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Freeman, Jeremy; Edwards, Andrew C.; Russell-Smith, Jeremy

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Article

Fire-Driven Decline of Endemic *Allosyncarpia* Monsoon Rainforests in Northern Australia

Jeremy Freeman *, Andrew C. Edwards and Jeremy Russell-Smith

Darwin Centre for Bushfire Research, Charles Darwin University, Darwin, NT 0909, Australia; andrew.edwards@cdu.edu.au (A.C.E.); jeremy.russell-smith@cdu.edu.au (J.R.-S.)

* Correspondence: freeman_jeremy@yahoo.com; Tel.: +61-427-270-835

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Abstract: Although contemporary fire regimes in fire-prone Australian savannas are recognised as having major impacts on an array of biodiversity and environmental values, a number of studies have observed significant monsoon rainforest expansion in recent decades. Here we assess the status of a locally extensive endemic monsoon rainforest type, dominated by *Allosyncarpia ternata* (Myrtaceae), restricted to sandstone terrain including in the World Heritage property, Kakadu National Park. We undertook assessments of: (1) geographic correlates of *Allosyncarpia* forest distribution; (2) change in canopy cover at 40 representative forest patches at topographically exposed sites with reference to a 60-year aerial photo and fine-scale image archive, and fire mapping data; and (3) structural characteristics associated with sites exhibiting stable, contracting, and increasing canopy cover. Mean canopy cover at sampled forest patches declined by 9.5% over the study period. Most canopy loss occurred at the most fire-susceptible patches. Assessment of structural characteristics at sampled sites illustrated that canopy expansion represented vegetative recovery rather than expansion de novo. The study (1) confirms the vulnerability of exposed margins of this forest type to fire incursions; (2) illustrates the magnitude of, and describes solutions for addressing, the regional conservation management challenge; and (3) serves as a reminder that, in savanna environments, severe fire regimes can substantially outweigh the woody growth-enhancing effects of other regional (e.g., increased rainfall) and global-scale (e.g., atmospheric CO₂ fertilisation) drivers.

Keywords: Arnhem Land; Kakadu National Park; Landsat Fire History; canopy change; woody thickening

1. Introduction

Fire regimes are recognised as being a major driver affecting the dynamics of savanna systems, including the balance of tree-grass components and the relative stability of savanna—forest margins. In the absence of burning, or even under conditions of frequent low intensity fires, substantial increases in savanna woody biomass, or extensive forest invasion of surrounding savanna vegetation, have been reported worldwide from a large number of observational and experimental studies [1–11]. Given that savannas contribute 30% of terrestrial net primary production [12] and the majority of global fire emissions [13], understanding the effects of savanna fire regimes and interactions with other drivers of savanna biomass change (e.g., land use, climate, CO₂ fertilisation) assumes increasing significance in an age of accelerating climate change including rising atmospheric CO₂ [14–16].

A substantial body of evidence indicates that contemporary anthropogenic fire regimes are causing significant impacts on a wide range of fire-vulnerable biodiversity elements in north Australia's relatively unmodified savannas, including: avifauna [17,18]; small mammals [19,20], and obligate seeding flora [21–23]. In large part, these recent fire regime impacts can be attributed to widespread Aboriginal societal collapse from the end of the nineteenth century, and associated breakdown of

former systematic fire management approaches focusing on imposing fine-scale spatial patchiness over the annual burning season [24–26]. In the absence of such intensive management, north Australian savanna fire regimes today are characterised generally by the incidence of frequent, extensive and relatively intense late dry season fires, especially in biodiverse higher-rainfall northern regions [27,28].

Despite these observed fire impacts, recent assessments suggest that contemporary regional fire regimes have had limited effects on vegetation structural attributes at landscape scales. For example, a recent meta-analysis incorporating a number of local and regional studies suggests there has been a general, if weak, trend of woody thickening in regional savannas in recent decades [29], especially in higher rainfall locations [10].

Likewise, many north Australian studies have reported expansion of typically small (<10 ha) patches or larger tracts of monsoon rainforest vegetation within the savanna matrix, particularly under higher rainfall conditions [5–7,30–35]. In different situations, such expansion has been attributed variously to reduced burning or browsing pressure, increased rainfall in recent decades, possible CO₂ fertilisation, and interactions between these. Although severe fires can incur significant impacts especially on closed forest margins [36], monsoon rainforest expansion has been observed in conjunction with frequent, presumably lower severity fires under relatively moist, productive site conditions [5,37].

Here, we present a reassessment of reported recent expansion of monsoon rainforest occupying mostly rugged sandstone uplands in the Top End of the Northern Territory, Australia. This monsoon rainforest type, dominated by the regional endemic *Allosyncarpia ternata* S.T. Blake (Myrtaceae), occupies an area of 113,800 ha [38], representing ~25% of all monsoon rainforest in northern and north-western Australia [36]. For a smaller region describing the western-most extent of *Allosyncarpia*'s range in Kakadu National Park, Bowman and Dingle [33] reported a net 21% expansion of this forest type based on analysis of canopy cover change at 12 sampled areas using the available aerial photographic record, 1964–2004. While observing that forest expansion was negatively correlated with fire activity, these authors attributed the expansion to a regionally wetter climate since the mid-20th century and elevated CO₂. Expansion of this forest type contrasts with other regional observations concerning current fire regime impacts on: (1) the catastrophic decline of the fire-sensitive conifer, *Callitris intratropica*, in adjoining savannas and on *Allosyncarpia* forest margins [21,39–42]; (2) stable or slightly contracting *Allosyncarpia* forest boundaries associated with decadal long-term monitoring plot observations [43]; and (3) contiguous extensive regional heathland floral diversity and associated fauna, now listed nationally as an Endangered Community with frequent wildfire identified as the key threatening process [22,44,45].

Given the critical regional biodiversity conservation management implications of these contrasting observations, in this paper we reassess decadal-scale *Allosyncarpia* forest change based on the available aerial photo record 1950–2010 at 40 exposed sites and associated fire records, and detailed on-ground vegetation structure sampling. The assessment seeks to address whether, in unmodified savanna landscape contexts, the influence of regional fire regimes may strongly outweigh climatic and global drivers.

2. Materials and Methods

2.1. Study Area

The Arnhem Plateau study area is located in Australia's remote tropical north, 350 km to the east of Darwin (Figure 1a). The Arnhem Plateau mostly comprises an extensive Proterozoic sandstone platform at 200–400 m elevation, covering approximately 25,000 km². This study focuses on an area of 29,600 km², slightly larger than the nationally mapped Arnhem Plateau Bioregion [46], home to around 1000 animal and 2000 plant taxa (Atlas of Living Australia: www.ala.org.au/). The region is recognised nationally and internationally for its biodiversity values [44,45,47,48].

