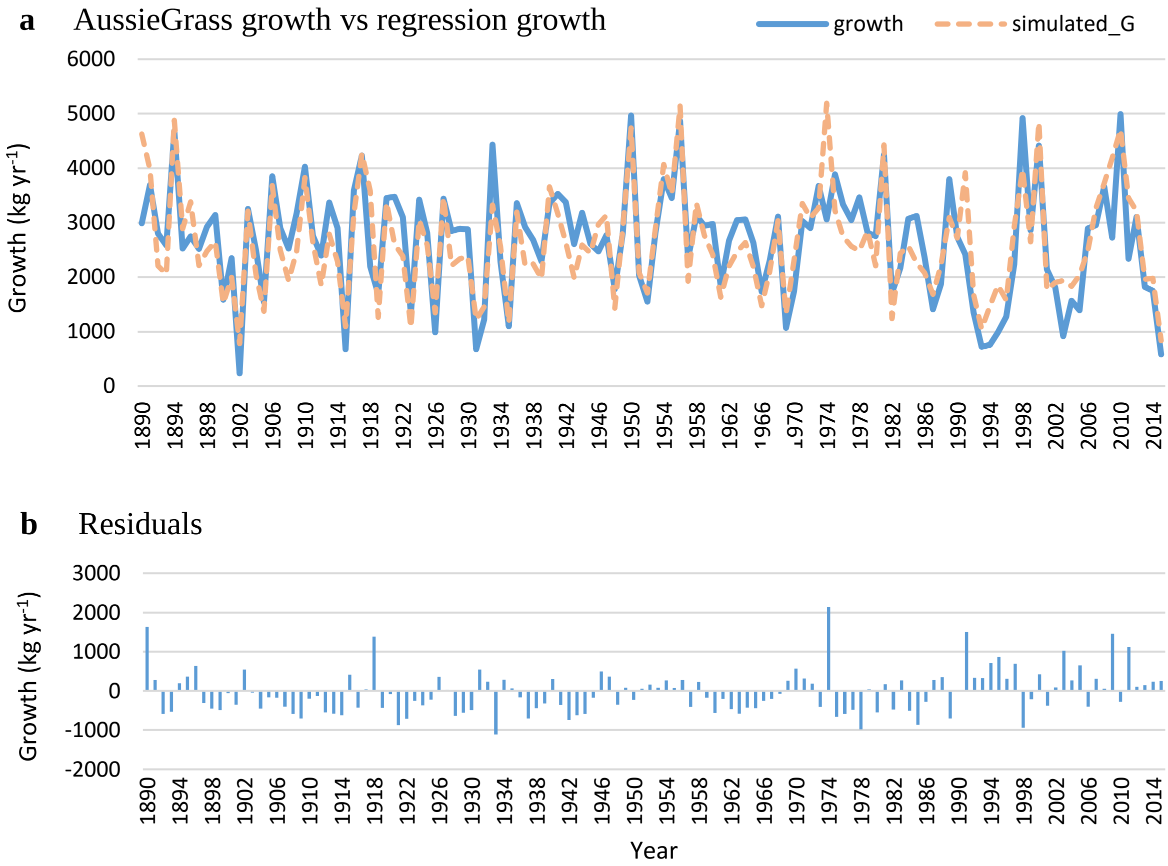


Figure S15 | Comparison of AussieGrass pasture production data and growth (a) simulated via regression equation with residuals (b) for each year in the Broken Riven sub region.

2.5 Results

Simulated pasture production values across the study area ranged from. 0.1 to 4.5 Mg ha^{-1} yr^{-1} although approximately 70% of the area produces between 1.5 and 3 Mg ha^{-1} yr^{-1}. Coastal areas were consistently more productive than inland reflecting the higher rainfall near the coast (Figure S15 and S16). Climate change effects on pasture production are negative under all scenarios and GCMs. Mean declines in production included 124 (CE2), 126 (MPI) and 74 (MR5) kg ha^{-1} yr^{-1} for the RCP 2.6 between 2013 and 2050. RCP 4.5 produced reductions of 161 (CE2), 163 (MPI) and 98 (MR5) kg ha^{-1} yr^{-1} while the worst case scenario RCP 8.5 resulted in 193 (CE2), 197 (MPI) and 121 (MR5) kg ha^{-1} yr^{-1} reductions (Figure S17).

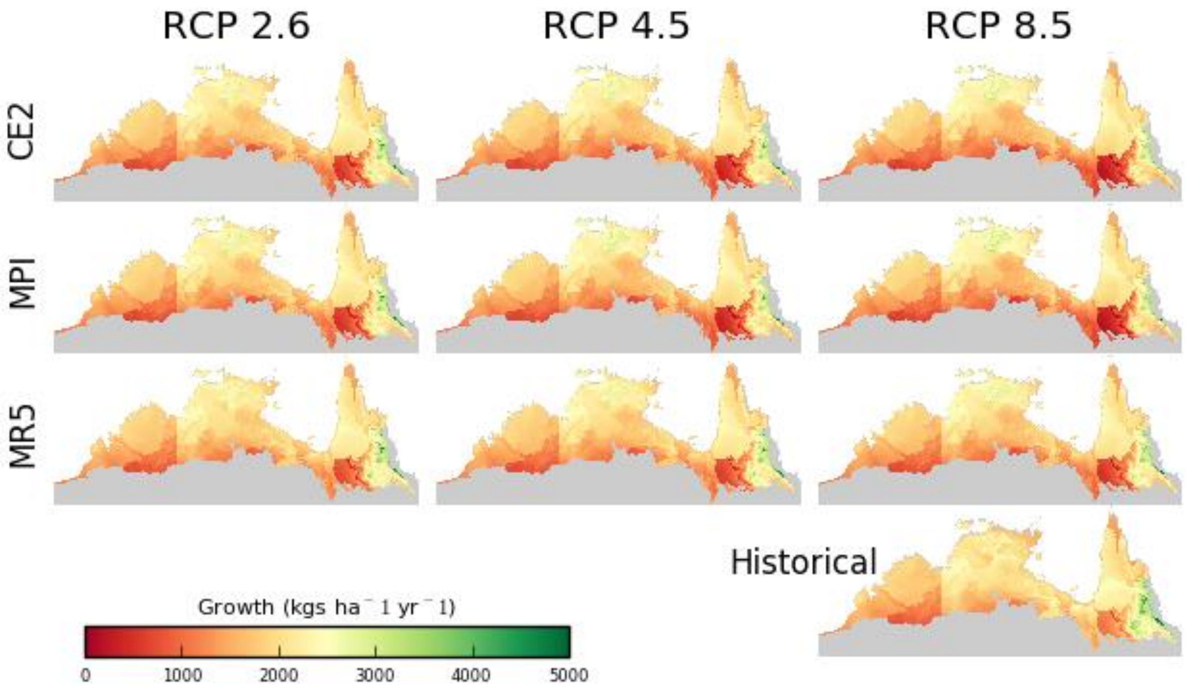


Figure S16 | Pasture growth (kg ha^{-1} yr^{-1}) under historical climate and each scenario and GCM in the year 2050.

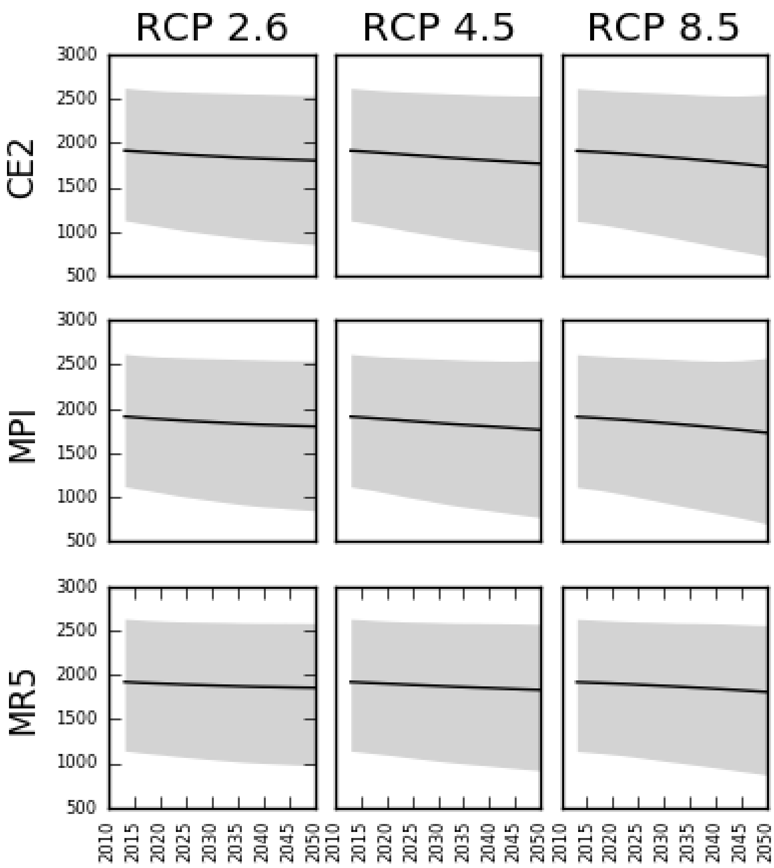


Figure S17 | Mean Pasture production (kg ha^{-1} yr^{-1}) across all locations for each scenario, GCM and future year with 5^{th} and 95^{th} percentile range in grey.

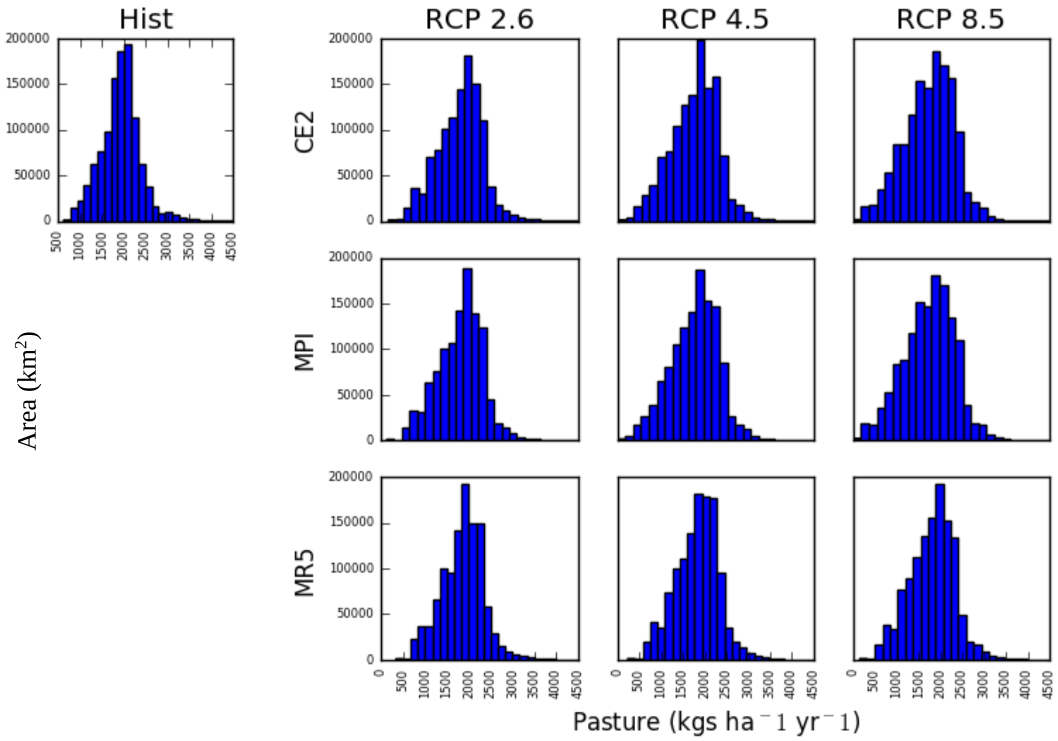


Figure S18 | Histograms of total area of pasture production rates (kg ha^{-1} yr^{-1}) under historic conditions and for each scenario and GCM at the year 2050.

3 Grazing and associated GHG emissions

3.1 Simulations of safe stocking rates

Our model aimed to simulate “safe” stocking rates / carrying capacity (the number of livestock that could be supported by the amount of simulated pasture growth in each year without adversely impacting land condition). It was designed so that we could model grazing under future climate (and economic) change. We assumed that the number of cattle could be varied from year to year in response to changing conditions. While this is a valid stocking strategy, there are constraints to its application in practice, as it can be challenging to rapidly increase of decrease stock numbers when managing a breeding herd in northern Australia (O’Reagain *et al* 2014). However, research results recommend applying flexible stocking rates to manage for climate variability (O’Reagain and Scanlan 2013). Adult equivalents per year were modelled from a combination of pasture growth, safe pasture utilisation rates, and pasture intake per animal (9 kg/day). Specifically, the safe stocking rate (adult equivalents per km^2) in each year was calculated using the following equation:

$$AE_{iy} = \frac{P_{iy} \times \underline{U}_i}{C} \tag{S10}$$

Where AE is the number of adult equivalents (~450 kg) in pixel *i* in year *y*, *P* is the annual amount of pasture growth (in kilograms) in pixel *i* in year *y*, $\underline{U}$ is the safe pasture utilisation rate for pixel *i*, and *C* is the amount of pasture consumed by an adult equivalent in a year (in kilograms). Northern Australia comprises many different pasture types which can each support different levels of grazing, so we applied an individual safe pasture utilisation rate (and variation for the sensitivity analysis) for each pasture type based on Tothill and Gilles (1992) (Figure S18 and S19, Table S7). The pasture consumption per adult equivalent was set at 9 kg per day (± 1 kg per day) based on a range of studies (Queensland Government Department of Agriculture Forestry and Fisheries (DAFF) 2013, Scanlan *et al* 1994, Pieper 1988, Holechek 1988, Walsh and Cowley 2011, Bernado 1989), and multiplied by 365 to give an annual value. We constrained the model to the broad area currently grazed by livestock to avoid unsuitable vegetation types, soils, or topographies, and ensure appropriate land tenure (primarily pastoral leasehold).

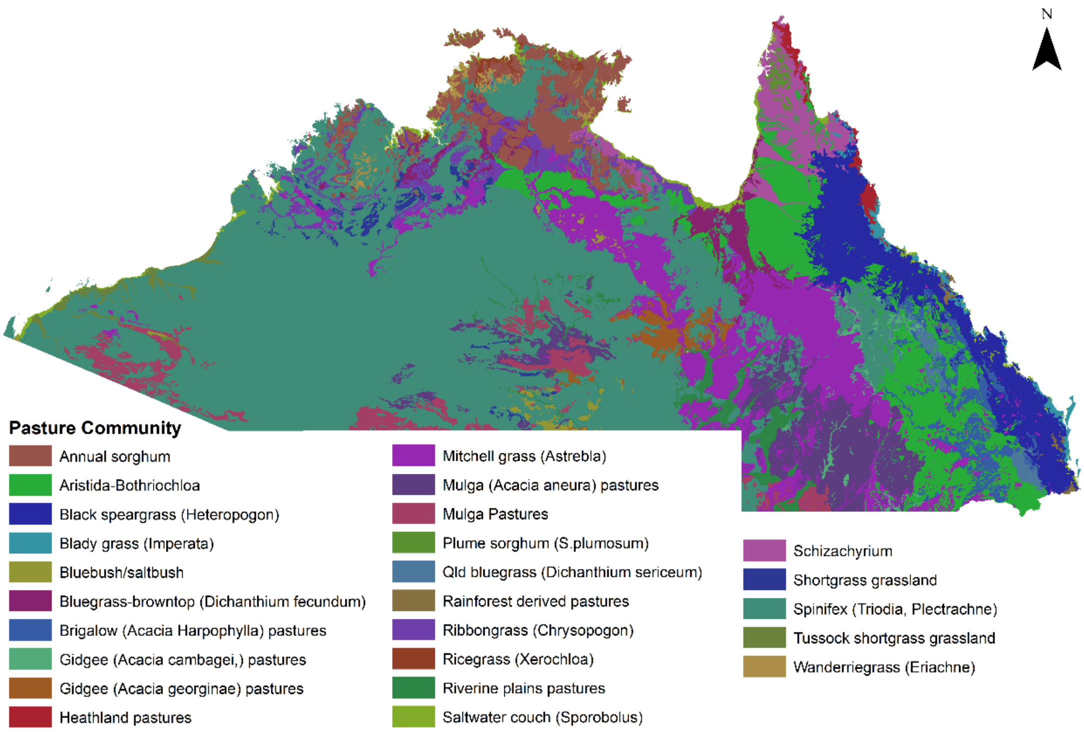


Figure S19 | Pasture land types in northern Australia (based on Tothill and Gillies (1992), data supplied by Javier Navarro)

Table S7 | Carrying capacity of all northern Australian pasture communities across each State. Unless otherwise stated, the mean, lower and upper bounds were from the mean, minimum, and maximum values (respectively) given in Tothill and Gillies (1992).

