Long–Term revegetation success of severely degraded chenopod shrublands

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Long-term revegetation success of severely degraded chenopod shrublands

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Abstract. The restoration of severely degraded vegetation communities is often said to require mechanical intervention. However, the degree of intervention required, and its capacity to successfully restore areas of bare (scalded) soil and high weed cover into functioning chenopod shrubland, is unknown. Ten years on from mechanical intervention and direct seeding using a Contour Seeder and Camel Pitter, the abundance and cover of species was compared across disturbed and undisturbed microtopographic zones using one-way repeated-measures ANOVA and pairwise *t*-tests. Along Contour Seeder rip lines, recruitment of perennial species was greatest in the furrow (e.g. direct seeded: $F_{2,78} = 27.15$, $P < 0.001$; wild-sourced: $F_{2,78} = 13.19$, $P < 0.001$), and annual (and short-lived perennial) species equal to, or greater, on the undisturbed flat (wild-sourced: $F_{2,98} = 43.91$, $P < 0.001$). At the species-level, these trends often coincided with the species' life-history strategy, but not always (e.g. the perennial *Atriplex stipitata* illustrated the annual trend; $F_{2,78} = 7.71$, $P < 0.001$). It is also important to note that a trend in recruitment could be driven by recruitment patterns in any one demographic phase. For example, the perennial trend in *Atriplex vesicaria* and *Sida fibulifera* abundance was driven by the recruitment of reproductive plants ($F_{2,48} = 15.57$, $P < 0.001$), or seedlings ($F_{2,36} = 5.24$, $P = 0.010$), respectively. These results indicate the functional benefits of mechanical intervention vary between species. The perennial and annual trends detected along Contour Seeder rip lines were less apparent along Camel Pitter rip lines where both perennial and annual species benefited from the initial disturbance. Overall, mechanical intervention facilitated the patchy recruitment of a *Maireana pyramidata* over *A. vesicaria* low shrubland, including several perennial and annual chenopods, indicating an early stage of recovery.

Additional keywords: direct seeding, functional restoration, Kimseed Camel Pitter (Disc Revegetation Seeder), Kimseed Linkage Rangeland Contour Seeder, regeneration, semi-arid.

Received 13 April 2017, accepted 25 September 2017, published online 31 October 2017

Introduction

In semi-arid and arid environments, heavy grazing and trampling by domestic livestock can reduce shrub cover, reduce leaf litter cover, break down soil crusts and increase soil erosion (Kinloch and Friedel 2005). These processes can have a snowballing effect on the availability of water and nutrients for seed germination and plant growth (Kinloch and Friedel 2005), leading to persistent changes in the vegetation community, soil structure, hydrological function, and nutrient cycling that may be difficult to reverse (Westoby *et al.* 1989; Hobbs and Cramer 2008; Cortina *et al.* 2011). These changes can lead to a vegetation community declining below an abiotic threshold into a degraded state (Hobbs and Cramer 2008). Degraded systems can have reduced system function (Hobbs and Cramer 2008; Tongway and Ludwig 2011), that can substantially reduce the production of domestic livestock on native pastures. Yet declines in productivity driven by intense grazing regimes can be difficult to distinguish from natural fluctuations, until persistent patches of bare ground form and signs of soil erosion become more apparent (e.g. as scalds; van den Berg and Kellner 2005; Thompson 2008).

Once the systems have degraded below an abiotic threshold, the removal of livestock alone may be insufficient for the system to recover (Westoby *et al.* 1989). In such cases mechanical intervention is considered necessary (Snyman 2003; van den Berg and Kellner 2005). Mechanical intervention can immediately improve the microclimate and soil moisture available for germination and plant growth. Machines built for this purpose vary in design (Wakelin-King 2011), but typically rip (break up) the soil surface and push soil into mounds, leaving a furrow on one or both sides of the mound. Ripping compacted soil improves infiltration and supports early root growth, and is particularly important for soils with a high clay content (Dalton 1993). Mounds (or contours) can be orientated to slow the movement of surface water, increase infiltration and water storage in the soil, and encourage deeper root systems (Tongway and Ludwig 2011; Azam *et al.* 2012). Deeper root systems aid the survival of growing plants during dry periods in regions with low rainfall (Dalton 1993; Azam *et al.* 2012). Furrows (or troughs) trap water (Snyman 2003; Tongway and Ludwig 2011; Wakelin-King 2011; DeFalco *et al.* 2012), leaf litter (Tongway and Ludwig 2011; Wakelin-King 2011), wind-blown seed (DeFalco *et al.* 2012), and

facilitate seedling establishment and survival (van den Berg and Kellner 2005). Deeper furrows can persist for longer (Powell 1998; Knight *et al*. 1998), but are still likely to fill after a few years (Tongway and Ludwig 2011; Wakelin-King 2011). The undisturbed area between rip lines (mounds and furrows) remains flat and compacted. Some machines allow for direct seeding or seedling planting. Burying seeds into the mound helps to reduce seed loss to predation and wind redistribution (Dalton 1993; Azam *et al*. 2012). Combining mechanical intervention with the direct seeding of native species can be a relatively cheap way to revegetate large areas.

There are few cases where scalded and erosion-prone soils in semi-arid environments have been successfully restored (e.g. Odermatt 1998; Thompson 2008). The processes underpinning the successful establishment of different species are not well understood (Knight *et al*. 1998; Azam *et al*. 2012). Perhaps more importantly, what constitutes success is usually poorly defined (Cortina *et al*. 2011), defaulting to measures of initial seedling survival (e.g. Knight *et al*. 1998; Maestre *et al*. 2003; van den Berg and Kellner 2005), rather than the longer-term restoration of a functional community (Hobbs and Cramer 2008; Cortina *et al*. 2011; Suding 2011) (although see Snyman 2003; Schneemann and McElhinny 2012). As a result, it is not clear that mechanical intervention and direct seeding can sufficiently restore degraded areas into self-sustaining, functional communities. There is a real possibility that the successful restoration of severely degraded systems in semi-arid systems will be expensive, difficult and slow (Snyman 2003; van den

Berg and Kellner 2005; Hobbs and Cramer 2008; Lawley *et al*. 2013). If so, we risk losing productive ecosystems and biodiversity (Menz *et al*. 2013), and mis-spending limited conservation dollars (Maron *et al*. 2012). This investigation aimed to evaluate the success of a restoration program 10 years following mechanical intervention and direct seeding, based on:

(i)   The survival, reproduction and recruitment of seedlings of direct seeded species,
(ii) The recruitment, survival, and subsequent reproduction of wild-sourced species, and
(iii) The degree to which the severely degraded areas have been restored into *Maireana pyramidata* over *Atriplex vesicaria* low shrubland.