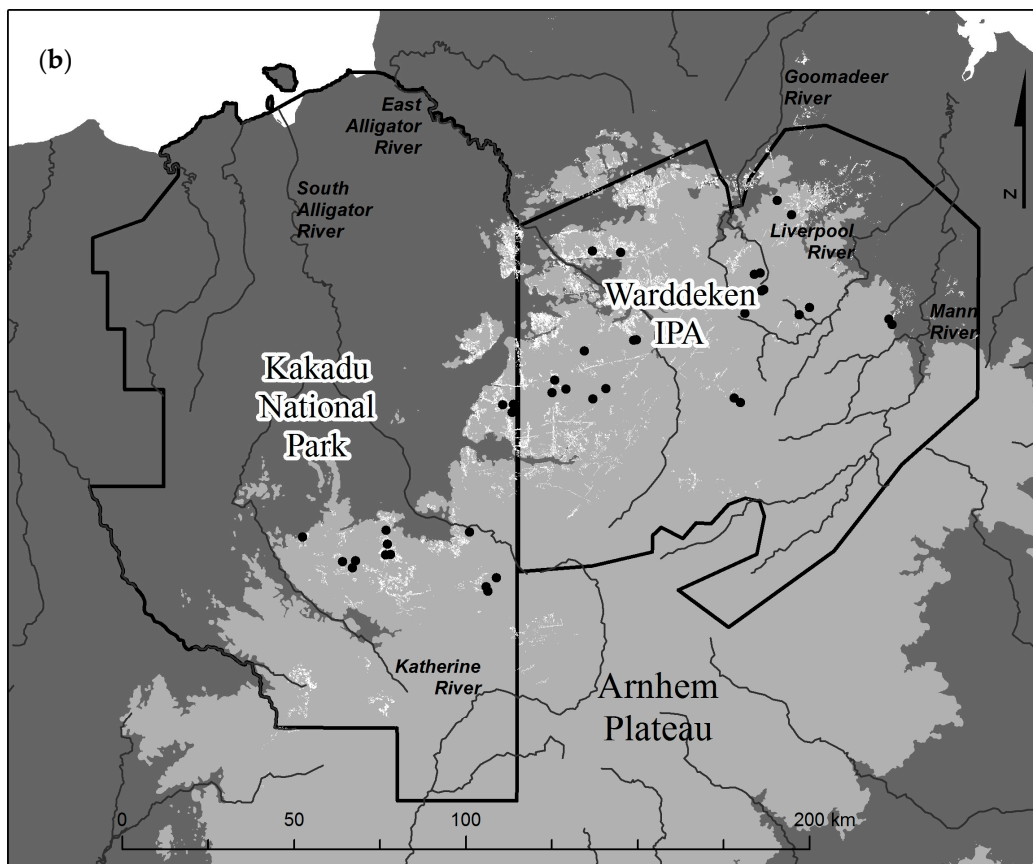
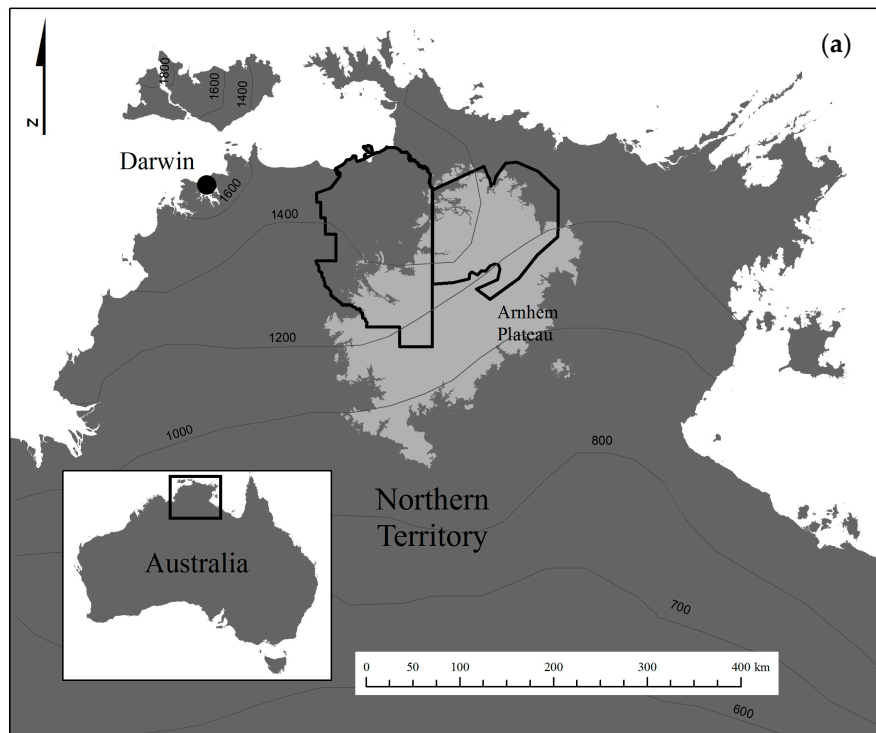


Figure 1. (a) Study region in the Northern Territory, Australia, including mean seasonal rainfall isohyets (mm); and (b) the mapped distribution of *Allosyncarpia* forest outlined in white, and the location of 40 sample sites used for the canopy cover change assessment.

The plateau is sharply defined on its western and northern edges by cliffs but less clearly defined in the south and east. It is dissected by major drainage lines and is topographically characterised by rocky gorges and extensive areas of heavily eroded rocky outcrops, platforms, and extensive areas of savanna plains. The sandstone rock is both porous and permeable which allows for significant ground water infiltration [49].

The regional climate is tropical monsoonal with temperatures consistently over 30 °C year-round [50]. Nearly all rain falls in the wet season between October and April, with peak rainfall from December to March. The region receives around 1450 mm of rainfall per annum with declining falls to the south and west of around 1200 mm.

Vegetation of the Arnhem Plateau is dominated by extensive savanna woodlands and open forests on shallow to deeper sandy soils, with tracts of shrubby heath vegetation occurring on shallow sandsheets and skeletal, rocky substrates [51]. *Allosyncarpia* forests occupy a range of sandstone-derived substrates, from moist valley sediments to steep, freely draining, rocky sites. *Allosyncarpia* is especially dominant on seasonally dry substrates where it provides over 80% basal area and effectively the entire canopy [38]. It is commonly associated with the obligate seeder conifer, *Callitris intratropica*. Although *Allosyncarpia* is relatively fire-tolerant given that established juveniles re-sprout prolifically after fire [52], termite-affected over-mature adults are susceptible to fire mortality once stems are ignited [41].

Climatic conditions promote rapid grass and litter fuel accumulation such that fires may recur within 1–3 years of having been burnt previously [53,54]. Fire severity generally increases with the progression of the dry season given increasingly severe fire weather (stronger winds, higher temperatures, lower humidity), and lower fuel moisture conditions [55]. Over much of the study region fire is a recurrent disturbance, especially in the south and south-east associated with more flat topography and prevailing south-easterly winds in the dry season [56]. Densities of browsing and grazing feral animals (Asian water buffalo [*Bubalus bubalis*], cattle [*Bos Taurus*, *B. indicus*]) were largely uncontrolled over the study period, and are consistently high in localised situations in western Arnhem Land, especially along watercourses [57].

