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Old Man Saltbush mortality following fire challenges the resilience of post-mine rehabilitation in central Queensland, Australia

By Phillip B. McKenna, Natasha Ufer, Vanessa Glenn, David Doley, Stuart Phinn and Peter D. Erskine

Phillip B. McKenna *is a Senior Research Officer specialising in the fire resilience of rehabilitated ecosystems post-mining (Centre for Mined Land Rehabilitation, Sustainable Minerals Institute, The University of Queensland, Brisbane, QLD 4072, Australia; Tel: 0419 563 102).* ***Natasha Ufer*** *is a Research Officer specialising in ecosystem monitoring (Centre for Mined Land Rehabilitation, Sustainable Minerals Institute, The University of Queensland, Brisbane, QLD 4072, Australia).* ***Vanessa Glenn*** *is a Research Officer specialising in botany and ecosystem monitoring (Centre for Mined Land Rehabilitation, Sustainable Minerals Institute, The University of Queensland, Brisbane, QLD 4072, Australia).* ***David Doley*** *is an Honorary Associate Professor specialising in environmental plant physiology and forest tree physiology (Centre for Mined Land Rehabilitation, Sustainable Minerals Institute, The University of Queensland, Brisbane, QLD 4072, Australia).* ***Stuart Phinn*** *is a Professor with research interests in measuring and monitoring environmental changes using earth observation data and publishing/sharing ecosystem data (School of Earth and Environmental Sciences, The University of Queensland, Brisbane, QLD 4072, Australia).* ***Peter D. Erskine*** *is a Professor at UQ specialising in the environmental impacts and mitigation impacts from mining on ecosystem function (Centre for Mined Land Rehabilitation, Sustainable Minerals Institute, The University of Queensland, Brisbane, QLD 4072, Australia).*

Summary Landscape rehabilitation following mining is required to be resilient to disturbance impacts such as fire, drought and disease. As mining companies undergo the process of rehabilitation certification and mine closure, there are notable knowledge gaps on the ecological risks associated with mature rehabilitated landscapes, based largely on the assumption that rehabilitation is analogous to reference communities. However, the response to fire disturbance across a range of landscapes remains largely untested and in particular there is limited understanding of recovery traits of plant species that occur naturally or are commonly seeded into rehabilitation. In August 2018, a controlled fire was applied to 37 hectares of 12-year-old coal-mine rehabilitation in central Queensland, Australia. We used a combination of (i) ground plot surveys and (ii) drone imagery to compare the vegetation response of burnt woody species to unburnt controls prior to, and for, two years following the fire. The survival of the most dominant shrub species found on the rehabilitation site was significantly impacted by the fire. Old Man Saltbush (*Atriplex nummularia* Lindl. subsp. *nummularia*) recorded significant post-fire mortality, with ground surveys recording an average reduction of 89% of stems per hectare across the burnt site, while unburnt controls remained unchanged. The plot data analysis was supported with high spatial and temporal resolution drone imagery, classified using a Random Forest machine-learning approach. Change analysis of these maps showed a significant decline of 82% in Old Man Saltbush plant density and 92% reduction in foliage cover following the fire. In addition, the mean canopy area of individual Old Man Saltbush shrubs reduced significantly from a pre-fire mean of 11.3 to 4.8 m^2 two years following the fire. A spatial proximity analysis showed that those individuals that survived the fire were located significantly closer to unburnt areas and bare spoil, indicating that discontinuous ground fuel loads can greatly improve the survivability of individuals. This study provides new evidence on the contested fire sensitivity of Old Man Salt bush and demonstrates the risk that future climate-driven extreme events may have on the resilience of novel ecosystems.

Key words: *Atriplex nummularia* subsp. *nummularia*, machine learning, novel ecosystems, post-mine rehabilitation, reclamation, restoration.

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Implications for Managers

- Old Man Saltbush recorded high mortality rates when impacted by fire in post-mine rehabilitation containing high-ground fuel loads dominated by Buffel Grass.
- The resilience of novel ecosystems may be compromised if post-mining fire regimes and recovery traits of desired species are incompatible.
- Practitioners can assist the survivability of Old Man Saltbush individuals after fire by creating a mosaic of fuel loads and reducing the continuity of fuel loads associated with Buffel Grass pastures.
- Regulators should consider future fire events as a residual risk to rehabilitation sustainability, particularly when high fuel loads are not controlled.
- Multispectral drone imagery and machine-learning algorithms such as Random Forest can be used to improve monitoring of post-mine rehabilitation.

Introduction

In Australia, post-mining site rehabilitation aims to achieve four key goals – safe, stable, self-sustaining, and non-polluting landscapes (Queensland Treasury 2017; Government of Western Australia 2019). As policy, regulation and legislation are intended to deliver self-sustaining ecosystems, the concept of ecological resilience may be considered a primary and urgent objective, given the already changing climate and the increased exposure of rehabilitated ecosystems to more frequent disturbance events such as drought and uncontrolled fires (IPCC 2021; Commonwealth of Australia 2022). A question arises, therefore, around the risk of rehabilitation decline, loss or failure at some future point following mine closure and lease relinquishment. Recently published guidance documents suggest that the Queensland Government is concerned primarily with residual engineering risks (geotechnical and geochemical) rather than the risk of ecosystem failure following disturbance events (Government of Queensland 2018, 2022). Both mine operators and government regulators consider that rehabilitated vegetation communities do not pose a significant residual risk, based largely on the assumption that once a mine lease is relinquished, vegetation on a post-mining landform will function in a manner analogous to unmined landform vegetation (Government of Queensland 2022). While this may prove to be the case, there is currently no data to support this assumption, and its validity in eastern Australia is yet to be tested.

The Bowen Basin, in central Queensland, contains over 40 coal mine sites on which the predominant rehabilitation activity is seeding a mixture of exotic and native plant species to reinstate pasture for grazing or to re-establish native ecosystems for biological conservation (Erskine & Fletcher 2013). Due to the radical chemical and physical changes in properties of topsoil and spoil, mine site rehabilitation in the region is often conceptualised as securing 'novel' ecosystems which permit ecologically achievable outcomes for the industry and stakeholders (Hobbs *et al*. 2006; Doley & Audet 2013). Novel ecosystems, as indicated by their name, are 'new' ecosystems, and the unique combination and assemblage of flora and fauna is largely untested for resilience following disturbance impacts such as fire, drought, floods and disease. Novel ecosystems in the central Queensland region are mostly dominated by the exotic and invasive Buffel Grass (*Cenchrus ciliaris* L.) which has historically been favoured by rehabilitation practitioners for its wide environmental tolerance and rapid growth to stabilise highly erodible slopes (Erskine & Fletcher 2013). Additionally, Buffel Grass is often included in rehabilitation seed mixes at the request of future pastoralists.

Research on fire on rehabilitated mine sites is almost exclusively represented by Australian-based studies, especially in Western Australia's bauxite mines (Grant *et al*. 1997a,b; Smith *et al*. 2004; Bell & Hobbs 2007). This work has been foundational for the development of conceptual succession models for multiple theoretical ecological states (Grant 2006) and the recognition of achievable rehabilitation targets in highly disturbed environments (Doley & Audet 2013). However, comparatively little fire-related rehabilitation research has been conducted in eastern Australia, including in central Queensland, where Buffel Grass pastures have been shown to increase the risk of high-frequency and high-intensity fires (Butler & Fairfax 2003; Erskine & Fletcher 2013; McKenna *et al*. 2017).