Methods

Study sites

This investigation was performed within the Ikara-Flinders National Park, located in the Flinders Ranges (Fig. 1). The Flinders Ranges span temperate and arid regions, receive higher winter rainfall, and are geomorphologically and geologically complex (Brandle 2001). These factors enable the Flinders Ranges to support a greater diversity of vegetation communities than the surrounding area (Brandle 2001). Pastoralism was developed in 1850s (Northern Flinders Ranges Soil Conservation Board 1997), leading to the general area being heavily grazed by high numbers of domestic livestock (e.g. 100–260 sheep km^{-2}), feral goats, introduced European rabbits and abundant native
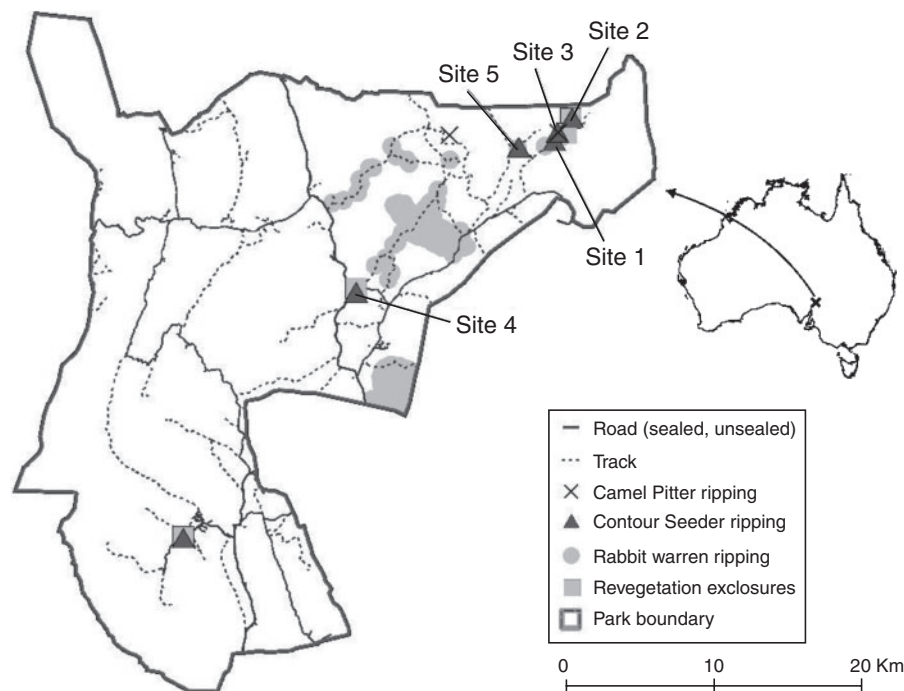
Fig. 1.   Location of revegetation activities in the Ikara-Flinders Ranges National Park (Australia). Activities included establishing herbivore exclosures, and ripping and direct seeding using a Camel Pitter, Contour Seeder, and bulldozer to destroy rabbit warrens (Watkins 2006). This investigation targeted five sites where the Contour Seeder was used (Sites 1 to 5), including Site 3 where the Camel Pitter was used in between Contour Seeder rip lines.

Macropod species (Northern Flinders Ranges Soil Conservation Board 1997; Lay 1998). In localised areas favoured by herbivores, degradation was more severe. By the late 20th century, these areas had transformed into bare (scalded) soil, in close proximity to erosion gullies, supporting a dominant understorey of invasive weeds (Watkins 2006). The few species persisting in the overstorey, included *Nitraria billardierei* (nitre-bush), *Acacia victoriae* ssp. *victoriae* (elegant wattle) (Sites 1–3, 5) and *Callitris glaucophylla* (white cypress-pine) (Site 4; Fig. 1).

Severely degraded sites within the park appeared resistant to passive recovery following the removal of stock in 1971 (National Parks and Wildlife Service 1983). The lack of recovery indicates the original vegetation community had declined beyond an abiotic threshold into a degraded state (as per the state and transition approach in Hobbs and Cramer 2008). As a result, active revegetation was attempted to aid the restoration of a *M. pyramidata* over *A. vesicaria* low shrubland (i.e. an intermediate chenopod shrubland community resilient to grazing).

### Condition states of M. pyramidata *over* A. vesicaria low shrubland

The five severely degraded sites targeted in this investigation may have originally supported a mixed chenopod shrubland. High and persistent grazing is likely to have removed the palatable saltbush and low bluebush, destabilising the top soil, and causing the mixed chenopod shrubland to transform into *Maireana pyramidata* (black bluebush) over *Atriplex vesicaria* (bladder saltbush) low shrubland (Lay and Tynan 2005). *M. pyramidata* over *A. vesicaria* low shrubland is relatively resilient to moderate grazing (Lay and Tynan 2005), but intensive grazing can cause

the condition of this community to decline (Cunningham *et al.* 1992; Lay and Tynan 2005). Traits of this community, in good to poor condition, include (Lay and Tynan 2005):

- Good condition: *M. pyramidata* are healthy and *A. vesicaria* are lightly grazed. Ephemeral species are diverse and abundant and *Dissocarpus paradoxus* (ball bindyi) is sparse. Any palatable overstorey species are regenerating.
- Fair condition: *M. pyramidata* are healthy, but *A. vesicaria* has died or been removed and *Atriplex stipitata* (bitter saltbush) may be present and healthy. *D. paradoxus* is the dominant ephemeral species. Regenerative growth on palatable shrubs is grazed.
- Poor condition: *M. pyramidata* and *A. stipitata* are heavily grazed, show no sign of regenerating and may be dead. *A. vesicaria* remains absent. Topsoil is lost. *N. billardierei*, *Maireana aphylla* (cotton-bush), *Sclerolaena* spp., annual saltbushes associated with soil scalding, and/or ephemeral species have increased in abundance.

Alternatively, mechanical intervention may encourage the natural colonisation of *Cymbopogon ambiguus* (lemon-grass) over *Aristida nitidula* (brush three-awn) tussock grassland (with or without emergent *C. glaucophylla*). This community is relatively common in the Flinders Ranges (Lay and Tynan 2005), and regularly colonises degraded patches in neighbouring communities (Gell and Bickford 1996). Over time, colonisation by sparse trees and shrubs may cause this community to transition into other communities (it is unknown whether this community is a persistent or temporary alternative state in the recovery of chenopod shrublands). In the meantime, intensive grazing can still cause a decline in the condition of this community, as indicated by the following traits (Lay and Tynan 2005):
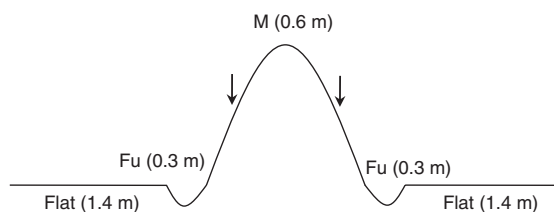
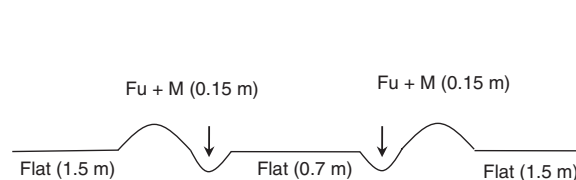**Fig. 2.** Cross-section of a (*a*) Contour Seeder rip line and (*b*) Camel Pitter rip line. Seeds were deposited (arrows) into the mound ('M'), or furrow ('Fu'), respectively. Wild, wind-blown seeds and surface water may also be trapped in the furrows. The flat zones received no direct seeding or mechanical disturbance, but had the potential for passive recovery over the course of this program. Photographer: P. Watkins.

Table 1.　Direct seeded and wild-sourced species detected during the survey and an opportunistic search of the local area

Most vouchers comprised sufficient material for a positive identification by botanists at the State Herbarium of South Australia. Nomenclature followed the 'Census of SA Plants, Algae and Fungi' (State Herbarium of South Australia 2015), which summarises the current scientific knowledge of the flora of South Australia, reviewed by taxonomic botanists and research associates of the State Herbarium of South Australia. For some species, one or more vouchers were tentatively identified[†], or adopted from positively identified vouchers with the same apparent morphology[††]. Species were classified as having an annual (A), short-lived perennial (SLP) or perennial (P) life history strategy (Peter Lang and Helen Vonow, pers. comm.), and plant phase as seedling (S), vegetative (V) and reproductive (R) or dead (D). Perennial species with over 100 plants recorded from seedling, vegetative and reproductive demographic phase with the potential for further analyses[‡]

