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North-Kimberley
Landscape Conservation Initiative

2010–12
Performance Report

North-Kimberley Landscape Conservation Initiative: 2010–12 Performance Report

Department of Parks and Wildlife
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The Landscape Conservation Initiative (LCI) Monitoring and Evaluation Framework was developed over several years by the Department of Parks and Wildlife's (DPaW) Kimberley Region staff (principally Erica Shedley but including Amanda Moncrieff, Ed Hatherley and Daryl Moncrieff), with considerable input and guidance from Nature Conservation Division (Gordon Wyre), Science Division (Neil Burrows), and research staff from the Australian Wildlife Conservancy (AWC). A workshop hosted at Mornington Station by AWC in November 2009 was invaluable in raising ideas and establishing protocols for the framework.

The gains made in terms of meeting fire management targets presented in this report are the result of a collaborative approach to fire management in the North-Kimberley and have involved the Australian Wildlife Conservancy (through the EcoFire project), the Uunguu Rangers and the Wunambal Gaambera Aboriginal Corporation, the Wunggurr Rangers and the Wilinggin Aboriginal Corporation, the Balanggarra Rangers and the Balanggarra Aboriginal Corporation, and the Dambimangari Rangers and the Dambimangari Aboriginal Corporation. In short, the positive trends in fire management in the North-Kimberley are the result of many groups working together.

Many regional DPaW staff have contributed to meeting fire, invasive animal and plant targets through multiple works programs delivered through the Landscape Conservation Initiative. This includes aerial cattle culls, weed management and prescribed burning, as well as implementing the monitoring program itself. The LCI project has also involved many DPaW staff from outside of the Kimberley. Ricky van Dongen, Bart Huntley and Graeme Behn (from the Remote Sensing section of GIS Branch) have been working on the Rapideye and Landsat imagery analysis (as well as the initial field work) which forms a component of this report. Similarly, Janine Kinloch and Georgina Pitt (from the GIS Applications section of GIS Branch) ran the fire scar analysis and assisted in the development of the metrics used to report against meeting fire targets. These metrics are based on methods developed by Sarah Legge, Terry Webb and colleagues from AWC. The LCI project has benefited greatly from support and advice from DPaWs Science Division, in particular Norm McKenzie and Greg Keighery. Butch Maher, from All North Helicopters has played a critical role in AWC's EcoFire project and DPaWs aerial cattle culls. The Land and Sea Management Unit at the Kimberley Land Council, in particular Ari Gorring, Sam Bailey, Rob Warren, Anna Pickworth, Rowan Clarke and Danyl Wolfe, and Tom Vigilante from Bush Heritage Australia, have played an important role in joint planning and facilitation with Native Title and ranger groups.

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The views expressed in this report, reflect those of the authors and not necessarily those of the collaborating partners. Any errors or omissions remain the responsibility of the authors.

SUMMARY

Context. Despite its relatively intact landscapes, northern Australia has been experiencing worrying biodiversity declines, with strong evidence that these herald an imminent wave of extinctions – similar to those experienced in southern Australia. The vast and remote North-Kimberley has been far less affected by these declines, and is considered a stronghold for biodiversity.

Aims. The Landscape Conservation Initiative (LCI) was established under the Kimberley Science and Conservation Strategy (KSCS) and aims to retain and enhance current natural biodiversity and landscape values in the North-Kimberley.

Key results. Rainforest patches have been mapped using Rapideye remote sensing technology and will provide a baseline for measuring changes in both the extent and canopy cover of these patches. Native vegetation condition monitoring plots have been established in conjunction with small mammal monitoring plots and Landsat satellite imagery is being used to monitor changes in canopy cover at these and additional sites. Native vegetation and small-medium sized mammal abundance and diversity have (on average) remained stable from baseline surveys, although there are species of concern, most notably the black-footed tree rat (*Mesembriomys gouldii*). Early results suggest that both increasing vegetation age and patchiness are correlated with mammal abundance and diversity at several spatial scales. Similarly, increasing fire frequency was negatively correlated with both mammal abundance and diversity. This provides further support for the notion that both fire size and frequency are drivers of mammal decline in northern Australia. Early results also suggest a negative correlation between indices of cattle damage at monitoring sites and mammal abundance and diversity. Management of fire has shifted the seasonality of burning from predominately late to a mix of early and late season fires. Late season fires made up 76.3 % of annual fire scars from 2000–07 but from 2008–12 this was reduced to 53.2 %. However, the overall extent of annual fire scars has not changed significantly (47.7 % from 2000–07 vs. 39.6 % from 2008–12). The availability of unburnt vegetation has improved. The mean distance from burnt areas to the nearest unburnt vegetation has decreased from 3.8 km to 1.7 km for all unburnt vegetation, from 4.6 km to 2.2 km for three year old vegetation and from 8.9 km to 5.1 km for five year old vegetation. The extent of old-growth vegetation has not improved (vegetation that is 1–2 years of age still dominates), but is now more patchily distributed across the landscape. Fire frequency has improved between the periods 2003–07 and 2008–12. A total of 12,564 feral cattle were removed from areas of the LCI during the period 2007–08 to 2011–12, delivering 12–46 % reductions in population size. Cattle damage monitoring plots were established in high-priority areas to measure concurrent reductions in landscape impacts. Weed management has concentrated on high priority species (e.g. Weeds of National Significance) in high priority areas, particularly those with high public visitation.

Conclusion. Whilst it is very early days in the LCI program, the collaborative approach to threat management – in particular fire – across the North-Kimberley by DPaW, the Australian Wildlife Conservancy and the Balanggarra, Dambimangari, Uunguu and Wunggurr ranger groups demonstrates that threats to biodiversity can be managed across protected area boundaries.

Implications. Based on early monitoring data and lessons learnt from 2010–12, we suggest the following activities and priorities as part of the LCI adaptive management program:

Management of fire

- Continue to decrease distances between burnt and unburnt vegetation;
- Continue to reduce the extent of late dry season fire scars;
- Increase the extent of old-growth vegetation; and
- Reduce fire frequency across inland and southern regions of the LCI.

Management of feral cattle

- Continue to reduce cattle numbers and their impacts;
- Undertake feral animal aerial surveys and derive sound population estimates; and
- Align current management zones with priorities outlined in the Healthy Country Plans of partners.

Management of weeds

- Implement CyberTracker technology for mapping the occurrence of weed infestations and recording on-ground weed management; and
- Take a more strategic approach to weed management.

Biodiversity indicators

- Undertake analysis of vegetation, fire scar, cattle density and impact, rainfall and mammal data with complex ecological models to better understand the roles that threatening processes play in the persistence of biodiversity across the North-Kimberley.

In all cases, threat management and monitoring programs should align with those of collaborating partners.

BACKGROUND

The Kimberley is one of the most ecologically important regions in Australia, home to both diverse and unique assemblages of plants and animals. Whilst northern Australia is considered relatively intact in comparison with other landscapes in Australia, ecologists as well as land and wildlife managers are concerned by the rapid decline of biodiversity, particularly of small to medium sized mammals (Fitzsimmons et al. 2010; Woinarski et al. 2010; Ziembicki et al. 2013) and granivorous birds (Franklin 1999; Franklin et al. 2005; Murphy et al. 2010; Woinarksi and Legge 2013), across northern Australia. There are strong indications that current declines herald an imminent wave of extinctions, similar to that experienced in other regions of Australia (Johnson 2006; McKenzie et al. 2007; Fisher et al. in press).

Biodiversity values across northern Australia are threatened by a range of pervasive and interacting processes: inappropriate fire regimes (Andersen et al. 2005; Legge et al. 2008), introduced and domestic herbivores (Legge et al. 2011a), feral predators and prey (Johnson 2006; Woinarksi et al. 2011; Phillips et al. 2003) and invasive plants (Rossiter et al. 2009). The Kimberley has been far less affected by these declines, in particular the North-Kimberley bioregion (Fig. 1), which is considered a bastion for wildlife (McKenzie et al. 2007; Start et al. 2007; Burbidge et al. 2008; Start et al. 2012).

The Landscape Conservation Initiative (LCI) was established under The Kimberley Science and Conservation Strategy (Government of Western Australia 2011), to retain and enhance current natural biodiversity and landscape values. The LCI is managed by the Department of Parks and Wildlife (DPaW; formerly the Department of Environment and Conservation) in collaboration with Native Title and Indigenous ranger groups, environmental NGOs and pastoralists in a tenure blind approach to protect biodiversity values in the North-Kimberley. This is currently being achieved by managing: (1) inappropriate fire regimes; (2) the impacts of feral animals; and (3) the impacts of invasive plants across an area of more than 65,000 km^2 that includes DPaW managed conservation estate, pastoral estate, lands held under Native Title and Indigenous Protected Areas, and private conservation estate (Fig. 1).

A monitoring and evaluation program was established to measure and report on progress towards achieving these goals (see Shedley et al. 2012) and to ensure that the increased funding in threat management is achieving positive benefits for biodiversity. Indicators for measuring management effectiveness and reporting on the overall condition of biodiversity at a landscape scale are: (1) rainforest patch extent and condition; (2) small–medium sized mammal diversity and abundance; (3) native vegetation condition.

Rainforest patches are considered key habitat across the North-Kimberley, and are one of the most sensitive indicators of landscape health (Russell-Smith and Bowman 1992). These are remnants of the natural environment dating back millions of years and many species depend on these for their continued persistence in the region. Rainforest patches are threatened by a combination of inappropriate fire, feral animals and weeds (Russell-Smith and Bowman 1992). These threatening processes will result in reductions in the extent and condition of rainforest patch sizes (McKenzie 1991) and if unmanaged, losses of entire patches (but see Banfai and Bowman 2006). Retaining the extent and quality (species diversity and vegetation structure) of rainforest patches by managing landscape-scale threatening process is fundamental to the desired LCI outcome and is indicative of healthy landscapes more broadly.

Small to medium sized mammals (those weighing < 5 kg) that inhabit the ground layer of vegetation are regarded as one of the most vulnerable groups of native animals in northern Australia (Woinarksi et al. 2011). This group of animals has suffered extinctions elsewhere in northern Australia as a result of habitat degradation by inappropriate fire regimes and introduced herbivores, predation by feral predators and poisoning from introduced animals (Johnson 2006; Woinarksi et al. 2010). Thus, retaining an intact assemblage of smaller native mammals in the North-Kimberley will be a key indicator of the success of this program.

Finally, landscape health will be measured via long-term changes in vegetation and substrate condition. Changes in vegetation cover and structure (including trees, shrubs and grasses) and substrate condition (including litter and soil) will indicate changes in overstorey dominance, structural and floristic diversity, and site productivity which lead to changes in habitat condition (McKenzie et al. 2007). This measure is deemed responsive to gross landscape vegetation changes through impacts of fire, feral animals and weeds (Shedley et al. 2012), as well as rainfall trends (Lehmann et al. 2009), and will be monitored across the North-Kimberley.

Performance in achieving management actions (see Tables 4, 10) will be measured locally against North-Kimberley historical work and baseline surveys (e.g. Bradley et al. 1987; Palmer 2004; Start et al. 2007; 2012; Radford et al. submitted) and regionally, against more recent trends observed across northern Australia (e.g. Woinarski et al. 2001; Corbett et al. 2003; Andersen et al. 2005; Legge et al. 2008; Woinarski et al. 2010; Legge et al. 2011b). This performance report for the LCI covers the period 2010–12. Although increased fire management in the North-

Kimberley – in close collaboration with the Australian Wildlife Conservancy (EcoFire, Legge et al. 2013), the Kimberley Land Council (KLC) and the Balanggarra, Dambimangari, Wilinggin and Wunambal Gaambera Aboriginal Corporations– began in 2008, monitoring under the LCI program did not begin in earnest until 2010–11.

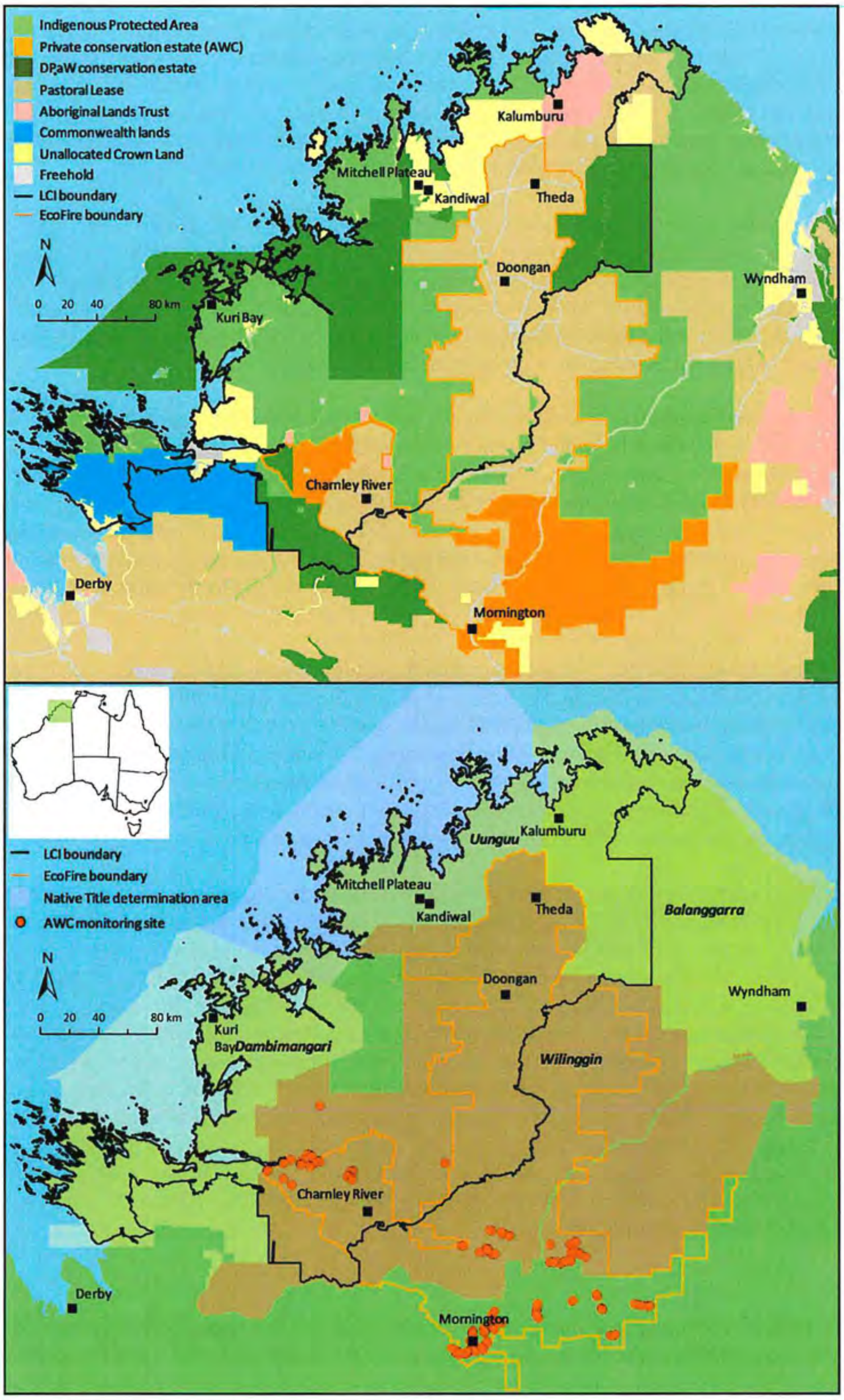

Fig. 1. The North-Kimberley region with dominant land tenures (A) and the same region with Native Title determinations (shaded areas) and the LCI (black line) and EcoFire (orange line) project area boundaries (B).

APPROACH

The Landscape Conservation Initiative (LCI) project area

The LCI project area (65,194 km^2 in total) contains a complex mix of DPaW managed conservation estate, unallocated Crown land, pastoral leases (including those managed principally for conservation), Commonwealth managed lands, and Aboriginal Lands Trust estate (Fig. 1a). Native Title has been determined over nearly the entire area, with much of it recognised as Exclusive Possession Native Title (Fig. 1b). Indigenous Protected Areas (IPAs) have also been declared over large areas (Fig. 1a). The LCI area is delineated by the Mitchell sub-region of the North-Kimberley IBRA (Interim Biogeographic Regionalisation for Australia) region, with the addition of Drysdale River National Park to the east and the King Leopold Ranges Conservation Park (north of Gibb River Road) to the south (Fig. 1a).

The Balanggarra, Dambimangari, Wilinggin and Wunambal Gaambera Native Title groups have all developed Healthy Country Plans to assist with management of their whole country, as well as their declared IPAs (Wunambal Gaambera Aboriginal Corporation 2010; Balanggarra Aboriginal Corporation 2011; Dambimangari Aboriginal Corporation 2012; Wilinggin Aboriginal Corporation 2012; see also Moorcroft et al. 2012). All provide clearly articulated targets for managing fire, feral animals and weeds, with activities implemented by Indigenous Ranger groups which are mostly funded by the Commonwealth Government's Working on Country (WoC) program.

The expansion of on-ground activities throughout the LCI area relating to fire, feral animal and weed management is being negotiated and planned with individual Aboriginal Corporations and Indigenous Ranger groups. For some groups the preference has been to instigate collaborative projects to help build capacity and develop working relationships, and for others there has been a preference to negotiate broader agreements for the management of country, some of which now include formal Joint Management Arrangements. In all instances, there are pre-existing Healthy Country works programs being implemented by WoC funded Ranger groups, and these activities complement the objectives and scope of the LCI, that being the protection of biodiversity values through the management of threatening processes.

Until recently (2011–2012) annual fire management plans have been developed and implemented collaboratively by DPaW, the Kimberley Land Council (KLC) and the four Native Title Groups. Each of the claim groups also participate in the North Kimberley Fire Abatement Project (NKFAP), which is coordinated by the KLC. The NKFAP focuses on the creation of firebreaks that aim to compartmentalise country and exclude fire from areas in order to maximise potential income from carbon credits (areas unburnt or burnt early in the season). During 2012 the KLC and DPaW delivered early prescribed burning programs – in partnership with the four Native Title claim groups – but, with differing objectives, i.e. the creation of fire control blocks (NKFAP) vs. patch mosaic burning (DPaW).

The LCI area also includes a large section of the EcoFire project area (Figs. 1a, b), which is coordinated by the Australian Wildlife Conservancy (AWC, Legge et al. 2011b; 2013). EcoFire is a regional fire management project covering 39,360 km^2 of the central and north Kimberley (Figs. 1a, b; 16,998 km^2 (26.1 %) of the EcoFire project area lies within the LCI area) and is a partnership of pastoralists, Indigenous communities, AWC, DPaW and the Department of Fire and Emergency Services which have collaborated since 2007 to manage fire. The objective is to reduce the occurrence and impacts of late dry season wildfires and increase the area of old growth vegetation. This is achieved by delivering a prescribed burning program that is coordinated across property boundaries and land tenures. The extent to which the project has modified fire patterns has been significant (Legge et al. 2013). The impacts of changes in fire patterns on biodiversity indicators are measured at numerous sites on those pastoral leases directly managed by AWC (Fig. 1b) in the central Kimberley. AWC also undertake their own feral animal and weed management programs on some of these areas. EcoFire has previously been funded by Rangelands NRM, the Commonwealth Government's Caring for our Country program, and more recently by DPaW through the Kimberley Science and Conservation Strategy. DPaW now funds EcoFire in its entirety.

