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Research article

The cost of not acting: Delaying invasive grass management increases costs and threatens assets in a national park, northern Australia

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ABSTRACT

Globally, invasive grasses are a major threat to protected areas (PAs) due to their ability to alter community structure and function, reduce biodiversity, and alter fire regimes. However, there is often a mismatch between the threat posed by invasive grasses and the management response. We document a case study of the spread and management of the ecosystem-transforming invasive grass, *Andropogon gayanus* Kunth. (gamba grass), in Litchfield National Park; an iconic PA in northern Australia that contains significant natural, cultural and social values. We undertook helicopter-based surveys of *A. gayanus* across 143,931 ha of Litchfield National Park in 2014 and 2021–2022. We used these data to parametrise a spatially-explicit spread model, interfaced with a management simulation model to predict 10-year patterns of spread, and associated management costs, under three scenarios. Our survey showed that between 2014 and 2021–22 *A. gayanus* spread by 9463 ha, and 47%. The gross *A. gayanus* infestation covered 29,713 ha of the total survey area, making it the largest national park infestation in Australia. *A. gayanus* had not been locally eradicated within the Park's small existing 'gamba grass eradication zone', and instead increased by 206 ha over the 7-year timeframe. Our modelled scenarios predict that without active management *A. gayanus* will continue spreading, covering 42,388 ha of Litchfield within a decade. Alternative scenarios predict that: (i) eradicating *A. gayanus* in the small existing eradication zone would likely protect 18% of visitor sites, and cost ~AU$825,000 over 5 years – more than double the original predicted cost in 2014; or (ii) eradicating *A. gayanus* in a much larger eradication zone would likely protect 86% of visitor sites and several species of conservation significance, and cost ~AU$6.6 million over 5 years. Totally eradicating *A. gayanus* from the Park is no longer viable due to substantial spread since 2014. Our study demonstrates the value of systematic landscape-scale surveys and costed management scenarios to enable assessment and prioritisation of weed management. It also demonstrates the increased environmental and financial costs of delaying invasive grass management, and the serious threat *A. gayanus* poses to PAs across northern Australia.

1. Introduction

Invasive alien species are a growing challenge for protected area (PA) managers worldwide (Foxcroft et al., 2013a, 2017; Pyšek et al., 2020), and are acknowledged as one of the five principal drivers of global biodiversity loss in the UN Convention on Biological Diversity (CBD), Kunming-Montreal Global Biodiversity Framework (CBD, 2022). Managing high-biomass invasive grasses is a particular challenge for PA managers due to the broad scale at which these invasions often occur, and their ability to radically transform ecosystems and fire regimes (Setterfield et al., 2010; Fusco et al., 2019; Kerns et al., 2020). Globally invasive grasses are recognised as some of the most important ecosystem-transforming species (Gaertner et al., 2014). Despite the serious threat that invasive grasses pose to PAs, there is often insufficient resourcing, and staffing to implement effective weed management and monitoring strategies (Read et al., 2020; Craigie and Pressey, 2022; Cuthbert et al., 2022). Many PAs experience budget shortfalls to achieve their diverse management objectives, and often this leads to managers

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having to prioritise actions related to visitors' experience, over actions related to biodiversity management (Craigie and Pressey, 2022).

Australia's tropical savannas are impacted by a suite of highly competitive invasive grasses (van Klinken and Friedel, 2017; Setterfield et al., 2018; Creswell et al., 2021). These tropical savannas cover ~2 million km^2, approximately a quarter of the Australian continent, and remain largely intact (Hutley and Setterfield, 2019); however, invasive grasses increasingly threaten these ecologically diverse, and culturally important landscapes (Setterfield et al., 2018; Read et al., 2020; Bergstrom et al., 2021). Government-supported programmes to increase agricultural productivity in northern Australia historically imported and tested hundreds of potential pasture plants (Cook and Dias, 2006). The vast majority of these grasses have since become weeds in the region (Lonsdale, 1994; van Klinken and Friedel, 2017). This includes *Andropogon gayanus* Kunth. (gamba grass), *Urochloa mutica* (Forssk.) T. Q. Nguyen (para grass), *Hymenachne amplexicaulis* (Rudge) Nees (olive hymenachne), *Cenchrus polystachios* (L.) Morrone (perennial mission grass) and *Cenchrus pedicellatus* (Trin.) Morrone (annual mission grass) (Creswell et al., 2021).