Pasture Community	*Head Km^{-2} lower*	*Head Km^{-2} mean*	*Head Km^{-2} upper*	*Notes/ Source*
		Queensland		
Aristida-Bothriochloa	2.5	3.8	9.1	Tothill and Gillies (1992)
Black speargrass (Heteropogon)	6.7	10.0	20.0	Tothill and Gillies (1992)
Blady grass (Imperata)	2.9	5.2	12.5	Tothill and Gillies (1992)
Bluegrass-browntop (Dichanthium fecundum)	6.3	7.1	8.3	Tothill and Gillies (1992)
Gidgee (Acacia cambagei) pastures	2.9	4.3	8.3	Tothill and Gillies (1992)
Heathland pastures	0.0	0.0	0.0	Cannot be grazed in natural state (Tothill and Gillies 1992)
Mitchell grass (Astrebla)	6.7	8.0	10.0	Tothill and Gillies (1992)
Plume sorghum (S.plumosum)	2.5	3.3	5.0	Tothill and Gillies (1992)
Ribbongrass (Chrysopogon)	2.5	3.8	8.3	Tothill and Gillies (1992)
Saltwater couch (Sporobolus)	3.3	4.0	5.0	Tothill and Gillies (1992)
Schizachyrium	2.0	2.7	5.0	Tothill and Gillies (1992)
Spinifex (Triodia, Plectrachne)	0.7	1.0	2.9	Tothill and Gillies (1992)
		Northern Territory		
Annual sorghum	1.0	1.3	1.9	Only one value given (Tothill and Gillies 1992), apply ± 30% for upper/lower bounds
Aristida-Bothriochloa	1.0	2.5	3.8	“Low” given in Tothill and Gillies (1992), applied 80th percentile (± 20) from all NT values
Bluebush/saltbush	2.0	2.2	2.5	Tothill and Gillies (1992)
Bluegrass-browntop (Dichanthium fecundum)	3.8	6.0	8.3	Tothill and Gillies (1992)
Mitchell grass (Astrebla)	2.0	4.3	8.3	Tothill and Gillies (1992)

Ribbongrass (Chrysopogon)	2.5	4.5	8.3	Tothill and Gillies (1992)
Ricegrass (Xerochloa)	0.0	0.7	1.7	Typically on saline mud soils and is generally unproductive. Upper bound equivalent to saltwater couch. The lowest bound of other grazed pastures (spinifex) was given as the mean. Lower bound is ungrazed.
Saltwater couch (Sporobolus)	1.3	1.7	2.4	Only one value given (Tothill and Gillies 1992), apply ± 30% for upper/lower bounds
Schizachyrium	1.4	2.6	3.8	"Low" given in Tothill and Gillies (1992), applied 75th percentile (± 20) from all NT values. This supports similar, but slightly less head km^{-2} then the same pasture type in QLD.
Shortgrass grassland	2.0	3.6	6.7	Tothill and Gillies (1992)
Spinifex (Triodia, Plectrachne)	0.7	0.9	1.7	Applied the average between WA and QLD
Wanderriegrass (Eriachne)	3.0	3.8	5.5	Only one value given (Tothill and Gillies 1992), apply ± 30% for upper/lower bounds
Western Australia				
Annual sorghum	1.5	1.9	2.5	Tothill and Gillies (1992)
Bluegrass-browntop (Dichanthium fecundum)	4.0	4.4	5.0	Tothill and Gillies (1992)
Mitchell grass (Astrebla)	6.7	8.0	10.0	Only one value given (Tothill and Gillies 1992), applied QLD values
Ribbongrass (Chrysopogon)	1.9	3.5	8.3	Tothill and Gillies (1992)
Ricegrass (Xerochloa)	0.0	0.7	1.7	No values given in Tothill and Gillies (1992), but area is on the NT border, so NT values were applied
Saltwater couch (Sporobolus)	1.3	1.7	2.4	Only one value given (Tothill and Gillies 1992), apply ± 30% for upper/lower bounds
Shortgrass grassland	1.5	2.5	6.3	Tothill and Gillies (1992)
Spinifex (Triodia, Plectrachne)	0.7	0.9	1.3	Tothill and Gillies (1992)
Tussock shortgrass grassland	7.7	10.0	14.3	Only one value given (Tothill and Gillies 1992), apply ± 30% for upper/lower bounds
Wanderriegrass (Eriachne)	1.0	1.3	1.9	Only one value given (Tothill and Gillies 1992), apply ± 30% for upper/lower bounds

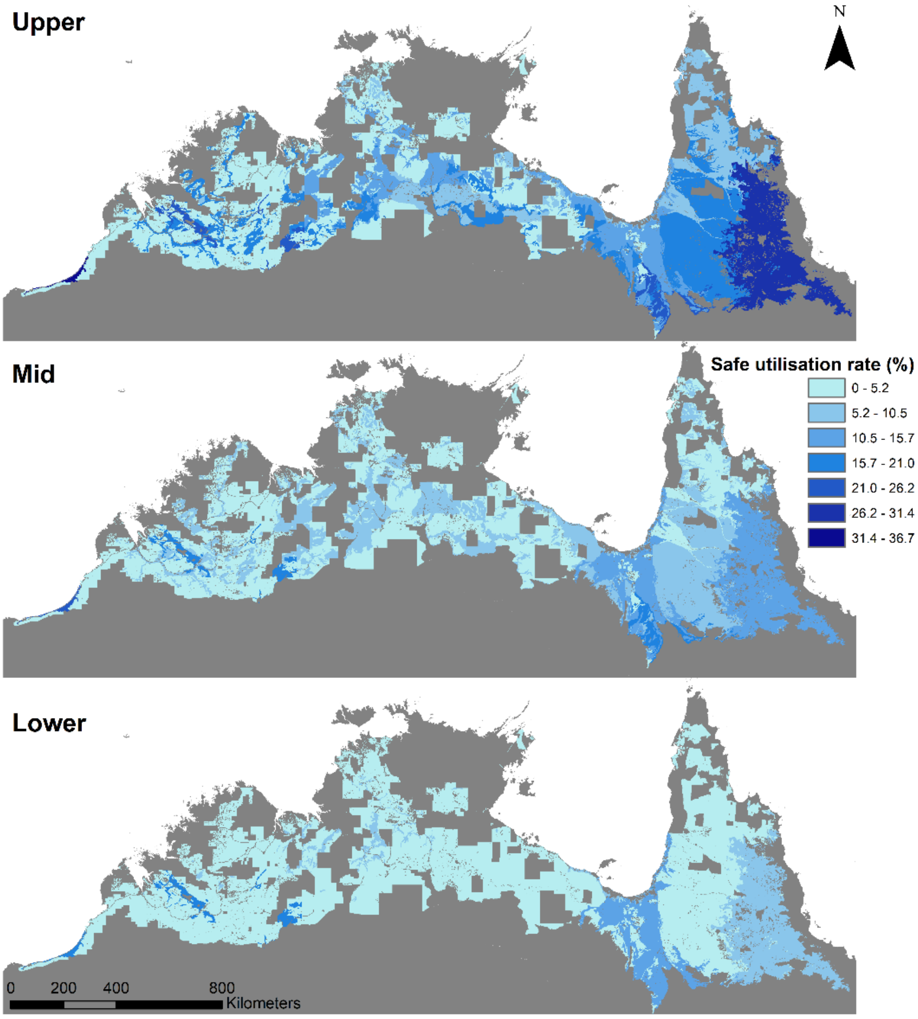


Figure S20 | The spatial variation in safe utilisation rates, including the upper and lower bounds used to inform the sensitivity analysis.

3.2 Calculating baseline stocking levels

To simulate a continuation of the baseline stocking level, we also included a spatial approximation of these stocking rates by adapting three existing data sources. A map of baseline stocking levels was adapted from stocking rate maps produced by the Queensland Department of the Environment and Resource Management (Carter *et al* 1996, 2003), which considered location-specific factors such as land use, pasture type, pasture growth rate, presence of noxious weeds and predators, and topography. Stocking rates were modified for beef cattle, dairy cattle and sheep by combining them with livestock numbers at the Statistical Local Area level from the 2010/11 agricultural census, and restricted their spatial extent to match that of the 2005 Australian Land Use Map (Navarro *et al* 2016). These livestock numbers were given in DSE (dry sheep equivalents), which were converted to adult equivalents (per km^2). Each broadacre region had a different typical herd structure, so the conversion to adult equivalents were specific to each region based on modelling using *Breedcow* software (Navarro *et al* 2016, Queensland Government Department of Agriculture Forestry and Fisheries (DAFF) 2013). For pixels where adult equivalents were above the 95th percentile, focal statistics were applied (taking median of a 5x5 km window), to avoid unrealistically high values (see Supplementary Information for a spatial comparison of the historical and simulated safe stocking rates).

3.3 Comparison

Table S8 | Comparison of summary statistics between "Our model" (simulations of historical safe stocking rates) and census data for historical stocking rates.

	Our model	***Census***
Mean	3.85	3.95
Median	2.91	3.46
Max	24.94	791.92
95th	10.41	8.20
5th	0.73	1.15
Min	0.00	0.00
Sum	2,658,099	2,726,938

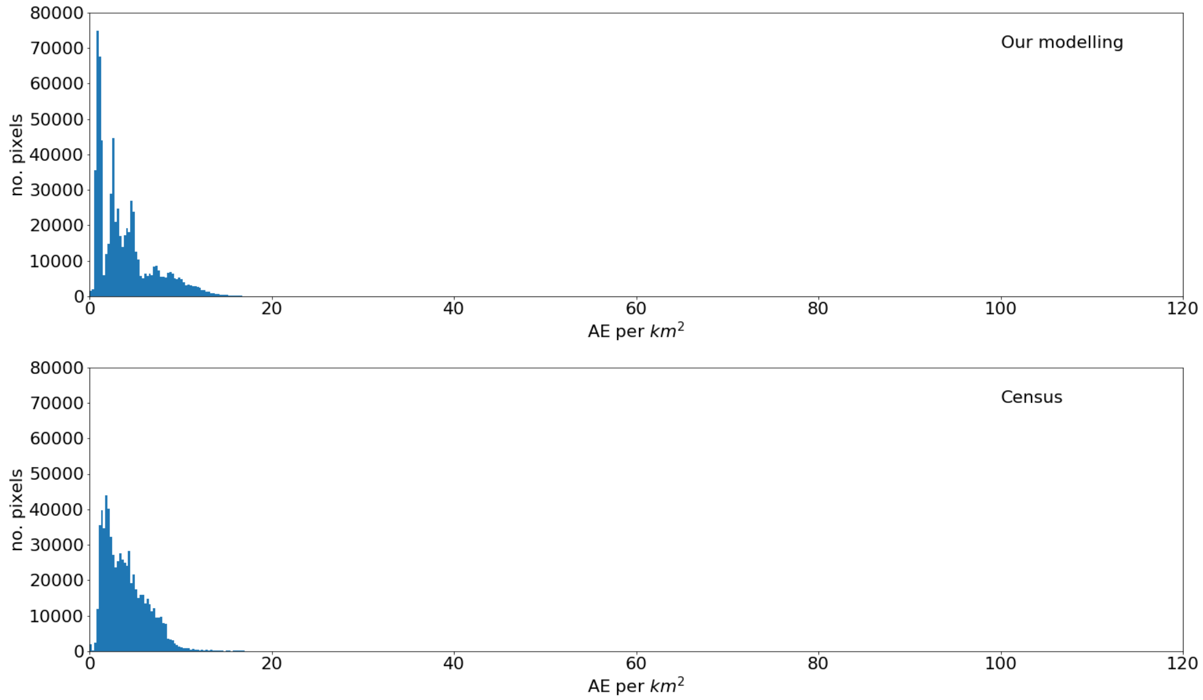


Figure S21 | Frequency histograms comparing "Our model" (simulations of historical safe stocking rates) and census data for historic stocking rates.

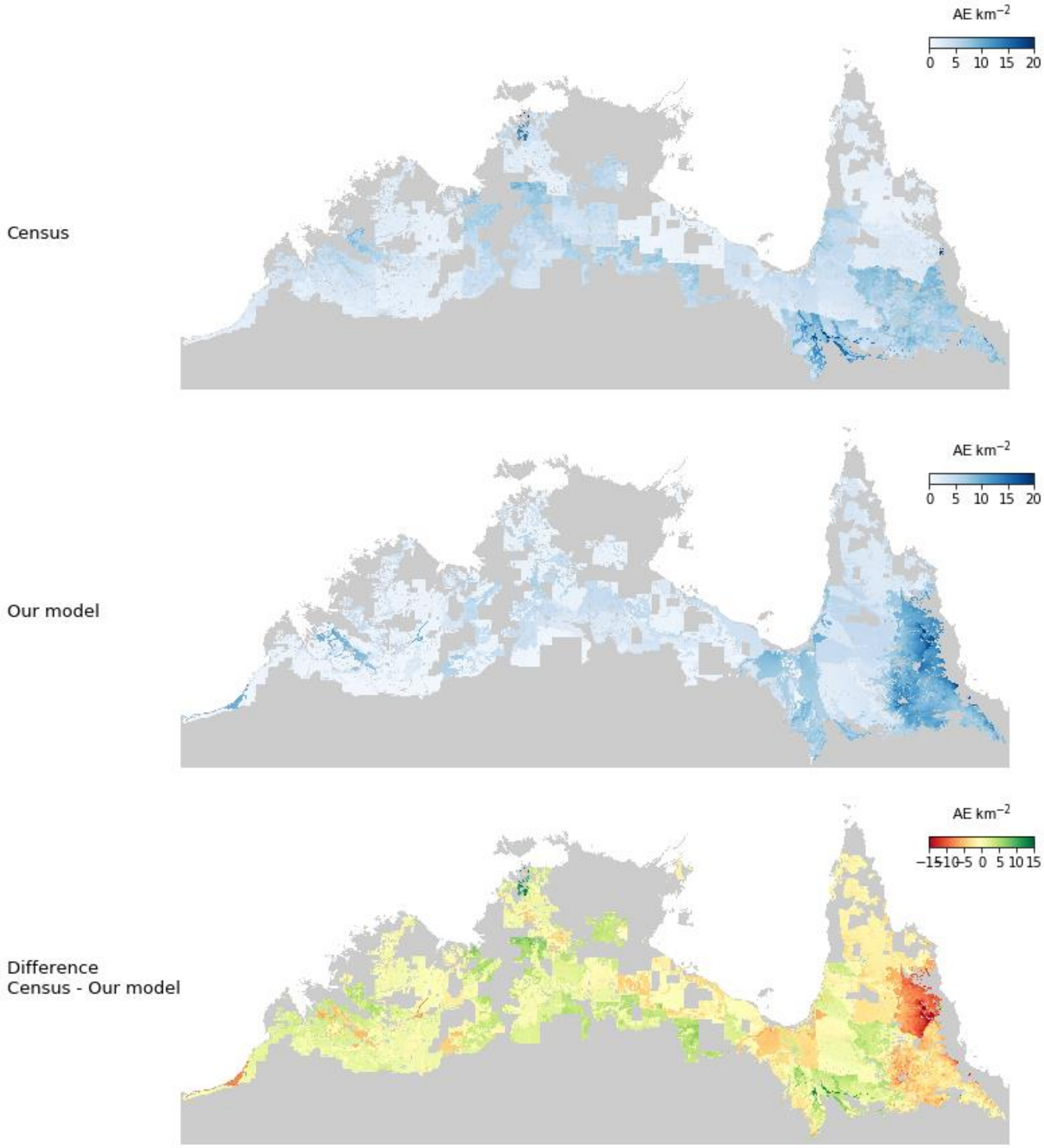


Figure S22 | Spatial comparison between historic stocking rates (census) and simulated 'safe' stocking rates (our model). These differences can be due to historic over/under stocking, changed land use, rotating cattle (during census), and generalisations in our model.

3.4 Livestock GHG emissions

Livestock also produce GHG emissions, primarily from enteric fermentation (microbial action in the digestive system) (Cottle et al 2011). GHG emissions per head were calculated in a similar way to profitability: the mean (± the standard deviation) biogenic GHG emissions per head of beef cattle were taken from time-series data (1997-2013) for each Australian broadacre region (Navarro et al 2016), and converted to emissions per adult equivalents (Table S10). These beef cattle biogenic emissions were calculated by applying the data on total head and herd structure into the Greenhouse

Gas Accounting Framework (Navarro et al 2016, Eckard et al 2008). Whilst this analysis does not capture greenhouse gas emissions from farm operations, these additional sources are considered to be relatively minor in extensive grazing systems relative to biogenic emissions (Steinfeld and Wassenaar 2007).

3.5 Supplementation

There is potential to reduce biogenic emissions from cattle without impacting livestock production, but this comes with additional costs (Grainger and Beauchemin 2011). Emerging research has demonstrated that methane emissions from cattle can be virtually eliminated by supplementing livestock feed with red macroalgae (*Asparagopsis taxiformis*) (Kinley *et al* 2016, 2020). Methane emissions were reduced by >99% in a laboratory setting (Kinley *et al* 2016) and up to 98% in a feedlot setting (Kinley *et al* 2020). While there is potential to supplement extensively grazed cattle macroalgae using lick blocks (Tomkins and Kinley 2015, Machado *et al* 2018), this is unlikely to achieve the reductions seen in feedlots, due to highly variable intake overtime and between individuals (Ridoutt *et al* 2022). Further, a field study on methane reduction from calcium nitrate molasses lick blocks in an Australian extensive grazing system found no difference in methane emitted between the control group, while the calcium nitrate molasses lick blocks resulted in lower liveweight gain and poorer body condition scores, due to poor uptake of the supplement (Callaghan *et al* 2021).

To reflect the uncertainty in this management action, we assumed a large range in potential methane reduction from macroalgae supplementation via lick blocks. For an upper estimate, we took the lowest value from Roque et al (2021), a 36.3% reduction, representing a lower macroalgae dose in a high forage feedlot mix. Translating results from the feedlot to an extensive grazing scenario is ambitious, even for the lower range of results. Zero was set as the lower value to represent a poor uptake scenario as seen with calcium nitrate molasses lick blocks in Callaghan et al (2021), with the midpoint between these two values used in the main runs. This intervention is also likely to be more costly than supplementing with calcium nitrate molasses lick blocks, so we applied a multiplier of 1.5 to the cost of nitrate supplementation (an additional $0.255 per animal per day) and varied this between 1 and 2 in the sensitivity analysis. The total factor productivity was also applied here to reflect potential increases in methane reduction, and reduced costs. The additional cost was subtracted from the profit per animal (equation S10).

3.6 Modified Pastures

Productivity can be increased by exotic pastures, which are not currently utilised across much of northern Australia. The method that is most likely to be feasible in the north involves the aerial sowing of seed by helicopter or light aircraft, where most of the property is oversown with legumes (e.g., stylo (*Stylosanthes* spp.)), which likely carries a once-off cost of \$45 ha^{-1} (Andrew Ash, pers. comm. 26 March 2018). This cost was annualised over the period from 2013 – 2050 at a 5% discount rate giving a cost of \$2.70 ha^{-1} yr^{-1} for use in the integrated assessment model. For the sensitivity analysis, a lower bound of \$25 ha^{-1} was applied based on a case study near Charters Towers, Queensland (Hunt *et al* 2013) (annualised to \$1.50 ha^{-1} yr^{-1}). An upper bound of \$120 ha^{-1} (\$7.20 ha^{-1} yr^{-1} annualised) was included to represent to represent cases were poor conditions necessitated a second sowing over much of the area. The safe stocking rate was increased to represent the higher carrying capacity achieved by the additional forage available (Hunt *et al* 2013). In addition, revenue was increased per adult equivalent to represent faster liveweight gain (and higher turnoff) due to the lower seasonal decline in forage (Hunt *et al* 2013). The faster liveweight gain also reduced the Mg of CO_2e per adult equivalent as this meant fewer years until turnoff and a lower emissions intensity (Hunt *et al* 2013). Values for stocking rate increase, revenue increase, and methane reduction varied for each broadacre region across the north (Table S9). The values were taken form the most relevant regional case studies in Hunt et al (Hunt *et al* 2013).