Reporting

Reporting under the LCI Performance Reporting Program (this report) only includes outcomes from DPaW related activities, or those undertaken as fee-for-service arrangements. The exception to this is the fire scar analysis of prescribed burning carried out within the LCI area. DPaW has worked very closely with Native Title groups (and their Rangers), the Kimberley Land Council and AWC since 2008 to manage fire across multiple land tenures in the North-Kimberley. Thus, the fire metrics reported against in this document represent the collaborative efforts of many groups working together to better manage fire. For the purpose of this report it would be counter-productive to dissect the LCI area into smaller compartments because (1) the report is about regional, gross fire patterns; (2) multiple boundary intersects create artificial fire pattern metrics; and (3) the units of analysis need to be large relative to the historical size of late dry season fires.

Site selection and stratification

Monitoring sites (Fig. 2) were selected based on historical survey locations (e.g. Bradley et al. 1987; Start et al. 2007; 2012; Radford et al. submitted) and stratified by geological vegetation types (Fig. 3). These included: sandstone scree (rugged sandstone boulder fields with low tree cover), sandstone woodland (sandstone woodland with hummock grass spinifex understory), Carson volcanic riparian (black clay soil associated with creek line habitat), Carson volcanic woodland (sparse woodland on clay), rainforest or vine thicket scrub (closed forest on laterite-volcanic interface) and laterite forest (forest on lateritic plateau). Within the constraints of historical survey sites, selection was based on the sites containing 'typical' habitat attributes. For instance, laterite forest was selected based on dominant tree species composition (e.g. *Eucalyptus tetrodonta*, *E. miniata*, *Corymbia nesophila* and *Livistona eastoni*). On-ground monitoring sites (mammals and vegetation) are referred to throughout this report as 'clusters' based on geographic groupings of individual sites.

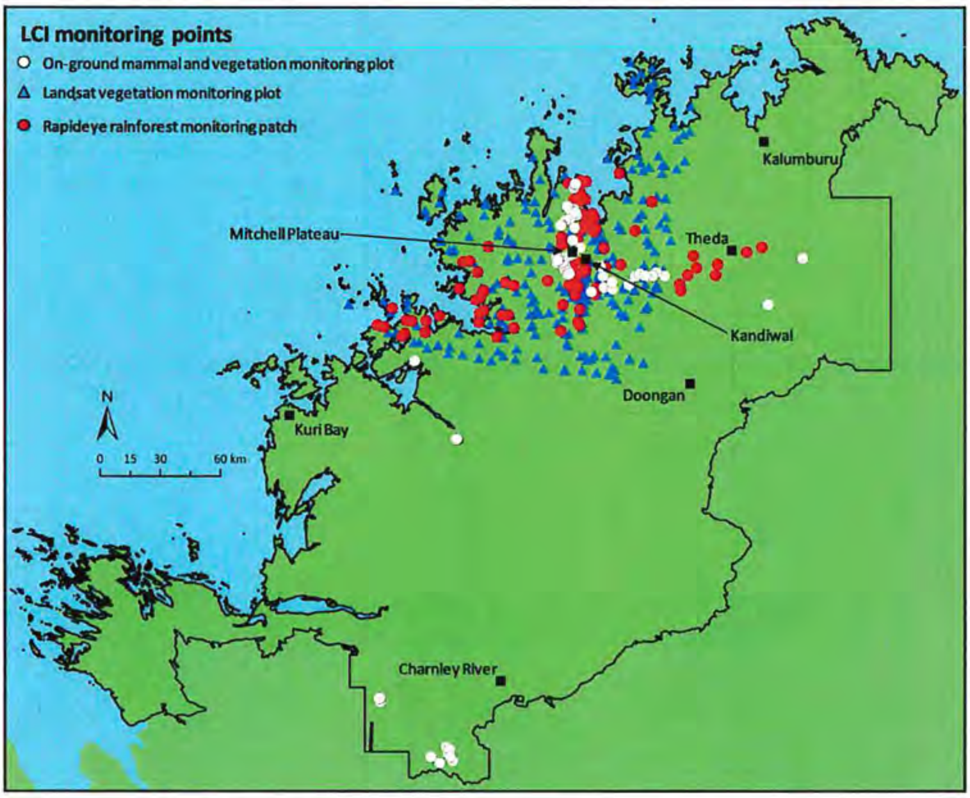

Fig. 2. Monitoring plots in the LCI project area as of December 2012.

Fig. 3. Major vegetation types used in the stratification and selection of LCI monitoring sites.

Climatic variables

The tropical savannas of northern Australia experience extremes of high rainfall during the monsoonal 'wet season' (November/December–March/April) and the near absence of rainfall in the intervening 'dry season' (May/June – October/November), as well as unpredictability in the quantity that falls and its timing (Taylor and Tulloch 1985). Historical and current rainfall records were obtained from Bureau of Meteorology stations deemed representative of annual rainfall patterns at LCI sampling clusters throughout the north Kimberley (Table 1). Summary statistics were generated from historical records and graphed against wet season rainfall records (Fig. 4). With the exception of Mitchell Plateau and Kuri Bay, all stations are still active (Table 1). The station at the Mitchell Plateau will be on line again in 2014.

Table 1. Bureau of Meteorology weather stations used to obtain rainfall records for Landscape Conservation Initiative monitoring sites and associated mean monthly rainfall records.

BoM station	Rainfall isohyet (mm)	Mean monthly rainfall (mm)												Mean wet season rainfall (mm)
		Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	
Mitchell Plateau	1400	9.1	1.4	12.1	50.6	126.9	182.1	381.4	364.5	321.6	41.3	24.9	8.9	1524.0
Kuri Bay	1200	4.7	1.2	1.3	8.9	26.9	162.7	408.5	341.9	306.3	67.1	52.5	8.6	1375.4
Kalumburu	1100	0.3	0.3	1.8	38.8	80.4	235.8	314.4	308.7	234.9	46.6	18.1	3.5	1279.0
Theda	1100	3.8	1.3	10.4	59.0	108.4	199.8	302.4	280.9	225.7	55.5	18.0	2.6	1245.5
Doongan	1000	0.4	1.0	5.4	45.0	115.1	222.7	303.8	251.4	208.5	49.5	12.0	5.9	1216.7
Charnley River	800	4.0	3.2	2.9	17.6	49.9	157.6	256.2	239.6	172.8	48.0	13.6	6.5	965.3

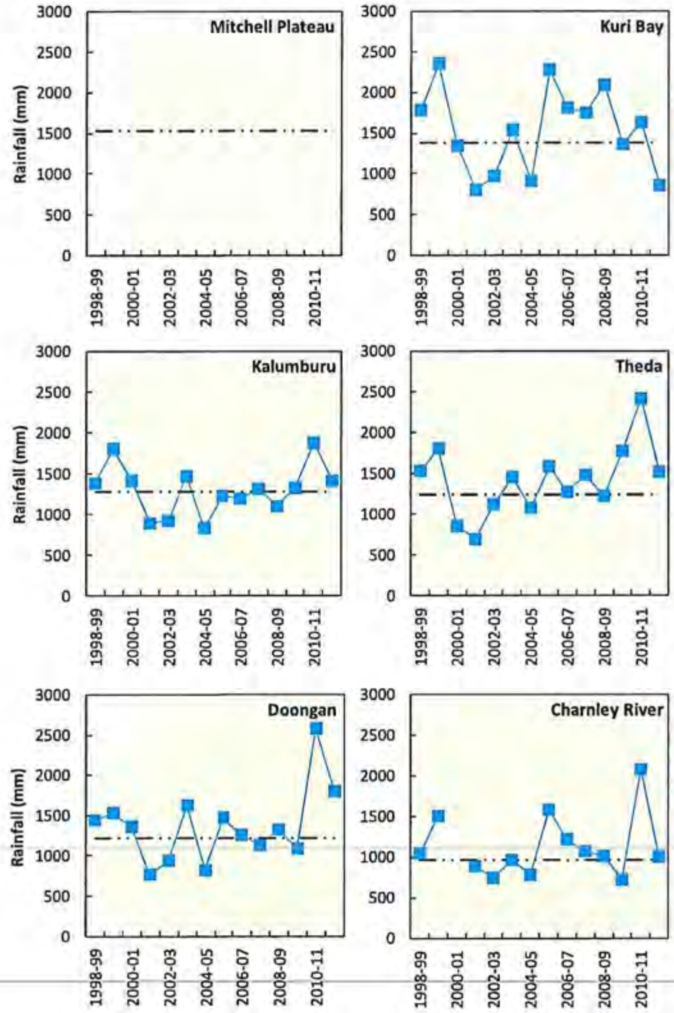

Fig. 4. Annual wet-season rainfall records (blue squares) vs. long-term averages (black dashed line) for Bureau of Meteorology stations considered representative of LCI monitoring sites (see Table 1).

Native vegetation condition

Habitat variables, such as type and distance to closest water body, landform pattern, landform element and soil surface texture and colour are described for each site following methods outlined by CSIRO (2009). Vegetation and ground layer habitat are quantified along a 50 m transect running diagonally across mammal trapping quadrats (see below). A line intersect method is used to estimate perennial grass, annual grass, forb, leaf litter, sub-shrub, shrub and tree, log (> 50 mm diameter), exposed rock and gravel, bare ground and termite mound cover along the transect. Major changes in vegetation attributes along the transect are noted, and the change described (for instance, change from grass dominated to litter dominated understorey or transition from more recently burnt to longer unburnt vegetation).

Litter and soil layers are described for each quadrat as follows: Erosion pegs are placed at three points chosen to represent major components of vegetation architecture (e.g. under tree canopy, in the open away from tree canopy, etc.). Pegs are driven deep enough into the ground to remain stable over time, and the distance from the top of the erosion peg to the compacted soil or litter surface is measured annually to determine soil mobility over time. Within close proximity of each erosion peg, litter and soil layers are exposed by slicing into the ground with a metal spatula. The depth of intact litter (unbroken leaves and stems), fragmented and decaying litter, crust, worm cast layer and organic mineral soil are then measured.

Woody vegetation structure is estimated from the central point of each plot (37 m along vegetation transect). An estimate of projected canopy cover (%) and basal area ($m^2.ha^{-1}$) is made for dominant plant species within each of the major vegetation strata (e.g. trees > 4 m; shrubs < 4 m; low shrubs). A Bitterlich gauge (Lindsey et al. 1958) is used to estimate projected canopy cover. Basal area is estimated using a factor one forestry prism. For each dominant species notes are taken in regards to height, presence of different life stages, condition and reproduction.

Disturbance indicators recorded for every site include a visual rating of fire status including the percentage of the quadrat burnt, scorch height and fire intensity on a standard savanna scale (Russell-Smith and Edwards 2006; Edwards 2009). Impacts of cattle at each site are assessed in terms of (1) number of cattle sighted, (2) grazing level, (3) tracks and trampling (4) amount of dung present and (5) the percentage of vegetation affected within the site. Sites are then ranked from 1–5, based on the number of categories with values. For instance, a site with a score of 5 has values for cattle impacts across all 5 categories, whereas a site with a score of 0 has no values across categories. Weed infestations are recorded including information on extent, density and invasiveness.

Several other habitat and vegetation attributes are measured at monitoring sites, and will feature in future analyses, once there are more than two years of data, because there is no previous data for comparison with current values. These include: fauna habitat (tree hollows, vegetation cover, rock crevices and hollow logs) and food availability values (fruiting plants, flowering vegetation, seeding vegetation, termite mounds), as well as fire sensitive species such as obligate seeders, (e.g. *Callitris intratropica*) and rainforest species, based on previous studies (Russell-Smith and Bowman 1992; Woinarski and Fisher 1995; Russell-Smith et al. 1998; Bowman et al. 2001; Vigilante and Bowman 2004a; Woinarski et al. 2004; Prior et al. 2011; Radford et al. 2013).

Landsat satellite imagery[1] is being used to monitor vegetation condition over time following methods adapted from Zhu et al. (2012). Specifically, Landsat time series imagery from 2003 to 2012 is being used to monitor and assess changes in vegetation canopy cover (van Dongen et al. 2013). Canopy cover estimates were validated by field work undertaken in 2012 and 136 images have been included in the analysis to date (van Dongen et al. 2013).

Rainforest patch extent and condition

The condition of rainforest patches is being measured in two ways. Predominately through satellite imagery captured in 2012 from the Rapideye satellite system[2] (Fig. 2), which was used to map rainforest patches and determine size/extent and density/cover, which were validated by fieldwork carried out in 2012 (van Dongen et al. 2013). These metrics will be compared following new image captures every second year. The physical condition of rainforest patches is also measured as per vegetation transects (see above).

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[1] The Landsat series of satellites began capturing data in the 1973. Imagery is collected at 30 m pixel size across six spectral bands. Landsat data is a fundamental dataset used globally to monitor long-term vegetation changes (see Zhu et al. 2012).

[2] The Rapideye program (see http://www.rapideye.com/) was launched in 2009 and is made up of a constellation of five satellites. Imagery is collected across five spectral bands at 6.5 m pixel size, which is re-sampled to 5 m during rectification. These specifications result in the timely and cost effective capture of data at a regional scale.

Small mammal diversity and abundance

Small-medium sized mammal sampling quadrats (comprised of Elliot and cage traps on a 0.25 ha grid) have been established in association with vegetation plots (Fig. 2). These follow published methods (e.g. Woinarski et al. 2010) used widely across northern Australia, with the exception that camera traps are used instead of night-time spotlight searches. Larger species (> 150 g) are permanently tagged using micro-chip insertion under the skin (between shoulder blades) to allow individuals to be identified upon recapture. Smaller species are earmarked using permanent marker pens to allow recaptures to be identified during a single trapping session. Standard body measurements are taken and identification follows Menkhorst and Knight (2004). Recaptures are not included in any analysis. We have set 2 % trap success as a threshold for concern for mammal assemblages. This arbitrary value is based on changes in trap success in Kakadu, Litchfield and Nitmiluk National Parks in the Northern Territory which declined from > 5 % in 1996 to < 2 % in 2008 (Woinarski et al. 2001; 2010).

Data analysis

Spatial metrics used to assess fire patterns

Fire pattern targets are based on key assumptions about the ecological effects of fire; namely that:

- The landscape should not be dominated by late dry season fires, as these are more intense and cover larger areas in a single event;
- Early dry season fires are generally less intense, smaller, and leave more internal unburnt patches within fire scars which are more likely to escape late dry season fires and thus, act as refuges for wildlife;
- Unburnt patches should be distributed evenly across the landscape to make them more accessible to wildlife; this means that the distance from any fire scar to the nearest unburnt vegetation should be smaller, rather than larger; and
- The landscape should contain a mix of vegetation age classes, including older vegetation (> 3 years) and not be solely comprised of vegetation that is 1–2 years of age.

Mapping fire patterns

Analyses and maps presented in this report are based on fire scars derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery, which provide a resolution of 250 m (http://modis.gsfc.nasa.gov/). MODIS data for each year were obtained from the North Australia Fire Information (NAFI) website (http://www.firenorth.org.au/nafi2/) and were clipped by the LCI area defined in Figure 1. Fire scars were attributed by year and season (early dry season (EDS): January–June; late dry season (LDS): July–December). The following metrics were then calculated for each year: (1) proportion of the LCI burnt; (2) proportion of the LCI burnt in the EDS; (3) total area burnt for the EDS and LDS. Unburnt patches were then generated by erasing the clipped fire scar polygons from the LCI boundary. All spatial analyses were carried out in ArcMap 10.1 using tools in the Spatial Analyst extension and the burnt and unburnt summary statistics were calculated in MS Access. All maps were produced using ArcMap 9.2.

A raster analysis approach was used to determine the distance to an unburnt patch. For each 250 m burnt pixel the shortest land distance to the nearest unburnt pixel was calculated. Burnt and unburnt patches 20 ha or less (3 x 3 MODIS pixels) were then excluded from future analysis. To ensure the distance values on the edge of the LCI boundary were accurate the fire scars within a 20 km buffer area outside of the LCI mainland boundary were included. Two raster datasets were generated for the LCI area including the buffer: (1) coast raster with all land areas set to 1 and ocean areas set to null; (2) unburnt raster with all unburnt areas greater or equal to 20 ha set to 1. The 'cost distance', 'reclassify', 'zonal statistics as table' and 'zonal histogram' tools were then used to calculate the following metrics for each year: (1) average distance to an unburnt patch; (2) maximum distance to an unburnt patch. The above process was then repeated restricting the fire scars to vegetation ages of three years of age (data only available from 2002–12) and five years of age (data only available from 2004–12). The area of each age class and the number of patches (> 20 ha) that made up those classes was then calculated.

A fire history spatial dataset was developed by unioning the clipped fire scars for all years and the LCI boundary dataset together and attributing each polygon for each year with the following year age classes: < 1 (burnt in current year), 1, 2, 3, 4, 5 + (not burnt for 5 or more years). For the years 2002 and 2003 the full range of age classes could not be assessed as MODIS fire scar data is not available prior to 2000. The following metrics were then calculated for each year: (1) total area in each year age class; (2) proportion in each year age class.

Time since last burn (TSLB) and fire frequency (FF) were also attributed for the years 2011 and 2012 in the aforementioned fire history spatial dataset. Fire frequency was calculated for the periods 2003–07 (before increased fire management) and 2008–12 (following increased fire management) and attributed to the fire history spatial data set, and ranged from 0 (never burnt in that period) to 5 (burnt every year). The following metrics were then calculated for each time period: (1) total area in each FF class; (2) proportion in each FF class.

Statistical analyses

Analysis of Variance (ANOVA) was used to compare mammal trapping data between years, habitats and areas, and regression analysis was used to examine the effects of fire and cattle on mammal abundance and diversity. For each monitoring site we derived several metrics to explore the role that vegetation age and patchiness might play within a 1, 3 and 5 km radius: the diversity of vegetation age classes, calculated as the number of different vegetation age classes in each radius; the mean age of all vegetation within each radius, and the median age of all vegetation within each radius. Means were then used for each geographic cluster of sites in a regression analysis to examine the relationships between mammal abundance (trap success) and species richness, and the fire metrics described above. No analysis of vegetation structural change was undertaken as only two years of data were available (there is likely to be annual variation due to fire events) and because there is no pre-LCI vegetation data available.

Fire metrics from 2000–12 were graphed for visual examination, and means for all metrics were compared pre- (2000–07), during a transition period where fire management shifted from the use of fire breaks or buffers to the application of landscape mosaics (patch mosaic burning (PMB), 2008–10) and following LCI management (where the PMB continued, 2010–12) using ANOVA. This trajectory of management changes applies only to DPaW managed areas (not the EcoFire managed areas). The Tukey method was used for all follow-up comparisons and in all cases, the assumptions of equality of variances and normality were examined, and when data failed to meet these, they were transformed accordingly. Statistical significance was accepted at 0.05 level and means are presented with standard errors except where stated otherwise.

Managing inappropriate fire regimes

Approach and implementation

The DPaW fire program uses prescribed burning late in the wet season and very early in the dry season (as does EcoFire, see Fig. 1), when fires are less intense, and more likely to leave internal unburnt patches of vegetation. Planning and delivery of the fire program involves DPaW staff, traditional landowners, Indigenous ranger groups and property owners/managers. Annual burn plans are developed based on fuel loads and group objectives. Given the scale and remoteness of the North-Kimberley LCI area, most of the prescribed burning is carried out using aerial incendiaries dropped from a helicopter or fixed-wing aircraft. The fire crew typically flies 25,000 km each season across the North-Kimberley, dropping around 50,000 incendiaries, following routes outlined in burn plans to achieve a strategic mix of vegetation age classes and firebreaks. EcoFire covers a similar distance and drops the equivalent number of incendiaries (Legge et al. 2013). Hand-burning from the ground is also carried out to protect infrastructure, key biological assets and cultural sites.

