OVERVIEW ARTICLE

Degraded or just dusty? Examining ecological change in arid lands

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Abstract

The ecological history of rangelands is often presented as a tale of devastation, where fragile drylands are irreversibly degraded through inappropriate land-use. However there is confusion about how to recognise and measure degradation, especially in low productivity environments characterised by extreme natural variability and where abrupt and comprehensive management upheavals preclude benchmarks. These issues have important consequences for rangeland management programs, which are typically founded on presumptions of substantial and ongoing degradation from former 'natural' states. We explore complementary approaches to critically assess degradation: the historical record; long-term grazing exclosures; surveys for potentially rare and sensitive plant species; and assessment of water-remote areas in relation to rare plant occurrence. Employing these approaches in inland

Australia, we show that prevailing paradigms have become entrenched despite being inconsistent with empirical evidence. Our methodology can be applied to drylands with abrupt changes in management and contentious ecological narratives.

Keywords: desertification, grazing, rangelands, threatened species, vegetation change

Introduction

The ecological history of arid-lands is often presented as a tale of destruction and devastation. 'Degradation narratives' involving undesirable environmental change and productivity declines date from the earliest dryland civilisations. Archaeological excavations and ancient records show that salinisation and silting began to affect Lower Mesopotamian irrigation schemes from about 2400 B.C., ultimately playing an important part in the demise of Sumerian civilisation (Jacobsen and Adams 1958). Abandonment of Khorezm oasis settlements in Uzbekistan in the first century A.D. (Thomas and Middleton 1994) and Roman settlements in north African and Arabian deserts around 500 A.D. (Barker 2002, Pyatt et al. 1999) have also been attributed to environmental disintegration. The perception that rangelands surrounding the Mediterranean were degraded was expressed by Plato in 400 B.C., who described the hills around Athens as being '…like the skeleton of an old man, all the fat and soft earth wasted away and only the bare framework of the land being left' (in Sinclair and Sinclair 2010:117). Notions of arid regions as landscapes ruined and deforested by human mismanagement became entrenched in the 19th century, and were used to justify expanding western imperialism and land 'improvement' policies (Davis 2016).

However, it was the 1930s western American 'Dust Bowl' that swept arid-lands degradation to the forefront of popular imagination and reverberated through a generation of scientific thinking (Schubert et al. 2004, Worster 1979). The 1930s also saw early scientific investigations of land degradation in Africa, with Stebbing (1935), a European forester who visited Africa, describing the encroachment of the Sahara southwards into the Sahelian savannah as ‘one of the most silent menaces of the world’. Four decades later, events in the Sahel thrust dryland degradation to the forefront of the global environmental agenda (Davis 2016). The ‘Great Sahelian Drought’ of 1968-73 followed a long period of colonial rule, which had disrupted traditional food production systems at a time of increasing population. Images of mass starvation were beamed into Western living rooms and stimulated international concern (Thomas and Middleton 1994). In the drought aftermath, an assessment of environmental conditions in northern Sudan reported that the Sahara had encroached 100 km south into semi-desert scrub in the two decades since its boundary was originally mapped (Lamprey 1988, cited in Dodd 1994).

As with events in North America, drought was viewed as the catalyst which had exposed harmful effects of overgrazing, inappropriate cultivation and deforestation. The Sahelian disaster was seen as a vivid expression of the global problem of desert expansion, and led to the convening of a United Nations Conference on Desertification in Nairobi in 1977. Through the 1970s and 1980s, the United Nations Environment Program (UNEP) made a series of dramatic public announcements regarding the extent of the problem (Davis 2016). In terms of capturing the world’s imagination, desertification was the first ‘big’ environmental issue, preceding the ozone hole, acid rain and global warming in its adoption by the popular and pseudo-scientific press and policy makers, and its appeal to growing environmental concerns (Thomas and Middleton 1994). Today, the emblematic image conjured by the term remains one of rampaging deserts smothering villages and destroying farmland and pasture, resulting in declines of productivity and man-made famine among subsistence farmers (Imeson 2012).

Other evocative symbols of degradation have also become axiomatic: barren earth strewn with bleached animal carcasses, erosion scars, eerily twisted trees, crumbling relics and the lonely tenants of a broken land (Fig. 1).

Dryland degradation: themes and examples

A central tenet of degradation narratives is that semi-arid and arid drylands (here defined as those areas receiving < 500 mm average annual rainfall) are inherently fragile and vulnerable to exploitation, particularly during dry times and with the imposition of management regimes that diverge from their evolutionary history. In the Americas and Australia, this involves the imposition of European 'ranch style' pastoralism since the 1830s (Aagesen 2000, Heathcote 1983). In African, Arabian and Central Asian deserts, the upheavals associated with increasing populations, breakdown of traditional semi-nomadic or nomadic pastoral systems, encroachment of sedentary agriculture on traditional grazing lands, and in some areas restrictions on movements due to conflicts since the 1950s, are blamed for widespread degradation (Tewari and Arya 2004, Verstraete et al. 2009).

Maps of desertification and degradation risk essentially show all of the world's arid and semi-arid regions shaded with varying degrees of severity (Mabbutt 1984, UNCCD 2014). Peer-reviewed articles routinely report that 70% of all drylands are affected by desertification, a figure based on a heavily-questioned 1992 UNEP report (Verón et al. 2006). Some authors have suggested that degradation is an inevitable consequence of land-use in some arid environments (Beaumont 1993, Caughley 1986), a view encapsulated in the ancient proverb: *'Man strides over the earth, and deserts follow in his footsteps'* (in Worster 1979). Some examples of arid lands degradation are well-documented and unequivocal, particularly the consequences of inappropriate dryland cultivation, including on the Great Plains of North America (Schubert et al. 2004), eastern Europe (Goudie and Middleton 2006) and central

Asia and China (Youlin et al. 2001). Desert water resources have borne the brunt of arid zone land-use in many regions, with the desiccation of Lake Chad (Gao et al. 2011) and the Aral Sea (Breckle et al. 2012) being well-known examples. The depletion of desert aquifers and loss of artesian springs are well documented (Idris 1996, Patten et al. 2008, Powell et al. 2015).