Table S9 | Variation in the safe stocking rate increase, revenue increase and methane decrease with modified pastures (oversowing with legumes).

Broadacre Region*	**AE % increase**	**Gross margin AE^{-1} % increase**	**Mg CO_2e AE^{-1} % decrease**	**Notes - Hunt et al (2013) case study region and variation**
QLD: Cape York and the QLD Gulf, West and South West	0.095 (±0.0475)	0.1095 (±0.0547)	0.0645 (±0.0323)	All from Barkly-NW Queensland ± 50%
QLD: Central North	0.198 (±0.099)	0.224 (±0.112)	0.107 (±0.0535)	All from Northern Queensland ± 50%
WA: The Kimberly	0.1949 (±0.0974)	0.18 (-0.09, +0.6)	0.0897 (-0.0448, +0.1449)	Modelled values from the Kimerley were unexpectedly high, so values from the adjacent Pilbra region were used instead. All main values were taken from the Pilbra region, % all lower bounds set at -50%. For AE, the upper value was +50%. Values from the Kimberley were taken as the upper bound for GM & Mg CO_2e.
NT: Barkly Tablelands, Victoria River District – Katherine, Top End Darwin and the Gulf of NT	0.158 (±0.079)	0.1922 (±0.1405)	0.08685 (±0.05685)	For the main values the mean was taken from 2 proximal case studies – the Victoria River District and Central Australia. For AE, the bounds were set at ± 50%. For GM & Mg

CO_2e, the value from Central Australia represented the lower bounds and the value from the Victoria River District represented the upper bound.

*QLD = Queensland, WA = Western Australia, NT = Northern Territory. AE = adult equivalents.

4 Profit

First, we created a baseline of the potential profit from safe stocking rates using historic (1997-2013) time series data for each Australian broadacre region in our study area (Navarro *et al* 2016). Time series data (including revenue, costs, cattle heads and herd structures) was compiled from ABARES Farm Survey data on specialist beef farms (ABARES 2015), and values with high relative standard error (> 0.9) were discarded. We calculated the mean (± the standard deviation) of revenue and costs per head of cattle for each region and converted these to a value per adult equivalent (stocked, Table S10) using regionally specific conversion values. The range of gross margin values used here also encompass the range given by other sources of financial information for the northern beef sector (e.g. Chilcott et al. (2020)).

Table S10 | The baseline revenue, costs and greenhouse gas emissions per AE from beef cattle for each broadacre region in northern Australia.

Broadacre Region*	Revenue AE^{-1}	Costs AE^{-1}	Mg CO_2e AE^{-1}
QLD: Cape York and the QLD Gulf	$98.98 (±33.43)	$45.11 (± 14.60)	1.72 (± 0.78)
QLD: West and South West	$225.98 (± 58.88)	$101.38 (± 38.27)	2.49 (± 0.84)
QLD: Central North	$157.20 (± 59.92)	$65.99 (± 24.37)	2.04 (± 0.64)
WA: The Kimberly	$130.29 (± 68.35)	$56.22 (± 28.03)	1.55 (± 0.66)
NT: Barkly Tablelands	$123.58 (±52.47)	$73.66 (± 36.49)	2.21 (± 0.91)
NT: Victoria River District - Katherine	$125.20 (± 64.37)	$60.81 (± 22.30)	2.26 (± .0.93)
NT: Top End Darwin and the Gulf of NT	$166.08 (± 57.91)	$98.66 (± 25.27)	2.11 (± 0.93)

*QLD = Queensland, WA = Western Australia, NT = Northern Territory. AE = adult equivalents.

The economic outlook for livestock production could change in the future due to technological innovation and changes in livestock demand and costs of production. To calculate the potential change in profit, the projected changes in livestock price for each global outlook (from Hatfield-Dodds et al. (2015a)) were applied to the baseline revenues. We used the projected changes in oil price as a proxy for trends in the cost of farm inputs, due to the energy intensive inputs (Bryan *et al* 2014, 2015), and applied these to the baseline costs. We also increased yields by the total factor productivity in each year to 2050. For the main analysis, this was set at 1% representing the average increase in northern Australia beef production between 1977-78 to 2006-07 (Nossal *et al* 2008). In the sensitivity analysis, the total factor productivity was varied between 0% (no growth, a pessimistic scenario) and 2% (a scenario representing accelerated investment in northern Australia). The profit was calculated for each global outlook and GCM combination (with upper and lower extrema) using the equation:

$$PF_{iy} = AE_{iy}P_{iy}(1 + \Delta P_y)(1 + TFP_y) - AE_{iy}C_{iy}(1 + \Delta C_y) \qquad \text{(S10)}$$

Where PF_{iy} is the profit (or loss) for pixel *i* in year *y*, AE_{iy} is the number of adult equivalents in pixel *i* in year *y*, P_{iy} and C_{iy} represent the price and costs for an adult equivalent for pixel *i* and year *y* respectively, ΔP_y and ΔC_y are the changes in livestock price and oil price, and TFP_y is the total factor productivity increase.

4.1 Carbon price

For global outlooks that include a carbon price (L1, M2, M3), payments for reductions in GHG emissions could also contribute to profits. The calculation of profit remained the same for ‘baseline stocking’ (equation S10) as there was no emissions abatement. However, the equations for other management actions changed. For the safe stocking management action, in pixels where the safe stocking rate was less than the baseline stocking rate, additional revenue from emissions abatement was calculated as:

$$CR_{iy} = \begin{cases} (CAE_i - SAE_{iy})E_iCP_y & \text{if } CAE_{iy} > SAE_{iy} \\ 0 & \text{if } CAE_{iy} \leq SAE_{iy} \end{cases} \qquad \text{(S11)}$$

Where CR_{iy} is the additional revenue from carbon pricing, CAE_i is the number of historical adult equivalents in pixel *i*, SAE_i is the simulated safe number of adult equivalents in pixel *i* for year *y*, E_i is the biogenic GHG emissions per adult equivalent in pixel *i*, and CP_y is the carbon price in year *y*. For supplementation with macroalgae, the equation was:

$$SPF_{iy} = AE_{iy}P_{iy}(1 + \Delta P_y)(1 + TFP_y) - AE_{iy}\big(C_{iy} + (SC(1 - TFP_y)\big)(1 + \Delta C_y) + AE_{iy}CP_yER \qquad \text{(S12)}$$

Where SPF_{iy} is the profit from historical stocking with supplementation for pixel *i* in year *y*, *SC* is the additional annual cost of supplementation compared to urea per animal, *ER* is the emissions reduction from supplementation per animal, and CP_y is the carbon price in year *y*. All other parameters are as per equation S10. The cost of macroalgae supplementation was reduced overtime in line with the total factor productivity to account for future innovation in this area. The potential profit from destocking was calculated as:

$$DPF_{iy} = AE_{iy}CP_yE_i \qquad \text{(S13)}$$

Where DPF_{iy} is the profit from destocking for pixel i in year y, E_i is the biogenic GHG emissions per animal in pixel i, and the remaining parameters are as above. The profit from prescribed burning was calculated as:

$$BPF_{iy} = ER_{iy}CP_y - BC(1 + \Delta C_y) \tag{S14}$$

Where BPF_{iy} is the profit from prescribed burning for pixel i in year y, ER_{iy} is the emission reductions (in Mg of CO_2e) from prescribed burning in pixel i in year y, and BC is the cost of conducting a prescribed burn, which was set at an initial value of \$0.4685 ha^{-1} (± 30%), based on data from Heckbert et al. (2012). The change in oil price ΔC is also used here as a proxy for the trends in farm costs. Where multiple actions were undertaken simultaneously, these costs and emissions reductions were summed. Together, this allowed a comparison of GHG emissions and profits for each of the management combinations under a range of carbon prices.

5 Overgrazing and Land Condition

While it is well-established that overgrazing leads to land degradation, the exact functional form in northern Australia is unknown (McIvor 2010). Here we developed a function with a threshold effect linking pasture utilisation to land condition with different forms. For our main analyses we assumed a linear function with a threshold effect:

$$D_{iy} = \begin{cases} 0 & \text{if } U_{iy} \leq \underline{U}_{iy} \\ \dfrac{U_{iy} - \underline{U}_{iy}}{1 - \underline{U}_{iy}} & \text{if } U_{iy} > \underline{U}_{iy} \end{cases} \tag{S15}$$

Where D_{iy} is the land degradation [0,1] in pixel i for year y, $\underline{U}_{iy}$ is the safe pasture utilisation rate in pixel i for year y, and U_{iy} is the actual pasture utilisation rate in pixel i for year y, calculated as:

$$U_{iy} = \frac{AE_{iy}C}{G_{iy}} \tag{S16}$$

Were AE_{iy} is the number of adult equivalents in pixel i for year y, G_{iy} is the annual amount of pasture growth (kg) in pixel i for year y, and C is the amount of pasture consumed by an adult equivalent in a year (kg). However, the response to overgrazing may not always be linear, so we also evaluated concave and convex functional forms, each with a threshold effect, in the sensitivity analysis:

$$D_{iy} = \begin{cases} 0 & \text{if } U_{iy} \leq \underline{U}_{iy} \\ \dfrac{e^{U_{iy}z} - e^{\underline{U}_{iy}z}}{e^{z} - e^{\underline{U}_{iy}z}} & \text{if } U_{iy} > \underline{U}_{iy} \end{cases} \tag{S17}$$

Where z was varied between -2.5 and 2.5 and all other parameters are as above. Altering the safe utilisation rates would also alter the land degradation index (Figure S22). Note that the land degradation index depends only on direct pasture utilisation by livestock, and does not incorporate interactions with wildfire or prescribed burning.

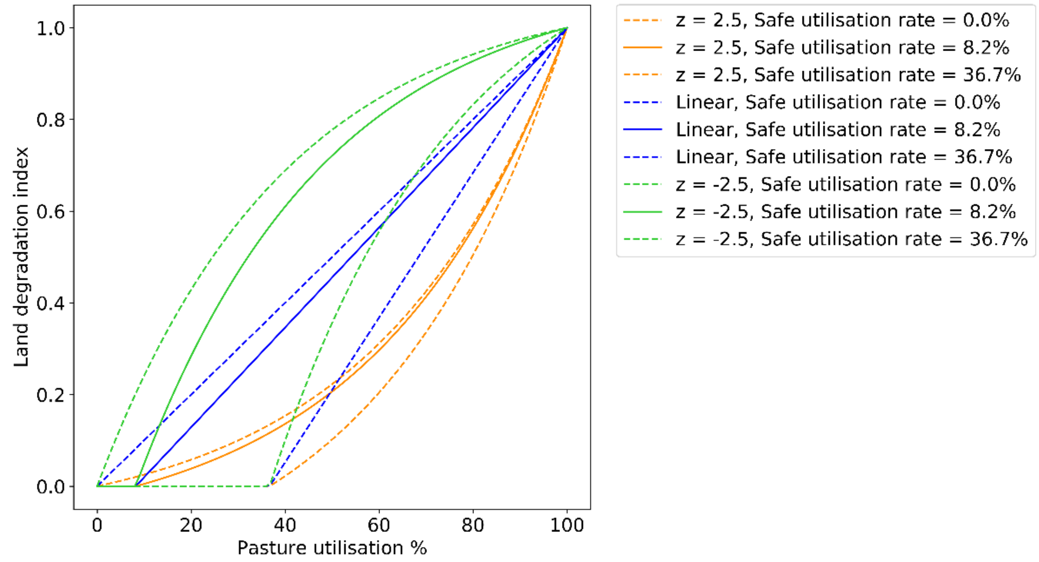


Figure S23 | Land degradation in response to pasture utilisation and variation in the safe utilisation rates. The utilisation rates used for illustration here are the mean (8.2%), maximum (36.7%) and minimum (0% - i.e. the pasture is not safe to graze at any level) across all pasture types and variations. This encompasses cases where exceeding the safe pasture utilisation rate degrades the land more (green) or less (orange) relative to a linear function (blue). In all cases, the land was not degraded if the pasture utilisation rate remained below the safe level for any given pixel.

To account for the impacts of overgrazing on liveweight gain and profit, we produced a linear function from data in Reagain et al (2014). Specifically, we used the negative linear trend between stocking rate (once above carrying capacity) and liveweight gain and annual returns (Fig 1, Reagain et al 2014), and built a linear model in R using the *lm* function. This model was then applied in pixels where overgrazing was occurring, to reduce the profit and liveweight gain (reflected in total AE) accordingly. For liveweight gain, this took the form of:

$$AE\ Impact = 1.65573 - 0.62237\left(\frac{current\ stocking\ rate}{safe\ stocking\ rate}\right)x \qquad \text{S18}$$

and for profit:

$$Profit\ Impact = 1.6931 - 0.6994\left(\frac{current\ stocking\ rate}{safe\ stocking\ rate}\right)x \qquad \text{S19}$$

This returns the relative amount of AE and profit compared to the safe stocking rate. In both cases, x was used as a modifier in the sensitivity analysis (1 in main runs, and varied between 0.85 and 1.15 for the sensitivity analysis). In all cases the metric was capped at 1 such that overgrazing did not improve profit and liveweight gain beyond the peak carrying capacity. The form of these relationships, and how they relate to the original data, is shown in Fig S23. The values from eqn.

S18 and S19 were applied to the total AE and profit in the integrated model (where overgrazing was occurring).

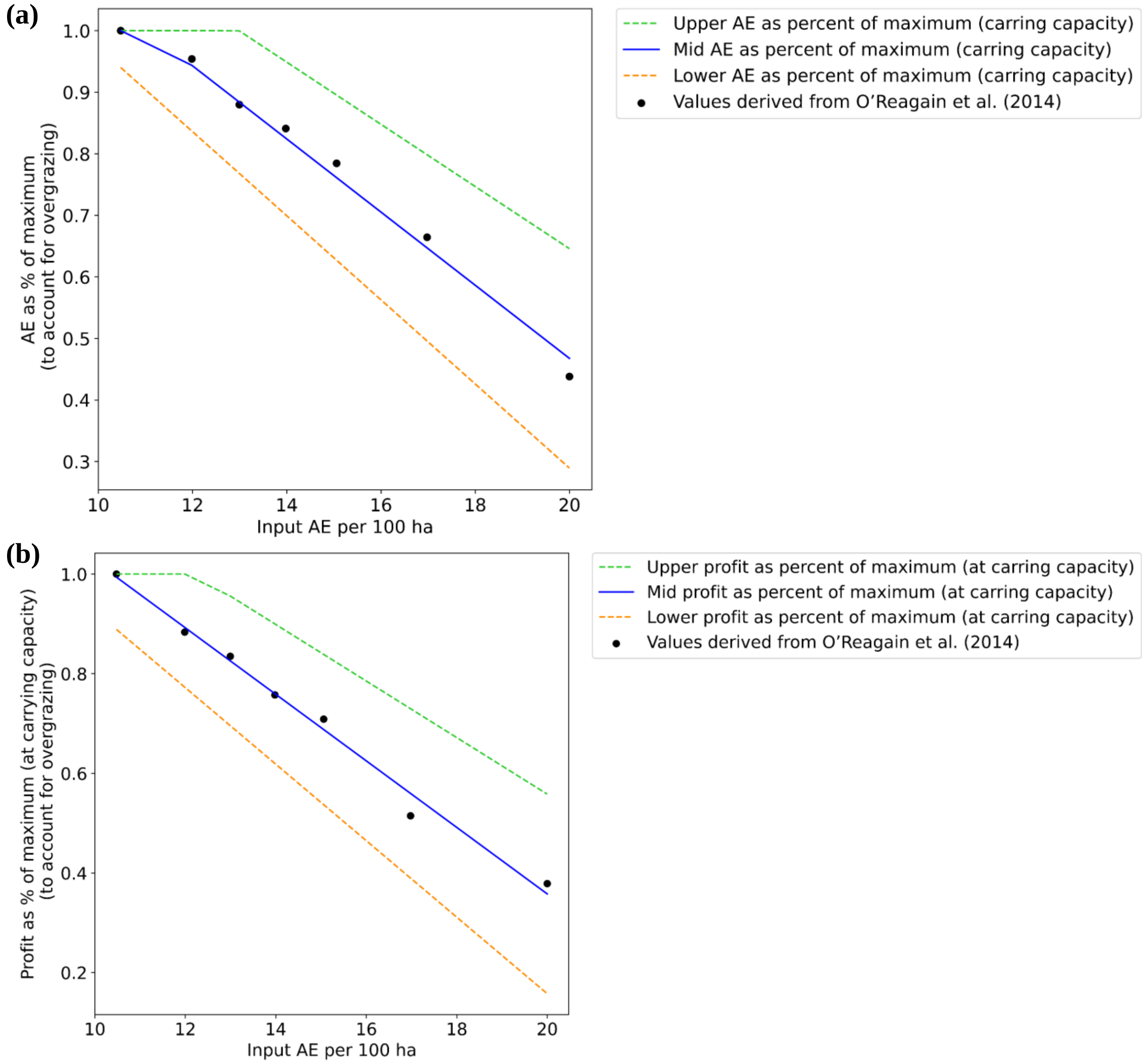


Figure S24 | Modification to AE (a) and profit (b) based on the degree of overgrazing. The values shown here are the range from the peak carrying capacity used by Reagain et al (2014). However, the input values in the integrated model would differ based on the given carrying capacity for each pixel.

6 Biodiversity

To account for the impact of climate change on biodiversity, we used species distributions for vertebrates (mammals, birds, reptiles, amphibians) under each RCP, averaged across 18 GCMs (Graham *et al* 2019). We included 609 species (43 amphibians, 286 birds, 93 mammals and 187 reptiles (Table S11) that were located in our study region. These species distributions include suitable bio-climatic envelopes (which includes the probability of presence), but do not consider the limitation of species dispersal. To account for the realities of species dispersal, we applied taxa-specific dispersal kernels (4km yr^{-1} for mammals and birds, 0.5km yr^{-1} for reptiles and amphibians) (Reside *et al* 2017).

To determine the impact of each management type on biodiversity we used data from an expert elicitation on threats facing northern Australian species (Alvarez-Romero *et al* 2021). This study used the 4-point estimation method, taking the 'best guess', upper bound, lower bound and confidence for each threat, threat level (1-3), and species functional group (see Table S12 for a list of species and their groups). This information was used to quantify the best guess, upper and lower bound for each threat, threat level, and species group combination at a 90% confidence interval using the formula from McBride et al (McBride *et al* 2012):

$$lower = \gamma - (\gamma - \alpha) * (c/p) \qquad \text{S20}$$

$$upper = \gamma + (\beta - \gamma) * (c/p) \qquad \text{S21}$$

Where γ is the expert's best guess, α is their lower bound, β is their upper bound, p is the expert's stated confidence and c is the required possibility level (here 90%). The mean best guess, upper and lower bound was taken across all experts.

