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WILDLIFE RESEARCH

Controlling feral ruminants to reduce greenhouse gas emissions: a case study of buffalo in northern Australia

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ABSTRACT

Context. The bourgeoning carbon economy is creating novel ways to incentivise conservation management activities that have the co-benefits of reducing greenhouse gas (GHG) emissions and social inequality. **Aim.** To estimate the monetary value of carbon credits that landowners could generate by reducing ecologically destructive feral populations of the Asian water buffalo (*Bubalus bubalis*) in northern Australia. **Methods.** First, we estimated buffalo enteric emissions based on the population structure of feral buffalo in northern Australia, and discounted the reduction of fire emissions due to the consumption of grassy fuel by feral buffalo. We then predicted the change in buffalo population size across the South Alligator River region of Kakadu National Park under four buffalo management scenarios: (1) no buffalo control; (2) low-intensity buffalo control; (3) moderate-intensity buffalo control; and (4) high-intensity buffalo control. We quantified the reduction of GHG emissions under the three buffalo control scenarios, relative to the scenario of no buffalo control, while discounting the GHG emissions that directly result from buffalo control actions (e.g. helicopter emissions). **Key results.** All three buffalo control scenarios substantially reduced the estimated GHG emissions that would otherwise have been produced. The low-intensity buffalo control scenario was predicted to abate 790 513 t CO_2-e over the 20-year simulation, worth USD15 076 085 (or USD753 804 year^{-1}). Our high-intensity buffalo control scenario had the greatest reduction in GHG emissions, with a total net abatement of 913 231 t CO_2-e, worth USD17 176 437 (or USD858 822 year^{-1}). **Conclusions.** The potential value of carbon credits generated by controlling feral buffalo populations in northern Australian savannas far exceeds the management costs. **Implications.** The management of feral ruminants could be incentivised by the generation of carbon credits. Such management could simultaneously avoid GHG emissions, generate income for landowners and offer significant ecological benefits.

Keywords: carbon credits, climate change, conservation, feral herbivores, greenhouse gas emissions, northern Australia, ruminants, tropical savanna.

Introduction

In response to the increasingly evident effects of anthropogenic climate change, there is growing impetus to reduce greenhouse gas (GHG) emissions. Stemming from global commitments to reduce GHG emissions made under various treaties (i.e. Kyoto Protocol, Paris Agreement), the pricing of GHG pollution has emerged as a financial mechanism to incentivise GHG emissions reduction (Bradshaw *et al.* 2013). This has created new ways to incentivise conservation management activities that demonstrably reduce net GHG emissions. For example, the Australian Government's legislated offset scheme, the Emissions Reduction Fund (ERF), provides a financial basis for fire management across northern Australia's fire-prone tropical savannas via a carbon-crediting mechanism (Russell-Smith *et al.* 2013). The approved 'savanna burning' methodology allows land managers to earn an Australian Carbon Credit Unit (ACCU) for each (net) tonne of carbon dioxide equivalent (CO_2-e) prevented from release into the atmosphere (Corey *et al.* 2020). ACCUs can then be sold to third parties looking to offset their emissions. Such fire management programs now cover more than 380 000 km^2 of northern

Australia and are delivering important economic, social and ecological benefits (Russell-Smith *et al.* 2013; Edwards *et al.* 2021).

Given the scale of GHG emissions from the Australian agricultural sector, programs to reduce emissions associated with the management of livestock represent a key component of the ERF. In contrast, there has been little attention paid to the potential development of a methodology to incentivise the control of feral herbivores (Bradshaw *et al.* 2013). This is despite the devastating impacts that numerous species of feral herbivore have had on Australian ecosystems. Importantly, Australia is home to large feral populations of ruminants, including camel (*Camelus dromedarius*), cattle (*Bos taurus*, *Bos indicus*), buffalo (*Bubalus bubalis*), goat (*Capra hircus*) and numerous deer species (*Rusa unicolor*, *Dama dama*, *Cervus elephus*, *Axis axis*, *Axis porcinus*, *Rusa timorensis*). Ruminant digestion can produce large amounts of methane (CH_4), a potent GHG. For example, an individual camel or buffalo produces >50 kg CH_4 year^{-1}, equivalent to >1.25 t CO_2 year^{-1} (Bradshaw *et al.* 2013).

Quantifying the GHG emissions abatement that can be achieved through the management of populations of feral ruminants could lead to a market-based mechanism to incentivise their ongoing management. Although previous research has investigated the change in GHG emissions as a result of feral camel management in Australia (Drucker *et al.* 2010; Zeng 2015), to our knowledge no such work has focused on feral buffalo. Here we investigate the potential monetary value of carbon credits that could be generated by reducing feral buffalo populations in northern Australia. We focus on a region where the population dynamics of feral buffalo and the costs of their control have been well documented: the South Alligator River region of Kakadu National Park. We aimed to:

1. Estimate the enteric emissions from feral buffalo;
2. Use population models and published costs of buffalo control to investigate the net value of GHG abatement of feral buffalo control.

Materials and methods

Study system

Our study focused on the South Alligator River region of Kakadu National Park, 250 km east of Darwin in northern Australia (Fig. 1). This region experiences a tropical monsoonal climate with an intense wet season (November–April) followed by a dry season (May–October). Mean annual rainfall is

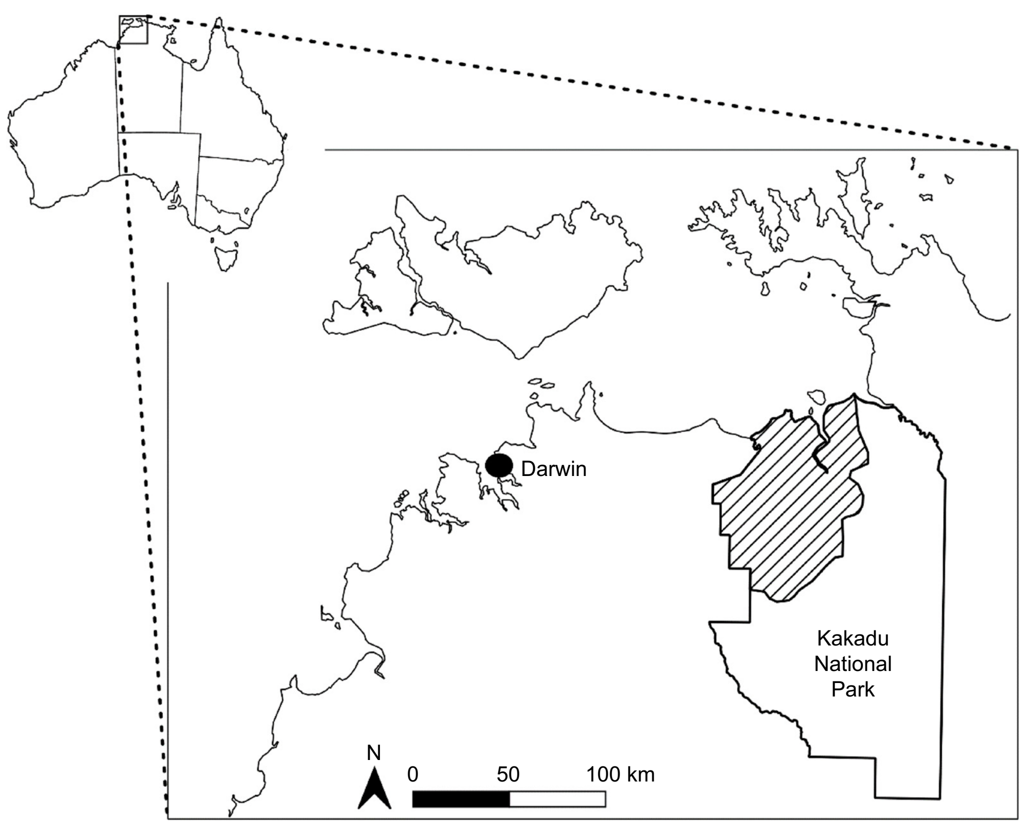

Fig. 1. The location of the South Alligator River region (hatched area), covering 5300 km^2 of Kakadu National Park in northern Australia.