Reflecting the national importance and scale of their environmental impact, in 2009 this group of invasive grasses were formally declared a Key Threatening Process (KTP) under Australia's *Environment Protection and Biodiversity Conservation (EPBC) Act* 1999 (TSSC, 2009). This was followed by a national Threat Abatement Plan (TAP) outlining actions required to reduce this threatening process (Australian Government, 2012), and a range of national and State/Territory weed declarations and weed management strategies, acknowledging the seriousness of this biodiversity conservation threat (See Setterfield et al., 2018). *A. gayanus* is of the greatest concern to PA managers in northern Australia, due to its invasiveness and the very high intensity wildfires it fuels, and the damage it causes to natural values (Setterfield et al., 2013, 2018).

A. gayanus could potentially spread across much of Australia's tropical savanna region (Adams and Setterfield, 2013) and it is already established in locations across northern Australia (in the states of Queensland, the Northern Territory and Western Australia). There has been limited landscape-scale mapping of *A. gayanus* since a national assessment was undertaken in 2012 (March et al., 2013). Therefore, the extent and potential management cost to mitigate *A. gayanus* risk in PAs is likely to be greatly underestimated. To address this, we mapped changes in *A. gayanus* extent and density over a ~7-year period, and modelled the potential cost-effectiveness of management scenarios to protect key natural and visitor values in Litchfield National Park (hereafter referred to as Litchfield or the Park). Litchfield is one of the most visited national parks in the Northern Territory (NTG, 2017), recognised as a 'Class 1' Park (most important) due to biodiversity values of international and national significance, including threatened species (Ferdinands et al., 2016).

Litchfield is one of the few areas in northern Australia where repeated, spatially-continuous aerial surveys of *A. gayanus* extent has been undertaken. It therefore provides a valuable insight into change that may occur across the broader region. The data from a survey in 2014 was used to propose a range of management scenarios to protect park values, published in Adams and Setterfield (2016), and was presented to park managers. In the current study, we present results from a survey of *A. gayanus* distribution undertaken in 2021–22. This dataset was used to: (i) determine the change in *A. gayanus* extent and density between 2014 and 2021–22, (ii) model the predicted distribution of *A. gayanus* over the next decade, (iii) compare three potential management scenarios and evaluate their effectiveness to protect key Park assets and estimate associated management costs to 2031–32, and (iv) compare the difference in the cost of commencing one of these scenarios in 2014 versus 2021–22, to determine the additional costs of eradication incurred through delayed action. Although we recognise there are significant cultural values throughout the Park, assessing the impacts of *A. gayanus* on these values was beyond the scope of our study. However, this should be taken into account in future planning scenarios (e.g Adams et al., 2018). Our intention was to provide tools that park managers could then use to develop a range of cost-effective management scenarios across the Park, taking in account natural, cultural and social assets.