Shifts in plant species composition and abundance can indicate degradation, with palatable and perennial species typically replaced by unpalatable and annual species in grazed areas (Cingolani et al. 2005, Seymour et al. 2010). Palatable perennials can be particularly impacted by grazing during dry periods (Hacker et al. 2006, Watson et al. 1997), sometimes becoming locally extinct in more heavily grazed areas due to lack of recruitment (Hunt 2001). Some rangelands have become degraded through invasion by exotic plants, especially where they change the structure and function of an ecosystem and have adverse effects on native species and/or agricultural productivity (Brooks et al. 2010, Muturi et al. 2013). Unpalatable native species or 'woody weeds' may increase at the expense of palatable species in grazed landscapes (Roques et al. 2001, Van Auken 2000).

The combined effects of habitat modification, introduced predators, prey depletion and direct exploitation and extermination of indigenous herbivores have led to extinctions and declines of many arid-land fauna species globally over the past two centuries. Large mammals have been especially susceptible to range contractions and extinctions (Cardillo et al. 2005). The drastic demise of mammals since pastoral settlement in Australia is well-documented (Woinarski et al. 2015) while large birds have also fared badly in many drylands (Goriup 1997, Thiollay 2006). Loss of fauna may in turn affect ecosystem function, through changes in landscape processes, nutrient cycling and plant species composition (Chillo and Ojeda 2012, Waldram et al. 2008).

Questioning degradation narratives

These examples show that degradation has occurred in some drylands and is sometimes irreversible. However, perceptions that it is a universal and perhaps inevitable consequence of land-use in arid zones are simplistic and problematic. Where do they leave arid-lands that have high background rates of wind and/or water erosion and inherently low or variable groundcover (Pickup 1989, Wiegand and Jeltsch 2000)? Particularly during drought, landscapes may display the hallmarks of degradation, but can recover rapidly after rain. Other landscapes appear 'degraded' most of the time, due to soil characteristics such as high salinity and/or sodicity and low and erratic rainfall (Qadir and Schubert 2002). Annual plant cover in good times and bare ground during drought represent the 'healthy' state of many communities, rendering above-ground biomass or life-cycle traits poor indicators of degradation. Even dust storms, the archetypal symbol of degradation, are natural occurrences in most arid-lands and there is little evidence that dust production is associated with widespread land degradation (Brooks and Legrand 2000; McTainsh et al. 2005).

Over the past three decades, numerous authors have questioned basic tenets underlying simplistic notions of degradation, particularly the 'myth of the marching desert' (Behnke and Mortimore 2016, Dodd 1994, Peters et al. 2015, Thomas 1997, Verón et al. 2006). It is now widely accepted that the conclusions of Stebbings and Lamprey which became so influential were based largely on limited direct observation and uncorroborated information from local authorities. Contemporary research has demonstrated that the Sahara 'expands' and 'contracts' in concert with rainfall fluxes (Herrmann et al. 2005). Davis (2004, 2016) details how the narrative of decline and decay was constructed during the French colonial period in Algeria, Tunisia and Morocco. Founded on historical inaccuracies and environmental misunderstandings, it blamed ‘hordes of Arab nomads and their rapacious herds’ for deforestation and desertification of what was erroneously believed to have been a fertile forested landscape, and helped to justify Colonial policies aimed at restoring the region to its ‘past glory’. In northern African and Arabian deserts, the long-term dynamics of long-lived *Acacia* species seem more complex than the oft-cited decline due to overharvesting and grazing (Lahav-Ginott et al. 2001, Rohner and Ward 1999). Recent studies have shown how perceptions of severe livestock-induced degradation is overstated in Mongolia (Addison et al. 2012, Jamsranjav et al. 2018).

Research has highlighted the resilience of many rangelands to disturbance, including in the Middle East (Batanouny 2001, Blumler 1998), Mediterranean (Dell et al. 1986, Perevolotsky and Seligman 1998), north America (Bestelmeyer et al. 2013) and Africa (Davis 2016, Oba et al. 2000), as well as the role of climate variability and other natural factors in desertification and vegetation change (Herrmann and Hutchinson 2005, Lehnert et al. 2016, Romme et al. 2009, UNEP 2011). The view that climate is the primary driver of vegetation dynamics in highly variable drylands (specifically when the coefficient of variation of inter-annual rainfall exceeds 33%; von Wehrden et al. 2012), with grazing playing a secondary role or even having little effect, has gained traction over the past two decades. Proponents of this non-equilibrium theory argue that the risk of degradation through overgrazing in such systems is limited, because the ephemeral forage is only abundant for brief sporadic periods amidst frequent protracted drought, keeping livestock numbers well below the level where they can affect the vegetation community (Illius and O’Connor 1999, Sullivan and Rohde 2002, von Wehrden et al. 2012). Moreover, beneficial effects of grazing have been documented in some instances, including the promotion of tillering in grasses (Crawley 1987), improved germination of some perennial species (Reid and Ellis 1995, Rohner and Ward 1999), and control of shrub encroachment and invasive weeds (Perevolotsky and Seligman 1998, Popay and Field 1996). Rather than being universally viewed as a negative imposition, current theory predicts that the effects of long-term grazing on plant species diversity will be variable across ecosystems, depending upon evolutionary history, ecosystem productivity and herbivore type (Bakker et al. 2006, Cingolani et al. 2005, Koerner et al. 2018, Milchunas and Lauenroth 1993).