Scientific name	Common name	Strategy	Phase	Contour Seeder Plants ha^{-1}	Cover (m²) ha^{-1}	No. quadrats	Camel Pitter Plants ha^{-1}	Cover (m²) ha^{-1}	No. quadrats	Local area
Direct seeded species						41/50			7/10	
Acacia victoriae ssp. *victoriae*	Elegant Wattle	P	S, V, R	358	123	22	13	4	1	X
Atriplex vesicaria	Bladder Saltbush	P‡	S, V, R, D	983	215	19	71	4	2	X
Maireana astrotricha	Low Bluebush	P	V	4	1	1	4	2	1	X
Maireana georgei/turbinata†	Satiny Bluebush	P‡	S, V, R, D	1192	113	21	163	4	4	X
Maireana pyramidata	Black Bluebush	P‡	S, V, R, D	1467	251	20	408	78	5	X
Senecio magnificus	Showy Groundsel	SLP	S, R, D	513	38	7	–	–	–	X
Vittadinia gracilis	Woolly New Holland Daisy	SLP	S, V, R, D	3808	27	14	433	1	1	X
Wild-sourced species						49/50			9/10	
Atriplex eardleyae	Eardley's Saltbush	SLP	V	17	<1	1	–	–	–	–
Atriplex nummularia ssp. *nummularia*†	Old-man Saltbush	P	V	4	<1	1	–	–	–	–
Atriplex sp.	Saltbush	P	V	4	<1	1	–	–	–	–
Atriplex stipitata	Bitter Saltbush	P‡	S, V, R, D	3042	81	26	304	5	4	X
Austrostipa sp.	Spear-grass	P	V	433	1	3	–	–	–	–
Boerhavia schomburgkiana	Schomburgk's Tar-vine	SLP	S, V, R	558	9	23	4	<1	1	–
Brachyscome ciliaris var. *lanuginosa*	Woolly Variable Daisy	SLP	V, R	117	1	3	–	–	–	–
Chenopodiaceae sp. 1††		A	S	13	<1	2	–	–	–	–
Chenopodiaceae sp. 2††		P	S, V	38	<1	2	–	–	–	–
Chenopodiaceae sp. 3††		P	S, D	38	1	1	–	–	–	–
Cymbopogon ambiguus	Lemon-grass	P	V, R, D	46	2	7	–	–	–	X
Dactyloctenium radulans	Button-grass	A	V, R	8	<1	2	–	–	–	–
Dissocarpus paradoxus	Ball Bindyi	SLP	S, V, R, D	7579	71	8	–	–	–	–
Echium plantagineum*	Salvation Jane	A	D	58	3	1	–	–	–	–
Einadia nutans ssp. *eremaea*	Dryland Climbing Saltbush	P	V	38	<1	3	–	–	–	–
Enchylaena tomentosa var. *tomentosa*†	Ruby Saltbush	P‡	S, V, R, D	488	102	13	8	1	1	X
Enneapogon avenaceus	Common Bottle-washers	SLP	V, R	458	1	8	–	–	–	–
Eremophila longifolia	Weeping Emubush	P	V	8	1	1	–	–	–	X
Eriochiton sclerolaenoides	Woolly-fruit Bluebush	SLP	V	42	<1	1	–	–	–	–
Gramineae sp. 1		P	V	1542	3	6	–	–	–	–
Gramineae sp. 2		P	V	13	<1	1	–	–	–	–
Leguminosae sp.		P	V	4	<1	1	–	–	–	–
Leiocarpa tomentosa	Woolly Plover-daisy	P	V	8	<1	1	–	–	–	–
Lycium australe	Australian Boxthorn	P	V	8	23	2	–	–	–	–
Maireana brevifolia	Short-leaf Bluebush	P	S, V, R, D	71	16	6	4	<1	1	X
Maireana integra	Entire-wing Bluebush	P	V	17	<1	2	4	<1	1	–
Maireana trichoptera	Hairy-fruit Bluebush	P	S, V	42	1	2	–	–	–	X
Malvaceae sp.		P	S, V, D	79	9	8	–	–	–	–
Marrubium vulgare*	Horehound	P	S, R, D	54	13	4	–	–	–	–
Osteocarpum spp.	Bonefruit	A	S, V, D	7596	52	27	3442	15	9	–

Ptilotus nobilis ssp. nobilis	Yellow-tails	SLP	S	1	<1	–	4	1	–	–
Ptilotus obovatus	Silver Mulla Mulla	P	S, V	1	1	4	8	1	<1	X
Rhagodia spinescens	Spiny Saltbush	P	V, R	6	12	–	63	–	–	–
Rytidosperma spp.	Wallaby-grass	P	S, V	2	<1	–	167	4	–	–
Salsola australis	Buckbush	A	S, V, R, D	23	10	108	671	4	<1	–
Sclerolaena brachyptera†	Short-wing Bindyi	SLP	S	2	<1	–	29	1	–	–
Sclerolaena diacantha	Grey Bindyi	SLP	S, V	1	<1	4	13	1	<1	–
Sclerolaena lanicuspis	Spinach Bindyi	SLP	S, V, R	2	1	4	29	1	<1	–
Sclerolaena limbata	Pearl Bindyi	SLP	S, R	3	<1	–	75	–	<1	–
Sclerolaena obliquicuspis	Oblique-spined Bindyi	SLP	D	–	<1	4	–	1	–	–
Sclerolaena patenticuspis†	Spear-fruit Bindyi	SLP	S, V, R, D	46	112	17271	43475	9	36	–
Senecio dolichocephalus	Woodland Groundsel	SLP	S, R, D	4	19	167	1117	1	–	–
Sida corrugata var. corrugata††	Corrugated Sida	P	S, V, D	9	2	–	1650	–	–	–
Sida fibulifera†	Pin Sida	P‡	S, V, R, D	19	51	46	5667	2	3	X
Sida intricata†	Twiggy Sida	P	V, D	1	2	–	217	–	–	X
Solanum ellipticum†	Velvet Potato-bush	P	S, V	2	<1	–	8	–	–	–
Solanum esuriale	Quena	P	S, R	–	–	275	–	1	<1	–
Solanum quadriloculatum†	Plains Nightshade	P	S, V	7	1	–	92	–	–	–
Solanum sp.	Nightshade/Potato-bush	P	D	1	1	–	4	–	–	–
Zygophyllum iodocarpum†	Violet Twinleaf	A	V	2	2	–	4	–	–	–

- Good condition: intact grassland with annual species present between tussocks. *Aristida nitidula* may occur. Any indicator species present are lightly grazed (e.g. *A. victoriae* ssp. *victoriae*). No soil movement.
- Fair condition: intact grassland with evidence of grazing on *C. ambiguus* and other palatable perennial grasses. Any palatable indicator or annual species are grazed. Signs of accelerating erosion with some areas showing recent colonisation by *C. ambiguus*.
- Poor condition: *Cymbopogon ambiguus* and *A. nitidula* are moderately to heavily grazed. Any palatable species are heavily grazed or absent (e.g. *A. victoriae* ssp. *victoriae*), including less palatable species (e.g. *Ptilotis obovatus*, silva mulla mulla). The soil surface is bare and unstable and actively gullying or rilling on susceptible soil types. Disturbed areas may show limited colonisation by *C. ambiguus*.

Revegetation method and seed mix

To restore the severely degraded areas, ripping and direct seeding were applied using two machines. Ripping using the Kimseed Linkage Rangeland Contour Seeder created a single notable mound, capable of redirecting surface water flow, with a shallow furrow on each side (Kimseed International Pty Ltd 2013a; Fig. 2a). The Contour Seeder rip lines were ~20–30 m in length, with a 5-m gap between consecutive lines, and 5–20-m gap between parallel lines (Watkins 2006). Pitting using the Kimseed Camel Pitter (Disc Revegetation Seeder) created two smaller mounds, each with a single, shallow furrow (Kimseed International Pty Ltd 2013b; Fig. 2b). In both cases, mechanical intervention (disturbance) formed the mound and furrow microtopographic zones, and seeds were directed into either the mound or furrow (Fig. 2). The neighbouring flat microtopographic zone was considered to be undisturbed.

Both machines were used in autumn (April) of 2004 to maximise germination (Knight *et al.* 1998), and allow plants to establish before summer (Watkins 2006). Seeds were selected to represent perennial species that occurred naturally in the area, and could be obtained from propagated plants, plants growing in the local area, or purchased (Watkins 2006). The direct seeded species included *A. victoriae* ssp. *victoriae*, *A. vesicaria*, *Maireana astrotricha* (low bluebush), *Maireana georgei* (satiny bluebush), *M. pyramidata*, *Senecio magnificus* (showy groundsel), *Vittadinia gracilis* (woolly new Holland daisy), *Acacia calamifolia* (wallowa), and *Eucalyptus socialis* ssp. (beaked red mallee). Although insufficient records were available to determine which combination of species were sown at each of the sites, all direct seeded species were present in the local area (Table 1).

