Assessing biodiversity outcomes from waterpoint interventions in the patchy, gibber-gilgai arid rangelands

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1. PROJECT TITLE

Assessing biodiversity outcomes from waterpoint management in the arid rangelands.

2. INTRODUCTION

Domestic and feral herbivores need daily access to water during summer, and every few days during winter. The risk to biodiversity and ecosystem function depends on the type of herbivore activity, its intensity, and how long an area is exposed to grazing and seasonal conditions (rainfall). We explore whether waterpoint manipulation is a useful management tool for achieving biodiversity and ecosystem outcomes in the arid grazing lands of remote Australia. We used the 'gibber gilgai' systems of the Stony Plains Bioregion in northern South Australia for our case study.

3. PROJECT OBJECTIVES

The project objectives listed in the original schedule in April 2006 were revised in the first milestone report (Smyth 2007) in response two types of feedback: (i) outcomes of project planning meetings with the project team in the early stages of our work (Activity 1), and (ii) new information gained from other projects after the proposal of this work was submitted in 2005/06 (Smyth *et al.* 2007). The reasons for the revisions related to scientific and logistic challenges of field research conducted in extremely hot and remote locations on working cattle properties that were isolated from research centres (Appendix A). The revised objectives were:

1. Review early project proposal and define the desired biodiversity outcomes from waterpoint management in the case study region (Problem definition).

2. Develop indices to measure biodiversity condition and to assess improvements from changes in waterpoint management (Metric design).

3. Undertake calibration and intervention experiments to understand the biological responses to rainfall seasonality, soil condition and current grazing pressure (Waterpoint field experiments)

4. Investigate the influence of rainfall seasonality on native vegetation under controlled conditions (Rainfall seasonality experiments)

5. Assess the capacity of remote sensing techniques to provide a broader context to field vegetation data, using the correlation between spectral signature and cover (Spatio-temporal analysis of field vegetation data).

4. SUMMARY OF METHODS

Five activities were undertaken: (1) problem definition, (2) metric design, (3) waterpoint field experiments, (4) rainfall seasonality experiments and (5) spatio-temporal context of field vegetation data. Methods involved a technical workshop, two desk-top studies, a field study and glasshouse trials. Modifications to the methods in the original schedule via variations are highlighted where appropriate.

4.1 Problem definition (Activity 1)

These methods resolved project planning issues in the early stages of our work and tightened the focus of our project, making it achievable. Our project had three major dependencies at the outset influencing the metric design and both the waterpoint and rainfall seasonality experimental components.

Progress on the ‘biodiversity condition and outcomes’ metric design was stalled until we could obtain a clear statement from government environmental agencies as to what biodiversity outcomes from water management they expected in the arid rangelands and which biological attributes were best to measure condition and management outcomes. The major issue with the waterpoint experiments was finding land managers willing to manipulate their grazing management for the life of the field experiments (approximately 30 months). The other major issue was about the logistics of running *in situ* rainfall seasonality experiments.

The methods used in refining the Problem Definition were: (a) hold a technical meeting with key representative SA government agencies to define desired outcomes of waterpoint management and identify potential measures of biodiversity condition and management outcomes, (b) seek expressions of interest (EOI) to participate in our project, and (c) follow-up EOIs with a field visit to assess logistics and waterpoint and ‘rainfall seasonality’ experimental design options and (d) to obtain long-term formal agreement. In the technical meeting, the desired outcomes and biodiversity measures were obtained in seven steps: (1) define biodiversity assets and values, (2) identify threats and pressures, (3) prioritise key threats and pressures, (4) define desired outcomes, (5) identify surrogates and biological and management indicators for metrics, and (6) select robust indicators by applying selection criteria.

All of these methods were part of the developmental phase of the project schedule. No modifications were necessary.

4.2 Metric design (Activity 2)

Another major issue affecting metric design was the merit of environmental metrics as a tool for assessing ‘biodiversity condition and outcomes’ in biological terms. Most metrics in Australia have been designed for assessing improvements in management outcomes for stewardships in terms of ecological economics. In doing so, it is assumed, with small modifications, that these are easily transferrable for monitoring and assessing changes in any biological phenomena in different places over time. As a consequence, the metric design was carried out in three steps: (1) critically review the scientific literature and environmental agency reports to assess the merit of environmental indices for assessing biodiversity in Australia’s rangelands, (2) develop ‘biodiversity condition’ and ‘outcome’ (performance) metrics, using the biodiversity measures selected in Activity 1 and the lessons learnt from the review, (3) validate metrics using data from waterpoint experiments (Activity 3).

All of these methods are more focussed then those articulated in the original schedule and continue to meet the original objectives. No modifications were necessary.

4.3 Waterpoint field experiments (Activity 3)

Study area and sites

We chose the Stony Plains Bioregion (IBRA) as a case study to understand biological responses to sustained cattle grazing pressure and soil condition. Field sites were located on two cattle properties and the Oodnadatta Town Common in the Oodnadatta saltbush (*Atriplex nummularia* ssp. *omissa*) – mitchel grass (*Astrebla pectinata*)-neverfail (*Eragrostis setifolia*) low sparse shrubland/sparse grassland community as it is the most widespread and productive for livestock production. Due to its aridity, cattle grazing is mostly centred on artificial waterpoints (troughs from bores and rain-fed dams) within only approximately 4 to 5 km from water. After significant rains, the gilgais act as natural water sources, which allow cattle to disperse across the landscape away from permanent sources, thus, areas remote from permanent water sources experience pulses of grazing depending on the amount and spatial patterning of rain.

Experimental design and sampling

To measure the effects of waterpoint intervention, the experimental design comprised two treatments with each having two levels: grazing exposure (sustained, relaxed) and present grazing intensity (relatively heavily grazed, lightly grazed to negligible). The main experiment was waterpoint interventions but we also set-up cross-fence experiments so we could confirm the intervention results. Our intent was not to duplicate past paired grazing studies using cross-fence comparisons but to establish the patterns in the gibber-gilgai biome for the first time.

The waterpoint intervention experiments followed a ‘beyond BACI (before-after, control-impact) design where the ‘before’ experiments surveyed responses to sustained grazing of about 80-100 head of cattle per waterpoint (control). The ‘after’ experiments tracked changes in biological responses after the intervention (treatment). The intervention involved stock removal off the waterpoints. Responses were surveyed four times, twice during each of ‘typical’ rains and extreme dry periods. Vascular flora (in plots), ground-dwelling mammals and reptiles were sampled in a total of 40 sites for the waterpoint interventions.

The design of the calibration experiments were cross-fence comparisons of grazing exposure (sustained, relaxed from stocking for > 15years) relative two waterpoints with grazing intensity nested within each level. This design was replicated for a total of 40 sites. Sites were surveyed once after ‘typical’ rains and once after a long dry period.

Soil condition was not sampled as it appeared to be comparatively well structured probably because of the underlying landscape processes. Nevertheless, we undertook some pilot samples of low and high intensity sites of the controls to assess soil biological resilience. The measures used were soil microbial activity and catabolic profiling. We also undertook resident, ground bird surveys but densities were so low (~1 individual per 4 km traversed) at the spatial scale of other experiments that a new study would be needed to assess impacts.

4.4 Rainfall seasonality and soil seedbank experiments (Activity 4)

It was apparent after Activity 1 that it would not be practical to undertake *in situ* experiments in the field to examine vegetation response to summer and winter rains. Instead, we conducted a study comparing the species composition and abundance of germinable soil seedbank under winter (5-25°C & short day length) and summer (15-35°C & long day length) glasshouse conditions (with all other factors held constant). Soil seedbank results were also compared with similar attributes measured *in situ* for standing herbage in the 40 vegetation plots as described for the calibration experiments in Activity 3. For the glasshouse trials, eleven bulked soil cores were sampled for each site and placed with sterile potting mix in trays in a temperature-regulated, 'automatically watered' glasshouse under natural day length conditions. Plant species richness and density were determined from seedlings and for 'above ground' herbage, the measures were species richness and foliar cover. These measures were determined for a variety of longevity/lifeform and palatability classes. Also determined were litter cover; and the density, height and average canopy area of *Atriplex nummularia* ssp. *omissa* individuals. We chose this species because it was the only common, widespread long-lived perennial species at all replicates in the study area. Canonical correspondence analysis was used to depict multi-dimensional patterns by treatments and linear mixed-effect models to examine differences between treatments.

4.5 Spatio-temporal context of field vegetation data (Activity 5)

Analysis of satellite imagery can provide additional spatial and temporal context to field-based measurement. Two separate remote-sensing analyses were conducted for this waterpoint study.

1. Very high spatial resolution images were analysed to better understand spatial characteristics of gilgais for areas within and beyond the field-study sites.

2. Historic Landsat images were analysed to build a profile of change in vegetation cover across a range of seasonal conditions.

In the first study, Ikonos and Quickbird images (~1-m spatial resolution) were classified to map gilgai structures. Descriptive statistics (mean gilgai size, nearest-neighbour distance, gilgai density etc) were then used to build a 'typology' of gilgai types for each study site. Analysis of cover trends between 1974 and 2006 based on Landsat data (25 to 50-m pixel resolution) was conducted to assess change in response to seasonal conditions.

5. SUMMARY OF RESULTS

5.1 Biodiversity condition assessment framework

Results of the technical workshop revealed that the desired biodiversity outcomes from waterpoint management were integral to broader aspirations for biodiversity of the Stony Plains Bioregion. Based on a priority setting, four broad biodiversity management outcomes were identified: (1) native vegetation maintenance or restoration, (2) no species, population or community losses; (3) natural water flows, and (4) natural mosaic of water remoteness. The latter directly relates to waterpoint management but waterpoint management also influences the other outcomes via total grazing pressure by livestock, feral herbivores (donkeys, camels) and kangaroos. It was revealed in the workshop and the metric review that despite considerable work being done on measuring rangeland condition, it had yet to include assessments of biodiversity (plants, animals and microbes) as a primary focus. Rangeland managers struggle with assessing it because it is rarely defined and difficult to pin down as biodiversity is everywhere (so what do you assess), is always changing in response to natural (and unnatural) disturbances (so how do you know when it has changed and what was the trigger) and what amount of change signals changes in management action is perplexing.

We first defined biodiversity and then developed the first framework for assessing biodiversity condition in the arid rangelands. We applied it to select indicators that were scientifically defensible in biological and planning terms for assessing biodiversity. We defined 'biodiversity condition' in terms of differences between trends in biodiversity indicators for relatively unmodified (reference) and modified ecological systems of the same type monitored for the same time period, where biodiversity is the composition, structure and function measured at a range of biotic scales (genes, species, ecological communities). We were not able to illustrate the interpretation of condition because of the absence of long-term biodiversity monitoring data in Australian rangelands. We do however provide guiding principles about sampling design and analytical methods for interpretation. They are based on raw data rather than summaries of data produced by (multi)metrics. We discovered that: (i) the choice of biodiversity surrogates and indicators were driven by the choice of management outcomes, and (ii) a number indicators were not robust when assessed on biological relevance, measurement qualities, feasibility of implementation, and policy and management relevance for the three different waterpoint-related management outcomes.

Our work highlights the importance of stating the expected outcomes of biodiversity condition assessments up front, so that indicators relevant to future management outcomes are chosen. It also shows that critical thought on the robustness of indicators is warranted, especially as condition assessments under climate change will require information on the functional traits of species. This is the first framework of its type and although it has strengths, it also has weaknesses in relation to environmental planning and should be used as one of many tools in the biodiversity management toolbox.

5.2 Metrics in biodiversity management – an ecological perspective

Metrics most commonly applied in the management of Australia's biodiversity are through stewardships. Although US economists pioneered the work on environmental metrics in 1985, Australia developed and applied them independently as early as 2001 with the inaugural *BushTender*. Metrics when used in biodiversity stewardships are a standard of measurement used to objectively classify and rank multi-objective landholder proposals for the purpose of allocating stewardship contracts. Structurally, they combine information on the existing biodiversity condition and its potential improvement or "benefits" under changed management for a number of biodiversity surrogates assessed at the site, landscape and regional scales. Information on biodiversity condition is assessed by indicators, which are justifiable more in planning than ecological terms. They have great appeal among NRM practitioners as they distil the condition and restorative capacity of ecological systems into a single number. There is mounting evidence by ecologists though to suggest they may mislead or at worst deceive management. This is because (multi) metrics are only as rigorous as the indicators on which they are based. In our review, we concluded that: (1) It is too early to test the credibility of biodiversity metrics in Australia as they have not been in placed long enough; consequently serious attention needs to be given to their design. (2) Rarely are biodiversity indicators explicitly justified in terms of ecosystem functionality (biological properties, their goods and services that benefit humans) and resilience (to measure recovery). (3) Indicators are often selected on theoretical and anecdotal grounds rather than empirical ones. (4) Information for indicators is combined into a single number and obscures useful information for decision-making and hence compromises the transparency in their application. For these reasons, we decided not to pursue the development of metrics but to focus on the selection of scientifically defensible indicators in terms of ecology and biodiversity planning as described above and leave interpretation to the end-users.

5.3 Waterpoint field experiments

Standing herbage (one sample) and seedbank

Gilgais containing sparse-shrublands dominated by *Atriplex nummularia* ssp. *omissa* are common on cattle pastoral leases in the Stony Plains Bioregions of northern South Australia and contain a diverse range of ephemeral-wetland plant species. The present study compared the germinable soil seedbank of gilgais in relatively heavily grazed areas near waterpoints with others in lightly grazed areas far from waterpoints, all of which were nested in areas that were consistently stocked with cattle and areas relaxed from grazing for more than 15 years. The community was found to be relatively resistant to grazing pressure with no evidence of significant decline in soil seedbank species richness, even near long-established waterpoints (since 1890s). We believe this is due to three interacting phenomena: (i) the predominance of short-lived species which are able to complete their life cycle rapidly after significant rainfall events, (ii) the ability of many gilgais to hold water for a number of weeks, thereby enabling herbivores to disperse widely, and (iii) leading to reduced grazing pressure around permanent watering points at such critical times. The relatively unpalatable dominant shrub *A. nummularia* also appears to provide protection to “highly palatable” species. The use of individual plant species as indicators of gilgai range condition is problematic due to the absence of dominant, palatable long-lived perennial species. The only widespread species identified as a potential indicator of prolonged grazing pressure was *Rhodanthe stricta,* which is a short-lived ephemeral. Significant correlations between combined above and below-ground total species richness, and two above-ground attributes (viz. numbers of “highly palatable” species, and numbers of longer lived perennial species) suggest that these attributes can be used to rank gilgai sites according to species richness even during periods of below-average rainfall. However, this probably only applies to sites which are geographically close and have experienced the same recent rainfall history.

Fauna

Preliminary vertebrate fauna results indicate two patterns. There were no significant differences in the fauna community composition between sites and current levels of grazing intensity within our sites did not consistently affect species richness and relative abundance. At this level of biotic organisation, there does not appear to be a grazing intensity effect as measured by two levels of grazing (relatively heavily grazed and lightly grazed to negligible). However, for individual species, there were some preliminary significant trends within the reptiles. Two species were more abundant in lightly grazed sites whilst another was more abundant in relatively heavily grazed sites close to waterpoints. Further analysis of these responses will be conducted following the final field sample.

Preliminary analyses of pre and post intervention at the Macumba sites indicated that there were statistically significant responses in both mammal and reptile productivity following grazing relaxation. When these were combined for the total animal productivity, average abundance showed a significant increase in fauna following resting from grazing in the more heavily grazed sites close to waterpoints. This indicates an interaction between grazing exposure and intensity resembling a ‘pulse’-like response to grazing relaxation (following Underwood and Chapman 2003). Fauna declined in average abundance compared with the first sample in sites where grazing was maintained. Mammals responded with a significant increase in abundance following cattle removal in sites both near and far from waterpoints, suggesting that grazing exposure may play a greater role than intensity in affecting mammal numbers. As semi-permanent water during the hottest months becomes available throughout the landscape, grazing pressure is released near artificial waterpoints but spreads unevenly throughout the landscape. Alternatively, mammals could be reflecting responses to vegetation productivity as the summer rainfall was relatively higher in the post-intervention sites. In the pre-intervention sites, rainfall occurred in autumn, favouring ephemeral recruitment (Davies *et al.* manuscript). Summer rains favour grass recruitment and the abundance of seeds may have increased granivorous/omnivorous mammal abundance. Where there was no relaxation of grazing, reptiles responded with a significant decrease in abundance post intervention at both near and far sites. There was also a less marked and non-significant decrease in the areas following reduced grazing pressure. It is likely that the trend for reduced reptile productivity relates to the differences in timing of the sampling (late spring for pre-intervention and early autumn for post-intervention). Reptiles are generally more active in spring compared to autumn, as more mobile adults in the population are replaced by less mobile hatchlings. The results could therefore be interpreted as relaxation of grazing, having a positive effect on reptile productivity when compared with sustained grazing. This is because, despite post-intervention sampling occurring when productivity would be seasonally lower, the replicates in the post-intervention areas were as productive as during the pre-intervention sample when reptile productivity was at its peak. In the sustained grazing areas, productivity dropped significantly more than in the areas relaxed from grazing.

In summary, there is a suggestion that grazing relaxation may benefit the fauna at least but it in conclusive and requires completion of our field work.