~1500 mm. The major vegetation types are lowland savanna (characterised by moderately dense trees of *Eucalyptus miniata* and *Eucalyptus tetrodonta*, with a grassy understorey), as well as floodplain and *Melaleuca* forest.

Buffalo in northern Australia

The buffalo (*B. bubalis*) was introduced to northern Australia in the 19th century, and by the 1980s had reached densities as high as 34 km^{-2} on the South Alligator River floodplains, and 15 km^{-2} in the region's lowland savannas (Ridpath *et al.* 1983). Following the commencement of a major control programme in the late-1980s (the Brucellosis and Tuberculosis Eradication Campaign [BTEC]), buffalo densities decreased dramatically to <0.1 km^{-2} by the mid-1990s (Skeat *et al.* 1996). Although there have been numerous small-scale buffalo control operations since the cessation of BTEC in 1995, there has been no large-scale, coordinated control of buffalo. High buffalo densities cause significant ecological impacts, including a reduction in vegetation biomass, changes to species composition, soil compaction and erosion, changes to surface hydrology including the intrusion of saltwater into freshwater swamps, and reduced water quality (Skeat *et al.* 1996; Werner 2005). It has been suggested that these impacts drive long-term ecological cascades (Petty *et al.* 2007), and there is growing evidence of the role of large feral herbivores, such as buffalo, in the widespread collapse of northern Australian native mammal populations (Legge *et al.* 2019; Davies *et al.* 2020; Stobo-Wilson *et al.* 2020). However, it is important to note that the well documented collapse of native mammal populations in Kakadu National Park coincided with the BTEC (Woinarski *et al.* 2001), with mammal decline continuing despite the massive reduction in densities of feral herbivores.

Estimating enteric emissions from feral buffalo

We calculated the net GHG emissions from feral buffalo populations using estimates of buffalo enteric emissions (CH_4), adjusted for both buffalo population structure and the reduced emissions from fire in lowland savanna (CH_4 and N_2O) resulting from the reduction of grassy fuel loads due to buffalo consumption. The emissions of carbon dioxide (CO_2) were not accounted for based on the assumption that savanna fires produce no net CO_2 flux. All GHG emissions were converted to a CO_2 equivalent (i.e. standardising for the global warming potential of each gas), expressed in units of t CO_2-e $year^{-1}$.

Enteric emissions (E_e)

The enteric emissions produced by ruminants are determined by a number of factors, including the rate and volume of feed intake, and the conversion rate of feed energy to CH_4 (Calvo Buendia *et al.* 2019). These factors depend on feed quality (with lower-quality feed generally having higher rates of conversion to CH_4) as well as the demographic structure of the ruminant population, which determines the feed biomass consumed (e.g. the proportion of adults and juveniles, reproductive stages, weight etc.). In absence of the detailed information required to estimate CH_4 emissions from feral buffalo across northern Australia using higher-tier Intergovernmental Panel on Climate Change (IPCC) methods (i.e. Tiers 2 and 3), we used the Tier 1 method (Calvo Buendia *et al.* 2019). Tier 1 methods use default emissions factors based on previous studies, across different regions. Our analysis is based on the estimate that enteric emissions from an individual adult buffalo contributes 76 kg CH_4 $year^{-1}$, equivalent to 2.1 t CO_2 $year^{-1}$.

Adjusting E_e for population structure

Because feral buffalo populations in northern Australia include both adults and juveniles, we adjusted the estimate of buffalo enteric emissions based on the structure of these populations. To estimate the relative proportion of juveniles and adults, we used the buffalo age frequency distributions from four northern Australian buffalo populations harvested between 2006 and 2008 by McMahon *et al.* (2011). Because buffalo in northern Australia become reproductively active between 2 and 3 years of age (Tulloch and Grassia 1981), we classified juvenile buffalo as those <2 years of age, and adult buffalo as those ≥2 years of age. On average across the four populations, the proportion of juveniles (P_j) and adults (P_a) was 43% and 57% respectively. We acknowledge that the accuracy of this population structure is contingent on an unbiased sampling protocol, but note that they are likely reliable because entire family groups were culled at similar times of the year (McMahon *et al.* 2011). We assumed that a juvenile buffalo produces half the enteric emissions of an adult (38 kg CH_4 $year^{-1}$, equivalent to 1.05 t CO_2-e $year^{-1}$).

We estimated the enteric emissions (E_e) from northern Australian feral buffalo populations as:

$$E_e = (D \times P_j \times E_j) + (D \times P_a \times E_a)$$

where D is the population density of feral buffalo, P_j is the proportion of the buffalo population that are juveniles (0.43), E_j is the annual enteric emissions of juvenile buffalo (1.05 t CO_2-e $year^{-1}$), P_a is the proportion of the buffalo population that are adults (0.57), E_j is the annual enteric emissions of adult buffalo (2.1 t CO_2-e $year^{-1}$).

Accounting for the reduction in fire emissions from lowland savanna due to the consumption of grassy fuel by feral buffalo

Feral buffalo can consume large amounts of herbaceous plant biomass, particularly grasses (Bowman *et al.* 2010). As a result, they have the potential to reduce fuel loads across savanna landscapes, thereby reducing GHG emissions due to fire. We accounted for this by estimating the net emissions from feral buffalo in lowland savanna (E_s) as:

$$E_s = E_e - E_f$$

where E_e is estimated enteric emissions (see above), and E_f is estimated reduction in fire emissions due to the consumption of grassy fuels by buffalo in northern Australian lowland savannas. Because adult and juvenile buffalo consume different amounts of vegetation, we again adjusted our estimates based on feral buffalo population structure (outlined above) such that:

$$E_f = (E_{fba} - E_{fjb}) + (E_{fba} - E_{fab})$$

where E_{fba} is the estimated emissions from fire in the absence of feral buffalo, E_{fjb} is the estimated emissions from fire at a given density of juvenile buffalo, and E_{fab} is the estimated emissions from fire at a given density of adult buffalo.

Following the methods outlined by Cook *et al.* (2016), we estimated the emissions from a typical lowland savanna fire regime in the absence of feral buffalo (E_{fba}). First, we estimated the mean proportion of fuel remaining after a fire under a particular fire regime, R, as:

$$R = 1 - \frac{B_E U_E f_E + B_L U_L f_L}{f_E + f_L}$$

where B_E and B_L are the mean burning efficiencies in the early and late dry season respectively, U_E and U_L is the burn uniformity (i.e. the proportion of area within a fire scar that remains unburnt) in the early and late dry season respectively, and F_E and F_L is the mean fire frequency in the early and late dry season respectively (Cook *et al.* 2016). These values were chosen to represent a typical lowland savanna fire regime (Table 1).

