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TITLE

Dryland communities find little refuge from grazing due to long-term changes in water availability

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ABSTRACT

Surface water availability in drylands has changed with the introduction of artificial water points. Despite known ecological impacts, detailed mapping of this change has not occurred in most drylands. We aimed to quantify the extent and distribution of changes in water availability. We tested whether water availability increased more in pastorally productive areas than less fertile areas, and whether remaining water remote areas are restricted to low productivity landscapes. Our new spatially-explicit method mapped access to water at fine spatial scale, weighting locations by their distance to water and the permanence of those water sources. We demonstrated our method in a study area of over 700,000 km2 in Queensland, Australia, with our mapping showing large changes in water availability since pastoral development. Less than 5% of the study area is now more than 10 km from water, compared with almost 60% previously. Few refuges for grazing-vulnerable communities remain. Even low fertility landscapes showed marked increases in water availability. This has conservation implications for managing production landscapes. Our approach can be applied in any dryland landscapes that have experienced changes in water availability, and can help guide actions such as removing artificial water points to recreate ecological refuges.

KEYWORDS

drylands, grazing rangelands, surface water, water remoteness, anthropogenic threat, spatiotemporal analysis

1 INTRODUCTION

The development of artificial water points for livestock farming has increased water availability in drylands of the world, leading to a redistribution of grazing in the landscape by both native and non-native fauna. Grazing impacts from livestock farming are among the top four most prevalent threats to global biodiversity (Maxwell, Fuller et al. 2016) and understanding where they occur is crucial for effective fauna and flora management. Changes in water availability can be mapped as a proxy for changes in land use, providing a way to quantify and compare the distribution and intensity of grazing in landscapes where water is a limiting resource (Fensham and Fairfax 2008). However, there is no standardised approach for mapping historical and spatial change in water availability in dynamic environments such as drylands with ephemeral water sources. This hampers our ability to evaluate ecological impacts of changed water availability and to prioritise management strategies for biodiversity recovery.

Changes in water availability have a range of ecological impacts. In drylands characterised by low total precipitation with high inter-annual variability, water is a limiting factor for vegetation growth and the persistence of many fauna species (Noy-Meir 1973, Morton, Stafford Smith et al. 2011). Providing freely available additional water allows both domestic and non-domestic grazing species to occupy areas for a longer duration (James, Landsberg et al. 1999, Chamaillé-Jammes, Valeix et al. 2007). Increased grazing is associated with changes in ecosystem structure and a reduction in vegetative cover (Asner, Elmore et al. 2004, Goirán, Aranibar et al. 2012), while even low intensity grazing can negatively impact ecosystem function (Eldridge, Poore et al. 2016). By preferentially selecting palatable plants, grazing animals also change vegetation composition (Cingolani, Noy-Meir et al. 2005), and intense grazing has led to declines in preferred species close to water sources (Landsberg, James et al. 2003, Todd 2006). Changes in vegetation composition and structure from grazing can also affect fauna by decreasing shelter, increasing exposure to predators, and reducing food resources (James 2003, Davies, Melbourne et al. 2010) Artificial water points also provide a resource subsidy that can influence the distribution of predators (De Boer, Vis et al. 2010, Brawata and Neeman 2011), and human settlements (Millán, Goirán et al. 2017). Groundwater-derived water sources are particularly important in dynamic environments, as they provide permanent water sources disconnected from rainfall and associated plant growth (James, Landsberg et al. 1999). When available, semi-permanent and ephemeral water sources can also allow grazing to extend away from permanent waters (Redfern, Grant et al. 2005, Smit and Grant 2009).

Water sources separated more widely than grazing animals' preferred travel distances will create 'water remote' areas, sufficiently distant from permanent water to have minimal grazing impacts (Fensham and Fairfax 2008). By mapping historical and current water sources, we can identify areas where water remoteness has remained high and thus land use change has been minimal. We assume that these water remote areas provide refuges from grazing impacts and water-driven changes in predator behaviour.

One challenge for mapping water availability is that data on water source location and permanence varies in quality and availability. While permanency can be assumed for groundwater-fed sources such as springs and bores, it cannot be determined from infrequent observations for dynamic sources such as rivers and dams. To identify the permanence and location of ephemeral waters, we therefore need data at a fine temporal scale to distinguish different levels of permanence, long temporal extent to account for inter-annual variability, fine spatial scale to identify small features in the landscape, and broad spatial extent to identify patterns across large areas. Mapping based on data with a relatively coarse temporal scale (e.g., once per year) may either miss non-permanent sources (errors of omission) or falsely identify them as permanent (errors of commission). Field surveys and land manager interviews can supply information on permanency (Silcock 2009), but this type of intensive effort is usually too expensive to implement across a broad spatial extent, and may not keep pace with rapid development of artificial water points.

In light of these challenges, water permanence has not been comprehensively addressed in previous studies of water availability, with many focussing only on permanent waters (James, Landsberg et al. 1999, Smit, Grant et al. 2007, Shannon, Matthews et al. 2009), or on measuring all surface waters during a constrained field period without distinguishing their permanence (Brawata and Neeman 2011). The few studies that have incorporated ephemeral as well as permanent waters only investigated short-term changes between seasons or a limited number of wet and dry years (Western 1975, Redfern, Grant et al. 2005), or averaged water availability over several years for a constrained area (Smit and Grant 2009). Archives of remote sensing data are now available at suitable temporal and spatial resolution to determine water availability for water sources of a certain size (e.g., larger than 30m × 30m) (Mueller, Lewis et al. 2016, Pekel, Cottam et al. 2016).