5.4 Rainfall seasonality

Overall it was found that of all seed germinating, over twice as many (69%) germinated under “winter” conditions than under “summer” conditions (31%). This was due to the majority (78%) of forb seedlings germinating in winter, as did 64% of seedlings of long-lived perennial shrubs. Ninety eight percent of all exotic seed, which germinated did so under “winter” conditions as did 66% of seedlings of “highly palatable” species, and 66% for “unpalatable” species. However, 81% of grass seed, which germinated emerged under “summer” conditions.

5.5 Spatio-temporal analysis of field vegetation data

Analysis of satellite imagery can provide additional spatial and temporal context to field-based measurement. Two separate remote-sensing analyses were conducted for this waterpoint study.

1. Very high spatial resolution images were analysed to better understand spatial characteristics of gilgais for areas within and beyond the field-study sites.
2. Historic Landsat images were analysed to build a profile of change in vegetation cover across a range of seasonal conditions.

In the first study, Ikonos and Quickbird images (~1-m spatial resolution) were classified to map gilgai structures. Descriptive statistics (mean gilgai size, nearest-neighbour distance, gilgai density etc) were then used to build a ‘typology’ of gilgai types for each study site. Difficulties in precisely classifying gilgais and verifying results on the ground prevented accurate maps of gilgai shape and location. Thus the spatial statistics are only indicative. Nevertheless, there were considerable differences in gilgai-type between sites (e.g. size and separation) and these differences need to be accounted for when interpreting field data.

Analysis of cover trends based on Landsat data (25 to 50-m pixel resolution) showed that cover remained relatively stable between 1974 and 1994 and was much more variable over the next 10 years. It is likely that seasonal variation in rainfall accounted for most of the cover change. This historical pattern provides important context for interpreting vegetation data collected on the ground.

6. ADOPTION AND COMMUNICATIONS

6.1 Adoption

Although it is still early days for policy adoption, the biodiversity condition assessment framework has been modified and used as an example framework for monitoring and evaluating biodiversity changes in response to feral goat control by the Department for Environment and Heritage, SA. Smyth *et al.* (2009) has received 3 reprint requests at this stage.

6.2 Communication

Presentations to stakeholders – 4
Newsletter articles – 5
Internet weblinks – CSIRO project web address linked with Desert Channels and Lake Eyre Basin.
Publications sent to: ACRIS (Jenny Boshier) and all project collaborators and some of their colleagues.
Field Day has not occurred but we are in the early stages of planning a Landline grab.

7. COMMERCIAL POTENTIAL

The knowledge collected, derived or synthesized during this project has no commercial potential. Most information will be published in the scientific literature for the general readership.

8. PUBLICATIONS

Bastin, G. (2009). Spatio-temporal analysis of vegetation cover of gilgai ecosystems. Report to Land and Water Australia. (Attached)

Brandle, R. and Smyth, A. (2009). Preliminary report of vertebrate fauna analyses resulting from the waterpoint manipulation for biodiversity project. Report to Land and Water Australia. (Attached)

Davies, R.J-P, Mackay, D. A., Whalan, M. A., Smyth, A.K. (2009). A comparison of above-ground vegetation and soil seedbank in the ‘gibber gilgais’ of the arid rangelands of central Australia**.** (Manuscript)(Attached)

Davies, R.J-P, Mackay, D. A., Whalan, M. A. (2008). A comparison of above-ground vegetation and soil seedbank in the gilgais of the arid rangelands of central Australia**.** Report to the Native Vegetation Council of South Australia by Flinders University. (Attached)

Smyth, A.K., Brandle, R., Chewings, V., Read, J., Brook, A., Fleming, M. (2009). A framework for assessing regional biodiversity condition under the changing climates of the arid rangelands. **The Rangelands Journal** 31:87-101. (Attached)

Davies, R.J-P., Smyth, A. K. (2008). A review of metric design applied in stewardship management of Australia's biodiversity – an ecological perspective. (Manuscript being revised) (Attached)

In preparation:

Brandle, R., Smyth, A. Predicting the vulnerability of small ground mammals to cattle grazing in Australia's arid gibber-gilgai rangelands.

Smyth, A.K., Brandle, R., Davies, R.J-P, Cook, G., Dawes, T., Foulkes, J. Biodiversity responses to waterpoint invention in the arid gibber-gilgai rangelands.

9. ADDITIONAL INFORMATION

Additional information can be found at:
http://www.csiro.au/science/WaterpointManagement.html

APPENDIX A – TECHNICAL REPORT

Assessing biodiversity outcomes from waterpoint management in the gibber-gilgai grazing lands of arid Australia

INTRODUCTION

Artificial waterpoints control the distribution of grazing animals both spatially and temporally in the arid rangelands (Lange 1969; Squires 1974; Low *et al.* 1978). Many scientific studies have addressed the responses of plants and animals to water-focussed grazing gradients with some recommending waterpoint closures to rest biodiversity from grazing pressure (e.g. Noble *et al.* 1998; James *et al.* 1999). However, questions still arise among environmental planners as to whether the benefits to biodiversity outweigh the production losses and lost opportunities costs that waterpoint closures imposed on producers. An implicit assumption of waterpoint closure is that that grazing relaxation can stimulate inherent ecosystem resilience and lessen ‘persistent grazing gradients’. Assuming degraded ecosystems can bounce back from decades of grazing, how then can change be reliably measured in the highly, changeable arid rangelands? In other words, how do we assess and interpret biodiversity gains from waterpoint interventions in patchy dynamic systems? Presently, there are no strategic planning tools for waterpoint management to answer these questions in a transparent way. Our work addresses this issue for cattle grazing in the gibber-gilgai biome of Stony Desert bioregion of South Australia.

In arid Australia, rainfall is discontinuous and comes in “pulses” of very short duration relative to the long-term absence of rain between rain events. An effective rain event activates biological processes (especially production and reproduction) and the biomass of plants and animals build up in response (following Noy-Meir 1973). It is not only discontinuous but it is also stochastic in amount, intensity and timing of rainfall. Generally, between-year variability of rain events increases with lower average rainfall. On a monthly basis, rainfall is mostly aseasonal and falls sequentially for only one or two days at a time on average. Imposed on the temporal patchiness of rainfall is the spatial patchiness. The latter is persistent and random at all scales, depending on topography (influencing run-off), wind direction and speed, and rain angle. Erratic thunderstorm cells play an important role in biological responses. Coupled with rainfall seasonality is the variability in the structure and function of arid rangelands systems in space and time (Friedel 1990). Consequently, any assessment of biodiversity outcomes has an extremely stochastic quality, making it potentially difficult to assess in a repeated fashion and to tease apart changes due to rainfall seasonality and human interventions.

Rainfall via runoff also recharges natural drinking sources, thereby increasing access to forage over much larger areas. When conditions are good, cattle tend to disperse away from permanent water sources. During long dry periods, they rely on artificial waterpoints and sparse permanent natural water sources (Fleming 1998). Therefore, waterpoint management requires an intimate knowledge of the rainfall seasonality, the patchiness of nutritious forage, livestock movements and livestock defoliation activity (dietary preferences and amount consumed).

Cattle tend to have a less selective diet than kangaroos and probably the least selective of all dominant herbivores in the Australia rangelands (Fensham & Fairfax 2008). They are likely to switch preferences depending on food availability but preferentially graze on forbs after rain (Dawson & Ellis 1994). Overall, cattle have a low defoliation impact due to their reluctance to forage pastures to the same level as other herbivores because of their smaller bite (Graetz 1980). Grazing animals tend to favour the habitats that provide the most nutritious forage (Hunt *et al.* 2007). Cattle tend to have a greater threshold area (area occupied by 95% of the population relative to water-focussed grazing) under dry than wet conditions (Fensham & Fairfax 2008). Grazing is also more likely to be constrained around waterpoints with fertile substrates in arid systems(Smith *et al.* 2007).

The vulnerability of areas has been defined in terms of one or more threatening processes or human disturbances that puts at risk "the survival, abundance or evolutionary development of a native species or ecological community". Three properties are believed to mediate vulnerability in environmental planning (Wilson *et al.* 2005): (i) exposure – the time a threatening process affects an area, often measured categorically as low, medium or high, (ii) intensity – measured by magnitude, frequency or duration, and (iii) impact – the positive, negative or neutral effect of a threatening process on biodiversity such as water-focussed grazing (Waite 1896; Osborn *et al.* 1932; Ratcliffe 1936; Lange 1969; James *et al.* 1995; Friedel 1997; Landsberg *et al.* 1997; Noble *et al.* 1998; James *et al.* 1999; Hunt 2001; Fisher 2001; Harrington 2002; Heshmatti e*t al.* 2002; Landsberg *et al.* 2003; Letnic 2004; Montague-Drake *et al.* 2004; Fukuda 2006; Underhill *et al.* 2007; Fensham & Faifax 2008). The level of intervention controls the impact of a threatening process. Waterpoint intervention therefore can possibly control vulnerability of rangeland areas through the intensity and exposure to grazing. Grazing intensity is measured by stocking rate per water but intensity changes around the waterpoint, depending on the threshold area over which cattle wander from water (Andrew & Lange 1986a, b). Over time, sustained grazing creates persistent grazing gradients (Bastin *et al*. 1993) which tend to radiate out from waterpoints. This can occur in many different patterns by cattle depending on the topography, patchiness of nutritious forage (Hunt *et al.* 2007), wind and rocky terrain which can create uneven grazing exposure in some ecosystems. Grazing exposure can also be uneven depending on grazing land management practices and rainfall seasonality. If stock start to loose condition on waterpoints, land mangers usually take them off and transport them to other waterpoints if conditions allow or send them to market.

Most work on the 'vulnerability' of Australia's rangelands over the last 50 years has been as condition assessments of native vegetation (pastoral plant species) and the land for the purpose of sustaining livestock production for private benefits (Smyth *et al.* 2009). Work on 'biodiversity condition' has occurred at a slower pace and at most provide *ad hoc* 'snapshots' of condition in space and time (Smyth *et al*. 2009). Attempts have been made to use rangeland monitoring to indicate *post hoc* biodiversity changes (e.g. plant structural dynamics, Watson *et al.* 2007) but are limited by sampling design issues.

The assessment of 'biodiversity condition' and changes from waterpoint interventions is important for environmental planning by many different managers of biodiversity. The SA Department for Environment and Heritage is interested in terms of biodiversity conservation, the SA Department of Water, Land and Biodiversity Conservation, S. Kidman Pastoral Company and Todmorden Pastoral Company are engaged in terms of managing pastoral leases sustainably and the South Australian Arid Lands NRM Board in terms of managing the rangeland's natural resources. The status and changes in biodiversity and its trends can inform conservation policies, the management and rehabilitation of biodiversity, locations for protected areas and environmental stewardship programs. In the rangelands, the information can also be useful for assessing 'duty of care' compliance of leases and the accreditation of niche livestock products (ISO 14001 standard for sustainable environments) (Smyth *et al.* 2003). Most attempts at assessing biodiversity for policy and planning purposes have used or modified (multi)metrics for environmental stewardships (Smyth *et al.* 2009). However, the lack of a guiding framework is urgently needed especially for claiming evidence-based outcomes from management intervention.

In this work, we undertake five related activities.

1. We develop and apply a biodiversity assessment framework as a guide on how to assess and evaluate outcomes from waterpoint interventions (Problem definition).
2. We assess the merit of environmental stewardship metrics for reporting biodiversity outcomes from waterpoint intervention (Metric design).
3. We undertake waterpoint intervention experiments to test the predictions in an homogeneous ecological community that (Waterpoint field experiment):
    a. Grazing relaxation via waterpoint intervention (or land reservation) will have a richer biodiversity than areas under sustained grazing exposure
    b. High grazing intensity (relatively heavily grazed areas near the waterpoint) will have lower biodiversity compared to low intensity grazing (lightly to negligibly grazed areas far from the waterpoint).
4. Because of the dynamic patchiness of arid rangelands caused by rainfall, we study its effect under controlled conditions in the glasshouse, using the soil seedbank from experimental field sites. We test the same hypotheses as above (Rainfall seasonality).
5. To extrapolate our findings on vegetation in the context of broader landscape patchiness, we explored the spatial variation in our study system and the long-term temporal effects of rainfall on vegetation cover.

METHODS

Five activities were undertaken: (1) application of a new biodiversity condition assessment framework, (2) a review of the use of existing biodiversity metrics, (3) waterpoint ‘closure’ field experiments, (4) rainfall seasonality and seedbank experiments, and (5) spatio-temporal patchiness of gibber-gilgais vegetation in a broader context. The Stony Plains Bioregion was used to assess evidence-based biodiversity outcomes from waterpoint closure.

1. Case study region

The Stony Plains Bioregion covers an area of 129, 240 km^2 in the central northern half of the State of South Australia. Its key features are the vast undulating gibber and gypsum plains that can be separated into five major landforms, each supporting broadly different ecological communities. The stony plains and tablelands are the dominant landforms (about 70% of the bioregion), are varied and include sloping gibber plains with cobble-sized stone cover and gibber pavements of small pebbles. Other landforms include the breakaways (tablelands) and other residual landforms of mesas and tabletops, dunefields and sandy plains, drainage lines and flood plains and the Great Artesian Basin springs (Smyth *et al.* 2009).

The bioregion supports 17 major vegetation types, 784 plant species, approximately 230 bird species, 100 reptile, five frog and 41 native and 11 feral mammal species (Brandle 1998). Seventy-seven land types (smallest sub-regional management unit) have been mapped for the region (Fleming and Brook 2008). They range in area from 457 to 11,079 km^2 and represent the diversity of ecosystems at a broad scale.

The bioregion is situated in the most arid region of Australia with a median annual rainfall of 150 mm that is spatially and temporally patchy. Rainfall for most of the region is stochastic and difficult to quantify, nevertheless, Oodnadatta typifies annual rainfall for the bioregion. Unlike seasonal areas, the rain year extends from August to July. Since 1892, annual rainfall peaked at its maximum deviation below a century average (rainfall data for 1892 - 2007) in 1955. Thereafter, annual rainfall gradually increased although most of it was still below the century average. In 1999/2000, annual rainfall peaked above the century average, the wettest it has ever been in the past 116 years. By 2007, annual rainfall had dropped to the century average. However, it is soil moisture that drives vegetation responses and is affected by daily temperatures through evaporation (Table 1). Summer temperatures are consistently high, reaching a maximum of 50.7^{o}C and winter temperatures rarely drop below zero. Under climate change (worst scenario by 2150), the climate is predicted to become hotter (increase by 2, 5, 7 ^{o}C for 2030, 2070, 2100) and drier (rainfall decreasing by up to 10, 20, 30% for 2030, 2070, 2100) (CSIRO OZClim 2007).

Land uses are livestock grazing, mining, ecotourism, protected areas and regional towns, with grazing being the most widespread activity such that most of the region is under pastoral lease. Livestock graze on chenopod *Atriplex/Sclerolaena/Maireana* vegetation association of low open shrubland over sub-shrubs and grasses, which cover about 75% of the bioregion (Brandle 1998). Cattle grazing dominates the northern two-thirds of the region.

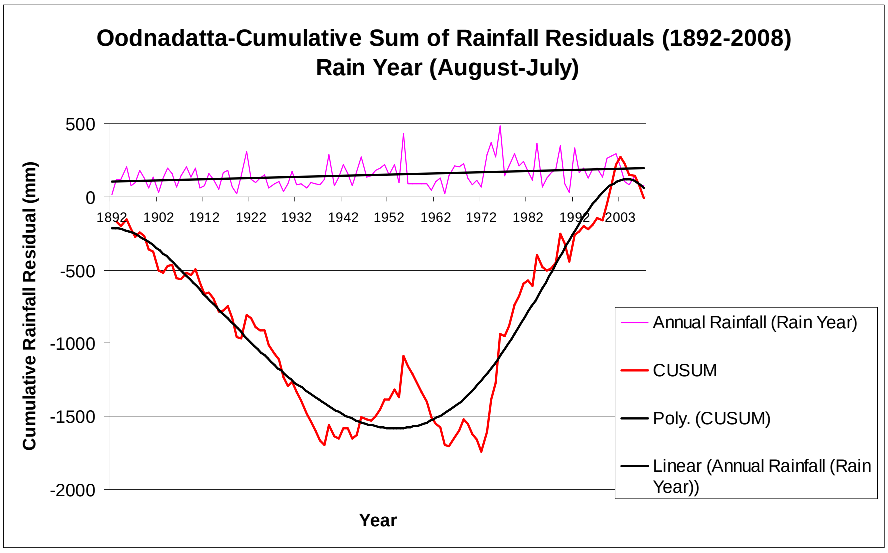


Figure 1. Annual rainfall and cumulative rainfall residuals above and below the century mean annual rainfall for Oodnadatta, South Australia, 1892-2008. Polynominal curves are fitted for each rainfall variable. (Rain year is August to July, cum sum = cumulative sum, Poly. (cum sum) = polynomial curve for cum sum; Source: Foulkes, Department for Environment and Heritage, 2009).