Identifying and measuring degradation

Confusion remains about how to recognise and measure degradation, despite considerable effort devoted to defining desertification and identifying indicators (Bestelmeyer et al. 2015). In particular, how can we distinguish characteristics inherent to arid-lands from symptoms of anthropogenic degradation, especially in the face of extreme natural variability? These questions are more than semantics. Arid and semi-arid lands cover 40% of the global land surface and support over a third of the world's population, including many in the most economically vulnerable and politically unstable regions of the world (Reynolds et al. 2007). They are also a significant repository of global biodiversity (Durant et al. 2012).

Most rangeland development and management programs are founded on a perceived crisis of degradation and desertification; a perception that these systems have declined from a more pristine historical state (Fairhead and Leach 1995, Witt et al. 2000). Decades of work to arrest this decline have had little success in many parts of Africa, the Middle East and Asia (Chatty 2001, Goldschmidt 1981), and in some places have had unintended negative consequences (Davis 2016). Is it possible that at least some of this failure is due to misunderstandings or dubious interpretations of the ‘natural’ state and functioning of these systems, in the absence of a reference state (Romme et al. 2009, Sprugel 1991)? Are these programs aiming for a desired state that never existed; an unattainable Eden? Or, alternatively, are some systems so degraded that recovery is not possible, or occurs over such long timescales that results are not yet discernible? The implications of degradation narratives and their continued acceptance in public policy are profound, so there is an obvious need for robust methods of assessing degradation in arid lands.

Examining ecological change and degradation in inland eastern Australia

Semi-arid and arid eastern Australia is part of one of the largest desert systems in the world and one subject to recurring arguments about the nature and extent of ecological change. We refined four approaches to examining ecological change in this region: (i) the historical record, in the form of early explorer journals dating from the 1840s, straddling the critical watershed of pastoral settlement (Silcock et al. 2013); (ii) a network of long-term grazing exclosures across four major land types (Fensham et al. 2011b, Fensham et al. 2014, Silcock and Fensham 2013); (iii) assessment of water-remote areas and gradients in relation to rare plant occurrence (Fensham et al. 2010, Fensham et al. in press, Silcock and Fensham 2014), and (iv) systematic surveys for potentially rare and sensitive elements of the flora (Silcock et al. 2014).

Inland eastern Australia is defined as that portion of Queensland, northern New South Wales, north-eastern South Australia and the eastern Northern Territory receiving <500 mm of average rainfall per annum, encompassing four Biogeographic Regions: (Thackway and Cresswell 1995; Fig. 2). The Mulga Lands are dominated by extensive tracts of mulga (*Acacia aneura*) shrubland; the Mitchell Grass Downs are characterised by open clay soil plains dominated by long-lived Mitchell grass (*Astrebla* species) tussocks; the channels and floodplains of the Channel Country are interspersed with a matrix of stony plains, open downs and chenopod shrublands, while the Simpson-Strzelecki Dunefields are comprised of aeolian dunefields with extensive ephemeral lake systems in the west and south. Dunes are dominated by spinifex (*Triodia* species) hummock grassland in the north, and spinifex species also dominate smaller areas in the eastern Mulga Lands and the northern Channel Country. All bioregions are intersected by low sandstone ranges and major river systems which exist as a network of permanent and semi-permanent waterholes between sporadic flood events. Springs emanating from the Great Artesian Basin (GAB) and local aquifers occur across the study area, respectively clustered around the edges of the Basin and in sandstone ranges.

Average annual rainfall decreases on a south-westerly gradient from 500 mm along the eastern and northern boundary of the study area to 120 mm in the Simpson Desert, but rainfall is highly variable both within and between years (Table 1). Summer temperatures are hot with maximums through December-February averaging 35-38°C and regularly exceeding 40°C, while short winters are characterised by cold nights (5-10°C), often falling below zero and warm days averaging 20-27°C.

Aboriginal people have lived in generally low densities across inland Australia for at least 35 000 years (Smith 2013), and influenced ecosystems through hunting, burning, manipulation of water sources, dispersal of plant propagules and deliberate sowing of some plant species (Gammage 2011, Silcock 2018, Wilson et al. 2004). The nature and magnitude of Aboriginal environmental impacts is contentious, and while some of our conclusions may prove relevant we will not enter this debate here.

From the 1840s, two decades of concerted European exploration merged into rapid pastoral settlement from the 1860s. By the 1890s, the pastoral frontier had enveloped nearly all suitable country across inland eastern Australia. Today, Australia has more land area under managed grazing than any other country (Asner et al. 2004). Most of the study area is used for extensive cattle and, in the eastern and southern portions, sheep grazing, with relatively small areas occupied by mining leases and conservation reserves. Large macropods occur across the area and are most abundant in semi-arid regions (Pople and Grigg 2001), with high densities of feral and, increasingly, semi-domestic goats in eastern parts of the study areas (Pople and Froese 2012). Rabbits were historically in plague proportions throughout large areas south of the Tropic of Capricorn (23 °26′ 22″), but have declined since the introduction of *myxomatosis* and *calici* virus in the 1950s and 1990s respectively, although they remain in high densities in some areas (Scanlan et al. 2006). Feral camels roam the Simpson-Strzelecki dunefields, while horses, donkeys and pigs occur patchily throughout the study area (Edwards et al. 2004).

The introduction of domestic and feral herbivores represents the largest Holocene environmental change in inland Australia, which had supported relatively low densities of native macropods since the extinction of the Pleistocene megafauna ≈45 000 years ago (Fensham and Fairfax 2008). Since the 1901 New South Wales Royal Commission into land degradation (Noble 1997), much work has focused on ascertaining the impacts of this major land-use upheaval. However, the magnitude and causes of ecological change since pastoral settlement remain hotly debated, both in the scientific literature and the public sphere. Substantial degradation of Australian rangelands over the past 150 years has been attributed to European land management practices (Letnic 2000, McKeon et al. 2004, White 1997).