The Contour Seeder was used at all sites, covering a total area of 3032.3 ha (Fig. 1). The Camel Pitter was only used at Site 3, in between Contour Seeder rip lines (979.6 ha; Fig. 1). The sites targeted in this survey were a maximum of 19 km apart and assumed to receive a similar rainfall. Annual rainfall before the millennium drought (1997–2009) averaged 321.1 ± 145.0 mm year^{-1}; on the wetter side of the range for semi-arid areas (between 250 and 350 mm; Fig. 3). After direct seeding, rainfall occurred over the subsequent 8 months, encouraging direct seeded and wild-sourced species to germinate, in particular *A. vesicaria*
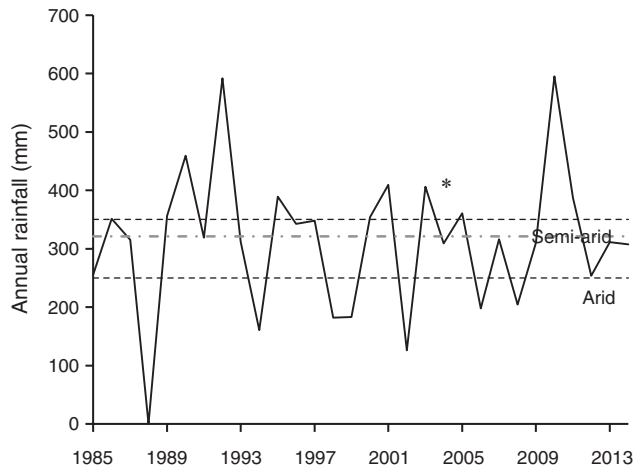
Fig. 3. Annual rainfall recorded at Oraparinna from 1985 to 2014 (Station number 19107; Bureau of Meteorology 2015). Shown is the average annual rainfall derived from between 1985 and 1996 (- · -), the 250-mm and 350-mm limits that define semi-arid Australia (- - -; Commonwealth of Australia 2014) and the year in which mechanical intervention and direct seeding were performed (star).

(Watkins 2006). Fewer plants appeared to germinate along Camel Pitter rip lines, and rabbit and kangaroo grazing was evident along Contour Seeder rip lines over summer after the perennial grasses had died back (Watkins 2006). Annual rainfall did not change between direct seeding in 2004 and the survey in 2014, but was slightly below average in 2004, 2006–2009, and 2012–2013, and below arid rainfall levels in 2006 and 2008 (Fig. 3).

Field survey

In April of 2014, at each of the five sites, 10 quadrats (4 m × 10 m) were centred over Contour Seeder rip lines and Camel Pitter rip lines (Fig. 2, Appendix 1). Quadrat locations were randomly determined using ArcGIS 9.3 (min. 30 m apart; ESRI 2009). The coordinates were then located in the field, and the nearest point along a rip line used to identify the start of a sampling quadrat. If the nearest point fell less than 10 m from the end of a rip line, the start of a sampling quadrat was moved along the rip line until the quadrat fitted. Rip lines near a track or less than 10 m in length were ignored.

General features recorded within each quadrat included the maximum height of the mound, cover of biocrust, and the abundance of herbivore tracks, diggings and scats. Specific features recorded for each plant included the species, maximum canopy diameter (to estimate plant cover), status (live or dead), demographic phase (seedling, vegetative, reproductive), signs of browse, and microtopographic position in the quadrat (flat, furrow, mound; Fig. 2), and any other notes (e.g. signs of resprouting). Annual weeds were generally not recorded (e.g. *Asphodelus fistulosus*, onion weed). Individual plants were distinguished based on the extent of continuous cover, and may represent multiple plants for small, abundant species (e.g. *Scleroleana* spp.). For each species, at each site, a voucher was collected for identification at the State Herbarium of South Australia (Table 1).

Statistical analyses

This investigation effectively applied a repeated-measures design, where quadrats were randomly spread throughout the study area, then split into microtopographic zones. The plant abundance and cover values were compared between zones using one-way repeated-measures analysis of variance in R (version 3.3.1; R Core Team 2016), with quadrat as the random effect. Where an analysis included three fixed treatments (e.g. disturbed mound, disturbed furrow, and undisturbed flat), *P*-values were checked against adjusted *P*-values using the Greenhouse–Geisser correction for non-sphericity (Quinn and Keough 2009). Where a significant difference was detected between microtopographic zones, a pairwise *t*-test using Bonferroni-adjusted *P*-values was applied to identify which zone(s) differed. To ensure the spatial clustering of quadrats within sites did not confound the analyses in each case, a linear mixed effects model with quadrats nested within sites as a random effect was compared against a model with the original model, using the package 'nlme' (Pinheiro *et al*. 2016). Where site improved performance (i.e. reduced the AIC value), this effect was successfully removed by excluding any sites dominated by quadrats where the species was absent (i.e. many zero values).

Analyses were performed for: (i) all direct seeded species, (ii) all wild-sourced species, (iii) all annual or short-lived perennial wild-sourced species, (iv) all perennial wild-sourced species, and (v) individual species detected in at least 19 of the 50 Contour Seeder rip line quadrats or four of the 10 Contour Seeder rip line quadrats. Abundance and cover values were standardised for a common sampling area per microtopographic zone (e.g. the 'flat' zone along Contour Seeder rip lines to 6 m^2) and square- or forth-root transformed before analysis. For perennial species with more than 100 live plants in seedling, vegetative, reproductive phases along Contour Seeder rip lines (Table 1), abundance and cover were compared between microtopographic zones, per demographic phase, using quadrats where the species was detected only. Recruitment between the combined 'mound' and 'furrow' zones of Contour Seeder and Camel Pitter rip lines were compared using the 10 paired quadrats sampled at Site 3. The abundance of macropod scats was compared between sites using quadrats sampling Contour Seeder rip lines.

Results

Recruitment of direct seeded species

Seven of the nine direct seeded species were detected in this investigation. *A. victoriae* ssp. *victoriae*, *A. vesicaria*, *M. georgei*, and *M. pyramidata* were moderately common, whereas *M. astrotricha*, *S. magnificus* and *V. gracilis* were less common (Table 1). Reproductively active plants and seedlings were observed at the time of survey for all direct seeded species, except *M. astrotricha*, which had a low frequency of occurrence, low abundance and was only present in the vegetative phase at the time of survey.