Table 1. Mid-summer and mid-winter temperature comparative statistics (based on 30-year climatology) for Oodnadatta, Coober Pedy and Alice Springs climate stations (Bureau of Meteorology 2005)

(Data: 1961 – 1990)	Oodnadatta (SA)	Coober Pedy (SA)	Alice Springs (NT)
January (mid-summer)			
Mean Min - Max	19.6 – 37.7	20.7 – 36.4	21.3 - 36.2
Daily range	31.9	37.6	34.7
Highest maximum	50.7	47.0	44.7
Mean No. days >= 40°C	10.4	7.7	6.5
Lowest minimum	11.7	9.4	10.0
Variability Index[1]	0.3	0.4	0.3
July (mid-winter)			
Mean Min - Max	5.8 – 19.6	6.3 – 18.7	3.7 – 19.5
Daily range	34.8	34.0	37.4
Highest maximum	32.2	32.0	29.9
Lowest minimum	-2.6	-2	-7.5
Mean No days <= 0°C	0.7	0.1	7.0
Variability Index[1]	1.6	1.0	3.1

[1] Variability Index = (90 percentile – 10 percentile)/50 percentile

2. Defining the problem with a biodiversity assessment framework

With no existing framework for assessing biodiversity outcomes from waterpoint intervention, we developed a framework for assessing biodiversity condition by building on developmental work on biodiversity stewardship metrics (Smyth *et al.* 2007) and measures (indicators) for assessing biodiversity condition (Smyth *et al.* 2009a) previously undertaken for the case study region. We used this approach because waterpoint management by relaxing total grazing pressure is an on-ground activity advanced to retain or restore (if possible) the resilience of ecological communities/habitats for the bioregion (DEHSA and SAAL NRM Board 2008). This also was the reason for undertaking a biodiversity condition assessment for the Stony Plains bioregion (Smyth *et al.* 2009a). The framework embraces biological diversity and its role in maintaining ecosystems (Smyth *et al.* 2009b). It has seven steps:

1. define biodiversity condition
2. prioritise the outcomes of a biodiversity condition assessment using priority setting related to management outcomes
3. identify biodiversity surrogates
4. select robust biodiversity indicators or measures
5. design and implement long-term monitoring
6. evaluate monitoring results
7. adapt biodiversity planning and management.

The justification for each step is published in detail in Smyth *et al.* (2009b), so here we summarise this information.

Step 1 - Defining of biodiversity condition

We defined 'biodiversity condition' as *differences between trends in biodiversity indicators for relatively unmodified/resilient (reference) and modified ecological systems of the same type monitored for the same survey period*, where 'biodiversity' is *terrestrial species* (vertebrates and vascular plants) *and ecological communities* (aka habitats, land type, regional ecosystem) of the Stony Plains Bioregion. We chose these biodiversity entities because that is the biotic level where most work has been done in the rangelands and are of interest to biodiversity planners.

The focus on differences between reference and modified systems solves the problem of 'how much' change is required to signal a threshold shift. We assume that ecosystem functioning is maintained if management activities operate within the range of conditions expected under natural disturbance regimes. We suggest using present day 'reference' conditions to represent relatively natural ecosystem variability, as it is impossible to reconstruct pre-settlement conditions.

Step 2 - Scoping the objectives and outcomes

Before identifying the objectives and the desired outcomes, a systematic prioritisation assessment of the environmental issues affecting changes in biodiversity and ecosystem functioning is a necessary first step for the reasons outlined in Smyth *et al.* (2009b). The prioritisation assessment involves four tasks:

1. identification of biodiversity values and assets,
2. analyses of environmental issues affecting biodiversity and ecosystem functioning,
3. identification of biodiversity management priorities, on-ground interventions and capacity to achieve management, and
4. assessment of ‘duty of care’ responsibilities under the legislation (leasehold permits in South Australia).

In our case study, information for each task was collected using scientific and ‘biodiversity manager’ workshops, and reviews of the scientific and grey literature about the biodiversity and ecological communities of the Stony Plains Bioregion.

Step 3 - Identifying biodiversity surrogates

The surrogates were based on the objectives and outcomes for species and ecological communities identified in Step 2, with the underlying assumption that they represented broader biodiversity. Where possible, we used cross-taxon surrogates as these have been shown to have the best possible surrogacy value (Rodrigues and Brooks 2007). To ensure that the surrogates affected or responded to changes in ecosystem functioning, we assessed the role of each. A coarse and admittedly incomplete list of key functions for each biodiversity value based on the best knowledge we had at the time were identified.

Step 4 - Selecting robust biodiversity indicators to measure surrogates

Indicators are often used interchangeably with surrogates (Rodrigues and Brooks 2007). They differ by being *measures* of the condition of environmental phenomena, as barometers for trends in natural resources (Suter 2001; Niemi and McDonald 2004). In our work, they are the biological (and sometimes abiotic) measures of surrogates for species and ecological communities that can be measured in the field or remotely. We ran an expert workshop of terrestrial ecologists familiar with the region to identify all potential indicators. Next, we undertook a desktop assessment of the robustness of each indicator in an attempt to have indicators that spread across the science-planning spectrum. We justified *robustness* of indicators by systematically subjecting each indicator to an integrated set of criteria (33 in total). The criteria relates to the biological relevance, historical dimensions, measurement qualities, feasibility of implementation, relevance to policy and management utility of indicators applied in natural resource management (see Table 1, Smyth *et al.* 2009b). We adopted the precautionary principle (following Gray and Brewers 1996) and assumed equal weightings among criteria to avoid applying importance to criterion based on the values, depending on where one works in the biological science-planning interface. The rule-of-thumb for a robust biodiversity indicator was it had to *meet approximately two-thirds (67%) of the selection criteria* (i.e. 22 or more criteria). We accept that the rule is subjective, but it is the best solution given the gaps in our knowledge of rangeland biodiversity. As a precaution, we used a top-down approach, with the intent of discarding indicators once evidence indicated they could be misleading.

Step 5 – Sampling design and implementation of long-term monitoring

Less attention has been given to sampling design issues for assessing condition in terms of biodiversity. The monitoring and assessment of biodiversity condition in any biome requires sophisticated sampling design and statistical analysis. Most environmental impact sampling designs (e.g. ‘beyond BACI’, ‘before-after, control-impact’) are best for assessing the impact of planned disturbances/interventions (e.g., waterpoint installations). Nevertheless, the knowledge gained by applying environmental impact designs especially ‘beyond BACI’ designs in Australia are useful for guiding sampling designs for assessing biodiversity condition in places where human disturbances already operate or are largely unplanned reactively.

Other important issues for sampling design monitoring biodiversity condition are statistical power, sample site arrangement to detect change, spatial and temporal scales (Underwood and Chapman 2003) and rainfall patterns (Hewitt *et al.* 2001).

Step 6 - Interpreting biodiversity condition

A range of approaches have been adopted for interpreting biodiversity condition, ranging from simple aggregates (e.g., Ant Index of Biological Integrity, Majer and Beeston 1995; Biohyets, Read *et al.* 2005); conditions score such as BSSs (e.g., Oliver and Parkes 2003), “BioCondition” (Eyre *et al*. 2006a, b) and graphical representations (e.g., radar plots, Suter 2001) to more sophisticated multivariate mathematical treatments (Andreasen *et al.* 2001). The use of a single score (multimetrics) has been challenged on the basis of being misleading and lacking transparency (see attached review by Smyth & Davies, under revision). Consequently, we prefer using raw biodiversity indicator data values in preference to derived values such the (multi)metrics.

There are a large number of published methods for interpreting change in trends. Some are regression techniques for disturbance gradients, time series analysis, a Bayesian approach and post-hoc ‘beyond BACI’ if a putative change is suspected (especially if the interest is to estimate temporal variation) and post hoc ‘impact vs control’ where sites are treated as experimental units. However, most approaches while uniformly powerful assume data conforms to the assumptions of eventual analysis, which can create practical difficulties for the researcher and is a constant challenge in environmental change assessments. There are statistical solutions to this problem (Underwood and Chapman 2003).

3. Metric design

We undertook a review of biodiversity metrics used in environmental stewardships to assess the value of metrics (see attached manuscript by Davies & Smyth). From an ecological perspective, we reviewed the designs of existing biodiversity metrics applied for environmental stewardships, the key issues affecting their designs and assessed the implications of their application for biodiversity planning in the dynamic environments of the arid rangelands. We consulted the international and Australian scientific literature and research reports for our review.

4. Waterpoint field experiments

The methods for this work have not been described in detail previously, so we present them here for this report as 'Methods' text for two future scientific papers devoted to the vegetation and fauna results of the waterpoint intervention experiment. In these papers, we would shorten the study area description but include Table 2.

The study area and sites

Sixteen sites were selected across the two study properties (Todmorden and Macumba Stations and the Oodnadatta Town Common via Oodnadatta area of northern SA) in the open gibber-gilgai landscape dominated by low chenopod shrubs and grasses (Fig. 2). These gilgais represent the productive parts of the landscape and were the focus of our research. Cattle grazing is commonly centred on artificial waterpoints (troughs from bores and rain-fed dams) due to the sparseness of permanent natural water sources. The rough cobblestones of the gibber pavement restrict cattle grazing to within only approximately 4 to 5 km from water, differing from other biomes of up to 10 km with softer substrates (Hodder & Low 1978; James *et al*. 1999). After significant rains, cattle move off the waterpoints to forage more widely, relying on the semi-permanent gilgais as water sources.

The gibber-gilgai landscapes are a unique landform not found anywhere else in the world and South Australia contains most of them. The patterning consists of two major elements: (1) extensive shelves of stone pavements in which the stones or gibbers consist mainly of quartzitic silcrete forming a layer usually only one stone thick overlying heavy loam to clay soil which contains few stones and (2) gilgai depressions from two to greater than 30 metres in diameter which mostly lack a stone pavement particularly on the downslope side (Jessup 1960). The soils of the stone pavements have extremely low rates of water infiltration, and so are water shedding in contrast to the gilgai depressions where open cracks can provide very high rates of water infiltration and the depression accumulates water and provides a long-lasting surface detention capacity (Hunter and Melville 1994) . Water can pond up to six months in winter and two months in summer, depending on the size and depth of the depression.

The gilgais are formed by the alternate cycles of clay soil shrinkage and swelling associated with wetting and drying (Ollier 1966; Stace *et al*. 1968; Ollier 1966). While the gibber covered shelves in between the gilgai are largely devoid of vascular plants, the depressions support a diversity of flora characteristic of both semi semi-arid tropical and temperate regions. Even slight variations in microtopography (< c. 30 mm) associated with gilgai in semi-arid New South Wales produced marked variation in vegetation, with greater productivity and diversity in the depression (Wilson and Leigh 1964). Cracks, which are open most of the time in the base of the gilgais provide habitat and shelter for a variety of reptiles, small mammals, small birds and invertebrates (Brandle 1998). Consequently, our fauna sites are centred on the gilgais (patches) with sampling extending into the pavement (the matrix). Vegetation sampling centred on the gilgai depressions.

The gibbers themselves vary from large boulders to a closely packed mosaic pavement that is highly resistant in parts to traffic and erosion (Bourman and Milnes 1985). In the gibber-gilgai landscapes of the Oodnadatta region, upward movement of rock clasts through the soil is likely to be responsible for their dominance at the soil surface and absence throughout most of the soil profile (Jessup 1960)(Fig. 3). This action is believed to maintain and restore soil structure once rested from repeated trampling by cattle (and other ungulates historically).

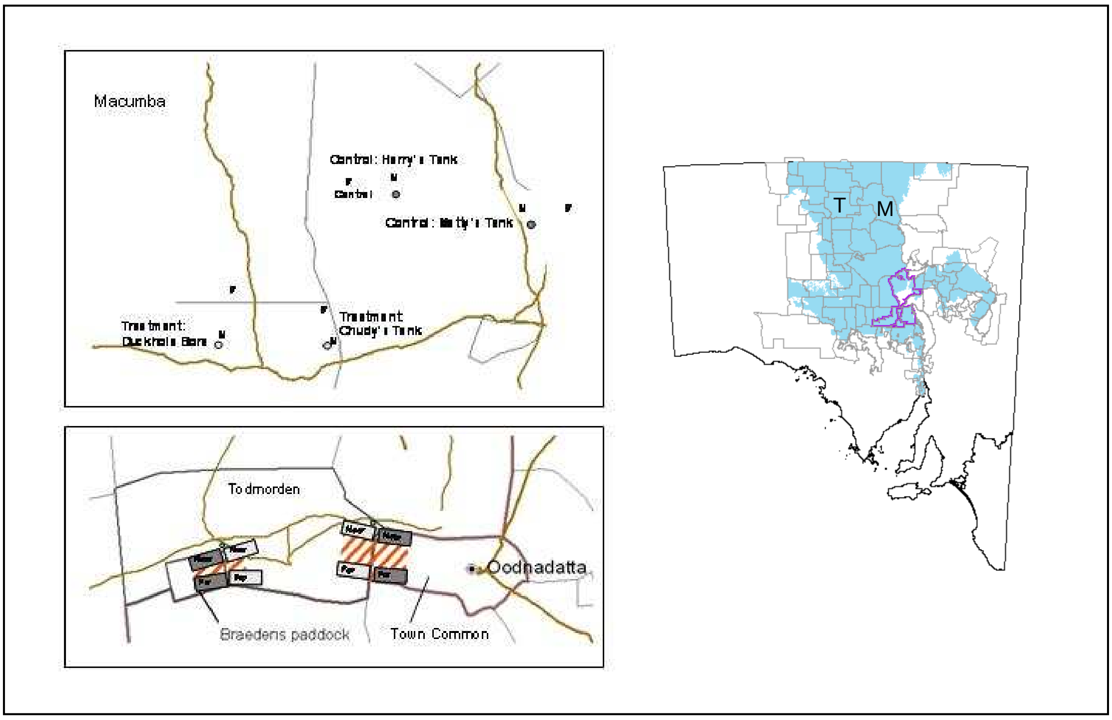


Figure 2 Map of study sites for intervention (**M**acumba) and calibration (**T**odmorden) experiments. (Intervention experiment: treatment – grazing relaxation (stock removed off water point); control – on-going grazing, N – near (~< 1 km from waterpoint), relatively high grazing intensity, F – far (~> 4 km from waterpoint); relatively low or negligible grazing intensity, grey lines – fences; mustard lines – roads, circles –waterpoints. Calibration experiment: treatment – dark areas; long-term grazing relaxation; control – light areas, on-going grazing; near and far as per intervention experiments; brown lines – tenure boundary)

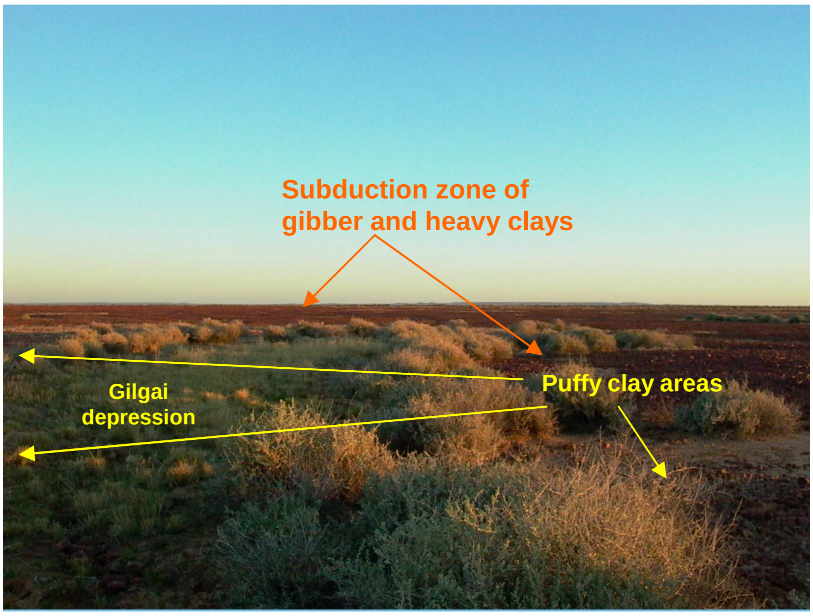
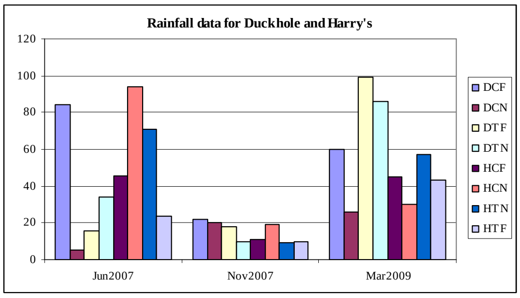


Figure 3 Part of a large, grazed gilgai (30 m long x 5 m wide) showing a base of heavy, cracking clay soil and the leading edges of gibber that forms a subduction zone and trailing edges of "puffy" loam, clay soil. The gilgai is surrounded by a matrix of non-vegetated, gibber pavement

Rainfall for the study sites prior to each sample was highly variable spatially and temporally (Fig. 4). Average minimum and maximum temperatures for the month of each 14-day sample period were typical for the bioregion: March 19.2 – 33.7°C, June 6.5 – 19.9°C, October 15.1 – 30.3°C, and November 18.5 – 33.7°C.