Managing the impacts of feral animals

Cattle

There have been no formal large-scale surveys of feral herbivores in the North-Kimberley as there has been in other areas of northern Australia (e.g. Bayliss and Yeomans 1989a). With no estimates of population size, assessing management is problematic; a central tenet of wildlife management, in particular achieving reductions in population size, is that the rate of decrease must be more than the rate of increase (Sinclair et al. 2006). The remoteness and scale of the North-Kimberley means that aerial surveys are impractical. Current departmental Aerial Operating Procedures (AOP) are also restrictive of aerial-based activities, particularly high-risk low-level flying required for aerial surveys. To overcome this constraint, carrying capacity figures developed by the Department of Agriculture, which were based on pastoral potential derived from land system mapping by the CSIRO (Speck et al. 1960), were used to estimate feral cattle numbers in the LCI. These were then validated with aerial surveys in selected areas. Physical signs of cattle damage at mammal and vegetation monitoring sites (see above) and additional sites (see below), and reductions in these are being used to demonstrate a reduction in the impacts of cattle on the landscape.

Estimating abundance

Pastoral potential classification maps were obtained from the Department of Agriculture, which are based on CSIRO land system mapping work (Speck et al. 1960). These land systems were assigned to one of five categories based on pastoral potential: (1) unsuitable (carrying capacity: 0.5–1 animal km^{-2}); (2) very low (1–2.5 animals km^{-2}); (3) low (2.5–4 animals km^{-2}); (4) moderate (4–8 animals km^{-2}) and (5) high (8 + animals km^{-2}). Land systems were mapped in

ArcView 9.2 and clipped by the LCI boundary (Fig. 5). The total area (km^2) of each land system was determined and cattle estimates were derived accordingly from these. We used the minimum and maximum values, as well as the middle (mean) value, to derive estimates (Table 2). For instance, to estimate abundance on land classed as having very low pastoral potential (Fig. 5), we multiplied the area (in this case 348 km^2) by the minimum carrying capacity (1; 348 x 1 = 348) and the maximum carrying capacity (2.5; 2.5 x 348 = 870); the mean was the average of the two values (in this case 1 and 2.5 = 1.75; 1.75 x 348 = 609; Table 2).

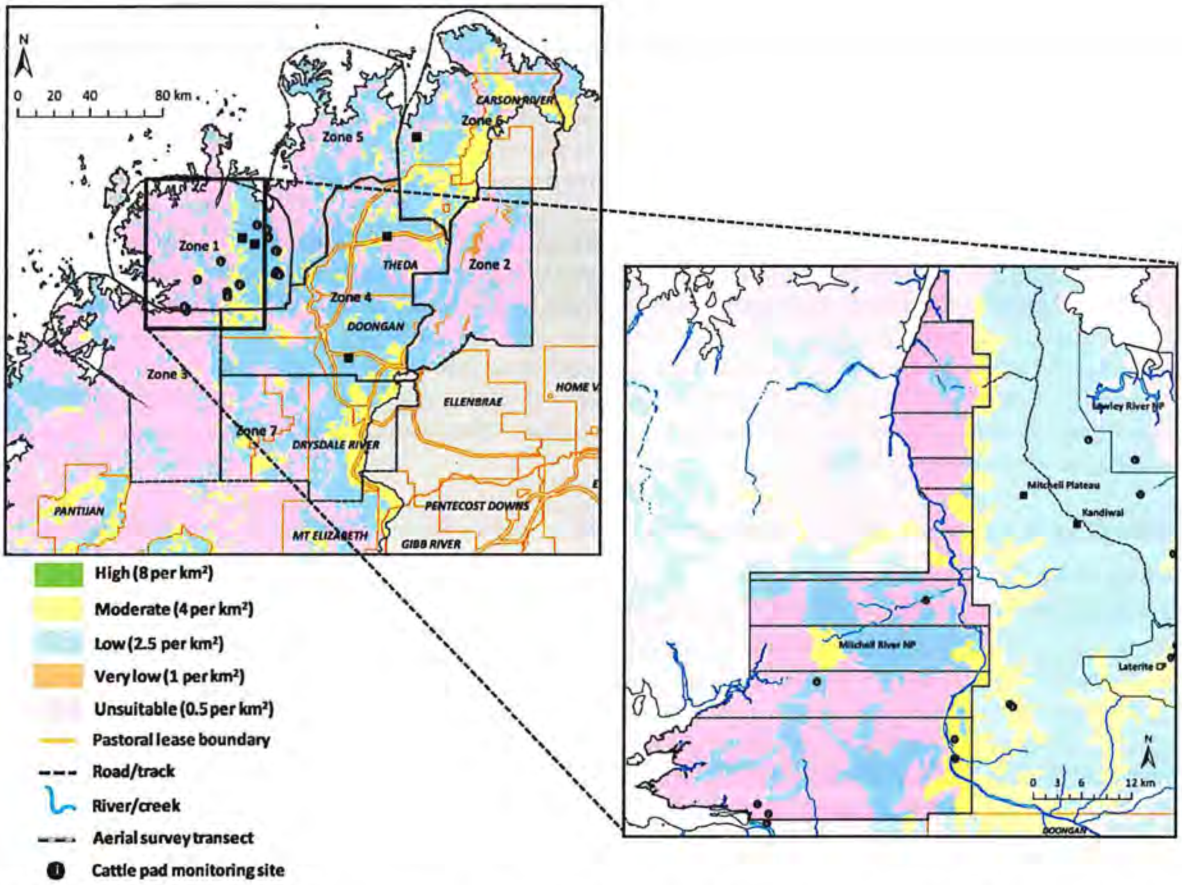


Fig. 5. Cattle management zones and pastoral potential based on Department of Agriculture and CSIRO land system mapping and Locations of cattle pad monitoring sites and aerial survey transects in the north-west Kimberley.

A population estimate of feral cattle was obtained from Mitchell River National Park via an aerial survey. This was carried out in the early dry season of 2008, using a Bell 206 Jetranger helicopter. Survey variables were standardised at a height of 300 feet, a speed of 60 knots and a 250 m wide counting transect for each observer. Transects were delineated by lightweight rods attached to the machine and carried out during mid-morning (0800–1100). The park was surveyed by systematic east–west transects spaced 5 km apart, giving a mean sampling rate of 10.1 %. Double counting techniques (see Caughley and Grice 1982) or habitat-specific correction factors (see Bayliss and Yeomans 1989b) were not used to account for the negative biases typical of aerial animal counts. Population size and associated errors were estimated using the ratio method (Table 3; see Caughley and Grigg 1981).

The population size of cattle in Mitchell River National Park only was estimated based on land systems and associated pastoral potential figures (Table 3). Estimates from both methods (aerial survey and pastoral potential) were verified by regional DPaW staff with more than 10 years experience shooting cattle in the North-Kimberley, previous aerial culling records, staff from DAFWA (M. Fletcher, pers. comm.) and previous station managers (B. Maher, pers. comm.) in the North-Kimberley. In lieu of large-scale formal aerial surveys, the mean carrying capacity was deemed the most appropriate estimator of feral cattle numbers in the LCI (Table 3). The North-Kimberley was then divided into seven management zones based on tenure (Fig. 5) and estimates derived for each of these (Table 2). Figures from aerial culling operations were compared to population estimates.

Table 2. Population estimates for North-Kimberley cattle management zones based on the classification of pastoral potential from CSIRO land system mapping. Note CC, carrying capacity and percentages in parentheses are the contribution of that land area to the total area of the LCI.

Zone	Area of pastoral potential (km^2)						Population estimate			Population density (km^{-2})		
	Unsuitable (CC: 0.5–1 km^{-2})	Very low (CC: 1–2.5 km^{-2})	Low (CC: 2.5–4 km^{-2})	Moderate (CC: 4–8 km^{-2})	High (CC: 8–12 km^{-2})	Total	Mean	Min.	Max.	Mean	Min.	Max.
1	2,947	0	1,774	475	0	5,198	10,828	7,810	13,847	2.1	1.5	2.7
2	2,448	149	1,835	42	0	4,476	8,314	6,130	10,499	2.4	1.7	3.1
3	4,857	0	862	107	0	5,830	7,090	5,014	9,165	1.9	1.3	2.4
4	1,852	0	4,933	963	0	7,753	23,201	17,112	29,290	1.4	1.0	1.8
5	1,622	0	1,961	155	0	3,743	8,520	6,334	10,707	2.9	2.1	3.7
6	1,419	90	3,388	1,451	0	6,355	20,942	15,076	26,808	1.7	1.2	2.2
7	1,914	108	2,634	1,010	0	5,674	16,245	11,690	20,800	1.9	1.4	2.4
Total	**17,060 (44 %)**	**348 (1 %)**	**17,388 (45 %)**	**4,204 (11 %)**	**0 (0 %)**	**39,000 (100 %)**	**95,140**	**69,165**	**121,115**	**2.0**	**1.5**	**2.6**

Table 3. Comparison of population estimates derived from CSIRO and Department of Agriculture land system carrying capacity mapping, and an aerial survey at Mitchell River National Park.

Pastoral potential	Area (km^2)	Estimate based on land system carrying capacity			Estimate based on aerial survey			
		Min.	Max.	Mean	Mean	SE	Lower	Upper
Unsuitable	752	376	752	564				
Very low	0	–	–	–				
Low	314	786	1257	1021				
Moderate	109	435	870	652				
High	0	–	–	–				
Total	**1,175**	**1,597**	**2,879**	**2,238**	**1,621**	**473**	**1,148**	**2,094**

Aerial culling operations at Mitchell Plateau (left) and a large scrub bull near Mitchell River campground (right). Photos: DPaW

Measuring reductions in cattle impacts

A simple method for measuring the effectiveness of feral herbivore control is to quantify levels of disturbance over time. Large feral herbivores leave conspicuous signs (pads, tracks, wallows and damage around freshwater; Hone 1988; Bowman and McDonough 1991; Taylor and Friend 1984; Fordham et al. 2006), and reductions in these over time can be used to demonstrate effective management. These methods have been used successfully elsewhere in northern Australia to monitor impacts of feral herbivores (Bowman and McDonough 1991; Bowman and Panton 1991; Fordham et al. 2006; 2007; Mitchell et al. 2007; Ens et al. 2010).

Cattle pad monitoring sites in the LCI area were chosen based on previous cattle control operations and the local knowledge of long-term staff (Fig. 5). Photos of each site were taken from a helicopter at 300 ft at the end of the wet season. Sites were assessed (and scored) as having: no (0), low (1), moderate (2) or high (3) impacts, by a panel of regional nature conservation staff. These will provide a baseline for comparison with future surveys and compliment the measurements of cattle impacts measured at mammal and vegetation monitoring plots (see above).

Donkeys

The North-Kimberley is largely donkey free, following intensive control programs from 2002 to 2008 where more than 8,000 animals were shot. A 'Judas' animal program is carried out by DAFWA each year on conservation estate and surrounding unallocated Crown land (uCl), and this is funded through the LCI. Staff from DAFWA provide operational data, including location of collared animals and the number of any animals shot.

Cane toads

Cane toads pose a serious threat to the high biodiversity values of Kimberley Islands, the majority of which are Exclusive Possession Native Title (Wunambal Gaambera Aboriginal Corporation 2010; Balanggarra Aboriginal Corporation 2011; Dambimangari Aboriginal Corporation 2012). The State Cane Toad Program, in partnership with Native Title holders, the Science and Conservation Division of DPaW and regional Nature Conservation staff are responsible for identifying priority North-Kimberley Islands and maintaining these as toad free. A cane toad quarantine strategy for Kimberley islands is currently being prepared by Native Title holders and DPaW.

Managing the impacts of invasive plants

High priority weed species were identified by Shedley et al. 2012 and control zones established in areas with high public visitation and thus the greatest potential to act as source populations for new infestations. On-ground works are managed and carried out by DPaW staff or on a fee-for-service arrangement with partners.

RESULTS

Landscape outcomes

Progress towards delivering landscape outcomes is summarised against monitoring and reporting targets (see Shedley et al. 2012) in Table 4, with supporting information and context provided in the ensuing text.

Palm forest at Mitchell Plateau. Photo: David Bettini

Table 4. Landscape Conservation Initiative outcome performance reporting: landscape health indicators across the North-Kimberley.

Condition monitoring	Monitoring targets	Method	Outcome
Rainforest patch extent retained in extent and condition			
OCM 1 Extent to which monitored rainforest areas maintain their size/extent	*OT1* Stable or increasing trend in rainforest patch/size	Rapideye high resolution satellite imagery used to delineate areas with greater than 65% canopy cover. Rainforest patches were then manually identified. The boundaries and cover classes of identified patches can then be assessed for changes over time with repeat Rapideye captures.	Regional knowledge was used to 'clean' the extent masks of non-rainforest patches. A sub-set of 94 rainforest patches have been selected and statistics such as area, perimeter and canopy cover have been generated from the extent mask (Table 7; Fig. 10); this will form the basis of future rainforest patch monitoring.
OCM1a Extent to which monitored rainforest areas maintain their vegetation condition (structure)	*OT1a* Maintenance of vegetation density/cover with no evidence of clearing by grazing or burning, or invasion of annual plant species	Quadrats (50 m x 50 m) at monitoring patch boundaries to measure changes in rainforest patch vegetation structure. Rapideye and Landsat imagery will be used to assess cover changes within rainforest patches.	Vegetation density/cover was maintained (Table 5). Tree canopy cover showed a slight increase, while shrub layers showed a slight decrease in cover from 2011 to 2012. No sites showed evidence of major grazing or burning damage or invasion by weeds (Table 14); some showed evidence of cattle (Fig. 21). Landsat temporal sequence will look at canopy cover and structure (Fig. 7), and provide information relating to patch condition and annual climatic conditions (Figs. 8, 9). Rapideye imagery (Fig. 10) will be used to assess changes in monitored rainforest patches (Table 7, Figs. 7, 8, 9)
Small-medium mammal species (< 5 kg) diversity and abundance retained			
OCM2 Changes in numbers of species detected	*OT2* Stable or increasing trend in number of small mammal species detected	Kimberley divided into sites by geology, landform and vegetation. Compare species numbers over time on average per site, with trend compared to 2004 data point.	There were no significant changes in mean abundance or species richness (Fig. 11). Slight increases in abundance and richness occurred from baseline surveys to 2012 in rocky and woodland habitats (Table 8). Rainforest mammal abundance and richness remained stable (Table 8). Site occupancy showed no consistent trends (Table 9). Species not recorded during 2012 include *Mesembriomys gouldii*, *Petrogale concinna* and *Petropseudes dahlia* (Table 9).
OCM2a Specific species presence/absence rates	*OT2a* Trends in average abundance of monitored species stable or increasing	Determine trends in approximate species capture/detection rates at above monitoring sites	Rocky habitats have shown slight declines in abundance, but are still well above the threshold of concern (Table 8). Arboreal rodents have re-established in rocky habitats and bandicoots also increased (Table 8). Woodland habitats showed increases in abundance of most groups compared to baseline surveys (Table 8). Rainforest sites also showed increases (Table 8). Overall, mean abundance and species richness increased at more than 70 % of monitoring clusters from 2011 to 2012 (Fig. 12).

Native vegetation condition retained, enhanced or re-established			
OCM3 Vegetation/habitat condition scores in six major vegetation types	*OT3* Stable or increasing trend in retention of natural habitat condition	Use 50 m x 50 m quadrats in replicated sites in major vegetation types. Measure and compare changes in canopy cover of each vegetation strata for comparison with initial data points (2003–04)	Vegetation was assessed as being stable. Shrub and grass layers showed declines with increased fire frequency among 2012 monitoring sites, tree canopy cover increased or remained stable (Table 5). The ephemeral nature of fires in savanna landscapes suggests that year to year variation in vegetation strata should not be taken as an indication of decline. Analysis of long term trends (> 5 years) in site data would be needed to provide rigorous evidence of vegetation decline.
OCM3a Litter layers enhanced (through ecological burning and cattle control) to increase habitat resilience	*OT3a* Stable or increasing trend in leaf litter accumulation	Assess changes in litter depth and soil erosion in quadrats (annually). Include at least six reference sites on selected NW Kimberley Islands (every five years)	There was a stable to declining trend at monitoring sites in litter depth (Table 6). Although likely related to ephemeral changes in fire regime, it will be important to follow this up in future years. No assessment of island litter attributes has been undertaken.
OCM3b Sub-region-wide vegetation canopy cover maintained or increased over the long term	*OT3b* A reduction in the proportion of annual grasses to perennial grasses and an increase in the proportion of shrubs and younger age tree species	Use 50 m x 50 m quadrats in replicated sites in major vegetation types. Measure and compare changes in canopy cover of each vegetation strata for comparison with initial data points (2003/04). Landsat temporal sequence used to assess long term changes in overstorey canopy cover since 2003 in 50 m x 50 m sites around monitoring quadrats (see above) and in at least 50 other sites per vegetation unit.	Annual *Sorghum* spp. was recorded as a dominant herbaceous species in only 22 % of 2011 sites and only 12 % of 2012 sites. Maximum cover was 3.3 % in 2012 compared to 60 % in 2011. At several sites this is due to 2012 fires which removed *Sorghum* biomass. It is too early to assess trends in annual grass cover with only two years of data. The Landsat time series plots give an indication of vegetation trends (particularly canopy cover) over time. Analysis of 2003–2012 data suggests a stable trend (Figs. 7, 8, 9), but this should be interpreted with caution owing to the short window of observation. Analysis of the full time series data set back to 1987 is needed to gauge current trends relative to historical fluctuations.
Rainfall trends			
Rf Monitoring Rainfall quantity and timing by year compared to historic average and trends	*Rf Reporting* Monthly rainfall data is obtained for representative sites from BoM weather stations in the Kimberley and used in analysis of trends.	Rainfall performance variance from averages (as available).	Historical annual and monthly rainfall records have been collated for Bureau of Meteorology (BoM) stations throughout the North-Kimberley (Table 1; Fig. 4). Stations closest to LCI monitoring stations were chosen. An automated BoM station has been established at Mitchell Plateau, and will be on line in 2014.

Native vegetation condition

Some vegetation structural indices varied, and some were stable, between 2011 and 2012 (Tables 5, 6, Fig. 6). Tree basal area changed little between 2011 and 2012 across all major habitat types (Fig. 6). Similarly tree cover changed little between 2011 and 2012 among habitat types (Table 5). The largest differences were recorded for the more dynamic and fire affected components of the vegetation: the grass and shrub layers (Table 5). Laterite forest sites showed a > 50 % reduction in shrub and grass project ground cover (Table 5). Grass cover was also substantially (> 50 %) reduced at volcanic and sandstone woodland sites (Table 5). Reduced ground layer vegetation at Laterite forest sites can be attributed to extensive early dry season fires in April/May 2012, with 77 % of sites burnt prior to measurement compared to only 43 % of sites in 2011. Low grass cover at woodland sites in 2012 (volcanic and sandstone) may be more correctly attributed to lower rainfall in the 2011-12 wet season compared to that of 2010-11 (> 2000 mm, see Fig. 4). The same percentage of volcanic and sandstone woodland sites were burnt in both 2011 and 2012 (38 % and 22 % respectively). There was a decline in both ground cover and tree cover at King Leopold sites from 2011 to 2012 (Table 5). This is probably attributable to much higher incidence of fire among King Leopold sites in 2012 (see below). Cover values for vegetation strata in the Prince Regent region were similar to equivalent strata in the Mitchell region (Table 5).

Table 5. Mean vegetation strata cover in 2011 and 2012 among vegetation habitat types at LCI monitoring sites. Note: values are percentages.