Water source locations can be identified using a variety of resources, but the datasets derived from these sources have varying spatial and temporal accuracy. Ground-based survey is accurate but costly and may not be possible due to security and logistics in remote regions (Owen, Duncan et al. 2015). Infrastructure databases, capturing groundwater bores and dams (for example Bureau of Meteorology 2013, Geoscience Australia 2015), can provide coverage but their accuracy requires validation, and these datasets are not necessarily available in drylands across the world (Duncan, Kretz et al. 2014). Manual interpretation of high spatial resolution imagery can provide accurate data on the location of infrastructure, though comprehensive coverage is limited by imagery costs (Duncan, Kretz et al. 2014). Freely available medium resolution imagery, such as Landsat, offers potential for locating and monitoring water sources larger than the imagery resolution (Owen, Duncan et al. 2015, Mueller, Lewis et al. 2016). Determining the location of historical water sources, such as now extinct springs, requires examination of historical sources and field survey (Fensham, Silcock et al. 2015). Because of these challenges, previous studies documenting the magnitude and extent of changes in water availability focussed on local areas (James, Landsberg et al. 1999, Smit, Grant et al. 2007, Shannon, Matthews et al. 2009) or at a broad scale that did not include smaller point sources (Pekel, Cottam et al. 2016). The information required for mapping water permanence and location with fine detail across broad scales is thus not available from any individual dataset. However, by combining several detailed datasets with the broad coverage of remote sensing-derived data, it may be possible to effectively map water permanence and location across very large areas.

This study develops a method to map changes in water availability in dynamic environments characterised by fluctuating ephemeral water sources, using the drylands of western Queensland, Australia, as a case study. Our approach provides a more comprehensive view of water availability and remoteness than has previously been possible, due to our incorporation of both permanent and non-permanent water sources. Through analysing distance from water, we identify areas that have remained water remote, as well as those where water remoteness has changed considerably since pastoral development. Our spatially-explicit method describes the extent and magnitude of water remoteness, enabling identification of both the absolute and relative remoteness of areas of interest. We predict that water availability will have increased more in pastorally productive areas than less fertile areas, and that remaining water remote areas will be limited to low productivity landscapes less preferred for livestock farming. We demonstrate how an understanding of water availability can address landscape ecology questions. In particular, where have changes occurred in the distribution of grazing impacts? Which vegetation communities have experienced the greatest change in water availability over time? The answer to these questions is crucial for targeting scarce conservation resources to the ecosystems most in need of threat mitigation and recovery actions.

2 MATERIALS AND METHODS

2.1 Study area

The study area covers 738,561 km^2 of the Australian State of Queensland, covering the area where mean annual rainfall is less than 500mm (Bureau of Meteorology 2011) (Figure 1). Small areas of higher mean annual rainfall within the study area were included to avoid gaps in the coverage. We divided the landscape into 820 million grid cells at 30m resolution for the analysis, to align with the resolution of the underlying remote sensing dataset.

The region has diverse landscapes, with the main vegetation communities comprised of Tussock grasslands, *Acacia* dominated open forests, Eucalypt open woodlands and Hummock grasslands (Neldner, Niehus et al. 2015). Grazing of native vegetation, mostly by cattle, is the dominant land use throughout the study area (Queensland Government 2016), with additional grazing pressure provided by herbivores such as macropods and goats (James, Landsberg et al. 1999). There is a mix of tenures across the study area, including freehold, leasehold and conservation areas (Queensland Government 2019). Grazing by sheep and cattle has occurred since the 1840s in the eastern part of the study area and the 1860s elsewhere, with pastoral development increasing in the 1890s following the introduction of artificial water points sourced from groundwater (Irvine 2016). Prior to the development of artificial water points in arid Australia, surface water availability was limited to permanent waterholes, rockholes and springs (Silcock 2009, Fensham, Silcock et al. 2015), and the ephemeral to semi-permanent waters of river channels, claypans and other temporary storages. These natural waters are now complemented by an extensive network of artificial water points, which capture surface water through dams and provide access to groundwater through bores (Bureau of Meteorology 2013, Geoscience Australia 2015). All water rights in the study area are held by the State of Queensland, with a framework of permits, licences and plans to govern the usage of water resources (State of Queensland 2019).

2.2 Water availability and remoteness mapping

We created maps of water availability to determine current and historic water remoteness across our study landscape, by following three steps:

(1) Combine spatial datasets from multiple sources on the location and permanence of natural and artificial waters into a water availability dataset for each time period of interest (here, pre-pastoral and current);

(2) Create maps of water remoteness using an algorithm that incorporates both distance from water and the permanence of the water source for each time period;

(3) Validate the water remoteness map for the current period by conducting targeted checks for unmapped water sources (omission errors) or falsely mapped sources (commission errors) using high resolution satellite imagery. Revise the water remoteness map as required. Conduct sensitivity analysis to test aspects of uncertainty.

We explain these steps in detail below. We conducted the analysis using ArcGIS 10.3 (Environmental Systems Research Institute 2015).

2.2.1 Step 1: Combine existing spatial datasets on natural and artificial waters

We collated six spatial datasets on surface water features (Table 1, with additional detail in Appendix A: Table A.1). The study area had high data availability for point-based water sources, with freely available Australian government databases providing the locations of bores and dams (Bureau of Meteorology 2013, Geoscience Australia 2015) and a recent extensive ground survey providing data on natural water features such as springs, waterholes and rockholes (Silcock 2009, Fensham, Silcock et al. 2015). These datasets on individual features were complemented by data on the prevalence of water features (e.g., rivers and lakes) across the landscape, derived from 27 years of Landsat data (Fisher, Flood et al. 2016). Water prevalence was calculated for each cell by dividing the number of times water was recorded in that pixel by the total number of valid records across time (~600) for that cell. This is a similar approach to the water index calculated by the Australian Water Observations from Space service (Mueller, Lewis et al. 2016), however our approach uses a water index calibrated to Queensland conditions (Fisher, Flood et al. 2016). This water index is known to underestimate green-brown water (e.g., shallow dams), and small (e.g., bores and springs) or narrow features (e.g., waterholes), for which other data are used in this analysis. To avoid edge effects in later data processing, we included data points up to 50km outside the study area, where available.

Using the water feature datasets, we created water permanence datasets to represent two time periods: pre-pastoral development (before ~1840) and current day (~2016). The dataset for the pre-pastoral period used all waterholes, rockholes and permanent springs, including those that are no longer active as derived from historical research (Fensham, Silcock et al. 2015). It also used the Landsat-derived water prevalence layer, with dams and other artificial waters removed. The dataset for the current period comprised all waterholes, rockholes and currently active permanent springs, as well as dams, currently active bores, and the Landsat-derived water prevalence layer. Our data did not include open bore drains that were historically extensive water sources after European settlement but are gradually being discontinued from use, nor does it include the network of piping and troughs that replaced and usually extended the open drains (Noble, Habermehl et al. 1998). However, these features are always associated with bores, which we have included in our mapping. For both datasets, we converted point and polygon data to a raster with a cell size of 30 × 30m, corresponding to the resolution of the Landsat data. We calculated the permanence of each water source as an annual percentage value ranging from 1 – 100%. The permanence value was based on a set of decision rules, developed from a combination of author knowledge of the landscape and literature review (Appendix A: Table A.2). As it was not possible to determine the permanence of small dams, we ascribed a consistent value of 70%. We also conducted a sensitivity analysis of this value, using 30%, 50% and 90% permanence values for small dams.