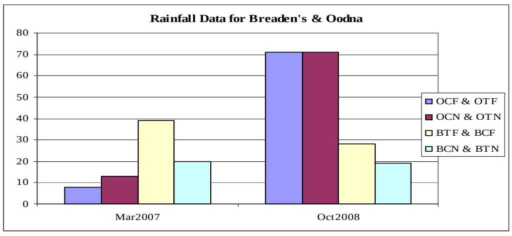


Figure 4. Total rainfall for intervention and calibration sites, November 2006 – March 2009 based on one event before the sampling dates. (March 2007 – 4 months total, June 2007 – 9 months total, November 2007 – 4 months total, October 2008 – 18 months total, March 2009 – 16 months total)

The vegetation is usually a low sparse shrubland most frequently dominated by the long-lived perennial shrub *Atriplex nummularia* ssp. *omissa*, (hereafter referred to as *A. nummularia*) and occasionally by *Atriplex vesicaria. Maireana aphylla* and occasionally *Chenopodium auricomum* also dominate on sites that receive more runoff or where water is retained for longer, while the halophytes *Sclerostegia medullosa* and *Frankenia serpyllifolia* are indicative of gilgais with more saline soils. *Astrebla pectinata*, *Eragrostis setifolia* and occasionally *Eriachne ovata* are the dominant longer-lived perennial grasses in the ground stratum although mostly sparse in cover. The short-lived perennial grasses and a number of short-lived perennial sub-shrubs also frequently dominate the ground stratum but die out during extended dry periods. The forbs are also frequent in the in ground stratum during wetter periods but persist as perennial root stock during dry periods. After soaking rains, a diverse range of short lived species germinate especially after summer rains and include a wide range of ephemeral grasses. The deeper parts of the gilgai that retain water for extended periods are dominated by species associated with wetlands. With the exception of dominant species, plant species composition and abundance (especially of ephemerals) vary greatly between gilgais within a local region. This is probably due to the great range of gilgai sizes, soil variation between gilgais, the differing extents to which water flows into gilgais from adjacent impervious areas, and the extent to which a gilgai is disjunct from adjacent gilgais.

Historically, the study area has been stocked (sheep, goats, horses, camels) since 1885 (Fleming 1998). Sheep shepherding (herds of 1,000-2,000 head protected from dingoes) occurred up until 1940s after which fencing concentrated stock in paddocks. During these times, grazing was restricted to permanent and semi-permanent natural water resources (springs and waterholes). It was self-regulating in that stock was forced to move on when semi-permanent waters dried up. However, when ephemeral forage (the historical basis of livestock production) dried off during long dry periods, starving stock retreated to a few permanent waters to eat all foliage they could reach. Consequently, riparian systems were over-grazed and some species were permanently damaged (Fleming 1998). Dingo predation of sheep in paddocks forced a switch to cattle. With intensive fencing in 1970s, came intensive installation of artificial waterpoints. Although this development rested the few permanent waters from grazing pressure, it extended grazing into almost all areas of the biome. However, because of the roughness of the gibber pavements on hoofs, cattle mostly graze within 4 km (occasionally 5 km) from water. We only once saw 2 head of cattle at about 5 km from water. However, cattle dung was detected in all of our sample areas, highlighting their ability to spread out when conditions allow. At the time of our study, the numbers of other herbivores was low. No camels were sighted, up 13 donkeys were observed on Macumba Station but were removed by the end of the study and 12 to 40 horses roamed the Oodnadatta Town Common but 8 km away from our study sites. Goats were never sighted and rabbit activity was minor and restricted to areas of soil where they can sustain burrows. Red Kangaroos, the only large native herbivores, were very sparse at the Macumba sites, but relatively common at Oodnadatta-Todmorden sites.

Experimental design

Two related experiments were carried out in the study area (Fig. 2). The first one determined the effects of waterpoint manipulation on plants, small ground mammals, reptiles and pilot samples of soil microbial activity at eight sites on Macumba Station using a ‘beyond BACI (Before, After, Control, Impact)’ design to study the response of biota to a waterpoint intervention (termed the intervention experiment). The second experiment (termed the calibration experiment) used ‘cross fence’ paired comparisons of the remaining eight sites on Todmorden and Oodnadatta Town Common for two reasons: (i) to understand the resilience of gibber-gilgai landscapes that have been largely free of livestock grazing for >15 years and (ii) to confirm the generality of results from the intervention experiment. Each of the 16 sites contained five replicates, giving a total of 40 sample plots for each of the intervention and calibration experiments.

The experimental design for intervention experiment had three factors with two levels in each: (i) grazing exposure – on-going (control), relaxed (treatment); (ii) existing grazing intensity – relatively heavy grazed (0.2 –1.5 km, near), lightly to negligible grazing (> 4 km, far) and (iii) times – before and after waterpoint intervention. Approximately 80 to 100 head of cattle per waterpoint sustained ‘on-going’ grazing exposure. To induce grazing relaxation, all cattle were moved off the waterpoint (closure) in September 2008 and remained that way for the duration of the study. The very low densities of kangaroos (probably due to dingo predation when compared with Todmorden on which dingoes were regularly poison bated) and negligible feral herbivores meant grazing relaxation was maintained. We controlled for grazing intensity to test the assumption of grazing decreasing with distance from water. Although several studies have shown it to be sound, the fact that gilgais can function as semi-permanent water sources and induce grazing relaxation at permanent waters meant we needed to confirm the existence of grazing gradients.

The calibration experiment had two factors with two levels in each: (i) grazing exposure – long-term grazing relaxation (> 15 years) (treatment), ‘on-going grazing’ (control); and (ii) grazing intensity – relatively heavily grazed (0.2 – 2 km from permanent waters), light grazing (>4 km from permanent waters). Distributions of feral herbivores and kangaroos were very sparse.

All sites were randomly chosen in homogenous gibber-gilgai away from gidgee drainage lines, with equal representation of relatively small and large gilgais on across rises and on plains (Table A3. Although the landscape context of the sites varied within a hectare, the cover of gibber pavement, puffy clay, gilgai depression and ‘hard pan’ were represented evenly in similar relative proportions (Table 2).

Sampling design and techniques

We stratified sampling to control for differences in plant productivity due to rainfall seasonality. Sampling of the intervention sites was planned twice before and after the intervention under no plant productivity (dry condition) and observable plant productivity (in response to rain), giving a total of four visits. The expected total number of samples for all intervention sites is 160 but is presently 140 as one sample is outstanding. Sampling took place before the intervention in June/July 2007 (autumn plant productivity) and November 2007 (no plant productivity). Cattle were removed in September 2008 and the first ‘after’ sample occurred March 2009 after moderate rains in November and December 2008 (summer plant productivity)(Fig. A4). We hope to complete a ‘no plant productivity’ sample in July/August/September. Calibration sites were sample under “no plant productivity” and “light plant productivity” (low rainfall) in March 2007 and October 2008 respectively. This makes two visits in total. These samples were supported by the rainfall seasonality experiments. The total number of samples for all calibration sites was 80.

Table 2. Variation in landform characteristics of replicates for each level of each factor for the waterpoint intervention and calibration experiments. (1. Landform and 3. Gibber size are presented as % cover classes: 1= <5%, 2=5-25%, 3=25-50%, 4=50-75%, 5= >75%). Characteristics of each replicate were mapped within a hectare grid with the centre of the gilgai as the centre point of the grid. Cover percentages across characteristics do not add up to 100%. ‘Puffy clay’ is very fine loamy, clay soil, gilgai depression is the ‘bowl-like’ shape of the heavy, cracking clays and ‘hard pan’ is the ‘claypan- like’ bare surface. Intervention Experiment: C – control, on-going grazing; T – treatment, grazing relaxation, N – near, relatively high grazing intensity; F – far, relatively low grazing intensity)

Characteristic	Waterpoint Intervention Experiment								Calibration Experiment							
	Harry’s Tank Sites (H & M)				Duckhole Bore Sites (D & C)				Braedens Sites (B)				Oodnadatta Sites (O)			
	HCN	HCF	MCN	MCF	DTN	DTF	CTN	CTF	BCN	BCF	BTN	BTF	OCN	OCF	OTN	OTF
1. Landform																
Plain	5	5	5	5	5		5	5	0	0	0	0	5		5	
Rise	0	0	0	0	0	5	0	0	5	5	5	5		5		5
2. Min-max Slope	-1 - 2	-1 - 3	-1 - 0	-1 - 3	0 - 2	-1 - -5	0 - 3	0 - 3	0	-1 - 0	-5 - 1	-2 - 1	0	0	0	0
3. Gibber size																
Pebble (5 - 50 mm)	5	5	5	5	5	0	5	5	5	5	0	4	5	0	5	0
Cobble (51 – 250 mm)	0	0	0	0	0	5	0	0	0	0	1	0	0	5	0	5
4. Min-max gibber cover (%)	42-83	54-89	42-83	54-89	42-74	70-80	51-70	60-74	50-80	60-80	30-82	50-80	50-80	45-60	30-60	50-85
5. Min-max ‘puffy clay’ cover (%)	2-17	5-27	1-12	0-9	2-11	5-10	10-25	5-50	8-40	2-22	5-19	0-20	10-40	30-45	35-60	15-50
6. Min-max ‘gilgai depression’ (%)	2-13	2-21	1-10	3-9	5-16	4-37	5-50	4-26	5-20	3-19	7-19	3-50	5-15	5-15	5-40	15-60
7. Min-max ‘hard pan’ cover (%)	5-19	0-13	2-13	5-14	2-16	1-15	2-15	2-15	0	0-1	0-1	0-10	0-10	0-1	0-15	0-1
8. Distance from water (km)	0.5 - 1	6	0.5 - 1	4	0.5 - 1	7	0.5 - 1	6*	~ 0.3	5	0.5	5	~0.5	7	0.2	7
9. Historical grazing history	Negligible grazing until tank about10 yrs ago		Semi-permanent waterhole 2 km to MCN, up to 500 head after big rains. Tank about 10 yrs old		Bore installed in 1940s, up to 500 head after big rains		Negligible grazing tank installed about 10 yrs ago.		Up to 500-600 head of cattle from 1890s		Up to 500-600 head of cattle from 1890s		Up to 300-400 head of cattle, horses, donkeys, goats and camels from 1890s		Up to 300-400 head of cattle, horses, donkeys, goats and camels from 1890s	
10. Present stocking rate (cattle/water)	80 – 100		80 – 100		80 – 100		80 – 100		Stocked with 100 head of cattle for 90% of time centred on South Gap Bank Dam		Infrequently stocked since 1980s by100 head of cattle for up to 2 mo/yr		Stocked since 2002 with 50-60 head of cattle centred on Clarrie’s Tank		Little grazing since 1960s with 40-50 horses and some cattle centred on water >8km away	

* large semi-permanent waterhole at about 2 km from the nearest site filled up after rain in June 2007.

Simple pseudoreplication was minimised as much as possible as plots were centred on the gilgai, which have their own microtopgraphy. Nevertheless, macrotopography at the landscape scale is likely to have an effect within a set of five replicates but not between sets because of the distances especially between the eight sites in the intervention experiment. In the calibrate sites, the four ‘near’ sites may be temporal pseudoreplicates if significant rainfall ran-on through gilgais across the landscape. This is most likely at BCN and BTN replicates as the gilgais were on slight slopes. However, no significant rainfall events (>100 m) occurred during the study that would create this run-on pattern.

Despite our original intention, it was not possible to centre the plant and fauna sampling on the same gilgais as the repeated visits of the fauna sampling disturbed the vegetation. Plant plots were located a close as possible to the fauna ones.

Plant sampling

A 10 x 10-m^2 plot was permanently pegged out across each gilgai. Eleven parallel transects, each perpendicular to the longest axis of the gilgai, were located at one metre intervals. Percentage cover of each species (along with litter, rock, and bare ground) was estimated using the “wheel point intercept” method, the species name of each plant intercepted by a wheel point being recorded along each transect at 75 cm intervals (Griffin 1989). Where a wheel point struck more than one species at one point, the first species struck (i.e. the highest) was differentiated from the other species. Plants were included in species cover estimates even when dead, but were recorded as litter when no longer rooted in the ground. Species richness for each quadrat was determined by recording the names of all species identifiable at least to genus level, including rooted dead plants.

The only consistently-present, dominant, perennial plant species in the quadrats was *A. nummularia.* For each quadrat, the height and foliage area of each plant in the quadrat of this species was determined, area (A) being estimated from measurements of the diameter at the widest axis (D_1) and of the perpendicular axis (D_2) using the equation $A = \pi((D_1 * D_2)/4)^2$.

Species were classed by palatability: (i) ‘palatable’, (ii) ‘highly palatable’ (preferentially grazed by cattle in all situations) and (iii) ‘highly unpalatable’ (unpalatable in all situations) based on a consensus of the literature and by longevity/lifeform (Davies *et al*. 2008 attached) for the purpose of summary statistics.

Fauna sampling

Small ground-dwelling mammals and reptiles (excluding large snakes) were sampled using a combination of pitfall and Elliott traps. These were set up four months before sampling in November 2006 at the calibration sites and completed at the intervention sites by March 2007. Each plot had four 250mm x 400mm plastic buckets and three 160mm x 500mm pipes set in a 3-pronged star arrangement connected from the central bucket by 15m of drift fence. The pipe pits were at 5m from the centre whilst the remaining buckets were set at 10m. Fifteen baited Elliott traps were set around this configuration. A total of 280 pits and 360 Elliott traps were set for each experiment on each visit. Traps were opened for 7 nights with Elliott traps being closed during the heat of the day and the bait removed to deter ants. Foil shelters (roof tile insulation) were placed in base of the pitfall traps to protect captured animals from heat and predators. Animals were temporarily marked with colour codes using paint pens to enable recaptures and movements to be tracked over the trapping period. Animals were handled for identification and marking purposes, and released at point of capture. All animals were measured and weighed to help assess condition. Pit traps were checked mornings and evenings.

Pitfall traps were left in the ground for the duration of the study. Large bucket pit traps (270mm x 400mm) were capped with a plastic bucket lid to which we screwed a 3mm thick galvanized steel plate (305mm x 305mm). The smaller pipe pit traps (166mm x 500mm) had a double thickness of flywire mesh and a clear plastic dish at the base to prevent escapes by fossorial reptiles and were capped with a 2mm thick galvanized steel plate. This plate was glued to the rim of the pipe using a continuous bead of silicone gap sealer. The 4 corners of the steel plate have been bent down and at two of these corners the plate will be screwed into the sides of the pipe to prevent the lid being moved. Both types of trap will be covered over with local soil to reduce the possibility of human interference and weathering of the trap materials.

Each replicate was located at distances greater than 200m apart based on published and anecdotal observations of small mammal and reptile movements to maintain independence between replicates. This was upheld in the study. Of 240 sampling events across the intervention and calibration experiments, independence was violated twice, one pair of replicates in each experiment. Temporal pseudoreplication was minimised as much as possible within resource constraints. However, multiple samples from each plot were taken simultaneously for a set of 20 replicates, followed by another set of 20 replicates over a 12-day period. Pseuoreplication is an issue when the climatic conditions at replicates vary during the sampling period thereby influencing the biotic responses differently. We observed a switch in climatic conditions only once in the calibration experiments when a dust storm in the second 6-day sampling period (March 2007) bought rain and lowered temperatures by 10^{o}C. This noticeably affected reptile captures.

A pilot study was undertaken to test an appropriate sampling scale for resident, ground-dependent birds but was not continued as the spatial scale required to obtain a representative sample was too resource intensive and beyond the capabilities of this study. Invertebrates were also captured in the pitfall traps but their collection was discontinued to allow more effort for the vegetation sampling component subsequent to the botanist leaving the project.

Soil condition

Soil condition was not sampled in further detail as it appeared to be comparatively well structured right up to the waterpoints (Dr Garry Cook, *pers. comm.*) probably because of the underlying landscape processes and relatively low stocking rates compared with less arid areas. Nevertheless, eight same-sized samples (200 g) of soil were collected from each of two relatively heavy grazed replicates grazed regularly and two lightly grazed replicates which had been rested from cattle grazing for 6 months under the same rainfall conditions. The purpose is to assess microbial activity and do catabolic profiling to quantify the resilience of soils.

Statistical analysis

To avoid sacrificial pseudoreplication, no multiple samples were pooled prior to analysis.

Vegetation

Differences in species composition for different factors (i.e., different grazing exposure and intensity) were described using 2 or 3 dimensional NMDS ordinations (following Clarke 1993). These techniques were applied to unstandardised, fourth root transformed density and cover data, using Bray-Curtis similarity measure and the Jaccard measure for absence/presence data. “Rare” species, defined as those occurring in less than 10% of quadrats (and represented by less than 20 seedlings over all quadrats in the case of seed density), were excluded from the ordinations. We assessed any discernable separations in the rank (dis)similarities of the data by conducting a one-way ANOSIM and IMDs (index of multivariate dispersion). SIMPER analysis was done to understand which levels contributed to differences in the factors. Canonical Correspondence Analysis (CCA) in the “R” statistical package was used to investigate differences between standing herbage and germinable soil seed bank (Davies *et al.*, manuscript).

‘Beyond BACI’ analyses of transformed (if appropriate), selected functional groupings will be undertaken to test hypothesis about grazing relaxation, if the ‘grazing gradient’ assumption holds. Power is always a concern with GLM analysis because of the eventual assumptions but Bayesian approaches (following Fox 2001) is an option.

Fauna

A similar multivariate approach for vegetation will be taken for fauna once all the data is collected. Preliminary analysis used statistics for small sample sizes. Species will be classified as ‘sensitive’ to ‘tolerant’ species and predictive modelling using information-theoretic approaches (following Burnham & Anderson 1998; Garnett & Brook 2007) to understand vulnerability to grazing pressure.

5. Rainfall seasonality and soil seed bank

Rainfall seasonality was investigated using glasshouse experiments for ‘winter’ and ‘summer’ growing conditions while holding all other confounding factors constant. Detailed methods can be found in the attached report and manuscript by Davies *et al.* (2008; manuscript), so we provide a summary for this report.