2. Materials and methods

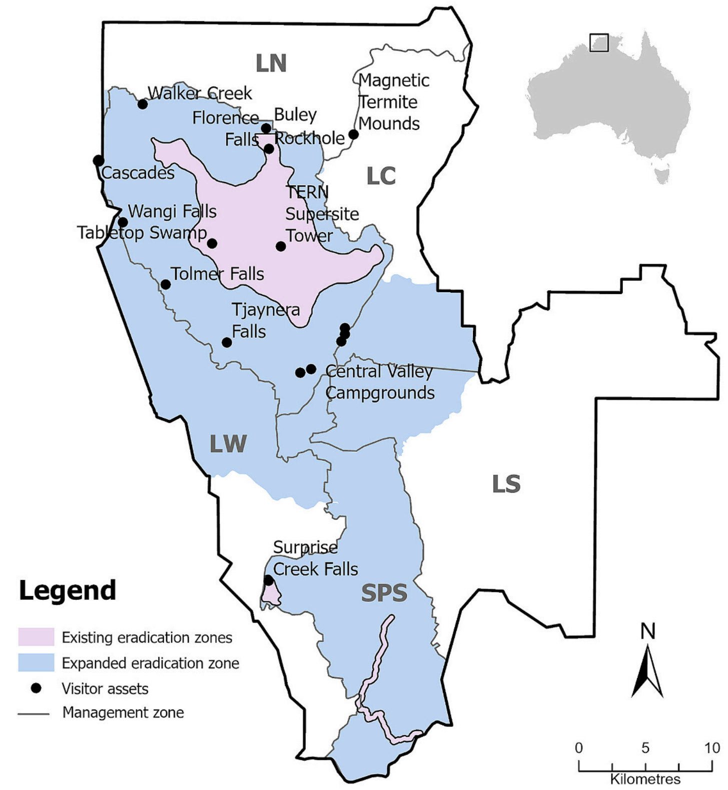
2.1. Study area and study species

Litchfield National Park is located 120 km south of Darwin, Northern Territory, Australia and covers 145,600 ha (NTG, 2017, Fig. 1). Litchfield contains several sites of national and international conservation significance (NTG, 2017), as well as sites of cultural and heritage significance, and is an iconic tourist attraction (Fig. 1; Ferdinands et al., 2016). The Park consists of an escarpment and sandstone plateau running along the western boundary of the Park, with a vast expanse of lowland savanna (Bowman et al., 2001). The region is within the wet-dry tropics with high temperature throughout the year (monthly mean maximum 33.8 °C), and highly seasonal rainfall (mean 1514.9 mm per annum) concentrated in the wet season (October–April). The sandstone plateau and lowlands of Litchfield are dominated by savanna woodlands, interspersed with patches of other high value vegetation types, including open forest, upland swamplands, upland sedgelands, wet heath, dry heath, alluvial grassland, *Melaleuca* spp. dominated woodlands, and riparian vegetation (Truman and Cuff, 2014; See Fig. S1).

Andropogon gayanus (gamba grass) has rapidly invaded extensive areas of Litchfield, replacing the diverse, shorter statute native grass communities (~0.5 m) with dense swards up to 4 m high (Adams and Setterfield, 2016). *A. gayanus* was introduced from tropical Africa, and the cultivar 'Kent' was bred, and commercially released as an introduced pasture grass in the Northern Territory in 1978 (Setterfield et al., 2018). When Litchfield National Park was initially established in 1991, the management plan reported no *A. gayanus* within the Park (Petty et al., 2012), although noted it as a threat from outside the Park boundary (NTG, 1992). For the next 17 years, records of *A. gayanus* were from roadside or other *ad hoc* data collection, with the first systematic aerial survey undertaken in 2008–9, and a second survey in 2014. These surveys (2008–9 and 2014) highlighted *A. gayanus* had invaded areas well beyond roadsides, and into a range of vegetation types, including habitats of high conservation value on the sandstone plateaus of Tabletop Range (Ferdinands et al., 2016). *A. gayanus* high rates of spread are due to its high seed production (~70,000 seeds m^{-2}; Flores et al., 2005) and ability to establish in a wide range of habitats (Setterfield et al., 2004; Petty et al., 2012). *A. gayanus* spreads via natural means (wind, water, and animals). Riparian corridors are critical pathways of spread, facilitating the movement of *A. gayanus* propagules and providing suitable conditions for establishment (Petty et al., 2012).

2.2. Specified management goals for *A. gayanus*

The 2014 distribution data, and the management scenarios presented to park managers/planners directly supported the Park's planning and prioritisation of *A. gayanus* control. This was carried out via a conservation action planning (CAP) process, and development of an Integrated Conservation Strategy (ICS) (See Ferdinands et al. (2016) for comprehensive planning and management elements of the Litchfield ICS). The ICS process identified fire and invasive grasses as priority threats to the values of Litchfield, with wildfire and *A. gayanus* ranked as the top two threats, with an 'extreme' threat rating (Ferdinands et al., 2016). The Litchfield ICS process identified management targets and threat reduction targets across six management zones in the Park (NTG, 2015). We use these six zones within our study (Fig. 1): Sandstone Plateau North (SPN), Sandstone Plateau South (SPS), and the Lowland North (LN), Lowland Central (LC), Lowland South (LS) and Lowland West (LW). The ICS plan defined a 10,525 ha *A. gayanus* (gamba grass) 'eradication



a) top of the waterfall on the sandstone plateau

b) magnetic termite mounds in the lowlands

c) *A. gayanus* infestation in lowlands

d) *A. gayanus* infestation in lowlands

Fig. 1. Litchfield National Park, Northern Territory (Australia) showing the location of high visitation tourist sites in the park; the existing *A. gayanus* eradication zone (scenario 2) and the expanded *A. gayanus* eradication zone (scenario 3); as well as the six Park management zones (Sandstone Plateau North (SPN), Sandstone Plateau South (SPS), and the Lowland North (LN), Lowland Central (LC), Lowland South (LS) and Lowland West (LW)). Photos show (a) Swimming hole in the SPN zone, (b) Magnetic termite mounds in LC zone, (c) Dense *A. gayanus* infestation (>50% cover) in the LC zone and (d) Dense *A. gayanus* infestation (>50% cover) in the LC zone, with *A. gayanus* (green understory throughout the photo) in foreground and brown senescent native grass understory in upper left of photo, with the SPN in the background (Litchfield, April 2021). Photos are by Patch Clapp (a, b) and Natalie Rossiter-Rachor (c,d).

zone' which incorporates the high visitation 'Tabletop Range' area in the SPN zone, and two small areas in the SPS management zone (Fig. 1) The Park-wide 2020 target was "All known infestations of gamba grass are eradicated within the eradication zone" (NTG, 2015). Actions to achieve the Litchfield ICS goals are supported by the Park's overarching statutory Plan of Management (NTG, 2017), and annual weed and fire action plans (Ferdinands et al., 2016). However, following the 2014 survey (Adams and Setterfield, 2016) and the 2016 ICS (Ferdinands et al., 2016) there was no substantial additional funding for intensive *A. gayanus* management, or implementation of a significant control program.