2.2.2 Step 2: Create maps of water remoteness

We created maps of water remoteness based on distance from each water source and the permanence of that water source. We classified the water permanence dataset for each time step into 20 classes in 5% increments from 0 – 100%, to provide a meaningful distinction between levels of permanence, without requiring excessive processing by using the unclassified raw data. Due to the potential underestimation of permanence using Landsat water indices (Fisher, Flood et al. 2016), we assigned each cell the upper value for its assigned class (e.g. water sources in the 5-10% class were assigned a value of 10%). These 20 classes were then split into individual layers for further processing. The very low permanence class (0-5%) layer was not used further in this analysis, as it was dominated by cells with extremely infrequent (<1%) water availability. Assigning these cells to 5% water availability would have overstated the total water availability in the landscape.

To extrapolate water availability across the landscape, we calculated an inverse distance raster layer for each of the 19 permanence-class layers in the historical and current time steps. Each permanence-availability layer represented the reciprocal distance from water sources in a given permanence class, whereby cells close to water had high values and cells distant from water had low values. Reciprocal distance has been used in previous grazing studies to highlight the decrease in grazing intensity from a resource (Manthey and Peper 2010). We then multiplied the inverse distance for each cell by the weighted permanence class for that layer, e.g., a cell 2000m distant from water features in the 45-50% class would be weighted at 50% and have a weighted permanence value of 1/2000 x 0.5 = 0.00025. To combine the weighted permanence values across the 19 permanence-class layers, we overlaid layers and selected the maximum value for each cell from across all classes for that time step. This approach assumes that an animal seeking water chooses to maximise energy efficiency by using available water sources in the landscape sequentially, visiting the closest points before visiting more distant water points. This approach provides greater discriminatory power than a simple threshold approach, where distance would be calculated simultaneously from all combined water sources with permanence greater than a certain arbitrary value. While rapid to calculate, a simple threshold method cannot distinguish areas with semi-permanent water sources from areas with no water at all.

Our maximum-value approach is calculated using the equation:

$$DPI_{max} = \begin{cases} \max_i \left(\frac{1}{d_i} * p_j \right) & \text{if } d_i > 0 \\ \max_i \left(p_j \right) & \text{if } d_i = 0 \end{cases}$$

(Eq. 1)

where DPI_{max} is the maximum water availability value, j represents the class (from n=2 to n =20 in this analysis), p is the weighting for class j, based on the permanence of water sources in the class (0.1 – 1), and d represents the Euclidean distance in metres from the nearest water source to cell i, (1 - ∞). We did not apply a maximum search distance, but the analysis was confined to the study area. To aid the more intuitive interpretation of the mapping data, the DPI_{max} values were then inverted to provide an ‘effective distance from water’, EDW_{max}, expressed in metres, where locations further from water had higher values of water remoteness than locations close to water (range of 0 – 110 000m). For the example above, a cell with a DPI_{max} value of 0.00025 would have an EDW_{max} value of 4000m.

Most landscapes experiencing fluctuations in water availability are also heterogeneous in other environmental characteristics such as rainfall and productivity. To enable comparison between similar landscapes in our study area and focus our validation effort, we stratified broad vegetation groups (BVGs) by rainfall. BVGs incorporate multiple biogeographic and landscape attributes in one measure (Neldner, Niehus et al. 2015). This stratification enabled us to reduce the bias towards detecting water remote areas only in lower rainfall, lower productivity areas. With this stratification, we identified areas of ‘relative remoteness’, i.e. the most water remote areas for a particular broad vegetation group and 100mm rainfall band. We calculated the mean and standard deviation of EDW_{max} for each BVG and 100mm rainfall band. We then converted the raw water remoteness map to a relative remoteness map by assigning cells with an EDW_{max} greater than one standard deviation above the relevant mean EDW_{max} a value of 1 and cells with

EDW_{max} greater than two standard deviations above the mean EDW_{max} a value of two. The rest of the cells in the landscape were assigned a value of zero. The relatively remote areas were then used for step 3 to identify high priority areas for validation.

2.2.3 Step 3: Validate water availability map

Data on water availability varies in accuracy and reliability. Omission errors can occur from use of datasets that are not comprehensive or have inaccurate locations for water points. Commission errors can also occur due to misclassification errors of remote sensing, the use of out of date datasets or incorrectly attributed features. We addressed omission and commission errors in the spatial coverage of our water source maps through a targeted validation process. This process addressed location errors, but not uncertainty about the permanence of the water. We used the relative remoteness map created in step 2 to select the most remote cells within each stratified class to ensure that the inspection effort was balanced across different regions of the study area.

For the omission error validation, we targeted our search to cells with a current EDW_{max} value that was two standard deviations above the mean for the relevant broad vegetation group and rainfall band, and with a contiguous area of over $10km^2$. As these are the most water remote areas, any additional water sources found within them will have a greater impact on water remoteness than additional sources found in less water remote areas. We selected 34 $371km^2$ (4.6%) of the study area for checking using this method.

For the commission error validation, we focussed our checks for false positives on the bores and dams datasets, as data on waterholes, springs and rockholes were recently compiled and ground-truthed (Fensham, Silcock et al. 2015). We first calculated the spatial average of the current EDW_{max} for a 5km radius around each bore and dam. This distance was selected as it

282 approximates the typical distance within which most cattle grazing occurs in these rangelands
283 (Fensham and Fairfax 2008, Frank, Dickman et al. 2012). The artificial water points with highest
284 EDW_{5km} are thus the most isolated from other water sources, and would have the greatest effect
285 on surrounding areas if incorrectly mapped. Within each broad vegetation group and rainfall
286 band, we selected artificial water points with a current era EDW_{5km} value that was two standard
287 deviations above the mean for bores (n = 229) and three standard deviations above the mean for
288 dams (n = 313). The more rigorous targeting of dams was due to the very large number of dam
289 features.