Species composition and species density in the germinable soil seed bank was determined using well-established methods. Soil samples (5cm diameter and 5cm deep) were extracted at one predetermined, random distance along each of the eleven transects crossing every quadrat in the vegetation plot. At each core site, two adjacent samples were taken, one for germination under winter conditions (5-25°C & short day length) and the other under “summer” condition (15-35°C & long day length). Each set of eleven samples was bulked and spread in a tray 35 cm by 29 cm over 3 cm of steam sterilized potting mix, resulting in one “summer” and one “winter” tray for each quadrat. All trays were placed in a temperature-regulated, automatically-watered glasshouse at Flinders University in Adelaide, under natural day length conditions.

To overcome potential problems of differentiated conditions within the glasshouse, the germination trays were randomly positioned and rotated from one side of the glasshouse to the other on a weekly basis. “Winter” trays were set up in the glasshouse in June 2007, seedlings being regularly harvested, identified to species level and recorded over the following four months, up until October 2007. “Summer” trays were setup in October 2007 and harvested and recorded until February 2008. Previous to being placed in the glasshouse, soil was stored in open bags in the dark and at room temperature (c. 22°C).

All the soil seed sampling was undertaken in March 2007, during a severe extended dry period. Annual rainfall in nearby Oodnadatta the year preceding the study (2006) had been only 43.2 mm this being the driest year since records began in 1939. In the two and a half months of 2007 preceding the study a further 37 mm had fallen but this had been confined to early and mid-January, no rain being recorded in the two months preceding the study.

Statistical analysis followed those described for the vegetation analysis. Similar biological variables were analysed for standing vegetation observed in the calibration sites and the germinable soil seed. Germinable seed density data consisted of both ‘winter’ and ‘summer’ data combined, this being logarithmically transformed.

6. Spatio-temporal patchiness of gibber-gilgai vegetation

We investigated the patchiness of gibber-gilgai vegetation at broader spatial and temporal scales than our study sites using satellite imagery and conclude by assessing the merit of our approach. The methods of this work are described in detail in the attached report by Bastin (2009), so a summary is provided in this final report.

Two separate remote-sensing analyses were conducted for this component of this ‘biodiversity and waterpoint management’ study.

1. Very high resolution images were analysed to better understand spatial characteristics of gilgais for areas within and beyond the field-study sites.
2. Historic Landsat images were analysed to build a profile of change in vegetation cover across a range of seasonal conditions.

Spatial analysis

In the first study, Ikonos and Quickbird images (~1-m spatial resolution) were classified to map gilgai structures. Descriptive statistics (mean gilgai size, nearest-neighbour distance, gilgai density etc) were then used to build a ‘typology’ of gilgai types for each study site. Analysis areas were ~100 to ~1100 hectares (median = 281 ha) surrounding each group of field sites.

Temporal analysis

Seasonal changes in vegetation cover between 1974 and 2006 were reported using Landsat imagery (50-m pixel resolution to 1988 and 25-m resolution since). Cover change was calculated for two mapped areas of the ‘chenopod (*Atriplex / Sclerolaena / Maireana* spp.) low open shrubland over sub-shrubs and grasses’ unit (SA Department of Environment and Heritage’s survey of the Stony Plains bioregion). One polygon (710 km^2) was located on Todmorden Station and included both the Oodnadatta and Breadens field sites. The other was on Macumba (2285 km^2) surrounding the Harry’s Tank and Duckhole field sites.

RESULTS

1. Application of biodiversity condition assessment framework for waterpoint management

For waterpoint management, we present the results of the prioritisation assessment, the management objectives and outcomes, the surrogates and indicator selection process (Steps 2 to 4) as biodiversity condition (Step 1) is already defined in the Methods. Steps 5 to 7 are not reported as they can only be applied after a monitoring program has been implemented for a considerable length of time. These results are presented as a summary of an overall assessment for the case study region published by Smyth *et al.* (2009).

The biodiversity management priorities for the region were:

- Manage total grazing pressure by domestic, native and feral herbivores across watered areas so as to avoid further degradation of biodiversity, production and cultural values.
- Manage threats and pressures on species vulnerable to extinction.
- Plan infrastructure development to avoid disruption of natural surface water flows.

Based on the priority setting, three management outcomes about biodiversity condition were identified for waterpoint management (Table 4):

- Native vegetation maintenance or restoration.
- No species, population or community losses.
- Natural mosaics of water remoteness.

Four biodiversity surrogates for achieving the desired biodiversity condition outcomes for waterpoint management were identified. No surrogates were common across the different outcomes. The number of robust biodiversity indicators for the whole case study region ranged from 2 to 21 across the different outcomes for biodiversity condition, reflecting the diversity of measures for biodiversity at different scales from sites to landscapes (Table A4). A total of 1,617 assessments of each potential indicator (total N_i = 49) against each criterion (total N_c =33) were made for all the desirable outcomes for biodiversity condition: When we applied the selection criteria to all potential indicators, about half (52%) of the indicators for Desired Biodiversity Condition (DBC) 1 and a third (33%) for DBC 2 were discarded on the basis of quality. No indicators were discarded for DBC 4 as each indicator met our two-thirds 'rule of thumb'. The biodiversity indicators for all outcomes when assessed using criteria for five different categories (CR - conceptual relevance, n_c = 6; HD - historical dimension, n_c = 2; MQ – measurement qualities, n_c = 11; FI – feasibility of implementation, n_c = 6; PM – policy and management, n_c = 8) met 48 to 100% of the criteria in total (see Fig. 2, Smyth *et al.* 2009b). HD indicators had the lowest performance overall but, when combined with MQ indicators, had the highest for 'water remote areas'.

Appropriateness of most indicators, independent of the categories of selection criteria and the outcomes for desirable biodiversity condition, met between 47 and 88% of the 33 criteria (Table 3). Indicators measured at sites for assessing 'native vegetation' and 'no loss of biodiversity' outcomes performed marginally lower on CR. The same indicators also tended to perform marginally lower in terms of their usefulness for policy and management.

Table 3. Management objective, desired biodiversity condition outcomes with key biodiversity values and robust indicators for assessing biodiversity condition after applying selection criteria for each waterpoint management outcome for the Stony Plains Bioregion. The robustness of indicators for each outcome is shown as percentages of the total indicators and criteria met per outcome, where 33 selection criteria were used in total.

Management objective	**Retain or restore (if possible) the resilience of ecological communities/habitats for land types of the stony plains bioregion**		
Outcomes for Desired Biodiversity Condition from waterpoint intervention	**DBC1: Native vegetation typical of the Stony Plains communities maintained or restored**	**DBC2: Amelioration of decline in rare and regionally significant native species, populations and ecological communities**	**DBC 3: Mosaic of water remote areas**
Key Biodiversity Surrogates	• Vegetation cover • Plant diversity • Structural complexity • Naturalness (e.g., few, if any, non-indigenous biota and feral animals and pre-settlement kangaroo densities.)	• Endemic and threatened species and communities • Species that decline when grazed • Significant ecological communities (gypsum clay plains, stony plains with gilgais, breakaway hills, drainage lines and flood plains, arid ranges, stony tableland with sand mounds) • Pre-settlement densities of dingos • Negligible densities of foxes and cats • Negligible densities of introduced weeds	• Interconnected tracts of intact ecological communities
Robust Biodiversity Indicators	Within land types: • Mean (or variance) dry-period vegetation cover[1] • Frequency and cover of *Abutilon halophilum* at 1.5 km from water sources • Cracking Index of gilgais systems[3] • Presence of non-native invasive weeds • Total grazing pressure[4]	Within land types: • Preferred grazing area of all stock most of the time • Number of domestic stock per water • Presence[2] of feral herbivores • Presence[2] of dingos or foxes • Presence of non-native invasive weeds • Presence of terrestrial endemic and threatened species • Presence of aquatic endemic and threatened fauna • Status of regionally significant aquatic	Within land types: • % area remote from water by length of time • Density of artificial waterpoints by age

		communities of drainage lines an floodplains • Water quality • CUMSUM (cumulative sum of the deviation from long-term rain year mean) for rainfall • Seasonality of temperature and rainfall	
% of total indicators for case study region (n out of N)[a]	48 (10/21)	77 (14/18)	100 (2/2)
% of criteria met (n =33)[b]	65-85	47-85	82-88

[1] Dry period vegetation cover is the vegetation cover that is typical after long periods of little or no rain and measured in terms of the % cover, frequency distribution of cover, spatial pattern of low cover areas according to what is most suitable for the ecological community.
[2] The use of 'presence of' was based on the constraints of reliability, technical effort and cost to measure the indicator. We acknowledge that the technical workshop members pre-empted the role of the selection criteria but it had little effect on the overall assessment other then sequencing of a task.
[3] % cover of cracks >1 mm wide within a 2 m radius summed over 9 radii within each gilgai (clay depression)(Smyth and Brandle, unpublished data)
[4] Measured by presence of feral herbivores, kangaroo abundance, preferred grazing area of all stock most of the time, productive value of grazed areas, number of domestic stock per water and historical numbers of all stock and kangaroos per water.
[a] Proportion of indicators identified for the outcome that met two-thirds or more of the selection criteria for appropriateness (see Appendix 1)

[b] Number of selection criteria met out of a total 33 criteria for assessing appropriateness of ecological indicators (see Appendix 1)

2. Metric design

Australia leads the world with designs of biodiversity metrics for stewardships. Biodiversity conservation planners have embraced environmental stewardships as an additional tool for managing biodiversity on private holdings as they have provided a standardised approach for comparative assessment of existing ‘biodiversity significance’ and its potential improvement. More recently, these metrics have been modified to assess biodiversity condition. However, with all new advances come its detractors. Wildlife biologists, working at the science end of the science-planning spectrum, are concerned about metrics misleading biodiversity conservation decision-making because a single number hides important biological information that may be critical for decision-making. To date, the planners have used biodiversity metrics virtually unchecked but calls for a dialogue via the Australian Centre for Ecological Analysis and Synthesis (ACEAS) highlight the concern.

In summary, key generic findings of our review about biodiversity metrics applied for environmental stewardships (and biodiversity condition more recently) were:

- The metrics are applied either as standalone tools or are embedded within multimetrics for environmental stewardships.

- The general structure of metrics are based on assessing three properties: (i) the ‘biodiversity value’ of places under current management, (ii) ‘improvement in its value ‘(or public benefits) expected from management changes for a specified term and (3) the area or extent of management change. Assessments of the first two properties are context-dependent and are likely to change from place to place. This raises questions about the generality of metrics for comparative purposes, as ‘apples’ may not be compared with ‘apples’.

- Three broad surrogates are commonly used to assess existing biodiversity and its value at three scales. (i) measures of ‘biodiversity condition’ (the quality of critical habitat and other resources required by biota) at the site level, (ii) biodiversity of a site relative to its position in a broader landscape context, and (iii) conservation significance of biodiversity at a regional scale.

- The second parameter of ‘anticipated improvement in biodiversity’, is mostly estimated using averages or ranges for indicators of ‘biodiversity condition’ (vegetation condition) at ecologically similar ‘benchmark’ sites that have had relatively little human disturbance. Data are collected for the same indicators used to assess existing biodiversity value. This parameter has proven problematic to estimate such that some developers have dropped it completely.

- The use of biodiversity metrics in environmental stewardships is not evidenced-based as no stewardship programs in Australia have run long enough to monitor and evaluate biodiversity outcomes. In the interim, sophisticated and scientifically defensible designs are paramount.

- Good metric design is only as good as the stewardship design. Strategic planning (purpose, expected outcomes, management objectives, capacity) is essential for good metric and policy design. These steps help to identify biodiversity surrogates and indicators for measuring potential outcomes. Many stewardships and metrics are designed in isolation to other NRM planning initiatives with little strategic planning but this is improving (e.g., Grassy Box Woodlands Stewardship).

- Indicators/measures to assess ‘biodiversity values’ and predict improvements in ‘biodiversity condition’ are not critically reviewed for conceptual relevance, historical dimension, measurement quality and policy and management utility. Their efficacy often goes unchallenged for the sake of convenience.
- Benchmarking to show change in ‘biodiversity condition’ has been debated since the inception of biodiversity metrics. Key issues centre on the reliance of a relatively “natural” site for comparative purposes. This can be overcome by having many benchmark sites (e.g., BioMetric) but they must be reasonably homogeneous and the best on offer. Other issues relate to measurement quality.
- Most dissatisfaction with metrics is the lack of transparency in decision-making. Even if the most sophisticated scoring methods are used, metrics (and multimetrics) condense important information that may be valuable in making decisions but it does not support transparency in decision-making. If contested legally, they may not hold up in court as they are based on many untested assumptions about biodiversity and its dynamics.

In summary, key findings of metric design specific to the arid rangelands were:

- Vegetation productivity drives animal responses and ecosystem services in the arid rangelands, consequently, vegetation condition will be an important property of a ‘biodiversity significance’ assessment.
- Many remote sensing techniques exist for measuring vegetation cover and its temporal changes but the efficacy of these approaches to predict biodiversity remain equivocal.
- Benchmarking has real potential in the arid rangelands because of its relative intactness.
- Grasslands and low open shrublands of the arid rangelands make it difficult to build on existing indicators used in biodiversity metrics.
- Measures of landscape context used in existing biodiversity metrics will be of little value in the arid rangelands as it is relatively intact compared to areas where most metrics have been applied. Instead, measures to water remoteness may prove a more useful way of contextualising a site in the overall landscape.
- Rainfall seasonality via soil moisture drives vegetation productivity. Rain varies decadally, annually and daily from place to place; it is stochastic in every respective and highly unpredictable. No metric has been (ever could be) developed that captures such dynamics as vegetation growth, ecological processes and animals that pulsate erratically to rainfall.

3. Waterpoint field experiments

Plant identification for the last two visits remains to be confirmed by the SA Herbarium, so analyses are not presented at this stage. Preliminary results on standing herbage for one field visit in the calibration sites (March 2007) as part of the rainfall seasonality work are presented. More detail can be found in Davies *et al*. (manuscript). We present these vegetation results recognising that they may change when the second sample after small rains in October 2008 is included.

Fauna sampling is incomplete and may change with further results and more detailed analyses. Nevertheless, we present some preliminary results on the small ground mammals and reptiles. A full report on the preliminary fauna results is attached in Brandle (2009).

Vegetation

(i) Standing herbage (one sample) and soil seedbank

A total of 108 species were recorded standing and in the germinable seedbank. All of these species were indigenous natives with the exception of three exotic ephemerals (*Rostraria pumila*, *Polygonum aviculare*, *Centaurium spicatum*) which were confined to only 30% of plots. The majority (81%) of indigenous species were short-lived, 70.5% being ephemerals and 10.5% short-lived perennials. Only 17 species of longer-lived perennials were recorded, consisting of ten shrub or dwarf shrub species, six grass species, and one sedge.

Of the 108 species recorded overall, only 63% were observed in the above ground herbage due to the dry conditions in the field (Fig 5). In particular, only 65% of a total of 54 ephemeral forbs were detected in the field. In contrast, all but two of the 28 grass species were detected in at least one quadrat over the two sites, despite the dry conditions. No new short-lived or longer-lived perennial species were found in the seedbank that were not already observed above the ground overall. However, new species for individual plots were found that were not observed in the field. Three new short-lived ephemerals and an undescribed species of *Poa* grass was discovered.

Short-lived and non-native forb species were greater in the seedbank than was observable in the field (Fig 5A). Of interest was the paucity of perennial species in the germinable seed bank. Two species listed as significant at the national and state level (Briggs and Leigh 1996; Barker *et al*. 2005) and a further species listed as rare at the state level (Barker *et al.* 2005) were germinated out of the soil seed bank.

(ii) Standing herbage species cover compared with soil seedbank

Native vegetation covered 18.2% of the ground and litter 10.5% (Fig 6A). Exotic species were absent from the standing herbage. Longer-lived perennial shrubs and dwarf-shrubs dominated the vegetation (11.3% cover), with *A. nummularia* the main shrub (10.4%). Perennial grasses and were very sparse contributing only 1.6%. Similarly, short-lived grass and forb cover were very sparse at 3.2% and 2.2% cover respectively.

Of the seed that germinated, 83% was from short-lived ephemeral forbs, and 13.2% from short-lived grasses and sedges (Fig. 6B). Only 1% of seed germinating was from long-lived perennial shrubs and dwarf shrubs, and 1.6% from perennial grasses. Exotic species made up only 0.6% of all germinating seed.