Known by its Aboriginal custodians as *warddewardde*, the ‘stone country’ is also rich in cultural assets. The Arnhem Plateau region has been continuously occupied by people for at least 60,000 years [58,59], and use of fire over thousands of years is integral to the regional ecology. Fire usage was, and in some localised regions of the Arnhem Plateau especially, remains multifaceted and intricate [26,40,57,60,61]. *Allosyncarpia* forests, or *anbinik anngarre*, are highly valued as important camping, food and cultural sites. Individual forest patches are known to have been actively protected by site-specific traditional burning practices [61]. Following major Aboriginal depopulation of the region from the late 19th century [25,62], breakdown of the traditional fire regime has resulted in less frequent but extensive late dry season wildfires sweeping across the region, with significant impacts on fire-sensitive species and vegetation types [21,42,45,57].

2.2. *Allosyncarpia* Distribution Mapping

To provide a geographic context for the study as a whole, we utilised new 1:25,000 mapping of the distribution of *Allosyncarpia*-dominated forest extent across its entire range (J. Freeman, unpublished) and the geographic relationships between *Allosyncarpia* forest distribution mapping at 1:25,000, with available cadastral and biophysical Geographic Information System (GIS) datasets. *Allosyncarpia* forest patches are characterised by distinctive dense, closed canopies, which can be readily mapped (Figure 2). The new *Allosyncarpia* mapping was undertaken predominantly by manual digitisation of sub-1m Digital Globe (Worldview and GeoEye satellites imagery (October 2014), supplemented with small sections derived from Quickbird (0.6–2.4 m) imagery. A minimum of four identifiable canopy trees in a cluster (~1 ha) was used as the smallest area to be digitised; open areas with *Allosyncarpia* canopy-free areas larger than 50 × 50 m were excised. Based on extensive validation assessments,

reliability of the revised *Allosyncarpia* distribution map was found to be >99%, and boundary locational errors ± 3 m. Greater detail on mapping methods will be available in a forthcoming paper.

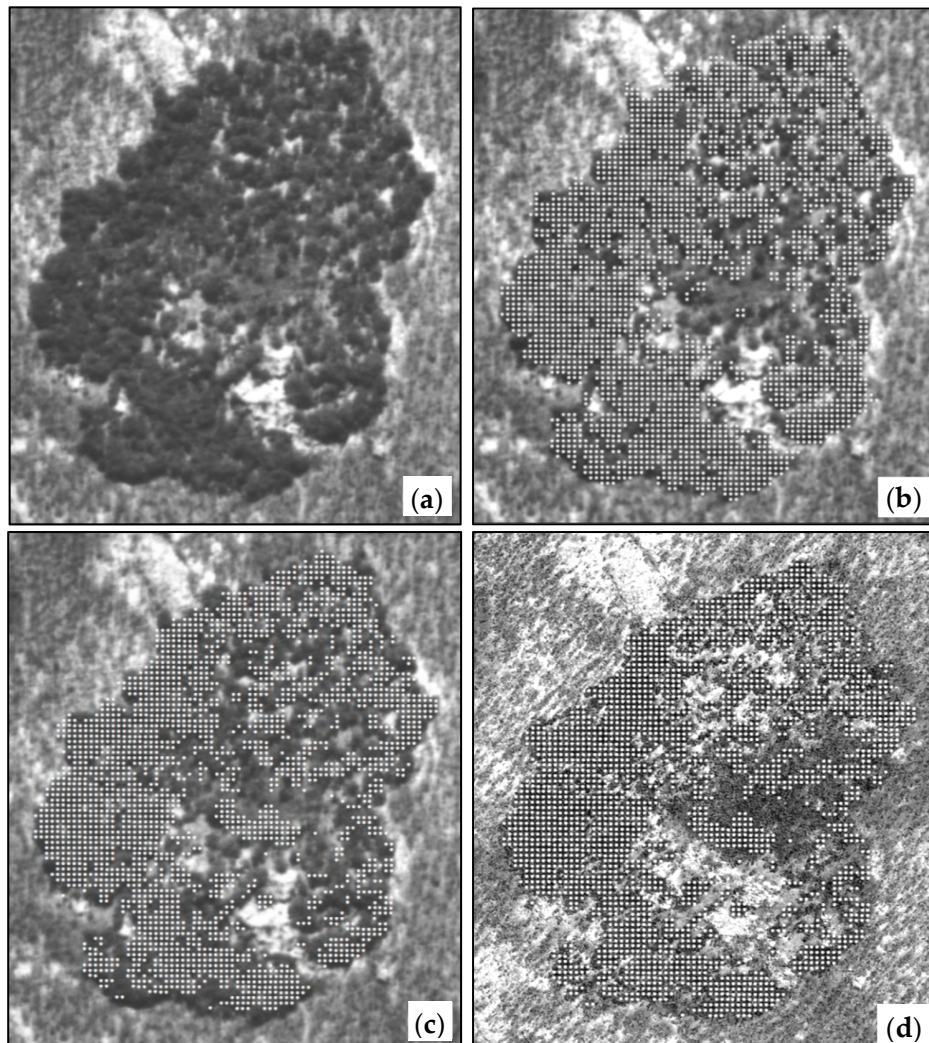


Figure 2. Assessing change at an *Allosyncarpia* sample site: (a) 1950 aerial photo; (b) 1950 image with 1982 canopy cover points; (c) 1950 photo with 2010 canopy cover points; (d) 2010 photo with 2010 canopy cover points.

The 1:25,000 *Allosyncarpia* map was intersected with available and derived GIS datasets, including: digital elevation and slope (derived from the SRTM 1 s (30 m) Digital Elevation Model (DEM) [63], tenure boundaries, biogeographic regions (after Thackway and Cresswell [46]), geology [64], and assembled hydrology and rainfall surfaces. A drainage map layer covering the study area, at 1:100,000, was derived from the DEM (using the hydrology functions: ESRI 2011. ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute, USA). This was refined with reference to existing topographical maps and high-resolution imagery. Drainage lines were allocated a stream order, with an order of 5 or above defined as a ‘major’ waterway. Using the same DEM, slope angles were classified as flat ($0\text{--}5^\circ$), rugged ($5\text{--}20^\circ$), and steep ($>20^\circ$). Available multi-decadal rainfall data for 11 regional rainfall stations (Source: Bureau of Meteorology, Commonwealth of Australia), were used to create an interpolated annual rainfall map for the study area. A GIS layer of 10,000 random points occurring within *Allosyncarpia* forest patches was used to determine relationships between the regional distribution of *Allosyncarpia* forest with above geographic descriptors.