We then calculated the mean fire return interval, r, as:

$$r = \frac{1}{f_E + f_L}$$

The maximum grass fuel load, Φ_{max}, was calculated as:

$$\Phi_{max} = \frac{L}{k}$$

where L is grass fuel load and k is the grass turnover rate.

The mean post-fire residue (i.e. dead organic matter remaining after a fire), $\Phi(0)$, was:

$$\Phi(0) = \Phi_{max} \frac{(1 - e^{-kr})}{(\frac{1}{R} - e^{-kr})}$$

The mean fuel load when a fire occurs, Φ_r, was:

$$\Phi_r = \Phi_{max} - (\Phi_{max} - \Phi(0))e^{-kr}$$

The mean mass of fuel emitted as gas, E_Φ, was:

$$E_\Phi = [\Phi_r - \Phi(0)][f_E + f_L]$$

The mean annual emissions of methane, E_{CH_4}, was then calculated as:

$$E_{CH_4} = 1.333 E_\Phi EF_{CH_4} \gamma$$

where EF_{CH_4} is the emission factor for methane, γ is the fuel carbon content and 1.333 is the molecular to elemental mass ratio for CH_4.

The mean annual emissions of N_2O, E_{N_2O}, was then calculated as:

$$E_{N_2O} = 1.571 E_\Phi EF_{N_2O} \gamma NC$$

where EF_{N_2O} is the emission factor for N_2O, γ is the fuel carbon content, 1.571 is the molecular to elemental mass ratio for N_2O, and NC is the fuel nitrogen to carbon ratio.

The total fire emissions in the absence of feral buffalo (E_{fba}) was then calculated as:

$$E_{fba} = (E_{CH_4} \times GWP_{CH_4}) + (E_{N_2O} \times GWP_{N_2O})$$

where GWP_{CH_4} and GWP_{N_2O} are the global warming potentials of CH_4 and N_2O, respectively, expressed relative to CO_2 (28 and 265 respectively).

To estimate the change in emissions due to the consumption of grassy fuel by buffalo, we estimated the emissions from a typical lowland savanna fire regime in the presence of juvenile (E_{fjb}) and adult (E_{fab}) buffalo. To do this, we followed the same approach as outlined above, but reduced the maximum grass fuel load estimate based on the amount of grass consumed by buffalos, such that:

$$\Phi_{max} = \frac{L - (D \times G)}{k}$$

where D is the population density of juvenile or adult buffalo, and G is the amount of grass consumed per individual juvenile or adult buffalo (t ha^{-1} $year^{-1}$).

Estimates of the daily consumption of vegetation by adult feral buffalo range from around 4–6 kg day^{-1} (Williams and Dudzinski 1982; Williams and Ridpath 1982), to as high as 30 kg day^{-1} (Jesser *et al.* 2016). To ensure our estimate of buffalo emissions was conservative, we used 30 kg day^{-1} when estimating the reduction in fire emissions due to the consumption of vegetation by adult buffalo. We assumed that consumption by juvenile buffalo was half of adult consumption (i.e. 15 kg day^{-1}). We note here that our approach assumes buffalo are only eating grass (not browsing on woody plants), that the reduction in fire emissions from buffalo consumption in habitats other than lowland savanna (i.e. floodplains) are negligible, and that fire intensity remains constant. The latter assumption is similar to other GHG

Table 1. Inputs for estimating the emissions from a typical lowland savanna fire regime in the absence of feral buffalo (E_{fba}), and when juvenile (E_{fjb}) and adult (E_{fab}) buffalo are present.

Inputs	Description	No buffalo (E_{fba})	Juvenile buffalo (E_{fjb})	Adult buffalo (E_{fab})
$G_{buffalo}$	Mean grass consumption by buffalo (kg buffalo^{-1} day^{-1})	0.000	15.000	30.000
f_E	Mean fire frequency in the EDS (fires year^{-1})	0.333	0.333	0.333
f_L	Mean fire frequency in the LDS (fires year^{-1})	0.167	0.167	0.167
B_E	Burning efficiency in the EDS (proportion)	0.658	0.658	0.658
B_L	Burning efficiency in the LDS (proportion)	0.761	0.761	0.761
U_E	Uniformity in the EDS (proportion)	0.709	0.709	0.709
U_L	Uniformity in the LDS (proportion)	0.889	0.889	0.889
R	Mean proportion of the fuel remaining after a fire	0.464	0.464	0.464
r	Mean fire return interval (years)	2.000	2.000	2.000
$L_{G-buffalo}$	Grass input (t ha^{-1} year^{-1}), with buffalo consumption deducted	0.660	0.528	0.310
k_G	Grass turnover (year^{-1})	0.615	0.615	0.615
Φ_{max}	Maximum grass fuel load (t ha^{-1})	1.073	0.859	0.505
$\Phi(0)$	Mean post-fire grass residue (t ha^{-1})	0.407	0.326	0.191
Φ_r	Mean fuel load when a fire occurs (t ha^{-1})	0.879	0.703	0.413
E_Φ	Mean mass of fuel emitted as gases (t ha^{-1} year^{-1})	0.236	0.189	0.111
γ	Carbon content of grass (proportion)	0.460	0.460	0.460
NC	Nitrogen to carbon ratio of grass	0.010	0.010	0.010
EF_{CH_4}	Emission factor for methane	0.003	0.003	0.003
EF_{N_2O}	Emission factor for nitrous oxide	0.008	0.008	0.008
E_{CH_4}	Mean emissions of methane (t ha^{-1} year^{-1})	4.47E−04	3.58E−04	2.11E−04
E_{N_2O}	Mean emissions of nitrous oxide (t ha^{-1} year^{-1})	1.23E−05	9.81E−06	5.76E−06
GWP_{CH_4}	Global warming potential of methane, relative to CO_2	28.000	28.000	28.000
GWP_{N_2O}	Global warming potential of nitrous oxide, relative to CO_2	265.000	265.000	265.000
$E_{CH_4\ (CO_2\text{-e})}$	Mean emissions of methane, converted to CO_2-e (t ha^{-1} year^{-1})	0.011	0.009	0.005
$E_{N_2O\ (CO_2\text{-e})}$	Mean emissions of nitrous oxide, converted to CO_2-e (t ha^{-1} year^{-1})	0.004	0.003	0.002
$E_{CH_4 + N_2O\ (CO_2\text{-e})}$	Mean emissions of both, converted to CO_2-e (t ha^{-1} year^{-1})	0.015	0.012	0.007

accounting approaches for Australian savannas, which assume that fire intensity is not affected by reductions in fire frequency (Russell-Smith *et al.* 2009; Cook *et al.* 2016). All inputs are summarised in Table 1.

Combining our estimate of enteric emissions with spatially explicit buffalo population models

To gauge the potential of greenhouse gas abatement of feral buffalo control, we combined our estimates of E_e and E_s with spatially explicit population models to contextualise the potential emission reductions within a realistic management framework. Importantly, our management scenarios were costed, thereby permitting us to gauge whether the potential economic benefit of reducing buffalo emissions (by generating and selling carbon credits) outweigh the cost of applying such management. To do this, we explored the net value of GHG abatement from feral buffalo control under different management scenarios across the South Alligator River region of Kakadu National Park.