Early accounts are especially stark, suggesting that initial impacts, particularly during drought when over-optimistic stocking rates combined with rabbit plagues rapidly denuded vegetation leaving the land exposed to wind and water erosion (Dixon 1892, Ratcliffe 1938). A century of research and management programs have aimed to stem perceived declines in biodiversity, land condition and productivity. Table 2 summarises prevailing paradigms of degradation in the study area, the presumed causes, key references outlining these paradigms, the bioregions affected, and the relevant hypothesis that examines each paradigm.

However some scientists and long-term land managers argue that grazing is a sustainable land use with few adverse effects in the study area, particularly in perennial grasslands and on floodplains (Orr 1992, Phelps et al. 2007), or at least that changes are less pronounced, and ecosystems more resilient, than is commonly assumed (Eldridge and Lunt 2010, Mitchell 1991, Witt et al. 2006). Clearly, in the absence of a known reference state prior to pastoralism, a method for resolving these debates underpins ecological understanding and future management directions.

Based on a review of literature from the study area and arid lands globally, if areas of inland eastern Australia are ecologically degraded, we hypothesise that:

1. There will be clear evidence from the historical record of increasing tree and shrub cover in concert with a reduction in burning and changes in the density of native animals
2. There will be shifts in plant species abundance under different management regimes, including palatable and perennial species being replaced by unpalatable and annual species in grazed areas and an overall decline in plant species diversity

3. Some plant and animal species will have become rare or disappeared from the landscape
4. Introduced species of plants and animals will have proliferated, changing ecosystem structure and function

We consider each of these hypotheses below. Details of the methodology used for each of the principal studies drawn on can be found in Fensham et al. (2010), Fensham et al. (2011b), Fensham et al. in press, Silcock and Fensham (2013), Silcock et al. (2013), Silcock and Fensham (2014) and Silcock et al. (2014).

Hypothesis 1. Evidence of landscape change in the historical record

The explorer record reveals little evidence of native woody vegetation thickening across inland eastern Australia (Silcock et al. 2013). Explorers recorded large areas of dense woodland and scrub, particularly *Acacia*-dominated communities in the semi-arid zone, and there are no observations of open country now characterised by thick vegetation. The explorer record provides strong evidence that pre-pastoral semi-arid Queensland was a mosaic of open plains, lightly wooded downs, grassy woodlands and dense sometimes impenetrable scrubs, which explorers tried their best to avoid or struggled to forge a path through. Other studies that have employed the historical record systematically also reveal little evidence of unidirectional vegetation change (Denny 1987, Fensham et al. 2011a). These findings are supported by the results of Witt et al. (2006, 2009), who found only a modest average increase in canopy cover and substantial variation between sites in the central and eastern Mulga Lands since the 1950s, the period when major thickening is purported to have occurred.

There were no references to fire in mulga communities in three early explorer journals examined, encompassing >800km travelled through mulga woodlands, including in summer when Aboriginal people were noted firing the spinifex. This is consistent with research showing that fire is an extremely rare occurrence in eastern Australian mulga communities (Silcock et al. 2016), and typical fires have negligible long-term impacts on woody vegetation structure (Silcock et al. 2017).

Thus the advance of woody vegetation and the mechanism for its advance (lack of fire) is not supported in the explorers' record (Silcock et al. 2013). The explorers were not ignoring fire and it is recorded in spinifex-dominated communities, grasslands on the eastern edge of the semi-arid zone and along watercourses. The explorer record supports the contentious hypothesis that large macropods have increased dramatically in the semi-arid zone. The numerous records of medium-sized mammals that are now locally extinct supports the already well-documented decline of Critical Weight Range (35 – 5500g) mammals (Johnson 2006).

Hypothesis 2. Shifts in plant species abundance under different management regimes

Data from long-term grazing exclosures (established between 1981 and 1996) reveals little evidence of irreversible degradation at typical levels of grazing in mulga forests (Fensham et al. 2011b), dunefields (Silcock and Fensham 2013), floodplains (Silcock and Fensham 2013) or Mitchell grasslands (Fensham et al. 2014). The floristic composition of dunefields and floodplains in north-eastern South Australia seem to be largely unaffected by domestic grazing, with no significant differences between grazed and ungrazed treatments in total species richness or abundance, life form richness or abundance, or herbaceous biomass (Silcock and Fensham 2013). In the low-productivity mulga forests, annuals were favoured by grazing and highest species richness for most lifeforms was found in the macropod-grazed treatment, an intermediate grazing disturbance that best approximates the evolutionary history of the environment. Palatable perennial grasses decreased but were not eliminated from grazed areas, and no species were found only in ungrazed treatments (Fensham et al. 2011b). Overall, the findings are not consistent with established assertions that long-grazed mulga has crossed functional thresholds that limit recovery. In the Mitchell grasslands, livestock grazing altered plant composition but did not cause declines in dominant perennial grasses or overall species richness. Neutral, positive, intermediate and negative responses to grazing were recorded, but no single lifeform group was associated with any response type (Fensham et al. 2014).