2.3. Canopy Cover Change Assessment

Based on mapping described above of *Allosyncarpia* forest extent, targeted aerial photo analyses and associated field surveys were undertaken at a subset of sites in topographically exposed and semi-exposed situations. Initial selection of potential study sites was conducted via a stratified GIS process designed to produce wide geographic sampling and to meet the requirements of sound aerial photography interpretation. The study aimed to map canopy dynamics of distinct individual forest patches, with topographically unimpeded capacity for expansion; e.g., we did not sample sites located in steep terrain ($>20^\circ$), including cliffs and deeply dissected canyons, which significantly inhibit potential forest expansion into adjacent vegetation, and where aerial photo interpretation of canopy change can be difficult to assess.

A total of 40 discrete *Allosyncarpia* patches, covering a total area of 1201 ha in 1950, were selected for canopy change assessment based on set criteria (Table 1). Size criteria included selection of larger patches (10–150 ha extent), at least 100 m from another patch (>10 ha). Patch shape criteria were applied to exclude highly complex or elongated forest patches based on edge/area, calculated as the percentage of patch perimeter (km) to patch area (ha); and threshold criteria were applied to patches of different sizes (Table 1). *Allosyncarpia* forest associated with major rivers, and within 50 km of the coast, were omitted to exclude effects of potential flooding and cyclonic impacts, respectively. Approximately equal numbers of patches were selected to be representative of relatively rugged sandstone (24 patches), and open sandy (16 patches) terrain, and in eastern (14 patches), western (14 patches) and southern (12 patches) regions.

Table 1. Selection criteria for canopy cover change assessments.

Selection Criteria	Metric
Functional analysis size	Area 10–150 ha
Distinct, isolated, forest patch	Minimum 100 m distance from another patch >10 ha
Functional shape	Patch perimeter (km): area (ha) ratio within parameters, as below: Patch 10–39 ha: <15 Patch 40–79 ha: <12.5 Patch 80–150 ha: <10
Broad geographic distribution	Patches from western, eastern, southern regions, as defined by contiguous 1:250,000 topographic mapping covering the majority of <i>Allosyncarpia</i> 's range
Not associated with major river	Exclude patches associated with stream order >5
Not in path of major cyclone, especially very damaging Cyclone Monica in 2006	Exclude patches <50 km from coast
Topographic variation	Include patches both in relatively subdued sandstone, and open sandy terrain, but exclude patches in steep ($>20^\circ$) terrain to ensure assessment reliability
Image availability	Available historical aerial photos at consistent scales, and 2010 high resolution satellite imagery
Image quality	Amenable for visual interpretation
Cost efficiency	If possible, select more than one patch meeting above criteria per imagery sample

For comparative purposes, near-consistent sample dates were required for images, as well as functionally similar scales of photography or imagery to provide comparable image resolution (Table 1). A GIS database of rectified aerial photo flight line diagrams was created, and pertinent air photos from 1950 (1:50,000), 1982/87 (1:50,000/1:60,000), and high-resolution satellite images from 2010 (sub-1 m pixels), were obtained.

Scanned aerial photos and satellite images were standardised to 1 m resolution in a GIS, and used to determine whole-of-patch canopy cover for each site. The 2010 imagery operated as the base layer,

and historical aerial photos were rectified onto these with very high accuracy (RMS error consistently <2 m). A 10 m grid-point GIS overlay was created for each selected site and period, with attribution of each point as 'canopy' or 'not canopy' (Figure 2). Attribution was undertaken visually since the human eye's capacity to pick up colour, tone, shadow, shape, texture and context exceeds current automated methods [65]. Sampling of ~120,000 points for 3 time periods was undertaken for this assessment. Net change in canopy cover extent was assessed for respective sample patches and time intervals.

Relationships between canopy cover change and frequency of annual and seasonal fire incidence at respective study sites were assessed with reference to the available Landsat imagery archive covering the second canopy cover change assessment period, 1982/87–2009, where: (1) the commencement year for fire incidence assessment was dependent on whether suitable aerial photo coverage was available for respective patches from either 1982 or 1987; (2) the end year describing the last fire season before *Allosyncarpia* mapping imagery was captured in 2010; (3) fire incidence was recorded from available fire mapping data (see below) for each instance where fire mapping overlapped with any portion of *Allosyncarpia* forest canopy extent. Fire mapping data were derived from published records [42,66], supplemented by Landsat imagery available from the online USGS *Landsat Look* viewer (<https://landsatlook.usgs.gov/>). Fire seasonality was determined from available imagery and recorded as occurring in either the early dry season (EDS; before end of July), or late dry season (LDS; August onwards). Fire seasonality is strongly correlated with fire intensity and resultant severity, with LDS fires typically being substantially more severe [55,67]. Two-tailed *t*-tests were used to assess the significance of net canopy cover change at the 40 sites over the assessment period.

The fire susceptibility of respective patches was derived, using the GIS, with reference to an index describing the proportion of a 250 m buffer surrounding each forest patch exposed to flammable savanna vegetation; i.e., not including unvegetated rocky terrain. For illustrative purposes, the fire susceptibility of respective patches exposed to flammable savanna was rated as low (<30%), moderate (30–49%), or high (\geq 50%). At all selected sites, at least some proportion of respective forest patch margins was exposed to flammable savanna vegetation and to allow for topographically unimpeded canopy expansion. Simple linear regressions were used to test for relationships between canopy cover change and EDS and LDS fire incidence in the second assessment period, 1982/87–2010. Differences in canopy cover change at sites with boundaries relatively little exposed to fire versus those more substantially exposed were assessed with two-tailed *t*-tests.

2.4. On-Ground Vegetation Structure Assessment of Canopy Change

Canopy cover change assessments derived from aerial photos at the 40 study sites identified areas where canopy extent had decreased, increased or was stable, using a threshold change in area of at least 2.5%. Considering net canopy cover change data only for the second assessment period, 1982/87–2010, on-ground assessments of *Allosyncarpia* size-class structures were conducted in 2011 at 29 locations from 10 of the 40 study sites broadly representative of *Allosyncarpia*'s geographic distributional envelope. Survey transects (50 × 20 m) were established parallel with and within patch margins, randomly at 14 locations exhibiting decreased canopy extent, 7 exhibiting increased canopy extent, and 8 where canopy extent was stable.

Numbers of established *Allosyncarpia* 'juveniles' (i.e., individuals without cotyledons) were recorded in ten 5 × 4 m quadrats along the central core of each 50 m transect in three classes (<50 cm tall; 50–200 cm tall; >200 cm–<10 cm DBH [diameter at breast height, 1.3 m]). Given the lack of clonal capacity in *Allosyncarpia*, each established juvenile represents an individual developed from seed [52]. The DBH of living and non-living *Allosyncarpia* tree stems \geq 10 cm DBH was measured in consecutive 5 × 10 m quadrats either side of a central tape. Where (dead) stumps were <1.3 height, an estimate of their basal diameter was recorded. Assessments concerning localised impacts of Asian water buffalo were also undertaken but, as considered further in Discussion, impacts were not evident at the great majority of surveyed patches and minor in others.