Old Man Saltbush (OMSB) is a large, sprawling native shrub, commonly used in post-mine rehabilitation projects throughout arid and semi-arid Australia (Preston *et al*. 2002). Prior to European settlement, the species formed part of shrub-woodland communities associated with *Acacia* species in inland Australia (Moore 1953; Leigh & Mulham 1977) and forms a critical component of chenopod shrubland communities in central and south-western Australia where it is considered largely inflammable, but is at an increased risk of fire damage following the invasion of exotic grasses such as Buffel Grass (Friedel *et al*. 2014; Van Etten & Burrows 2018). A considerable amount of research has been conducted on this species, but with a focus on its potential for contributing to the national and international pastoral industry as a perennial fodder species (Wilson 1966; Hobbs *et al*. 2009; McKenna *et al*. 2009; Norman & Barrett-Lennard 2019) and landscape values such as protecting soils, livestock and fauna from wind (Lefroy *et al*. 1992; Collard *et al*. 2011). OMSB is tolerant of highly saline soils and can persist in low rainfall and drought conditions (Ledger & Morgan 2007; McKenna *et al*. 2009), making it a valuable agricultural species in semi-arid parts of the world. There is considerably less information, and conflicting opinions, on the fire ecology of OMSB. Gillieson *et al*. (1996) suggest that OMSB is sensitive to fire impacts and that non-indigenous methods of fuel reduction

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burning have impacted on the distribution of the species in southern Australia. In contrast, other sources suggest that this species is fire tolerant and fire resistant (Government of South Australia 2015). While the primary source driving the fire resistance narrative is unknown, it is possible that this conclusion may be derived from studies on the related *Atriplex lentiformis* (Torr.) S.Watson which has been shown to contain high field leaf moisture and reduced flammability in early studies in southern California (Montgomery & Cheo 1969).

The scarcity of knowledge on the fire ecology of rehabilitated landscapes in eastern Australia necessitates further enquiry. There is an assumption by rehabilitation practitioners and industry regulators that species seeded into rehabilitated areas have the resilience to cope with fire disturbance both at the species level and at the novel community level (Government of Queensland 2022). To test this assumption, we assessed the fire resilience of a 12-year-old rehabilitated pasture site in central Queensland, Australia, by quantifying the fire impact on, and post-fire recovery of OMSB which had been widely seeded across the study area. The first objective of this study was to measure the stem density of OMSB and other woody species seeded in the rehabilitation to determine changes over time in relation to fire. The second objective was to quantify changes in *OMSB* canopy cover across the burnt and unburnt control blocks. The third objective was to develop a deeper empirical understanding of the resilience of rehabilitation dominated by OMSB, and the management implications for novel ecosystems impacted by fire. We conclude by discussing the capacity for improved monitoring efforts using drone imagery to complement ground surveys for future monitoring of rehabilitated areas.

Methods

Site description

The study site was located in the Bowen Basin, central Queensland, Australia, approximately 200 km west of Mackay (Fig. 1). The climate in the area is defined by the Köppen climate classification as sub-tropical (BOM 2022), with distinct

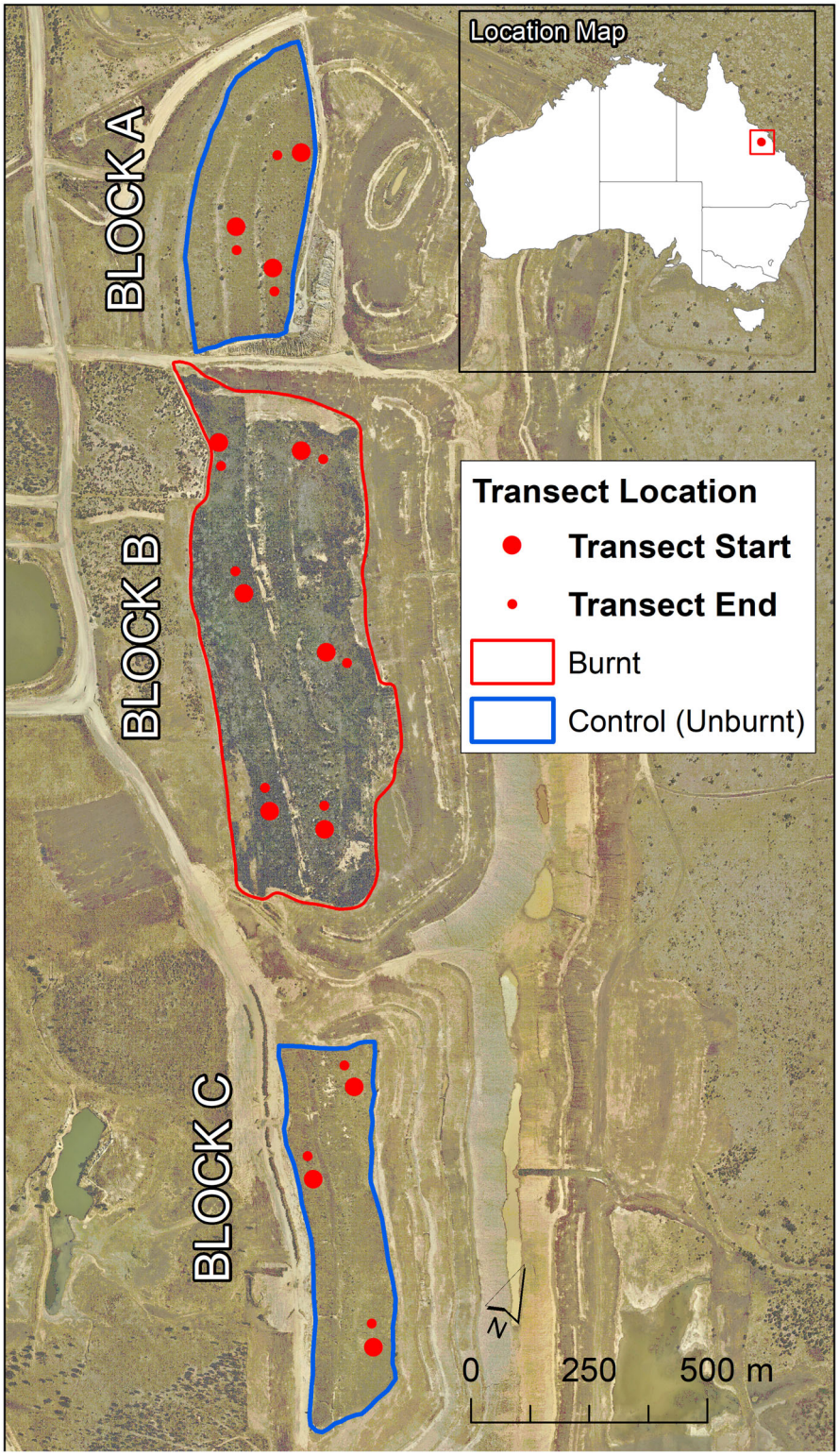


Figure 1. Study site location showing the burnt Block B and two unburnt control Blocks A and C. Each treatment had a total of six ground plots. Block A is 13 ha, Block B is 37 ha and Block C is 13 ha in size. Image is 15 cm red, green, blue aerial captured in November 2018 (two months post-fire) with the fire scar still visible.

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Table 1. Timeline of field data and drone imagery capture for the project

Epoch	Ground survey data	Drone image capture
Pre-fire	5–13 August 2018	12 August 2018
Post-fire	13–17 August 2018	14 August 2018†
+3 Months post-fire	12–15 November 2018	–
+6 Months post-fire	25–28 February 2019	25 February 2019
+12 Months post-fire	12–15 August 2019	12 August 2019
+18 Months post-fire	24–27 February 2020	–
+24 Months post-fire	10–14 August 2020	11 August 2020

The controlled fire occurred on the evening of 13 August 2018.
†Post-fire imagery was captured on the burnt Block B only.

wet and dry seasons and an average annual rainfall of 614 mm (BOM 2021).