	Grass (0.3–1.0 m)		Shrub 2 (0.5–1.5 m)		Shrub 1 (1–8 m)		Tree (4–20 m)	
	2011	2012	2011	2012	2011	2012	2011	2012
Rainforest	0	0	16	12	43	30	63	72
Laterite forest	29	14	7	3	14	6	36	32
Volcanic woodland	72	34	2	8	7	6	23	18
Volcanic riparian	64	70	2	5	8	11	14	16
Sandstone woodland	24	8	2	3	7	5	17	24
Sandstone rocky	15	17	1	4	7	7	9	9
King Leopold	52	37	7	5	9	8	26	21
Prince Regent VW	-	44	-	4	-	6	-	17
Prince Regent SS	-	24	-	3	-	11	-	28

Table 6. Mean litter component depth and percentage cover among habitat types at LCI monitoring sites from 2011 to 2012 in the Mitchell region only.

	Litter depth (mm)		Fragmented litter depth (mm)		Litter cover (%)	
	2011	2012	2011	2012	2011	2012
Rainforest	26.7	25.8	6.7	1.8	60.4	73.1
Laterite forest	21.4	5.8	1.0	2.0	38.4	31.0
Volcanic woodland	10.2	2.5	0.6	0.3	8.8	16.4
Volcanic riparian	10.0	3.1	2.1	0.7	17.0	8.9
Sandstone woodland	8.0	2.4	0.8	0.7	18.7	29.8
Sandstone rocky	6.0	4.4	2.1	2.9	13.4	10.0

Litter depth decreased from 2011 to 2012 in all vegetation types except for rainforest (Table 6). Depth of fragmented litter layers and litter cover showed no consistent patterns between years (Table 6). Changes in litter depth could be attributed to higher fire frequency among monitoring sites in 2012 in Laterite habitats (see below) or due to reduced wet season related vegetation production (see Fig. 4) in volcanic and sandstone habitats. In broad terms, litter cover was highest in rainforests (> 60 %), followed by laterite forest (> 30%) and sandstone woodlands (> 18 %), and lowest in vegetation on volcanic clay soils (< 18 %) and in rocky sandstone habitats (< 13 %; Table 6). This is consistent with a gradient of woody plant canopy cover from high cover (rainforests) to low cover (volcanic woodland and rocky sandstone) (Fig. 6). Cracking clay self mulching soils may preclude major leaf litter build-up on soil surfaces due to their self mulching properties.

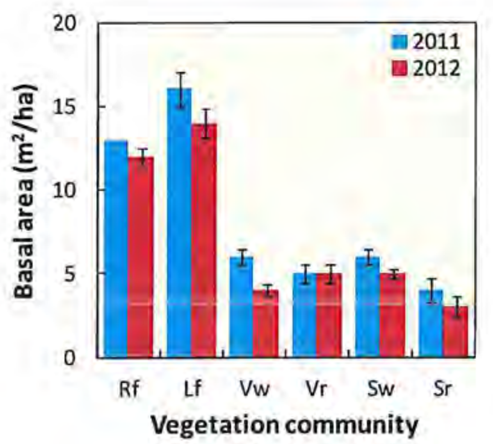


Fig. 6 Mean (± standard error) basal area during 2011 and 2012 surveys among vegetation habitat types at the Mitchell Plateau and Prince Regent regions. Note: Rf, rainforest; Lf, laterite forest; Vw, volcanic woodland; Vr, volcanic riparian; Sw, Sandstone woodland; Sr, Sandstone rocky.

No benchmark data is available at the plot level for either vegetation or litter attributes as there is for mammal abundance data. Future trends in vegetation cover and basal area and litter variable will be assessed using 2011–2012 as benchmark values. No statistical analyses have been undertaken for vegetation or litter structural values in this report as only two years of LCI data are available (see Approach), and no previous comparative data is available for the region. No comparison has been attempted in this report with vegetation and litter attributes reported elsewhere in northern Australia, but this will feature in future analyses.

The Landsat time series plots show the seasonal fluctuation of vegetation cover. The time series plot of LCI_015 (a rainforest patch) shows a small and consistent seasonal fluctuation in cover (Fig. 7a). Cover values peak in May at close to 80 % and drop to no lower than 65 % in late November (Fig. 7a). Canopy cover at this site has declined slightly over time (Fig. 7d), but was not significant (R^2 = 0.003; $F_{1,112}$ = 0.361; P = 0.549), and could suggest that this patch is slowly deteriorating. The time series plot of RFP_017 (another rainforest patch) fluctuates more than LCI_015 (Fig. 7b), but has remained stable over time (R^2 = 0.001; $F_{1,119}$ = 0.165; P = 0.685; Fig. 7e). These regular fluctuations, which are characteristic of most monitored rainforest patches, stand in stark contrast to the large annual fluctuations observed in other vegetation types. For instance, site LCI_003, which is a riparian site, showed cover in the months around March consistently above 50 % (Fig. 7c). This estimate includes the contribution of grasses which flush annually. Low points in the time series (July–September) are likely to be representative of remaining canopy cover. This pattern is consistent from 2003 to 2007, with low points on 17–November 2004 and 26–November 2011 (Fig. 7c) that corresponds with late dry season fire events. Canopy cover has remained stable over time at this site (R^2 = 0.004; $F_{1,124}$ = 0.492; P = 0.485; Fig. 7f).

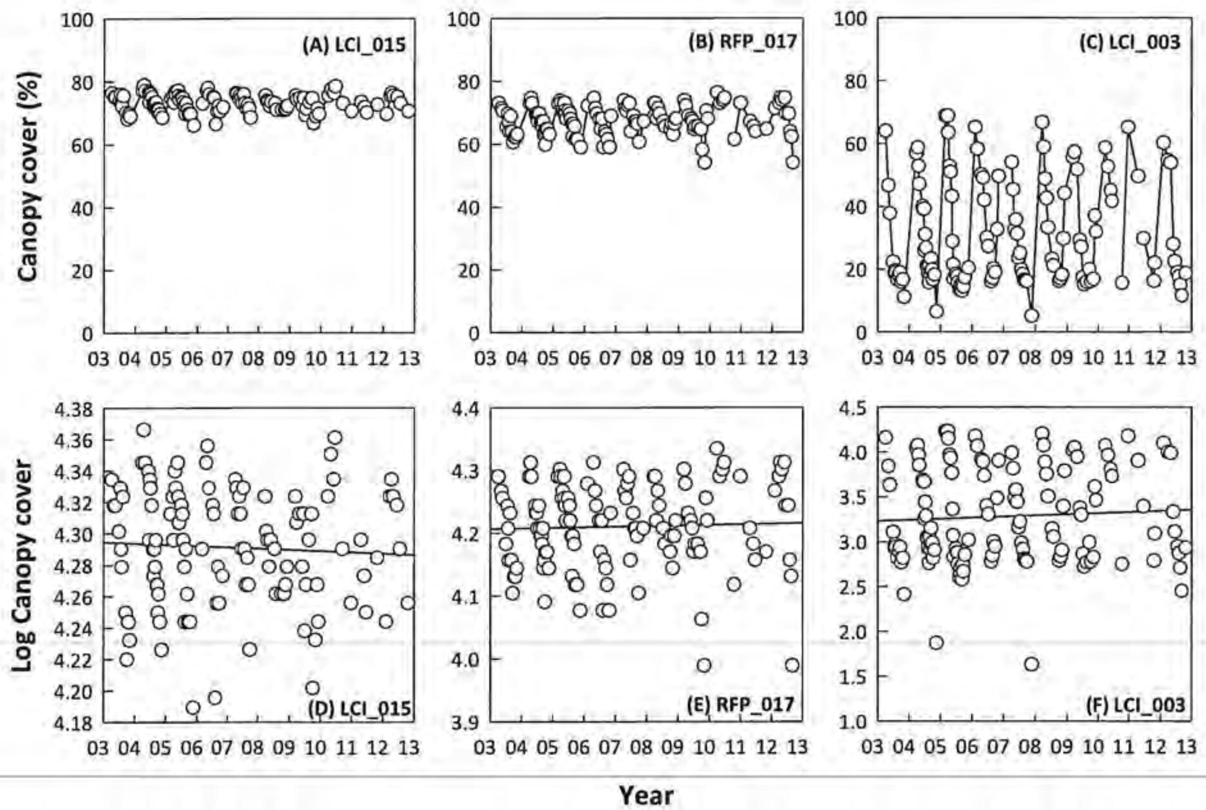


Fig. 7. Landsat time series plots for LCI monitoring sites: LCI_015 (rainforest site, left), RFP_017 (another rainforest site, middle) and LCI_003 (a riparian woodland site).

The values from each site can be pooled by habitat type (see Fig. 8) and trends examined over time (Fig. 9). For instance, canopy cover across rainforest (R^2 = 0.027; $F_{1, 125}$ = 3.438; P = 0.066; Fig. 9a), sandstone woodland (R^2 = 0.027; $F_{1, 127}$ = 3.477; P = 0.065; Fig. 9b), Volcanic riparian (R^2 = 0.027; $F_{1, 132}$ = 3.664; P = 0.058; Fig. 9e) and sandstone scree (R^2 = 0.023; $F_{1, 126}$ = 2.962; P = 0.088; Fig. 9f) sites have, on average, remained stable over time. In contrast, canopy cover has increased across volcanic woodland (R^2 = 0.102; $F_{1, 132}$ = 15.044; P < 0.001; Fig. 9e) and Laterite woodland (R^2 = 0.031; $F_{1, 131}$ = 4.170; P = 0.043; Fig. 9d) sites from 2003 to 2012.

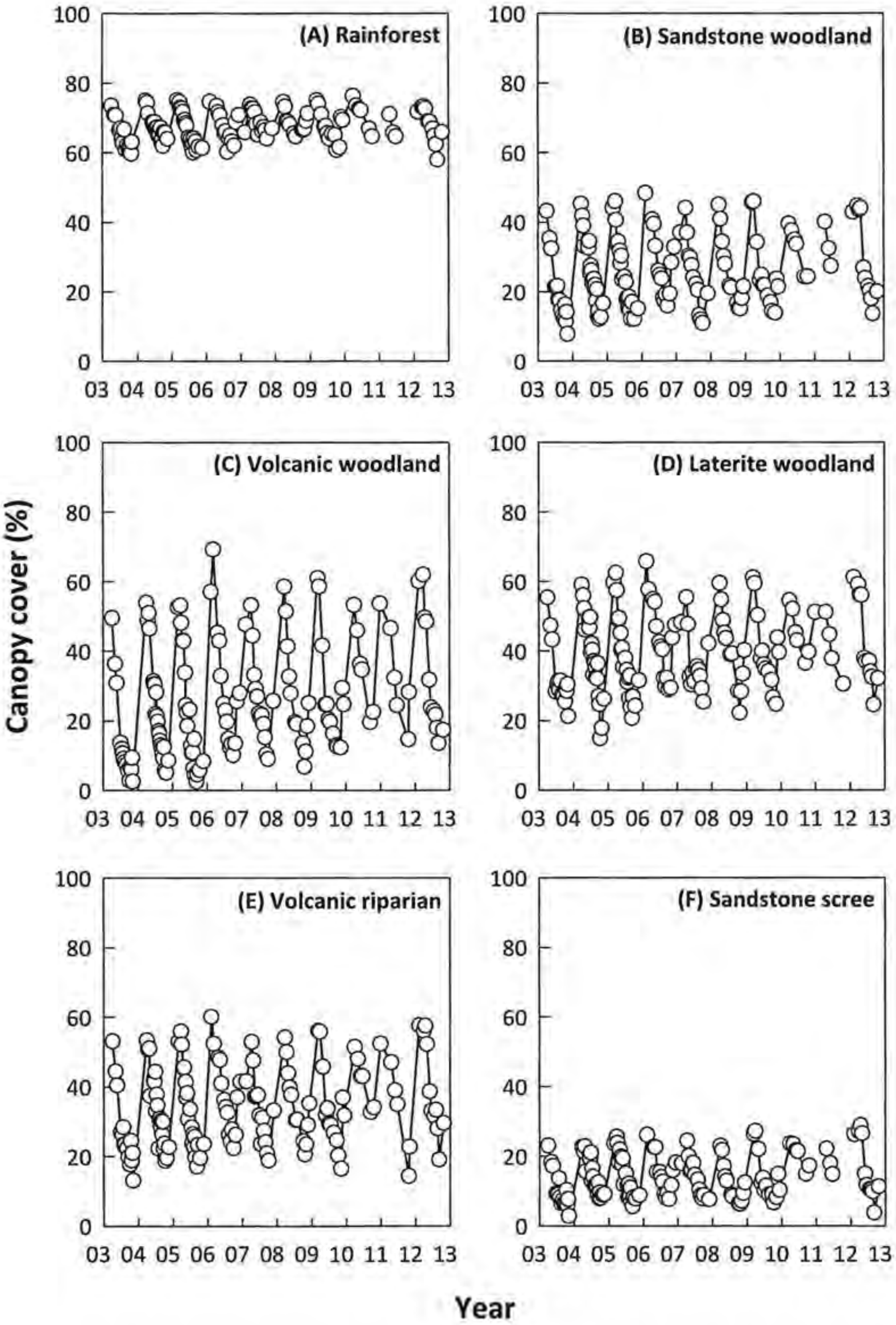


Fig. 8. Landsat time series plots of the major habitat types (see Fig. 3). Note: values are means for each habitat type.

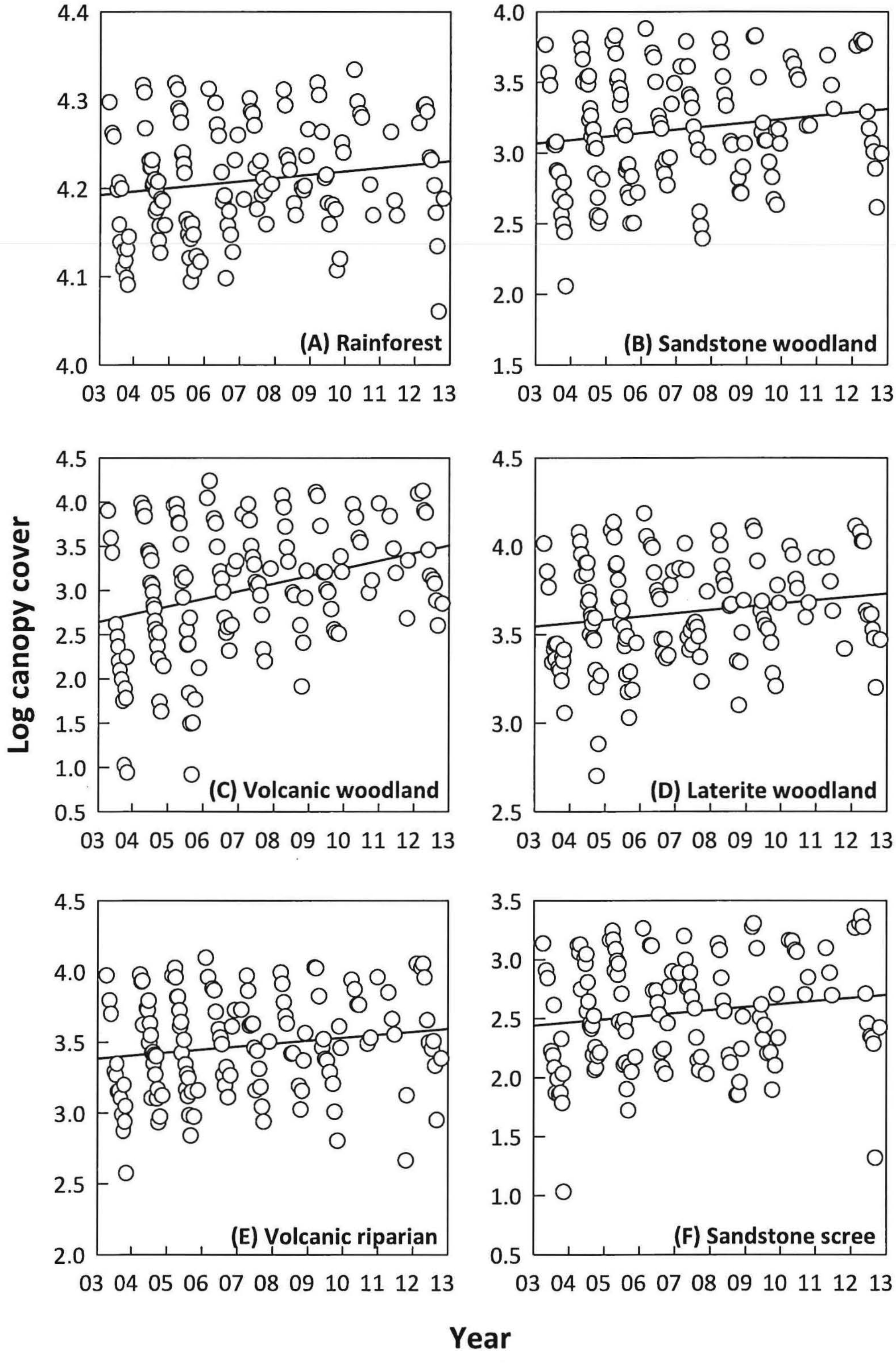


Fig. 9. Regression analysis of habitat type Landsat time series plots. Note values are means for each habitat type and have been log transformed to meet assumptions of normality and equal variances.

Rainforest patch extent and condition

The rainforest patch extent values created from the 2012 Rapideye imagery have provided baseline measures for future monitoring of rainforest patches (**Table 7**). These will provide a boundary and area of the patch and canopy cover metrics (e.g. **Fig. 10**) that will be compared with future Rapideye image captures to allow trends to be ascertained over time.

Table 7. North-Kimberley rainforest patch summary statistics.

	Canopy cover class area (ha)							Perimeter (km)	Area (ha)
	< 40 %	40–50 %	50–60 %	60–70 %	70–80 %	80–90 %	90–100 %		
Mean	0.01	0.02	0.07	1.2	2.7	1.6	0.09	15.16 (range = 0.02–124.2)	5.64 (range = 0.003–52.6)
Total	**0.7**	**1.7**	**6.8**	**109.1**	**252**	**144.9**	**8.1**	**1410**	**523.3**

Rainforest vegetation structure did not change between 2011 and 2012 in vegetation monitoring plots (Tables 5, 6, Fig. 6). Vegetation condition was typical closed vegetation (> 50 % total projected canopy cover) expected for rainforest and vine thickets in northern Australia (McKenzie et al. 1991). There had been no incursion by fire at monitored rainforest sites at the Mitchell River or Prince Regent National Parks in 2012. Cattle pads and dung were noted at most rainforest sites; however, this was associated with very limited damage to the rainforest canopy. Only at one site (LCI_047), was cattle damage sufficient to open up the ground layer across the site, however, this did not result in increased light penetration to ground level as tree cover was high. There was no evidence of invasive plant incursions into rainforest sites (see below), although Mount Trafalgar rainforest sites in the Prince Regent (LCI_313, LCI_314 and LCI_317) had *Passiflora foetida* patches recorded near boundaries. Mean rainforest basal area (BA) was 12–13 $m^2.ha^{-1}$ with minimum BA > 6 $m^2.ha^{-1}$ (Fig. 6). All rainforest patches, despite very rocky substrates, had a well-developed litter layer with strong decomposition of litter into decayed fragments, humus and organic soil layers (Table 6).