It is possible that grazing-sensitive species had been lost from these systems prior to erection of exclosures, as reported for some sites in the United States (Valone et al. 2002) and South Africa (Seymour et al. 2010). If a system had already passed into a degraded state prior to the erection of the exclosures, this could account for negligible differences between treatments. However, in Australian arid ecosystems it is difficult to identify candidates amongst the perennial grasses that have been completely removed from large areas. If perennial grasses have been diminished and not removed they should exhibit recovery after long periods of grazing protection, particularly in low-productivity environments where competition does not exert a strong influence and after periods of high rainfall such as those that preceded the studies reported upon here. Generally comparisons between grazing typical of commercial domestic stock and grazing by native macropods or from long-term total grazing protection reveal few significant differences in the abundance of palatable and unpalatable perennial grass species (Fensham et al. 2011b, Fensham et al. 2014, Silcock and Fensham 2013). Some of the perennial grass species that exhibit recovery with grazing relief in some situations do not exhibit a response in others (Table 3). An exception is the dominant palatable perennial grass *Monachather paradoxa* from mulga woodlands, although this species is typically more abundant in macropod-grazed treatments than both treatments that are ungrazed or grazed by domestic livestock (Fensham et al. 2011b; Table 3).

It is possible that the soil surface has been altered after long periods of livestock grazing to limit the recovery of perennial grasses and other grazing sensitive species after grazing. A study in desert dune swales employed an experimental framework utilising water-remote gradients to compare areas that have rarely been grazed by livestock (far from artificial water-points) to areas that have been intensively grazed by cattle for 40 years (close to artificial water-points) (Fensham et al. 2010). In these arid environments perennial grasses are generally rare, but one species, *Eriachne aristidea* was sufficiently abundant for statistical analysis but had no significant response along the grazing gradient. In general there was only one species that exhibited restriction to the rarely grazed water-remote areas and this pattern could be expected by chance.

Livestock grazing typical of current commercial enterprises seems consistent with conservation of plant species diversity across most of inland eastern Australia. The results from the exclosures suggest non-equilibrium vegetation dynamics (Illius and O'Connor 1999) may prevail in our study area. Periods of rainfall more than twice the yearly average occur between 6 and 13 times per century across the four bioregions (Table 1) and when they occur, available forage is sufficiently abundant that stocking rates are rarely high enough to inhibit seed-set of even most palatable species. During extended dry periods these perennial-dominated systems are mostly bare ground, with many shorter-lived grasses dying and persisting in the seedbank or the dominant perennials being reduced to twigs or stubble.

Livestock cannot be sustained and must be removed, ensuring persistence of palatable perennials (Silcock and Fensham 2012). The notable exception is mulga communities where the palatable *Acacia aneura* can sustain stock through drought, and this is considered a major factor predisposing these communities to degradation (Beeton et al. 2005). While this may contribute to the greater floristic differences between grazing treatments relative to the other systems studied, assertions of irreversible degradation are not supported by available data. It has also been suggested that productive habitats dominated by palatable perennial species situated within less productive ecosystems may be subject to long-term degradation (Illius and O'Connor 1999, Silcock and Fensham 2012), and further research is required to investigate these impacts.

Hypothesis 3. Some species have become rare or extinct

A small number of plant species are disfavoured by grazing in mulga (Fensham et al. 2011b), Mitchell grasslands (Fensham et al. 2014, Orr and Phelps 2013a) and the Simpson Desert (Fensham et al. 2010). Most species with negative responses to grazing were perennial grasses, legumes and vines. These 'decreaser' species are probably less abundant than in the pre-pastoral landscape. However, all remain widespread and common and very few have declined so dramatically to be considered rare at a landscape scale.

If grazing has driven species to the point of extinction they should persist in areas that livestock cannot readily access because of their remoteness from water. Grazing gradients revealed no evidence of plants that are so sensitive that they can only survive in water remote areas (Fensham et al. 2010, Silcock and Fensham 2014). Grazing gradients do reveal species favoured and disfavoured by grazing (Landsberg et al. 2003), but the few species that are only abundant at water-remote locations could easily have been expected to occur there by chance (Fensham and Fairfax 2008). While the association of grazing-sensitive plants with water-remote refuges has not been established for any species, the conclusion of Landsberg *et al.* (2003) that there are more species consistently favoured by water-remoteness than disfavoured appears robust (Fensham and Fairfax 2008).

An alternative approach, then, is to survey for species considered to be potentially rare and/or threatened and assess their viability in the grazed landscape. Targeted surveys (2800 hours) for plant species identified as rare and/or potentially threatened in western Queensland revealed many to be widespread and abundant at least in good seasons (Silcock et al. 2014). Their apparent rarity was due to sparse collections across vast and often inaccessible areas combined with temporal rarity for short-lived and geophytic species. Large (>1000 plants), healthy and regenerating populations of 61 of the 91 species (67%) considered to be potentially rare and/or threatened were found (Silcock et al. 2014). Although 27 species will remain listed under IUCN criteria by virtue of being naturally restricted, only six terrestrial species are threatened or declining under current management: three palatable long-lived trees and shrubs with limited recruitment at many populations, two grazing-sensitive woody forbs and an annual forb (Silcock et al. 2014, 2019). No species were found to be restricted to water-remote refuges (Fensham et al. in press). This runs counter to most conservation planning documents which cite grazing by domestic and feral herbivores and altered fire regimes as default threats for most rangeland species (Briggs and Leigh 1996). Overall, there is no evidence that any dryland plant species have become extinct since pastoral settlement, and only a handful have declined to the extent that they have become rare at a regional scale.

In contrast, GAB springs have been subject to extinction and diminishment due to aquifer drawdown since pastoral settlement, and their dependent species of plants and animals have also declined, including numerous local extinctions (Rossini et al. 2018). Numerous palatable trees and shrubs are in decline across the study area (Auld et al. 2019, Silcock et al. 2019, Tiver and Andrew 1997), although the long-term dynamics of their populations require further research (Fensham et al. in press).