We combined a subset of the information from the expert elicitation with bio-physical information from our study to calculate the impact on each species from fire management, the level of grazing, and presence of modified pastures (over-sowing of legumes). Mean fire events (0-1) from the fire event simulations (see above Fire modelling section) were converted to fire return intervals using the equation:

$$\text{FireReturnInterval} = 1/\text{MeanFireEvents} \qquad \text{S22}$$

To align with the threat levels from the expert elicitation (Alvarez-Romero *et al* 2021), the fire return interval was converted to one of the three threat levels in accordance with Table S11. As the simulated fire events changed overtime with changing temperature and rainfall, these threat levels were re-calculated in each year of the integrated assessment model. Applying prescribed burning as a management action reduced the simulated fire frequency by 34% (ranging from 25% to 48% in the sensitivity analysis) (Russell-Smith *et al* 2009a, 2013).

Table S11 | How each threat level from the expert elicitation (Alvarez-Romero *et al* 2021) was assigned to different categories of fire return intervals. Higher numbers indicate a greater threat level.

Fire return interval	**Threat level**
≥ 3.5 years	1
< 3.5 & > 1.5 years	2
≤ 1.5 years	3

The threat level for grazing was set relative to the simulated safe stocking rate in each year and scenario. If grazing was present, but below the level of safe stocking, a threat level of 1 was applied, as even if grazing is on native pastures and not degrading the land condition, it can still impact some species groups. If grazing occurred at the threshold of safe stocking a threat level of 2 was applied, and where grazing exceeded the threshold of safe stocking, a threat level of 3. Modified pastures was given an additional threat level of 2 for 'shrub-trees' as over-sowing of legumes would mean the introduced species would be common and widespread in the application area, but the pressure from grazing would suppress many shrubs from becoming fully established.
Supplementation via lick blocks does not impact biodiversity, as they do not increase the number of stock relative to urea lick blocks, so no calculations were included here.

Using eqns. S20 and S21, this gave us a multiplier (0-1) for the impact of each management action for each pixel, which also allowed us to determine the impact where management actions were combined (e.g., prescribed fire and grazing). The biodiversity index presented in the main text is the habitat quality adjusted species richness, calculated as:

$$SR_{iy} = \sum_{x=1}^{n} \left(Pp_{xiy} \prod_{z=1}^{z} Th_{xiyz} \right) \qquad \text{(S23)}$$

Where SR_{iy} is the species richness is pixel *i* for year *y*, Pp_{xiy} is the probability of presence of species *x* in pixel *i* for year *y*, and Th_{xiyz} is the impact of threat *z* on species *x* in pixel *i* for year *y*.

Table S12 | Species and groupings included in the study. Code refers to the code given for each functional group.

Scientific	**Group**	**Code**
Amphibians		
Litoria coplandi	Rock dwellers	A01
Litoria meiriana	Rock dwellers	A01
Litoria wilcoxii	Rock dwellers	A01
Crinia bilingua	Seasonal burrowers	A02
Crinia deserticola	Seasonal burrowers	A02
Cyclorana alboguttata	Seasonal burrowers	A02
Cyclorana australis	Seasonal burrowers	A02

Cyclorana brevipes	Seasonal burrowers	A02
Cyclorana cryptotis	Seasonal burrowers	A02
Cyclorana cultripes	Seasonal burrowers	A02
Cyclorana longipes	Seasonal burrowers	A02
Cyclorana maculosa	Seasonal burrowers	A02
Cyclorana novaehollandiae	Seasonal burrowers	A02
Limnodynastes convexiusculus	Seasonal burrowers	A02
Limnodynastes depressus	Seasonal burrowers	A02
Limnodynastes lignarius	Seasonal burrowers	A02
Limnodynastes terraereginae	Seasonal burrowers	A02
Notaden melanoscaphus	Seasonal burrowers	A02
Notaden nichollsi	Seasonal burrowers	A02
Uperoleia altissima	Seasonal burrowers	A02
Uperoleia borealis	Seasonal burrowers	A02
Uperoleia inundata	Seasonal burrowers	A02
Uperoleia lithomoda	Seasonal burrowers	A02
Uperoleia littlejohni	Seasonal burrowers	A02
Uperoleia mimula	Seasonal burrowers	A02
Uperoleia mjobergii	Seasonal burrowers	A02
Uperoleia trachyderma	Seasonal burrowers	A02
Litoria caerulea	Tree frogs	A03
Litoria gracilenta	Tree frogs	A03
Litoria rothii	Tree frogs	A03
Litoria rubella	Tree frogs	A03
Litoria splendida	Tree frogs	A03
Litoria bicolor	Wetland frogs	A04
Litoria dahlii	Wetland frogs	A04
Litoria fallax	Wetland frogs	A04
Litoria inermis	Wetland frogs	A04
Litoria latopalmata	Wetland frogs	A04
Litoria microbelos	Wetland frogs	A04
Litoria nasuta	Wetland frogs	A04
Litoria pallida	Wetland frogs	A04
Litoria personata	Wetland frogs	A04
Litoria tornieri	Wetland frogs	A04
Litoria wotjulumensis	Wetland frogs	A04
Birds		
Aprosmictus erythropterus	Cockatoos and parrots	B01
Barnardius zonarius	Cockatoos and parrots	B01
Cacatua galerita	Cockatoos and parrots	B01
Cacatua sanguinea	Cockatoos and parrots	B01
Calyptorhynchus banksii	Cockatoos and parrots	B01
Lophochroa leadbeateri	Cockatoos and parrots	B01
Platycercus adscitus	Cockatoos and parrots	B01
Platycercus venustus	Cockatoos and parrots	B01
Polytelis alexandrae	Cockatoos and parrots	B01
Psephotus dissimilis	Cockatoos and parrots	B01
Psitteuteles versicolor	Cockatoos and parrots	B01
Trichoglossus chlorolepidotus	Cockatoos and parrots	B01
Trichoglossus haematodus	Cockatoos and parrots	B01
Eolophus roseicapillus	Galah, cockatiel, budgerigar and crested pigeon	B02
Melopsittacus undulatus	Galah, cockatiel, budgerigar and crested pigeon	B02
Nymphicus hollandicus	Galah, cockatiel, budgerigar and crested pigeon	B02
Ocyphaps lophotes	Galah, cockatiel, budgerigar and crested pigeon	B02
Coturnix chinensis	Doves, pigeons, finches and quails	B03
Coturnix pectoralis	Doves, pigeons, finches and quails	B03
Coturnix ypsilophora	Doves, pigeons, finches and quails	B03
Emblema pictum	Doves, pigeons, finches and quails	B03
Erythrura gouldiae	Doves, pigeons, finches and quails	B03
Geopelia cuneata	Doves, pigeons, finches and quails	B03
Geopelia humeralis	Doves, pigeons, finches and quails	B03
Geopelia striata	Doves, pigeons, finches and quails	B03
Geophaps plumifera	Doves, pigeons, finches and quails	B03
Geophaps scripta	Doves, pigeons, finches and quails	B03
Geophaps smithii	Doves, pigeons, finches and quails	B03
Heteromunia pectoralis	Doves, pigeons, finches and quails	B03
Lonchura castaneothorax	Doves, pigeons, finches and quails	B03
Neochmia modesta	Doves, pigeons, finches and quails	B03
Neochmia phaeton	Doves, pigeons, finches and quails	B03
Neochmia ruficauda	Doves, pigeons, finches and quails	B03
Petrophassa albipennis	Doves, pigeons, finches and quails	B03
Petrophassa rufipennis	Doves, pigeons, finches and quails	B03
Phaps chalcoptera	Doves, pigeons, finches and quails	B03

Poephila acuticauda	Doves, pigeons, finches and quails	B03
Poephila cincta	Doves, pigeons, finches and quails	B03
Poephila personata	Doves, pigeons, finches and quails	B03
Taeniopygia bichenovii	Doves, pigeons, finches and quails	B03
Taeniopygia guttata	Doves, pigeons, finches and quails	B03
Turnix castanotus	Doves, pigeons, finches and quails	B03
Turnix maculosus	Doves, pigeons, finches and quails	B03
Turnix pyrrhothorax	Doves, pigeons, finches and quails	B03
Turnix velox	Doves, pigeons, finches and quails	B03
Chalcophaps indica	Frugivores	B04
Dicaeum hirundinaceum	Frugivores	B04
Ptilinopus cinctus	Frugivores	B04
Ptilinopus regina	Frugivores	B04
Ptilonorhynchus maculatus	Frugivores	B04
Ptilonorhynchus nuchalis	Frugivores	B04
Sphecotheres vieilloti	Frugivores	B04
Zosterops lateralis	Frugivores	B04
Zosterops luteus	Frugivores	B04
Acanthiza chrysorrhoa	Insectivore birds	B05
Aegotheles cristatus	Insectivore birds	B05
Apus pacificus	Insectivore birds	B05
Artamus cinereus	Insectivore birds	B05
Artamus leucorynchus	Insectivore birds	B05
Artamus minor	Insectivore birds	B05
Artamus personatus	Insectivore birds	B05
Artamus superciliosus	Insectivore birds	B05
Cacomantis flabelliformis	Insectivore birds	B05
Cacomantis variolosus	Insectivore birds	B05
Caprimulgus macrurus	Insectivore birds	B05
Centropus phasianinus	Insectivore birds	B05
Chalcites basalis	Insectivore birds	B05
Chalcites lucidus	Insectivore birds	B05
Chalcites minutillus	Insectivore birds	B05
Chalcites osculans	Insectivore birds	B05
Cheramoeca leucosterna	Insectivore birds	B05
Climacteris melanura	Insectivore birds	B05
Climacteris picumnus	Insectivore birds	B05
Colluricincla harmonica	Insectivore birds	B05
Colluricincla megarhyncha	Insectivore birds	B05
Colluricincla woodwardi	Insectivore birds	B05
Coracina maxima	Insectivore birds	B05
Coracina novaehollandiae	Insectivore birds	B05
Coracina papuensis	Insectivore birds	B05
Coracina tenuirostris	Insectivore birds	B05
Cuculus optatus	Insectivore birds	B05
Cuculus pallidus	Insectivore birds	B05
Daphoenositta chrysoptera	Insectivore birds	B05
Dicrurus bracteatus	Insectivore birds	B05
Eudynamys orientalis	Insectivore birds	B05
Eurostopodus argus	Insectivore birds	B05
Eurostopodus mystacalis	Insectivore birds	B05
Eurystomus orientalis	Insectivore birds	B05
Falcunculus frontatus	Insectivore birds	B05
Gerygone chloronota	Insectivore birds	B05
Gerygone fusca	Insectivore birds	B05
Gerygone levigaster	Insectivore birds	B05
Gerygone magnirostris	Insectivore birds	B05
Gerygone olivacea	Insectivore birds	B05
Gerygone palpebrosa	Insectivore birds	B05
Gerygone tenebrosa	Insectivore birds	B05
Grallina cyanoleuca	Insectivore birds	B05
Hirundo neoxena	Insectivore birds	B05
Lalage leucomela	Insectivore birds	B05
Lalage tricolor	Insectivore birds	B05
Malurus coronatus	Insectivore birds	B05
Malurus lamberti	Insectivore birds	B05
Malurus leucopterus	Insectivore birds	B05
Malurus melanocephalus	Insectivore birds	B05
Manorina flavigula	Insectivore birds	B05
Manorina melanocephala	Insectivore birds	B05
Melanodryas cucullata	Insectivore birds	B05
Merops ornatus	Insectivore birds	B05
Microeca fascinans	Insectivore birds	B05

Microeca flavigaster	Insectivore birds	B05
Myiagra alecto	Insectivore birds	B05
Myiagra inquieta	Insectivore birds	B05
Myiagra rubecula	Insectivore birds	B05
Myiagra ruficollis	Insectivore birds	B05
Nectarinia jugularis	Insectivore birds	B05
Oreoica gutturalis	Insectivore birds	B05
Oriolus flavocinctus	Insectivore birds	B05
Oriolus sagittatus	Insectivore birds	B05
Pachycephala lanioides	Insectivore birds	B05
Pachycephala melanura	Insectivore birds	B05
Pachycephala rufiventris	Insectivore birds	B05
Pachycephala simplex	Insectivore birds	B05
Pardalotus rubricatus	Insectivore birds	B05
Pardalotus striatus	Insectivore birds	B05
Peneoenanthe pulverulenta	Insectivore birds	B05
Petrochelidon ariel	Insectivore birds	B05
Petrochelidon nigricans	Insectivore birds	B05
Petroica goodenovii	Insectivore birds	B05
Pitta iris	Insectivore birds	B05
Poecilodryas cerviniventris	Insectivore birds	B05
Pomatostomus temporalis	Insectivore birds	B05
Rhipidura fuliginosa	Insectivore birds	B05
Rhipidura leucophrys	Insectivore birds	B05
Rhipidura phasiana	Insectivore birds	B05
Rhipidura rufifrons	Insectivore birds	B05
Rhipidura rufiventris	Insectivore birds	B05
Scythrops novaehollandiae	Insectivore birds	B05
Sericornis frontalis	Insectivore birds	B05
Smicrornis brevirostris	Insectivore birds	B05
Stipiturus ruficeps	Insectivore birds	B05
Struthidea cinerea	Insectivore birds	B05
Acanthagenys rufogularis	Honeyeaters, friarbirds and chats	B06
Certhionyx variegatus	Honeyeaters, friarbirds and chats	B06
Cissomela pectoralis	Honeyeaters, friarbirds and chats	B06
Conopophila albogularis	Honeyeaters, friarbirds and chats	B06
Conopophila rufogularis	Honeyeaters, friarbirds and chats	B06
Entomyzon cyanotis	Honeyeaters, friarbirds and chats	B06
Epthianura crocea	Honeyeaters, friarbirds and chats	B06
Epthianura tricolor	Honeyeaters, friarbirds and chats	B06
Lichenostomus flavescens	Honeyeaters, friarbirds and chats	B06
Lichenostomus flavus	Honeyeaters, friarbirds and chats	B06
Lichenostomus keartlandi	Honeyeaters, friarbirds and chats	B06
Lichenostomus penicillatus	Honeyeaters, friarbirds and chats	B06
Lichenostomus plumulus	Honeyeaters, friarbirds and chats	B06
Lichenostomus unicolor	Honeyeaters, friarbirds and chats	B06
Lichenostomus virescens	Honeyeaters, friarbirds and chats	B06
Lichmera indistincta	Honeyeaters, friarbirds and chats	B06
Meliphaga albilineata	Honeyeaters, friarbirds and chats	B06
Meliphaga lewinii	Honeyeaters, friarbirds and chats	B06
Melithreptus albogularis	Honeyeaters, friarbirds and chats	B06
Melithreptus gularis	Honeyeaters, friarbirds and chats	B06
Myzomela erythrocephala	Honeyeaters, friarbirds and chats	B06
Myzomela obscura	Honeyeaters, friarbirds and chats	B06
Myzomela sanguinolenta	Honeyeaters, friarbirds and chats	B06
Philemon argenticeps	Honeyeaters, friarbirds and chats	B06
Philemon buceroides	Honeyeaters, friarbirds and chats	B06
Philemon citreogularis	Honeyeaters, friarbirds and chats	B06
Philemon corniculatus	Honeyeaters, friarbirds and chats	B06
Ramsayornis fasciatus	Honeyeaters, friarbirds and chats	B06
Sugomel niger	Honeyeaters, friarbirds and chats	B06
Accipiter cirrocephalus	Raptors, owls, corvids and tree kingfishers	B07
Accipiter fasciatus	Raptors, owls, corvids and tree kingfishers	B07
Accipiter novaehollandiae	Raptors, owls, corvids and tree kingfishers	B07
Aquila audax	Raptors, owls, corvids and tree kingfishers	B07
Aviceda subcristata	Raptors, owls, corvids and tree kingfishers	B07
Circus approximans	Raptors, owls, corvids and tree kingfishers	B07
Circus assimilis	Raptors, owls, corvids and tree kingfishers	B07
Corvus bennetti	Raptors, owls, corvids and tree kingfishers	B07
Corvus coronoides	Raptors, owls, corvids and tree kingfishers	B07
Corvus orru	Raptors, owls, corvids and tree kingfishers	B07
Cracticus nigrogularis	Raptors, owls, corvids and tree kingfishers	B07
Cracticus quoyi	Raptors, owls, corvids and tree kingfishers	B07

