The dynamics of grazed woodlands in southwest Queensland, Australia and their effect on greenhouse gas emissions

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Abstract

This study outlines the development of an approach to evaluate the sources, sinks, and magnitudes of greenhouse gas emissions from a grazed semiarid rangeland dominated by mulga (*Acacia aneura*) and how these emissions may be altered by changes in management. This paper describes the modification of an existing pasture production model (GRASP) to include a gas emission component and a dynamic tree growth and population model. An exploratory study was completed to investigate the likely impact of changes in burning practices and stock management on emissions. This study indicates that there is a fundamental conflict between maintaining agricultural productivity and reducing greenhouse gas emissions on a given unit of land. Greater agricultural productivity is allied with the system being an emissions source while production declines and the system becomes a net emissions sink as mulga density increases. Effective management for sheep production results in the system acting as a net source ($\sim 60–200$ kg CO_2 equivalents/ha/year). The magnitude of the source depends on the management strategies used to maintain the productivity of the system and is largely determined by starting density and average density of the mulga over the simulation period. Prior to European settlement, it is believed that the mulga lands were burnt almost annually. Simulations indicate that such a management approach results in the system acting as a small net sink with an average net absorption of greenhouse gases of 14 kg CO_2 equivalents/ha/year through minimal growth of mulga stands. In contrast, the suppression of fire and the introduction of grazing results in thickening of mulga stands and the system can act as a significant net sink absorbing an average of 1000 kg CO_2 equivalents/ha/year. Although dense mulga will render the land largely useless for grazing, land in this region is relatively inexpensive and could possibly be developed as a cost-effective carbon offset for greenhouse gas emissions elsewhere. These results also provide support for the hypothesis that changes in land management, and particularly, suppression of fire is chiefly responsible for the observed increases in mulga density over the past century. © 2001 Elsevier Science Ltd. All rights reserved.

Keywords: Climate change; Greenhouse; Emissions; Woodlands; Grasslands; Carbon; Grazing; Fire

1. Introduction

Increasing atmospheric concentrations of radiatively active (greenhouse) gases are expected to result in significant changes to the global climate (Houghton et al., 1996). Agricultural systems are estimated to produce 20.2% of all greenhouse gas emissions in Australia (AGO, 2000). Hence, there is a need to develop methods to evaluate greenhouse gas emissions from all industry sectors including agriculture, and to develop strategies to reduce these emissions. The major greenhouse gases emitted from agriculture activ-

ities are carbon dioxide (CO_2), methane (CH_4), carbon monoxide (CO), and nitrous oxide (N_2O). This study develops a framework for evaluating emissions of these greenhouse gas emissions from sheep-grazed semiarid rangelands and focusses on the mulga (*Acacia aneura* F. Muell) lands of southwest Queensland. Mulga-dominated rangelands occur on red earth soils in a broad strip across the central part of the Australian continent, between latitudes 21°S and 33°S. They occupy 26% (1.7 million km²) of rangelands and occur in both arid and semiarid environments. Rainfall is low (150–500 mm/year) and erratic, becoming increasingly summer dominant toward the northern regions. Herbaceous vegetation dynamics are strongly coupled to rainfall and are consequently highly variable from year to year.

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Charleville (the site for this study) has an average annual rainfall of 489 mm with approximately 70% of this falling in the summer months (November–April). Originally open woodland, the land has changed significantly since grazing (Hodgkinson, 1991, Beale et al., 1986). Unchecked growth of mulga has resulted in dense stands of mulga growth, which can result in pasture yields as low as 100 kg/ha (Silcock, 1986) in affected areas resulting in mulga being viewed as a woody weed. The increase in mulga density in this region is usually attributed to the suppression of fire since European settlement (Harrington et al., 1984; Moore, 1973).

Rangeland grazing systems, including the semiarid mulga lands considered here, can act as a net source or sink of greenhouse gases depending on how they are used and managed (Howden et al., 1994). Ruminants, other herbivores, detritivores, and fire all result in greenhouse gas emission. Biomass increase and soil absorption of methane act as sinks for greenhouse gases. Whether the system acts as a net source or a sink is dependent on the interplay of these components, which are affected by management and climate.