Extinctions and declines of formerly abundant mammals, particularly ground-dwelling species falling within the Critical Weight Range, are catastrophic (Woinarski et al. 2015). Nineteen mammals have disappeared from inland eastern Australia, while a further 12 are listed as either Endangered or Vulnerable and nine as Near Threatened (Woinarski et al. 2014). Flow-on effects of some mammal declines on ecosystem structure and function have been suggested, however the magnitude of these effects remains speculative (Johnson 2006, Noble et al. 2007). Birds and reptiles have fared better, with no recorded extinctions from inland eastern Australia. Of the 17 birds (including subspecies) which are listed or considered potentially rare and/or threatened, 14 show no evidence of decline in the study area (Garnett et al. 2011), while there is no evidence that any reptiles have declined (Wilson and Swan 2010). However, there are historical declines documented for some unlisted species (Franklin et al. 2005), while many species remain too poorly known to allow confident assessments of their status. Mirroring the results of rare plant surveys, recent fauna surveys in the study area have revealed that some species considered rare or restricted are actually quite abundant and widespread, at least in certain seasons (Dickman 2013, Woinarski et al. 2014).

Myriad factors are implicated in arid zone mammal declines and extinctions, primarily introduced predators, habitat degradation, inappropriate fire regimes, competition from introduced herbivores and interactions between these factors (McKenzie et al. 2007, Woinarski et al. 2014, Woinarski et al. 2015). The results presented here, which show that changes to vegetation and fire regimes are more modest than often assumed, support the argument of Johnson (2006) that introduced predators are primarily responsible for mammal declines, at least in the study area. Feral cats and/or foxes are directly implicated in the decline of 28 threatened mammal species in the study area, while there is strong correlative evidence of negative impacts of domestic and/or feral herbivores for only four species. The timing of extinctions also supports this hypothesis, with many species disappearing before pastoralism or rabbits had become established in more arid areas, whereas feral cats had colonised all of Australia by the 1890s (Abbott 2002, Woinarski et al. 2014).

Hypothesis 4. Proliferation of introduced species

Although more than 200 exotic flora species have been recorded in the study area, most are restricted to disturbed areas such as towns, homesteads and roadsides, or have become naturalised as scattered components of native vegetation communities. Only a handful of species have proliferated to the extent that they substantially impact ecosystem structure and function.

Buffel grass (*Cenchrus ciliaris*) is the most widespread exotic species in inland eastern Australia, and the one with the clearest ecological impacts. It is restricted by soil type in the study area and is most abundant in the eastern Mulga Lands where its spread is promoted by broadscale clearing and sowing seed, but data collected following recent wet summers indicate that it continues to expand in other areas of inland Australia (Clarke et al. 2005, Fensham et al. 2013). The negative impacts of buffel grass on biodiversity and ecosystems are well documented (Fensham et al. 2015, Franks 2002, Miller et al. 2010), however it is productive fodder for cattle, continues to be sown by some pastoralists, and there is little prospect for broadscale control once it has become established (Friedel et al. 2011). The biggest impact of pastoralism in Mitchell grasslands is probably the spread of the woody leguminous trees prickly acacia (*Vachellia nilotica*) and, to a lesser extent, mesquite (*Prosopis* spp.), which have transformed over 60 000 km^2 of grassland into thorny scrubland (Osmond 2003, Spies and March 2004). Cattle ingest the pods of both species and are a vector for seed dispersal.

As discussed above, introduced cats and foxes are the major cause of mammal declines, represent the greatest ongoing threat to surviving threatened fauna and appear to be responsible for the striking disparity between dramatic fauna declines and the persistence of the arid zone flora since European settlement. Introduced feral herbivores occur patchily across the area. Although much-reduced from historical densities, rabbits continue to inhibit recruitment of some plant species (Denham and Auld 2004), while feral goats also appear to be limiting recruitment of some perennial species although the nature and extent of this problem requires further research. Camels inhabit the Simpson Desert, however the nature and severity of their impacts are not well documented (Edwards et al. 2010). Exotic and translocated aquatic species, particularly gambusia (*Gambusia holbrookii*), carp (*Cyprinus carpio*) and goldfish (*Carassius auratus auratus*), pose a major threat to the biological integrity of inland waterways (Haynes et al. 2009), and to the survival of native fish in spring-fed wetlands (Kerezsy and Fensham 2013).

Critical re-assessment of degradation narratives

Overall, there is little evidence for irreversible degradation of the vegetation of inland eastern Australia, and the results presented here provide little empirical support for the dominant prevailing narratives of ecological change (Table 2). The life histories of plant species and inherent climatic limitations have allowed much of the flora to persist despite the massive upheavals initiated by pastoral settlement. This is not to say that there are no impacts of overgrazing, including accelerated erosion, species declines and weed invasion. Heavily grazed areas close to water points have borne the brunt of grazing pressure with associated biotic and abiotic effects (Andrew and Lange 1986, Eldridge et al. 2011b). Even relatively resilient communities such as Mitchell grassland can be impacted by consistently high stocking rates (Hall and Lee 1980, Orr and Phelps 2013b). However, climate fluctuations and subtle soil differences often have greater effects on floristic composition than grazing (Fensham et al. 2010), and the conservation of plant biodiversity seems largely compatible with commercial pastoralism across most of the study area.

The main unequivocal examples of degradation are the loss of a suite of medium-sized mammals (Woinarski et al. 2015), extinction of GAB springs and their dependent organisms through aquifer drawdown (Rossini et al. 2018), lack of regeneration of some palatable perennial plants (Auld et al. 2015, Silock et al. 2019, Tiver and Andrew 1997), and invasion of prickly shrubs and buffel grass which have altered ecosystem structure and function across large areas (Fensham et al. 2015, Franks 2002, Spies and March 2004). There are also areas where the magnitude and effects of landscape change remain uncertain, particularly with regard to the long-term dynamics of shrubs in the Mulga Lands (Silcock et al. 2017). The possibility that productive habitats dominated by perennial species situated within less productive landscapes may be most vulnerable to degradation requires further research (Silcock and Fensham 2013).