Buffalo population model inputs

We used the Spatio-Temporal Animal Reduction (STAR) model, developed by McMahon *et al.* (2010) for several feral species in Kakadu National Park, to predict buffalo population change across Kakadu's South Alligator River region. We examined four scenarios over a 20-year period: (1) no buffalo control; (2) low-intensity buffalo control; (3) moderate-intensity buffalo control; and (4) high-intensity buffalo control. We based our management scenarios on the pre-specified management scenarios designed by McMahon *et al.* (2010) to examine how populations will change over time with no management actions (i.e. no buffalo control scenario), and those designed to reduce feral buffalo densities to specified target levels (i.e. our low-, moderate- and

Table 2. Parameter inputs for our buffalo management scenarios.

	No buffalo control	Low-intensity buffalo control	Moderate-intensity buffalo control	High-intensity buffalo control
Initial cull (proportion of population)	–	0.20	0.40	0.63
Maintenance cull (proportion of population)	–	0.17	0.25	0.38
Control target density (proportion of initial population size)	–	0.50	0.25	0.05
Initial buffalo density (buffalo/km^{-2})	0.93	0.93	0.93	0.93
Duration (years)	20.00	20.00	20.00	20.00
Cell size (km^2)	100.00	100.00	100.00	100.00
Helicopter cost (USD/h)	–	937.00	937.00	937.00
Other aerial culling costs (USD/h)	–	356.00	356.00	356.00

high-intensity buffalo control scenarios). Our model inputs are summarised in Table 2. We used the same population demographic parameters (i.e. carrying capacity, maximum intrinsic population growth rate, growth response shape parameter and dispersal probability) as outlined in McMahon *et al.* (2010). We estimated an initial buffalo density of 0.93 km^{-2}. This estimate was based on the density estimate across Kakadu in 1996 (0.1 km^{-2}, Skeat *et al.* 1996), adjusted for the habitat types across the South Alligator River region, and an annual population growth rate of 5.25% (Saalfeld 2014). Given the absence of a robust estimate of current buffalo density, this estimate represents our 'best guess' but is potentially inaccurate. For our culling scenarios, we adjusted the logistical costs associated with aerial culling based on a recently quoted price for aerial buffalo culling in comparable habitat. The cost of helicopter hire was quoted as USD937 h^{-1}, and other costs (including labour, ammunition, food, accommodation, reporting and administration) as USD356 h^{-1}.

Combining population model outputs with estimates of enteric emissions to estimate potential carbon credits

Our population model outputs included the predicted change in buffalo population size across the South Alligator River under our four buffalo management scenarios, as well as the number of animals culled each year, the annual number of helicopter hours required, and an estimated annual cost of management.

Using our estimates of buffalo enteric emissions, we quantified the GHG emissions from buffalo enteric emissions across the South Alligator River region for each of our management scenarios. To do this, we multiplied our estimates of buffalo enteric emissions (t CO_2-e km^{-1} year^{-1}) for the predicted density of buffalo (buffalo km^{-1}) across the entire South Alligator River region. Given the different habitat types, we multiplied E_s across the 2300 km^2 of lowland savanna, and E_e across the 3000 km^2 of floodplain, wetlands, and paperbark forest. The coarse spatial scale of the model (100 km^2) may reduce its ability for accurate predictions on which to base management actions; however, it still offers a useful heuristic tool for the purposes of this study (McMahon *et al.* 2010). We also acknowledge that this approach assumes an even density of buffalo across the lowland savannas and other habitat types of the South Alligator River region. Because buffalo density may be higher in floodplain areas (compared with savanna), this may overestimate the reduction of GHG emissions from fire across the region, thus producing a conservative estimate of the potential GHG abatement from buffalo management.

To gauge the potential GHG abatement from our buffalo management scenarios, we subtracted our estimates of annual buffalo enteric emissions across the South Alligator River region (Fig. 2) – predicted by our low, moderate and high buffalo control scenarios – from our no buffalo control scenario. As such, our annual GHG abatement estimates were quantified against the shifting baseline of buffalo population growth predicted under no buffalo control. This differs from established savanna burning methodology where a baseline estimate of emissions is quantified over a period of no fire management, against which the annual GHG abatement from fire management is calculated.

Once we had quantified the reduction in GHG emissions arising from our buffalo control scenarios, we estimated the net value of abatement. We did this by first calculating the dollar value for the amount of GHG emissions abatement using the current market price of around USD22 for an Australian carbon credit unit (ACCU; equivalent to 1 t CO_2-e stored or avoided by a project) (https://www.renewableenergyhub.com.au/market-prices/ accessed on 17/3/2022). To account for the cost of applying each management scenario, we subtracted the estimated cost of applying each management scenario.

To accurately quantify the net value of GHG abatement we also needed to estimate the GHG emitted during the application of management actions. Culling feral animals from aircraft is an effective, commonly used approach in northern Australian landscapes. To account for the GHG emissions released during our buffalo control scenarios, we used the

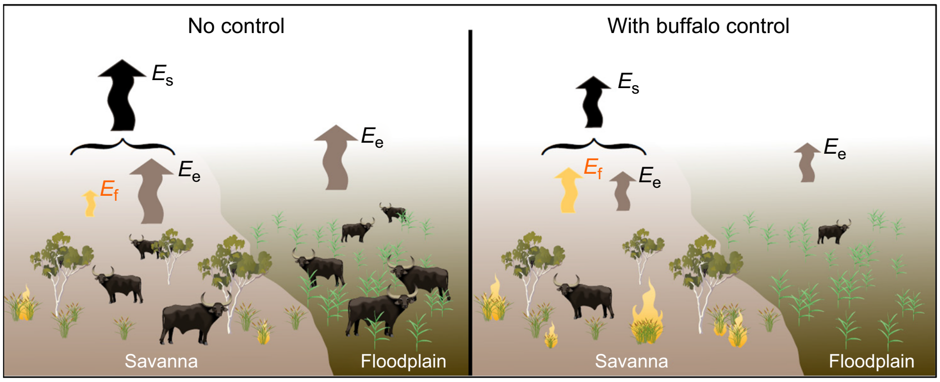

Fig. 2. A conceptual diagram of how buffalo emissions were estimated across the different habitat types of the South Alligator River region of Kakadu National Park. E_e is the estimated buffalo enteric emissions (adjusted for population structure), and E_s is the estimated net emissions from buffalo in lowland savanna (i.e. E_e adjusted for the reduction in fire emission [E_f] resulting from the consumption of grassy fuel).