In this study, we have extended an existing model (GRASP; McKeon et al., 1982, 1990; Rickert and McKeon, 1982) to include root pools, termites, livestock production, greenhouse gas emissions from the system, including those from burning and grazing, and a dynamic tree growth and population model. We use this extended model to assess how management, particularly stocking strategies and burning regimes, affects greenhouse gas emissions.

2. Modelling greenhouse emissions

GRASP is a daily timestep model that has been widely tested in the mulga rangelands. GRASP simulates the response of the system to climate variability, pasture growth, and management. Pasture growth is limited by water availability, radiation, temperature, and nitrogen content of the soil. Water availability is determined using a four-layer soil water model (Rickert and McKeon, 1982). Infiltration and runoff are functions of vegetative cover, rainfall intensity, and soil water levels. The processes of soil evaporation, grass, and tree transpiration are calculated separately with tree transpiration being calculated before grass transpiration each day (Scanlan and McKeon, 1993).

2.1. Dynamic tree growth and population model

Grazing, burning, and clearing affect trees differently depending on the size of the trees. Therefore, the population is divided into cohorts of different sized plants with identical individuals within each cohort. A new cohort is formed by each annual recruitment event. The dynamics of each cohort are calculated separately, and totals summed at the end of each day. For each cohort, i, a number of attributes are calculated including, the number of individuals in the cohort (D_i, plants/ha), their height (H_i, m), individual tree basal area (TBA_i, m²/ha), and the above (leaf and wood) and below ground biomass.

Although germination and establishment of mulga are sensitive to rainfall in other regions, Burrows (1973) concluded from work in the Charleville area that regeneration is less sensitive to rainfall in this region, and that it was continuous and sufficient to maintain the mulga population. He attributed lack of seedling survival to density effects, grazing pressure, and burning. Thus recruitment is calculated as a function of adult tree density (AD, $H_i > 5$ m) and is independent of rainfall. The impact of sheep and fire are incorporated separately and are discussed below. Recruitment was represented following Burrows (1973) as:

$$\text{Recruit. (plants/ha)} = 126 + 7.1482 \times AD, \text{ if } AD < 34.5$$

otherwise,

$$\text{Recruit. (plants/ha)} = 2217 \times AD^{-0.504}.$$

The height of these seedlings is set initially to 0.1 m and this height is used to calculate the biomass and other attributes of the cohort using the size–mass relationships developed below.

The tree growth model is run on a daily timestep and is linked with GRASP through tree basal area and transpiration. Tree transpiration (mm/day) is calculated by GRASP, and increases in proportion to the total tree basal area (TBA, m²/ha). Perennial grasses tend to outcompete mulga seedlings ($H_i < 0.3$ m) for water (Harrington et al., 1984; Mills, 1989). To incorporate this effect, seedling transpiration is calculated similarly to tree transpiration except that grasses have prior access to water over seedlings and seedlings can only access water from the top two soil layers (less than 50 cm deep). Total tree transpiration is partitioned between the cohorts in proportion to leaf mass. Growth is calculated separately for each cohort and is the product of transpiration (mm/ha) and transpiration use efficiency (TUE, in g/m²/mm), which is calculated as;

$$TUE = 0.0578 \times TBA + 0.756$$

The growth increment is then added to the total standing dry matter of the cohort ($TSDM_i$, kg/ha). This dry matter increment is partitioned between leaf and wood for each cohort using the following relationship derived from Pressland (1974):

$$(\text{leaf/wood})_i = \min(1.2, \ 0.824 \times H_i^{-0.7305}).$$

Seedlings <60 cm tall have a leaf/wood ratio of 1.2. From this, leaf and wood biomass are calculated. The average circumference (C_i, cm/plant) of the trees in each cohort is calculated from the average wood mass (W_i, kg/plant) of individual trees:

$$C_i = W_i \times \exp(5.2193)^{0.418}.$$

Individual tree basal area (TBA_i, cm²/plant) is calculated from the circumference assuming a circular cross-section,

which is used to calculate the average height (H_i, m) of plants in the cohort (Pressland, 1974). Then:

$$H_i \ (\mathrm{m}) = 1.845 \times \mathrm{TBA}_i - 0.4613, \qquad \text{if } \mathrm{TBA}_i$$