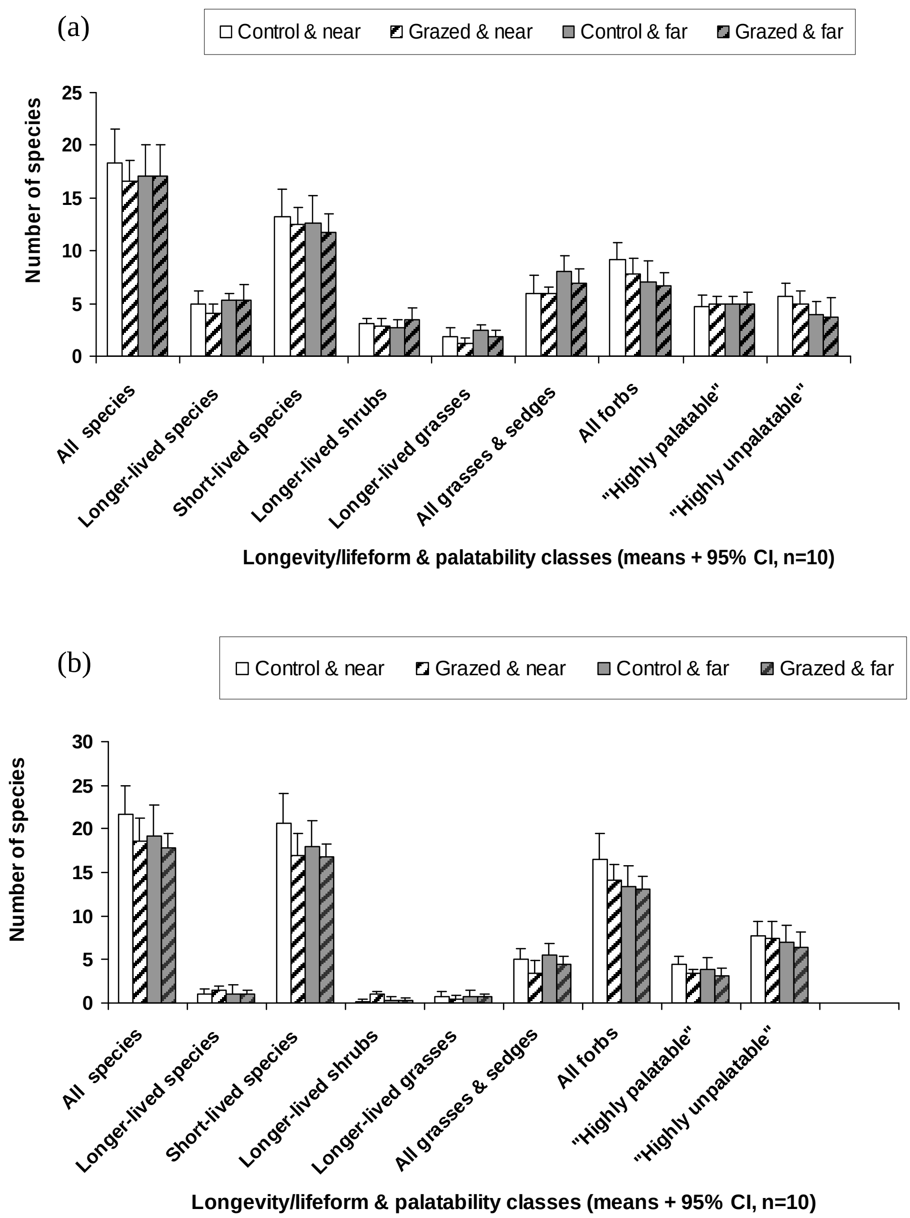


Figure 5. Species richness for different longevity/lifeform and palatability classes, and for different treatments: A) Above-ground herbage, B) Germinable soil seedbank for grazing exposure (on-going grazing -grazed, relaxation-control) and grazing intensity (relatively heavily grazed-near, lightly-negligible grazing-far) for the calibration sites during one visit, March 2007

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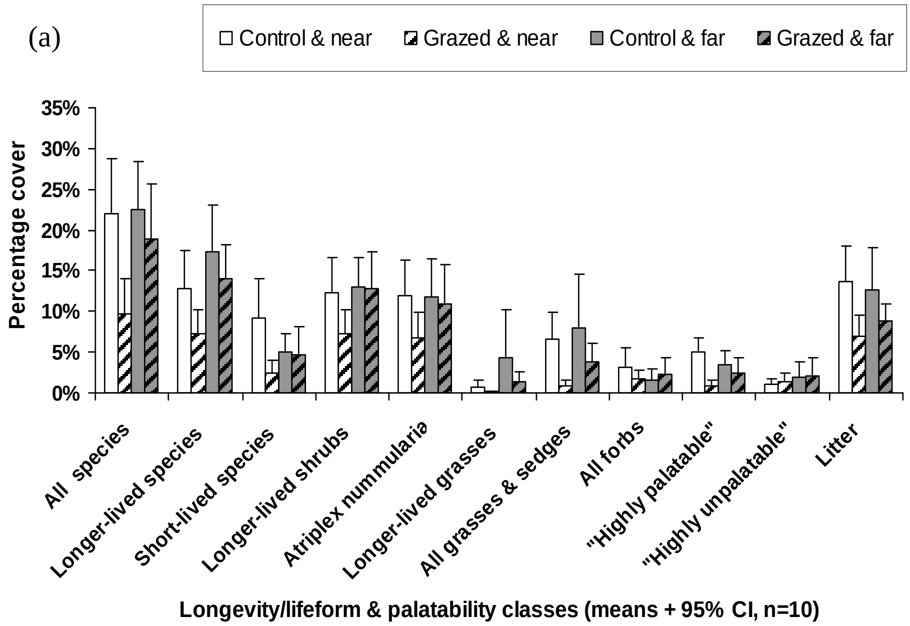
(a)



(b)

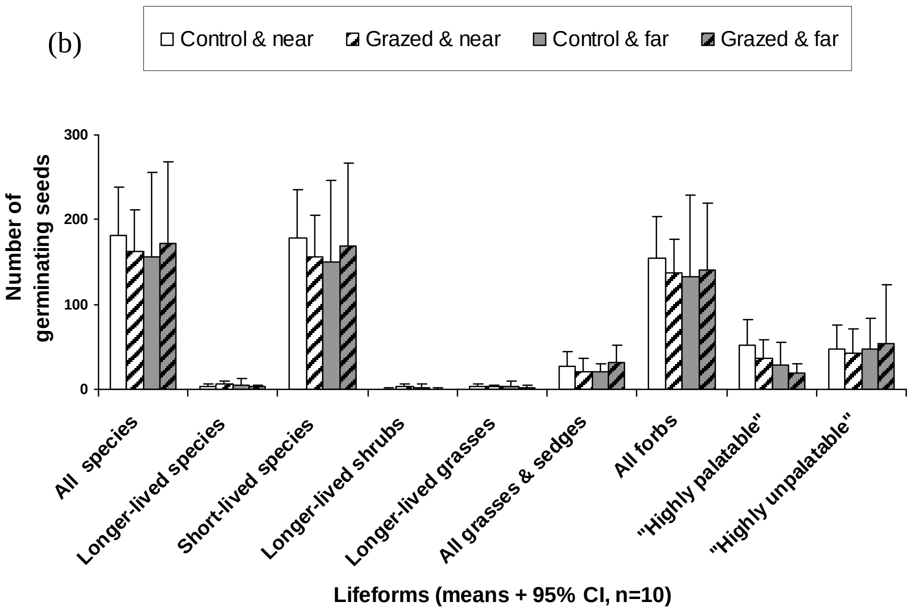


Figure 6. Abundance measures for different longevity/lifeform and palatability classes, and for different treatments: A) Percentage cover of above-ground herbage, B) Density of germinating seed from the soil seedbank for grazing exposure (on-going grazing -grazed, relaxation-control) and grazing intensity (relatively heavily grazed-near, lightly-negligible grazing-far) for the calibration sites during one visit, March 2007.

(iii) Responses to grazing exposure and intensity

The assumption of a grazing gradient was upheld for standing herbage ($p < 0.05$) in standing herbage between sites near to watering points compared with water remote sites, in relation to total plant species composition (see Fig. 5 in Davies *et al.* manuscript), cover of all species (Fig. 6, in Davies *et al*. manuscript), long-lived species composition (Fig. 7, in Davies *et al.* manuscript), short-lived species composition (Figure 8), and short-lived species cover (Fig. 9, in Davies *et al.* manuscript) for one sample in March 2007 after a long dry period. In comparison, no significant differences were found in response to either of these variables in relation to species composition and seed density in the soil seedbank. No significant cross-fence differences were found for any of these attributes.

Analysis using linear mixed-effect models indicated that no significant differences in soil seed-bank species-richness and seed-density; and above-ground species-richness and percentage-cover between treatments (Figs 2-3, Davies *et al*. manuscript). This applied to soil seed-bank species-richness and seed-density; and above-ground species-richness and percentage-cover. This was also true for the average height and area of individual *Atriplex nummularia* shrubs.

An interesting observation was that the ephemeral daisy, *Rhodanthe stricta*, which was absent from the soil seed bank (and above ground herbage) at all relatively heavily grazed sites subject to long-term, on-going grazing at the Braeden sites.

(iv) Reliability of species richness as an indicator for condition assessments

No significant correlations existed between above-ground species richness and soil seed bank species richness. The only exception was species richness values for above-ground forbs and “highly palatable” species, both of which were positively correlated with the species richness of “highly palatable” species in the soil seed bank.

Mammals

A total of eight mammal species were recorded across the 80 sample arrays in the 4 study areas. Only four of those species were common to all 4 study areas (Table 4). Three species were recorded twice or less and were excluded from any further analysis as vagrants of the gibber gilgai habitats sampled within the 4 study areas.

For the following analyses species data from the 5 replicates within each of the 16 sites have been combined. The maximum number of species recorded at any site was 5 (at 1 site) over a 6 night trapping visit, the average number of species being 1.5 with a median of 1 for all sites and visits (n=200).

(i) Species richness and grazing gradients

No trends were discernable for combined species richness (for replicates) in response to different grazing exposure and intensity for the intervention and calibration sites. (see Figs. 1 and 2 in Brandle 2009). Species richness was noticeably lower during the first visit to the two study areas. When data were combined for grazing exposure to test the assumption of a grazing gradient, there was no apparent difference between extremes for gradient during any sampling period at the intervention (Macumba) and calibration sites (Todmorden) (see Figs. 3 and 4 of Brandle 2009).

Table 4. Mammal species abundances recorded within each study area. Braeden's and Oodnadatta had two visits, whilst Duckhole and Harry's had three with one visit still required

Species	Braeden's Paddock	Oodnadatta Common	Duckhole Bore	Harry's Tank	Total
Sminthopsis macroura	39	34	218	103	394
Leggadina forresti	9	8	34	43	94
Sminthopsis crassicaudata	16	7	22	24	69
Planigale gilesi		7	11	20	38
Planigale tenuirostris	4	8	1	6	19
*Mus musculus**		1	1		2
Pseudomys australis				1	1
Pseudomys hermannsburgensis				1	1
Total	68	65	287	198	618

* non-native invasive species

(ii) Productivity and grazing gradients

Relative abundance of small mammals from trapping results was used to compare productivity. No trends in a range of abundance measures are discernable in response to different grazing exposure and intensity for the intervention and calibration sites or by visits (Brandle 2009). Productivity was much greater during visit 3 at the intervention sites, which appears to be a response to rainfall (Fig. 4; Fig.15 in Brandle 2009). The significant differences between the first and other visits (see Fig. 13 in Brandle 2009) to these sites may be related to a very dry 2006 prior to the first sample in June 2007 as most dasyurids (which comprised the majority of individuals) breed in winter and spring and their populations would not have had the opportunity to responded to the February, March and May rainfall during 2007, whereas production following these rainfall events could have provided impetus for breeding before visit 2 in November 2007.

(iii) Individual species abundance and grazing intensity (near-far) comparisons

Comparison of individual species highlighted difference in response to presence of permanent waterpoints and associated impacts. These analyses have been restricted to the intervention (Duckhole and Harry's Tank) sites for before (June and November 2007) the intervention (September 2007) because of the lower abundances in the other study areas.

The main observable trends were: *Leggadina forresti* more abundant at lightly grazed sites, *Sminthopsis crassicaudata* and *Sminthopsis macroura* more abundant at relatively heavily grazed sites. There was week support for these trends in some sites within the pre-treatment visits, but support was weaker in the single post-treatment visit (refer to Table 3 and 4, Brandle 2009).

(iv) Relaxation from Grazing Analyses (Intervention sites only)

As only one sample has been undertaken post grazing relaxation (compared with 2 pre) only a preliminary analysis can be performed. Given the similarities in average abundance between Visit 2 (on-going grazing) and Visit 3 (grazing relaxation) only these two visits will be included in the following analyses. Earlier analyses indicate that mammal species richness is unlikely to show a measurable response. The differences in responses in total relative abundance vs. averaged relative abundances indicate more detailed analysis is required.

The data indicates a significant increase in small mammal abundance following cattle removal from all sites (including relatively heavily and lightly grazed sites) (Fig. 7). The difference between pre and post intervention samples in the control sites where grazing was maintained were not significantly different, despite substantially higher rainfall prior to the post-intervention visit when compared with the pre-intervention visit (Fig. 8).

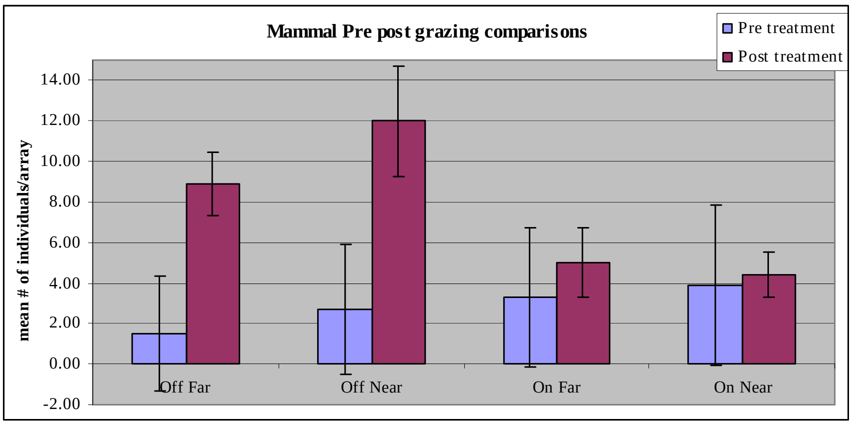


Figure 7. Mean small mammal abundance pre and post cattle removal for near and far sites. Pre treatment – on-going grazing, Post treatment –cattle removed. 'Off' indicates waterpoint closure = Post treatment. 'On' – Pre-treatment

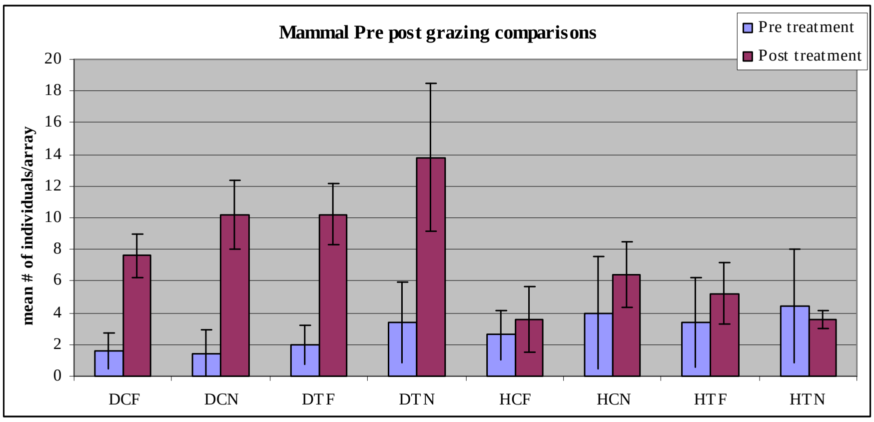


Figure 8. Average small mammal abundances pre and post cattle removal for near and far site. Pre treatment – on-going grazing, Post treatment – cattle removed. (Grazing relaxation sites (treatment) - D prefix, on-going grazing sites (control) – H prefix)

Reptiles

A total of 20 reptile species were recorded across the 80 sample arrays in the 4 study areas. Only four of those species were common to all 4 study areas (Table 5). Seven species were recorded twice or less and were excluded from any further analysis as vagrants of the gibber gilgai habitats sampled within the 4 study areas.

Table 5. Reptile species abundances recorded within each study area. Breaden's and Oodnadatta had two visits, whilst Duckhole and Harry's had three with one visit still required

Species	Breaden's Paddock	Oodnadatta Common	Duckhole Bore	Harry's Tank	Total
Tympanocryptis tetraporophora	24	124	95	53	296
Menetia greyii	22	52	153	16	243
Ctenotus olympicus	44	16	10	19	89
Lerista muelleri	12	11	41	6	70
Heteronotia binoei	19	13	9	15	56
Ctenotus strauchii		4	24	6	34
Cyclodomorphus venustus	2	3	22	2	29
Diplodactylus tessellatus	4	4	6	9	23
Tympanocryptis intima		1	16	1	18
Suta suta	2	5	2	1	10
Delma australis	1	1		3	5
Diplodactylus byrnei		3	1	1	5
Eremiascincus richardsonii	2			2	4
Pogona vitticeps	2				2
Ramphotyphlops endoterus				2	2
Rhynchoedura ornata	1		1		2
Ctenotus schomburgkii	1				1
Ctenotus sp.		1			1
Ctenotus taeniatus				1	1
Pygopus lepidopodus	1				1
Total	137	238	380	137	892

Species data from the 5 replicates within each site have been combined. The maximum number of species recorded at any replicate was 7 (at 1 site) over a 6 night trapping visit, the average number of species being 3.2 with a median of 3 for all spring and autumn samples (n=160). The intervention sites were sampled in winter (June 2007) on visit 1. The maximum number of species recorded at any one replicate was 3 and the average was 0.95 with a median of 1.

(i) Species richness and grazing gradients

As for mammals, no trends are discernable for combined species richness (for replicates) in response to different grazing exposure and intensity for the intervention and calibration sites. (see Figs. 16 and 17 in Brandle 2009). The differences between visit 1 (June 2007) and visits 2 and 3 (November 2007 and March 2009) at the intervention sites (Macumba) can be attributed to the cooler conditions during winter. At the calibration sites, it can be attributed to the sudden cold change toward the end of March 2007 that significantly suppressed reptile activity.