The experiment was conducted on an operational coal mine that has been progressively rehabilitated with a post-mine land use of pasture for cattle grazing. Three rehabilitated blocks were assigned to the study, with each seeded in 2006 making the rehabilitation 12 years old at the time of burning. Block A and Block C were unburnt controls, each 13 hectares (ha) in size, and Block B (37 ha) was the burnt treatment. The site is a 'box-cut' (first entry to the coal seam) and as a result, all blocks had largely westerly aspect slopes. The western boundary of Block B graded into unmined, remnant brigalow regrowth and contained a protected Cultural Heritage Area for the Traditional Owners, The Birriah and Jaanga People, as identified under the Aboriginal Cultural Heritage Act (2003). Each rehabilitation block was trisected into equal parts and plots were randomly located in each portion, to give six plots in the burnt site (Block B) and six in the unburnt control sites (three in Block A and three in Block C) (Fig. 1). Burnt and control treatments were therefore of the same age and had similar site preparation conditions (reshaped mine spoil, topsoil depth, deep ripping, seed mix) as well as experiencing similar rainfall and climate due to proximity.

The rehabilitation at the study site can be described as a novel ecosystem (Hobbs *et al*. 2006; Doley & Audet 2013; Erskine & Fletcher 2013). The area was seeded with a mix of native and exotic pasture and woody species, and at the time of the study, the understorey across all sites was dominated by the high biomass pasture grass Buffel Grass. The midstorey was dominated by OMSB which was common across all three rehabilitated blocks and formed a shelter for fauna including Kangaroos (Macropodidae) and a range of bird species. The canopy consisted of a sparse cover of mixed *Acacia* and *Eucalyptus* species.

In August 2018, a controlled fire was applied to the 37-hectare Block B. The fire was a research burn designed to test the capacity for the rehabilitation to recover from fire disturbance (i.e., resilience to fire), while also serving to reduce fuel loads and reduce the risk of future wildfire at the mine. The fire was conducted at night in low temperature and high relative humidity conditions, a measure that is generally expected to produce fires with low to moderate intensities (Cheney & Sullivan 2008). However, the region was in drought and the grasses were estimated to be 85%–90% cured. As a result, based on estimates of ground fuel loads and rate of spread, the grassfire produced moderate to high intensities ranging from 200 to 12,000 kW/m (Byram 1959).

The site was assessed using a combination of (i) ground survey methods and (ii) remote sensing methods.

Ground surveys

Ground surveys were conducted across seven time periods to record fire impacts and changes over time (Table 1). Plot dimensions were 50 m × 10 m (500 m^2), and these were aligned with the longest side orientated at right angles to the slope to avoid bias associated with soil erosion rills and gullies. Within each plot, every live OMSB shrub was counted, and height class recorded (cm) using a height pole. OMSB counts were summed for each plot and converted to stems per hectare for the analysis.

Statistical modelling was conducted in R (R Core Team 2014) v 2022.12.0, Build 353, using the packages 'lme4' (Bates *et al*. 2015), 'emmeans (Lenth 2022) and plotting using the package' ggpolt2 (Wickham 2016). Generalised linear mixed models have been used successfully to apply to non-normalised, complex data containing both fixed and random effects (Bolker *et al*. 2009). Thus, we used generalised linear mixed models (GLMM) to test if stem density changed overtime depending on the treatment (burnt vs unburnt), using Block as a random effect to account for between Block variation. If the effect of Block was found to be non-significant, then we proceeded to generate generalised linear models (GLM) to relate the response of stem density with the interaction between months post-fire and treatment. Models were fitted with Gaussian error distribution with log link function and were compared with and without Block as a random effect for significance using Analysis of Variance (ANOVA). We also compared the GLMM and GLM significance based on to the Akaike Information Criterion (AIC) where lower AIC values represent a better-fit model (Burnham & Anderson 2002). Following model selection, pairwise comparisons were conducted using Estimated Marginal Means (least-squares means) to determine significant differences for:

i pre-fire and post-fire mean stem density modelled values within each treatment; and

ii burnt and unburnt mean stem density modelled values between each treatment for each epoch.

To accurately determine the effect of treatment on OMSB individuals that were present at the start of the experiment, we excluded those plots that did not contain OMSB at the pre-fire measurement (n = 2 for each treatment).

Drone-based remote sensing

High spatial resolution (<10 cm) multispectral imagery was captured using a

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MicaSense RedEdge sensor in five spectral bands – blue, green, red, and red-edge (RE) and near-infrared (NIR) (Table S1) mounted on a DJI M210 quadcopter drone. Flight paths were programmed to capture imagery with 80% forward overlap and 75% side overlap to generate accurate reflectance orthomosaic imagery, point clouds and digital surface models. Individual tiles were mosaiced using Pix4D software version (4.6.4) (Pix4D 2011). Individual bands were composited using ArcGIS Pro version 2.8.0 (ESRI 2021) to create 5-band raster images (Supporting Information Fig. S3).

Post-fire imagery was georeferenced using ground control points established using a Trimble 7× Global Navigation Satellite System (GNSS). The field rover data was post-processed using RINEX base station files downloaded via HxGN SmartNet (https://hxgnsmartnet.com/en-au) so that the field rover files were <10 cm x, y, z accuracy. All imagery in Block B was georeferenced to the immediate post-fire georeferenced imagery and the unburnt control imagery was georeferenced to the pre-fire imagery using ArcGIS Pro with an RMSE less than one pixel.

The supervised classification technique Random Forest (RF) (Breiman 2001) was used to classify the imagery for each epoch. Random Forest classifiers are commonly used in remote sensing analysis, including in fire severity (Collins *et al*. 2018; Gibson *et al*. 2020). Aerial photograph interpretation (API) was used to select training and validation points from the drone imagery using six cover classes (Atriplex, Euc/Acacia, Buffel/Pasture, Bare Spoil, Bare Topsoil and Shadow). Additional classes were added for the post-fire image and 6-month post-fire image to account for the presence of ash, scorched vegetation and regrowth of forbs/herbs. Separate collections of API points were created independently for each epoch and these were selected from the imagery to correspond with each class. Using the "caret" package (Kuhn 2021) in R Studio, the API points were randomly split into 70% training and 30% validation. The number of trees (ntree) was set to 1,000 and the number of variables randomly sampled was set to 20 (mtry = 1:20) (Abdi 2020). The points used in validation were assessed using an error matrix to calculate overall map accuracy, class accuracies and the Kappa statistic which shows the level of agreement between the API and classified raster (Jensen 2005).

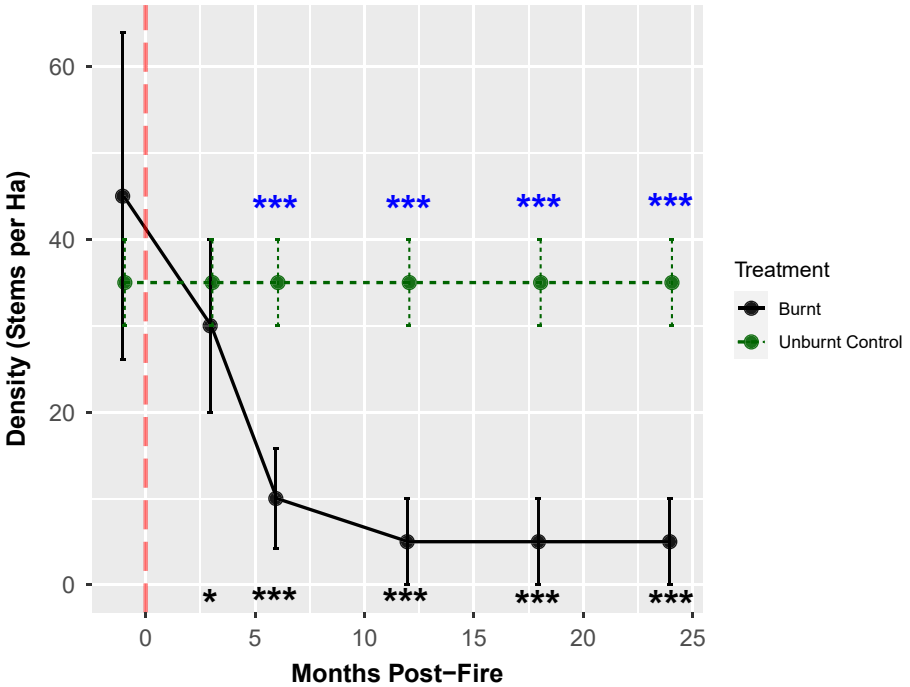


Figure 2. Time-series trends for stem density (stems per hectare) for Old Man Saltbush. Values are for those plots containing Old Man Saltbush (n = 4 in each treatment) expressed as mean values ± SE. Significance values derived from generalised linear modelling pairwise comparisons, blue stars = between treatment significance, black stars = within burnt treatment significance compared to pre-fire values (*p < 0.05, ***p < 0.001)