Direct seeded species had germinated, survived, and had recruited new plants across all three microtopographic zones along Contour Seeder rip lines (Fig. 4a). Plant abundance was greatest in the furrow ($F_{2,78} = 27.15$, $P < 0.001$). The trend
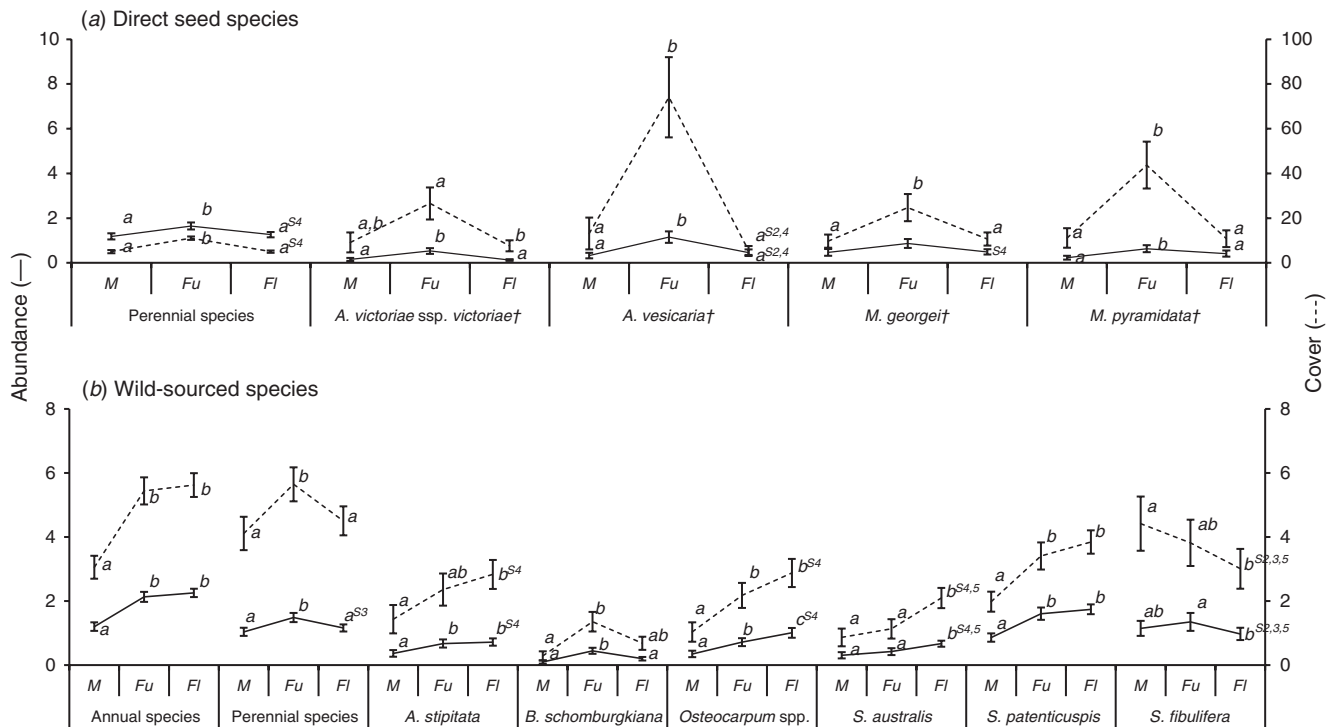
Fig. 4. Average abundance and cover (±s.e.) of (*a*) direct seeded species, and (*b*) wild-sourced species, surveyed along Contour Seeder rip line mound (M), furrow (Fu) and nearby flat (Fl) microtopographic zones. Values collected from the flat microtopographic zone (a larger portion of the quadrat) were standardised to 6 m². All values were forth-root transformed, except for all direct seeded species, which were square-root transformed (as indicated by †). Letters denote statistically different groups identified using a one-way repeated-measures analysis of variance and post hoc multiple comparisons using pairwise *t*-tests incorporating Bonferroni-adjusted *P*-values. Sites where species were predominantly absent from quadrats, supporting a random effect and potentially confounding analyses, were excluded as indicated (e.g. S4: Site 4).

for greater recruitment in the furrow ('perennial species trend') was supported by the abundance of several perennial species: *A. victoriae* ssp. *victoriae* ($F_{2,98} = 8.51$, $P < 0.001$), *A. vesicaria* ($F_{2,58} = 11.86$, $P < 0.001$), and *M. georgei* ($F_{2,78} = 3.89$, $P = 0.025$, although differences between zones were not supported by Bonferroni-adjusted *P*-values). However, *M. pyramidata* abundance was similar on the flat as per the 'annual species trend' (see below; $F_{2,98} = 7.23$, $P = 0.001$). This slight discrepancy disappears when looking at cover. The perennial species trend was supported by the combined cover of all perennial species ($F_{2,78} = 34.20$, $P < 0.001$; Fig. 4*a*), and perennial species: *A. victoriae* ssp. *victoriae* ($F_{2,98} = 5.50$, $P = 0.005$), *A. vesicaria* ($F_{2,58} = 13.16$, $P < 0.001$), *M. georgei* ($F_{2,98} = 6.79$, $P = 0.002$), and *M. pyramidata* ($F_{2,98} = 12.53$, $P < 0.001$; Fig. 4*a*).

Three direct seeded species were sufficiently abundant along the Contour Seeder rip lines sampled to investigate trends in demographic phases (i.e. were represented by more than 100 plants in vegetative, reproductive and seedling phases). The perennial species trend illustrated by all *A. vesicaria* plants above (Fig. 4*a*), appeared to be driven by the recruitment of reproductive plants (abundance: $F_{2,48} = 15.57$, $P < 0.001$; cover: $F_{2,48} = 17.07$, $P < 0.001$; Fig. 5*a*), vegetative plants (abundance: $F_{2,48} = 6.23$, $P = 0.005$, cover: $F_{2,48} = 6.02$, Adj. $P = 0.013$) and seedlings (abundance: $F_{2,48} = 3.57$, Adj. $P = 0.046$; Fig. 5*a*; cover: $F_{2,48} = 3.64$, Adj. $P = 0.045$, although differences between zones were not supported by Bonferroni-adjusted *P*-values for

the latter). The perennial trend illustrated by *M. georgei* cover (Fig. 4*a*), was not driven by any one demographic phase (Fig. 5*a*). The annual trend illustrated by *M. pyramidata* abundance (Fig. 4*a*), appeared to be driven by the abundance of vegetative plants ($F_{2,24} = 7.29$, $P = 0.002$). The perennial trend illustrated in the species' cover (Fig. 4*a*), was influenced by the cover of vegetative plants ($F_{2,24} = 9.15$, Adj. $P < 0.003$).

Fewer species were detected in the 10 quadrats sampled along Camel Pitter rip lines (Table 1). During the field survey it was also not possible to distinguish between the mound or furrow as the shallow furrows had mostly filled in (1.5 cm ± 0.4 cm (s.e.); max. 5 cm). Despite this, recruitment of direct seeded species was greater in the combined mound and furrow, than on the undisturbed flat (abundance: $F_{1,9} = 10.65$, $P < 0.010$; cover: $F_{1,9} = 16.93$, $P = 0.003$; Fig. 6*a*). This trend was supported by *M. pyramidata* (abundance: $F_{1,9} = 8.24$, $P = 0.019$; cover: $F_{1,9} = 8.72$, $P = 0.016$; Fig. 6*a*).