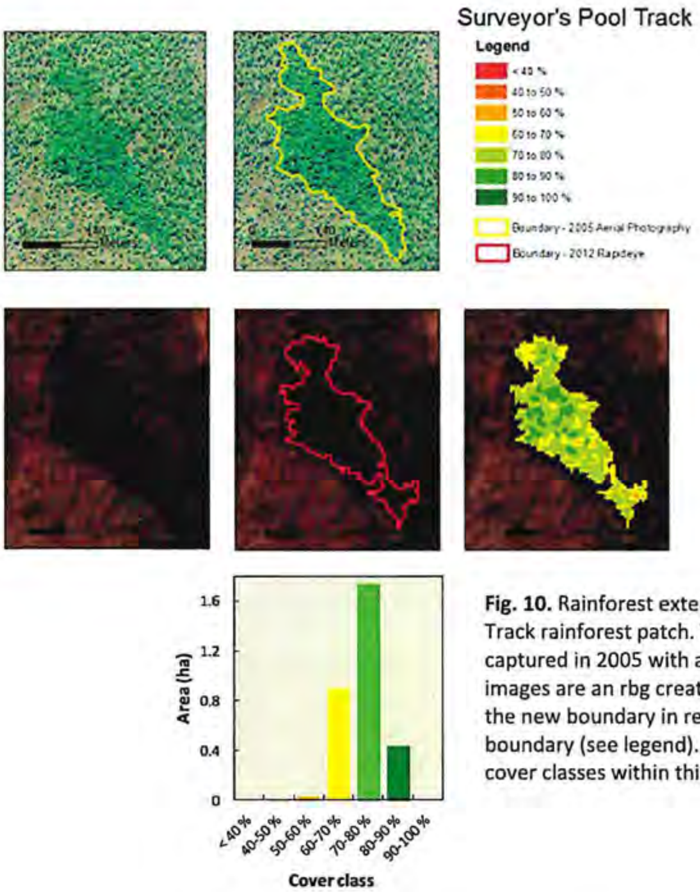


Fig. 10. Rainforest extent and cover density classes for Surveyor's Pool Track rainforest patch. The top two images are aerial photographs captured in 2005 with a digitised boundary in yellow. The middle three images are an rbg created from Rapideye data (left), superimposed with the new boundary in red (middle), and showing cover classes within the boundary (see legend). The graph on the bottom is the distribution of cover classes within this rainforest patch.

Small mammal diversity and abundance

Mean trap success, averaged across monitoring clusters, for small-medium sized mammals did not change over time (2-way ANOVA, year and monitoring cluster as factors: year: $F_{2,14} = 0.006$; $P = 0.994$; cluster: $F_{9,14} = 0.612$; $P = 0.768$; Fig. 11a). Similarly, mean species richness did not change over time (2-way ANOVA, year and monitoring cluster as factors: year: $F_{2,14} = 1.434$; $P = 0.271$; cluster: $F_{9,14} = 1.279$; $P = 0.328$; Fig. 11b). No site clusters had below threshold values for trap success or species richness during 2012 (Fig. 12a, b). However, below threshold trap success and richness was recorded across King Edward and the Laterite survey sites in 2011 (Fig. 12a, b). Five of the nine sites in the King Edward River and Laterite clusters still had below threshold values in 2012.

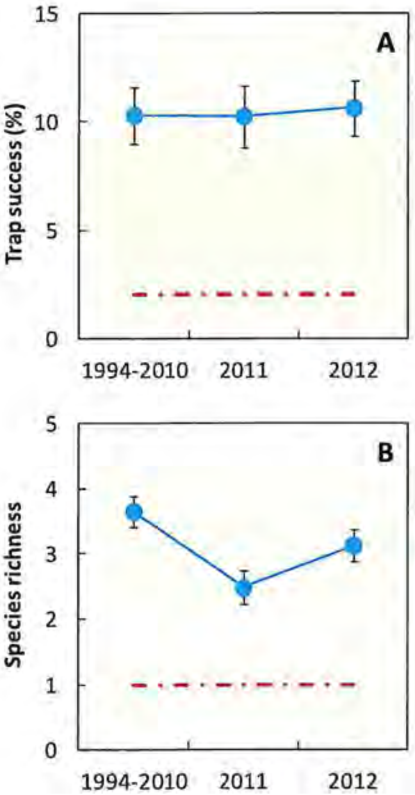


Fig. 11. Mean traps success (A) and species richness (B) for repeatedly surveyed small-medium sized mammal monitoring sites from 1994 to 2012 in the LCI project area. Note: errors are ± 1 standard error and dashed lines are critical thresholds from mammal collapse in Kakadu National Park, Northern Territory (see Approach).

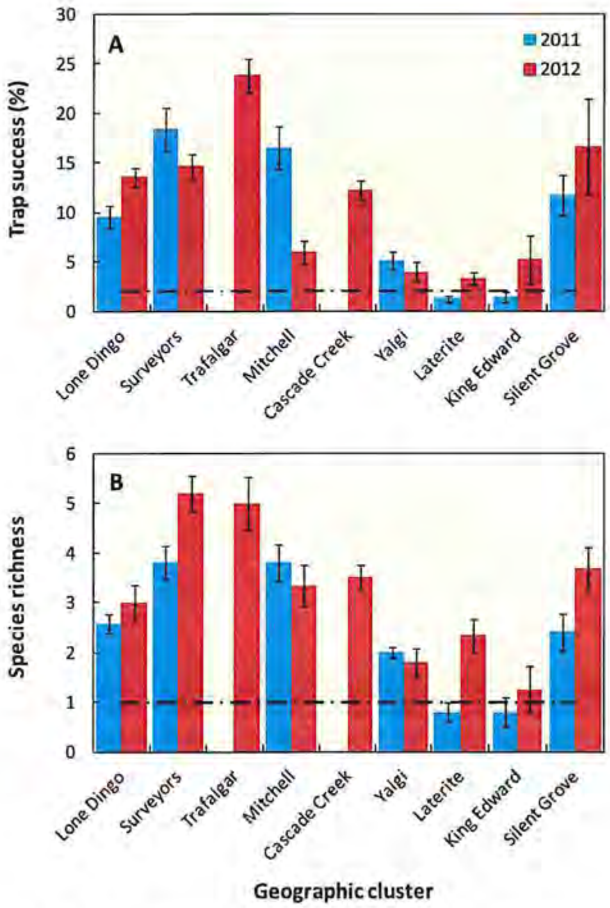


Fig. 12. Mean trap success (A) and species richness (B) of small-medium sized mammals at monitoring clusters in the LCI project area. Note: errors are ± 1 standard error and dashed lines represent critical thresholds.

Trap success for some small-medium sized mammal groups across habitat types appears to have increased over time, whereas others have decreased, fluctuated or remained stable (Table 8). Similarly, the occurrence of individual small-medium sized mammal species has increased at some repeatedly surveyed monitoring sites, whereas others have decreased, fluctuated or remained stable (Table 9). It is too early in the monitoring program to ascertain if these decreases are of concern owing to the ephemeral nature of small-medium sized mammal abundance.

Occurrence frequency of key threatened species at sites across the LCI area was stable or increasing, although there were a number of species of concern (Table 9). The brush-tailed rabbit rat and golden backed tree rat increased in site frequency by > 100 % from 2011 to 2012 and also from historic surveys (Table 9). Common species including the northern brown bandicoot, the grassland melomys, the Ningbing false antechinus, the western chestnut mouse and the red cheeked dunnart also increased their occurrence (Table 9). Northern quolls decreased in their occurrence at sites compared to 2011 and historical surveys (Table 9). However, the difference in relation to historical surveys is not of conservation concern because historically most surveys were conducted in preferred rocky quoll habitat, whereas LCI monitoring sites are more representative across habitat types.

Monitoring failed to record the Nabarlek in 2011–12 in the Mitchell or Price Regent regions. However historical survey data indicates this species has not been recorded in the Mitchell area, although it has been recorded in Prince Regent (Table 9). The closely related monjon is relatively common in rocky sites throughout the region. The rock ringtail is another species which has not been recorded during monitoring surveys (Table 9). The species is known to be present in the broader region (C. Myers, Dunkeld Pastoral Company, pers. comm.) and has not been historically recorded in the Mitchell region. The brush-tailed phascogale was last recorded in the Mitchell region in 2010, with a confirmed sighting of this species at the Mitchell campground (B. Stewart, pers. comm.). Targeted monitoring using nesting boxes (a more effective trapping approach, A. Burbidge, pers. comm.) will be used in 2014. The brushtail possum was recorded at the Mitchell in 2011 monitoring surveys, but not in 2012 (Table 9). This species appears to be elusive when using conventional Elliott or camera traps. Nest box traps will also be used for this species in 2014 and onwards.

Table 8. Mean trap success of small-medium sized mammal groups in rocky, woodland and rainforest habitats in the North-Kimberley during baseline surveys (1994–2010) and during 2011 and 2012 LCI monitoring. Note: *n* is the number of sites and *tn* is the mean number of trap nights per survey/monitoring site.

	1994–2010	2011	2012
Rocky	**(*n* = 109; *tn* = 200)**	**(*n* = 16; *tn* = 80)**	**(*n* = 12; *tn* = 114)**
Omnivorous rodents	6.33 ± 0.69	9.03 ± 1.77	5.92 ± 1.27
Arboreal rodents	0.64 ± 0.26	0	0.37 ± 0.22
Dasyurid predators	2.92 ± 0.31	2.43 ± 0.76	2.01 ± 0.46
Peramelids (bandicoots)	1.22 ± 0.18	1.69 ± 0.79	1.86 ± 0.60
Insectivorous dasyurids	0.03 ± 0.02	0.09 ± 0.09	0
Possums	0.05 ± 0.02	0.13 ± 0.09	0.07 ± 0.07)
Small Macropods	0.09 ± 0.03	0.09 ± 0.09	0.07 (± 0.07)
Woodland	**(*n* = 43; *tn* = 423)**	**(*n* = 30; *tn* = 98)**	**(*n* = 21; *tn* = 120)**
Omnivorous rodents	2.50 ± 0.59	3.87 ± 0.92	6.19 ± 2.07
Arboreal rodents	0.09 ± 0.05	0.93 ± 0.55	1.27 ± 0.60
Dasyurid predators	0.18 ± 0.09	0.46 ± 0.18	0.08 ± 0.08
Peramelids (bandicoots)	0.21 ± 0.08	1.18 ± 0.38	0.95 ± 0.30
Insectivorous dasyurids	0.01 ± 0.01	0.26 ± 0.12	0.64 ± 0.25
Possums	0	0	0
Small Macropods	0.01 ± 0.01	0	0
Rainforest	**(*n* = 7; *tn* = 127)**	**(*n* = 3; *tn* = 96)**	**(*n* = 8; *tn* = 117)**
Omnivorous rodents	6.72 ± 2.03	6.48 ± 3.24	10.23 ± 2.20
Arboreal rodents	0.16 ± 0.11	0.93 ± 0.93	0.56 ± 0.23
Dasyurid predator	4.44 ± 1.09	0.93 ± 0.46	0.63 ± 0.52
Peramelids (bandicoots)	1.22 ± 0.53	2.31 ± 1.22	1.35 ± 0.63
Insectivorous dasyurids	0	0	0
Possums	0	0	0.01 ± 0.01
Small Macropods	0.11 ± 0.11	0	0

Omnivorous rodents included *Leggadina lakedownensis*, *Melomys burtoni*, *Pseudomys delicatulus*, *P. johnsoni*, *P. nanus*, *Rattus tunneyi*, *Zyzomys argurus* and *Z. woowardi*. Arboreal rodents included threatened *Coniulurus penicillatus* and *Mesembriomys macrurus*. There was one species of dasyurid predator, *Dasyurus hallucatus*. Peramelids included *Isoodon auratus* and *I. macrourus*. Small insectivorous dasyurids included *Planigale maculata*, *Pseudantichinus ningbing* and *Sminthopsis virginiae*. Only one species of possum was recorded in 2012, *Wyulda squamicaudata*, while *Trichosaurus vulpecula* was recorded on a camera trap in 2011. Only the small macropod *Petrogale burbidgei* was represented in 2011 and 2012 monitoring surveys.

The golden-backed tree rat (listed as critically endangered in the Northern Territory and vulnerable under the EPBC Act) is relatively common across monitoring sites. Photo: David Bettini

Table 9. Percentage of surveyed sites in the North-Kimberley where small-medium sized mammals were recorded in baseline surveys (1994–2010) and in 2011 and 2012 LCI monitoring surveys.

Name	Common name	1994–2010	2011	2012
Conilurus penicillatus	brush-tailed rabbit rat	2.8	6.5	14.6
Dasyurus hallucatus	northern quoll	41.5	39.1	29.3
Hydromys chrysogaster	water rat	0.7	2.2	0
Isoodon auratus	golden bandicoot	15.5	8.7	14.6
Isoodon macrourus	northern brown bandicoot	25.4	23.9	46.3
Leggadina lakedownensis	tropical short-tailed mouse	2.8	0	2.4
Melomys burtoni	grassland melomys	6.3	8.7	19.5
Mesembriomys macrourus	golden-backed tree rat	11.3	4.3	19.5
Mesembriomys gouldii	black-footed tree rat	0	0	0
Petrogale burbidgei	monjon	2.8	2.2	2.4
Petrogale concinna	nabarlek	4.2	0	0
Petropseudes dahli	rock ring-tail possum	2.1	0	0
Phascogale tapoatafa	brush-tailed phascogale	2.1	2.1	0
Planigale maculata	common planigale	3.5	2.2	0
Pseudoantichinus ningbing	ning bing false antechinus	2.1	0	9.8
Pseudomys delicatulus	delicate mouse	23.9	4.3	2.4
Pseudomys nanus	western chestnut mouse	24.6	39.1	48.8
Pseudomys johnstoni	pebble mound mouse	0.7	2.2	7.3
Rattus tunneyi	pale field rat	27.5	23.9	24.4
Sminthopsis virginiae	red-cheeked dunnart	4.2	10.9	17.1
Trichosurus vulpecular arnhemensis	northern brushtail possum	0.4	2.9	0.0
Wyulda squamicaudata	scaly-tailed possum	3.5	4.3	4.9
Zyzomys argurus	common rock rat	43.7	34.8	34.1
Zyzomys woodwardi	Kimberley rock rat	14.8	8.7	22.0

Small-medium sized mammal monitoring at Lone Dingo in the North-Kimberley. From left to right: Cathy Goonack (and daughter), Kade Malay (Goonack) and Ian Radford. The brush-tailed rabbit rat (critically endangered in the NT and vulnerable under the EPBC Act, right) is relatively common across monitoring sites. Photos: Richard Tunnicliffe

Management action effectiveness

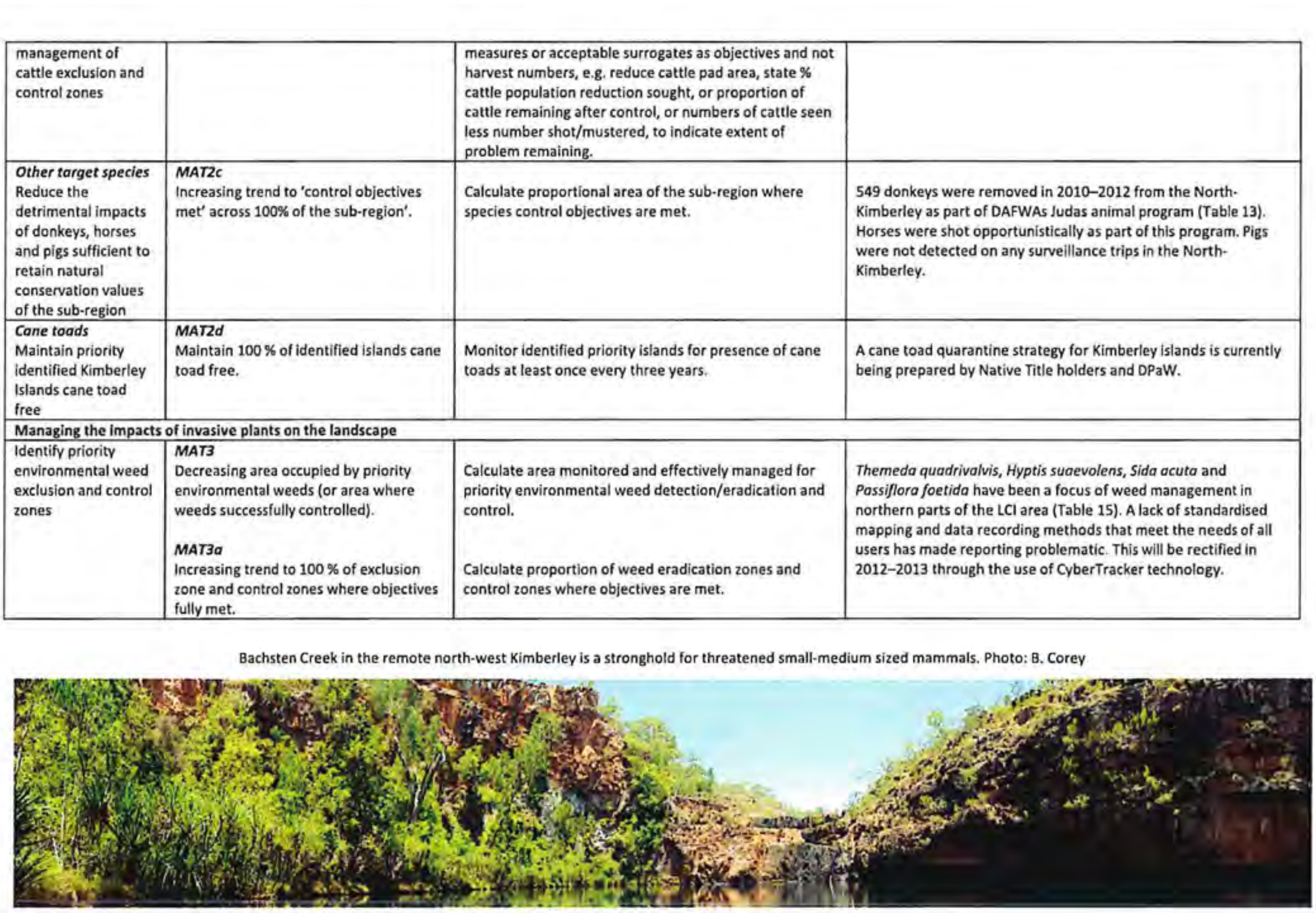
Progress towards delivering management actions is summarised against monitoring and reporting targets (see Shedley et al. 2012) in Table 10, with supporting information and context provided in the following text.

Table 10. Landscape Conservation Initiative management effectiveness in the North-Kimberley.