Following classification, the rasters were smoothed using a majority filter, masked to the rehabilitation boundary and converted to polygons. Shapefiles representing the classified OMSB were manually filtered and cleaned to remove false positives and each discrete canopy was numbered. We defined a canopy as a contiguous image area, consistent with the presence on the ground of one or more individual plants of OMSB. Canopy areas were averaged across each epoch and statistically compared (i) within each treatment (pre-fire vs post-fire) and (ii) between treatments (burnt vs control) using one-tailed t-tests using R software (R Core Team 2014) to determine directional significance, with alpha equal to 0.05. Paired t-tests were used for (i) and non-paired t-tests were used for (ii). Data were initially tested for equal variance using Levene's test and normality using Shapiro–Wilks and inference of outliers and skew using boxplots. Where the assumptions for t-tests were not met, we used the non-parametric equivalent Mann–Whitney Wilcoxon test. The p-values were adjusted for multiple comparisons using the Bonferroni method. For comparative purposes, unburnt control data were aggregated for both Blocks A and C.

A linear regression analysis was used to compare stem density measured in the field surveys with the RF classification stem density values. The centroid of each canopy was generated using the 'polygon-to-point' function in ArcGIS Pro, and points that fell within each ground survey plot area were included in a regression analysis. To account for the sprawling nature of OMSB we also included those centroid points that fell within 50 cm of the plot boundary line, and when >20% of the canopy polygon area was located inside the plot boundary.

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Table 2. Classification results showing overall mapping accuracy (OMA) and producers accuracy (PA) and users accuracy (UA) for the Old Man Saltbush class.

Block	Epoch	Overall classification			Atriplex classification accuracy		
		OMA (%)	Kappa	# Validation points	PA (%)	UA (%)	# Validation points
Burnt Block B	Pre-fire	94.4	0.930	555	96.5	96.0	172
Burnt Block B	Post-fire	90.2	0.876	1134	94.6	95.4	391
Burnt Block B	6 Months post-fire	86.0	0.822	1376	75.3	81.1	320
Burnt Block B	12 Months post-fire	97.1	0.964	800	98.8	99.4	165
Burnt Block B	24 Months post-fire	97.3	0.966	676	97.0	97.0	197
Unburnt Block A	Pre-fire	97.3	0.962	890	97.0	97.4	236
Unburnt Block A	6 Months post-fire	92.4	0.899	899	89.9	92.4	258
Unburnt Block A	12 Months post-fire	95.5	0.943	696	98.0	96.1	251
Unburnt Block A	24 Months post-fire	97.8	0.969	723	99.1	99.1	330
Unburnt Block C	Pre-fire	93.2	0.914	511	96.4	98.2	167
Unburnt Block C	6 Months post-fire	82.5	0.748	1621	81.6	83.6	505
Unburnt Block C	12 Months post-fire	96.5	0.955	769	98.6	97.9	283
Unburnt Block C	24 Months post-fire	97.2	0.964	712	98.3	96.7	242

Validation points were selected from the drone imagery using aerial photograph interpretation prior to the classification. Greyed cells represent those epochs with lower accuracy. Lower producers accuracy indicates a higher chance of missing Old Man Saltbush (omission error) and lower users accuracy indicates a higher chance of including other species erroneously (commission error).

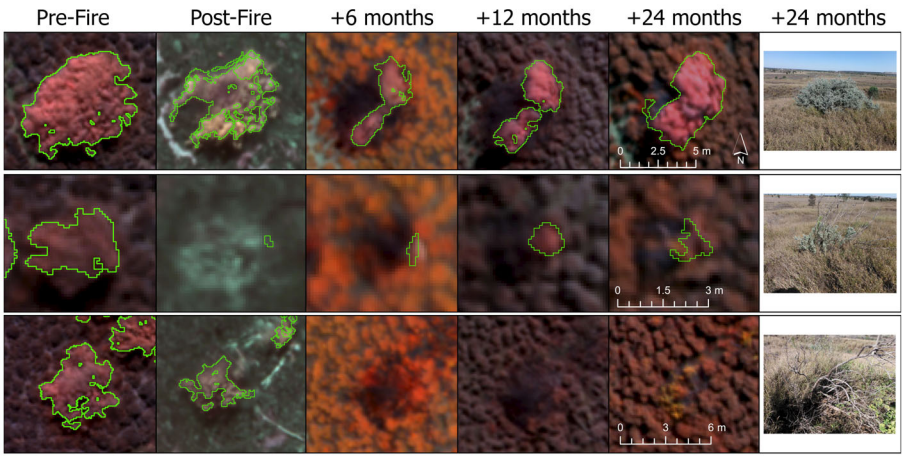


Figure 3. Examples of three fire-affected Old Man Saltbush canopies on the burnt treatment (Block B) identified in the time series drone imagery using Random Forest (RF) classification and associated field photographs taken at 24 months post-fire. Top two rows represent individuals that survived the fire, while the bottom row shows an individual that died following the fire.

The proximity analysis used bare spoil area derived from the RF classification as well as the unburnt areas derived from the fire severity map. Using the 'near' function in ArcGIS Pro, the distances from each pre-fire OMSB polygon to the nearest bare spoil and unburnt polygon (>2 m^2) were recorded and averaged for the burnt Block B. Then *t*-tests were conducted to compare the mean pre-fire distances to bare spoil and unburnt polygon edges for those individuals that were categorised as (i) alive and (ii) dead based on the 24-month post-fire classification. The fire severity map was created using image differencing of the green-red normalised difference vegetation index (GR NDVI) (Rouse *et al.* 1974; Wang *et al.* 2007). This index uses the green, red and near-infrared bands from each capture to generate rasters representing vegetation condition and biomass loss from fire impacts. See Supporting Information Table S21 and Fig. S12 for more information on fire severity methods and results.

Results

Vegetation densities from ground surveys

OMSB was recorded in plots across the two treatments at low to moderate densities. Pre-fire densities in those plots containing OMSB were 45 and 35 stems/ha^{-1} for burnt and unburnt treatments, respectively. Following the fire, density on the burnt block decreased significantly and by the end of the study, OMSB density had decreased to an average of five stems/ha^{-1}, a reduction of 89% over two years. Unburnt control averages for OMSB recorded consistent values throughout the monitoring period with an average of 35 stems/ha^{-1} that did not change over the two-year period (Fig. 2).