3. Results

3.1. *Allosyncarpia* Forest Distribution

The revised 1:25,000 *Allosyncarpia* distribution map, based on high resolution (metre-scale) imagery, provides a major refinement over earlier mapping and resulting in a substantial reduction on an earlier estimate of 113,800 ha [38] to 73,500 ha. Using this revised mapping, we determined that 62% of *Allosyncarpia* forest is in the Warddeken Indigenous Protected Area, 32% in Kakadu National Park, and the remainder on adjacent indigenously owned lands. *Allosyncarpia* forest occurrence is evenly distributed within the 1600–1300 mm rainfall isohyets despite only 35% of the Arnhem Plateau receiving mean annual rainfall in excess of 1400 mm (Table 2).

Table 2. Geographic affinities of the total distribution of *Allosyncarpia* on the Arnhem Plateau (29,600 km²), and 40 *Allosyncarpia* forest patches sampled for the canopy cover assessment.

Geographic Context	Arnhem Plateau Study Area (%)	<i>Allosyncarpia</i> Distribution (%)	Sampled Forest Patches (%; No. of Sites Given in Parentheses)
<i>Mean annual rainfall (mm)</i>			
1200–1300	23.0%	0	0
1300–1400	42.0%	34.0%	35.0% (14)
1400–1500	25.0%	32.0%	37.5% (15)
1500–1600	10.0%	33.0%	27.5% (11)
<i>Proximity to Drainage</i>			
<250 m from major river		1.0%	0
<25 m from stream		8.0%	8 sites
<i>Terrain</i>			
steep: >20%	3.0%	14.0%	0%
rugged: 5–20%	23.0%	59.0%	37.5% (15)
flat: <5%	74.0%	27.0%	62.5% (25)
<i>Surface type</i>			
Rocky	51.5%	90.0%	60.0% (23)
Sandy	48.5%	10.0%	40.0% (17)
<i>Interaction between terrain and surface</i>			
Steep x rocky	2.0%	12.0%	0%
Steep x sandy	1.0%	2.0%	0%
Rugged x rocky	17.0%	54.5%	37.5% (15)
Rugged x sandy	5.0%	4.5%	0.0%
Flat x rocky	32.5%	23.0%	22.5% (9)
Flat x sandy	42.5%	4.0%	40.0% (16)

Despite sandstone surfaces comprising only just over half of the study area, 90% of *Allosyncarpia* forest occurs on rocky sandstone surfaces, with the remainder on sandy sedimentary surfaces. Whilst terrain classified as Steep and Rugged comprise 3% and 23% respectively of the plateau surface, they account for 73% of *Allosyncarpia* forest extent; 27% of *Allosyncarpia* extent is located on flat terrain. Just 1% of *Allosyncarpia* forest occurs within 250 m of a major river, and 8% within 25 m of a drainage line (Table 2).

3.2. Canopy Cover Change

Canopy change assessment was undertaken at 40 sites, mostly (62.5%) on flat surfaces, and generally evenly distributed in Rocky and Sandy terrain (Table 2). The distribution of these sample sites, broadly representative of the geographic span of *Allosyncarpia* forest across the north-west sector of the Arnhem Plateau, is given in Figure 1b. Over the 60-year assessment period, canopy cover

declined significantly by a mean of 9.5% at the 40 sample sites (t -test, $p = 0.04$), including a net 3.7% (ns) in the first period 1950–1982/87, and a net 6.1% (ns) in the second period ending in 2010, (Table 3). Over the whole period, canopy cover declined by $>2.5\%$ at 29 patches, increased at 8, and was stable at 3, (Table A1). Canopy cover declined disproportionately at patches occurring under lower rainfall conditions, on flat terrain, and in the generally less rugged (and more fire-vulnerable) eastern sector (Table 3). Collectively, sample sites represented 1.6% of total mapped *Allosyncarpia* extent, and 86% of *Allosyncarpia* forest occurrence (i.e., in non-steep locations) (Table 2).

Table 3. Summary of canopy cover change, 1950–2010, at 40 study sites. Numbers in parentheses = % \pm S.E.M.

Geographic Context	No. of Patches	Canopy Cover Change		
		1950–1982/87	1982/87–2010	1950–2010
<i>Patch trend summary</i>				
Mean trend	40	(-3.7 ± 1.3)	(-6.1 ± 2.1)	(-9.5 ± 2.4)
No. declining patches		22	23	29
No. increasing patches		8	7	8
No. stable patches		10	10	3
Warddeken IPA ¹	24	(-4.9 ± 1.5)	(-8.4 ± 2.4)	(-8.4 ± 2.2)
Kakadu National Park	16	(-1.6 ± 0.5)	(-4.1 ± 1.4)	(-6.0 ± 1.9)
<i>Rainfall zone</i>				
1300–1400 mm	14	-8.1 ± 2.7	-12.2 ± 3.9	-19.2 ± 6.1
1400–1500 mm	15	-2.3 ± 0.5	-2.0 ± 0.7	-4.6 ± 1.5
1500–1600 mm	11	0.0 ± 0.2	-3.8 ± 1.2	-3.8 ± 1.6
<i>Terrain</i>				
Steep: $>20^\circ$	0			
Rugged: $5\text{--}20^\circ$	10	(-2.2 ± 0.8)	(-1.7 ± 0.6)	(-4.1 ± 1.4)
Flat: $<5^\circ$	19	(-7.5 ± 2.3)	(-7.7 ± 2.3)	(-14.3 ± 4.3)
Mixed (Flat and Rugged)	11	(1.4 ± 0.5)	(-7.2 ± 2.5)	(-6.1 ± 2.1)
<i>Surface type</i>				
Sandstone/Rocky	25	(-3.8 ± 1.0)	(-5.3 ± 1.4)	(-10.3 ± 0.5)
Sandy plain	15	(-3.5 ± 1.6)	(-7.3 ± 2.5)	(-9 ± 3.0)

¹ Indigenous Protected Area.

No significant relationships between canopy cover change and fire incidence were observed in the second assessment period, 1982/87–2010, either with frequency of all fires ($F = 3.1$; $p = 0.1$), or with just LDS fires ($F = 1.2$; $p = 0.7$). However, there was an observable difference (t -test; $p = 0.003$) in canopy cover change between those patches with boundaries relatively little exposed to fire (mean = $+2\%$) versus those more substantially exposed (mean = -11%) (Figure 3).