Management action	Management action effectiveness target	Method	Outcome
Managing inappropriate fire regimes			
MA1a Reduce proportion of the sub-region burnt on an annual basis	***MAT1a*** Increasing trend in the proportion of the sub-region (area of vegetation) unburnt in a single year.	Analysis of MODIS fire scar imagery to compare total area burnt by end of December each year.	There has been an increasing trend in the proportion of the total LCI area burnt each year from 2010 to 2012 (Fig. 13) but overall, no decrease in the proportion of the LCI area burnt annually (Fig. 13).
MA1b Reduce proportion of fires in late dry season	***MAT1b*** Increasing trend in proportion of fires that burnt in the early dry season.	Analysis of MODIS fire scar imagery, comparing total area burnt by end of June and December each year.	There has been an increasing trend in the proportion of early dry season fires each year from 2010 to 2012 (Fig. 13), and a substantial shift from less late dry season fires to more early dry season fires (Figs. 13, 14).
MA1c Reduce proportion of large aerial extent fires	***MAT1c*** Decrease the mean distance between unburnt patches.	Assess fire scars across the Kimberley and LCI area, in particular fauna and vegetation monitoring sites, determining average separation distances on an annual basis.	There has been a decrease in the mean and maximum distances from burnt to unburnt patches from 2010 to 2012 (Fig. 15), and from 2008 to 2012 (Fig. 15). There was a positive correlation between vegetation age and vegetation patchiness, and mammal abundance and diversity (Fig. 19), and a negative correlation between fire frequency and mammal abundance and diversity (Fig. 20).
MA1d Increase habitat value through effective fire management	***MAT1d*** An increasing trend in the proportion of fire sensitive vegetation in older age classes (> 3 years old).	Assess area of occupancy and age/diameter at breast height (DBH) of predetermined fire sensitive species in the landscape around fauna and vegetation quadrat monitoring sites.	There has been no change in the extent of old-growth vegetation (Fig. 16). Vegetation of 1 year old or less still comprises nearly 70 % of the total area of vegetation (Fig. 16). Old-growth vegetation is more patchily distributed across the landscape than before (Fig. 16). Fire sensitive, obligate seeding and fruit bearing species were often present at LCI sites, though never dominant. Further analysis of the distributions of these indicator species will be undertaken at the end of 2013, when three years of survey data are available.
Managing the impacts of introduced animals on the landscape			
Cattle Exclude/eradicate feral cattle from high conservation value areas wherever possible and reduce their impacts in other identified high conservation value areas through	***MAT2a*** Reducing trend in cumulative area of cattle pads in monitored areas ***MAT2b*** Increasing trend to 100 % achievement of targets in 'effective eradication', 'significant control' and 'control' zones.	Take aerial photographs along set transects at standard flying heights immediately after the wet season and compare density (number and cumulative area) of cattle pads as an index of cattle impacts each year. Calculate proportion of 'effective eradication', 'significant impact reduction' and 'impact reduction' areas where the control objectives are met. Use impact	Cattle pad monitoring sites were established in 2012 (Fig. 5), with more to be added in future years. Under conservative estimates, 12–46 % reductions in feral cattle numbers have been achieved across two of seven management zones (Table 12).

management of cattle exclusion and control zones		measures or acceptable surrogates as objectives and not harvest numbers, e.g. reduce cattle pad area, state % cattle population reduction sought, or proportion of cattle remaining after control, or numbers of cattle seen less number shot/mustered, to indicate extent of problem remaining.	
Other target species Reduce the detrimental impacts of donkeys, horses and pigs sufficient to retain natural conservation values of the sub-region	***MAT2c*** Increasing trend to 'control objectives met' across 100% of the sub-region'.	Calculate proportional area of the sub-region where species control objectives are met.	549 donkeys were removed in 2010–2012 from the North-Kimberley as part of DAFWAs Judas animal program (Table 13). Horses were shot opportunistically as part of this program. Pigs were not detected on any surveillance trips in the North-Kimberley.
Cane toads Maintain priority identified Kimberley Islands cane toad free	***MAT2d*** Maintain 100 % of identified islands cane toad free.	Monitor identified priority islands for presence of cane toads at least once every three years.	A cane toad quarantine strategy for Kimberley islands is currently being prepared by Native Title holders and DPaW.
Managing the impacts of invasive plants on the landscape			
Identify priority environmental weed exclusion and control zones	***MAT3*** Decreasing area occupied by priority environmental weeds (or area where weeds successfully controlled). ***MAT3a*** Increasing trend to 100 % of exclusion zone and control zones where objectives fully met.	Calculate area monitored and effectively managed for priority environmental weed detection/eradication and control. Calculate proportion of weed eradication zones and control zones where objectives are met.	*Themeda quadrivalvis*, *Hyptis suaevolens*, *Sida acuta* and *Passiflora foetida* have been a focus of weed management in northern parts of the LCI area (Table 15). A lack of standardised mapping and data recording methods that meet the needs of all users has made reporting problematic. This will be rectified in 2012–2013 through the use of CyberTracker technology.

Bachsten Creek in the remote north-west Kimberley is a stronghold for threatened small-medium sized mammals. Photo: B. Corey



Managing inappropriate fire regimes

There has been an increasing trend in the proportion of early dry season fires following increased fire management in the LCI area (ANOVA: $F_{2,10} = 6.099$; $P = 0.019$; Tukey pairwise comparisons: pre LCI vs. LCI, $P < 0.05$; LCI vs. transition, $P > 0.05$; transition vs. pre-LCI, $P > 0.05$; Figs. 13, 14) and a decreasing trend in the proportion of late dry season fires (ANOVA: $F_{2,10} = 6.099$; $P = 0.0019$; Tukey pairwise comparisons: pre LCI vs. LCI, $P < 0.05$; LCI vs. transition, $P > 0.05$; transition vs. pre-LCI, $P > 0.05$; Figs. 13, 14). There has been no change in the total area burnt following increased fire management (ANOVA: $F_{2,10} = 0.528$; $P = 0.605$; Fig. 13).

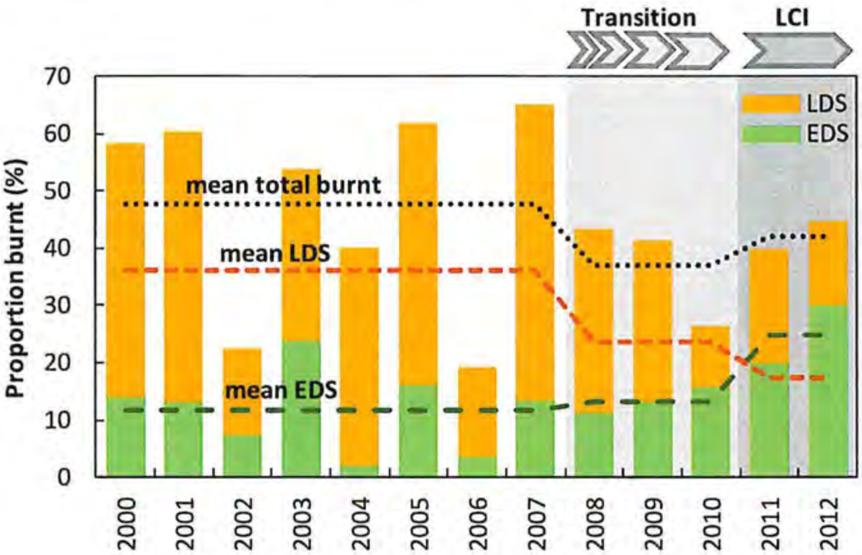


Fig. 13. Fire seasonality in the LCI project area from 2000–2012, derived from MODIS imagery. The lightly shaded area from 2008 to 2010 represents the transition period from a fire management approach that focused on the creation of blocks and buffers aimed to compartmentalise the landscape (2000–2007), to the introduction of the patch mosaic burning concept (2008–2010, lighter shaded area) which continued in 2011 and 2012 under LCI management of fire (darker shaded area).

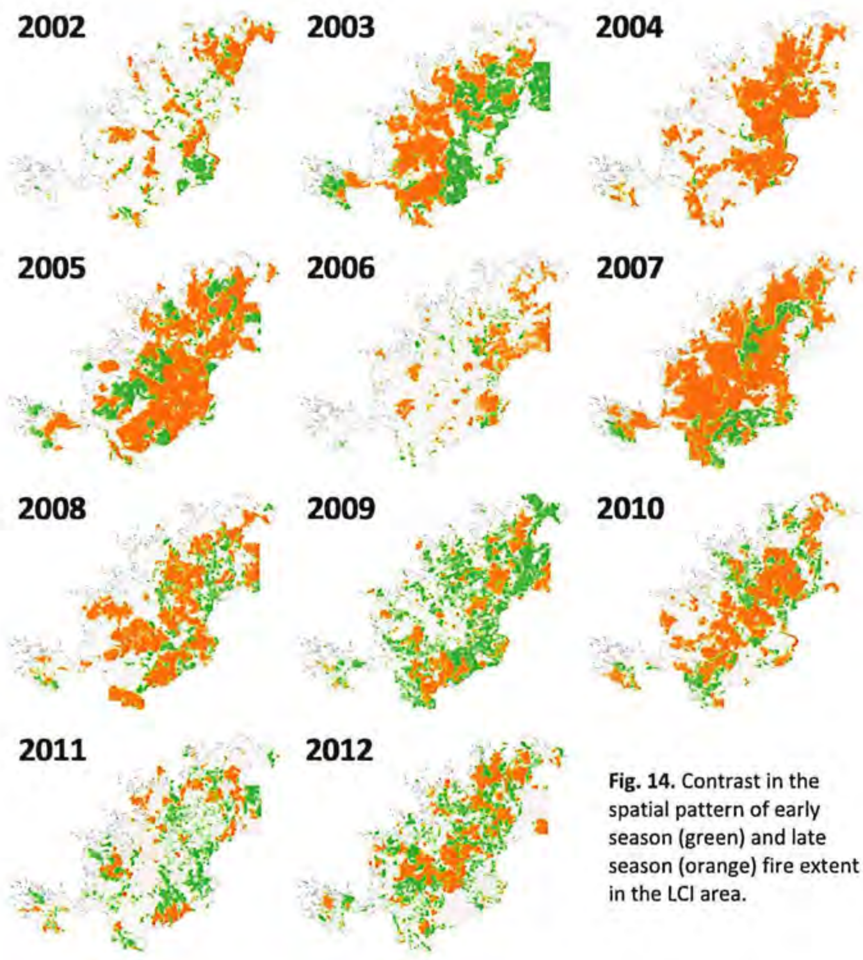


Fig. 14. Contrast in the spatial pattern of early season (green) and late season (orange) fire extent in the LCI area.

Both the mean and maximum distances from burnt to nearby unburnt vegetation that is three years of age (> 20 ha in size) decreased following increased management of fire (ANOVA: mean: $F_{2,8} = 13.728$; $P = 0.003$; Tukey pairwise comparisons: pre-LCI vs. LCI, $P < 0.05$; LCI vs. transition, $P > 0.05$; transition vs. pre-LCI, $P < 0.05$; maximum: $F_{2,8} = 70.150$; $P < 0.001$; Tukey pairwise comparisons: pre-LCI vs. LCI, $P < 0.05$; LCI vs. transition, $P < 0.05$; transition vs. pre-LCI, $P > 0.05$; Fig. 15a). The mean distance from burnt to nearby unburnt vegetation that is five years of age (> 20 ha in size) also decreased (ANOVA: mean $F_{2,6} = 5.833$; $P = 0.034$; Tukey pairwise comparisons: pre-LCI vs. LCI, $P < 0.05$; LCI vs. transition, $P > 0.05$; transition vs. pre-LCI, $P > 0.05$; Fig. 15b), whereas the maximum distance did not (ANOVA: maximum: $F_{2,6} = 3.140$; $P = 0.117$; Fig. 15b).

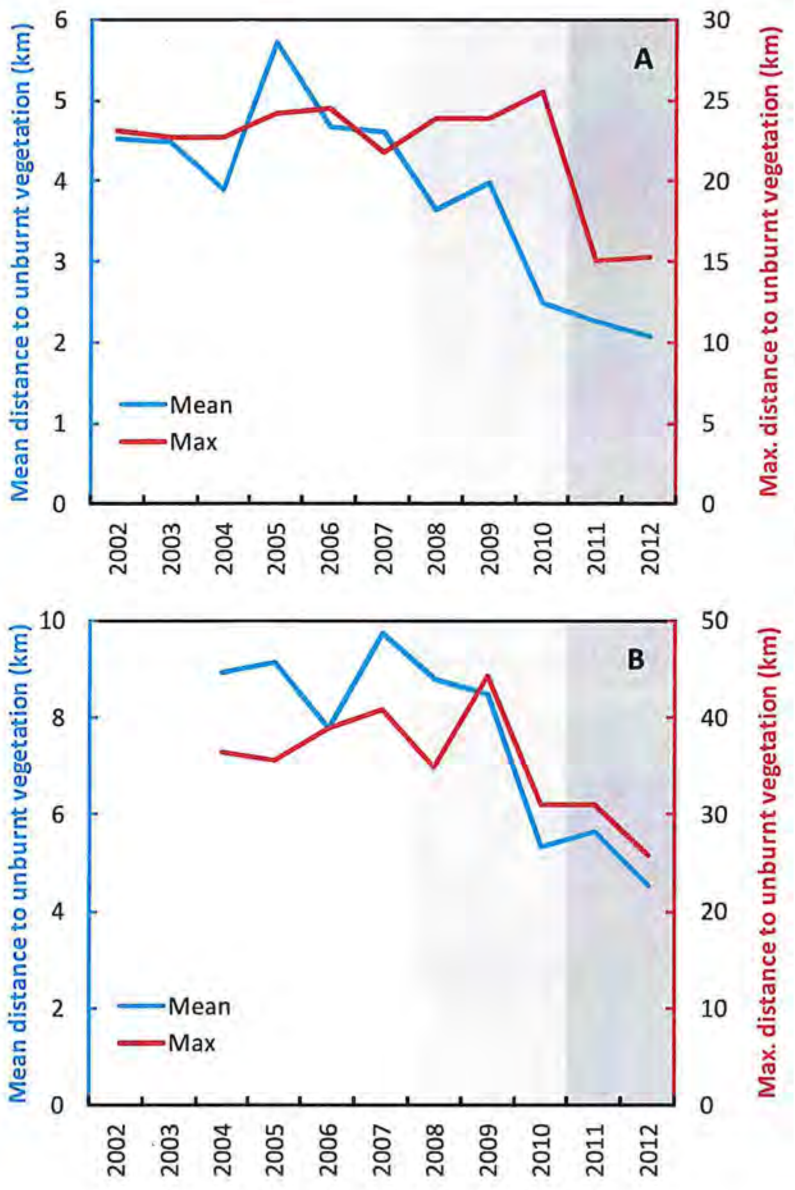


Fig. 15. The mean (blue line on first y-axis) and maximum (red line on second y-axis) distances from burnt areas to the nearest unburnt vegetation (> 20 ha in size) that is three years of age (A) and five years of age (B) from 2002 to 2012. The lightly shaded area from 2008–2010 represents when the management of fire increased and shifted from the creation of blocks or buffers, to the patch mosaic burning concept, and the darker shaded area from 2011–2012 represents the introduction of the LCI program, where this later method of fire management continued.

There has been no change in the extent of old growth vegetation with increased fire management (two-way ANOVA: management: $F_{2,45} = 0.047$; $P = 0.954$; age class: $F_{5,45} = 21.026$; $P < 0.001$), with vegetation that is < 1–2 years of age still comprising the majority of vegetation (Fig. 16). There has been an increase in the average number of patches (> 20 ha) that make up these age classes (two-way ANOVA: management: $F_{2,43} = 15.134$; $P < 0.001$; Tukey pairwise comparisons: pre-LCI vs. LCI, $P < 0.05$; LCI vs. transition, $P < 0.05$; pre-LCI vs. transition, $P < 0.05$; Fig. 16). Fire frequency in the LCI project area has decreased (Figs. 17, 18), but this was not significant (two-factor ANOVA: management: $F_{1,5} = 0.021$; $P = 0.992$; interval: $F_{5,11} = 9.304$; $P = 0.014$).

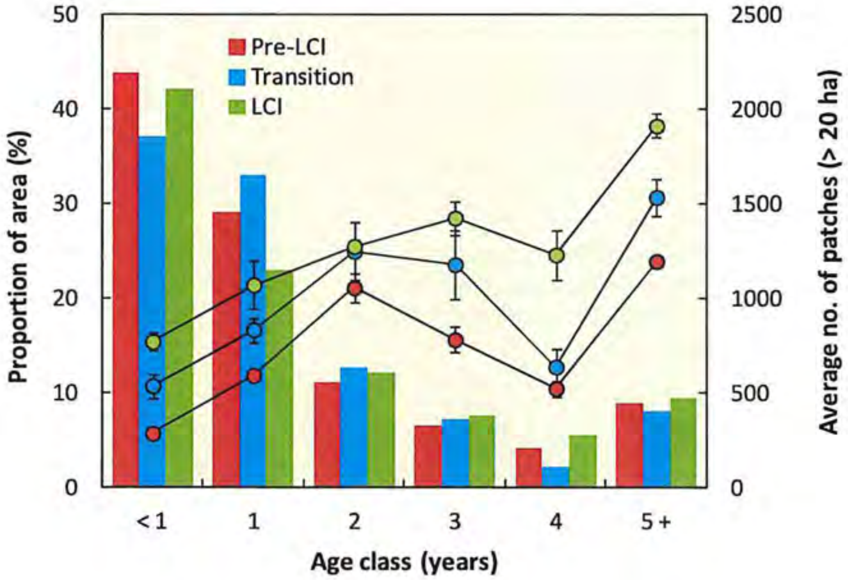


Fig. 16. The extent of vegetation age (measured as time since last fire) in the LCI project area, pre-LCI (2000–2007, red bars), during the transition period (2008–2010, blue bars) and following (2011–2012, green bars) LCI fire management on the first y-axis; the average number of patches (> 20 in size) for each age class is presented on the second y-axis (red circles, pre-LCI; blue circles, transition; green circles, LCI). Note: errors are ± 1 standard error.

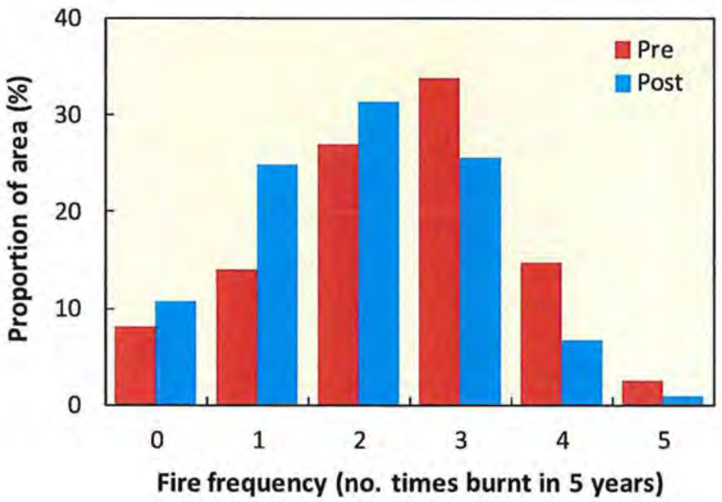


Fig. 17. Fire frequency (number of times burnt) in the LCI project area for the periods 2003–2007 (before increased fire management, red bars) and 2008–2012 (following increased fire management, blue bars).

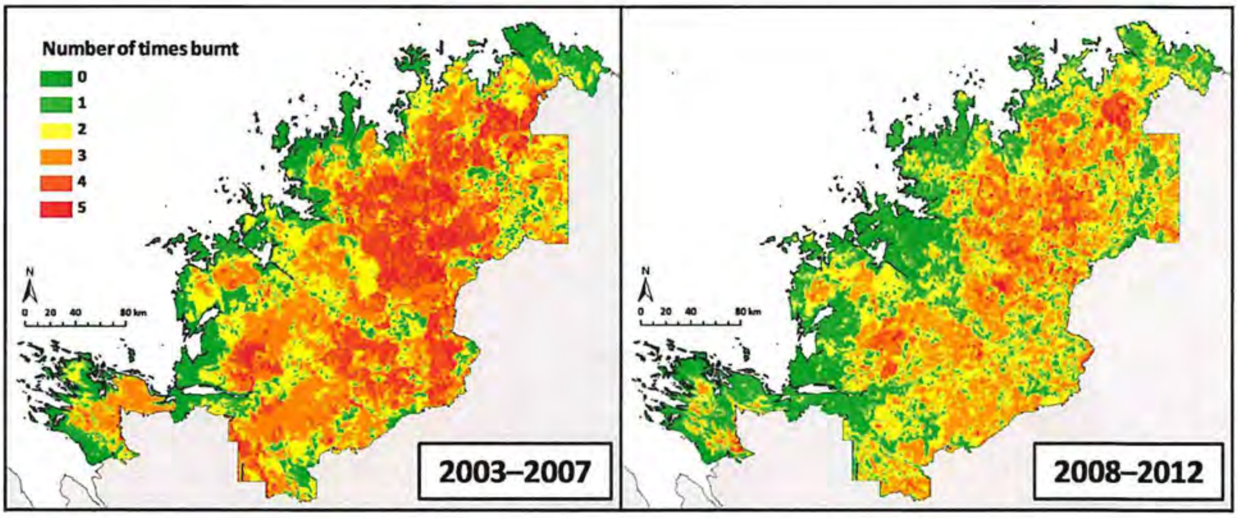


Fig. 18. Fire frequency (number of times burnt) in the five years prior to (2003–2007) and following (2008–2012) increased management of fire in the LCI project area. Red areas indicate high fire frequency (burnt every year) whilst green areas have not been subject to fire.