$$< 0.792 \ \mathrm{cm}^2$$

otherwise,

$$H_i \ (\mathrm{m}) = 1.0904 \times \mathrm{TBA}_i^{0.3615}$$

Total daily growth of the entire community is limited to the maximum daily growth measured by Pressland (1974) during the high rainfall years of 1973–1974 such that

$$\text{maximum growth (kg/ha)} = 0.306 \times \mathrm{TBA} + 1.375$$

If potential growth exceeds this then the water is transpired, but actual growth is set to the maximum limit. Also, trees are assumed to reach the commonly observed maximum height of about 12 m. There is also an imposed limit to the total tree basal area of the community, which is set at 13 m^2. This has been included because the data used to develop growth rates were for recently thinned stands and we were unable to develop a relationship between individual growth and a community limit.

Quantified information about the death rates of mulga seedlings and adults is largely unavailable. Mulga is very drought-resistant (Burrows, 1973; Winkworth, 1973) suggesting that drought-dependent death is uncommon and occurs only after severe and prolonged drought, but there was not enough information available to model the effect. In the model, annual mortalities are set at 50% for seedlings ($H_i < 0.3$ m), 22% for saplings (0.3 m $< H_i < 2.0$ m), and 1% for larger trees ($H_i > 2.0$ m; Cunningham and Walker, 1973). Sensitivity analysis of these settings showed changes in seedling and sapling death rates resulted in little change to the overall dynamics. However, the system is considerably more sensitive to adult death rates, which is consistent with the strong density-dependence of recruitment and TUE. This highlights the need for accurate information about shrub population death rates.

Sheep grazing will only reduce the survivorship of young plants under unsustainable stocking rates (>0.8 sheep/ha) (e.g., Cunningham and Walker, 1973), and so this effect was not incorporated explicitly into the model. However, grazing by sheep decreases the growth of mulga seedlings by browsing the growing tips. In the model, this results in greater mortality, as death rates are size-dependent. The proportion of growth eaten by sheep depends on the stocking rate (SR, sheep/ha) and, based on Brown (1985), is calculated as the reduction in height growth compared with potential growth where

$$\text{growth reduction} = 1.0 - (0.1/(0.1 + \mathrm{SR})).$$

Fire is potentially the most important management tool in the rangelands. It is commonly believed that reduction in fire frequency over the past 50 years has contributed to the increase in shrub density throughout the rangelands (Hodgkinson and Harrington, 1985). In recent years, burning of pasture has been strongly recommended as a tool to reduce woody shrub density and maintain pasture productivity and desirable pasture species composition (Hodgkinson and Harrington, 1985; Mills, 1989; O'Shea, 1993). In the model, all mulga seedlings ($H_i < 0.3$ m) are killed in a fire, along with 80% of plants less than 2 m and 20% of those less than 5 m (Hodgkinson and Harrington, 1985; O'Shea, 1993). Also killed by fire are 7.5% of those plants taller than 5.0 m (Wilson and Mulham, 1979).

2.2. Greenhouse emission calculation

The major pathways of greenhouse gas emissions in a subtropical rangeland are outlined in Howden et al. (1994). The emissions section of the model dealing with tree, grass and litter carbon flows, methane emissions from the livestock, and emissions from burning and termites are based on the mass balance approach of Howden et al., with modifications to the decay rate of mulga leaves from Burrows (1976). Carbon is emitted from the system as carbon dioxide due to decay, respiration and burning, methane due to sheep digestion, termite consumption, dung fermentation, and burning, and as carbon monoxide due to burning. Carbon inputs to the system in the form of CO_2 occur due to growth of trees and grass and as methane due to absorption by the soil. Carbon stores change with the size of tree leaf, wood, and root biomass, grass root and shoot biomass, grass and tree litter pools, standing and fallen wood pools, and dead root pools. Soil carbon is not currently modelled so it is assumed to remain constant between scenarios. Additionally, nitrous oxide is emitted from termites, sheep, and fire. Net emissions of the gases are calculated using a mass balance approach applied to carbon, as net emissions from the system in any period form a balance with carbon inputs due to growth and carbon stores. All emissions are converted to CO_2 equivalents following
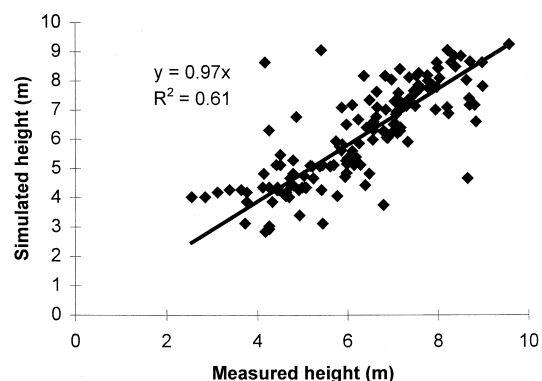