The *A. gayanus* management techniques used in the Park are consistent with those typically used in the region, but are dependent on adequate annual funding and resourcing. *A. gayanus* has a 1–2 year seed longevity (Flores et al., 2005), so can be successfully managed in three years if seeding is prevented (NTG, 2020). Chemical control is primary method of treatment, and is carried out via on-ground application of glyphosate-based chemical during the growing season. Chemical is sprayed over the entire tussock before seed fall (May), and high rates of mortality are generally achieved (NTG, 2020). Follow up inspections are required before the end of the following wet season, and the following year, to control seedlings that may have germinated. Herbicide application is generally via on-ground control methods (Quickspray units on 4WD or all-terrain vehicles) and knapsack sprayers are used for inaccessible infestations (NTG, 2020). Historically the use of aerial application of herbicide has been limited, although this technique could be used to treat dense infestations in inaccessible locations, in order to protect high-priority areas.

2.3. *Aerial survey of A. gayanus distribution 2021–22*

A visual aerial survey of *A. gayanus* extent (occurrence) and density (percentage cover) was undertaken for ~99% of the Park (143,931 ha) and was completed over three dates (29–30 April and 14 May 2021, and 27–29 April 2022). Surveys occurred during the early dry season when native grasses have senesced, whereas *A. gayanus* remains green, tall and clearly visible (Petty et al., 2012; Shendryk et al., 2020). The same low-level helicopter survey method and the senior observer (a staff member of the Northern Territory Government) were used during the 2014 and current surveys to allow comparison over time (for full methods see Petty et al., 2012; McMaster et al., 2014; Adams and Setterfield, 2015, 2016). In brief, the helicopter flew along predetermined transects spaced 500 m apart. *A. gayanus* percentage cover was estimated by two observers, sitting on either side of the helicopter, recording *A. gayanus* cover over adjacent grid areas of 250 × 250 m (~6.25 ha; hereafter, 'grid cell'). An audible alarm sounded every 8 s to notify observers when the helicopter had travelled 250 m. Observers then recorded *A. gayanus* percentage cover in each grid cell on a five-point cover scale (1: no *A. gayanus* present, 2: <1% cover, 3: 1–10% cover, 4: 10–50% cover, 5: >50% cover) following the weed cover classes used for survey and mapping Australian Weeds of National Significance (WoNS; McNaught et al., 2008). The survey resulted in consecutive grid cells of cover data along the survey lines. The 2021–22 survey did not cover a small section (~1%) of the Park between Cascades and Wangi Falls due to a technical issue with survey equipment, so this area was excluded from our analysis.

The survey data from 2014 and 2021–22 was aligned to a 250 m × 250 m grid for analysis. There was generally good agreement between point observations and grid cells, with 68% (in 2014) and 81% (in 2021-2) of grid cells containing either a single observation or multiple observations, in which case the highest value was used. Where no observations occurred in a grid cell the nearest observation within either a 250 m or 500 m buffer was applied. Less than 1% of cells had no point observation within 500 m and these cells were assigned 'no *A. gayanus*'. The number of grid cells in each cover class in 2014 and 2021–22 was then determined.

To assess change in *A. gayanus* extent (occurrence) and density (percentage cover) between 2014 and 2021–22, and the implications for park management, two estimates of infestation size were calculated, gross and net infestation area, following the terminology of Rejmánek and Pitcairn (2002) and Panetta and Timmins (2004). Gross infestation is the area over which the weed is distributed, and was calculated as the summed area of all the grid cells containing *A. gayanus* within a defined area. Net infestation is the smaller area, to which weed control treatment is actually applied, and was estimated by multiplying the number of grid cells counts in each cover class by the mid-point values of the cover class (1: 0.0%, 2: 0.05%, 3: 5%, 4: 30%, and 5: 75%). Gross and net infestation area were determined for: (i) the total survey area, (ii) the existing and expanded *A. gayanus* eradication zones, and (iii) the Park management zones (SPN, SPS, LN, LC, LS and LW). Data were pre-processed using the Geopandas package in Python (Jordahl, 2014) and maps produced in ArcGIS Pro v.2.9.0.