Diversity of vegetation age classes was correlated with mammal species richness at a larger scale (3 and 5 km), but not within a 1 km radius of monitoring sites (Table 11; Fig. 19). Diversity of vegetation age classes was not related to abundance at any scale (Table 11; Fig. 19). Increasing vegetation age at all examined spatial scales, had a positive influence on both abundance and species richness (Table 11; Fig. 19). These were heavily influenced by the Mount Trafalgar cluster of sites, which had very high abundance, low vegetation patchiness, but long unburnt vegetation (7–8 years old). When this cluster was removed from the analysis, mean vegetation age was still correlated with abundance and species richness at all scales ($P < 0.05$). However, diversity of age classes became a better predictor of abundance and species richness within a 3 and 5 km radius ($P < 0.05$), but not within 1 km ($P > 0.05$) of monitoring sites. Fire frequency in the ten years preceding each survey was negatively correlated with both mammal abundance and diversity (abundance: $R^2 = 0.510$; $F_{1,14} = 14.549$; $P = 0.002$; diversity: $R^2 = 0.443$; $F_{1,14} = 11.143$; $P = 0.005$; Fig. 20), even when the Mount Trafalgar cluster was removed from the analysis (abundance: $R^2 = 0.311$; $F_{1,13} = 5.873$; $P = 0.031$; diversity: $R^2 = 0.322$; $F_{1,13} = 6.168$; $P = 0.027$).

	R^2	$F_{1,14}$	P
Trap success			
1 km search radius			
No. vegetation age classes	0.001	0.014	0.908
Mean vegetation age	0.432	10.669	0.006*
3 km search radius			
No. vegetation age classes	0.186	3.200	0.095
Mean vegetation age	0.477	12.752	0.003*
5 km search radius			
No. vegetation age classes	0.203	3.567	0.080
Mean vegetation age	0.393	9.062	0.009*
Species diversity			
1 km search radius			
No. vegetation age classes	0.003	0.048	0.829
Mean vegetation age	0.320	6.573	0.023*
3 km search radius			
No. vegetation age classes	0.281	5.483	0.035*
Mean vegetation age	0.427	10.415	0.006*
5 km search radius			
No. vegetation age classes	0.392	9.024	0.009*
Mean vegetation age	0.483	13.056	0.003*

Table 11. Summary of regression analysis for both trap success and species diversity, and vegetation age and diversity at LCI monitoring sites for 2011 and 2012. Note: data are pooled by geographic cluster and * denotes significant ($P < 0.05$) values.

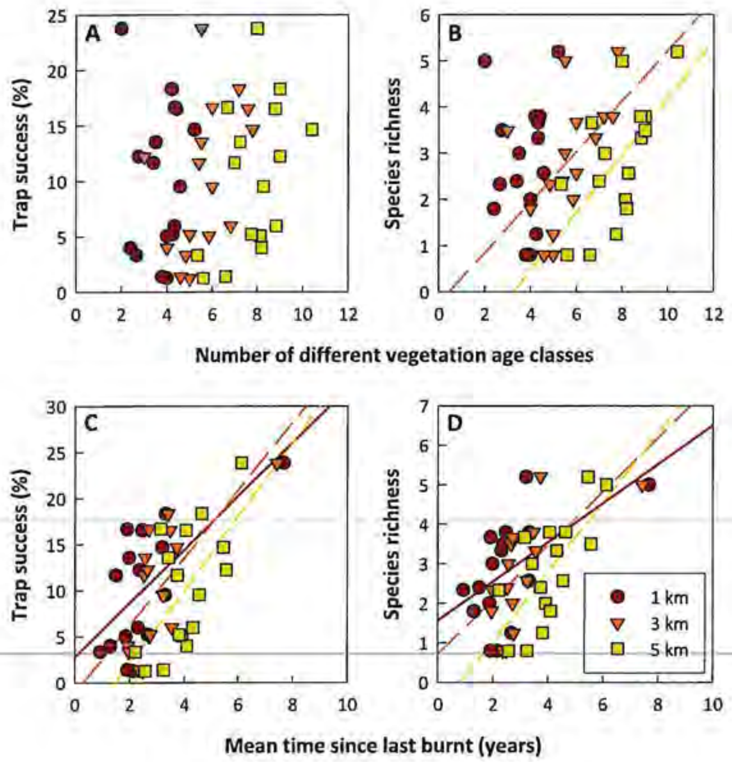


Fig. 19. The relationship between mean abundance and species richness of small-medium sized mammals and: diversity of vegetation age classes within a 1, 3 and 5 km radius of monitoring sites (A, B); and mean vegetation age (C, D). Note values are means of site clusters.

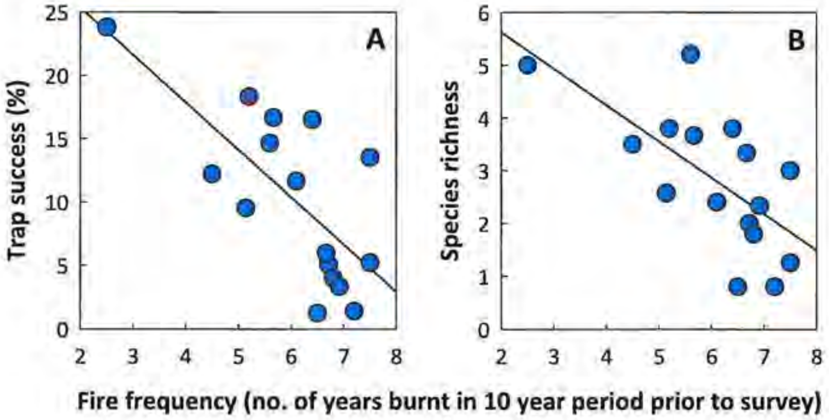


Fig. 20. The relationship between mean abundance (A) and species richness (B) of small-medium sized mammals and mean site fire frequency (number of fires) in the 10 years prior to survey. Note: values are means of site clusters.

Managing the impacts of invasive animals

Cattle

Aerial culling operations were focused in management zone one in 2010–11, and zones one and four in 2011–12 (Table 12; Fig. 5). Management zone one has experienced the greatest reduction in cattle numbers, where current estimates (see Approach) suggest a 45.6 % reduction since culls begun in 2007–08 (Table 12). Culls in management zone four have achieved an 11.9 % reduction since 2010–11. Advice from staff at Mitchell River National Park and nearby community members (see Fig. 5) suggests these figures are accurate, with cattle becoming increasingly less prevalent around the Mitchell Plateau area.

Table 12. Summary of North-Kimberley aerial culling operations. Note: *Y*, estimated population size; *K*, maximum carrying capacity. Final reductions (*n*) account for the rate of increase (*r*) generated from each annual reduction using the equation: $r = r_m(1–N/K)$ where r_m is the maximum rate of increase (set at 0.17) and *N* is the reduced population size following each cull (see Bayliss and Yeomans 1989a).

Zone	*Y*	*K*	Number of animals removed					*N*	Reduction	
			2007-08	2008-09	2009-10	2010-11	2011-12		*n*	%
MZ1	10,828	13,847	1,701	747	1,153	1,971	2,855	5,891	4,937	45.6
MZ4	23,201	29,290	0	0	0	893	3244	20,441	2,760	11.9
Total	**34,029**	**43,137**	**1,701**	**747**	**1,153**	**2,864**	**6,099**	**26,332**	**7,697**	**22.6**

Cattle impacts were highest at monitoring sites in the King Edward, Laterite and Surveyors Pool clusters (Fig. 21). Monitoring sites within the Cascade Creek and Trafalgar clusters had no evidence of feral cattle impacts (Fig. 21). There were pronounced decreases in cattle impact scores from 2011 to 2012 across the Laterite, Surveyors Pool and Yalgi clusters following cattle culls in management zone one in 2011 (Fig. 21; see Table 12).

There was a negative correlation between small-medium sized mammal trap success and impacts from feral cattle at associated monitoring sites ($R^2 = 0.447$; $F_{1,8} = 6.461$; $P = 0.035$; Fig. 22a). Similarly, there was a negative relationship between species richness and feral cattle impacts at monitoring sites ($R^2 = 0.529$; $F_{1,8} = 8.997$; $P = 0.017$; Fig. 22b). Because cattle impacts increased among sites further from the coast, cattle impacts, mammal abundance/richness and geography may be confounded. Hence we cannot be confident this early in the monitoring program that reduced mammal abundance in these areas is actually due to cattle impacts.

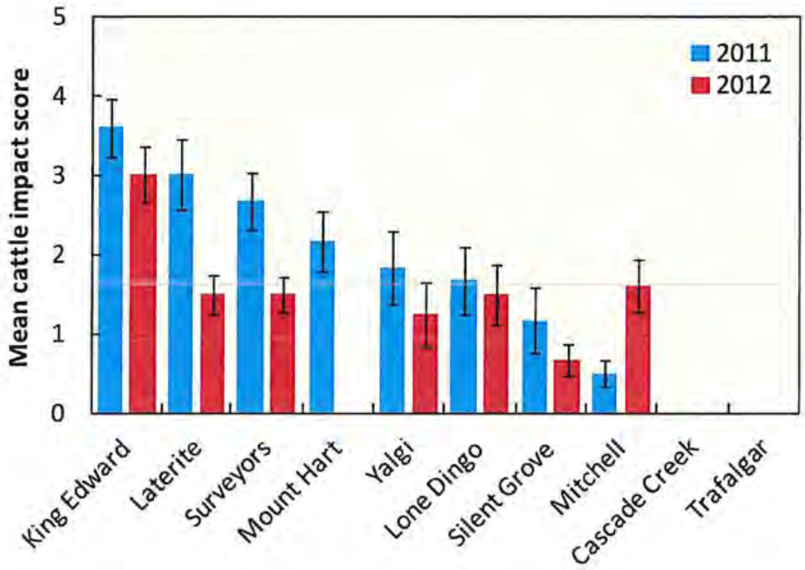


Fig. 21. Mean scores of feral cattle impacts across monitoring clusters in 2011 (blue bars) and 2012 (red bars) in the LCI project area. Note: errors are ± 1 standard error

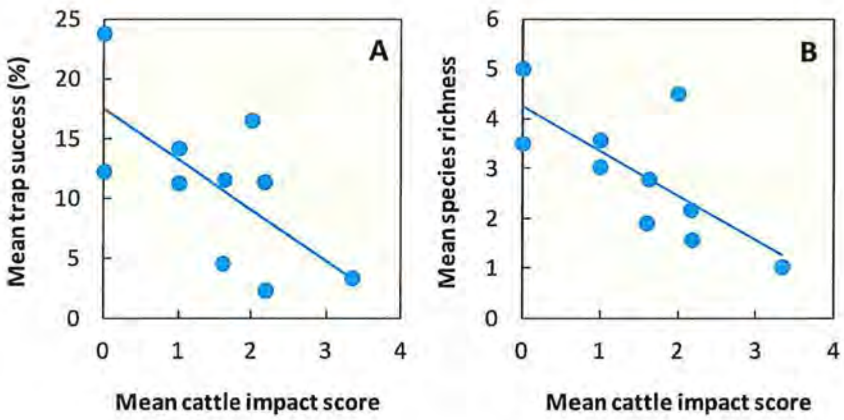


Fig. 22. The relationship between mean trap success and mean species richness of small-medium sized mammals and mean cattle impact scores at monitoring sites. Note: values are means of site clusters, pooled across years.

Photo monitoring sites were established at 18 remote sites across the North-Kimberley in May 2012 (see Fig. 5). These were scored and will provide a baseline for future monitoring. These sites will be increased throughout the north-Kimberley in future years in collaboration with Wunambal Gaambera Aboriginal Corporation and Uunguu Rangers as well as Wilinggin Aboriginal Corporation and Wunggur Rangers, and will be visited annually at the end of the wet season and dry season.

Donkeys

A total of 599 donkeys were removed from the North-Kimberley from 2010–12 (Table 13), as part of DAFWAs Judas donkey program.

Table 13. Summary of DAFWA delivered donkey 'Judas' animal program in the North-Kimberley (*denotes horses).

Location	2010–11			2011–12		
	Donkey culled	Other species culled	Donkey collared	Donkey culled	Other species culled	Donkey collared
Forest River	12	20*	7	83	0	6
Carson River Station	57	9*	16	66	0	3
Drysdale River National Park	182	0	32	199	0	32
Total	**251**	**29**	**55**	**348**	**0**	**41**

Managing the impacts of invasive plants

Invasive plants were a minor component of vegetation at monitoring sites in 2011 and 2012. Almost all weeds were recorded at sites with volcanic geology (14 of 15 occurrences), which represent the more fertile landscape elements in the North-Kimberley and sites with the greatest evidence of cattle impacts on vegetation. There were three site incidences of *Stylosanthes scabra*, four incidences of *Hyptis suaveolans* and nine incidences of *Passiflora foetida* (Table 14). The former two species were associated with sites frequently visited by cattle or with disturbed vegetation and were associated with watering points (see above) where cattle impacts were greatest. Infestations of these species were small, consisting of either scattered individuals or small clumps up to 25 m^2 in size. The latter species was ubiquitous throughout volcanic woodland and at the edges of rainforest patches on the side of Mount Trafalgar in the remote north Kimberley coastal region. *Passiflora* was generally present at < 1 % of ground cover; however a few clumps of this species near rainforest edges (possibly associated with underground soaks) had become semi-invasive. These latter sites showed little impact of recent fire or cattle, indicating that this invader either does not require disturbance in these habitats, or that it benefits from lack of disturbance through increases in available soil nutrients.

Table 14. Localities and condition of weed occurrences at LCI monitoring sites in the North-Kimberley.

Species	Site name	Latitude	Longitude	Infestation
Stylosanthes scabra	LCI027	-14.96896	125.86941	Individuals, scattered
	LCI028	-14.97009	125.86943	Individuals, scattered
	LCI029	-14.95825	125.96404	25 m^2 patch, scattered
Hyptis suaveolans	LCI201	-17.06883	125.24725	30 m^2, common in patches
	LCI204	-17.00703	125.21838	Individuals, scattered
	LCI209	-16.78791	124.92009	Individuals, scattered
	LCI213	-17.01915	125.23212	3 m^2, scattered
Passiflora foetida	LCI213	-17.01915	125.23212	5 m^2, scattered
	LCI311	-15.279917	125.07405	20 m^2, scattered
	LCI313	-15.2803	125.07245	Individuals, scattered
	LCI314	-15.27897	125.07339	Patch 100 m^2, common
	LCI315	-15.27788	125.07486	Individuals, scattered
	LCI316	-15.28091	125.07185	Individuals, scattered
	LCI317	-15.2816	125.07139	100 m^2, common
	LCI318	-15.27921	125.07355	100 m^2, common
	LCI319	-15.27764	125.07607	Individuals, scattered

Invasive plants that were a major focus of weed management in 2010–2011 and 2011–2012 are presented in Table 15. Work to control *Themeda quadrivalvis* (grader grass), a Weed of National Significance, in the Mitchell River work Zone has been particularly effective in treating infestations before they set seed.

Table 15. Summary of weed management activities in the North-Kimberley LCI region.

Area/Location	Species	2010–11 and 2011–12
Mitchell River Work Zone	*Hyptis suaevolens* (hyptis/stinking horehound)	DPaW, Uunguu Rangers
	Passiflora foetida (passionfruit vine)	
	Sida acuta (spinyhead sida)	
	*Themeda quadrivalvis** (grader grass)	
Mount Hart	*Colocasia esculenta* (taro)	DPaW, Wunggurr Rangers

Wet season control of grader grass at the APT tourist camp at the Mitchell Plateau. From left to right: Lionel Catada and Selwyn Malay (employed on a fee-for-service agreement with Wunambal Gaambera Aboriginal Corporation) and Greg Goonack (DPaW ranger). Photo: Richard Tunnicliffe

DISCUSSION

Landscape outcomes

Rainforest patches

The Rapideye remote sensing work by van Dongen et al. (2013) has provided a baseline from which to monitor changes in both canopy cover and extent of rainforest patches (Table 7; Figs. 9, 10). Whilst remote sensing technology has been used to monitor changes in vegetation in other areas of northern Australia, the Rapideye work in the North-Kimberley is the first of its kind.

Despite evidence that many rainforest patches are severely damaged by fire and cattle, leading to intrusions by invasive plants in the Northern Territory (Russell-Smith and Bowman 1992), monitored rainforest patches in the Mitchell and Prince Regent regions in 2012 were apparently in good condition (in terms of closed tree canopy cover, limited ground layer disturbance by fire and cattle and incursion by invasive plants; Table 5; Fig. 6). No sites had severe damage resulting in loss of tree canopy cover or reduced structural complexity. Although structural vegetation integrity was maintained at all monitored rainforest sites, mammal abundance and diversity was low in the more southern Mitchell Plateau areas, coinciding with greater cattle and fire impacts.

Mammal abundance

Mammal assemblages (on average) appear to have remained stable (Figs. 11, 12; Tables 8, 9). However, it is probably too early in the monitoring program to make any inferences about mammals owing to the ephemeral nature of their populations. Mammal abundance was high relative to other regions of northern Australian (Kutt and Woinarski 2007, Woinarski et al. 2010). Abundance was high relative to baseline surveys in the North-Kimberley (Bradley et al. 1987, Start et al. 2007, Legge et al. 2008, 2011a, Radford 2012, Start et al. 2012, Radford et al. submitted). There was no evidence of collapse of mammal assemblages as in the Northern Territory (Woinarski et al. 2001, 2010).

The black-footed tree rat (*Mesembriomys gouldii*), has not been recorded in the North-Kimberley since 1981–82 (from the northern end of Mitchell Plateau in bauxite woodland; Bradley et al. 1987), despite numerous mammal survey programs between 1964 and 2013. Its Northern Territory population is now classed as Vulnerable, having suffered a rapid and severe decline in abundance and range during the last decade (Woinarski et al. 2010). Its distribution has contracted in all parts of its mainland range as a result of frequent, extensive and intense fires (Ziembicki et al. 2013). Determining the current status of this species in the North-Kimberley is a priority.

Other arboreal species are infrequently encountered (e.g. *Phascogale tapoatafa, Trichosurus vulpecular arnhemensis*), but this may be an artefact of current sampling techniques which are biased towards terrestrial mammals. Other monitoring techniques are now planned to address this (camera traps, nesting boxes), and a project is currently being developed by regional staff and the Quantitative and Applied Ecology Group at the University of Melbourne to examine whether the availability of tree hollows limits the abundance and diversity of arboreal mammals in frequently burnt savannas of the North-Kimberley.

Our preliminary analysis of 2011 and 2012 mammal monitoring data and fire metrics, in particular vegetation patchiness (Fig. 19a, b), vegetation age (Fig. 19c, d) and fire frequency (Fig. 20a, b) provides further support to the notion that both high fire frequency and size are drivers of decline across northern Australia (e.g. Woinarski et al. 2005; 2010; Radford 2012; M. Lawes, unpublished data). What is less clear is the role of patchiness (Fig. 19a, b). Our early analysis of patchiness and abundance and species richness with only two years of data suggests that patchiness plays a role at multiple scales, but further analysis taking into account the size/contribution of different vegetation age patches, the proportion burnt early vs. late and fire frequency is needed before any firm conclusion can be drawn.