The statistical modelling comparison between GLMM and GLM showed that the random effect of Block was not significant ($p = 0.5073$, AIC = 618) and as a result, the GLM was used to test the months post-fire/treatment interaction (Supporting Information Table S16). Model outputs following pairwise comparisons suggested that there was a significant effect within the burnt treatment when compared with pre-fire values starting at three months post-fire ($p < 0.05$) and for the rest of the time series ($p < 0.001$). Comparisons between

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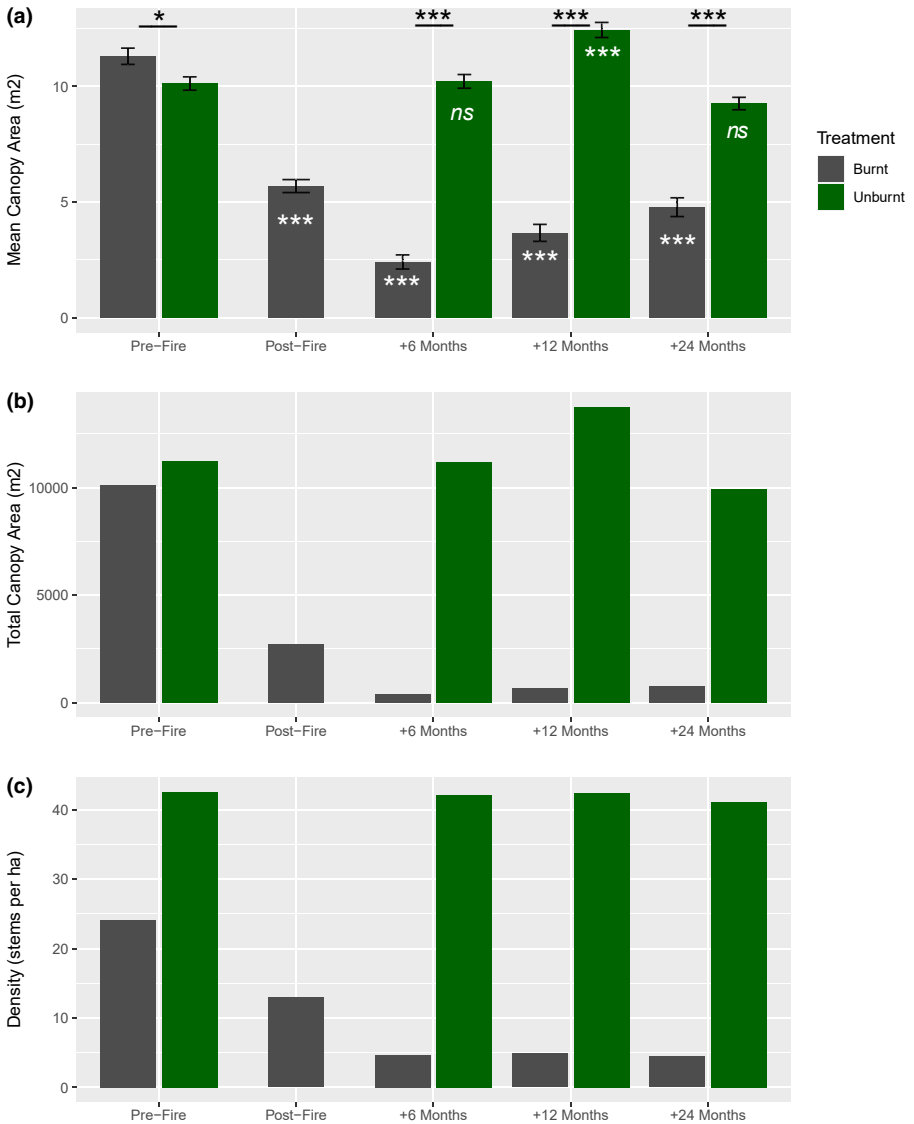


Figure 4. Old Man Saltbush canopy area summary derived by Random Forest (RF) classification. (a) Mean canopy area changes over time (*$p < 0.05$, ***$p < 0.001$, ns = not significant). Stars inside bars represent within-treatment pre-fire comparisons and stars above the bars represent between-treatment comparisons for each epoch. (b) Total canopy area and (c) density (stems per ha) changes over time. Note that the unburnt blocks did not have a post-fire capture. Further detail on statistical tests can be found in Supporting Information Tables S16-S19

treatments at each time period resulted in significant differences from six months onwards ($p < 0.001$) (Fig. 2, Supporting Information Tables S17–S19).

Vegetation mapping from drone imagery

The RF classification overall map accuracy ranged from 82.5%–97.8%, resulting in moderate to high accuracy and Kappa between 0.75 and 0.96 suggesting moderate to high agreement between API training data and the final classifications (Congalton 1991; Jensen 2005) (Table 2). Error matrices for each classification can be found in Supporting Information Tables S3–S15 and Supporting Information Figs S9–S11. In general, the results show high class accuracies, with reduced accuracy in the six-month post-fire assessments, due largely to the spectral confusion between OMSB and actively growing Buffel Grass tussocks.

Classification of OMSB canopies showed that mean canopy size within the burnt treatment decreased significantly following the fire from 11.3 to 5.69 m^2 ($p < 0.001$) and at 24-months post-fire were significantly lower than pre-fire values at 4.78 m^2 ($p < 0.001$) (Figs 3 and 4a). Canopies on unburnt control blocks were on average 10.1 m^2 pre-fire and by the end of the study decreased to 9.26 m^2. The 12-month post-fire epoch canopy area on the unburnt block (12.4 m^2) was significantly higher than the pre-fire mean ($p < 0.001$). This result was an artefact of the classification rather than an actual change in the site (Fig. 4a), as can be seen by the fewer fragments classified canopies compared to classifications in other epochs (Supporting Information Fig. S4).

Area calculations show that OMSB total cover decreased by 92% on the burnt block from 1.01 ha to 0.08 ha, compared to the unburnt blocks where cover remained relatively constant throughout the study with the exception of the 12-month epoch (Fig. 4b).

Plant density decreased by 82% from 24.16 to 4.46 stems/ha^{-1} on the burnt treatment, compared with the unburnt controls that remained constant and recorded 42.58 and 41.19 stems/ha^{-1} for the same epochs (Fig. 4c).

The linear regression between stem densities measured in the RF classification and the ground surveys showed a strong positive relationship for all sites ($R^2 = 0.84$, $p < 0.001$), particularly for the plots in the burnt treatment ($R^2 = 0.93$, $p < 0.001$) and for the unburnt treatment ($R^2 = 0.81$, $p < 0.001$) suggesting a high level of agreement between the plot data and classification results (Supporting Information Fig. S5).

Post-fire mortality for OMSB can be seen across the burnt treatment in Fig. 5. Analysis of the spatial distribution of mortality showed that those individuals that survived the fire were on average significantly closer to unburnt areas compared to those that were dead at the end of the study (6.29 and 9.1 m, respectively ($p < 0.001$)). Similarly, individuals that

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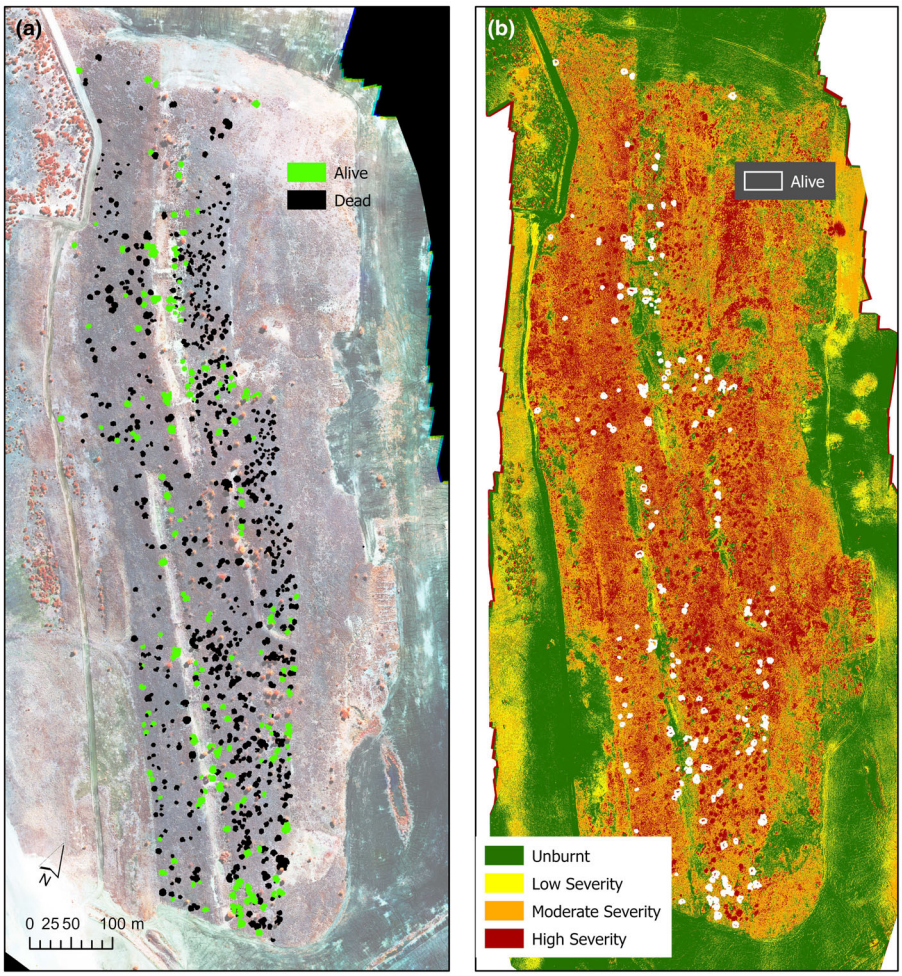