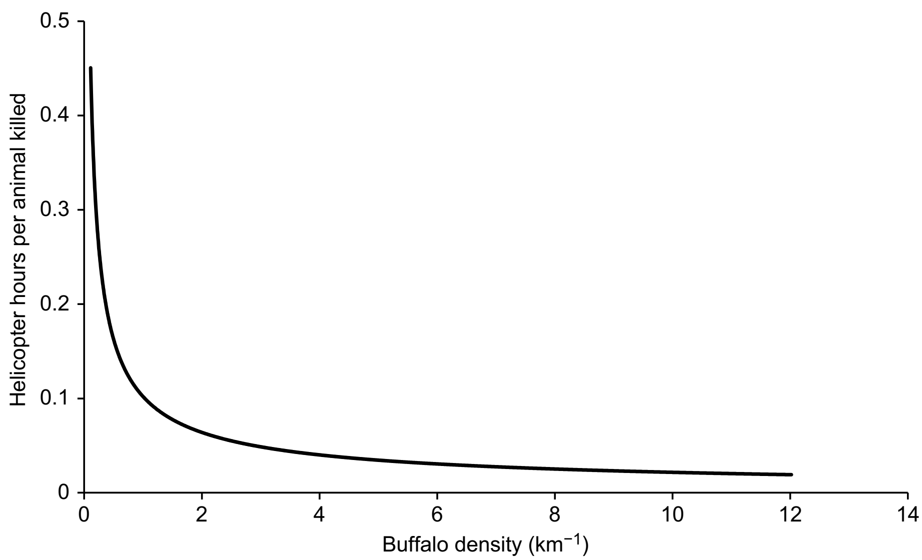

Fig. 3. The relationship between feral buffalo density and the helicopter hours required per animal culled (adapted from McMahon *et al.* 2010).

relationship between feral buffalo density and the helicopter flying time required per buffalo killed (McMahon *et al.* 2010) (Fig. 3). We used this relationship to combine the annual buffalo density, and number of individual buffalo culled each year under our management scenarios, to calculate the total annual helicopter hours. We then combined the number of helicopter hours with information on fuel consumption rate for a suitable helicopter model (Bell 206 Jetranger) of 120 L h^{-1} (Department of Environment and Conservation 2011) to calculate the total fuel consumed each year. We then combined the total amount of fuel (L) consumed each year with the emission factors per litre consumed of the fuel type (Jet A1 kerosene-type jet fuel: 2.58 kg CO_2-e L^{-1}) for this aircraft (Solomon *et al.* 2007) to estimate the annual amount of GHG emissions from each buffalo control scenario. Our analysis was guided by the Australian Government-approved methods for determining the abatement of greenhouse gas from savanna fire management (Commonwealth of Australia 2015, 2018), and therefore did not include emissions from the extraction, refinement or transportation of fuel.

We were then able to calculate the net value of abatement as: the estimated dollar amount of ACCUs generated through the reduction in buffalo enteric GHG emissions (compared to no management) minus the cost of each management scenario, minus the dollar amount of ACCUs lost due to the emissions of GHG during control operations.

Results

Estimated enteric emissions from feral buffalo

By combining IPCC estimates of buffalo enteric emissions, with information of buffalo population structure, we established the relationship between buffalo density and the estimated enteric emissions from buffalo (E_e) (Fig. 4*a*). To estimate the net emissions from buffalo in lowland savanna, we accounted for the reduction in fire emissions due the consumption of grassy fuels by buffalo (Fig. 4*b*).

Combining our estimate of enteric emissions with buffalo population models

With no population control, the number of feral buffalo across the South Alligator River region of Kakadu National Park increased from 4942 to 61 541 individuals (Fig. 5). This increase in abundance corresponds to an estimated increase in annual GHG emissions from 7792 t CO_2-e in the first year, to 97 282 t CO_2-e by the 20th year, totalling 951 703 t CO_2-e emitted over the 20-year simulation. This predicted population trajectory (and associated emissions) sets the baseline scenario against which we estimated the potential value of abatement from our three feral buffalo management scenarios.

Under our low-intensity buffalo control scenario, the buffalo population increased from 4942 to 7673 individuals, remaining well below that predicted with no population control (Fig. 5). The population reduction achieved by our low-intensity buffalo control scenario was estimated to reduce the total amount of emissions by 791 158 t CO_2-e. Our low-intensity buffalo control scenario cost a total of USD2 337 953. This scenario required a total of 2080

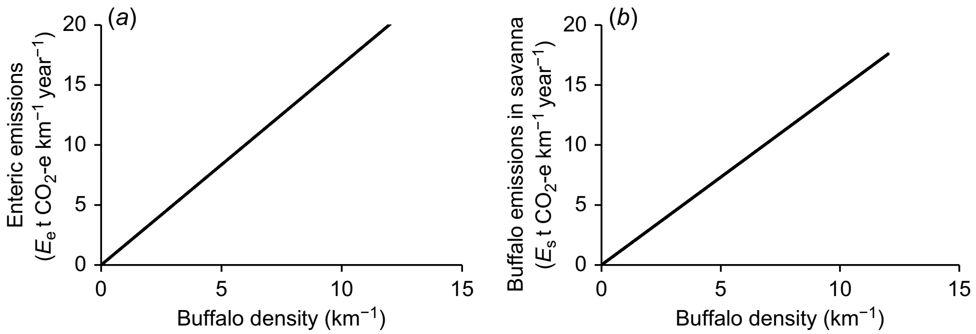

Fig. 4. The estimated relationships between buffalo density and (*a*) enteric emissions E_e and (*b*) net emissions from buffalo in savannas E_s (i.e. accounting for the reduction in fire emissions due to the consumption of grassy fuels).

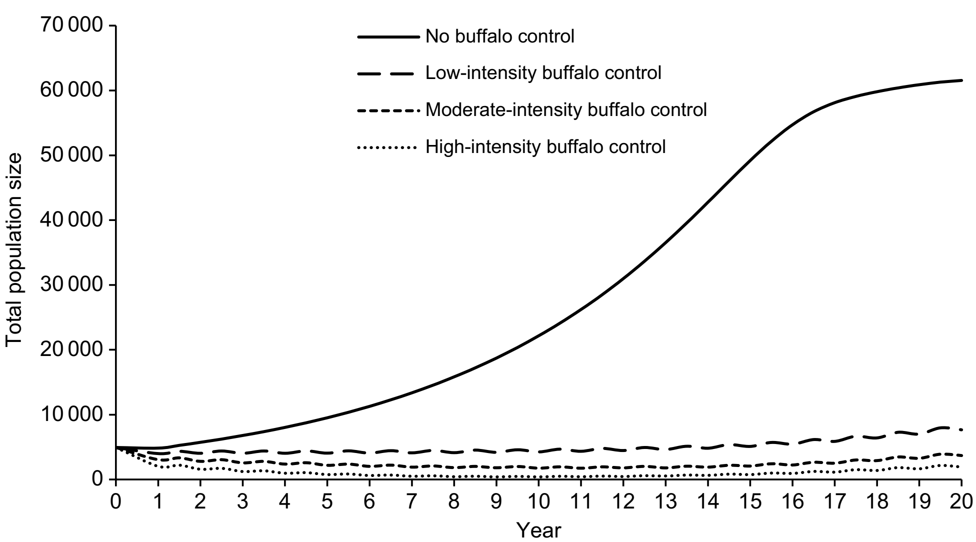

Fig. 5. Predictions of the total population size of feral buffalo across the South Alligator River region of Kakadu over 20 years under four different management scenarios. The 'No buffalo control' population trajectory is the scenario against which the net value of GHG emissions abatement from buffalo population reduction was quantified.

helicopter hours, resulting in the emission of 645 t CO_2-e, and an estimated net abatement of 790 513 t CO_2-e. This abatement equates to USD17 399 189 worth of carbon credits. Subtracting the estimated total cost of implementing our low-intensity buffalo control scenario resulted in a net potential value of abatement over 20 years of USD15 061 236 (or USD753 062 year^{-1}) (Fig. 6).

Under our moderate-intensity buffalo control scenario, buffalo population size decreased from 4942 to 3699 individuals (Fig. 5), abating 870 844 t CO_2-e for a total cost of USD2 657 097. This management scenario required 2503 helicopter hours, estimated to emit 776 t CO_2-e. For this scenario, we estimated a net abatement of 870 068 t CO_2-e, equal to USD19 150 197 worth of carbon credits. Subtracting the estimated total cost of implementing our moderate-intensity buffalo control scenario resulted in a net potential value of abatement of USD16 493 100 (or USD824 655 year^{-1}) (Fig. 6).