(ii) Productivity

As for mammals, no trends in a range of abundance measures are evident in response to different grazing exposure and intensity for the intervention and calibration sites or by visits (Brandle 2009). The higher abundances observed in the spring sample (November 2007 -Visit 2) at the intervention sites may reflect higher levels of activity for reptiles at this time of the year as most reptiles breed at this time of the year. By autumn (March 2009 - Visit 3), breeding activity has ceased and mature populations of some species suffer significant die back over summer. The very young hatchlings that are present at this time of the year are less mobile and more timid, so sampling may under-represent the population when compared to a spring sample.

(iii) Individual species abundance and grazing intensity (near-far) comparisons

Comparison of individual species highlighted difference in response to presence of permanent waterpoints and associated impacts. These analyses have been restricted to the intervention (Duckhole and Harry's Tank) sites for before (June and November 2007) the intervention (September 2007).

A number of species appear to show response to the proximity of waterpoints. Those with significantly ($\alpha = 0.05$) higher abundance in lightly grazed sites: Pre treatment - *Ctenotus olympicus*, *Heteronotia binoei* and Post treatment - *Cyclodomorphus venustus*. One species, *Ctenotus strauchii*, was significantly more abundant in relatively heavily grazed sites both Pre and Post treatment.

The lack of significance in trends for two of the species across all visits highlights the variability of fauna responses in the region over time, exacerbating the problem of small sample sizes. Comparison of mean abundances was precluded because of low numbers for species highlighted through the Chi squared analyses.

(iv) Relaxation from Grazing Analyses (Intervention sites only)

In contrast to the small mammal data, the reptile abundance following intervention did not increase at sites. Instead, abundances decreased (Fig. 9). This most likely reflects timing of the samples. The pre sample in November took place during the most active period in most reptile species annual activity cycle where the post sample was in March when activity has decreased substantially. However, the only sites where these decreases were significant coincided with control near and far sites where there was no relaxation of grazing. (Fig. 10). The patterns in reptile response do not closely mimic the rainfall data in each sample area indicating that release from grazing may be partially responsible for the significant decrease in abundance where cattle grazing was maintained.

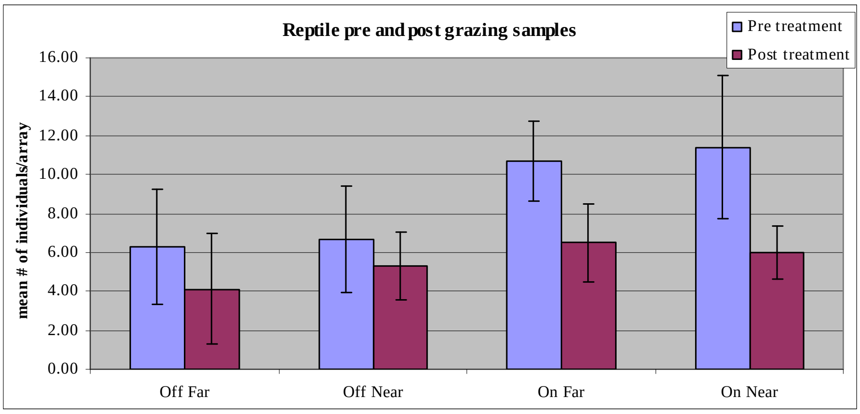


Figure 9. Mean reptile abundance pre and post cattle removal for near and far sites. Pre treatment – on-going grazing, Post treatment –cattle removed. 'Off' indicates waterpoint closure = Post treatment. 'On' – Pre-treatment.

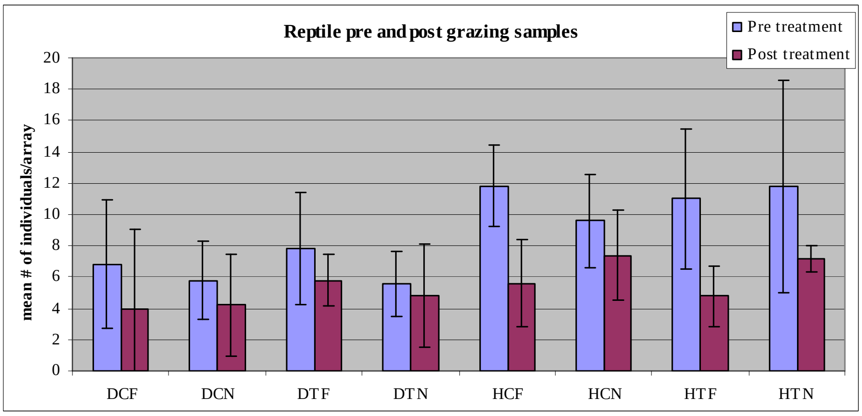


Figure 10. Average reptile abundances pre and post cattle removal for near and far site. Pre treatment – on-going grazing, Post treatment – cattle removed. (Grazing relaxation sites (treatment) - D prefix, on-going grazing sites (control) – H prefix)

4. Rainfall seasonality

Overall it was found that of all seed germinating, over twice as much (69%) germinated under "winter" conditions than under "summer" conditions (31%). This was due to the majority (78%) of forb seedlings germinating in winter, as did 64% of seedlings of long-lived perennial shrubs. Ninety eight percent of all exotic seed, which germinated did so under "winter" conditions as did 66% of seedlings of "highly palatable" species, and 66% for "unpalatable" species. However, 81% of grass seed, which germinated emerged under "summer" condition. A full species list of 'winter' and 'summer' responses can be found in Tables 1 and 2 of Davies *et al.* (manuscript).

5. Spatio-temporal patchiness of gibber-gilgais vegetation data

Detailed results of this work are presented in the attached report by Bastin (2009).

Spatial analysis

Difficulties in precisely classifying gilgais and verifying results on the ground prevented accurate maps of gilgai shape and location. Thus the spatial statistics presented are only indicative. Nevertheless, there were considerable differences in gilgai-type between sites (e.g. size and separation, see Figure 11) and these differences need to be accounted for when interpreting field data.

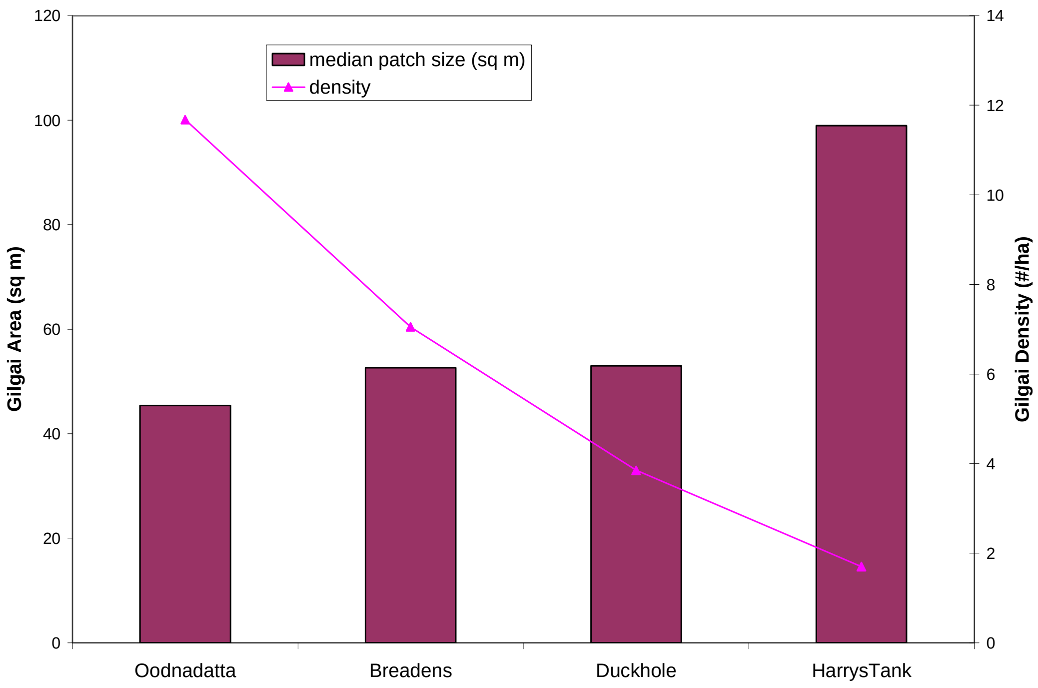


Figure 11. Median size (sq m), and density (number per hectare) of gilgais at the four study sites.

In summary:

- Descriptive spatial statistics indicated that the analysed area surrounding the Oodnadatta field sites had a higher density (number per ha) of smaller gilgais (both mean and median area) that were located closer together, compared with treatment areas at other sites.
- Harry's Tank sites had a relatively low density of larger gilgais that were located further from each other. This analysis area contrasted most strongly with the Oodnadatta area.
- Analysis areas surrounding field sites in the Braeden's area had intermediate characteristics for the spatial distribution of classified gilgais. Classification results were less certain for this analysis region.

- There was a strong contrast for the two areas analysed at Duckhole (Treatment areas close to and further from water). The 'Near' area had a lower density of larger gilgais which were further apart compared with the 'Far' area.

A separate analysis based on the lacunarity index suggests some differences in the uniformity of gilgai distribution amongst sites. (The lacunarity index provides a measure of the distribution of 'holes' or 'gaps' in a spatial grid [in this case, classified gilgais]). Analysis areas close to water at Oodnadatta, Treatment sites at Braeden's and the complete set of Harry's Tank sites were relatively 'gappy' in terms of gilgai distribution. Gilgais appeared to be more uniformly distributed for polygon areas further from water at Oodnadatta, on Braeden's Control sites and at Duckhole (Treatment) sites.

Temporal analysis

Seasonal changes in vegetation cover between 1974 and 2006 were reported using Landsat imagery (50-m pixel resolution to 1988 and 25-m resolution since). Cover change was calculated for two mapped areas of the 'chenopod (*Atriplex / Sclerolaena / Maireana* spp.) low open shrubland over sub-shrubs and grasses' unit (SA Department of Environment and Heritage's survey of the Stony Plains bioregion). One polygon (710 km^2) was located on Todmorden station and included both the Oodnadatta and Breadens field sites. The other was on Macumba (2285 km^2) surrounding the Harry's Tank and Duckhole field sites.

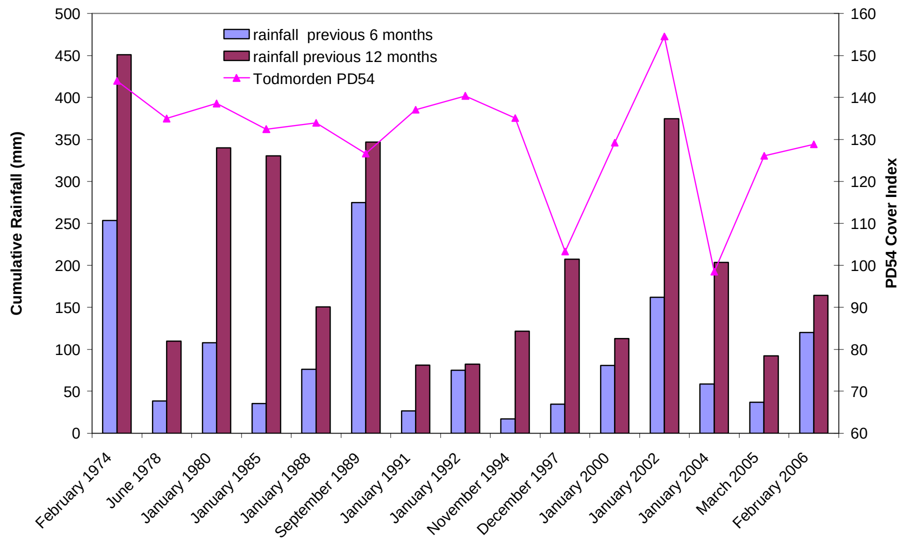


Figure 12. Spatially averaged vegetation cover as indicated by the PD54 index for the chenopod – low open shrubland vegetation type on Todmorden station. Rainfalls recorded at Oodnadatta for the 6 and 12 months prior to acquisition of Landsat images are also shown.

In summary:

- Historical levels of vegetation cover were indicated by two indices, PD54 and STVI (stress-related vegetation index). The PD54 index has been used extensively in central Australia to monitor cover. STVI was developed more recently and can only be used with Landsat TM data (i.e. image dates since 1989).
- Qualitative assessment showed that the PD54 index more reliably indicated vegetation cover for the Oodnadatta region (compared with STVI). Both indices were validated by comparing spatial patterning of index values with visual interpretation of hyper-

spatial true-colour imagery (i.e. Ikonos and Quickbird images). PD54 values extracted for the locations of field sites better conformed with cover levels indicated by landscape photos than was the case for the STVI index.
Some cover data were collected at field sites but measurements were restricted to gilgais. This meant that the data were not sufficiently extensive to validate either index at landscape scale. Robust validation of vegetation indices requires more extensive ground-based measurement of cover stratified for the different landscape components (gilgai, gibber pavement, gidgee-lined watercourses, and other cover types). These measurements should be contemporaneous with the Landsat image used for comparison.

- Cover trends were determined for mapped areas of the chenopod – low open shrubland vegetation type on Todmorden and Macumba. Available Landsat data were more continuous through time for the Todmorden area and results are summarised here (Figure 12) in preference to the temporal trend for Macumba. Spatially-averaged cover (as indicated by PD54) remained relatively stable between 1974 and 1994, and then was much more variable over the next 10 years. Cover declined appreciably in late 1997 then increased to the highest level recorded following wetter years in 2000 and 2001. Cover then declined over the next two years (i.e. early 2004) to match the low level present in late 1997.
  It is likely that seasonal variation in rainfall accounted for most of the change in cover.
- Despite dry conditions in 2004 and early 2005 (i.e. prior to the March 2005 image date), cover indicated by the PD54 index increased to be slightly below the level present through the 1980s and early 1990s. Average cover in early 2006 was similar (to 2005) following approximately median rainfall for the preceding 12 months. This suggests that the chenopod – low open shrubland vegetation type on Todmorden retains good resilience in terms of the capacity of vegetation cover to respond to rainfall.
- It was not possible to make ecological sense of trends in cover based on the stress related vegetation index. The index probably does reliably indicate moisture stress but the spatial patterning of index values does not closely relate to that of cover. Index values increased and were at their maximum (i.e. least stress) on the Todmorden area of chenopod – low open shrubland in the early part of this decade at the end of a notably wet period when vegetation stress should have been minimal.

This historical pattern provides important context for interpreting vegetation data collected on the ground.

DISCUSSION

The use of artificial waterpoints by livestock degrades habitats and can reduce biodiversity over time in the arid rangelands. Waterpoint intervention (e.g., closure) has been proposed as a management strategy benefiting arid biota through grazing relaxation (Landsberg *et al.* 1997; Noble *et al.* 1998; James *et al.* 1999). However, it is well recognised that arid environments are driven by a cascade of spatio-temporal, patchy dynamics initiated by rainfall. At the beginning of this report, we posed the question as to whether it is possible to assess biodiversity outcomes from waterpoint intervention in such changing patchy landscapes of the arid gibber-gilgai biome. Although our field studies are not complete, we are cautiously confident that it is possible provided the assessment is strategically planned and the on-ground measures of change are scientifically defensible. Despite their appeal at the planning level, metrics will need to be sophisticated and incorporate stochastic processes to be useful in waterpoint management. How then can outcomes from waterpoint manipulations be assessed for biodiversity conservation planning purposes in such dynamic systems? If this is possible, then opportunities exist for managing resilient, relatively intact arid landscapes for grazing and biodiversity protection in a dynamic way by relaxing patches of vegetation in space and time. We discuss these opportunities in relation to our results in terms of the science and waterpoint management for biodiversity outcomes. We recognise other socio-economic outcomes should be given equal attention (pastoral values paper) but it was outside the scope of this study.

Science underpinning waterpoint interventions and biodiversity outcomes

Standing herbage (one sample) and soil seedbank

The present study indicates the importance of rainfall seasonality in driving gilgai vegetation and fauna productivity in arid central Australia. It also demonstrates the resilience and uniqueness of the gibber-gilgai biome for conserving a range of ephemeral wetland plant species; communities which are critical for animal survival. Although strong indications, both of these are inconclusive due to the preliminary nature of our analyses.

Despite long-term grazing which historically has been heavy in places, the gilgai communities have retained a relatively high species richness (average >31 species per 100 m^2). No significant differences were found in above or below-ground species richness between areas which were relatively heavily grazed compared to lightly grazed areas, and between areas which were consistently grazed compared with areas rested from grazing for more than 15 years. The same applied to vegetation cover and seed density. The widespread presence of Oodnadatta saltbush *A. nummularia* ssp. *omissa* appears to provide resilience to the gibber-gilgai plant community. This shrub is relatively unpalatable and appears to provide protection to more palatable ground strata species. The positive correlation between Oodnadatta saltbush cover and “highly palatable” species cover and the positive correlation of total long-lived shrub cover (predominantly Oodnadatta saltbush) with total grass/sedge cover and short-lived species cover is further evidence of the functional role of Oodnadatta saltbush in the biome. Both Oodnadatta saltbush and that of total long-lived shrubs generally, were also found to be positively correlated with above-ground, long-lived grass species-richness.