Recruitment of direct seeded plants was similar between Contour Seeder rip lines and Camel Pitter rip lines (Fig. 7*a*). Although had more quadrats been sampled, it is possible a significant difference may have been detected for greater recruitment of *M. georgei* along Contour Seeder rip lines than Camel Pitter rip lines (Fig. 7*a*). It should also be noted this particular analysis was performed on abundance and cover values standardised to 3 m². The actual area of the combined mound and furrow on a Contour Seeder rip line was four times greater for
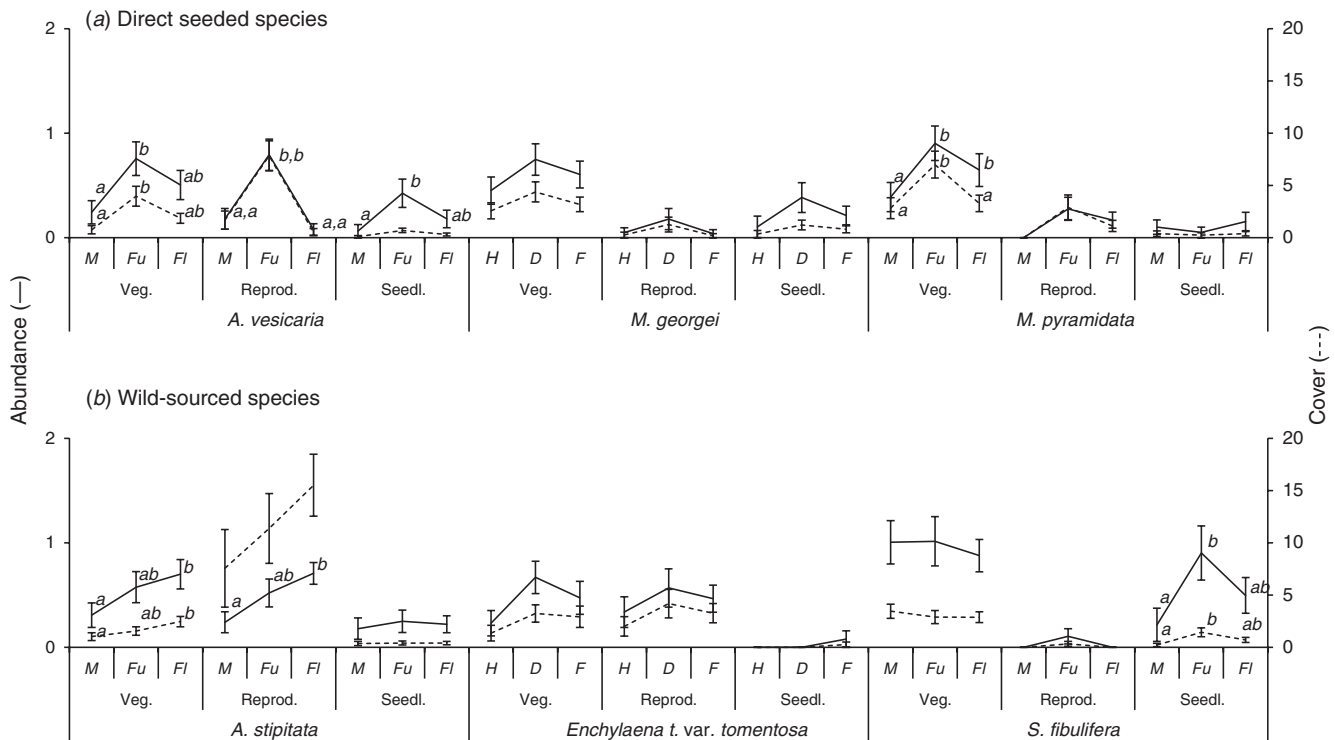
Fig. 5. The average abundance and cover (±s.e.) of (*a*) direct seeded or (*b*) wild-sourced abundant, perennial species surveyed along Contour Seeder rip line mound (M), furrow (Fu) and nearby flat (Fl) areas in vegetative, reproductive or seedling phase (live plants only). Values collected from the flat microtopographic zone (a larger portion of the quadrat) were standardised to 6 m². All values were forth-root transformed (except the cover of reproductive *Atriplex stipitata*, which was square-root transformed). Letters denote statistically different groups identified within demographic phases using one-way repeated-measures analysis of variance and post hoc multiple comparisons using pairwise *t*-tests with Bonferroni-adjusted *P*-values. Only quadrats where the species was present in any of the phases shown were included in the analyses.

a single run (Fig. 2), and is likely to have a greater logistical efficiency that is not considered here.

Opportunistic recruitment of wild-sourced species

An additional 50 wild-sourced species were detected, including a 2 : 3 ratio of annual and short-lived perennial species to perennial species (Table 1). Of these, 18 species were reproductively active and 26 species were recruiting seedlings at the time of survey (Table 1).

Wild-sourced species had colonised all microtopographic zones, in particular the furrow and flat (abundance: $F_{2,98} = 41.26$, $P < 0.001$; cover: $F_{2,98} = 16.30$, $P < 0.001$). This pattern appeared to be driven by annual and short-lived perennial species (i.e. 'annual species trend'; abundance: $F_{2,98} = 43.91$, $P < 0.001$; cover: $F_{2,98} = 33.79$, $P < 0.001$; Fig. 4b), rather than perennial species, which had greater recruitment in the furrow (i.e. perennial species trend; abundance: $F_{2,78} = 13.19$, $P < 0.001$; cover: $F_{2,98} = 6.24$, $P = 0.003$; Fig. 4b). The annual species trend was reflected in the abundance of four wild-sourced species, including the annual *Osteocarpum* spp. (abundance: $F_{2,78} = 14.66$, $P < 0.001$; cover: $F_{2,78} = 12.19$, $P < 0.001$), annual *S. australis* (abundance: $F_{2,58} = 5.46$, $P = 0.007$; cover: $F_{2,58} = 6.14$, $P = 0.004$), annual *S. patenticuspis* (abundance: $F_{2,98} = 20.48$, $P < 0.001$; cover: $F_{2,98} = 15.45$, $P < 0.001$), and perennial *A. stipitata* (abundance: $F_{2,78} = 7.71$, $P < 0.001$; cover:

$F_{2,78} = 5.74$, Adj. $P = 0.010$; Fig. 4b). The perennial species trend was supported by the short-lived perennial *B. schomburgkiana* (abundance: $F_{2,98} = 7.10$, $P = 0.001$; cover: $F_{2,98} = 5.93$, $P = 0.004$; Fig. 4b), and the abundance of perennial *S. fibulifera* ($F_{2,38} = 6.47$, Adj. $P = 0.012$). Unusually, *S. fibulifera* cover was greatest on the mound ($F_{2,38} = 4.65$, Adj. $P = 0.022$). As no individual species appeared to be singly driving the perennial pattern for greater cover in the furrow, the furrow appeared to be useful for a range of patchily distributed and uncommon perennial species.

Three perennial wild-sourced species were represented by more than 100 plants in vegetative, reproductive and seedling phases (Fig. 5b). The annual species trend illustrated by *A. stipitata* (Fig. 4b), was supported by vegetative plants (abundance: $F_{2,40} = 4.86$, $P = 0.012$; cover: $F_{2,40} = 5.20$, $P = 0.009$), and reproductive plants (abundance: $F_{2,40} = 7.28$, $P = 0.002$; Fig. 5b). The perennial species trend illustrated by *S. fubulifera* abundance appeared to be driven by the recruitment of seedlings (abundance: $F_{2,36} = 5.24$, $P = 0.010$; cover: $F_{2,36} = 4.09$, Adj. $P = 0.022$; Fig. 5b). It is likely the lack of discrimination between zones by established vegetative plants confounds the overall pattern in *S. fubulifera* cover (Fig. 4b).