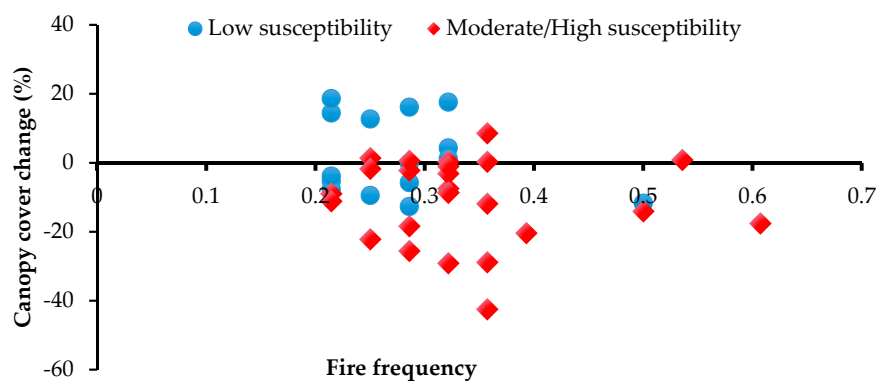


Figure 3. Relationship between fire frequency and canopy cover change (1982/87–2010) at 40 sampled patches, for sites of low vs. moderate/high susceptibility to fire.

3.3. On-Ground Assessment

The size-class distribution of *Allosyncarpia ternata* juveniles, and living and dead tree stems (≥ 10 cm DBH) at the time of survey in 2010, is given in Figure 4. Relative to transects exhibiting stable canopy conditions over the 1982/87–2010 assessment period, assembled data illustrate: (i) significantly less individuals in the <50 cm juvenile class at both decrease and increase canopy cover sites; (ii) significantly less very large (>60 cm DBH) living stems at both decrease and increase canopy cover sites; (iii) substantially more stems in younger tree size classes (10–30 cm DBH) at increase canopy cover sites; and (iv) relatively similar distributions of dead stems at decrease and increase canopy cover sites. Collectively, the occurrence at increase canopy sites of very large dead stems together with large numbers of smaller-sized tree stems is indicative that these transects sampled sites undergoing recovery.

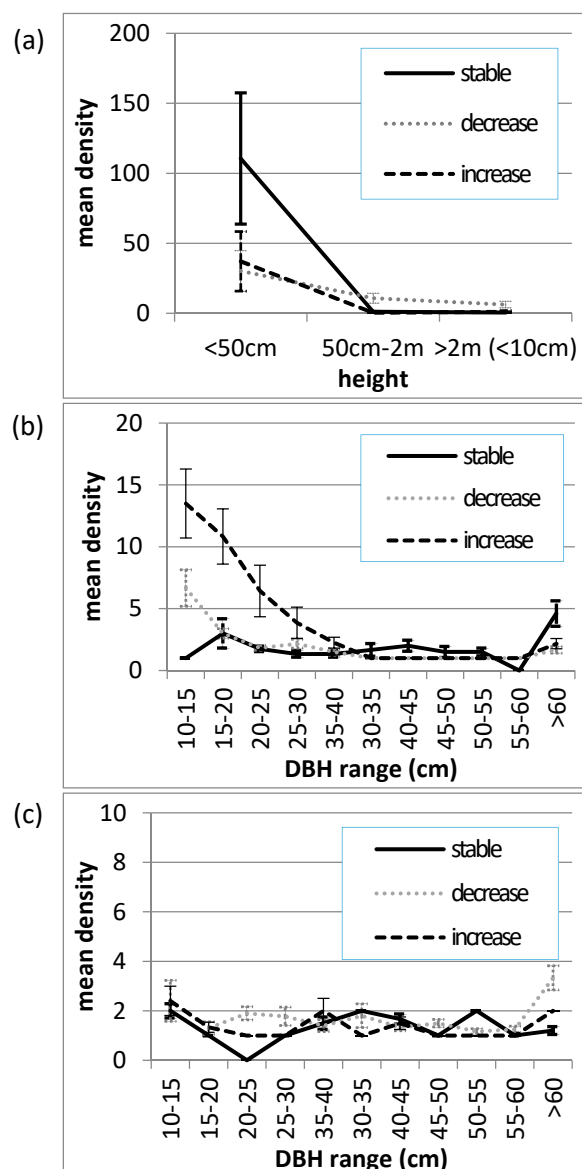


Figure 4. Mean size-class distributions of *Allosyncarpia ternata* (a) juveniles (<10 cm DBH); (b) living stems; (c) dead stems, at 0.1 ha transects ($n = 29$) describing stable ($n = 8$), decreasing ($n = 14$), and increasing ($n = 7$) canopy cover conditions, over the period 1982/87–2010. (Error bars represent the standard error of the mean).

4. Discussion

Current evidence indicates that across the relatively unmodified savanna landscapes of northern Australia (where tree clearing has, to date, been limited), there has been a general, if weak, trend of woody thickening in recent decades [29]. Such an assessment, however, disguises considerable regional variability, for example: little evidence for woody biomass change under lower rainfall conditions [10,68]; marked woody biomass tree death and resultant woody biomass stochasticity under drought conditions especially in lower rainfall savannas [69]; and substantial variability in regional woody biomass dynamics across the 426,000 km² mesic (>1000 mm y⁻¹) northern savannas associated with contemporary fire severity patterning [28].

Similarly, many regional studies support the observation of recent monsoon rainforest expansion in a variety of regional, climatic and topographic contexts [5–7,30–35]. In various cases, the observed expansion of monsoon rainforests has occurred under relatively fire-prone conditions (e.g., [32,33,35]), indicating that enhanced growing conditions associated with increased regional rainfall especially from the 1950s [70,71], reduced evaporation from at least the 1970s [72], global CO₂ fertilisation [14,73], but see [74], and possibly grazing interactions through reducing fuel loads, are likely to have been contributory. Trends of recent woody thickening and encroachment in savanna systems have been reported globally [75].

In contrast, observations reported here indicate that, since at least the 1950s, there has been substantial net decline in the canopy cover of especially relatively fire-susceptible *Allosyncarpia* forest patches. Decline in canopy cover occurred particularly on flat terrain, in regions exposed to predominantly fire-carrying south-easterly dry season winds, and under more marginal rainfall conditions (Table 3). Notably, mean canopy cover increased slightly at topographically less fire-susceptible sites but declined by >10% at 44% of more susceptible sites (Figure 3). Over the period 1990–2009, remote sensing-based studies describing fire regimes encompassing most of the Arnhem Plateau report regional mean fire frequencies of at least 0.37 y⁻¹, of which >60% occurred in the LDS period [42,56]. In the Arnhem Plateau study region, such fire regimes have resulted in significant impacts to fire-vulnerable fauna [44], obligate seeder taxa [5,21,22,39,42,56,76,77], habitats and communities [44,45].