Our critical assessment of ecological change makes priorities for conservation and sustainable management relatively straightforward. Conservation of the remaining GAB springs and their endemic species is a priority, particularly in the face of increasing demands on groundwater for extractive industries (Currell et al. 2017). Grazing-sensitive species, particularly palatable perennials with limited recruitment, form a small but vulnerable component of the inland eastern Australian flora and will benefit from the creation of large water-remote reserves where the impacts of domestic, feral and native herbivores are minimised. Research targeting palatable perennials with limited recruitment will shed further light on their life histories, threat status and management required to ensure their long-term persistence. Control of foxes and cats, particularly in response to infrequent irruptions following exceptional rainfall events, is critical to the survival of vulnerable mammals (Woinarski et al. 2014).

Fundamentally, the term degradation is value-laden, referring to a change of state that is judged to be negative relative to subjectively-chosen criteria, usually grounded in utilitarian considerations. Changes resulting in reduced pastoral productivity, such as perceived woody encroachment and decreases in groundcover, will be perceived as detrimental. However, some of the most disastrous changes from a biodiversity perspective, particularly conversion of native woodlands and pastures to monocultures of introduced perennial grasses, are rarely couched in terms of 'degradation'. Conversely, areas characterised by an abundance of 'woody weeds' and/or 'undesirable' native pasture species are routinely considered degraded, but in reality may harbour increased diversity of plant and animal species and show no deterioration in functional and structural indicators of landscape health (Bestelmeyer et al. 2018, Eldridge et al. 2011a, Good et al. 2012).

Ecological histories of rangeland areas will be as diverse as the rangelands themselves, and any global narratives will invariably involve simplifications and generalisations. Multidisciplinary regional studies combining historical sources, measurement of sites with different management histories and targeted surveys for sensitive and rare elements of the flora and fauna can allow critical assessments of ecological change in regions subject to abrupt management upheavals. In particular, focusing on the fate and trends of individual species identified as potentially rare and threatened holds substantial potential as both an unambiguous assessment of degradation and a means of prioritising conservation effort. The methods described here are broadly transferable across rangelands characterised by similar issues and debates, and other innovative approaches will emerge.

It seems the very characteristics long thought to render arid lands fragile, especially the prevalence of drought and dust, dominance of annuals and extended periods of low groundcover, may actually confer resilience to these ecosystems. The perception of drought as a stress on plants, animals and country is a hallmark of degradation narratives. Drought-affected land is often described as 'suffering…damaged, wrong, corrupted by lack of rain' (Arthur 2003:43). In reality, a climate characterised by long dry spells punctuated by unpredictable 'boom' events, and the associated adaptations of the flora, have conferred extraordinary resilience on the ecosystems of inland eastern Australia in the face of massive management upheavals imposed by pastoral management over the past 160 years. We should reframe our thinking to view arid lands as resilient but unpredictable, as dependent on extended drought as on the much-celebrated booms.

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Table 1. Rainfall summary for the four bioregions covered by this study. Climate data was derived for each 0.5 degree intersect of latitude and longitude and assigned to a bioregion to derive mean values, minimum and maximum. The frequency and magnitude of extreme rainfall events was derived for the location in the geographic centre of each bioregion.

Bioregion	Mean annual rainfall (Range)	Frequency of >200% rainfall.100y^{-1}	Frequency of extreme 3-y drought (<50% expected rainfall).100y^{-1}	Percentage of expected rainfall during most severe 3-yr drought (1900-2014)	Year of worst drought
Channel Country	202 (124-364)	9.9	10.8	32	1929
Mitchell Grass Downs	366 (179-530)	6.3	2.7	27	1929
Mulga Lands	345 (220-522)	7.2	4.5	39	1929
Simpson Strzelecki Dunefields	147 (112-244)	12.6	10.8	22	1929

Table 2. Prevailing paradigms/presumed changes indicating degradation in inland eastern Australia, summarising presumed causes of change, bioregion/s affected, key references outlining presumed changes, and the hypothesis/es that cover each presumed change in this study.

Prevailing paradigm/ presumed change	**Causes of change**	**Bioregion/s affected**	**Hypothesis**	**References**
General thickening of woody overstorey vegetation in semi-arid zone, especially *Acacia aneura* and *A. cambagei*	Preferential grazing of perennial grasses; reduced fire frequency and intensity	Mulga Lands; Mitchell Grass Downs	1	Gammage (2011), Noble (1997); see Silcock (2013) Appendix 1 for comprehensive list of references
Palatable and perennial species replaced by unpalatable and annual species	Increased total grazing pressure (domestic, feral and native herbivores)	All	2	Cingolani et al. (2005), Hacker et al. (2006), Hunt (2001), Seymour et al. (2010), Valone et al. (2002), Watson et al. (1997)
Overall decline in plant species diversity	Increased total grazing pressure (domestic, feral and native herbivores)	All	2	Cingolani et al. (2005) and references therein
Some grazing-sensitive plants only survive in water-remote refuges	Increased total grazing pressure (domestic, feral and native herbivores); provision of artificial watering points	All	3	Fensham and Fairfax (2008), Landsberg et al. (2003)
Some plant species have become extinct or rare, and/or are declining across the study area	Grazing pressure (domestic, feral and native herbivores)	All	3	Briggs and Leigh (1986), Hacker et al. (2006), Hunt (2001), Seymour et al. (2010), Watson et al. (1997)
Increase in range and abundance of macropods in semi-arid zone	Provision of artificial watering points; predator control;	Mulga Lands; Mitchell Grass Downs	1	Denny (1987), Pople and Grigg (2001); see Silcock et al.

	habitat modification			(2013) for comprehensive list of references
Contraction in range and abundance of medium-sized mammals	Feral predators (primarily cats and foxes), habitat modification	All	1, 3	Johnson (2006), Letnic (2000), McKenzie et al. (2007), Woinarski et al. (2014, 2015)
Proliferation of introduced plants and animals	Deliberate introductions, habitat modification	All	4	

Table 3. Summary of results for perennial grasses from studies using fenced grazing contrasts from a range of semi-arid and arid habitats. Treatments marked with the same letter are not significantly different. Upward arrows indicate a significantly higher abundance compared to another treatment marked with a different letter. Treatment combinations that are unavailable are left blank.