Figure 5. Mortality and survival of Old Man Saltbush shown in the pre-fire drone image (a) and derived fire severity map with those individuals that were still alive 24 months post-fire shown in white (b). The surviving individuals can be seen clustered around bare areas and unburnt areas, respectively.

survived the fire were on average 17.8 m distant from classified areas of bare spoil, compared to individuals that were dead at the end of the study and were on average further away at 26.4 m ($p < 0.001$) (Supporting Information Fig. S5).

Discussion

In the United Nations' Decade of Ecosystem Restoration, it is vital that mine site rehabilitation is resilient and capable of withstanding disturbance impacts. While the occurrence of fire on post-mine rehabilitation in eastern Australia is rare (McKenna *et al*. 2019), the mining industry has a responsibility as stewards of the land to create successful rehabilitation to provide for ongoing, sustainable land-use (Government of Western Australia 2014; Queensland Treasury 2017). This study showed that a controlled fire resulted in the widespread deaths of an important floristic and structural component of the rehabilitation at this site. A time series of drone imagery revealed a significant decrease in OMS stem density and cover, at both the plot and landscape scales following the fire. At the same time, unburnt controls showed no significant changes in stem density in the ground surveys, while unburnt control differences recorded in the drone imagery were in line with seasonal growth or were associated with classification limitations. Other species such as *Acacia*, *Eucalyptus and Corymbia* species showed no significant changes between the burnt and unburnt rehabilitation (Supporting Information Table S2). The weight of these outcomes provides new information on the fire sensitivity of OMSB and raises several questions on residual risk and the importance of species selection when developing novel ecosystems.

If the ultimate desired outcome of post-mine rehabilitation is long-term sustainability, then it is important that we understand the limitations of species that are commonly used across different sites. This includes the biotic and abiotic characteristics that define a species' geographical range as well as the niche tolerance and survival strategies for disturbances such as fire, flood, drought and disease. Novel ecosystems, by their nature, may include a collection of species drawn from disparate regions and locations, and as such, the seed mixes may contain species with mutually incompatible traits. For example, Buffel Grass (originally from Africa) has been shown to facilitate intense and frequent fires which can kill fire-sensitive components of an ecosystem (Fairfax & Fensham 2000; Franks 2002; Butler & Fairfax 2003). In turn, this creates areas for further migration, germination and establishment of the grass, which then fuel further fires in what has been called the 'grass-fire-cycle' (D'Antonio & Vitousek 1992; Bowman *et al*. 2014). The inclusion of species such as Buffel Grass in novel ecosystem seed mixes may therefore impact on the survival and recovery strategies of native species that have evolved under a different set of fire conditions, as defined through historic ecosystem associations. It is likely that the high fuel loads from Buffel Grass created intense fire conditions that impacted on the survival of OMSB and this is supported by our proximity analysis where individuals that were closer to bare areas and/or unburnt areas were more likely to survive fire. While many factors will contribute to fire conditions – weather, fire direction, topography, fuel loads and grass curing- it is feasible that unburnt areas and bare spoil are representative of lower-ground fuel loads and can be used as a proxy for reduced fire conditions that were more conducive to OMSB survival. This finding supports previous findings

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by McKenna *et al*. (2017) where discontinuous fuel layers reduced fire severity and intensity in Buffel Grass pastures, and promoted higher post-fire native species richness compared with topsoiled areas.

There is a dearth of information on the fire ecology of the vegetation communities where OMSB was found prior to widespread vegetation clearing on the Australian continent in the past few hundred years (McKenna *et al*. 2009; Van Etten & Burrows 2018). Leigh and Mulham (1977) suggest that prior to European settlement OMSB formed part of a woodland alliance with a discontinuous shrub stratum and a ground stratum that was "probably not well-developed". Gillieson *et al*. (1996) indicate that on the Nullarbor Plain, the original chenopod shrubland (including OMSB) was barely flammable apart from areas with grass cover, and that with increased frequency of controlled burns, OMSB cover has decreased and grasses such as *Stipa* spp. and *Danthonia* spp. are now more prevalent. Similarly, Burnside *et al*. (1995) suggest that fire and rabbit invasion resulted in community changes that favour fire-tolerant annual grasses and shrubs such as *Senna* and *Acacia* species. It is therefore highly probable that the persistence of OMSB is incompatible with high-intensity fires facilitated by Buffel Grass associations. OMSB is well known for survivability under drought conditions, but the degree of resistance to extremely high temperatures from high-intensity fire is unknown, and apart from our study, there is very little published information on the interactions between fire, OMSB and Buffel Grass.

A phenomenon affecting the resilience of the system in this study was the lack of post-fire recruitment. Surviving plants in the study area responded to fire by re-sprouting from stems, but no seedlings were recorded or observed in the post-fire environment. OMSB is predominantly dioecious with unisexual flowers in long panicles (BFF 1984). 'Fruits' (seeds encased in bracteoles) are up to 15 mm long and 11 mm wide, but few were observed during any epoch of the study. *Atriplex* species are known for complex dormancy and germination characteristics (Meyer 2008), and while the reasons behind the lack of recruitment are only speculative, the post-fire death of most OMSB plants and the failure of seedlings to establish after fire disturbance has impacted its persistence and future seed production at the site.

This study highlights the potential for remote sensing – conducted at appropriate spatial and temporal scales – to play a part in the ecological monitoring of rehabilitated and restored ecosystems. Historically, remote sensing has been used primarily for land cover assessments in rehabilitated landscapes, but in the last decade there has been an increase in new approaches to monitoring of post-mine rehabilitation using remote sensing (McKenna *et al*. 2020). Here we demonstrated the utility of multispectral drone imagery to classify changes over time to support ground surveys and introduce additional metrics such as shrub canopy area to assess impacts of fire disturbance and fire resilience. The statistically significant results for the metric mean canopy size were representative of the entire landscape and demonstrated comparable trends for stem density decline while unburnt controls remained relatively constant.

The seasonal variation observed in the drone imagery offered several insights into the limitations of using drones for time-series monitoring. When grasses are cured and the contrast is greatest between shrubs, trees and ground cover, the classification accuracies were highest (Table 2). However, following the wet season, the re-greening of grasses and forbs created notable confusion between all vegetation classes, which resulted in higher commission and omission errors (Supporting Information Tables S3–S15). As a result, the area calculations for the six-month post-fire epoch did not accurately depict ground cover, particularly for the burnt Block B. Manual filtering and numbering of individual canopies greatly improved the classification result, but was labour-intensive and time-consuming and is a process that would need to be automated in the future using code or geospatial tools to improve the utility of the remote sensing monitoring approach. Another challenge for the remote sensing assessments included the presence of canopy shadows in different locations due to capture times and seasons. This is largely dependent on the timing of image capture and can be minimised by appropriate planning – such as flying in the middle of the day and aligning the time of capture with each epoch. Additionally, *Acacia* and *Eucalyptus* canopies were highly variable in the imagery – sometimes represented by cloudy/convoluted circles and other times mostly absent, and only noticeable because of the adjoining shadow. This is primarily an error associated with the point cloud coverage of each canopy during the image capture and Pix4D processing and to some extent can be improved through multiple drone flights at different solar elevations (Fletcher & Mather 2020). Other improvements could be made by combining structural data such as LiDAR or canopy height models derived from the drone point clouds to improve the classification (Dash *et al*. 2019; Xu *et al*. 2020). Additionally, one of the benefits of using an RF pixel classifier is the option of increasing training sample sizes and re-running the classifier to improve classification accuracy. This was used to improve the six-month post-fire classifications, but the threshold to determine improvements in accuracy would require further study. Similarly, various parameters within the RF modeller such as predictor variables at each node and number of trees may be varied to improve class separation and class accuracy. It is clear therefore that converged monitoring of remote sensing and ground surveys has considerable potential, but improvements are required to instil confidence and encourage operational deployment of drones by rehabilitation managers.