Our observations contrast substantially with those of Bowman and Dingle [33] who reported 21% expansion of *Allosyncarpia* forest over four decades in Kakadu National Park, in the western sector of our study region. A number of methodological issues likely contribute to this disparity. Firstly, Bowman and Dingle [33] (p 709) undertook their canopy cover change, over a 40 year study period (1964–2004), through assessment at 12 geographically distinct sites containing sizeable tracts of *Allosyncarpia* forest ‘designed to capture the environmental variation of *A. ternata* forests in [the park]’. With knowledge of the study sites used by Bowman and Dingle [33] we calculate that their sample included ~58% rugged and steep sites compared to 38% (all rugged) in this study (Table 2). Canopy cover change detection is difficult to assess reliably in steep situations [78].

By contrast, the assessment method in the present study sampled discrete forest patches at seasonally dry sites in both flat and rugged terrain representative of >80% of *Allosyncarpia*'s current distribution, explicitly in fire-susceptible situations. Although not directly assessed here, topographically protected sites are generally associated with very rugged terrain (including deep gorges and scarps), especially in the north-western sector of the Arnhem Plateau (Figure 1b). Those locations support the bulk of *Allosyncarpia*'s current distribution [56], and are infrequently burnt [56].

Similarly, whereas Bowman and Dingle [33] (p. 710) undertook their aerial photo assessment in each period with respect to a grid of 519 one ha random samples containing *Allosyncarpia* in which canopy cover was estimated (in 5% intervals), the present study applied a lattice of 10 × 10 m cells over entire patches (totaling ~120,000 sample points per period), where canopy cover was scored in each cell as either present or absent. In short, the two approaches differ in geographic scope, purpose, and the precision with which canopy cover estimates were derived.

Based on vegetation structure surveys at sites representative of stable, increase, and decrease canopy cover (Figure 5a–c), the present study found that, in contrast to Bowman and Dingle [33], all ‘increase’ sites examined exhibited characteristic post-disturbance recovery size-class distributions (Figure 4)—as opposed to distributions lacking large-sized (mature) dead individuals indicative of canopy or boundary expansion *de novo*. The capacity for *Allosyncarpia* to establish new forest sites through dispersal, or for rapid *Allosyncarpia* forest boundary expansion at decadal scales, is severely restricted, given: short-lived (weeks) dry seeds with no evidence of animal dispersal [79]; gravity-based dispersal is generally limited (except through down-slope and water transport) to beneath, or within metres of adult canopies [79,80]; intolerance of germinants to open insolation [79,80]; intolerance of young juveniles (without lignotubers) to fire [52,80,81]; very slow juvenile growth rates on typical seasonally harsh sites, until individuals develop lignotubers with deep roots [52,82]; and lack of clonal (e.g., rhizomatous) regenerative capacity [38,52]. As illustration, over a ten-year observational period for 20 long-term monitoring plots located across *Allosyncarpia*/savanna boundaries, mean change in the location of outer *Allosyncarpia* canopies was just -0.2 m—with a maximum of $+3$ m at a recovering site, and minima of -3 m at two sites damaged by fire [43].



Figure 5. Photos illustrating examples of (a) stable; (b) decrease canopy; (c) increase canopy (recovery) sites, noting: (a) stable site—typical example of sharply defined *Allosyncarpia* forest boundary embedded in savanna matrix; (b) decrease site—recently fire-killed small stems and very large *Allosyncarpia* stem; (c) increase site—recovery region in foreground interspersed with standing dead *Allosyncarpia* stems, with mature (stable) *Allosyncarpia* forest at rear.

Fire impacts observed directly at ‘decrease’ and ‘increase’ field assessment sites were self-evidently substantial given large numbers of burnt dead stems (Figure 4). Otherwise relationships between canopy cover change and frequency of fire incidence derived from the Landsat imagery record were not particularly strong—perhaps reflecting that the minimum incidence of fire occurrence observed at all sampled sites was still 0.21 fires y^{-1} (Figure 3). As observed here and in other studies, exposed sites are most vulnerable to fire incursions, particularly but not exclusively due to typically severe late dry season fires [33,38,41,43,76,80,83]. Even fires of low intensity can effect stem death, especially where old, large stems are hollowed out by termites [41]. Death of a single large *Allosyncarpia* tree has significant impacts on canopy cover; e.g., from observed relationships between canopy extent, cover and stem size, the loss of a tree 100 cm DBH can immediately reduce canopy cover by ~ 300 m², or 3% of a 1 ha-sized forest patch (J. Freeman, unpublished data).

Other disturbances affecting *Allosyncarpia* forests in the present era include localised major impacts from cyclones and severe storms, and lesser browsing and trampling impacts associated with Asian water buffalo. Immediate effects of severe storms on forest vegetation, exemplified by severe Cyclone Monica in 2006, include substantial tree death, wind-throw and defoliation, whereas longer-term effects include invasion of flammable grasses and increased fire susceptibility [84,85]. Although most cyclonic-scale wind events occur within 50–100 km of the coast, effects of Cyclone Monica were evident up to 130 km inland. Based on historical records for a coastal area immediately to the west of our study region, Cook and Goyens [86] estimated that, within 25 km of the coast, the average recurrence interval for severe events (incurring as much as 40% wind-throw) is ~ 100 years.

As noted in other studies, significant buffalo impacts on regional closed forests typically are confined to sites with available surface water. At seasonally dry sites (which characterise nearly all *Allosyncarpia* forest (Table 2), buffalo grazing has been observed to reduce grassy fuel loads (hence fire risk) in surrounding savanna, as well as promote grass and weed invasion (hence increase fire risk) along ‘fire tongue’ thoroughfares through the interior of forest patches [57,87,88].

In sum, given the generally favourable climatic conditions that have prevailed over the study period, and absence of other evident significant disturbance (e.g., browsing, trampling; disease) drivers, we conclude that fire regimes in the contemporary era have been a major cause driving the landscape-scale decline of especially fire-susceptible *Allosyncarpia* forest sites. A large body of ethnographic and ecologic evidence suggests that contemporary fire regimes over the Arnhem Plateau arose through the latter half of the twentieth century associated with Aboriginal depopulation and the replacement of former intensive fire management practices with a boom and bust wildfire cycle [5,25,26,39,40,57,60–62,77]. Such observations stand in contrast to many other savanna regions, including in Australia, where key drivers influencing woody thickening and encroachment trends have involved reduced or no burning typically in combination with intense pastoral utilisation, enhanced rainfall and evapotranspiration conditions, and possible CO₂ fertilisation [11,29,75,89–91]. The present study serves as a useful reminder that, in fire-prone savanna landscapes, severe fire regimes can substantially outweigh the woody growth-enhancing effects of other regional and global-scale drivers.

Conservation Management

Despite the vast majority of *Allosyncarpia*'s distribution occurring today in regional conservation reserves (Figure 1b), very significant logistical challenges confront the implementation of conservation-effective fire management over this vast, very sparsely inhabited, rugged region. Fire management in Kakadu National Park's sandstone uplands has proceeded progressively with establishment of three main Park Stages, 1979–1987. Kakadu is responsible for conservation management of 32% of *Allosyncarpia* forest extent (Table 2). Recent assessment of fire regimes affecting *Allosyncarpia* forest in Kakadu show that effective management, as measured against specific performance criteria, increased progressively over the period 1989–2011 [92], but has then declined significantly [93]. The latter study demonstrates that over a 10-year assessment period, 2006–2015, fire management of *Allosyncarpia* forest has consistently not met minimum ecological criteria (i.e., <10% of forest extent should be affected by one severe fire in any five-year period), including the years leading up to 2011.