290 For each validation process, we then reviewed each set of selected cells (for omission error
291 detection) or water points (for commission error detection) using SPOT satellite imagery
292 (AIRBUS Defence and Space 2016). The 1.5m resolution of this imagery is sufficient to observe
293 small water points, such as water tanks and associated infrastructure. Unmapped water sources
294 (omission errors) were added to a revised dataset and falsely mapped points (commission errors)
295 were removed or shifted to their correct location.

296 To examine the sensitivity of the mapping to uncertainties in the location and permanence of
297 artificial water points, we created multiple versions of the water availability map for the current
298 period, using different water permanencies for artificial water points for each version.

299 2.2.4 Application of water remoteness mapping to conservation problem: *Using water* 300 *remoteness to evaluate broad changes in threats to vegetation groups*

301 To evaluate the usefulness of our maps at informing conservation management questions, we
302 explored an application of the water remoteness mapping. We demonstrate how to use the water
303 remoteness dataset to inform an ecosystem- or region-focused conservation problem about where

to allocate management efforts aimed at reducing grazing impacts. We asked the question: Have all vegetation groups been equally affected by changes in water remoteness and associated grazing pressure? If not, which vegetation groups are most impacted and might require greater recovery efforts than others? Our hypothesis was that vegetation groups with higher pasture productivity will have seen a greater increase in water availability than those with lower pasture productivity, as they provide a greater return-on-investment in water infrastructure for livestock farmers.

To derive a measure of pasture productivity for each broad vegetation group (BVG), we used the AussieGRASS pasture growth model (Queensland Department of Environment and Science 2015), which has a cell size of 0.05 degrees. We selected cells with at least 90% coverage of an individual BVG to determine the mean probability that 1ha of the relevant broad vegetation group will produce at least 1000kg/year of pasture. For BVGs with a wide range in pasture productivity, we used expert opinion to subset these BVGs into categories reflecting their productivity. Due to the diversity of pastoral productivity among subgroups within the ‘other *Acacia* dominated forests’ group, we further divided this group into separate subgroups of Gidgee, Brigalow and Other *Acacia* Species on Residuals (i.e. *Acacia* communities on rocky hills). This reclassification resulted in 8 categories for the detailed productivity analysis. We also considered vegetation species suitable for browsing, for example Mulga-dominated forests, when determining grazing potential. For this application, *EDWmax* values were sampled on a 900m grid, providing over 900,000 sample points.

3 RESULTS

Water remoteness has declined considerably since the development of artificial water points in the study area, with few areas remaining outside the effective grazing range (5km) of domestic stock in arid areas (Frank, Dickman et al. 2012) or native macropods (10km) (Lavery, Pople et al. 2018) (Figure 2b). During the pre-pastoral period, 80.1% of the study area (591,616 km^2) was moderately water remote (i.e. EDW_{max} > 5000), but this has dropped to 21.6% (159,577 km^2) in the current era. Only 4.6% (34,255km^2) of the study area is now very water remote (i.e. EDW_{max} > 10000), compared with 59.6% (440,094km^2) in the pre-pastoral period (Figure 2a and b). The decline in water remoteness has been uneven across the study area (Figure 2c), with extensive areas of sand dunes, rocky plateaux, and large conservation areas protected from grazing providing the highest water remoteness values and largest contiguous very water remote areas (Appendix A: Figure A.1). Only 0.6% of the region has experienced an increase in water remoteness, associated with springs becoming extinct (Figure 2c). The relatively remote areas are also not distributed evenly, with more remote areas tending to be clustered (Figure 2d, Appendix A: Figure A.1).

3.1 Accuracy assessment of datasets

The omission rate was low, with an additional 169 water points mapped within the most remote 34,371km^2 (4.6% of the study area), representing an additional 0.5 water points / 100 km^2. Most additional points (n = 129) were in the Tussock grasslands, representing one additional unmapped point per 100km^2. However, there was a high-moderate commission rate in the bore and dams datasets. Inspection of the most isolated water points showed that 38% (n = 87) of the 229 validated bores were not visible in the available imagery, and did not have accompanying infrastructure such as dams or troughs within 500m. Commission errors were moderate for dams, with 14% (n = 44) of the 313 dams not present, typically due to misclassification of ephemeral wetlands or heavily vegetated areas as water sources.

Considering this commission rate, we conducted an additional sensitivity analysis for bores and dams. We created a layer with bores weighted at 60% (i.e. approximating the reliability of 62% x permanence of 100%), and incorporated this into a new current EDW_{max} layer, EDW_{max_bore60}. As the permanence value for dams had been determined arbitrarily, we created three additional versions of the current EDW_{max} layer, with dams at 30, 50 and 90% permanence (EDW_{max_dam30}, EDW_{max_dam50}, EDW_{max_dam90}) to support a sensitivity analysis. This procedure was implemented to assess the impact of the assigned permanency and also commission errors in the dams dataset. The results from this sensitivity analysis are incorporated into discussion of the application below and highlighted in a series of maps (Appendix A: Figures A.2a-A.2d).

3.2 Application: Change in water remoteness of broad vegetation groups

Although change in water availability has been uneven across broad vegetation groups, all groups declined in water remoteness (Tables 2 and 3, Figures 3, 4 and 5). Only the low pastoral productivity Hummock grasslands and other *Acacia* communities on rocky hills retained a mean effective distance from water of over 10km. In contrast, all vegetation groups had a mean effective distance of more than 10km under the pre-pastoral scenario (Figure 5, Appendix A, Table A.3).

Increases in water availability were not limited to pastorally productive vegetation groups (Figure 4). For example, the four *Acacia*-dominated vegetation groups vary in productivity (Figure 4), but all experienced at least a 55% reduction in water remoteness (Table 2). Low-productivity *Acacia* communities on rocky hills maintained higher water remoteness than higher productivity Brigalow and Gidgee communities, although the magnitude of the change was similar to that for Gidgee (Figure 5). Mulga (*Acacia aneura*) dominated woodlands had a similar magnitude of change in water availability as Brigalow (Figure 5), with more than 85% of the area currently less than 5km from water and less than 2% remaining as very water remote (Table 2). Mulga-dominated forests have only moderate pasture productivity (Figure 4), however mulga itself can be harvested as fodder, complementing grass production for grazing.