Fig. 1. Validation graph of simulated against measured mulga heights for the three plots monitored by Beale (1973). Data are from both the Boatman and Monamby sites.
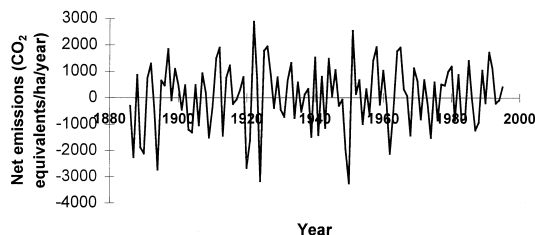
Fig. 2. Net annual greenhouse gas emissions (kg CO_2 equivalents) when moderately grazed (20% utilisation) under the "burn sometimes" regime (12 yearly if conditions suitable).

Houghton et al. (1996) to allow for radiative and lifecycle differences. The values used in this study are for an integration horizon of 100 years.

2.3. Validation

The difficulties in validating complex models such as this are detailed in Howden et al. (1994). The growth component of the mulga model was validated using data collected by Beale (1973), over a 7-year period (Fig. 1). The model gives a satisfactory fit to the data, but tends to underestimate individual growth rates. The short timescale over which data were available means that only the growth component of the model has been validated, and hence we are unable to draw any conclusions about the long-term dynamics as affected by death and recruitment rates. There is a need for a long-term data set for model validation.

3. Exploratory study

An exploratory study was undertaken using the modified GRASP model. The study focussed on a single paddock on a property typical of the region near Charleville. The aim of the study was to explore how pasture management impacts on greenhouse gas emissions and to identify possibilities for reducing these emissions.

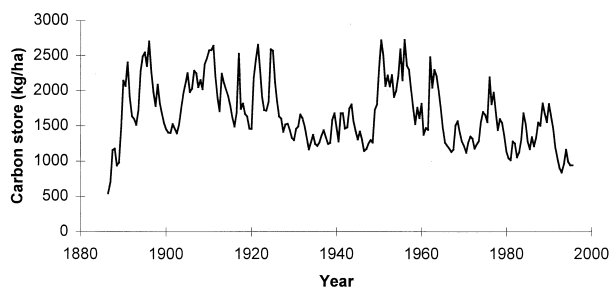As described previously, two of the major management issues in these rangelands are: (1) maintaining low shrub

Fig. 3. Biomass and litter carbon stores over the simulation period when the community was moderately grazed (20% utilisation) under the "burn sometimes" regime (12 yearly if conditions suitable).
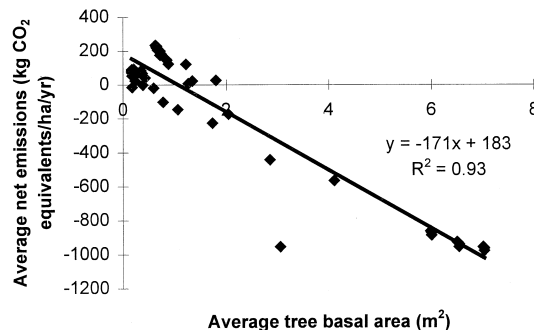

Fig. 4. Correlation between net emissions (kg CO_2 equivalents) and tree basal area (m^2/ha) both averaged over 110 years. The treatments cover the full spectrum of stocking rates, mulga densities, clearing regimes, and burning scenarios simulated.

densities largely via burning and (2) setting appropriate stocking rates for sheep. In this study, burning treatments were: never burnt, burnt sometimes (every 12 years if fuel loads sufficient), burnt frequently (every 6 years if fuel loads sufficient), and always burnt each year if the fuel load is sufficient. Stocking rate treatments ranged from 0% to 50% utilisation in 10% increments. The utilisation approach is recommended to account for the high variability in pasture growth (Mills, 1989) and is implemented by evaluating the feed available at the end of the growing season (1 June) and setting stock densities so that a target percentage (e.g., 20%) of the feed is consumed in the following year. Pasture utilisation of 50% is very high and is unlikely to be sustainable in the long term (Orr et al., 1986). An initial mulga population structure was derived from Beale (1973). Each simulation was run for 110 years (1885–1995) using historical climate data recorded at Charleville.