Species	Palatable	Ungrazed	Macropod grazed	Open grazed
Mitchell grasslands				
Aristida latifolia[1]	Yes	A^	A^	B
Aristida latifolia[2]	Yes	A		A
Astrebla elymoides[1]	Yes	A	A	A
Astrebla lappacea[1]	Yes	A	A	A
Astrebla pectinata[1]	Yes	A	A	A
Dichanthium sericeum[1]	Yes	A^	AB	B
Panicum decompositum[1]	Yes	A^	A^	B
Mulga woodland				
Aristida jerichoensis[3]	No	A	A	A
Aristida jerichoensis[4]	No	A	B^	AB
Eragrostis eriopoda[5]	Yes	A	A	A
Eragrostis microcarpa[6]	No	A	A	A
Eriachne helmsii	No	A	A	A
Monachather paradoxus[3]	Yes	A^	B	AB
Monachather paradoxus[4]	Yes	AB	A^	B
Monachather paradoxus[5]	Yes	AB	A^	B
Panicum effusum[6]	Yes	A	A	A
Themeda triandra	Yes	A	A	A
Thyridolepis mitchelliana[3]	Yes	A	A	A
Thyridolepis mitchelliana[5]	Yes	A^	A^	B

[1] Fensham et al. 2014; [2] (Foran and Bastin 1984); [3]Fensham et al. (Croxdale), [4]Fensham et al. (Lanherne), [5]Fensham et al. (Wallen)

Fig. 1. Archetypal degradation images: **(a)** Dust storm approaching Spearman, Texas, April 14 1935; **(b)** Farmer and sons walking in the face of a dust storm, Cimarron County, Oklahoma, April 1936 (US National Oceanic and Atmospheric Administration); **(c)** Russell Drysdale 'Crucifixion' (1946) and **(d)** 'Western Landscape' (1945), based on sketches made for *The Sydney Morning Herald* during drought in New South Wales (Art Gallery of NSW) **(e)** Sidney Nolan 'Untitled' (1952), part of a series of drought photographs taken on the Birdsville Track, originally commissioned by *The Courier Mail*, who ultimately deemed them too graphic to publish (National Library of Australia); **(f)** On the move through the dry dusty landscape in Burkina Faso, African Sahel (Andy Hall/Oxfam)

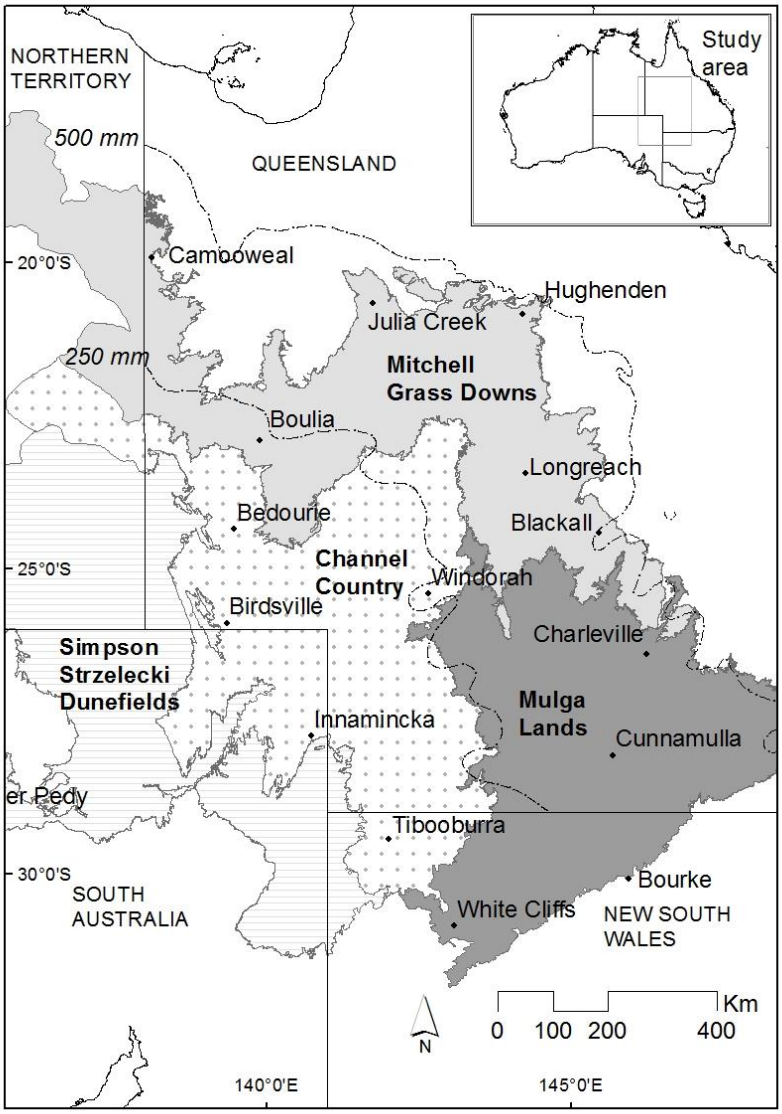

Fig. 2. Inland eastern Australia study area, showing state boundaries, biogeographic regions (shaded, labels bolded), 250 mm and 500 mm rainfall isohyets and major towns