Pastorally productive communities, such as Tussock grasslands, had some of the greatest declines in effective distance from water (Figure 5), driven by higher densities of bores and dams to service cattle grazing across large, contiguous areas (Table 2). However, the magnitude of change was similar to Gidgee, and only slightly higher than the lower productivity *Acacia* on rocky hills (Figure 5). Although almost 85% of pre-pastoral Tussock grasslands occurred more than 5km from water, now less than 20% of the Tussock grasslands remain water remote (Table 2).

The greatest overall change between pre-pastoral and current effective distance from water was in Eucalypt low open woodlands (Figures 4 and 5). This BVG previously had very low water availability, with less than 8% of the area found within 5km from water, and has relatively high pastoral productivity based on the AussieGRASS analysis (Figure 4). However, this may over-represent the stocking rate for less productive parts of this vegetation group on hills and ranges, due to the fine scale variation in this vegetation group compared to the broad scale of the AussieGRASS grid cells. In addition, the marked increase in water availability may not be completely associated with an increase in grazing by domestic stock. Most of these woodlands are found on the hill slopes and plateaus of the North West Highlands (Neldner, Niehus et al. 2015), where large dams have been established to support mining activities and settlements.

Although domestic stock may not be present in high numbers, this increase in water availability may still have ecological impacts by providing a resource for herbivores such as native macropods and feral goats, as well as predators such as dingos.

Even low pastoral productivity Hummock grasslands (mostly spinifex-dominated communities in the far west of the study area; Figure 1) lost half of their very water remote area (from 57% to 27% of its extent; Table 2), while the area within grazing range of 5km doubled (from 25% to 52%; Table 2). The magnitude of change was relatively low compared to the other vegetation groups (Figure 5).

The smallest overall change in water remoteness occurred in the Eucalypt open forests on floodplains (Figures 4 and 5), which previously had higher water availability due to their proximity to watercourses. However, as many of these watercourses are ephemeral, this vegetation group still had a notable increase in water availability, mostly through dam construction (Table 2).

Sensitivity analyses adjusting water permanence showed that even under a scenario with dams at only 30% permanency, the decline in water remoteness between pre-pastoral and current conditions was still strong across all broad vegetation groups (Appendix A: Table A.3). Some differences were evident, reflecting the relative dominance of dams or bores for each broad vegetation group. For example, due to the higher number of dams in the Tussock grasslands (Table 2), changes in dam permanency had a greater impact on water remoteness than the Hummock grasslands (Appendix A, Figure A.3A and A.3E). However, the rankings of ecosystems in terms of overall water remoteness were robust to changes in water permanence (Appendix A, Table A.3).

4 DISCUSSION

Despite large losses of surface water globally over recent decades, likely linked to climate change and associated drought (Pekel, Cottam et al. 2016), surface water across all continents has increased. This change is not restricted to recent years. Here, we demonstrate a substantial increase in dryland water availability in Australia since the introduction of artificial water points for pastoral production in the 1840s. Importantly, we find that this change is not limited to vegetation groups with high pastoral productivity (Figures 4 and 5). The current limited number and extent of water-remote areas implies that few refuges remain for grazing-vulnerable communities.

Our finding that water availability changes are not limited to more productive areas for livestock farming has implications for managing production landscapes for conservation. Less productive ecological communities can often experience greater negative impacts from grazing due to lower evolutionary exposure to grazing animals and therefore fewer adaptations that might help tolerate increased grazing pressure (Eldridge, Delgado- Baquerizo et al. 2017). Although less productive vegetation groups in our study (e.g, Hummock grasslands) had fewer artificial water points (Table 2), these areas can be affected by water points in neighboring more pastorally productive areas. For example, the Hummock grasslands of the Simpson Desert in the far south-west of the study area are intermixed with other *Acacia*-dominated woodlands used for grazing (Fensham, Fairfax et al. 2010). This proximity effect can have a particularly strong impact when less productive areas are intermingled with more productive areas, such as rocky hills within Tussock grasslands. Rocky hills are known to be important refuges for species within the arid zone (Silcock and Fensham 2014, Fitzsimons and Michael 2017), but these specialised habitats can be completely contained within the grazing range of water points located in more productive pastoral lands. Maintaining or re-establishing water remoteness for small rocky refuges will require careful management of artificial water points along their edge. Closing water points will increase water remoteness, but the cost-effectiveness of this strategy in our study area is limited by the high density of water points across much of the landscape. Previous government action to manage water usage has focussed on upgrading bores to reduce uncontrolled flows, with some ecological benefits to spring ecosystems due to the restoration of artesian pressure (Noble, Habermehl et al. 1998, Hill 2011). Incurring the cost of reduced pastoral production by closing water points will require a clear demonstration of the conservation benefits. Future studies could aim to optimise the points selected for closure by minimising lost production opportunity whilst maximising water remoteness for particular conservation or resource management objectives.

The water remoteness dataset created in this study enables the identification of relatively remote areas, which can form the basis for prioritising water point closure. For example, water points could be closed to reduce overall grazing pressure across a target area, to increase the total water remote area within a broad vegetation group, or to create the largest contiguous water remote area within a bioregion. These different landscape-level approaches could be complemented by a focus on target species, such as native herbivores or small mammals. For example, the impact of grazing pressure from macropods could be tested by closing water points to create areas at least 10km from water (Lavery, Pople et al. 2018). Alternatively, areas free from grazing are required to conserve small mammals (Kutt and Gordon 2012), and re-establishing water remote areas around known populations may support achievement of this objective. For example, reduction in water point density is included as a specific action in the draft recovery plan for the threatened Greater Bilby (*Macrotis lagotis*) (Commonwealth of Australia 2019). The water remoteness dataset could be used to identify the water points near Greater Bilby populations that will provide the greatest reduction in threatening processes if closed. In combination with other threat and habitat suitability modelling, the water remoteness dataset could also support identification of suitable translocation locations for this vulnerable species.