The loss of OMSB from the burnt rehabilitation at this site does not disqualify the rehabilitation from Government certification and nor will it reduce the likelihood of successful mine closure. However, the loss of a key species indicates that ecological residual risk of post-mine rehabilitation has been largely underestimated. This has implications for government regulators who acquire parcels

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of land from mining companies that are rehabilitated to an acceptable level and for practitioners who are responsible for generating and managing rehabilitation activities including any future disturbances. While a grazing outcome may still be achieved, the loss of OMSB may have additional implications for soil water availability, soil stability and the loss of shelter for macropods and future stock as well as fodder to provide additional feed, and this may impair the overall quality of the site. There is evidence of recovery of surviving individuals, with significant increases in canopy area recorded after the six-month post-fire assessments, but given the lack of any observed seedlings, and ongoing dominant cover of Buffel Grass, it is possible that recovery of OMSB density may not occur on this site without management intervention. It is therefore recommended that future rehabilitation activities with OMSB should attempt to limit the cover of Buffel Grass to protect the shrubs from inevitable fire impacts.

Conclusions

The mining industry in Australia is responsible for rehabilitating vast tracts of land to functioning ecosystems that are self-sustaining and able to withstand future challenges from fire, drought, flood and disease. However, the development of novel ecosystems includes a species mix of unknown capacity to cope with disturbance events at both the plant level and the community level. We found that a controlled fire on 12-year-old rehabilitation resulted in an 89% decline in stem density for Old Man Saltbush at the plot scale, and an 82% decline in stem density and 92% decline in cover at the landscape scale. The results suggest that the assumption of low ecological risk for established rehabilitation requires further investigation, and a deeper understanding of disturbance response traits for species used in rehabilitation is required. The capacity for converged monitoring using remote sensing and ground surveys has been highlighted in this study, and the time-series imagery was able to reliably quantify land cover classes of interest to rehabilitation managers, including to the species level. The time-series drone imagery was highly sensitive to seasonal changes, including the re-greening of Buffel Grass tussocks and the associated class confusion needs to be accounted for when planning image capture to quantify changes over time for future rehabilitation monitoring.

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References

Abdi A. M. (2020) Land cover and land use classification performance of machine learning algorithms in a boreal landscape using Sentinel-2 data. *GIScience and Remote Sensing* **57**, 1–20.

Bates D., Mächler M., Bolker B. M. and Walker S. C. (2015) Fitting linear mixed-effects models using lme4. *Journal of Statistical Software* **67**, 1–48.

Bell D. T. and Hobbs R. J. (2007) Jarrah Forest ecosystem restoration: a foreword. *Restoration Ecology* **15**, S1–S2.

BFF (1984). Chenopodiaceae, Bureau of Flora and Fauna. *In: Flora of Australia Volume 4 Phytolaccaceae to Chenopodiaceae*. (ed A. S. George) pp. 81–317. Australian Government Publishing Service, Canberra, ACT.

Bolker B. M., Brooks M. E., Clark C. J. *et al.* (2009) Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology and Evolution* **24**, 127–135.

BOM (2021) *Bureau of Meteorology*. Australian Government. Available from URL: http://www.bom.gov.au/climate/averages/tables/cw_034038.shtml [Accessed 27 August 2021].

BOM (2022) *Bureau of Meteorology*. Australian Government. Climate Classification Maps. Available from URL: http://www.bom.gov.au/jsp/ncc/climate_averages/climate-classifications/index.jsp?maptype=kpngrp#maps [Accessed 18 December 2022].

Bowman D. M., MacDermott H. J., Nichols S. C. and Murphy B. P. (2014) A grass-fire cycle eliminates an obligate-seeding tree in a tropical savanna. *Ecology and Evolution* **4**, 4185–4194.

Breiman L. (2001) Random forests. *Machine Learning* **45**, 5–32.

Burnham, K. P.; Anderson, D. R., 2002: *Model Selection and Inference: A Practical Information-theoretic Approach*, 2nd edn, Vol. **65**. pp. 1–515. Springer-Verlag. The Journal of Wildlife Management, New York.

Burnside D. G., Holm A., Payne A. and Wilson G. (1995) *Reading the Rangeland: A Guide to the Arid Shrublands of Western Australia*. Department of Agriculture and Food, Western Australia, Perth.

Butler D. B. and Fairfax R. J. (2003) Buffel grass and fire in a gidgee and Brigalow woodland: a case study from Central Queensland. *Ecological Management & Restoration*. **4**, 120–125.

Byram G. M. (1959) Combustion of forest fuels. *In: Forest Fire: Control and Use*. (ed K. Davis) pp. 69–89. McGraw-Hill, New York.

Cheney P. and Sullivan A. (2008) *Grassfires. Fuel, Weather and Fire Behaviour*, 2nd edn. CSIRO Publishing, Collingwood, Victoria.

Collard S. J., Fisher A. M. and McKenna D. J. (2011) Planted Saltbush (Atriplex nummularia) and its value for birds in farming landscapes of the south Australian Murray Mallee. *Ecological Management and Restoration* **12**, 37–45.

Collins L., Griffioen P., Newell G. and Mellor A. (2018) The utility of random forests for wildfire severity mapping. *Remote Sensing of Environment* **216**, 374–384.

Commonwealth of Australia (2022) *State of the Climate 2022*. Canberra, ACT.

Congalton R. G. (1991) A review of assessing the accuracy of classifications of remotely sensed data. *Remote Sensing of Environment* **37**, 35–46.

D'Antonio C. M. and Vitousek P. M. (1992) Biological invasions by exotic grasses, the grass fire cycle and global change. *Annual Review of Ecology and Systematics* **23**, 63–87.

Dash J. P., Watt M. S., Paul T. S. H., Morgenroth J. and Pearse G. D. (2019) Early detection of invasive exotic trees using UAV and manned aircraft multispectral and LiDAR data. *Remote Sensing* **11**, 1–21.

Doley D. and Audet P. (2013) Adopting novel ecosystems as suitable rehabilitation alternatives for former mine sites. *Ecological Processes* **2**, 1–11.

Erskine P. D. and Fletcher A. T. (2013) Novel ecosystems created by coal mines in Central Queensland's Bowen Basin. *Ecological Processes* **2**, 1–12.

Fairfax R. J. and Fensham R. J. (2000) The effect of exotic pasture development on floristic diversity in Central Queensland, Australia. *Biological Conservation* **94**, 11–21.

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Fletcher A. and Mather R. (2020) Hypertemporal imaging capability of uas improves photogrammetric tree canopy models. *Remote Sensing* **12**, 1–26.

Franks A. J. (2002) The ecological consequences of buffel grass Cenchrus ciliaris establishment within remnant vegetation of Queensland. *Pacific Conservation Biology* **8**, 99–107.

Friedel M., Allan G. E. and Duguid A. (2014) Do we know enough about vegetation dynamics to manage fire regimes in Central Australia? *Ecological Management and Restoration* **15**, 128–132.

Gibson R., Danaher T., Hehir W. and Collins L. (2020) A remote sensing approach to mapping fire severity in South-Eastern Australia using sentinel 2 and random forest. *Remote Sensing of Environment* **240**, 111702.