Our high-intensity buffalo control scenario achieved the largest reduction in feral buffalo population size (Fig. 5), an estimated abatement of 914 248 t CO_2-e, and cost a total of USD2 940 695. This management scenario required a total of 3277 helicopter hours, emitting a total of 1017 t CO_2-e, resulting in an estimated net abatement of 913 231 t CO_2-e equal to USD20 100 214 worth of carbon credits. The total net potential value of abatement for this scenario was USD17 159 519 (or USD857 976 year^{-1}) (Fig. 6).

For our three buffalo management scenarios, we predicted that the potential value of abatement would outweigh the costs and emissions of conducting feral buffalo management within 4 years (Fig. 6).

Discussion

The bourgeoning carbon economy is creating novel ways to incentivise conservation management activities that demonstrably avoid greenhouse gas (GHG) emissions. We have demonstrated that the potential economic value of carbon credits generated by reducing feral buffalo populations in northern Australia far outweighs the costs of management. Our estimates of GHG abatement represents an important demonstration that the management of feral ruminants could be incentivised by the generation of carbon credits in a similar way to savanna fire management. Such management would not only avoid GHG emissions while generating income for landowners, but it also has the potential to offer important ecological benefits, especially when applied in concert with fire management.

All three of our buffalo management scenarios constrained buffalo population size well below that predicted under no management. In doing so, all three scenarios substantially reduced the estimated buffalo enteric GHG emissions that would otherwise have been produced. Initially, the cost of applying each buffalo management scenario outweighed the potential net value from GHG abatement. However, we estimated that by the fourth year, the economic potential of the GHG abatement achieved under all three management scenarios would outweigh management costs. Even our low-intensity buffalo control scenario was predicted to have large economic returns, abating 790 513 t CO_2-e over 20 years, with the estimated net potential value of abatement being USD15 061 236 over 20 years (or USD753 062 year^{-1}). Although our high-intensity buffalo control scenario had the largest initial investment,

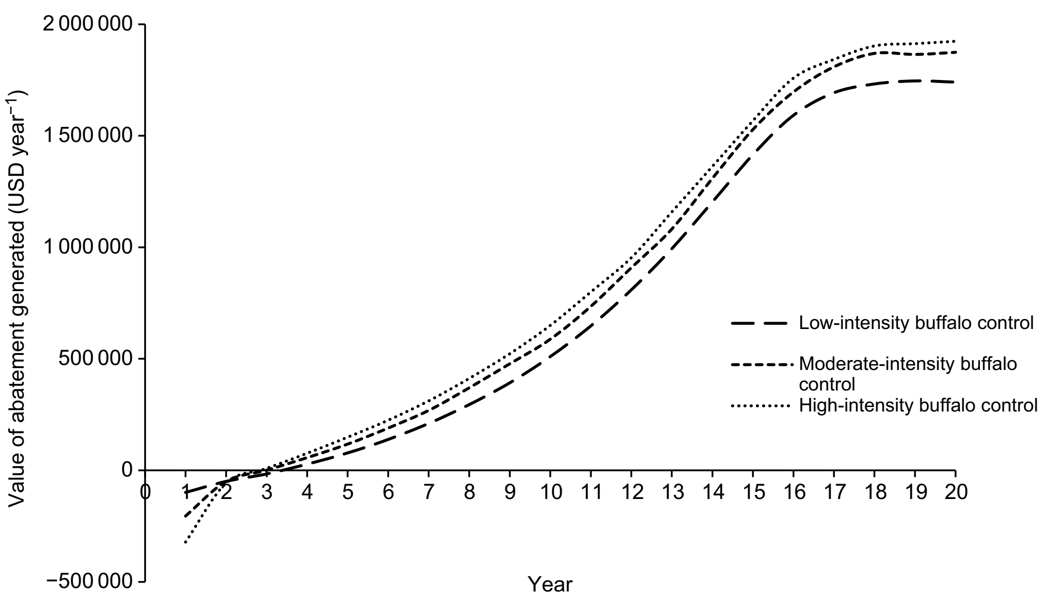

Fig. 6. The estimated annual value of abatement from our three buffalo management scenarios across the South Alligator River region of Kakadu National Park.

as well as higher annual management costs and emissions, it had the highest potential economic return due to the greatest reduction in buffalo enteric GHG emissions, totalling a net abatement of 913 231 t CO_2-e over the 20-year simulation, potentially worth USD17 159 519 (or USD857 976 year^{-1}).

Our estimates of GHG abatement from feral buffalo control provide a noteworthy comparison to the demonstrated GHG abatement currently being achieved under intensive fire management programs across northern Australian savannas. For example, by increasing the proportion of fires occurring under the benign fire conditions of the early dry season (April–July), the West Arnhem Land Fire Abatement (WALFA) project has delivered a mean annual emissions reduction of 116 968 t CO_2-e, while also delivering social, biodiversity and long-term biomass sequestration benefits (Russell-Smith *et al.* 2013). Comparatively, the maximum estimated annual net GHG abatement in our analysis was 92 594 t CO_2-e, recorded in the 20th year of the high-intensity buffalo control scenario. However, the WALFA project area (28 000 km^2) is over five times larger than the South Alligator area (5300 km^2). When this difference in area is accounted for, the GHG abatement recorded from fire management at WALFA is 4.2 t CO_2-e km^2, whereas our estimated potential GHG abatement from feral buffalo control across the South Alligator River region could be as high as 17.5 t CO_2-e km^2. The simultaneous management of fire and feral buffalo could substantially increase the amount of GHG emissions abatement and potential economic gain for landowners.

Our estimates of emissions abatement were quantified against the predicted growth of the buffalo population under no control. Under our 'no buffalo control' scenario, the predicted population size remained comparable to previous observations of buffalo population size across the South Alligator River region (Freeland and Boulton 1990; Skeat *et al.* 1996). Therefore, in this circumstance, we can be confident in the magnitude of our estimated abatement. Regardless, our use of a predicted baseline, rather than an empirically observed baseline, has important implications. It may be risky when applying this method to other areas without robust observations of the maximum size of the buffalo population (i.e. at carrying capacity) in that particular area. Such uncertainty may reduce confidence in the estimated abatement and impact on the ability to sell carbon credits. Another option might be to quantify the emissions abatement from buffalo control against a robust region-specific baseline buffalo density. However, if the buffalo population happened to be well below carrying capacity at the time the baseline estimate was established, this approach would underestimate the true magnitude of emissions abatement. Establishing a defensible, but widely applicable, baseline is one of the difficulties that must be addressed when developing a methodology for generating carbon credits through the management of feral ruminants.

There is evidence that severe disturbance regimes, characterised by frequent high-severity fires, and/or heavy grazing by feral herbivores, have disrupted savanna processes across northern Australia (Skeat *et al.* 1996; Legge *et al.* 2019; Stobo-Wilson *et al.* 2020). Currently, there is now an economic incentive driving improved fire management across more than 380 000 km^2 of northern Australia, facilitated by the approved 'savanna burning' accounting methodology under the Australian Government's Emissions Reduction Fund (Corey *et al.* 2020; Edwards *et al.* 2021). Unfortunately, such an incentive does not yet exist for the active management of large feral herbivores across northern Australia. This is despite research demonstrating the need for the concurrent management of both fire and feral herbivores for the conservation of native wildlife (Legge *et al.* 2019). As such, an economic incentive for feral herbivore control across northern Australia (similar to that currently incentivising fire management) could help maximise the economic and ecological benefits of these programs.