The availability and accuracy of datasets on water sources is a key constraint for mapping water remoteness. While the datasets available for this study area may not be available for all locations, Landsat-derived data can provide an initial indication of water prevalence. This can then assist in targeting areas for further investigation of smaller water sources, both natural and artificial. Even with high data availability for our study area, there were still several limitations. Our validation assessment of the bores and dams datasets showed inaccuracies in the location of some of these features, with a higher commission rate for bores than for other water sources. These inaccuracies did not affect the overall patterns for our example application, but finer-scale applications of our approach may need a more detailed local assessment of infrastructure. Given the error rates in the source data, use of our water remoteness dataset for spatial modelling or prioritisations should also incorporate uncertainty mapping, e.g. (Tulloch, Possingham et al. 2013). In addition, targeted reduction of uncertainty, as used in this analysis through map validation using alternative datasets, provides an efficient way to improve accuracy in specific areas.

We made several assumptions about the permanency value for water points in our water remoteness mapping approach. Although bores were assumed to provide water with 100% permanence, individual bores may be turned off or fenced for management purposes (e.g. Russell, Letnic et al. 2011). For small dams, we used a constant permanence value across the study area regardless of their position in the landscape. This could be improved by modelling permanence and reliability across the rainfall gradient to incorporate heterogeneity in dam water availability across the study area. Alternatively, as the imagery archive increases for high frequency, medium-high spatial resolution platforms such as PlanetScope (3m) (Planet Labs Inc. 2018), it will also be possible to improve the mapping of water permanence of smaller dynamic water sources, which will reduce their impact on water availability. The simple weighting by permanence used in our approach could be refined using additional temporal data from remote sensing products to examine the seasonality of water availability, similar to the approach of Pekel, Cottam et al. (2016), or using the establishment date of artificial water points to investigate the cumulative impact of water availability in the landscape.

Our method of mapping water remoteness can be applied in any dynamic environments that may have been affected by changes in water availability over time. The continuous and comprehensive measure of water availability produced by mapping water across space and time allows water remoteness analyses and maps to be tailored to numerous ecological and management questions. Because we explicitly include permanency of the water source when calculating water remoteness, this allows the comparison of refuge locations between different seasons, and the evaluation of different strategies of dispersing species, e.g., those that use ephemeral waters versus those dependent on permanent waters. For example, how might the preferred grazing distances for different species affect perceived water remoteness? Where in the landscape would a reduction in water permanence have the greatest impacts on improving water remoteness for at-risk species, or reducing the dispersal of invasive species? How could the invasion of a highly-dispersive species with tolerance to long distances from water affect water remote refuges in the landscape? It is also possible to incorporate water remoteness mapping directly into species distribution modelling to improve predictions of where species are likely to persist and investigate locations for translocations. Future studies could also test the impact of water availability on vegetation cover across the landscape, using the broad, consistent spatial coverage of this dataset in combination with remote sensing indices. For example, are there consistently higher proportions of bare ground in areas closer to water, if factors such as rainfall, soil type and vegetation type are kept constant?

We developed a practical and intuitive method for mapping water availability and water remoteness in dynamic environments, which showed that changes in water availability were not limited to pastorally productive areas. The method incorporates permanence and distance into broad scale analyses of water availability across a dryland landscape. The spatial data created by this method provides flexibility to evaluate water remoteness across multiple components of biodiversity and areas of interest. In the study area of over 700,000km2, the method detected regionally significant differences in changed water availability over time. Mapping water remoteness across different time periods can help mitigate human impacts on surface water distribution through identifying important refuges from grazing impacts, supporting species and ecosystem risk assessments, and informing allocation of conservation actions such as closing water points to create a water-remote reserve network (Fensham and Fairfax 2008).

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7 TABLES

Table 1 – Data sources for water source mapping used to develop water availability maps for two time periods, pre-pastoral (1840) and today (2016).

Time period	Water type	Water source	Water permanence	Data source
Pre	Natural	Springs	100%	Queensland Springs database (Queensland Herbarium 2016), supplemented by Springs data (Geoscience Australia 2006)
Post	Natural	Springs	100%	
Pre	Natural	Rockholes	70-100%	Western Queensland rockholes dataset (supplied by Silcock, J.) Permanence based on landholder interview.
Post	Natural	Rockholes	70-100%	
Pre	Natural	Waterholes	25-100%	Western Queensland waterholes dataset (supplied by Silcock, J.) Permanence based on landholder interview for most waterholes. Landsat-based water prevalence mapping used for large waterholes (> 50 hectares)
Post	Natural	Waterholes	25-100%	
Pre	Natural	Other surface waters	0-100%	Landsat-based water prevalence mapping (Fisher, Flood et al. 2016)
Post	Natural	Other surface waters	0-100%	
Post	Artificial	Boreholes	100%	Active wells from NGIS database (Bureau of Meteorology 2013)

Post	Artificial	Reservoirs, including rural water storage	30-100%	Water storage locations (Geoscience Australia 2015) Permanence based on Landsat-based water prevalence mapping for large reservoirs (> 1ha), and a consistent value (70%) for small reservoirs (< 1ha).

Table 2 - Changes in water remoteness for the six largest broad vegetation groups in western Queensland between two time periods, pre-pastoral (1840) and today (2016).

Broad vegetation group	Relative Pasture Productivity*	Percentage area > 5km from water		Percentage area >10km from water		Artificial Water Points/ 100km^2	
		Past	Current	Past	Current	Bores	Dams
Tussock Grasslands, Forblands	High	84.45	19.06	65.89	2.79	1.39	5.22
Acacia aneura (Mulga) Dominated Open Forests, Woodlands And Shrublands	Medium	76.28	14.71	50.26	1.39	1.26	3.26
Other *Acacia* Dominated Open Forests, Woodlands And Shrublands, consisting of:	Medium	86.17	26.15	64.84	5.11	0.90	3.92
- *Acacia* spp on residuals (i.e. rocky hills) (42%)	Low-Medium	86.87	39.79	67.03	9.44	-	-
- *Acacia harpophylla* (Brigalow) (5%)	High	84.66	10.05	55.98	1.46	-	-
- *Acacia cambagei* (Gidgee) woodlands (53%)	Medium	85.76	16.88	63.97	2.03	-	-
Eucalypt Low Open Woodlands	Medium-High	92.73	34.47	78.96	5.38	1.84	1.41
Hummock Grasslands	Low	75.01	48.12	57.31	27.83	0.63	0.69