Gillieson D., Wallbrink P. and Cochrane A. (1996) Vegetation change, erosion risk and land management on the Nullarbor Plain, Australia. *Environmental Geology* **28**, 145–153.

Government of Queensland (2018) *Managing Residual Risks in Queensland*. Discussion Paper. Part of the Financial Assurance Framework Reform Package, Department of Environment and Science, Brisbane.

Government of Queensland (2022) *Residual Risk Assessment Guideline—Interim*. Department of Environment and Science, Brisbane.

Government of Western Australia (2014). *Mining Rehabilitation Fund Act 2012*, 1–31.

Government of Western Australia (2019) *Guidance – How to Prepare a Mine Closure Plan in Accordance with Part 1 of Guidelines for Mine Closure Plans in Western Australia*, pp. 1–74. Department of Mines, Industry Regulation and Safety, Perth, WA.

Grant C. D. (2006) State-and-transition successional model for bauxite mining rehabilitation in the jarrah forest of Western Australia. *Restoration Ecology* **14**, 28–37.

Grant C. D., Koch J. M., Bell D. T. and Loneragan W. A. (1997b) Tree species response to prescribed burns in rehabilitated bauxite mines in Western Australia. *Australian Forestry* **60**, 84–89.

Grant C. D., Loneragan W. A., Koch J. M. and Bell D. T. (1997a) Fuel characteristics, vegetation structure and fire behaviour of 11–15 year-old rehabilitated bauxite mines in Western Australia. *Australian Forestry* **60**, 147–157.

Hobbs R. J., Arico S., Aronson J. *et al.* (2006) Novel ecosystems: theoretical and management aspects of the new ecological world order. *Global Ecology and Biogeography* **15**, 1–270.

Hobbs T. J., Bennell M. and Bartle J. (2009). *Developing Species for Woody Biomass Crops in Lower Rainfall Southern Australia – FloraSearch 3a Publication No. 09/043 Project No. UWA-98A*. (eds T. J. Hobbs, M. Bennell and J. Bartle) pp. 1–270. Rural Industries Research and Development Corporation, Barton, ACT, Australia.

IPCC (2021) Summary for policymakers. *In: Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. (eds V. P. Masson-Delmotte, A. Zhai, S. L. Pirani *et al.*) pp. 1–40. Cambridge University Press, Cambridge, UK.

Jensen J. R. (2005) *Introductory Digital Image Processing a Remote Sensing Perspective*, 3rd edn. Prentice Hall, Upper Saddle River, NJ.

Kuhn M. (2021) caret: Classification and Regression Training. R package version 6.0-90. Available from URL: https://cran.r-project.org/web/packages/caret/caret.pdf [Accessed 10 December 2022].

Ledger P. and Morgan C. (2007) Saltbush—a case for reintroduction. *Australian Farm Business Management Journal* **4**, 31–42.

Lefroy E. C., Dann P. R., Wildin J. H., Wesley-Smith R. N. and McGowan A. A. (1992) Trees and shrubs as sources of fodder in Australia. *Agroforestry Systems* **20**, 117–139.

Leigh J. and Mulham W. (1977) Vascular plants of the riverine plain of New South Wales with notes on the distribution and pastoral use. *Telopea* **1**, 225–293.

Lenth R. V. (2022) emmeans: Estimated Marginal Means, aka Least-Squares Means. R package version 1.8.3. Available from URL: https://cran.r-project.org/web/packages/emmeans/emmeans.pdf [Accessed 15 December 2022].

McKenna, David, J.; Bennell, M.; Mazanec, R., 2009: Review of Atriplex nummularia (Old-Man saltbush) and its potential for domestication as a fodder plant in Australian dryland farming systems. RIRDC publication No 09/044. In: Hobbs, T. J., J. Bartle & M. Bennell (eds.), *Reviews of High Priority Species for Woody Biomass Crops in Lower Rainfall Southern Australia FloraSearch 3b*. Barton, ACT, Australia, pp. 81–124.

McKenna P. B., Erskine P. D., Glenn V. and Doley D. (2019) Response of open woodland and grassland mine site rehabilitation to fire disturbance on engineered landforms. *Ecological Engineering* **133**, 98–108.

McKenna P. B., Glenn V., Erskine P. D., Doley D. and Sturgess A. (2017) Fire behaviour on engineered landforms stabilised with high biomass buffel grass. *Ecological Engineering* **101**, 237–246.

McKenna P. B., Lechner A. M., Phinn S. and Erskine P. D. (2020) Remote sensing of mine site rehabilitation for ecological outcomes: a global systematic review. *Remote Sensing* **12**, 1–34.

Meyer S. (2008) Part 2 specific handing methods and data for 236 genera: Atriplex L. Chenopodiaceae. *In: The Woody Plant Seed Manual Agriculture Handbook 727*. (eds F. T. Bonner and R. P. Karrfalt) pp. 283–290. Uniter States Department of Agriculture, Washington, DC.

Montgomery K. R. and Cheo P. C. (1969) Moisture and salt effects on fire Retardance in plants. *American Journal of Botany* **56**, 1028–1032.

Moore C. W. E. (1953) The vegetation of the south-eastern Riverina, New South Wales. I. the climax communities. *Australian Journal of Botany.* **1**, 485–587.

Norman H. and Barrett-Lennard E. (2019) *Potential of New Australian Oldman Saltbush Varieties to Fill Ruminant Feed-Gaps in Arid and Saline Areas of Pakistan*. Final report. ACIAR Project LPS/2016/022, Canberra, ACT.

Pix4D. (2011) Pix4Dmapper Pro: Version 2.1.

Preston C., Bellairs S. M. and Adkins S. W. (2002) Dormancy in selected species of Atriplex and Maireana. *In: Proceedings for the Fourth Workshop on Native Seed Biology for Revegetation*. (eds S. W. Adkins, S. M. Bellairs and L. C. Bell) pp. 91–97. Victoria, Australian Centre for Mining Environmental Research, Brisbane, Mildura.

Queensland Treasury (2017) Better mine rehabilitation for Queensland: discussion paper. *In: Prepared for the Queensland Government Interdepartmental Committee on Financial Assurance for the Resource Sector*, Queensland Treasury Corporation, Brisbane.

R Core Team. (2014) R: a language and environment for statistical computing.

Rouse J. W., Haas R. H., Shell J. A. and Deering D. W. (1974) Monitoring vegetation systems in the Great Plains with ERTS-1. *3rd Earth Resources Technology Satellite Symposium* **1**, 309–317.

Government of South Australia. (2015) *Fire Resistant Plants*. Government of South Australia, Adelaide, SA.

Smith M. A., Grant C. D., Loneragan W. A. and Koch J. M. (2004) Fire management implications of fuel loads and vegetation structure in jarrah forest restoration on bauxite mines in Western Australia. *Forest Ecology and Management* **187**, 247–266.

Van Etten E. and Burrows N. (2018) Fire regimes and ecology of arid Australia. *In: On the Ecology of Australia's Arid Zone*. (ed H. Lambers) pp. 243–282. Springer, Cham.

Wang F. M., Huang J. F., Tang Y. L. and Wang X. Z. (2007) New vegetation index and its application in estimating leaf area index of rice. *Rice Science* **14**, 195–203.

Wickham H. (2016) ggplot2 – elegant graphics for data analysis. *Journal of Statistical Software* **77**, 3–5 2nd edn.

Wilson D. (1966) The value of atriplex (saltbush) and kochia (bluebush) species as food for sheep. *Australian Journal of Agricultural Research* **17**, 147–153.

Xu Z., Shen X., Cao L. *et al.* (2020) Tree species classification using UAS-based digital aerial photogrammetry point clouds and multispectral imageries in subtropical natural forests. *International Journal of Applied Earth Observation and Geoinformation* **92**, 102173.

Supporting Information

Additional supporting information can be found in the following online files.

Data S1

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