2.4. *Management scenarios*

We used a weed spread model developed for invasive grasses (Adams and Setterfield, 2015) interfaced with a management simulation model developed for *A. gayanus* (Adams et al., 2018) to compare three management scenarios:

(1) 'no active management'; this provided a prediction of the pattern of *A. gayanus* spread in the absence of active management and therefore a baseline to compare with other scenarios,
(2) 'existing eradication zone' modelled the spread of *A. gayanus* and cost of intensive *A. gayanus* management to achieve the aim of localised eradication (sometimes referred to as 'extirpation', sensu Panetta, 2007) within the existing defined 10,525 ha 'eradication zone' as defined in the Park ICS (Fig. 1) with no active management elsewhere in the Park.
(3) 'expanded eradication zone' modelled the spread of *A. gayanus* and cost of intensive *A. gayanus* management to achieve the aim of localised eradication within a larger 74,331 ha eradication zone (Fig. 1) which would is likely to protect the majority of the visitor assets in the Park. We defined this area using the existing delineated Litchfield ICS park management zones because these align with natural features such as creeks or tracks that are important for weed management. We also sought advice from Park Rangers and Park planners. No other active management elsewhere in the Park was included in this scenario.

2.5. *Description of spread and management models*

2.5.1. *A. gayanus spread model*

To predict the pattern of *A. gayanus* spread, we used the Adams and Setterfield (2015) weed spread model, with improved predictions of distance and spread rate. This model is a spatially explicit, individual-based spread model that uses a comparison of regional distribution in multiple time-steps, together with habitat suitability variables and basic population data to predict patterns of spread over defined time periods (see Adams et al. (2015, 2018); Adams and Setterfield (2016) for previous application; Kool (2018) for software; and Adams et al. (2018) for methodological details). The 2014 survey data was used to develop the model, and the 2021–22 aerial survey provided additional data to validate the predicted distance and rate of spread from that used previously (e.g. Adams and Setterfield, 2016), particularly for the SPS, LS and LW zones in which propagule dispersal is primarily through natural means (wind, water and animals) because there is no strong human influence (either as vectors of spread via seed movement or as managers controlling spread). We re-calibrated the Adams and Setterfield (2016) model, based on the natural *A. gayanus* spread in the Park between 2014 and 2021–22 (See Supplementary methods for details). The re-calibrated model has an improved fit (Kappa of 0.729) from the original parameterisation (Kappa of 0.436, Adams et al., 2015).

2.5.2. *Management simulation model*

To estimate the resources required to achieve the weed management goals, the spread model was interfaced with a management simulation model (Adams et al., 2018). The dynamic management model applied action based on our specified management maps (Fig. 1) to grid cells, depending on a set of rules defined by the weed management goal for the zone in which the grid cell occurs. In our study, the grid cells occurred either: (i) inside *A. gayanus* eradication zone, in which case the model simulated management to achieve local eradication, or (ii) outside the *A. gayanus* eradication zone, in which case no management is undertaken. Within the eradication zone, the infestations are reduced to zero over a period of time dependent on the initial cover, with infestations locally eradicated by 5 years, based on best practice weed management. Outside the eradication zones, the model applies the default parameters for natural spread to all grid cells (Adams et al., 2018). Each year the model runs, it simulates an aerial survey that detects new infestations and then allocates management based on the specified spatially-explicit management scenario. Once *A. gayanus* is eradicated from grid cells within the eradication zone, they are placed within a maintenance programme and there is immediate deployment of on-ground control to any new infestations detected within the zone. It is worth noting that within the modelling approach used the spread model and monitoring model are stochastic, with levels of variability in spread rate, location and detection; however the management model is deterministic. Due to variation in spread distance, spread rate, and infestation detection parameters, we ran each scenario 100 times for 10 years. We assumed that an aerial survey of the eradication zones would be undertaken every year and any newly detected *A. gayanus* infestations within the eradication zone would be immediately treated (full model and tool details presented in Adams et al., 2018 and Adams and Setterfield, 2016).