Eucalypt Open Forests To Woodlands On Floodplains	Medium-High	63.86	15.67	39.74	2.57	1.60	8.06

*Assessment based on the mean probability that 1 hectare will produce at least 1000kg/year of pasture, based on Aussiegrass pasture growth model (Queensland Department of Environment and Science 2015)

9 FIGURES

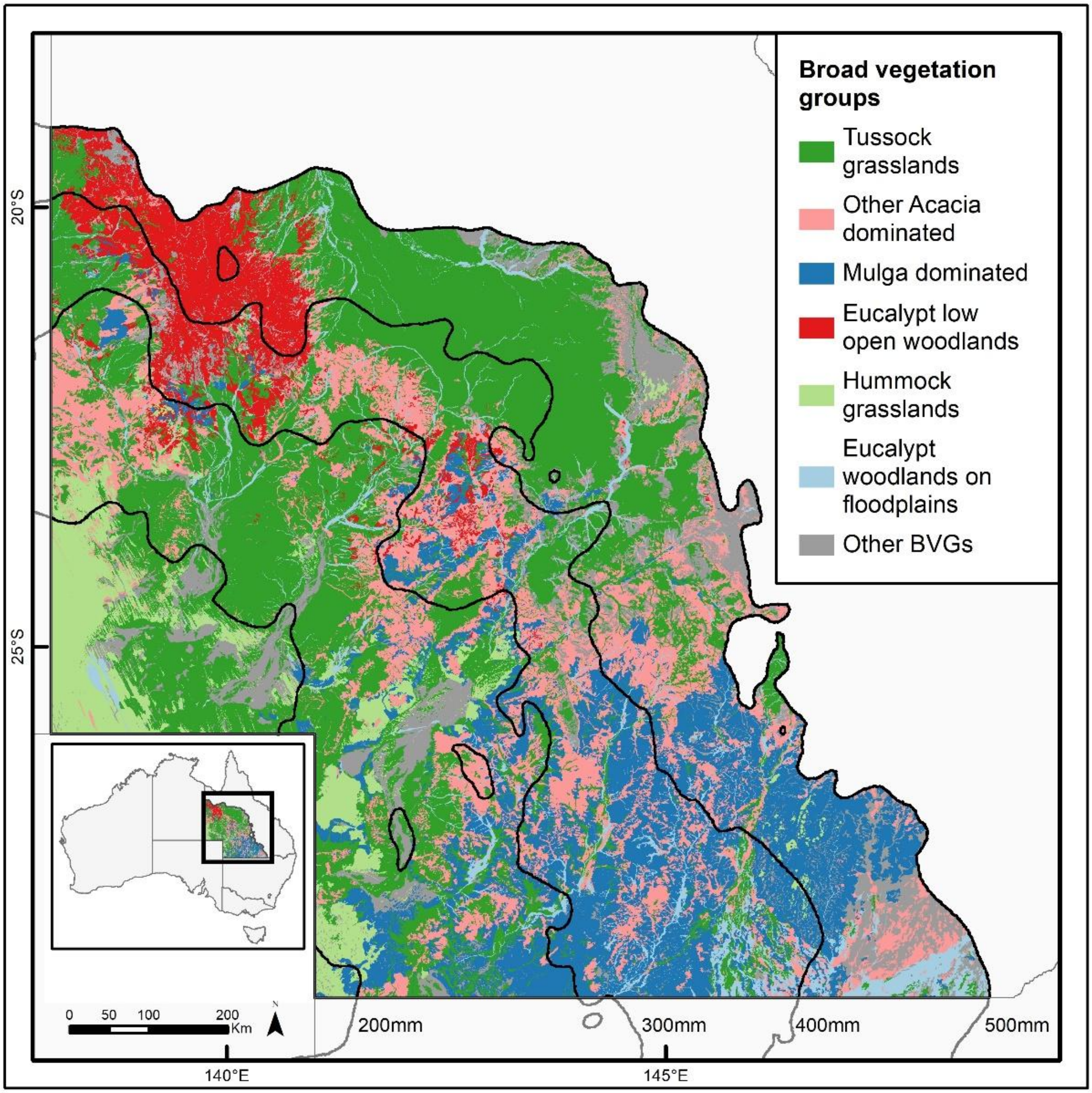

Fig. 1 Study area, representing the portion of the Australian state of Queensland with mean annual rainfall less than 500mm, showing 100mm isohyets and the six largest broad vegetation groups (BVGs).