Cracticus tibicen	Raptors, owls, corvids and tree kingfishers	B07
Cracticus torquatus	Raptors, owls, corvids and tree kingfishers	B07
Dacelo leachii	Raptors, owls, corvids and tree kingfishers	B07
Dacelo novaeguineae	Raptors, owls, corvids and tree kingfishers	B07
Elanus axillaris	Raptors, owls, corvids and tree kingfishers	B07
Erythrotriorchis radiatus	Raptors, owls, corvids and tree kingfishers	B07
Falco berigora	Raptors, owls, corvids and tree kingfishers	B07
Falco cenchroides	Raptors, owls, corvids and tree kingfishers	B07
Falco hypoleucos	Raptors, owls, corvids and tree kingfishers	B07
Falco longipennis	Raptors, owls, corvids and tree kingfishers	B07
Falco peregrinus	Raptors, owls, corvids and tree kingfishers	B07
Falco subniger	Raptors, owls, corvids and tree kingfishers	B07
Haliaeetus leucogaster	Raptors, owls, corvids and tree kingfishers	B07
Haliastur indus	Raptors, owls, corvids and tree kingfishers	B07
Haliastur sphenurus	Raptors, owls, corvids and tree kingfishers	B07
Hamirostra melanosternon	Raptors, owls, corvids and tree kingfishers	B07
Hieraaetus morphnoides	Raptors, owls, corvids and tree kingfishers	B07
Lophoictinia isura	Raptors, owls, corvids and tree kingfishers	B07
Milvus migrans	Raptors, owls, corvids and tree kingfishers	B07
Ninox connivens	Raptors, owls, corvids and tree kingfishers	B07
Ninox novaeseelandiae	Raptors, owls, corvids and tree kingfishers	B07
Ninox rufa	Raptors, owls, corvids and tree kingfishers	B07
Pandion haliaetus	Raptors, owls, corvids and tree kingfishers	B07
Podargus strigoides	Raptors, owls, corvids and tree kingfishers	B07
Strepera graculina	Raptors, owls, corvids and tree kingfishers	B07
Todiramphus chloris	Raptors, owls, corvids and tree kingfishers	B07
Todiramphus macleayii	Raptors, owls, corvids and tree kingfishers	B07
Todiramphus pyrrhopygius	Raptors, owls, corvids and tree kingfishers	B07
Todiramphus sanctus	Raptors, owls, corvids and tree kingfishers	B07
Tyto alba	Raptors, owls, corvids and tree kingfishers	B07
Tyto longimembris	Raptors, owls, corvids and tree kingfishers	B07
Tyto novaehollandiae	Raptors, owls, corvids and tree kingfishers	B07
Ceyx azureus	River kingfishers	B08
Ceyx pusilla	River kingfishers	B08
Acrocephalus australis	Grassland and swamp birds	B09
Amytornis woodwardi	Grassland and swamp birds	B09
Anthus novaeseelandiae	Grassland and swamp birds	B09
Ardeotis australis	Grassland and swamp birds	B09
Burhinus grallarius	Grassland and swamp birds	B09
Cincloramphus cruralis	Grassland and swamp birds	B09
Cincloramphus mathewsi	Grassland and swamp birds	B09
Cisticola exilis	Grassland and swamp birds	B09
Cisticola juncidis	Grassland and swamp birds	B09
Eremiornis carteri	Grassland and swamp birds	B09
Megalurus gramineus	Grassland and swamp birds	B09
Megalurus timoriensis	Grassland and swamp birds	B09
Mirafra javanica	Grassland and swamp birds	B09
Dromaius novaehollandiae	Emu	B10
Alectura lathami	Megapodes	B11
Megapodius reinwardt	Megapodes	B11
Amaurornis cinerea	Waterfowl	B12
Anas gracilis	Waterfowl	B12
Anas superciliosa	Waterfowl	B12
Anseranas semipalmata	Waterfowl	B12
Ardea alba	Waterfowl	B12
Ardea ibis	Waterfowl	B12
Ardea intermedia	Waterfowl	B12
Ardea pacifica	Waterfowl	B12
Ardea sumatrana	Waterfowl	B12
Aythya australis	Waterfowl	B12
Butorides striatus	Waterfowl	B12
Chenonetta jubata	Waterfowl	B12
Cygnus atratus	Waterfowl	B12
Dendrocygna arcuata	Waterfowl	B12
Dendrocygna eytoni	Waterfowl	B12
Egretta garzetta	Waterfowl	B12
Egretta novaehollandiae	Waterfowl	B12
Egretta picata	Waterfowl	B12
Ephippiorhynchus asiaticus	Waterfowl	B12
Eulabeornis castaneoventris	Waterfowl	B12
Fulica atra	Waterfowl	B12
Gallinago hardwickii	Waterfowl	B12
Gallinago megala	Waterfowl	B12

Gallinula tenebrosa	Waterfowl	B12
Gallirallus philippensis	Waterfowl	B12
Glareola maldivarum	Waterfowl	B12
Grus antigone	Waterfowl	B12
Grus rubicunda	Waterfowl	B12
Himantopus himantopus	Waterfowl	B12
Irediparra gallinacea	Waterfowl	B12
Ixobrychus flavicollis	Waterfowl	B12
Malacorhynchus membranaceus	Waterfowl	B12
Microcarbo melanoleucos	Waterfowl	B12
Nettapus coromandelianus	Waterfowl	B12
Nettapus pulchellus	Waterfowl	B12
Nycticorax caledonicus	Waterfowl	B12
Pelecanus conspicillatus	Waterfowl	B12
Phalacrocorax carbo	Waterfowl	B12
Phalacrocorax sulcirostris	Waterfowl	B12
Phalacrocorax varius	Waterfowl	B12
Platalea flavipes	Waterfowl	B12
Platalea regia	Waterfowl	B12
Plegadis falcinellus	Waterfowl	B12
Podiceps cristatus	Waterfowl	B12
Poliocephalus poliocephalus	Waterfowl	B12
Porphyrio porphyrio	Waterfowl	B12
Porzana tabuensis	Waterfowl	B12
Recurvirostra novaehollandiae	Waterfowl	B12
Rostratula australis	Waterfowl	B12
Stiltia isabella	Waterfowl	B12
Tachybaptus novaehollandiae	Waterfowl	B12
Tadorna radjah	Waterfowl	B12
Threskiornis molucca	Waterfowl	B12
Threskiornis spinicollis	Waterfowl	B12
Mammals		
Hydromys chrysogaster	Aquatic mammals	M01
Xeromys myoides	Aquatic mammals	M01
Macroglossus minimus	Mega bats	M02
Nyctimene robinsoni	Mega bats	M02
Pteropus alecto	Mega bats	M02
Pteropus scapulatus	Mega bats	M02
Syconycteris australis	Mega bats	M02
Chaerephon jobensis	Micro bats	M03
Chalinolobus gouldii	Micro bats	M03
Chalinolobus morio	Micro bats	M03
Chalinolobus nigrogriseus	Micro bats	M03
Hipposideros ater	Micro bats	M03
Hipposideros diadema	Micro bats	M03
Hipposideros stenotis	Micro bats	M03
Macroderma gigas	Micro bats	M03
Miniopterus australis	Micro bats	M03
Miniopterus schreibersii	Micro bats	M03
Mormopterus beccarii	Micro bats	M03
Myotis macropus	Micro bats	M03
Nyctophilus arnhemensis	Micro bats	M03
Nyctophilus bifax	Micro bats	M03
Nyctophilus geoffroyi	Micro bats	M03
Nyctophilus walkeri	Micro bats	M03
Pipistrellus adamsi	Micro bats	M03
Pipistrellus westralis	Micro bats	M03
Rhinolophus megaphyllus	Micro bats	M03
Rhinonicteris aurantia	Micro bats	M03
Saccolaimus flaviventris	Micro bats	M03
Scotorepens balstoni	Micro bats	M03
Scotorepens greyii	Micro bats	M03
Scotorepens sanborni	Micro bats	M03
Tadarida australis	Micro bats	M03
Taphozous georgianus	Micro bats	M03
Taphozous hilli	Micro bats	M03
Vespadelus caurinus	Micro bats	M03
Vespadelus douglasorum	Micro bats	M03
Vespadelus finlaysoni	Micro bats	M03
Vespadelus troughtoni	Micro bats	M03
Conilurus penicillatus	Arboreal marsupials and tree rats	M04
Melomys burtoni	Arboreal marsupials and tree rats	M04
Mesembriomys gouldii	Arboreal marsupials and tree rats	M04

Mesembriomys macrurus	Arboreal marsupials and tree rats	M04
Petaurus breviceps	Arboreal marsupials and tree rats	M04
Phascogale tapoatafa	Arboreal marsupials and tree rats	M04
Pseudocheirus peregrinus	Arboreal marsupials and tree rats	M04
Trichosurus vulpecula	Arboreal marsupials and tree rats	M04
Macropus agilis	Large macropods	M05
Macropus antilopinus	Large macropods	M05
Macropus bernardus	Large macropods	M05
Macropus fuliginosus	Large macropods	M05
Macropus parryi	Large macropods	M05
Macropus robustus	Large macropods	M05
Macropus rufus	Large macropods	M05
Onychogalea unguifera	Large macropods	M05
Aepyprymnus rufescens	Potoroos, bandicoots and hare-wallaby	M06
Isoodon macrourus	Potoroos, bandicoots and hare-wallaby	M06
Petrogale brachyotis	Rock-dwelling mammals	M07
Petrogale concinna	Rock-dwelling mammals	M07
Petrogale lateralis	Rock-dwelling mammals	M07
Petrogale mareeba	Rock-dwelling mammals	M07
Petrogale purpureicollis	Rock-dwelling mammals	M07
Petrogale rothschildi	Rock-dwelling mammals	M07
Petropseudes dahli	Rock-dwelling mammals	M07
Zyzomys argurus	Rock-dwelling mammals	M07
Zyzomys maini	Rock-dwelling mammals	M07
Antechinus bellus	Small ground mammals	M08
Dasycercus cristicauda	Small ground mammals	M08
Leggadina forresti	Small ground mammals	M08
Leggadina lakedownensis	Small ground mammals	M08
Macrotis lagotis	Small ground mammals	M08
Ningaui ridei	Small ground mammals	M08
Notomys alexis	Small ground mammals	M08
Notomys aquilo	Small ground mammals	M08
Planigale ingrami	Small ground mammals	M08
Planigale maculata	Small ground mammals	M08
Pseudantechinus bilarni	Small ground mammals	M08
Pseudantechinus ningbing	Small ground mammals	M08
Pseudomys calabyi	Small ground mammals	M08
Pseudomys delicatulus	Small ground mammals	M08
Pseudomys desertor	Small ground mammals	M08
Pseudomys hermannsburgensis	Small ground mammals	M08
Pseudomys johnsoni	Small ground mammals	M08
Pseudomys nanus	Small ground mammals	M08
Rattus colletti	Small ground mammals	M08
Rattus sordidus	Small ground mammals	M08
Rattus tunneyi	Small ground mammals	M08
Rattus villosissimus	Small ground mammals	M08
Sminthopsis bindi	Small ground mammals	M08
Sminthopsis macroura	Small ground mammals	M08
Sminthopsis virginiae	Small ground mammals	M08
Sminthopsis youngsoni	Small ground mammals	M08
Tachyglossus aculeatus	Echidna	M09
Dasyurus hallucatus	Quoll	M11
Reptiles		
Acrochordus arafurae	Aquatic snakes (except water python)	R01
Tropidonophis mairii	Aquatic snakes (except water python)	R01
Ramphotyphlops diversus	Blind snakes	R02
Ramphotyphlops grypus	Blind snakes	R02
Ramphotyphlops guentheri	Blind snakes	R02
Ramphotyphlops ligatus	Blind snakes	R02
Ramphotyphlops unguirostris	Blind snakes	R02
Acanthophis antarcticus	Death adders	R03
Demansia olivacea	Fast diurnal snakes	R04
Demansia papuensis	Fast diurnal snakes	R04
Demansia psammophis	Fast diurnal snakes	R04
Demansia quaesitor	Fast diurnal snakes	R04
Demansia rimicola	Fast diurnal snakes	R04
Demansia simplex	Fast diurnal snakes	R04
Demansia vestigiata	Fast diurnal snakes	R04
Dendrelaphis punctulatus	Fast diurnal snakes	R04
Aspidites melanocephalus	Large pythons	R05
Aspidites ramsayi	Large pythons	R05
Liasis olivaceus	Large pythons	R05
Morelia spilota	Large pythons	R05

Oxyuranus scutellatus	Large elapids	R06
Pseudechis australis	Large elapids	R06
Pseudonaja guttata	Large elapids	R06
Pseudonaja ingrami	Large elapids	R06
Pseudonaja modesta	Large elapids	R06
Pseudonaja textilis	Large elapids	R06
Brachyurophis australis	Small nocturnal elapids and pygopids	R07
Brachyurophis incinctus	Small nocturnal elapids and pygopids	R07
Brachyurophis roperi	Small nocturnal elapids and pygopids	R07
Brachyurophis semifasciatus	Small nocturnal elapids and pygopids	R07
Cryptophis boschmai	Small nocturnal elapids and pygopids	R07
Cryptophis nigrescens	Small nocturnal elapids and pygopids	R07
Cryptophis pallidiceps	Small nocturnal elapids and pygopids	R07
Furina ornata	Small nocturnal elapids and pygopids	R07
Hoplocephalus bitorquatus	Small nocturnal elapids and pygopids	R07
Lialis burtonis	Small nocturnal elapids and pygopids	R07
Pygopus nigriceps	Small nocturnal elapids and pygopids	R07
Pygopus steelescotti	Small nocturnal elapids and pygopids	R07
Simoselaps anomalus	Small nocturnal elapids and pygopids	R07
Suta punctata	Small nocturnal elapids and pygopids	R07
Suta suta	Small nocturnal elapids and pygopids	R07
Vermicella annulata	Small nocturnal elapids and pygopids	R07
Vermicella multifasciata	Small nocturnal elapids and pygopids	R07
Antaresia maculosa	Small to medium nocturnal non-elapid snakes	R08
Antaresia stimsoni	Small to medium nocturnal non-elapid snakes	R08
Boiga irregularis	Small to medium nocturnal non-elapid snakes	R08
Stegonotus cucullatus	Small to medium nocturnal non-elapid snakes	R08
Diporiphora albilabris	Partially-arboreal small agamids	R09
Diporiphora arnhemica	Partially-arboreal small agamids	R09
Diporiphora australis	Partially-arboreal small agamids	R09
Diporiphora bennettii	Partially-arboreal small agamids	R09
Diporiphora bilineata	Partially-arboreal small agamids	R09
Diporiphora lalliae	Partially-arboreal small agamids	R09
Diporiphora magna	Partially-arboreal small agamids	R09
Diporiphora pindan	Partially-arboreal small agamids	R09
Diporiphora winneckei	Partially-arboreal small agamids	R09
Lophognathus gilberti	Partially-arboreal small agamids	R09
Lophognathus temporalis	Partially-arboreal small agamids	R09
Pogona minor	Partially-arboreal small agamids	R09
Delma borea	Fossorial reptiles	R10
Delma nasuta	Fossorial reptiles	R10
Delma tincta	Fossorial reptiles	R10
Eremiascincus fasciolatus	Fossorial reptiles	R10
Eremiascincus richardsonii	Fossorial reptiles	R10
Glaphyromorphus cracens	Fossorial reptiles	R10
Glaphyromorphus darwiniensis	Fossorial reptiles	R10
Glaphyromorphus douglasi	Fossorial reptiles	R10
Glaphyromorphus isolepis	Fossorial reptiles	R10
Glaphyromorphus nigricaudis	Fossorial reptiles	R10
Lerista bipes	Fossorial reptiles	R10
Lerista borealis	Fossorial reptiles	R10
Lerista greeri	Fossorial reptiles	R10
Lerista griffini	Fossorial reptiles	R10
Lerista ips	Fossorial reptiles	R10
Lerista karlschmidti	Fossorial reptiles	R10
Lerista labialis	Fossorial reptiles	R10
Lerista orientalis	Fossorial reptiles	R10
Lerista vermicularis	Fossorial reptiles	R10
Lerista zonulata	Fossorial reptiles	R10
Liopholis kintorei	Fossorial reptiles	R10
Lygisaurus foliorum	Fossorial reptiles	R10
Menetia alanae	Fossorial reptiles	R10
Menetia greyii	Fossorial reptiles	R10
Menetia maini	Fossorial reptiles	R10
Morethia ruficauda	Fossorial reptiles	R10
Morethia storri	Fossorial reptiles	R10
Morethia taeniopleura	Fossorial reptiles	R10
Notoscincus ornatus	Fossorial reptiles	R10
Proablepharus kinghorni	Fossorial reptiles	R10
Proablepharus reginae	Fossorial reptiles	R10
Proablepharus tenuis	Fossorial reptiles	R10
Bellatorias obiri	Large terrestrial skinks	R11
Tiliqua multifasciata	Large terrestrial skinks	R11

Tiliqua rugosa	Large terrestrial skinks	R11
Tiliqua scincoides	Large terrestrial skinks	R11
Carlia amax	Small terrestrial skinks	R12
Carlia gracilis	Small terrestrial skinks	R12
Carlia jarnoldae	Small terrestrial skinks	R12
Carlia johnstonei	Small terrestrial skinks	R12
Carlia munda	Small terrestrial skinks	R12
Carlia pectoralis	Small terrestrial skinks	R12
Carlia rufilatus	Small terrestrial skinks	R12
Carlia schmeltzii	Small terrestrial skinks	R12
Carlia triacantha	Small terrestrial skinks	R12
Carlia vivax	Small terrestrial skinks	R12
Ctenotus borealis	Small terrestrial skinks	R12
Ctenotus brevipes	Small terrestrial skinks	R12
Ctenotus brooksi	Small terrestrial skinks	R12
Ctenotus coggeri	Small terrestrial skinks	R12
Ctenotus decaneurus	Small terrestrial skinks	R12
Ctenotus essingtonii	Small terrestrial skinks	R12
Ctenotus greeri	Small terrestrial skinks	R12
Ctenotus helenae	Small terrestrial skinks	R12
Ctenotus hilli	Small terrestrial skinks	R12
Ctenotus inornatus	Small terrestrial skinks	R12
Ctenotus leonhardii	Small terrestrial skinks	R12
Ctenotus militaris	Small terrestrial skinks	R12
Ctenotus pallescens	Small terrestrial skinks	R12
Ctenotus pantherinus	Small terrestrial skinks	R12
Ctenotus piankai	Small terrestrial skinks	R12
Ctenotus pulchellus	Small terrestrial skinks	R12
Ctenotus quattuordecimlineatus	Small terrestrial skinks	R12
Ctenotus rimacolus	Small terrestrial skinks	R12
Ctenotus robustus	Small terrestrial skinks	R12
Ctenotus saxatilis	Small terrestrial skinks	R12
Ctenotus spaldingi	Small terrestrial skinks	R12
Ctenotus storri	Small terrestrial skinks	R12
Ctenotus striaticeps	Small terrestrial skinks	R12
Ctenotus tantillus	Small terrestrial skinks	R12
Ctenotus vertebralis	Small terrestrial skinks	R12
Cryptoblepharus pannosus	Small arboreal geckos and skinks	R13
Cryptoblepharus ruber	Small arboreal geckos and skinks	R13
Gehyra australis	Small arboreal geckos and skinks	R13
Gehyra dubia	Small arboreal geckos and skinks	R13
Gehyra purpurascens	Small arboreal geckos and skinks	R13
Gehyra variegata	Small arboreal geckos and skinks	R13
Oedura castelnaui	Small arboreal geckos and skinks	R13
Oedura coggeri	Small arboreal geckos and skinks	R13
Oedura marmorata	Small arboreal geckos and skinks	R13
Oedura monilis	Small arboreal geckos and skinks	R13
Oedura rhombifer	Small arboreal geckos and skinks	R13
Crenadactylus ocellatus	Ground-dwelling geckos and small agamids	R14
Ctenophorus caudicinctus	Ground-dwelling geckos and small agamids	R14
Ctenophorus nuchalis	Ground-dwelling geckos and small agamids	R14
Gehyra borroloola	Ground-dwelling geckos and small agamids	R14
Gehyra occidentalis	Ground-dwelling geckos and small agamids	R14
Gehyra pilbara	Ground-dwelling geckos and small agamids	R14
Gehyra punctata	Ground-dwelling geckos and small agamids	R14
Heteronotia binoei	Ground-dwelling geckos and small agamids	R14
Lucasium immaculatum	Ground-dwelling geckos and small agamids	R14
Lucasium stenodactylum	Ground-dwelling geckos and small agamids	R14
Nephrurus laevissimus	Ground-dwelling geckos and small agamids	R14
Nephrurus milii	Ground-dwelling geckos and small agamids	R14
Rhynchoedura ornata	Ground-dwelling geckos and small agamids	R14
Strophurus ciliaris	Ground-dwelling geckos and small agamids	R14
Strophurus elderi	Ground-dwelling geckos and small agamids	R14
Strophurus jeanae	Ground-dwelling geckos and small agamids	R14
Strophurus krisalys	Ground-dwelling geckos and small agamids	R14
Strophurus taeniatus	Ground-dwelling geckos and small agamids	R14
Carlia mundivensis	Rock-dwelling lizards	R15
Egernia hosmeri	Rock-dwelling lizards	R15
Gehyra nana	Rock-dwelling lizards	R15
Gehyra pamela	Rock-dwelling lizards	R15
Gehyra robusta	Rock-dwelling lizards	R15
Heteronotia planiceps	Rock-dwelling lizards	R15
Nephrurus asper	Rock-dwelling lizards	R15