Native vegetation condition

Vegetation structural attributes were consistent with those reported for similar savanna vegetation in the Kimberley (Vigilante and Bowman 2004b) and the Northern Territory (Williams et al. 1999; 2003). Grass cover was high (> 70 % when unburnt) and tree and shrub cover low (< 25 %) in volcanic woodlands (Vigilante and Bowman 2004b, Table 5). Sandstone woodland and rocky habitats in the LCI area (Table 5) had higher tree cover (24 and 9 %) than reported in equivalent vegetation in the North-Kimberley in a lower rainfall zone (3.5 % and 1.9 %; Vigilante and Bowman 2004b). *Eucalyptus tetrodonta/E. miniata* woodland/forest savanna vegetation in the North-Kimberley (Laterite) had higher basal area (14-16 $m^2.ha^{-1}$, Fig. 6) than in similar vegetation in the Northern Territory (11.4 $m^2.ha^{-1}$; Williams et al. 2003). Williams et al. (1999; 2003) reported a 20 % reduction in basal area under sustained annual high intensity fires, and a 40 % reduction in basal area after a single very high intensity fire respectively, due to sapling and large tree mortality. Similarly, basal area declined in riparian woodland under higher intensity dry season fire regimes relative to low intensity wet season burning in north Queensland savanna woodlands (Radford et al. 2008). No equivalent changes in tree canopy cover or basal area were detected from 2010–11 to 2011–12 among North-Kimberley monitoring sites (Table 5, Fig. 6). Annual low intensity fires caused no change in tree basal area in the Kapalga study in the NT (Williams et al. 1999; 2003), and little change was evident at our monitoring results. This suggests that most fires experienced at monitoring sites were of relatively low intensity, or if changes have occurred, they occurred prior to LCI monitoring.

In the broader context, North-Kimberley savannas are among the best in Australia in terms of standard vegetation condition indices. Annual grasses including annual *Sorghum spp.* (also known as *Sarga* spp. in the NT) remain a small component of the grass layer at the majority of sites. This contrasts with many NT savannas where *Sorghum* spp. dominates grass layers (Russell-Smith et al. 2003; Andersen et al. 2005). Perennial tussock grasses dominated at almost all monitoring sites. This is a strong indicator of healthy savanna ecosystems across northern Australia (Speck et al. 1960; Craig 1997; Ash et al. 1997; Ash and Mclvor 1998). There was no evidence of significant patch grazing or erosion associated with feral cattle damage, nor any indication of transition to alternative degraded vegetation states as described for savannas of north Queensland under grazing (Ash et al. 1997). Herbaceous and woody alien invasive plants made up only a minor component of vegetation at a small number of sites, which contrasts with many savannas both in the NT (Rossiter et al. 2003; Setterfield et al. 2005) and in QLD (Ash et al. 1997; Grice et al. 2000; Grice 2006; Radford et al. 2008) where invasive plants often dominate vegetation.

Fire sensitive (e.g. *Calytrix exstipulata, C. brownii*), obligate seeding species (e.g. *Acacia translucens*), and fruit bearing species (e.g. *Planchonia careya, Ficus platypoda*) were often present among LCI sites, though never dominant. Further analysis of the distributions of these indicator species will be undertaken at the end of 2013, when three years of survey data are available.

Native vegetation condition is considered healthy (Tables 5, 6, Fig. 6) and (on average) possibly improving (Table 5, Fig. 9) and the time series analysis of the Landsat imagery will provide a means for assessing vegetation cover over time. Caution must be exercised when interpreting the Landsat data, as the window of analysis (2003 to 2012) is relatively short and a longer history of images may suggest different trends (R. Van Dongen, unpublished data). At the commencement of the project, Landsat imagery was only available back to 2003, but imagery dating back to 1987 is now available. Processing the entire archive will provide a longer baseline of vegetation dynamics to compare management actions against.

Management action effectiveness

Managing inappropriate fire regimes

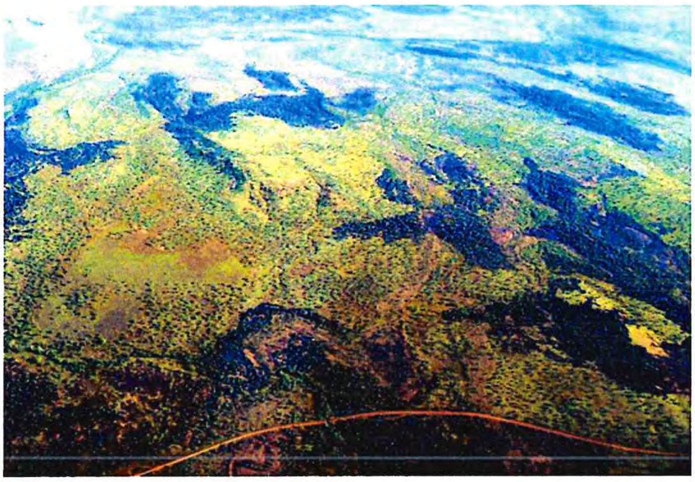
The proportion of the LCI area burnt annually has decreased only slightly, but substantially less of this is from late season fires (Fig. 13). There has been a large shift from a pattern of extensive late dry season fires, to more early dry season fires with increased fire management (Figs. 13, 14). Historically, late dry season fires (on average) made up 76.3 % of annual fire scars, but now comprise 53.2 % of annual fires (Figs. 13, 14). This trend is a significant improvement, but the target in future years should be to reduce this even further.

How do these results compare to other landscape-scale fire management programs across northern Australia? To answer this question we compiled fire seasonality data from fire programs in the Gulf of Carpentaria, Western Arnhem Land, Kakadu National Park and Fish River Station (Table 16). These programs have achieved 3.4–40.2 % reductions in the total area burnt, 3.3–18.1 % increases in the proportion of early dry season fires (they decreased at Fish River) and 10.2–34.8 % reductions in late dry season fires following improved fire management (Table 16). The respective metrics in the LCI area are generally within these ranges.

Differences in seasons and curing rates (e.g. windows for prescribed burning and timing of the fire season – later vs. earlier), fuel types (e.g. annual vs. perennial grasses) and rainfall, not to mention the size of the area that fire is being managed over, suggest that it may not be sensible to take too much from such broad-brush comparisons. They do suggest that metrics such as the total area burnt may be relatively insensitive to management, at least over larger areas (> 2,000 km^2; see also van Wilgen et al. 2004; Vigilante et al. 2004), and largely driven by rainfall in the preceding wet seasons (Legge et al. in prep.). Collectively, this suggests that land managers have far more control over when fire occurs (e.g. early vs. late) and hence the severity of fire, rather than how much fire occurs.

Table 16. Contrast in fire seasonality across northern Australian fire management programs (* no data available for 2012). Note: EcoFire is not presented as a standalone program, but is included in the LCI.

Project	Region	Size of project area (km^2)	Total area burnt (%) Pre	Total area burnt (%) Post	Reduction	EDS (% of total area) Pre	EDS (% of total area) Post	Increase	LDS (% of total area) Pre	LDS (% of total area) Post	Reduction	Baseline period (years)	Imagery used to derive data	No. of increased management years (at 30-Dec 2012)	Reference
NK LCI	North-Kimberley	65,194	47.7	39.1	8.6	11.5	17.9	6.4	36.2	21.1	15.1	8	MODIS	5	This report
WALFA	Western Arnhem Land	28,282	39.6	31.8	7.8	7.6	20.9	13.3	32.0	10.9	21.1	15	Landsat	9	Russell-Smith et al. 2013
NT Gulf	Gulf of Carpentaria	16,172	30.3	26.9	3.4	2.0	20.1	18.1	28.4	6.7	21.7	5	MODIS	3*	Jarrad Holmes, unpublished data
KNP	Kakadu National Park 'stone country'	3,899	27.6	20.5	7.1	9.0	12.3	3.3	18.5	8.3	10.2	26	Landsat	5	Murphy 2013
Fish River	Daly River	1,780	74.3	34.1	40.2	38.2	32.9	-5.3	36.0	1.2	34.8	9	MODIS	3	Shaun Ansell, unpublished data
		Mean	**43.9**	**30.5**	**13.4**	**13.7**	**20.8**	**7.2**	**30.2**	**9.6**	**20.6**	**13**		**5**	



An example of an early dry season landscape mosaic at Silent Grove in the King Leopold Ranges (left) and Dambimangari ranger Charles Numandumah undertaking prescribed burning with DPaW on Kunmunya Aboriginal Lands Trust (below). Photos: Ed Hatherley

There has been a large reduction in the distances from burnt to nearby old-growth vegetation (Figs. 15a, b). The average distance from burnt areas to nearby three year old unburnt vegetation (> 20 ha in size) has decreased from 4.6 km (SD = 0.59 km) to 2.2 km (SD = 0.14 km) and from 8.9 km (SD = 0.83 km) to 5.1 km (SD = 0.79) for five year old vegetation (> 20 ha). The overall extent of this old-growth vegetation has not changed however (Fig. 16). Vegetation that is one year old or less still makes up nearly 70 % on average of the LCI area (Fig. 16). The good news is that this old-growth vegetation is far more patchily distributed across the landscape (Fig. 16), rather than in a few larger patches, greatly reducing the risk of all old-growth vegetation being lost in a single wildlife. Fire frequency has been somewhat reduced (Figs. 17, 18), although not significantly so, suggesting greater emphasis may need to be placed on reducing this in future years.

Fire management in the LCI area has achieved two of the four initial targets (see Table 10): increased the proportion of early dry season fires (MAT1b) and decreased the distance between burnt and unburnt vegetation (MAT1c). Given that the total area burnt (MAT1a) may be relatively insensitive to management over large areas (see Table 16), it may be best to set a target specifying that the total area burnt should not exceed 40–45 % each year and that late season fires should not contribute more than 30–40 % of annual fire scars. The extent of old-growth vegetation has not increased (MAT1c), and thus a priority in future years will be to significantly improve this metric. Reducing fire frequency, to intervals greater than three years unburnt should also be a target in future years.

The focus of prescribed burning should not only be to limit the extent of large wildfires, but also to increase the area of old growth vegetation. This may be achieved via the application of landscape scale mosaics of different vegetation age classes and where appropriate, targeted fire management that utilises natural geographic barriers to prevent the loss of large areas of old growth vegetation in late dry season fire events. The Kimberley Region Fire Management service plan is currently being reviewed, and specific strategies to address this can be formulated in this plan.

Managing the impacts of invasive animals

Without firm estimates of cattle numbers in the North-Kimberley it is hard to gauge management success. Using conservative estimates, 11.9 % and 45.6 % reductions have been achieved across two of the seven management zones (after accounting for rates of increase generated from culls; Table 12). Our monitoring results – that show higher mammal richness and abundance in the absence of cattle impacts (Fig. 22) – are consistent with the recent findings of Legge et al. (2011a), where small-medium sized mammals increased in abundance following the removal of invasive herbivores.

A priority is to find aerial survey methods that meet Departmental AOPs and provide sound estimates of feral cattle numbers. These methods should be relatively inexpensive and be useable by all partners in the North-Kimberley. Ideally, these should be repeated every 1–2 years so progress towards achieving reductions can be tracked. Sound estimates (density) can then be linked to measures of impact (cattle pad monitoring), and in turn vegetation and mammal data, allowing us to explicitly demonstrate that reducing feral cattle numbers has a positive influence on biodiversity. This will also enable us to identify if density thresholds exist for maintaining numbers of 'killer' (or food) animals for community members in Kandiwal and Kalumburu (see Figs. 1, 5). Furthermore, this data can be linked with mammal, vegetation condition and fire data in complex ecological models to identify the relative effects and interactions of each; such analyses have not previously been undertaken for northern Australia as this type of data has not been available.

The number of feral animal monitoring sites throughout the LCI needs to be increased and other impact measurements such as water quality, aquatic diversity, etc. at waterholes (e.g. Ens et al. 2010), need to be incorporated into the monitoring program. Water holes are utilised more frequently by cattle in the late dry season, and thus impacts are more concentrated and easily measured. A suite of appropriately chosen sites would yield more valuable data on the positive effects of removing cattle from the landscape. Such methods align with the Healthy Country Plans of partners (e.g. Wunambal Gaambera Aboriginal Corporation 2010; Balanggarra Aboriginal Corporation 2011; Dambimangari Aboriginal Corporation 2012; Wilinggin Aboriginal Corporation 2012), and thus would provide further opportunities for collaboration.

Feral pigs are largely absent from the northern part of the LCI area, but sightings are increasing in the more southern areas, particularly around Mount Elizabeth Station and in the northern part of King Leopold Ranges Conservation Park. The surveillance of pigs needs to continue and incursions monitored, and any sightings need to be followed up on.

Feral cats have not been a focus of feral animal control programs in the North-Kimberley. Although Carwardine et al. (2012) suggest that ceasing dingo baiting programs is the most cost effective method for preventing further biodiversity losses as a result of predation by cats in the North-Kimberley; the studies used to justify the mesopredator effect (dingoes regulated cat activity) have recently been questioned by Allen et al. (2013). Managing inappropriate fire regimes and feral herbivores, although more expensive, was ranked as being more feasible (Carwardine et al. 2011; 2012) and as Legge et al. (2011a; 2011b) have demonstrated, provide positive benefits for biodiversity. However, DPaW is a partner in cat research with AWC which seeks to quantify the impacts of cats on native mammals, and to find ways of limiting those impacts across the landscape. This work is funded through the KSCS.

Managing the impacts of invasive plants

Whilst on-ground weed management works have been effective at a localised scale, mapping techniques and data management have not been consistent across the LCI area which has made it difficult to report against management action targets (see Table 10; see also Shedley et al. 2012). This will be overcome in future years, by the adoption of CyberTracker software on ruggedized Personal Digital Assistants (PDAs). This technology is used widely across northern Australia by Indigenous land management (ranger) groups and other land management agencies (see Ansell and Koenig, 2011). CyberTracker is a free online software program that was initially developed to record data by animal trackers in Africa (see http://www.cybertracker.org). All data recorded is geo-referenced and stored in a format that allows easy access, display and analysis irrespective of user capacity and this technology has revolutionised data recording and management for land management activities across northern Australia.

Implications for management

Based on early monitoring data and lessons learnt from 2010–12, we suggest the following activities and priorities as part of the LCI adaptive management program:

Biodiversity indicators

- Use Landsat and Rapideye imagery to compare the trajectories of rainforest patches (canopy cover and extent) in both northern regions of the LCI project area (which generally have a better fire history and fewer feral cattle) and southern and inland regions (that have a higher fire frequency and more cattle);
- Use Landsat imagery to analyse images from 1987–2003 and add these to the archive for further analysis of native vegetation condition. Compare trends in better and poorer performing areas in terms of fire and feral herbivores (as above);
- Undertake targeted surveys for *Mesembriomys gouldii* in likely habitats;
- Establish a project to examine the role that fire plays in tree hollow availability and the persistence of arboreal mammals;
- Examine the role that patch size, vegetation age diversity, fire frequency and proportion of vegetation burnt early vs. late play in small-medium sized mammal abundance and diversity, as well as native vegetation condition. Such analyses may have implications for the emerging Carbon Farming Initiative (CFI).

  The KLC, Native Title and Indigenous ranger groups throughout the LCI project area are currently pursuing options under the CFI (see Approach, as is EcoFire). This program allows for an income stream for landscape-scale fire management and additional employment opportunities for Indigenous Rangers and Traditional Owners, and is based on the highly successful West Arnhem Land Fire Abatement Project – see Russell-Smith et al. 2013). It is possible however, that fire strategies aimed at maximising greenhouse benefits – and hence the value of carbon credits – will not necessarily maximise biodiversity benefits. For instance, a fire regime that minimises carbon emissions (and hence greenhouse gases) may not represent an optimal strategy for small-medium sized mammals in general, or threatened specialist species such as Gouldian finches or purple-crowned fairy wrens (Skroblin and Legge 2012; 2013; Woinarski and Legge 2013). Evidence emerging from long-term monitoring of mammals and fire in Kakadu National Park (M. Lawes, unpublished data) suggests that land managers should carefully consider the relationships between biodiversity and carbon objectives when developing fire strategies throughout the North-Kimberley (see also Andersen et al. 2006; 2012; Bradshaw et al. 2013). DPaW will continue to work with groups in the North-Kimberley to ensure that biodiversity benefits are maximised with the implementation of future greenhouse gas abatement projects.

Fire management

- Time prescribed patch mosaic burning to ensure that small fire scars are maintained (< 100 ha in size; M. Lawes, unpublished data). Distances between burnt and unburnt vegetation should not (on average) exceed 1–1.5 km (see Fig. 15) as most small-medium sized mammals are unable to move over such distances (Woinarski et al. 2005; Yates et al. 2008; Radford 2012);

- Reduce the frequency of fire in inland and southern areas, recognising that some areas will need to be burnt annually (e.g. along parts of the Gibb River Road or around campgrounds) in order to prevent catastrophic loss from late season fires;
- Be aware of very old growth vegetation (> 7–8 years), for instance in Prince Regent Nature Reserve. Maintain this by breaking it up via small patchy fires in the wet season; and
- Increase the extent of old-growth vegetation and aim for no more than 45 % of the LCI area to be burnt in a single year, where late season fires contribute no more than 40 % of the annual fire scar.

Invasive animals

- Undertake feral animal aerial surveys and derive sound population estimates; these can then be used to obtain density estimates that can then be linked to landscape impacts to ascertain acceptable thresholds of impact (if they exist);
- Work with Native Title holders to realign current management zones and priority control zones in line with their Healthy Country Plans;
- Add more cattle pad monitoring sites and instigate billabong (waterhole) monitoring (e.g. turbidity, dissolved oxygen, etc. see Ens et al. 2010) to quantify impacts on critical freshwater habitats during the late dry season;
- Link cattle density estimates with mammal abundance and diversity to ascertain impacts. These can be used to determine acceptable thresholds (if they exist);
- Follow-up on any sightings of pigs and monitor and record signs (e.g. rooting); and
- Establish a project to examine the interaction between cats and dingoes (mesopredator hypothesis; Johnson et al. 2006) or support existing projects where appropriate.

Invasive plants

- Implement CyberTracker technology across the project area for mapping and recording weed occurrences and on-ground management;
- Continue wet season weed control in high priority areas, such as the Mitchell Plateau;
- Take a more strategic approach to weed management, particularly the management of grader grass which has the potential to radically alter fire patterns.

Conclusion

Whilst it is very early days in the LCI program, the collaborative approach to threat management – in particular fire – across the North-Kimberley by DPaW, the Australian Wildlife Conservancy and the Balanggarra, Dambimangari, Uunguu and Wunggurr ranger groups demonstrates that threats to biodiversity can be managed across protected area boundaries. With the recent formal declaration of several IPAs in the North-Kimberley and the recent proposal of a large and contiguous jointly managed National Park (which is currently being negotiated, see http://www.wa.liberal.org.au/article/liberals-create-huge-nw-national-park), more and more of the region is being managed for conservation (Fig. 1). Continued management of threatening processes both within and outside these areas is essential if biodiversity losses are to be averted. Catastrophic declines within many existing protected areas throughout northern Australia (e.g. Kakadu National Park; Woinarksi et al. 2001; 2010) indicate the importance of management across protected area boundaries (Woinarski et al. 2011). Moreover, recent work by AWC is demonstrating that good fire management without good feral herbivore management may have limited benefits for biodiversity (S. Legge, unpublished data). A cost-effectiveness analysis for prioritising threat management in the Kimberley by Carwardine et al. (2011; 2012) identified that actions to manage feral herbivores and fire would deliver improvements in probabilities of persistence for biodiversity. Monitoring and evaluation of LCI activities to date demonstrates encouraging trends in this direction.

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