Another attribute contributing to the conservation significance of arid zone gilgai landforms is that they provide ephemeral wetland habitats that are productive following minor rainfall events (<25 mm) because of the large impermeable catchments that the surrounding gibber pavements provide. They also appear to be relatively resistant to weed invasion in comparison to ephemeral arid wetlands in riparian systems, which can be rapidly colonised by weeds (e.g.,buffel grass) due to the presence of continuous belts of suitable habitat and the effect of stream flow as seed dispersal agent. In contrast, gilgais occur as more isolated, discrete wetland chains separated by barren gibber plains. Seventy percent of quadrats were free of exotic species, and such species made up only 0.6% of all seeds in the germinable soil seedbank.

Previous studies of semi arid rangelands have found that the presence and composition of the perennial grass component of the ground stratum is a reliable indicator of range condition (e.g., Ash *et al.* 2001). Evaluating biodiversity condition by measuring perennial grass abundance and composition cannot be used for gibber-gilgai vegetation in arid central Australia. We found that of the 108 indigenous species recorded at the study site, only six were perennial grasses with high variability in numbers of perennial grass species between similarly grazed adjacent sites. More interesting, perennial grass cover averaged only 1.6% and was also highly variable. Unlike *Atriplex vesicaria* in the southern chenopod rangelands, Oodnadatta saltbush is not a reliable indicator. No significant differences in cover were found between gilgais in heavily grazed compared with lightly grazed areas, or between consistently grazed areas compared to areas rested from grazing. This also applied to the average height and area of individual Oodnadatta saltbush plants. Oodnadatta saltbush has an abundance of meristems on the woody stems and can readily regrow after complete defoliation and also appears to be long lived, suggesting it can tolerate heavy grazing at the intensity levels experienced at our study sites. Investigation of nearby similar landform and soil type that were within 1 km of historically heavily utilized waterpoints, indicate that this perennial shrub layer can be completely removed from the landscape and that its return is unlikely under contemporary grazing management or complete removal of grazing.

We found short-lived plant species to be similarly important in arid zone gilgai communities. Eighty one percent of all species recorded from the study site were found to be short-lived and 71% ephemeral. This included the four plant species of conservation significance recorded during the survey (*Brachyscome ciliaris* var. *subintegrifolia, Brachyscome eriogona, Sclerolaena blackiana, Poa* sp. nov.) all of which were ephemeral. This indicates the importance considering ephemeral species when studying water-focussed grazing in arid rangelands. The monitoring of grazing impacts on ephemeral species in arid rangelands (either directly or through surrogate measures) is important given the finding of previous researchers that certain arid ephemeral species appear to be grazing intolerant (Landsberg *et al.* 1999; Nicol *et al.* 2007). Our study found 34% of ephemeral species occurring on arid zone gilgais were recorded only from sites with little grazing exposure (grazing relaxation) or low grazing intensity. Many of these had been identified in other studies as 'decreasers' under heavy cattle grazing in other plant communities at other arid rangeland locations (Appendix 1, Davies *et al.* manuscript).

The only potential indicator species of long-term changes to biodiversity condition in arid zone gibber-gilgai vegetation at this stage of the work was *Rhodanthe stricta.* It was absent from the germinable soil seed bank at all continually, long-grazed, "near-to-water" plots, but common in at least 80% of quadrats, which were either water remote, infrequently grazed, or were near to recently constructed watering points (~7 yrs). This species was found to be common component of the germinable soil seedbank at these sites, constituting 6% of all seed in the soil seed bank

(Davies *et al.* manuscript). It is absent in dry times, so as an indicator it would only be detected after rain or if germinable seed was collected. Our work also found that measuring above-ground plant species richness, in dry times, provided an unreliable absolute indicator of plant biodiversity at a site. It was found that of 108 plant species recorded from the combined germinable soil seed bank data and the above ground herbage data, 37% were not observed above ground in any of the quadrats. On average, only 55% of all species recorded from each plot was recorded above ground. Above ground survey also failed to detect three of the four significant plant species mentioned above. The timing of rainfall can also have a bearing on above-ground species richness and composition. Species richness in the arid rangelands is dynamic across landscapes, making it an unreliable indicator of biodiversity. This is probably due to the great range of gilgai sizes (2-30m in diameter), soil variations, the differing extents to which water flows into gilgais from adjacent impervious areas, and metapopulation dynamics of poor dispersers.

An alternative method for ranking gilgai area in terms of plant species richness is the recording of numbers of "highly palatable" species ("**Pal**" species in Appendix 1, Davies *et al.*, manuscript) or longer lived species ("P" in Appendix 1, Davies *et al.*, manuscript) in the above-ground herbage, again using quadrats. The present study found significant correlations between these above-ground measures, and combined above and below-ground total species richness. Where interest is in the relative species richness for a specific longevity and lifeform class of plant (e.g., to evaluate different areas for fauna habitat), field survey of above-ground herbage is also of value even if undertaken in dry times as shown by our preliminary results for one visit.

Fauna

Preliminary 'ground dwelling' vertebrate fauna results indicate that there were no significant differences in the fauna community composition between sites and that current levels of grazing intensity did not consistently affect species richness and relative abundance. At this level of biotic organisation, there does not appear to be a grazing intensity effect as measured by two extremes of grazing. However, for individual species, there were some preliminary significant trends within the reptiles. Two species were more abundant in lightly grazed sites whilst one was more abundant in relatively heavily grazed sites close to waterpoints. Further analysis of these responses will be conducted following the final field sample.

Early statistical comparison of pre and post intervention at the Macumba sites indicated that there were significant responses to both mammal and reptile productivity following relaxation of grazing. When these were combined for the total animal productivity, average abundance showed a significant increase in fauna following grazing relaxation in the more heavily grazed sites close to waterpoints. This indicates an interaction between grazing exposure and intensity resembling a 'pulse'-like response to grazing relaxation (following Underwood and Chapman 2003). Fauna declined in average abundance compared with the first sample in sites where grazing was maintained. Mammals responded with a significant increase in abundance following cattle removal in sites both near and far from the waterpoint, suggesting that grazing exposure may play a greater role than intensity in affecting mammal numbers. As semi-permanent water during the hottest months becomes available throughout the landscape, grazing pressure is released near artificial waterpoints and becomes diluted throughout the landscape. Alternatively, mammals could be reflecting responses to vegetation productivity as the summer rainfall was relatively higher in the post-intervention sites. In the pre-intervention sites, rainfall occurred in autumn favouring ephemeral recruitment (Davies *et al*. manuscript). Summer rains favour grass recruitment and the abundance of seeds may have increased mammal abundance. Where there was no relaxation of grazing, reptiles responded with a significant decrease in abundance post intervention at both near and far sites which may reflect a response in the area where grazing was relaxed when compared to where grazing was maintained. For both mammals and reptiles, the results are inconclusive because of their preliminary nature. More in depth analyses are required to tease out the complexities associated with environmental and temporal differences across the sample areas.

Due to the preliminary nature of the fauna data, we do not discuss any ecological and environmental relationships or whether the results are reflecting a statistical rather than biological result. In terms of our experimental design, we found studying waterpoint interventions on a working property challenging. Both pastoral companies were extremely supportive of our work and managed their stock to maintain our experimental design. However, with one of set of control sites, it had such little rain that the forage was very low and cattle were losing condition. The cattle were removed so these sites became a treatment for grazing relaxation. At another set of sites which did have cattle removed, the rains in summer were sufficient to create a series of semi-permanent waterholes along a drainage line and the cattle wandered from a control site (on-going grazing) back onto the treatment site. Consequently, this set of sites became a control which was not ideal for our experimental design in that all of our control sites were now clustered together, well away from the treatment sites instead of being spatially paired.

Landscape patchiness of gibber-gilgai biome

The spatial analysis has shown that satellite imagery of very high spatial resolution can characterise landscapes in terms of their gilgai distribution. It thus represents “proof of concept” for quantifying gilgai typologies of variable landscapes to provide broader spatial context for field data collected at a restricted number of ground sites. Pixel resolution (~1 m) was adequate to distinguish gilgais. Spatial extent (>100 km^2 for all site areas combined) was also sufficient to provide a reasonable sample of the variability in gilgai types in the Oodnadatta region. Relative remoteness hindered repeat field access to improve classification accuracy to the level where gilgais were mapped with suitable precision. This limitation could be improved with greater effort and would be assisted with increased accessibility. Building an improved typology (based on spatial analysis of gilgai characteristics) logically follows from satisfactory discrimination of gilgais. The long-term trend analysis of vegetation cover of the gibber-gilgai biome on Todmorden retains good resilience in terms of the capacity of vegetation cover to respond to rainfall.

Strategic waterpoint management for biodiversity outcomes

We developed for the first time a biodiversity assessment framework (based on condition) and then applied it to our case study region to demonstrate strategic planning for biodiversity outcomes from waterpoint intervention. Based on the expected outcomes, we were able to identify a robust set of indicators suitable for assessment at the planning level. They were grounded in biological science, historical dimension, sound measurement quality and had relevance to planning and policy. The approach also links into existing biodiversity conservation planning initiatives. The next challenge is the evaluation of outcomes as evidence of the merit of waterpoint management.

An important aspect of any program assessing biodiversity outcomes via condition assessments is that the results relate to the regional biodiversity management objectives by guiding effective management action including communicating how the method works, stimulating management research, social learning (Knight *et al.* 2005) and adaptive management (Noble and Brown 1997). Our framework achieves these qualities via the priority assessment step, where objectives for assessing biodiversity condition are linked to statements about outcomes for desired biodiversity condition and management action thereby effectively linking back to the purpose of the assessment in the first instance. Overall, our approach is grounded in biology and regional biodiversity conservation planning. It could also incorporate socioeconomic and political contexts of implementation, should that be an important consideration in a biodiversity condition assessment for a region. The use of selection criteria not only communicates what qualities were important in assessing biodiversity condition, but it also shows transparency in how they were selected. As far as we are aware, no previous work has explicitly evaluated indicator selection when developing condition scores. A significant finding of our work was that different indicators would most likely be needed for different planned outcomes of biodiversity condition. Nevertheless, the uniqueness of indicators for different desired outcomes of biodiversity condition means that the design of monitoring programs may be complex and expensive; thereby limiting what may be monitored as identified in a priority assessment of a region. For example, our field study found that indicators for total plant species richness and composition of the perennial grass component (indicators identified by experts in technical workshops) not to be reliable measures.

Our framework can, however, hamper biodiversity condition assessment efforts when adopted without critical thinking, innovation and a practitioner's intimate understanding of a region. The framework still needs to be fully tested using a long-term dataset for the monitoring and reference sites. Ideally, it would also be beneficial to capture the temporal variability in the indicators quantitatively (e.g., Coulson and Joyce 2006; Howe *et al.* 2007) and calculate the degree of uncertainty associated with the data. Despite using the best knowledge at the time, we also acknowledge that readers will question some filtering decisions about the indicators. It is possible that local knowledge about indicators will improve filtering decisions.

We believe the framework improves our overall knowledge about the challenges and practicalities of biodiversity condition assessment for evaluating outcomes especially from waterpoint interventions. It should also provide biodiversity managers of government agencies and other NRM organisations with guidelines on how to conceptualise such assessments so that they stimulate effective biodiversity management at the science end of the biological science-planning interface.

A range of techniques have been adopted for evaluating biodiversity outcomes. Most are based on condition assessments of monitoring data. They range from simple aggregates (e.g., Ant Index of Biological Integrity, Majer and Beeston 1995; Biohyets, Read *et al.* 2005); conditions score such as BSSs (e.g., Oliver and Parkes 2003), "BioCondition" (Eyre *et al*., 2006a, b) and graphical representations (e.g., radar plots, Suter 2001) to more sophisticated multivariate mathematical treatments (Andreasen *et al.* 2001). Australia has lead the way in the development of biodiversity metrics for application in environmental stewardships (e.g., *BushTender*, Stoneham *et al.* 2003).

In our work, we critically reviewed (multi)metrics as they becoming mainstream in biodiversity conservation planning on private holdings (Smyth & Davies, manuscript). We found, despite their simplicity and convenience, the use of metrics for evaluating biodiversity outcomes is problematical for the arid rangelands. When assessed in scientific and not MBI policy terms, we agree with other biologists that the use of a single aggregate, though attractively simple, is "arbitrary at best and seriously deceptive at worst". The key impediments specifically relating to the arid rangelands now are environmental stochasticity which is not incorporated into metrics and baseline science underpinning reliable prediction by metric indicators in dynamic environments is absent. Overlaying the generic lack of transparency of (multi) metrics in decision-making and the fact that a metric actually obscures biodiversity condition in all its forms bought us to the conclusion that (multi)metrics are not appropriate for evaluating biodiversity outcomes in the arid rangelands at this time. Instead, we recommend a comparative approach where benchmark and intervention indicators are depicted visually over time like standard trend analyses (Smyth *et al.* 2009; Smyth & Davies, manuscript).

Synthesis

In this study, we aimed to test whether biodiversity outcomes could be assessed from waterpoint intervention in the arid rangelands renowned for patchy dynamics in space and time and driven mainly by variable rainfall. Our first challenge was to identify the expected outcomes from waterpoint interventions for biodiversity and, secondly, to then identify how to assess changes in biodiversity to invention and what was the best way to interpret them for management purposes. With the development of the framework we believe we achieved the first challenge and outlined a blueprint for the second challenge. We proposed that the assessment of biodiversity outcomes was best achieved by undertaking a condition assessment approach. To do this, we had to clearly define 'biodiversity condition' in the first instance. To assess change from waterpoint intervention in the arid rangelands with rainfall seasonality controlled experimentally, many reference (benchmark) sites need to be established (controls in our study) and each need to have rainfall measured at the site so that covariate analyses can be undertaken. For example, the Oodnadatta rain station recorded no rain from June 2007 to November 2007 yet we recorded between 10 to 20 mm on our Macumba sites at approximately 90 km away. This issue about the scale of measurement also has implications for modelling of vegetation cover, using remote techniques. Biodiversity (multi) metrics although convenient, are not appropriate for representing change in the arid rangelands mainly due to their inherent patchy (spatially and temporally) dynamics (Bastin 2009). Instead, we recommend using standard trend analysis approaches, especially creative graphics (e.g., radar plots, Smyth *et al.* 2009b).

At the beginning of this work, we chose a case study region in the arid South Australian rangelands where waterpoints have been established for a long time and intensification was advanced. We particularly focussed on cattle grazing because cattle per DSE (dry sheep equivalent) create less defoliation damage than sheep (mouth anatomy, Fensham & Fairfax 2008) and if we could obtained noticeable changes in grazing relaxation with subtle disturbance processes during the short length of this study, then it meant may have broader implications for other systems defoliated more severely (e.g., sheep and goats grazing). Another important pre-condition was the cooperation of producers for the duration of the study. As it happened, the gibber-gilgai biome in Oodnadatta land type of Stony Plains Bioregion was the most widespread pastorally productive system in the North East pastoral District of south Australia. Based on the extensive literature, we expected our predictions about grazing exposure and intensity to be met but we did not expect the spatial patchiness of the gilgais to challenge the notion of permanent grazing gradients for water-focussed grazing.

Our work indicates the importance of gilgai vegetation in arid South Australian stony deserts (which contain the most extensive stony deserts in Australia) for conserving a range of ephemeral wetland plant species. Due to high summer temperatures, cattle-grazing is often centred on water courses, which contain the only trees and therefore the only shade. Where permanent waterholes occur along these watercourses, they are infrequent but very heavily used by stock, resulting in ephemeral wetlands being heavily impacted by grazing and pugging. In comparison, the Oodnadatta saltbush dominated gibber-gilgai biome appears to be less heavily impacted by grazing since they are all tree-free, widespread, relatively common and occur over extensive areas, include locations relatively remote from artificial and natural permanent water points. While riparian vegetation contains a range of species not found in gilgai communities (and therefore need to be protected in their own right) the latter community is important in conserving a significant number of ephemeral wetland species common to both systems. The fact that the gilgais act as semi-permanent water sources after rains means that permanent water sources are rested and allowed to recover at a time when recruitment is at its most critical. When the gilgai water dries up, cattle are forced back to the permanent sources and gilgais beyond the threshold distances from water for cattle (approximately 4-5 km) are rested from grazing pressure until the next rains. Overlay these spatial dynamics with the landscape processes of gilgai formation and we believe we have a biome, which is resilient spatially and temporally. In some respects, it is a natural experiment in waterpoint management. Although our waterpoint invention studies are not complete, we have advanced present knowledge about the potential merit of waterpoint management and biodiversity resilience systems in the arid rangelands and produced a new assessment tool for assessing and interpreting biodiversity outcomes.

The impact of our work at this stage is new information, development of a new tool (a generic condition assessment framework) and the creation of momentum for strategic waterpoint management via interventions at the planning level. Already, the condition framework is being modified to report on a new framework for monitoring and evaluation of goat impacts in South Australia as part of “Caring for Our Country” program. However, our vegetation and fauna results are preliminary and inconclusive at this stage and it is critical that the final results be completed so that government planners (our collaborators) have confidence in our results and tool. For these reasons, we do not make any recommendations about the role of waterpoint interventions in achieving biodiversity outcomes.

Once we complete our field studies, we will submit a scientific paper on the responses of vegetation and fauna to waterpoint interventions and the implications for strategic waterpoint management. In this manuscript, we will discuss the role of grazing exposure, grazing intensity and the influence of rainfall seasonality on biotic responses.

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