Nephrurus sheai	Rock-dwelling lizards	R15
Oedura gemmata	Rock-dwelling lizards	R15
Oedura gracilis	Rock-dwelling lizards	R15
Pseudothecadactylus lindneri	Rock-dwelling lizards	R15
Varanus acanthurus	Rock-dwelling lizards	R15
Varanus baritji	Rock-dwelling lizards	R15
Varanus glebopalma	Rock-dwelling lizards	R15
Chelosania brunnea	Partially-arboreal varanids and large agamids	R16
Chlamydosaurus kingii	Partially-arboreal varanids and large agamids	R16
Varanus gilleni	Partially-arboreal varanids and large agamids	R16
Varanus scalaris	Partially-arboreal varanids and large agamids	R16
Varanus tristis	Partially-arboreal varanids and large agamids	R16
Varanus brevicauda	Small terrestrial Varanids	R17
Varanus eremius	Small terrestrial Varanids	R17
Varanus primordius	Small terrestrial Varanids	R17
Varanus storri	Small terrestrial Varanids	R17
Varanus gouldii	Large terrestrial varanids	R18
Varanus panoptes	Large terrestrial varanids	R18
Varanus indicus	Water varanids	R19
Varanus mertensi	Water varanids	R19
Varanus mitchelli	Water varanids	R19

7 Water intake

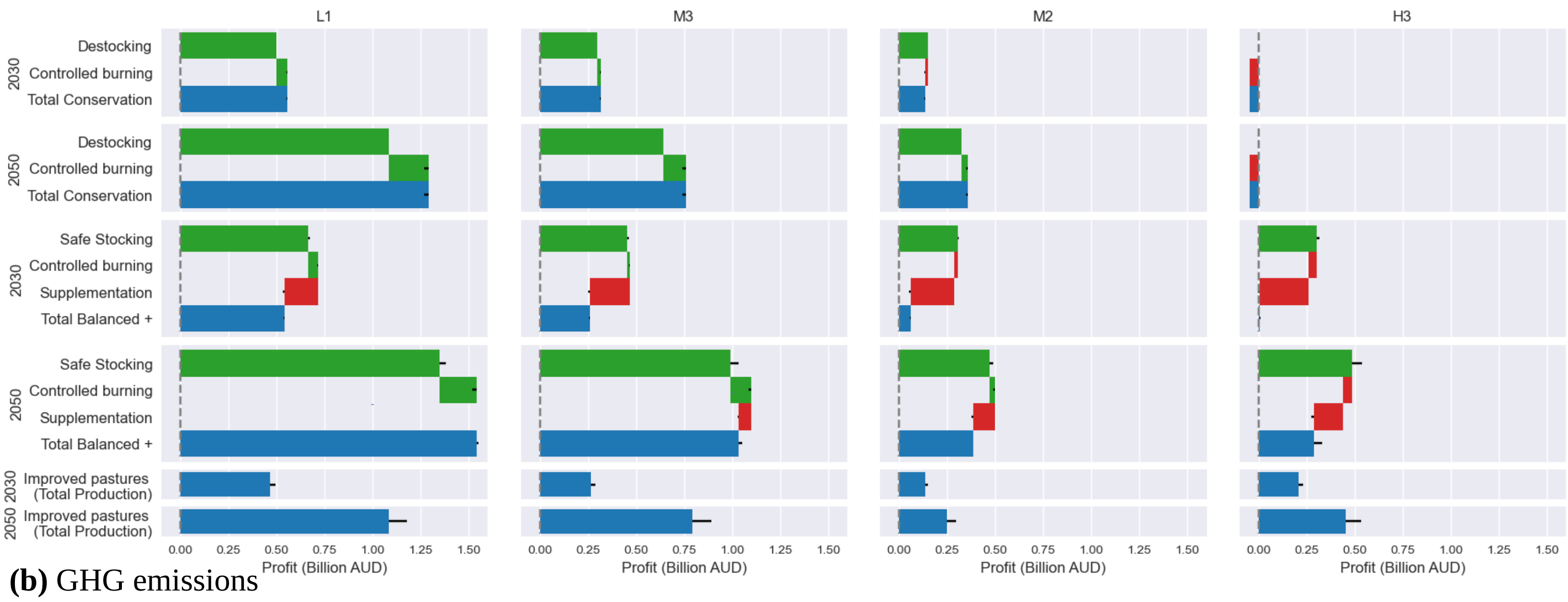
Water use is an increasingly important issue in northern Australia's rangelands. Despite high annual rainfall, it is seasonal in nature and many enterprises rely on bore water. Increasing stock numbers will inevitably increase the demand for water, and this is likely to be exacerbated by climate change, with stock requiring a greater water intake in higher temperatures. The functional form of the relationship between water intake and temperature for *Bos indicus* cattle has been developed by Watts, Tucker and Casey (1994), using the collated data of Winchester and Morris (1956). We modified this equation to simulate water intake over the study region under climate change:

$$WI_{iy} = AE_{iy}\left(DMI\left(3.076 + 0.008461e^{0.17596(\mathrm{T}_i+\Delta\mathrm{T}_{iy})}\right)\right) \quad \text{(S24)}$$

Where WI_{iy} is the water intake per pixel i in year y in litres per day, AE_{iy} is the simulated number of adult equivalents in pixel i in year y, *DMI* is the dry matter intake per AE (in kg per day), T_i is the baseline historic daily maximum temperature (in °C) (the median was taken for the main analyses but varied between the 10th and 90th percentiles in the sensitivity analysis), and ΔT_{iy} was the predicted change in temperature (under different global outlooks and GCMs) for pixel i in year y.

8 Supplementary Results

(a) Profit



(b) GHG emissions

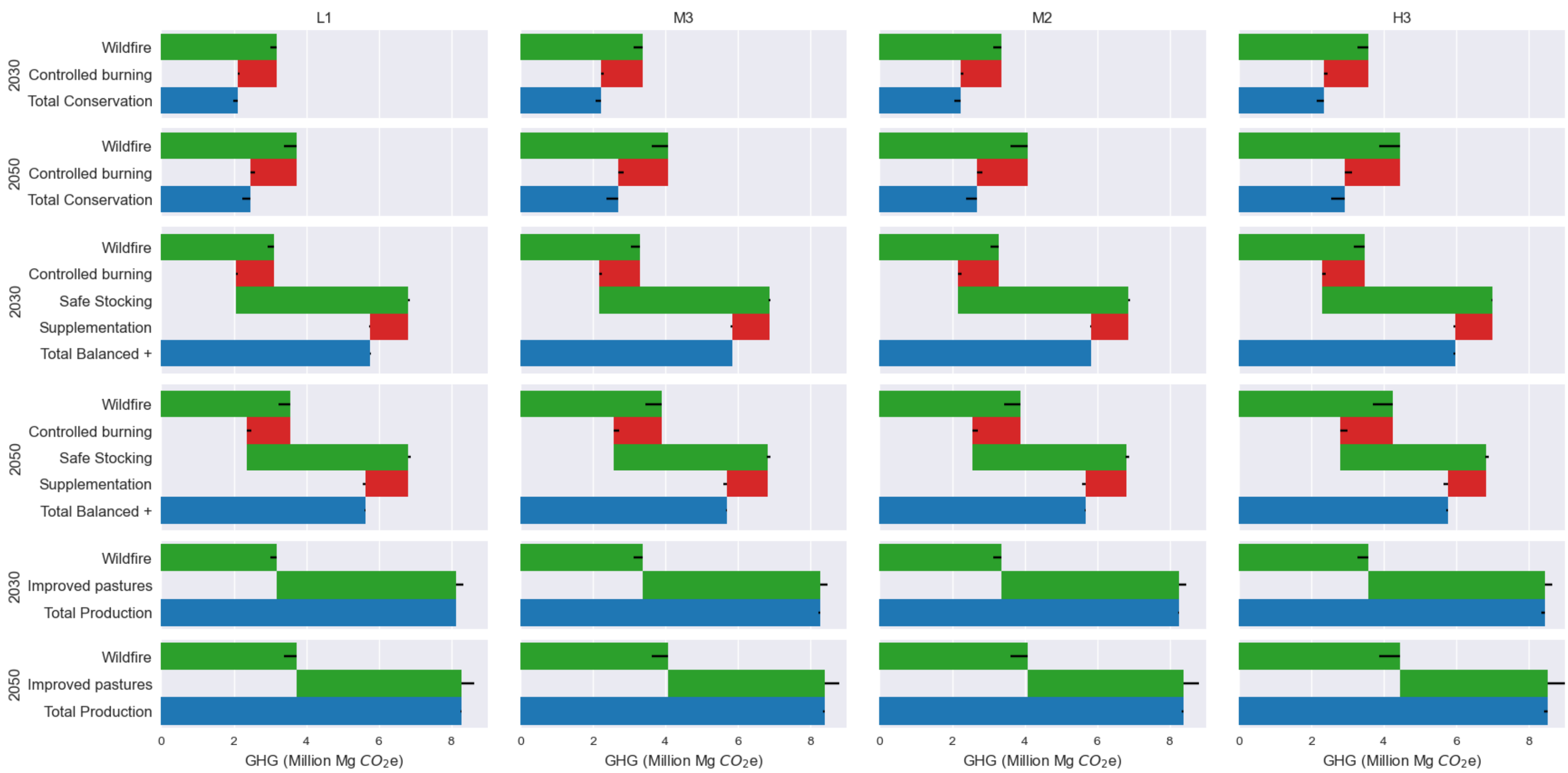


Figure S25 | Contributions to profit (a) and GHG emissions (b) for each management scenario for 2030 and 2050 across global outlooks (L1, M2, M3, H3). Error bars represent the range in outcomes from different GCMs. Here the 'Balanced' scenario is omitted, as it has the same actions as "Balanced +" except for supplementation. Supplementation for 2050 in L1 essentially broke even (an aggregated \$1.5 million loss, which is imperceptible on this scale).

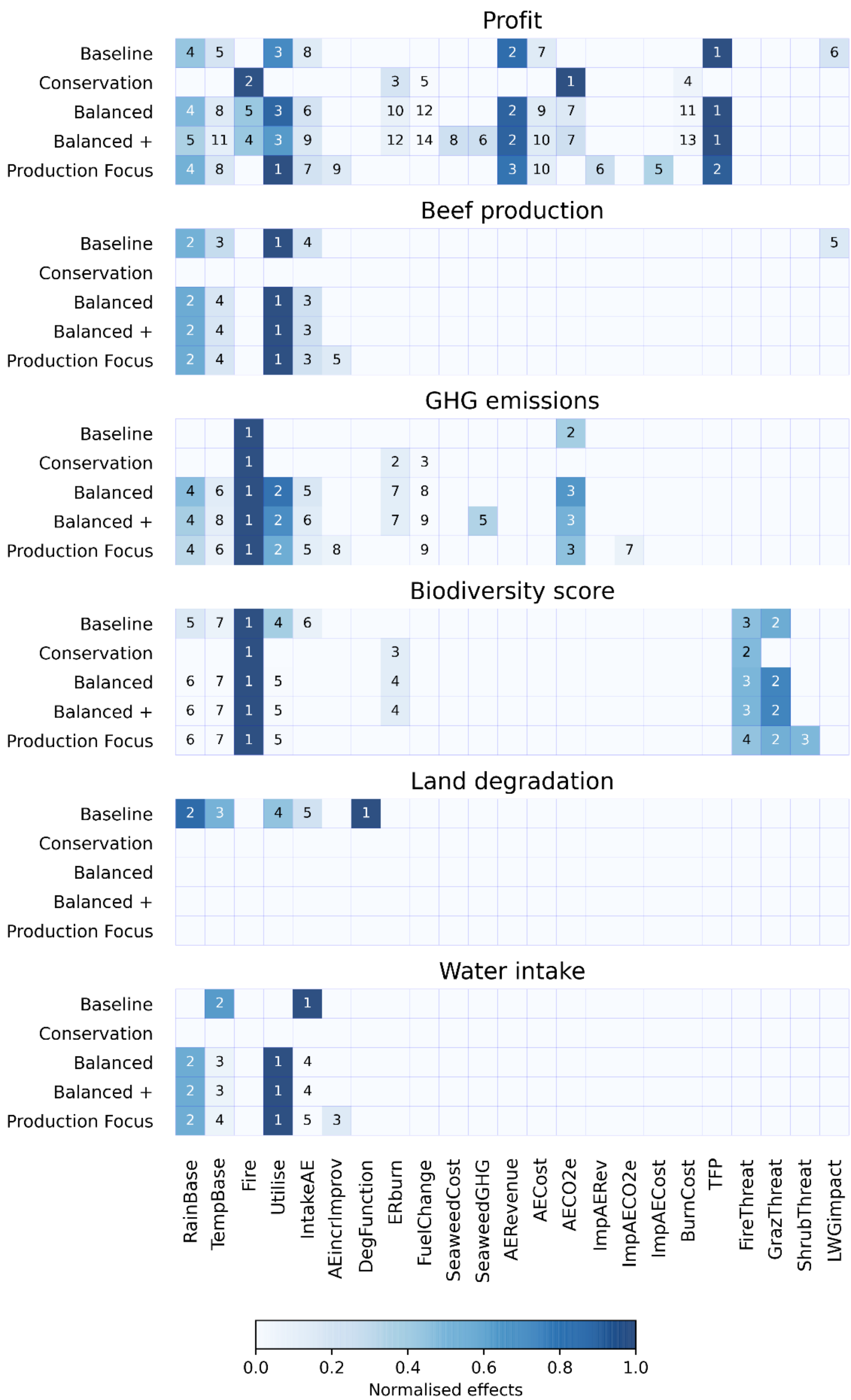


Figure S26 | Normalised Elementary Effects of each parameter to a given output variable (landholder profit, beef production, GHG emissions, biodiversity, land degradation, and water intake). The bigger the value the more influential the parameter on the outcome. The numbers inside each box are a ranking of parameters (in terms of influence) for each management scenario and outcome. All outcomes are for global outlook M3, GCM MPI, and year 2050.

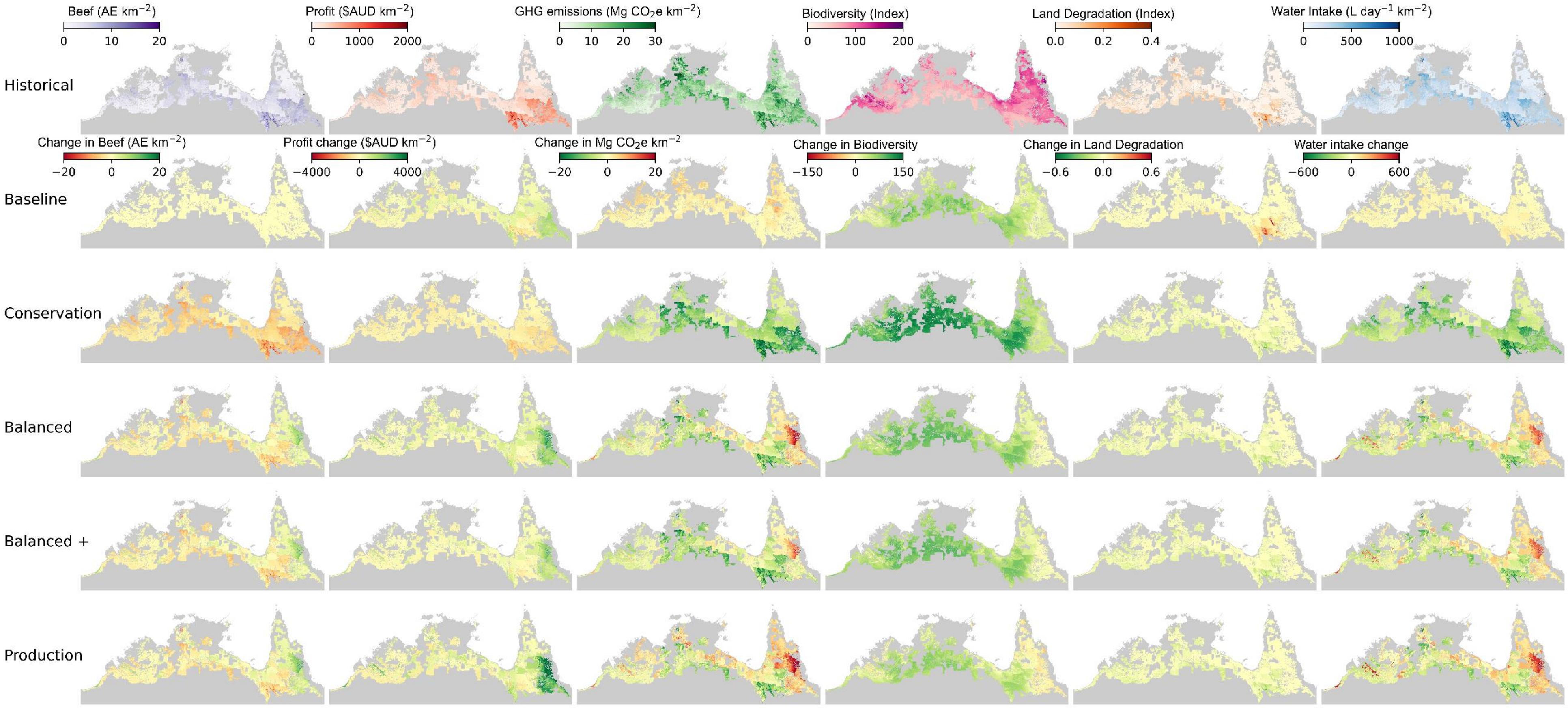


Figure S27 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM CE2 and global outlook L1. “Historical” represents stocking rates and climatic conditions representative of the period from 1987-2010. The remaining rows show the change from historical conditions to 2050 for each outcome.