Along Camel Pitter rip lines, the overall recruitment of wild-sourced species was greater in the combined mound and furrow (abundance: $F_{1,9} = 35.09$, $P < 0.001$, cover: $F_{1,9} = 50.78$, $P < 0.001$). This trend primarily reflected the abundance and
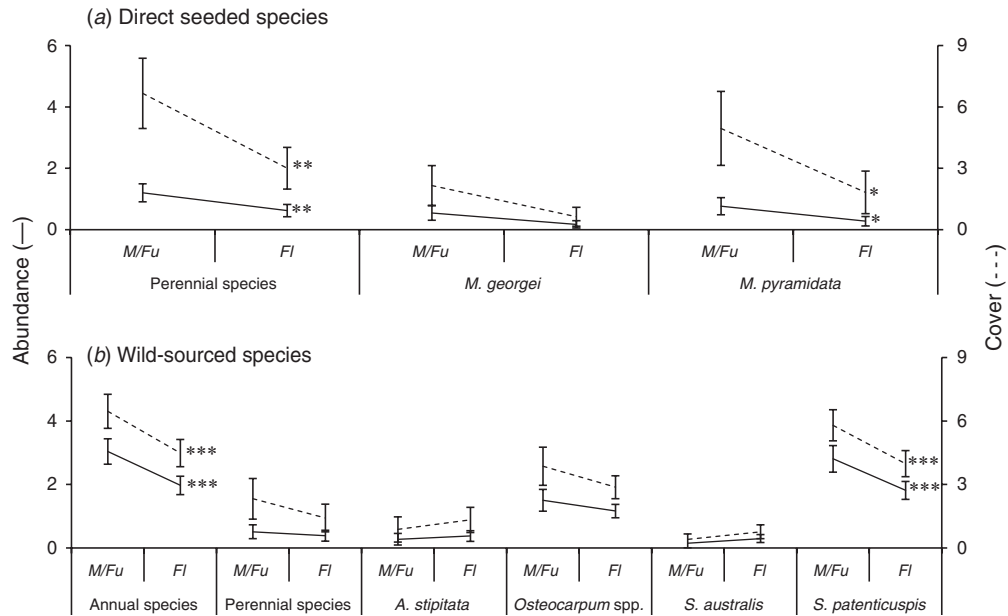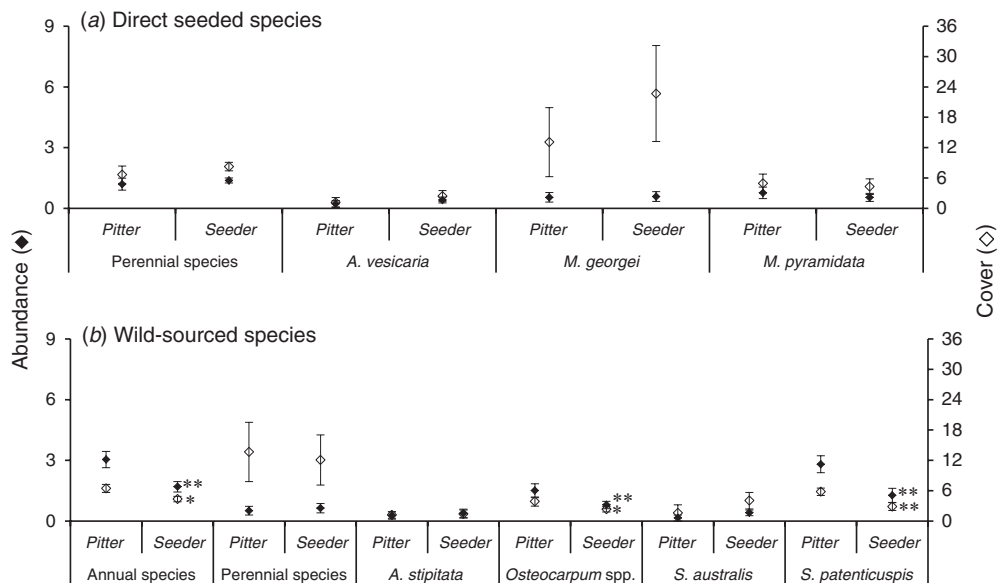
Fig. 6. Average abundance and cover (±s.e.) of (*a*) direct seeded or (*b*) wild-sourced species surveyed within combined mound and furrow (M/Fu) and nearby flat (Fl) microtopograhic zones along Camel Pitter rip lines (*n* = 10). Values collected from the flat microtopographic zone (a larger portion of the quadrat) were standardised for 3 m². All values were forth-root transformed. Statistically different groups were identified using a one-way repeated-measures analysis of variance and post hoc multiple comparisons using pairwise *t*-tests incorporating Bonferroni-adjusted *P*-values. Levels of significance indicated include: *$P<0.05$; **$P<0.01$; ***$P<0.001$.

Fig. 7. Average abundance and cover (±s.e.) of (*a*) direct seeded and (*b*) wild-sourced species surveyed within the combined mound and furrow along Camel Pitter and Contour Seeder rip lines (*n* = 10 paired quadrats). Values collected from the flat microtopographic zone (a larger portion of the quadrat) were standardised for 3 m². All values were forth-root transformed (except the cover of *M. georgei*, *S. australis*, and wild-sourced perennial plants, which were square-root transformed). Statistically different groups were identified using a one-way repeated-measures analysis of variance and post hoc multiple comparisons using pairwise *t*-tests incorporating Bonferroni-adjusted *P*-values. Levels of significance indicated include: *$P<0.05$; **$P<0.01$; ***$P<0.001$.

cover of annual and short-lived perennial species ($F_{1,9} = 34.48$, $P < 0.001$ and $F_{1,9} = 41.58$, $P < 0.001$, respectively; Fig. 6*b*). However, it is possible this trend may be driven by the relatively more widespread and abundant *S. patenticuspis* (abundance: $F_{1,9} = 27.33$, $P < 0.001$; cover: $F_{1,9} = 30.87$, $P < 0.001$; Fig. 6*b*).

For the species that were present in the 10 paired quadrats sampled along Camel Pitter and Contour Seeder rip lines at Site 3, wild-sourced species were more abundant in the combined mound and furrow of Camel Pitter rip lines ($F_{1,9} = 9.43$, $P = 0.013$; cover: $F_{1,9} = 4.24$, $P = 0.070$). This trend appears to reflect the abundance and cover of annual or short-lived perennial species ($F_{1,9} = 12.12$, $P = 0.007$ and $F_{1,9} = 6.25$, $P = 0.034$, respectively; Fig. 7*b*), in particular the more widespread and abundant annual species, *Osteocarpum* spp. ($F_{1,9} = 12.43$, $P = 0.006$ and $F_{1,9} = 8.91$, $P = 0.015$; Fig. 7*b*) and *S. patenticuspis* ($F_{1,9} = 14.54$, $P = 0.004$, $F_{1,9} = 17.46$, $P = 0.002$; Fig. 7*b*).

Current herbivore levels and signs of browsing

Signs of macropods were present in all quadrats sampled. Scats were significantly more abundant at Site 4 ($F_{4,45} = 23.80$, $P < 0.001$), which averaged 860.5 ± 31.2 (s.e.) scats per quadrat along Contour Seeder rip lines, compared with an average 139.6 ± 35.0 (s.e.) scats per quadrat at the remaining sites. Rabbit (*Oryctolagus cuniculus*) scats and diggings were present at 35 sites, including diggings in the sides of mounds of 19 of the 51 Contour Seeder rip lines sampled.

Herbivores had browsed five perennial grasses to butts, including *Austrostipa* sp., *E. avenaceus*, *Gramineae* sp. 1, *Gramineae* sp. 2 (100%), and *C. ambiguus* (27.3%). Heavily browsed shrubs, included perennial *M. georgei* (0.9%), *M. integra* (40%), *M. trichoptera* (60%), *S. fibulifera* (31%), *S. intricata* (100%) and *Leguminosae* sp. (100%), and annual *B. schomburgkiana* (0.7%), *B. ciliaris* var. *lanuginosa* (46.4%), *Osteocarpum* spp. (0.2%), and *S. lanicuspis* (25.0%). However, resprouting was also observed on several *E. avenaceus*, *M. georgei*, *S. fibulifera* and apparently dead *S. patenticuspis*.

Recovery of chenopod shrubland

The eastern degraded sites (Sites 1–3, and 5; Fig. 1) appeared to be in the early stages of *M. pyramidata* over *A. vesicaria* low shrubland recovery. Widespread and abundant *Maireana pyramidata*, *M. georgei* and *A. vesicaria*, active reproduction and seedling recruitment (Table 1), and the absence of browsing on these species, indicate the community was in good condition where it was present. Small patches of *M. pyramidata* over *A. vesicaria* low shrubland were present at approximately half of the quadrats sampled (i.e. recorded in 27/50 quadrats along Contour Seeder rip lines and 5/10 quadrats along Camel Pitter rip lines). Additional chenopod species that had colonised the area in moderate to low abundance, included palatable *E. tomentosa* var. *tomentosa*, *S. fibulifera* and *S. corrugata* var. *corrugata* and non-palatable *A. stipitata*. Annual chenopods of low palatability were most abundant (*S. patenticuspis*, *Osteocarpum* spp.). The annual *D. paradoxus* was also abundant, but was only detected at Site 5. The abundance of *A. stipitata*, *D. paradoxus* and *Sclerolaena* spp. indicate the community being in fair to poor condition. Other moderately

abundant annual and short-lived perennial species that had colonised some sites are associated with wetter environments (Cunningham *et al*. 1992), and included the less palatable daisies and groundsels: *V. gracilis*, *S. dolichocephalus* and *S. magnificus*. The palatable annuals and short-lived perennials *B. schomburgkiana*, *E. avenaceus* and *S. australis* had also colonised the area. Greater than 50% biocrust cover was recorded from an average of 4.5 ± 0.09 (s.e.) quadrats per site.