The great majority (62%) of *Allosyncarpia* forest is included in Aboriginal-owned lands of the Warddeken Indigenous Protected Area (IPA), established in 2009. From the early 20th century until 2006, very limited fire management was undertaken in the Warddeken IPA region, its Aboriginal landowners mostly having dispersed to surrounding settlements [62]. Fire regimes were dominated by very extensive LDS wildfires [23], with *Allosyncarpia* forest margins being burnt a mean frequency of 0.2 y⁻¹, including 0.16 y⁻¹ in the LDS, over the period 1990–2005 [42]. Thereafter, fire regimes in the Warddeken IPA have improved markedly associated with the implementation of a commercial landscape-scale fire management project aimed at reducing greenhouse gas emissions; for example, in the period 2010–2014 just a third of all fire extent in sandstone areas (annual mean = 0.26 fires y⁻¹) occurred as LDS fires [94]. That program has also involved reinstatement of preventative fire management around culturally significant *Allosyncarpia* forest sites.

Although it is still too early to assess the longer-term ecological benefits associated with the re-imposition of more conservative fire regimes in the Warddeken IPA, one lesson is clear: strategically implemented commercial savanna burning projects can substantially help reduce the frequency of LDS wildfires. As of 2017, two new savanna burning projects managed by Aboriginal people are in the process of being established in Kakadu National Park.

5. Conclusions

In summary, we report that through a multi-temporal change detection process, mean canopy cover of 40 sampled representative *Allosyncarpia ternata* patches decreased by 9.5% since the 1950's, where canopy loss was greatest at the most fire-susceptible patches. Importantly, canopy expansion represents vegetation recovery, rather than patch expansion. The study demonstrates the vulnerability to wildfire of exposed *Allosyncarpia* margins, and the magnitude of the effect, over and above regional and global drivers of woody growth enhancement.

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Appendix A

Table A1. The 40 individual *Allosyncarpia ternata* sites describing: the position (longitude and latitude) of the centroid of the patch; the initial patch area in 1950; the change in the patch area between the three assessment periods (1950–1980, 1980–2010 and 1950–2010); the proportion of the patch perimeter surrounded by fire-prone vegetation; and the categorical fire susceptibility based on the proportion of surrounding fire-prone vegetation and the topographic position of the patch. The site name lettering (a–e) infers a sub-patch of *Allosyncarpia* within a given local area.

Site Name	Long.	Lat.	Area in 1950 (ha)	%Change			%Fire-Prone	Fire Susceptibility
				1950–1980's	1980's–2010	1950–2010		
Mimolorrk a	133.7130	−12.4673	79	0%	−12%	−12%	25	Low
Mimolorrk b	133.6800	−12.4266	29	2%	−18%	−16%	35	Moderate
Tin camp south a	133.1920	−12.5620	24	3%	1%	4%	40	Moderate
Tin camp south b	133.1920	−12.5620	32	−5%	−9%	−14%	35	Moderate
Korbelak a	133.6180	−12.6217	27	−10%	−29%	−36%	60	High
Korbelak b	133.6320	−12.6204	48	−7%	−22%	−28%	75	High
Kondulgai a	133.7610	−12.7112	23	−2%	−20%	−22%	55	High
Andulgai b	133.7360	−12.7289	14	−7%	13%	5%	10	Low
Makkalarl head a	133.6410	−12.6631	15	3%	0%	3%	45	Moderate
Makkalarl tail b	133.6410	−12.6631	35	−5%	−42%	−46%	75	High
Nakarriken b	133.5880	−12.7226	36	−10%	−11%	−20%	80	High
Yidngarremanneng a	133.9770	−12.7566	122	−16%	9%	−9%	60	High
Yidngarremanneng b	133.9680	−12.7414	35	−8%	18%	9%	2	Low
East alligator a	133.3040	−12.7953	14	12%	19%	32%	25	Low
East alligator b	133.3100	−12.7935	22	−8%	14%	5%	30	Moderate
Maguk a	132.4050	−13.3153	13	2%	−14%	−13%	40	Moderate
Maguk b	132.4120	−13.3102	8	−1%	1%	0%	55	High
Magela creek a	133.1690	−12.8224	12	−6%	−2%	−7%	65	High
Namarrgon creek a	133.1190	−12.9210	14	−8%	−6%	−14%	30	Moderate
Namarrgon creek b	133.1190	−12.9210	15	−2%	−5%	−7%	5	Low
Namarrgon creek d	133.0990	−12.8991	12	0%	1%	1%	25	Low
Hill u662 a	133.1930	−12.9500	13	10%	−13%	−4%	25	Low
Hill u663 b	133.2280	−12.9214	32	−5%	−8%	−12%	25	Low
Deaf Adder north b	132.5700	−13.3727	4	−6%	−2%	−8%	70	High
Deaf Adder north c	132.9590	−12.9667	15	0%	−4%	−4%	25	Low
Deaf Adder north d	132.9850	−12.9641	24	5%	−9%	−5%	5	Low
Deaf Adder north e	132.9850	−12.9641	10	−1%	−9%	−9%	40	Moderate

Table A1. Cont.

Site Name	Long.	Lat.	Area in 1950 (ha)	%Change			%Fire-Prone	Fire Susceptibility
				1950–1980's	1980's–2010	1950–2010		
Kunbambuk a	133.5820	−12.9612	5	−5%	−9%	−14%	60	High
Kunbambuk b	133.5780	−12.9576	36	−21%	−29%	−44%	60	High
Kunbambuk c	133.5630	−12.9465	18	−26%	−18%	−40%	60	High
Round jungle a	133.0990	−12.8991	148	1%	1%	2%	65	High
Round jungle b	132.6550	−13.3294	37	−3%	−7%	−10%	50	High
Round jungle c	132.6480	−13.2931	55	4%	−1%	3%	55	High
Round jungle d	132.6480	−13.2931	5	10%	0%	10%	25	Low
Barramundie gorge a	132.5620	−13.3912	22	−5%	−3%	−8%	50	High
Barramundie gorge c	132.5700	−13.3727	9	6%	−26%	−21%	33	Moderate
Barramundie gorge d	132.5700	−13.3727	21	−8%	4%	−4%	15	Low
Gimbat north a	133.3100	−12.7935	57	−5%	−12%	−17%	40	Moderate
Gimbat north b	132.9190	−13.4550	35	−3%	0%	−3%	40	Moderate
Gimbat north c	132.9400	−13.4172	26	−22%	16%	−9%	20	Low

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