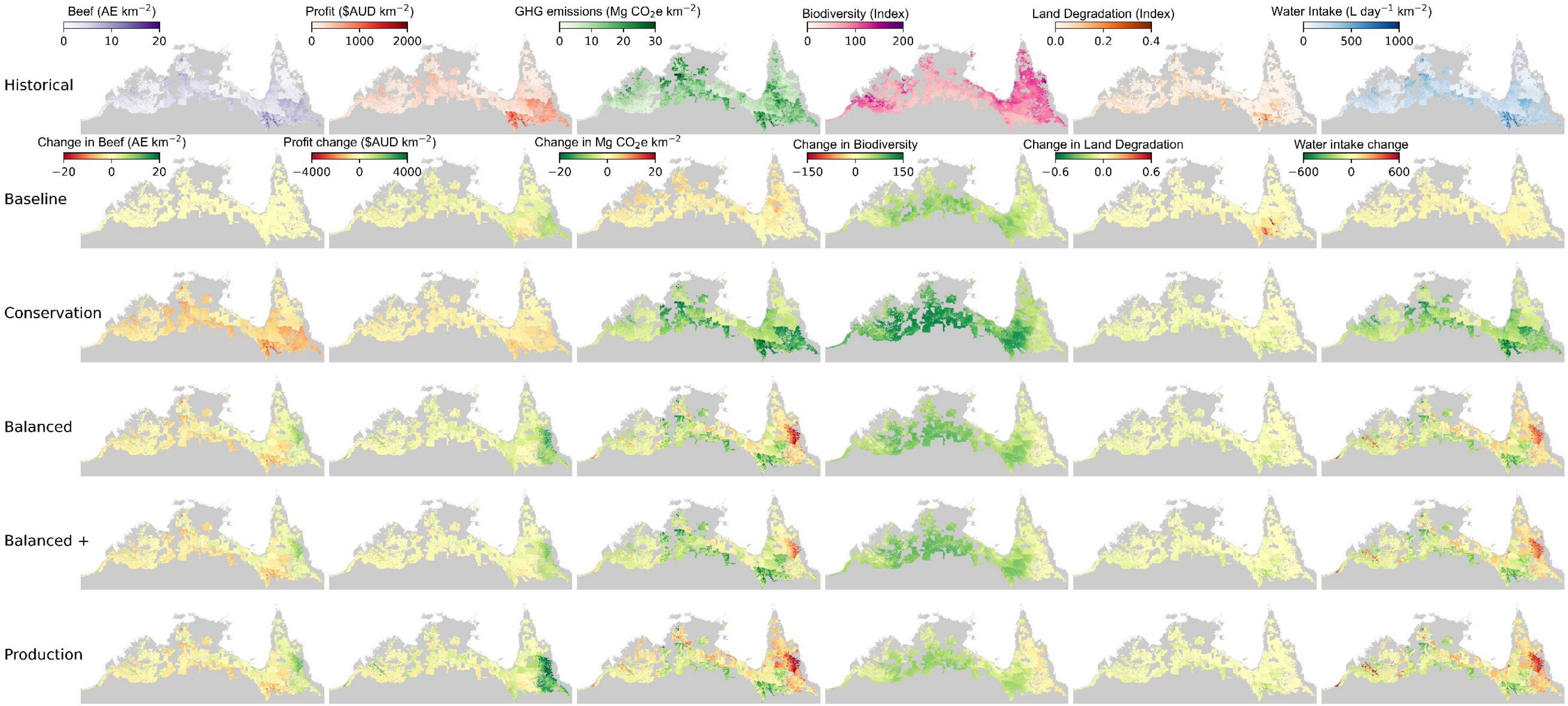


Figure S28 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM MPI and global outlook L1. "Historical" represents stocking rates and climatic conditions representative of the period from 1987-2010. The remaining rows show the change from historical conditions to 2050 for each outcome.

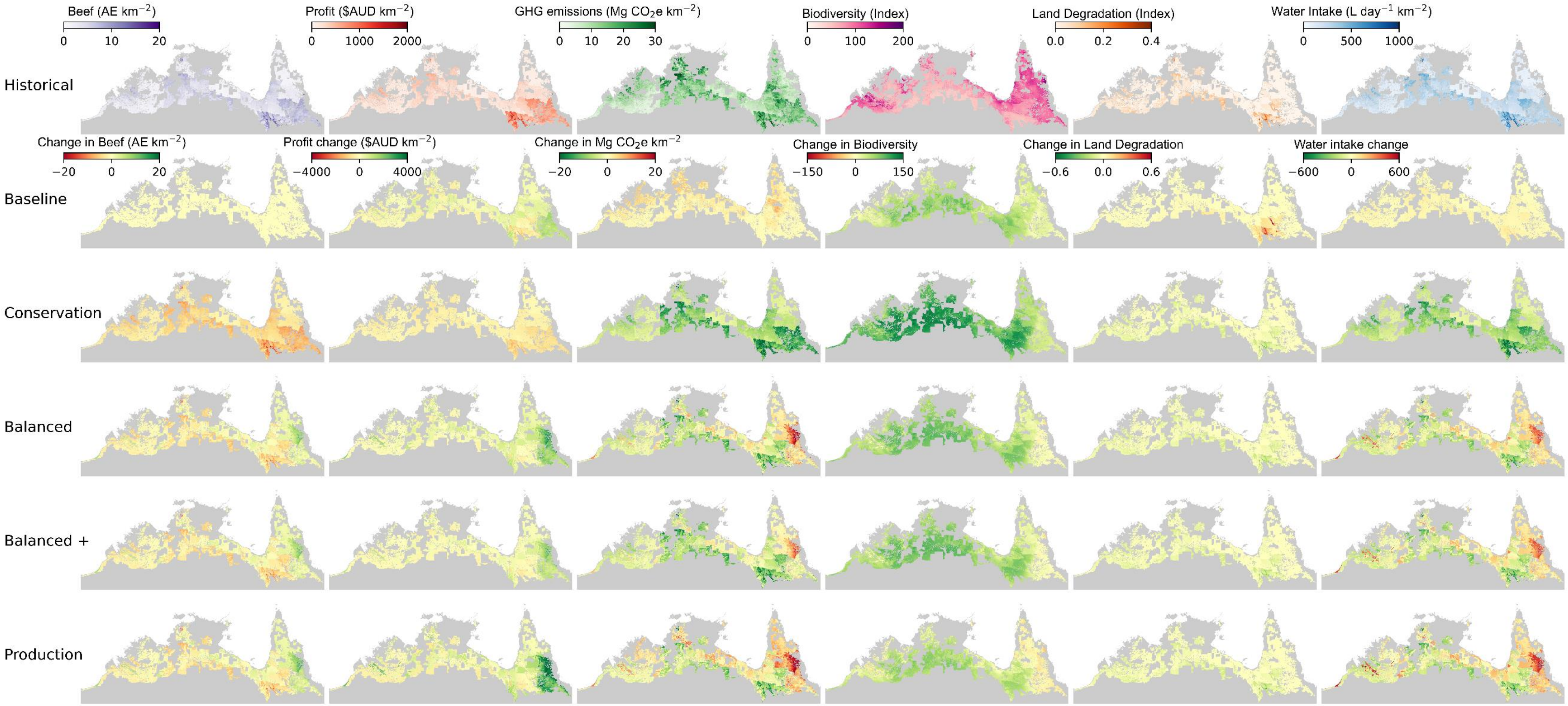


Figure S29 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM MR5 and global outlook L1. “Historical” represents stocking rates and climatic conditions representative of the period from 1987-2010. The remaining rows show the change from historical conditions to 2050 for each outcome.

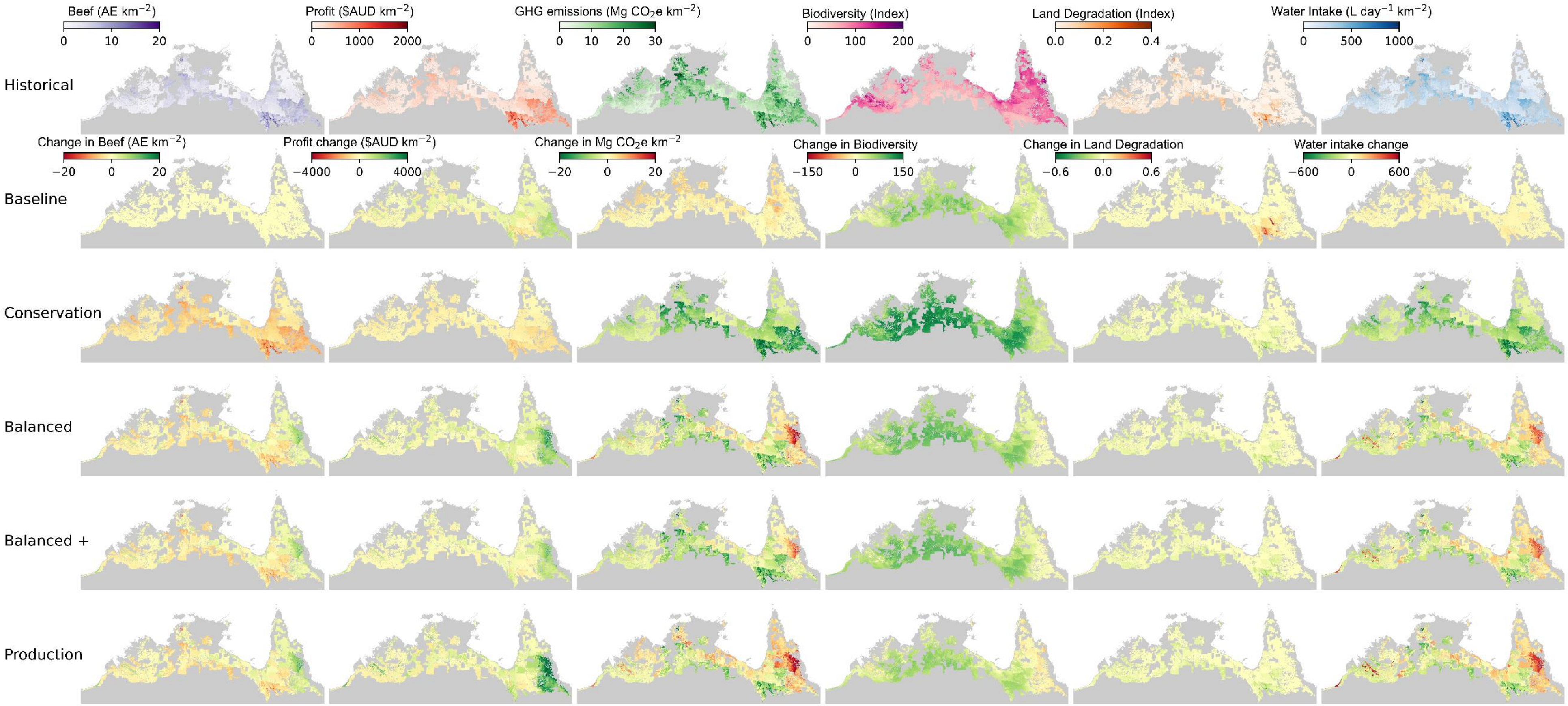


Figure S30 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM CE2 and global outlook M3.

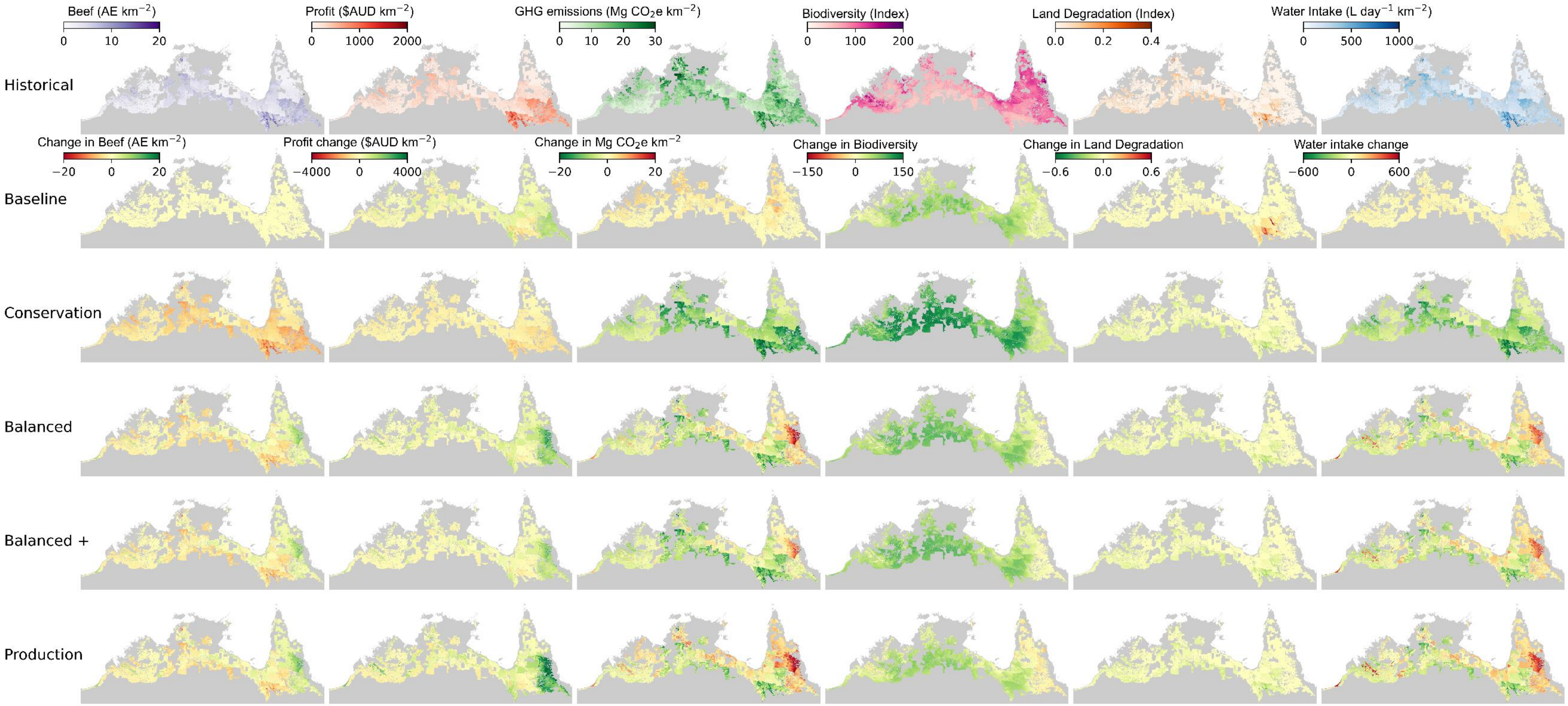


Figure S31 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM MR5 and global outlook M3.

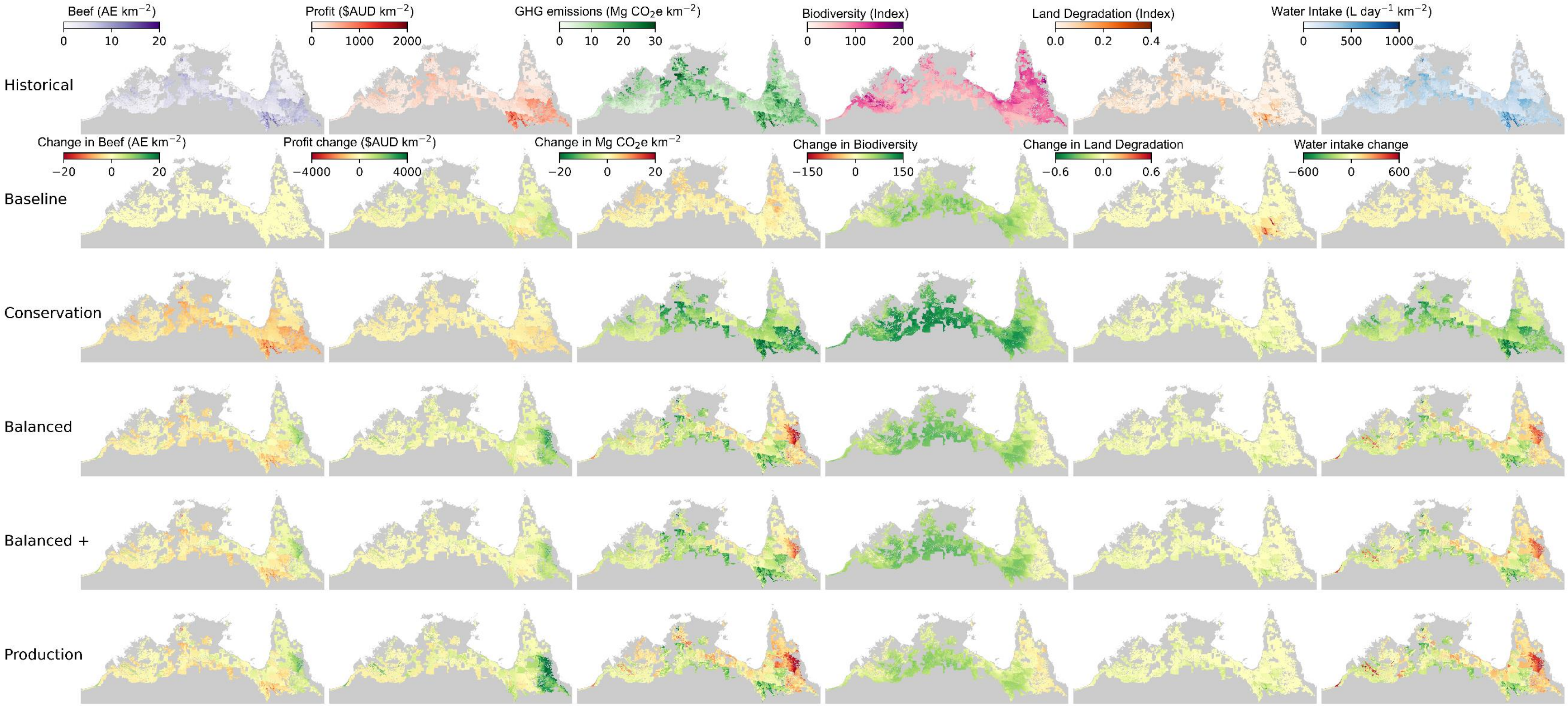


Figure S32 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM CE2 and global outlook M2.