4. Results

Annual variation in net emissions is large for all the treatments considered (e.g., Fig. 2) reflecting the highly variable nature of the ecosystem. Biomass and litter carbon
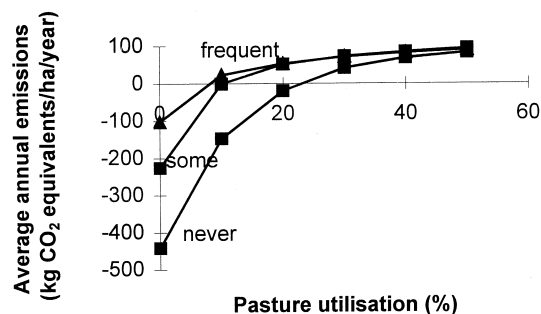

Fig. 5. The impact of pasture utilisation (%) on annual net emissions (kg CO_2 equivalents/ha/year) averaged over a 110-year simulation from grazing systems under three different burning scenarios (as described in the text).
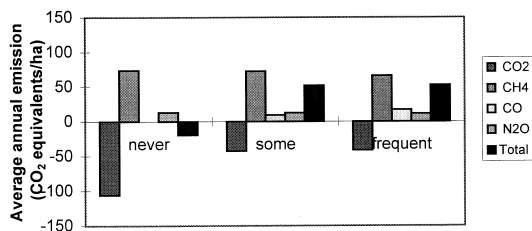
Fig. 6. The effect of different burning scenarios on the contribution that each gas makes to total emissions with 20% pasture utilisation: the burning scenarios are as described in the text.

stores also show considerable variability on both a yearly and decadal basis (Fig. 3).

If the land is managed for productive grazing (that is, if tree basal area is limited by management to less than 1.0 m^2), then the system acts as a net source of greenhouse gases. This source is ~ 60 kg CO_2 equivalents/ha/year with conservative grazing (20% utilisation and occasional burning; Fig. 6) up to ~ 200 kg CO_2 equivalents/ha/year with heavy grazing (40% utilisation — data not shown). The size of this source largely depends on the initial density of shrubs and the methods employed to control them. Burning only increases emissions if there is a significant amount of biomass available to burn. Sheep contribute directly a varying proportion of the net emission fluxes from methane from the digestive tract and from dung and nitrous oxide from excreta. Under moderate grazing (20% utilisation), these emissions average 68–75 kg CO_2 equivalents/ha/year, and under heavy grazing (40% utilisation), 108–119 kg CO_2 equivalents/ha/year. These emissions are only a small component of the fluxes under conditions that promote mulga growth (cf. Figs. 4 and 5) while being the dominant component with high stocking rates and when mulga growth is suppressed. In addition, the inhibitory effect of sheep on mulga growth significantly decreases carbon fixation and thus increases net emissions.

In contrast, if the system is not maintained for productive grazing (i.e., mulga increases in density significantly), then it can act as a significant greenhouse gas sink, absorbing up to an average of 1000 kg CO_2 equivalents/ha/year (Fig. 4). The magnitude of the emissions depends strongly on the density of mulga (data not shown) and to a lesser extent on grazing intensity and burning frequency (Figs. 5 and 6).

Simulation of the pre-European scenario (annual burning, no grazing) with an initial low mulga density effectively resulting in a balance between CO_2 absorption due to mulga growth and emissions as a result of regular burning. The system is a small net sink (~ 14 kg CO_2 equivalents/ha/year) in these simulations due to small increases in mulga biomass; however, this is maintained at low levels (Fig. 7). In contrast, if the model is initialised with a more dense mulga stand but with the same man-

agement, the mulga density will increase to a higher biomass (data not presented).

Wool production increases as mulga tree basal area decreases due to an increase in forage for the sheep and minor changes in greasy wool clip per ha were simulated under the different burning regimes (data not shown).