2.5.3. *Modelled management costs*

Ground control costs for the management scenarios were determined using the eradication cost model presented in Adams and Setterfield (2013). The management costs (2021 AU$) per 6.25 ha were determined in consultation with professional weed managers, based on best practice technique, tested with expert estimates (Adams and Setterfield, 2013) and found to be consistent for control of other grassy weeds – namely *Urochloa humidicola* (McMaster et al., 2014). The scenario model predicts the *A. gayanus* cover in a grid cell, based on the standard cover classes (1: no *A. gayanus* present, 2: <1% cover, 3: 1–10% cover, 4: 10–50% cover, 5: >50% cover; McNaught et al., 2008). This then forms the basis for a management category that reflects the ground control required to manage this cover of *A. gayanus*: scattered (1–10% cover), medium (10–50% cover) or dense (>50% cover) (See McNaught et al., 2008 for cover classes, and Adams and Setterfield, 2013 for previous application). For each management category (scattered, medium, or dense) a per grid cell cost is allocated, based on the published data in Adams and Setterfield (2013). The cost model assumes each grid cell containing *A. gayanus* were treated independently, ignoring potential economies of scale. The cost outputs also include proportionate total labour hours (as the largest component of costs) for treating a grid cell. All costs are in 2021 Australian Dollars (AU$). To reflect increases in labour and other operating costs from our initial cost estimates in 2008–9 we used a 30% increase to reflect estimated Consumer Price Index between 2008–9 and 2021–22 (Reserve Bank of Australia, 2022). The per grid cell costs by *A. gayanus* management category (scattered, medium, or dense) used to calculate costs per year in the scenarios are presented in Table S1. The model reported total per year cost for the management scenarios and total labour hours across all treated grid cells; with all management to be carried out by professional weed control contractors, including the annual aerial survey of the eradication zone/s. We report average costs across 100 runs of the model and standard deviation. For maps we present an indicative run out of the 100 model runs.

2.5.4. *The cost of not acting– comparing eradication costs for scenario 2 across time periods*

We compared the cost of achieving scenario 2 (eradicating *A. gayanus* in the existing eradication zone) at two times: in 2014 and 2021–22. We used the *A. gayanus* management simulation model to estimate and compare the cost of local eradication using the 2014 survey data as a start point, and the 2021–22 survey data as a start point. The cost difference provides an estimate of the additional costs incurred due to the 7-year delay in commencement of the intensive management action.

2.5.5. *Risk to visitor infrastructure assets*

We assessed the visitor infrastructure likely impacted by *A. gayanus* spread in 10 years under the three management scenarios, with visitor assets *within* an eradication zone considered protected due to ground control of *A. gayanus*, whereas assets *outside* the zone were potentially vulnerable to *A. gayanus* fuelled wildfires. High intensity, fast moving *A. gayanus* wildfires could result in the loss of infrastructure at visitor sites (e.g. campground and picnic area infrastructure, board walks near swimming areas, and signage), and presents a potential risk to visitors at these sites.

3. Results

3.1. *Current A. gayanus extent and density*

A. gayanus substantially spread in the Park between 2014 and 2021–22 (Fig. 2, Table 1). The gross infestation area was 20,250 ha, or 14.1% of the survey area in 2014; increasing to 29,713 ha, or 20.6% of the total survey area by 2021–22 (Fig. 2, Table 1). *A. gayanus* spread within the existing eradication zone, with the gross area increasing from 388 ha in 2014, to 594 in 2021–22, a 53% increase (Fig. 2, Table 2). *A. gayanus* also spread in all six park management zones, with the gross area increasing by up to 133% in some management zones (Fig. 2, Table 2). Our map shows the largest *A. gayanus* infestations were all within Lowland management zones (Fig. 2). The largest *A. gayanus* infestation was in the LC zone, covering a gross area of 12,119 ha, or 70% of that management zone (Fig. 2, Table 2).

The density of existing *A. gayanus* infestations increased in the survey area between 2014 and 2021–22. Grid cells that contained *A. gayanus* in 2014, generally experienced spread and an increase in *A. gayanus* cover within the cells, resulting in a higher cover class score when re-surveyed in 2021–22. This change was apparent at a landscape scale (See Fig. S2), with mapped grid cells generally moving up the cover scale by 1–2 cover classes over 7 years, but 10% of grid cells moved up 3–4 cover classes (Fig. 3). This increase was particularly evident at the higher end of the cover scale, with the number of class 5 cells (>50% cover) more than doubling in 7 years (Fig. 3, Fig. S3, Table 1). This resulted in a 148% increase in the net infestation area in the total survey area, increasing from 2196 ha in 2014, to 5454 ha in 2021–22 (Table 2). The net infestation in the existing eradication zone increased by 306%, increasing from 6 ha in 2014, to 23 ha in 20121–22 (Figs. 2, 3, Table 2). The net infestation in the six park management zones increased between 81% and 646% over the 7-year timeframe (Fig. 2, Table 2).