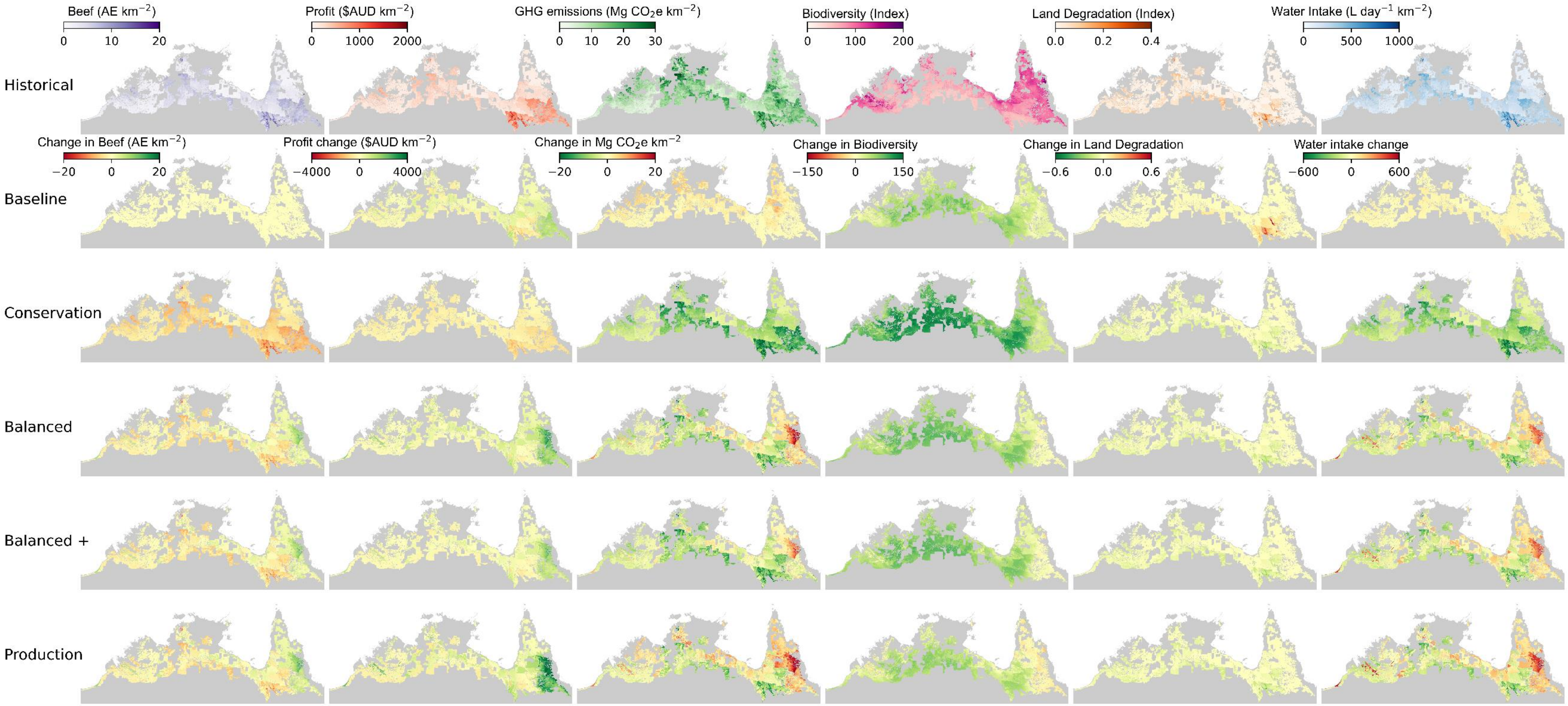


Figure S33 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM MPI and global outlook M2.

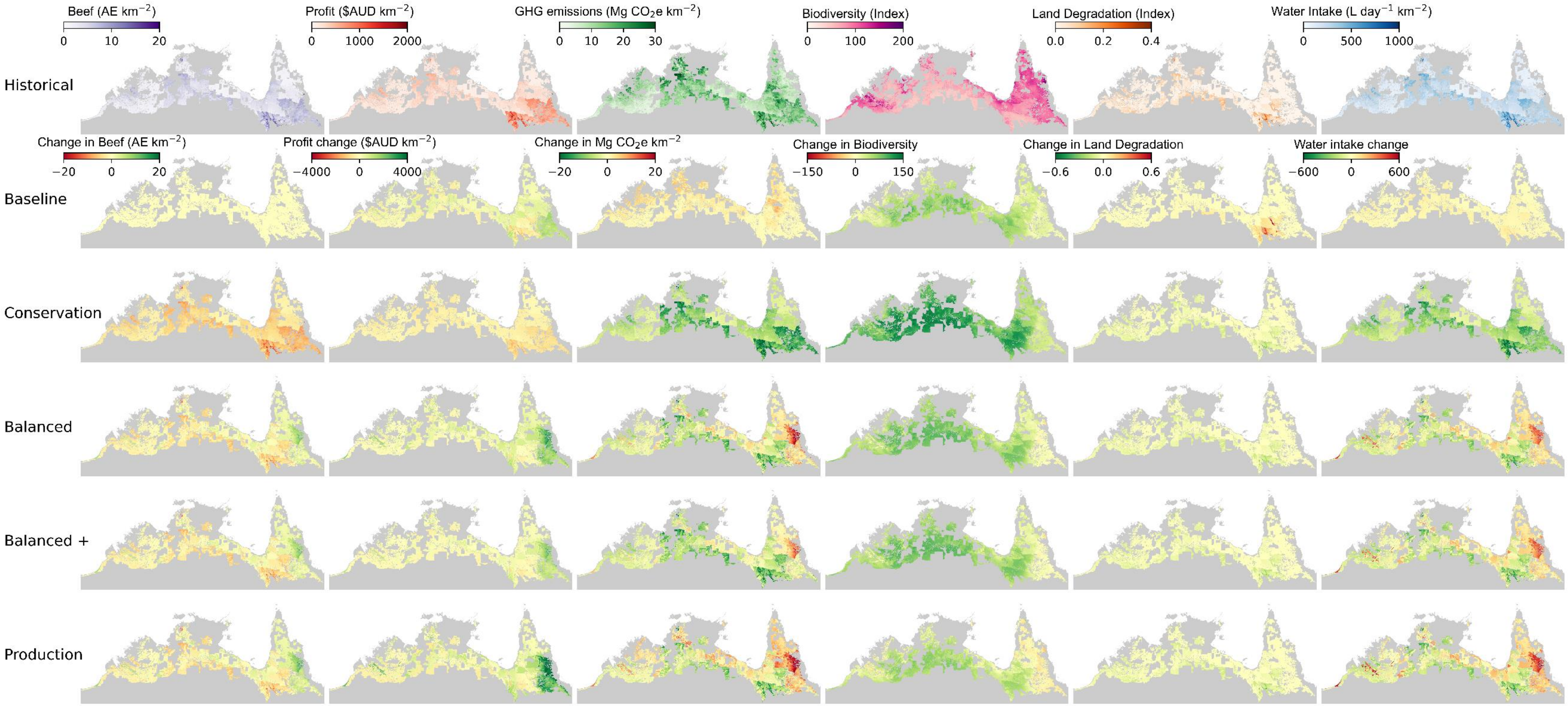


Figure S34 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM MR5 and global outlook M2.

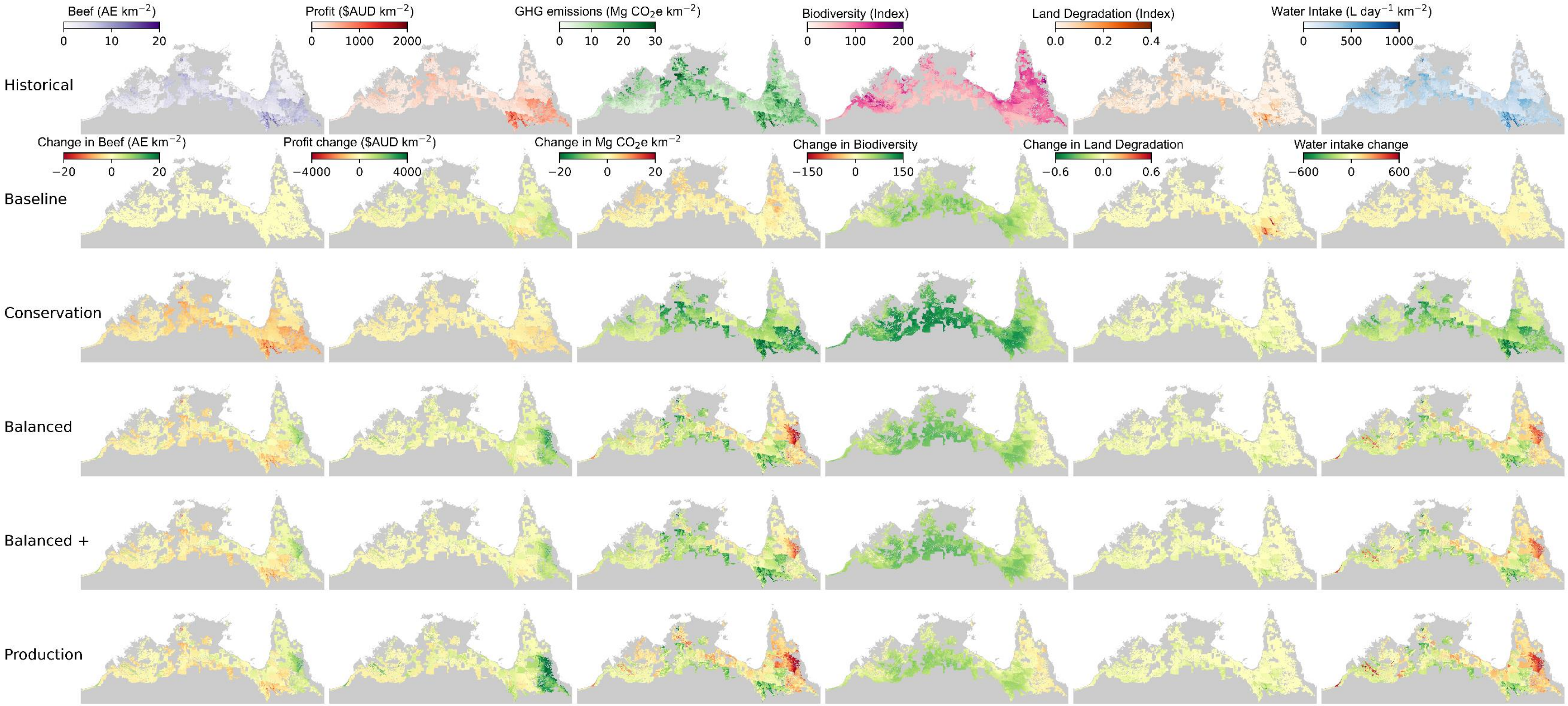


Figure S35 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM CE2 and global outlook H3.

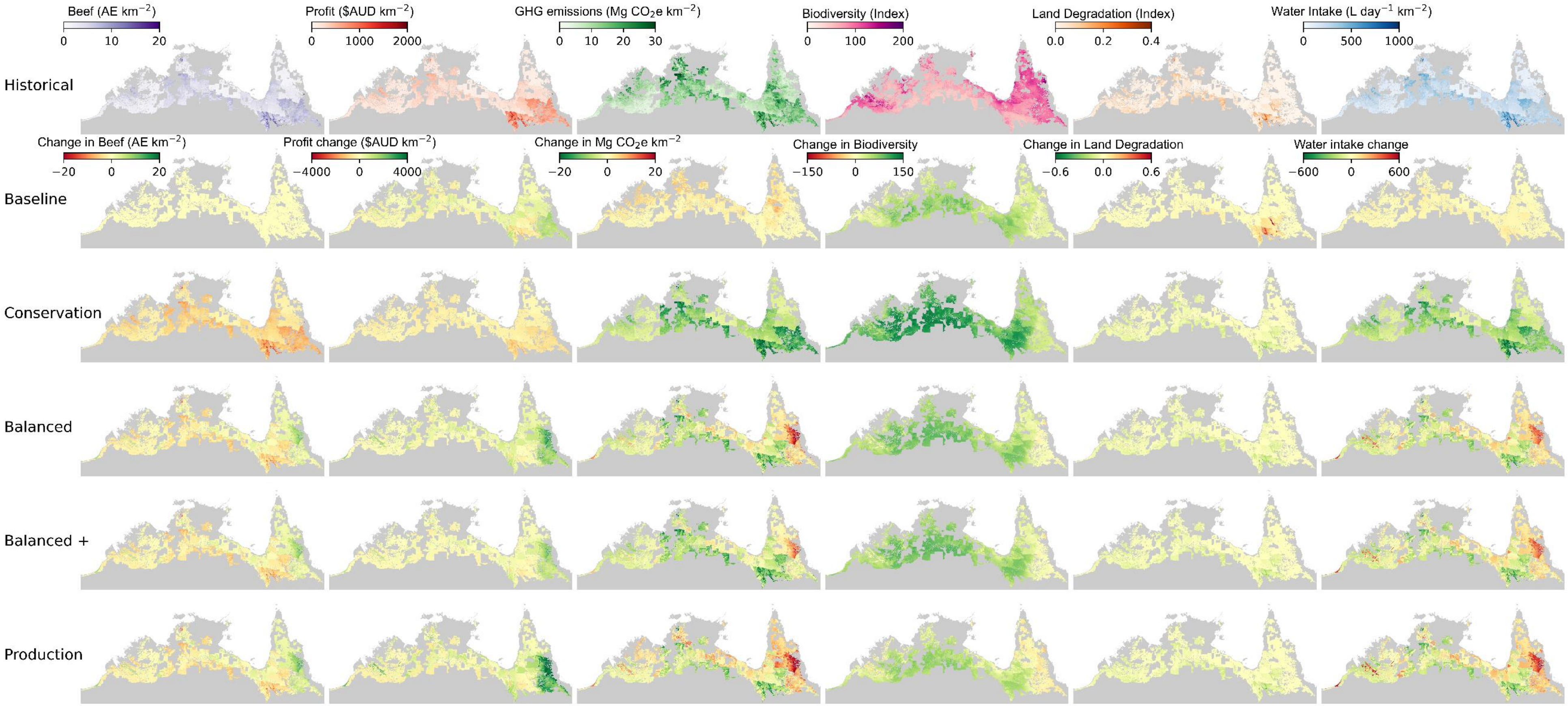


Figure S36 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM MPI and global outlook H3.

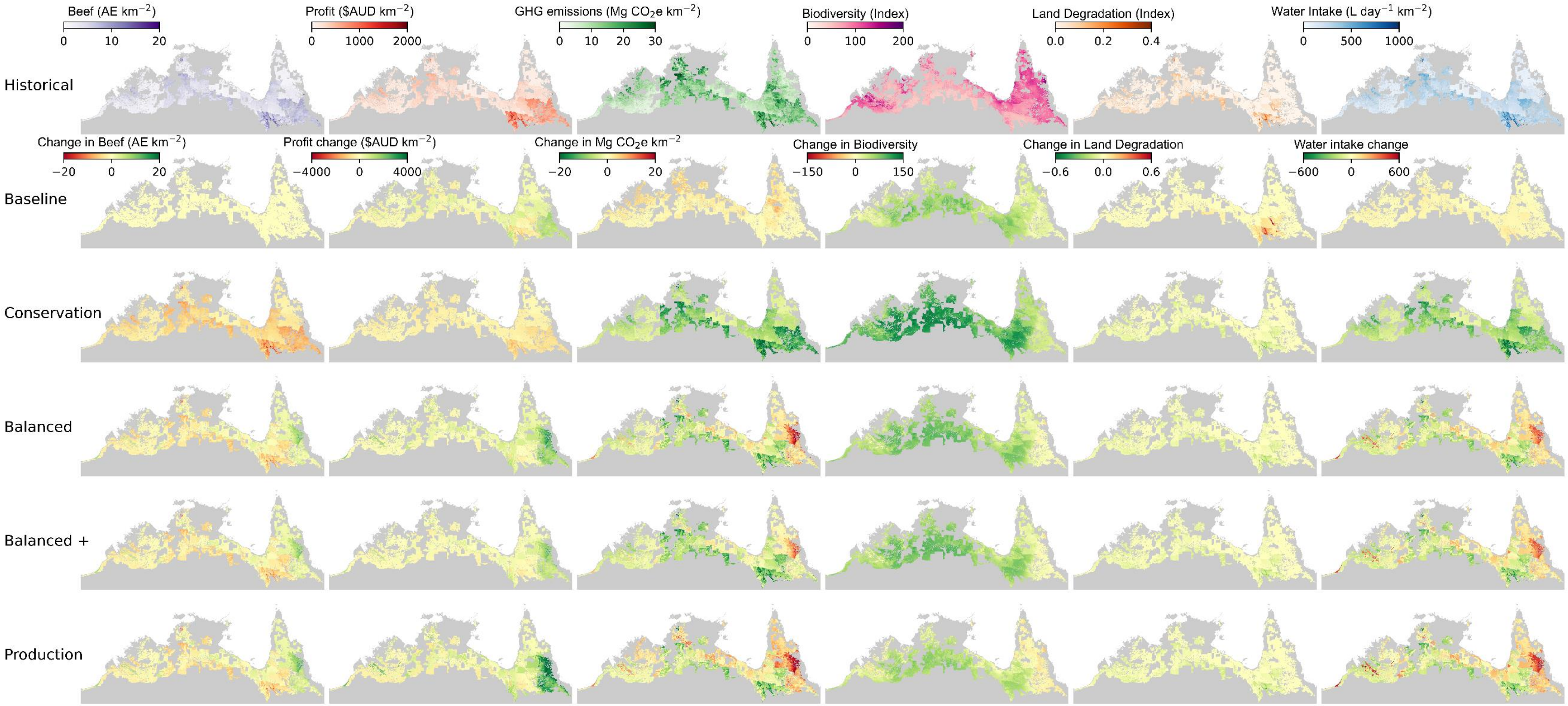


Figure S37 | Spatial variation in sustainability outcomes for the different future management scenarios in northern Australia by 2050 for GCM MR5 and global outlook H3.

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