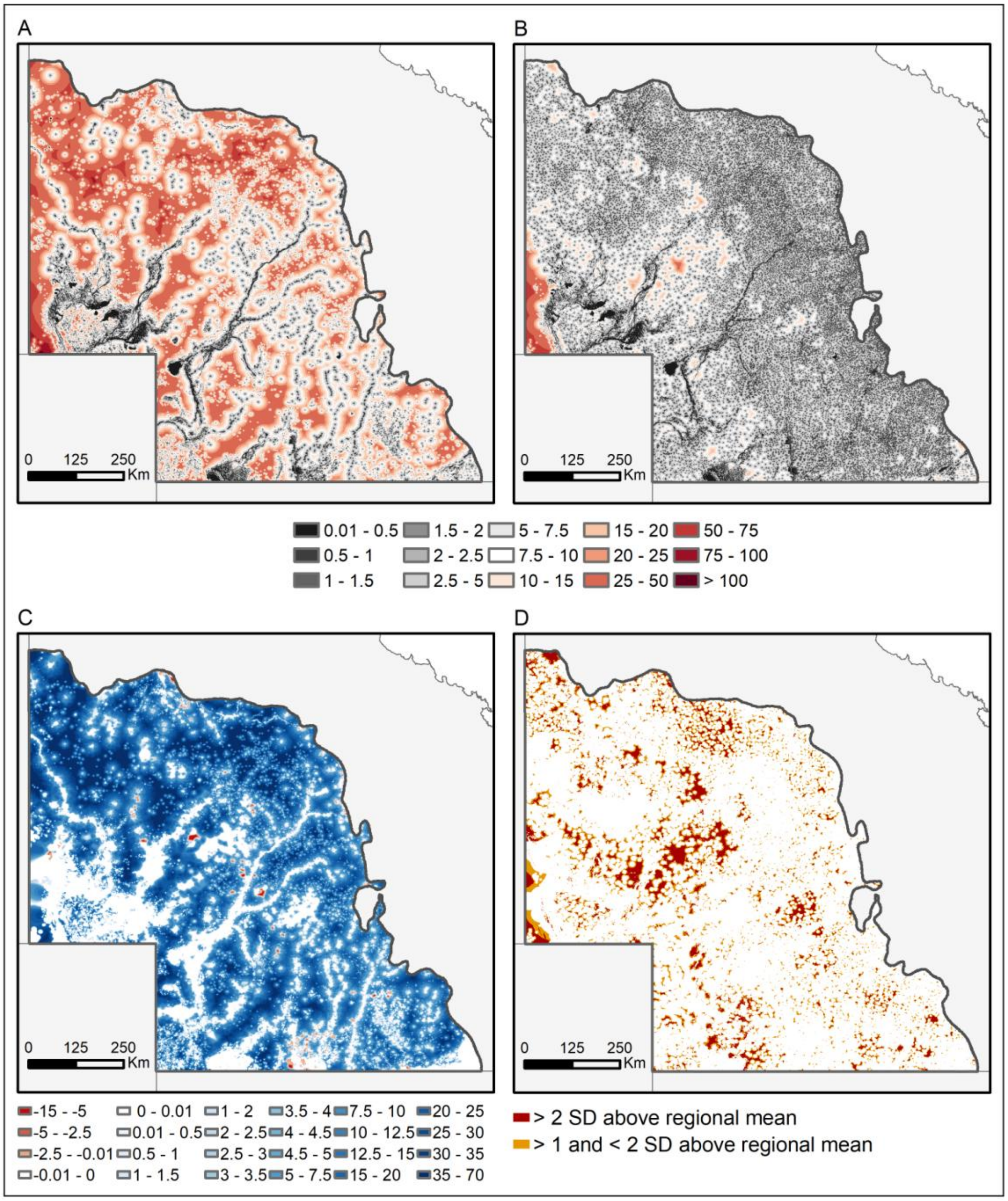

Fig. 2 The north-eastern Australia study area showing (a) pre-pastoral water-remoteness in km, (b) current water remoteness in km, (c) the difference between the two in km, using the validated dataset, and (d) the relatively water-remote areas.

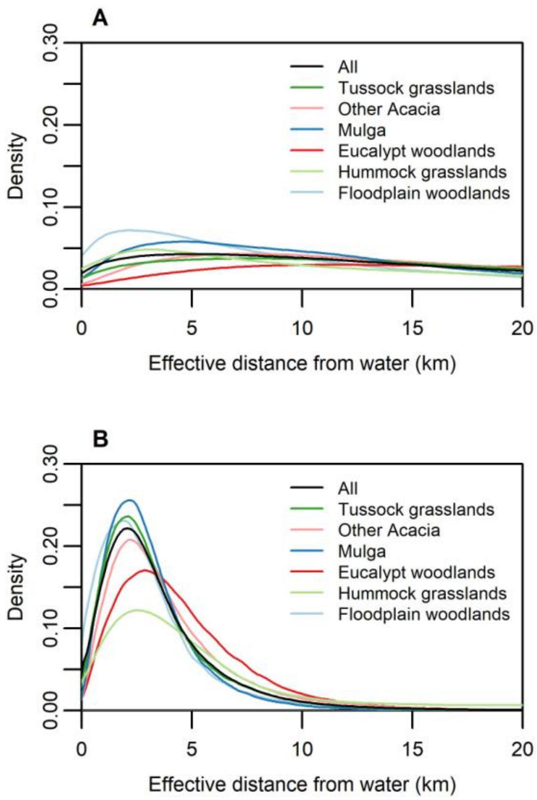

Fig. 3 Density distribution for EDW_{max} for the six largest broad vegetation groups in western Queensland, showing (a) pre-pastoral and (b) current conditions.

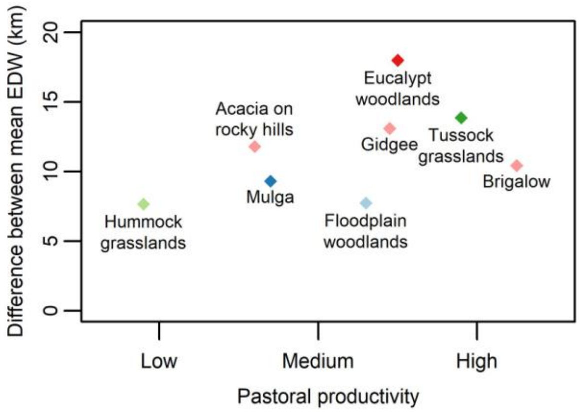

Fig. 4 The difference between means of pre-pastoral and current EDW_{max} for selected broad vegetation groups in western Queensland, by pastoral productivity. The 'Other Acacia' group is broken into three groups for this analysis – Acacia on rocky hills, Gidgee and Brigalow. A high difference indicates a high increase in water availability.

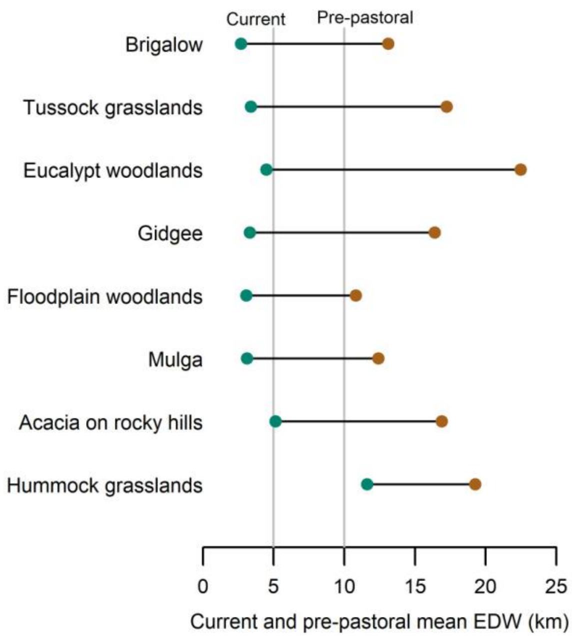

Fig. 5 The change between means of pre-pastoral and current EDW_{max} for selected broad vegetation groups in western Queensland. The threshold for water remote (5km) and very water remote (10km) is shown in grey.