A. gayanus infestations were found in a range of vegetation types in the Lowlands and Sandstone Plateaus in the 2021–22 survey (Fig. 2, Fig. S4). The largest infestations were all within Lowlands management zones, and the *A. gayanus* spread over the 7 years was particularly evident in the riparian corridors in these zones (Fig. 2, Fig. S2). While the largest infestations were generally in vegetation types dominated by Eucalypt woodland, *A. gayanus* was also found within fire sensitive vegetation types including dry heath on the Sandstone Plateaus (Fig. 2, Fig. S4). Numerous small *A. gayanus* infestations were recorded within 50 m of stands of fire sensitive vegetation including stands of Northern Cypress pine (*Callitris intratropica*) and monsoon rainforest patches (See photos in Fig. S2).

3.2. *Modelled predicted spread of A. gayanus in three management scenarios*

A. gayanus extent is predicted to substantially increase under all three management scenarios. A detailed summary of each scenario and the associated benefits and costs are provided below.

3.2.1. *Management scenario 1 – no active management*

Under scenario 1, the absence of active management, the gross infestation of *A. gayanus* is predicted to increase from 29,713 ha to 42,388 ha in 10 years (20.6%–29.4% of total survey area respectively; Fig. 4a; Table 3). This was largely driven by expansion within the four Lowland zones (LN, LC, LS and LW), and new infestations spreading into highly suitable habitat such as riparian corridors.

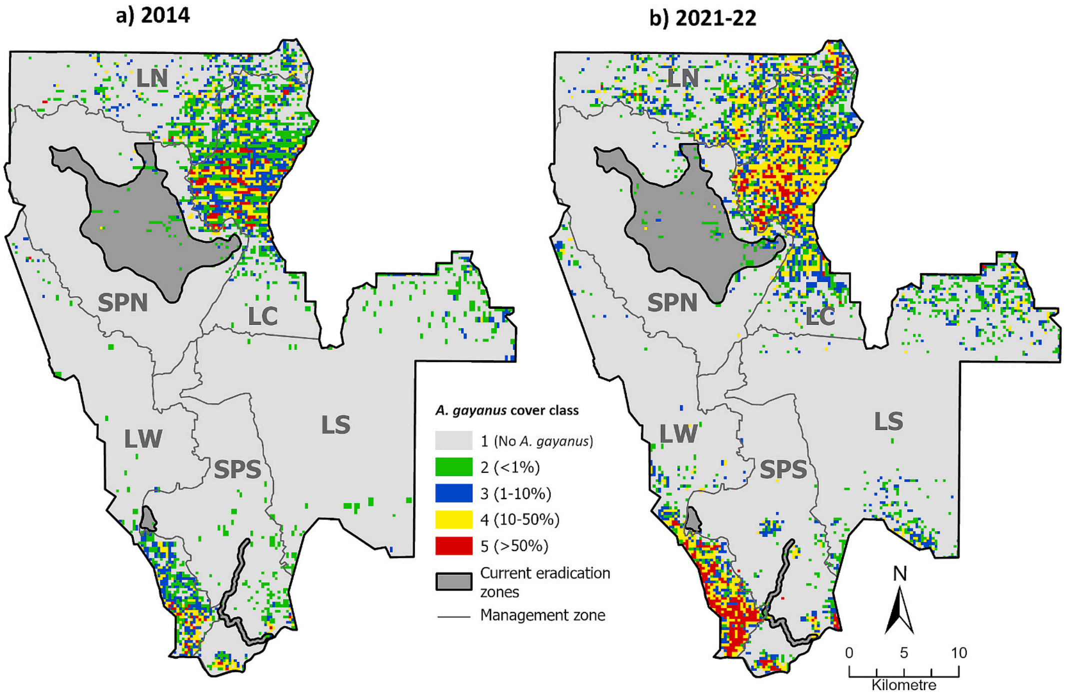


Fig. 2. Aerial survey map of Litchfield National Park (Australia) completed in (a) 2014 and (b) 2021–22 showing *A. gayanus* cover in grid cells (250 × 250 m) in five *A. gayanus* cover classes: grey = 1: no *A. gayanus* present, green = 2: <1% cover, blue = 3: 1–10% cover, yellow = 4: 10–50% cover, red = 5: >50% cover. Park management zones illustrated as base colour on the map (Sandstone Plateau North (SPN), Sandstone Plateau South (SPS), and the Lowland North (LN), Lowland Central (LC), Lowland South (LS) and Lowland West (LW)). The eradication zone implemented after the 2014 survey is outlined in black. See Table 2 for supporting data.