Because fire is influenced by a range of factors including weather, rainfall, fuel loads, and ignition points, the amount of GHG emissions abatement achieved through fire management can vary from year to year (Russell-Smith *et al.* 2013). This potentially poses issues when delivering an emissions abatement to meet contractual obligations (Evans and Russell-Smith 2019). On the other hand, populations of large feral ruminants do not experience large, unpredictable annual fluctuations, resulting in more predictable GHG emissions abatement and reliable income stream for landowners (notwithstanding fluctuations in the price of carbon). However, generating income from GHG abatement via the control of feral herbivores, such as buffalo, requires a range of important considerations. First, there are important animal welfare and ethical considerations that have the potential to influence the economic viability of programs generating carbon credits from the control of feral animals. The culling of feral animals can be highly controversial (Hagis and Gillespie 2021), which could potentially impact on the ability to sell carbon credits generated from the control of feral animals. On the other hand, there can be beneficial animal welfare outcomes from the humane culling of animals suffering from malnutrition and disease (Hampton and Forsyth 2016).

A second critical consideration is the diverse range of stakeholders and values engrained in buffalo exploitation and management across northern Australia, including those of Indigenous people, conservationists, pastoralists and trophy shooters (Albrecht *et al.* 2009). We note that feral herbivore eradication is not a logistically feasible option for most areas of northern Australia (apart from offshore islands and peninsulas), nor is it likely to be supported across such diverse stakeholders (Albrecht *et al.* 2009). As such, programs aiming to avoid GHG emissions through the sustained control of feral animals should be developed with broad consultation across stakeholder groups. Such stakeholder consultation could underpin the spatiotemporal application of control measures, thereby reducing conflict and maximising potential benefits.

The GHG emissions reduction from the control of feral ruminants is not included in accounting methodologies approved as part of the Emissions Reduction Fund, Australia's national carbon offset program. However, because projects aimed at reducing domestic livestock emissions are included, and given the potential economic and ecological benefits, the development of an accounting methodology that incentivises the control of feral ruminants is a conceivable next step (Bradshaw *et al.* 2013). Before this can happen, targeted research aimed at addressing remaining uncertainties is needed. Our study represents an important proof of concept, demonstrating that the economic potential of carbon credits generated from the control of feral buffalo populations far outweighs the cost of management. Importantly, this is true despite our study investigating a relatively small fraction of the total GHG emissions from large feral ruminants. For example, large feral animals substantially increase GHG emissions via impacts on soil, and disturbance of wetlands and floodplains (Limpert *et al.* 2021). Much research is still needed to understand the magnitude of these fluxes to accurately quantify the disturbance to these GHG emission pathways caused by feral ruminants. While we estimated the direct enteric emissions from only one species in northern Australia, the same approach could be applied to other species across Australia, including cattle, camels, goats, and deer.

Our results are based on Tier 1 Intergovernmental Panel on Climate Change (IPCC) estimates of buffalo enteric emissions (Calvo Buendia *et al.* 2019). The uncertainty of these estimates is an important limitation of our study, and future research should aim to refine estimates of GHG emissions from feral ruminant species, including how they vary throughout their lifespan. Our study did not account for the GHG emissions from decomposing carcasses. Although this source of GHG emissions may be negligible compared with those produced by a living animal (Zeng 2015), decomposing carcasses could provide supplementary food for predators (Forsyth *et al.* 2014). This could have important ecological implications that counteract the expected conservation benefits from feral ruminant management.

The impacts of global climate change are becoming increasingly evident, and motivating new, innovative methods to reduce and offset anthropogenic GHG emissions. The development of the carbon economy has created new ways to incentivise conservation management activities that demonstrably avoid greenhouse gas (GHG) emissions. A monetary incentive for the management of feral ruminants could offer substantial reductions in GHG emissions, while providing important economic and ecological outcomes.

References

Albrecht G, Mcmahon CR, Bowman DMJS, Bradshaw CJA (2009) Convergence of culture, ecology, and ethics: management of feral swamp buffalo in northern Australia. *Journal of Agricultural and Environmental Ethics* **22**, 361–378. doi:10.1007/s10806-009-9158-5

Bowman DMJS, Murphy BP, Mcmahon CR (2010) Using carbon isotope analysis of the diet of two introduced Australian megaherbivores to understand Pleistocene megafaunal extinctions. *Journal of Biogeography* **37**, 499–505. doi:10.1111/j.1365-2699.2009.02206.x

Bradshaw CJA, Bowman DMJS, Bond NR, Murphy BP, Moore AD, Fordham DA, Thackway R, Lawes MJ, McCallum H, Gregory SD, Dalal RC, Boer MM, Lynch AJJ, Bradstock RA, Brook BW, Henry BK, Hunt LP, Fisher DO, Hunter D, Johnson CN, Keith DA, Lefroy EC, Penman TD, Meyer WS, Thomson JR, Thornton CM, VanDerWal J, Williams RJ, Keniger L, Specht A (2013) Brave new green world – consequences of a carbon economy for the conservation of Australian biodiversity. *Biological Conservation* **161**, 71–90. doi:10.1016/j.biocon.2013.02.012

Calvo Buendia E, Tanabe K, Kranjc A, Baasansuren J, Fukuda M, Ngarize S, Osako A, Pyrozhenko Y, Shermanau P, Federici S (2019) 2019 refinement to the 2006 IPCC guidelines for national greenhouse gas inventories. (IPCC: Gland, Switzerland) Available at https://www.ipcc.ch/report/2019-refinement-to-the-2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/

Commonwealth of Australia (2015) Carbon credits (carbon farming initiative – emissions abatement through savanna fire management) methodology determination 2015. (Australian Government: Canberra, ACT, Australia)

Commonwealth of Australia (2018) Carbon credits (carbon farming initiative – savanna fire management – emissions avoidance) methodology determination 2018. (Australian Government: Canberra. ACT, Australia)

Cook GD, Meyer CP (Mick), Muepu M, Liedloff AC (2016) Dead organic matter and the dynamics of carbon and greenhouse gas emissions in frequently burnt savannas. *International Journal of Wildland Fire* **25**, 1252–1263. doi:10.1071/WF15218

Corey B, Andersen AN, Legge S, Woinarski JCZ, Radford IJ, Perry JJ (2020) Better biodiversity accounting is needed to prevent bioperversity and maximize co-benefits from savanna burning. *Conservation Letters* **13**, e12685. doi:10.1111/conl.12685

Davies HF, Maier SW, Murphy BP (2020) Feral cats are more abundant under severe disturbance regimes in an Australian tropical savanna. *Wildlife Research* **47**, 624–632. doi:10.1071/WR19198

Department of Environment and Conservation (2011) Fire management services aircraft factsheet. Perth. Available at https://www.dpaw.wa.gov.au/images/conservation-management/fire/aircraft/AircraftFactSheetBell206Jetranger.pdf [Accessed 6 July 2021]