Cymbopogon ambiguus was rare across all sites, but an unidentified perennial grass butt (Gramineae sp. 1) and *Austrostipa* sp. was moderately abundant at Site 4, which may indicate limited recovery of an alternative, *C. ambiguus* over *A. nitidula* tussock grassland community, at this site. These plants were extensively browsed down to the butt (an indicator of the community being in poor condition). Additional species that were abundant at this site, included perennial *S. fibulifera* and *S. intricata*, and annuals and short-lived perennials *S. patenticuspis*, *E. avenaceus* and *B. schomburgkiana*. Greater than 50% biocrust cover was recorded in seven quadrats at Site 4.

Discussion

The benefits of mechanical intervention

In this investigation, furrows enhanced the recruitment of perennial species. This was supported by greater abundance and cover of direct seeded species and perennial wild-sourced species in the furrow. This trend was often supported by individual perennial species (*A. victoriae* ssp. *victoriae*, *A. vesicaria*, *M. georgei*, *M. pyramidata* and *S. fibulifera*), and occasionally short-lived perennial species (*B. schomburgkiana*). Seeds were also observed beneath *A. vesicaria* and *Maireana* spp. It is likely the collection of water and litter in furrows enhanced plant survival during dry periods.

The risk that soil disturbance may encourage colonisation by annual species (Knight *et al*. 1998; van den Berg and Kellner 2005; Gibson-Roy *et al*. 2010), was not supported by the native species considered in this investigation. The recruitment of annual and short-lived perennial wild-sourced species in the furrow was equal to, or less than, their recruitment on the flat along Contour Seeder rip lines. The trend was reflected by several species considered to be annual or semi-perennial in nature (*Osteocarpum* spp., *S. australis*, and *S. patenticuspis*), and a perennial species (*A. stipitata*). The lack of discrimination between the furrow and the undisturbed flat might be explained by annual species being adapted to germinate during suitable conditions in autumn or winter (Facelli *et al*. 2005).

The mound also appeared to deter recruitment of annual and short-lived perennial wild-sourced species. The short life cycle of annual and short-lived perennial species inhibited further investigation into differences in abundance and cover across microtopographic zones. However, further investigation into abundant perennial species suggests the mechanism is associated with increased mortality rates. This is supported by (i) a bias towards greater *S. fibulifera* seedling recruitment on the mound that is lost as plants mature, and (ii) an even recruitment of *A. stipitata* seedlings across the microtopographic zones that develops into a bias towards increased recruitment of vegetative plants on the flat. Although both species were wild-sourced,

it should be noted the low rate of direct seeded species recruitment on the mound is potentially significant given the seeds were originally funnelled into the side of the mound.

Soil properties can affect the persistence of microtopographic changes made using machines such as the Contour Seeder and Camel Pitter. Shallow furrows have been unsuccessful elsewhere (Tongway and Ludwig 2011; Wakelin-King 2011). In this investigation, the shallow furrows created by the Camel Pitter had mostly filled in. Within the combined furrow and mound, annual species were more abundant and had greater cover (Fig. 7b), while direct seeded species were similarly abundant when standardised for area. Although this implies there is no overall benefit to using a Contour Seeder over a Camel Pitter, it should be noted the Contour Seeder rip line was four times the area of the Camel Pitter rip line, and more efficient per length of rip line. The Contour Seeder rip lines have also persisted over the 10 years, averaging a maximum height of $12.9 \, \text{cm} \pm 0.7 \, \text{cm}$ (s.e.; max. 20 cm). These rip lines may continue to weather into the future, or given their age, they may stabilise.

Current status of chenopod shrubland recovery

Ten years after restoration via mechanical intervention and direct seeding, four of the five severely degraded (scalded) sites continue to support patchy recruitment of *Maireana pyramidata* over *A. vesicaria* low shrubland. Recruitment of *M. pyramidata*, *M. georgei* and *A. vesicaria* seedlings was evident (an indicator of 'good' condition; Lay and Tynan 2005). However, additional indicators of 'fair' and 'poor' condition (Lay and Tynan 2005), suggest the community is still in an early stage of recovery, including: heavy browsing on some palatable species, presence of *A. stipitata*, and diversity and abundance of annual saltbushes associated with soil scalding. At one site, chenopods present in the seed mix have failed, and the site been colonised by *C. ambiguus* over *A. nitidula* tussock grassland. The very low floristic composition and extensive signs of browse indicate this community was in 'poor' condition (Lay and Tynan 2005), and in a very early stage of recovery. In either community, it is unknown whether the current level of recruitment will be sufficient to sustain, or increase, plant diversity, abundance and cover. Elsewhere revegetated communities of a similar age have been reported to be unstable (consist of many pioneer species; Snyman 2003), and to have declined (Schneemann and McElhinny 2012).

Despite the uncertainty about the future persistence of the current revegetation, existing shrubs are expected to be acting as 'resource islands' and provide an important function for the landscape (Cortina *et al*. 2011). These resource islands have the potential to create seed and organic matter, support soil biota and invertebrates, attract soil-disturbing fauna, enhance the microclimate available for new plants, slow surface soil runoff, improve infiltration, increase soil water-holding capacity, trap soil, nutrients, leaf litter and seeds, and prolong seed viability (Hunt 2001; Stavi *et al*. 2009; Chaneton *et al*. 2010; Tongway and Ludwig 2011; Daryanto and Eldridge 2012; Farrell *et al*. 2012). These plants may also facilitate the recruitment of other species (Cortina *et al*. 2011). For example, both *A. victoriae* ssp. *victoriae* and *Cymbopogon ambiguous* have been observed to

provide a safe environment for seedling germination and growth (B. McIntosh 2013, Gum Creek Station, pers. comm.). While interactions between species can be facilitative or competitive (e.g. through deterring seedling recruitment; Facelli and Springbett 2009; Cortina *et al*. 2011), in the restoration of semi-arid environments, established plants are likely to have an overall positive effect (Facelli and Springbett 2009). So in future, recruitment may continue to increase the vegetative cover in the furrow (currently with $48 \pm 8\%$ s.e. cover along Contour Seeder rip lines), mound ($11 \pm 2\%$ s.e. cover) and flat microtopographies ($10 \pm 2\%$ s.e. cover).

Recommendations for continued recovery

The continued recovery of *M. pyramidata* over *A. vesicaria* low shrubland may depend on reducing macropod and rabbit densities (Murphy 2014). Macropods and rabbits were present while the revegetation became established, but signs of their impact were clear at the time of this investigation. Signs of browse were recorded on 13 of the 57 species, and were prevalent or common on many species. This herbivory has influenced community composition by elevating recruitment of *M. pyramidata* and depressing recruitment of *A. vesicaria* (as per the species' known response to grazing; Facelli and Springbett 2009), relative to densities recorded elsewhere (Wilson *et al*. 1987; Eldridge *et al*. 1990). Reducing herbivory is likely to enhance the recruitment of palatable perennial grasses and chenopod species. For example, reduced herbivory on *A. vesicaria* has been documented to improve the persistence of established plants through drought (Read 2004), increase seed production and storage in the seed bank (Hunt 2001), and protect resource traps, or safe sites, for seed germination (e.g. perennial shrubs, depressions and patches of leaf litter; Hunt 2001; Kinloch and Friedel 2005). Additional actions that may support the ongoing revegetation of these degraded sites include continuing cultivating the soil to improve infiltration (Snyman 2003), or use piles of branches to moderate the micro-climate (Cortina *et al*. 2011).

Acknowledgements

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Appendix 1.   A selection of photos showing the alignment of quadrats along Contour Seeder (left) and Camel Pitter (right) rip lines

Quadrats were centred over the rip line and 10 m long and 4 m wide. Each plant was recorded as being within the mound, furrow or flat microtopographic zone