5. Discussion

The simulations indicate that this system can be managed to act as either a net source or a net sink of greenhouse gases. The direction in which the system acts is strongly influenced by land management, and particularly by how management influences the mulga density. Prior to European settlement, these lands were burnt frequently by the Aborigines. We simulated this management approach, where the vegetation is burnt whenever sufficient fuel is available. This land management approach with low initial mulga densities results in the system acting as a small emission sink over the period of the simulation and maintains the mulga at low densities. At higher initial densities of mulga, feedbacks on grass biomass (and hence fire frequency) tended to maintain the woodland structure. If implemented in a stochastic spatial framework, we can hypothesise that the resultant patches of grassland and open woodland would be consistent with both descriptions of the region prior to settlement and the changes observed since (Noble, 1997).

In contrast, management of the land for wool production results in the system acting as a source of greenhouse gases. The emissions comprise both an additional component of CH_4 and N_2O from the livestock as well as a reduction in mulga biomass caused by grazing of seedlings. The size of the net source depends on the specific land management techniques and initial density of mulga. For example, a pasture utilisation of 20% and an infrequent burning regime emits an average of 62 kg CO_2 equivalents/ha annually with this increasing with utilisation and decreasing if fire is suppressed.

Alternatively, suppressing both fire and grazing results in the growth of mulga into dense stands. During the growth phase of this vegetation, the system can absorb up to an average of 1000 kg CO_2 equivalents/ha/year or about 27 t C/ha at potential maximum biomass excluding soil carbon
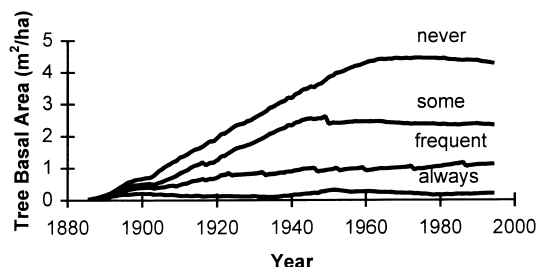

Fig. 7. Impact of different burning scenarios on the tree basal area (m^2/ha) of mulga when ungrazed. The burning scenarios are as described in the text.

components. This raises the possibility of using some of this land, particularly that with potential to develop dense mulga cover, as a carbon offset for greenhouse gases emitted elsewhere (e.g., coal-fired power stations). This option could be attractive as land in the region is relatively inexpensive (often $12–20/ha), and so could be used as a cost-effective carbon offset, although monitoring and verification costs need to be accounted for as they may significant. It is expected that the sequestration rate will decline after several decades as the mulga reaches maximum size and density. While much of the carbon will be stored in the mulga biomass, a significant additional amount (not modelled here) may also be transferred to the soil where carbon turnover times can be thousands of years (Ojima et al., 1993). However, establishment of carbon sink areas will of course reduce regional agricultural production. Furthermore, such areas will need to be fenced off as dense mulga causes significant mustering and husbandry problems as well as providing a refuge for pest animals such as feral pigs (Harrington et al., 1984). Alternatively, a mosaic of grasslands and dense mulga may have positive implications for biodiversity that has been negatively affected by grazing (Landsberg et al., 1997; Freudenberger, 2000).

The model adequately describes the broad changes in woody weed composition that have occurred under different management regimes since European settlement (e.g., Noble, 1997). However, we cannot definitively attribute recorded changes in shrub density to the processes that are described in this model as a number of processes thought to contribute to woody weed thickening are not adequately simulated. These include CO_2 fertilisation (e.g., Idso, 1992; Johnson et al., 1993), climate variability impacts on seeding and death of adult shrubs (e.g., Rolls, 1981; Mitchell, 1991) and changes in the vegetation community due to local extinction of native grazers and browsers (Noble, 1997). However, our simulations indicate that changes in land management, particularly changes in the use of fire, are sufficient to explain both the landscapes before pastoralist settlement and changes since this time. This agrees with the assessments of Archer et al. (1995), Grover and Musick 1990, and others for similar systems elsewhere. However, more definitive analysis requires improved sensitivity of the model to the impact of climatic events on germination, establishment and cohort death rates, and the ability to simulate enhanced germination after fire (Harrington et al., 1984). Further improvements include the need to account for changes in grass species composition, long-term degradation of the soil resource, and the incorporation of a soil carbon–nitrogen model.

Acknowledgments

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