Drucker AG, Edwards GP, Saalfeld WK (2010) Economics of camel control in central Australia. *The Rangeland Journal* **32**, 117–127. doi:10.1071/RJ09046

Edwards A, Archer R, De Bruyn P, Evans J, Lewis B, Vigilante T, Whyte S, Russell-Smith J (2021) Transforming fire management in northern Australia through successful implementation of savanna burning emissions reductions projects. *Journal of Environmental Management* **290**, 112568. doi:10.1016/j.jenvman.2021.112568

Evans J, Russell-Smith J (2019) Delivering effective savanna fire management for defined biodiversity conservation outcomes: an Arnhem Land case study. *International Journal of Wildland Fire* **29**, 386–400. doi:10.1071/WF18126

Forsyth DM, Woodford L, Moloney PD, Hampton JO, Woolnough AP, Tucker M (2014) How does a carnivore guild utilise a substantial but unpredictable anthropogenic food source? Scavenging on hunter-shot ungulate carcasses by wild dogs/dingoes, red foxes and feral cats in south-eastern Australia revealed by camera traps. *PLoS ONE* **9**, e97937. doi:10.1371/journal.pone.0097937

Freeland WJ, Boulton WJ (1990) Feral water buffalo (*Bubalus bubalis*) in the major floodplains of the Top End, Northern Territory, Australia – population growth and the brucellosis and tuberculosis eradication campaign. *Australian Wildlife Research* **17**, 411–420. doi:10.1071/WR9900411

Hagis E, Gillespie J (2021) Kosciuszko National Park, Brumbies, law and ecological justice. *Australian Geographer* **52**, 225–241. doi:10.1080/00049182.2021.1928359

Hampton JO, Forsyth DM (2016) An assessment of animal welfare for the culling of peri-urban kangaroos. *Wildlife Research* **43**, 261–266. doi:10.1071/WR16023

Jesser P, Markula A, Csurhes S (2016) Invasive animal risk assessment: water buffalo. *Bubalus bubalis*. Queensland Department of Agriculture and Fisheries Biosecurity, Brisbane, Qld, Australia. Available at https://www.daf.qld.gov.au/_data/assets/pdf_file/0018/62640/IPA-Water-Buffalo-Risk-Assessment.pdf

Legge S, Smith JG, James A, Tuft KD, Webb T, Woinarski JCZ (2019) Interactions among threats affect conservation management outcomes: livestock grazing removes the benefits of fire management for small mammals in Australian tropical savannas. *Conservation Science and Practice* **1**, e52. doi:10.1111/csp2.52

Limpert KE, Carnell PE, Macreadie PI (2021) Managing agricultural grazing to enhance the carbon sequestration capacity of freshwater wetlands. *Wetlands Ecology and Management* **29**, 231–244. doi:10.1007/s11273-020-09780-7

McMahon CR, Brook BW, Collier N, Bradshaw CJA (2010) Spatially explicit spreadsheet modelling for optimising the efficiency of reducing invasive animal density. *Methods in Ecology and Evolution* **1**, 53–68. doi:10.1111/j.2041-210X.2009.00002.x

McMahon CR, Brook BW, Bowman DMJS, Williamson GJ, Bradshaw CJA (2011) Fertility partially drives the relative success of two introduced bovines (*Bubalus bubalis* and *Bos javanicus*) in the Australian tropics. *Wildlife Research* **38**, 386–395. doi:10.1071/WR10174

Petty AM, Werner PA, Lehmann CER, Riley JE, Banfai DS, Elliott LP (2007) Savanna responses to feral buffalo in Kakadu National Park, Australia. *Ecological Monographs* **77**, 441–463. doi:10.1890/06-1599.1

Ridpath MG, Begg RJ, Dudzinski ML, Forbes MA, Graham A (1983) Counting the same populations of large tropical mammals from the ground and from the air. *Wildlife Research* **10**, 487–498. doi:10.1071/WR9830487

Russell-Smith J, Murphy BP, Meyer CP (Mick), Cook GD, Maier S, Edwards AC, Schatz J, Brocklehurst P (2009) Improving estimates of savanna burning emissions for greenhouse accounting in northern Australia: limitations, challenges, applications. *International Journal of Wildland Fire* **18**, 1–18. doi:10.1071/WF08009

Russell-Smith J, Cook GD, Cooke PM, Edwards AC, Lendrum M, Meyer CP (Mick), Whitehead PJ (2013) Managing fire regimes in north Australian savannas: applying Aboriginal approaches to contemporary global problems. *Frontiers in Ecology and the Environment* **11**, e55–e63. doi:10.1890/120251

Saalfeld K (2014) Feral buffalo (*Bubalus bubalis*): distribution and abundance in Arnhem Land, Northern Territory. NT Department of Land Resource Management, Darwin, NT, Australia.

Skeat AJ, East TJ, Corbett LK (1996) Impact of feral water buffalo. In 'Landscape and Vegetation Ecology of the Kakadu Region, Northern Australia'. (Eds CM Finlayson, I Von Oertzen) pp. 155–177. (Springer: Dordrecht, The Netherlands)

Solomon S, Qin D, Manning M, Marquis M, Averyt K, Tignor MMB, LeRoy Miller H, Chen Z (2007) Climate change 2007 – the physical science basis. Contribution of Working Group I to the fourth assessment report of the Intergovernmental Panel on Climate Change. (Cambridge University Press: Cambridge, UK) Available at https://www.ipcc.ch/report/ar4/wg1/

Stobo-Wilson AM, Stokeld D, Einoder LD, Davies HF, Fisher A, Hill BM, Mahney T, Murphy BP, Scroggie MP, Stevens A, Woinarski JCZ, Bawinanga Rangers, Warddeken Rangers, Gillespie GR (2020) Bottom-up and top-down processes influence contemporary patterns of mammal species richness in Australia's monsoonal tropics. *Biological Conservation* **247**, 108638. doi:10.1016/j.biocon.2020.108638

Tulloch DG, Grassia A (1981) A study of reproduction in water buffalo in the Northern Territory of Australia. *Wildlife Research* **8**, 335–348. doi:10.1071/WR9810335

Werner PA (2005) Impact of feral water buffalo and fire on growth and survival of mature savanna trees: an experimental field study in Kakadu National Park, northern Australia. *Austral Ecology* **30**, 625–647. doi:10.1111/j.1442-9993.2005.01491.x

Williams CK, Dudzinski ML (1982) Ingestion rates, food utilization and turnover of water and sodium in grazing buffaloes, *Bubalus bubalis*, and cattle, *Bos taurus* × *B. indicus*, in monsoonal Northern Territory. *Australian Journal of Agricultural Research* **33**, 743–754. doi:10.1071/AR9820743

Williams CK, Ridpath MG (1982) Rates of herbage ingestion and turnover of water and sodium in feral swamp buffalo, *Bubalus bubalis*, in relation to primary production in a cyperaceous swamp in monsoonal northern Australia. *Wildlife Research* **9**, 397–408. doi:10.1071/WR9820397

Woinarski JCZ, Milne DJ, Wanganeen G (2001) Changes in mammal populations in relatively intact landscapes of Kakadu National Park, Northern Territory, Australia. *Austral Ecology* **26**, 360–370. doi:10.1046/j.1442-9993.2001.01121.x

Zeng B (2015) Camel culling and carbon emissions in rangelands in central Australia. *Journal of Environmental Planning and Management* **58**, 270–282. doi:10.1080/09640568.2013.852077

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