Table 1

Comparison of *A. gayanus* occurrence, percentage cover and infestation size based on aerial survey in 2014 and 2021–22, covering 143,931 ha of Litchfield National Park. Results from the full survey area showing: (i) the number of grid cells in each cover class, with the percentage of surveyed grid cells in that cover class in brackets (ii) the percentage of surveyed grid cells containing *A. gayanus*, (iii) the gross infestation area (ha)[a], with the percentage of the *A. gayanus* infestation in that cover class in brackets, and (iv) the net infestation area (ha)[a], with the percentage of the *A. gayanus* infestation in that cover class in brackets.

Cover class and *A. gayanus* cover	Number of grid cells in cover class		% of surveyed grid cells containing *A. gayanus*		*A. gayanus* infestation area			
					Gross hectares		Net hectares	
	2014	2021–22	2014	2021–22	2014	2021–22	2014	2021–22
1: 0% no *A. gayanus* present	19789 (86%)	18275 (79%)	–	–	–	–	–	–
2: <1% cover	1542 (7%)	1339 (6%)	6.7%	5.8%	9638 (48%)	8369 (28%)	48 (2.2%)	42 (0.8%)
3: 1–10% cover	978 (4%)	1489 (6%)	4.2%	6.5%	6113 (30%)	9306 (31%)	306 (13.9%)	465 (8.5%)
4: 10–50% cover	545 (2%)	1451 (6%)	2.4%	6.3%	3406 (17%)	9069 (31%)	1022 (46.5%)	2721 (49.9%)
5: >50% cover	175 (1%)	475 (2%)	0.8%	2.1%	1094 (5%)	2969 (10%)	820 (37.4%)	2227 (40.8%)
Total	**23029**	**23029**	**14.1%**	**20.6%**	**20250**	**29713**	**2196**	**5454**

[a] Gross infestation area was the summed area of all the grid cells containing *A. gayanus* within a defined area. The net infestation area was estimated by multiplying the number of grid cells counts in each cover class by the mid-point values of the cover class.

3.2.2. Management scenario 2 – eradication in existing eradication zones

Under scenario 2, intensive *A. gayanus* management within the 'existing eradication zones' is predicted to result in a marginally reduced spread compared to Scenario 1, with the gross infestation predicted to increase from 29,713 ha to 41,231 ha in 10 years (20.6%–28.6% of total survey area respectively; Fig. 4b; Table 3). The reduced spread relative to scenario 1 is due to the eradication/ongoing maintenance of *A. gayanus* from 594 ha by the exiting eradication zone (Table 2), an area that includes four key visitor assets (Fig. 4b; Table S2). This scenario resulted in 4% of avoided gross infestation due to the active management. However, continued *A. gayanus* spread outside of this zone, resulted in an overall 39% increase in the gross infestation in the total surveyed area (Table 3).

Achieving eradication in the existing *A. gayanus* eradication zone is estimated to cost a total of AU$970,000 ± $29,600 over 10 years (Table 4). The initial eradication phase would cost AU $165,000 ± $2500 per year, over a 5-year work program (total of AU $825,000 ± $16,000; Table 4). The total labour hours per year required to achieve this is 830 ± 15 hrs. After eradication is achieved, annual cost during the ongoing maintenance period includes an annual aerial survey (AU$7000 per year for eradication zones and adjacent surrounding area), and 115 ± 30 labour hours for ground spraying (Table 4). The total cost of the maintenance program is AU$29,000 ± $5,400 per year, over a 5-year work program (total of AU$145,000 ± $23,000; Table 4).

3.2.3. Management scenario 3 –eradication in expanded eradication zone

Under scenario 3, focused management effort in the 'expanded eradication zone' is predicted to result in a greater reduced spread compared to Scenario 1, with the gross infestation predicted to increase from 29,713 ha to 32,225 ha in 10 years (20.6% and 23.1% of total survey area respectively; Fig. 4c; Table 3). The reduced spread relative to scenario 1 is due to the eradication/ongoing maintenance of *A.*

Table 2

Comparison of *A. gayanus* gross and net infestation area (ha)[a] based on aerial survey in 2014 and 2021–22, covering 143,931 ha of Litchfield. Results show the *A. gayanus* gross and net infestation area (ha) for the: (i) 'existing eradication zone' (as per the Park's ICS), (ii) the proposed expanded eradication zone (as proposed by this study), and the Park management zones, (iii) Sandstone Plateau North (SPN), (vi) Sandstone Plateau South (SPS), and the (v) Lowland North (LN), (vi) Lowland Central (LC), (vii) Lowland South (LS) and (viii) Lowland West (LW).

Park Management Zone	Zone area (ha)	*A. gayanus* infestation area					
		Gross hectares		Percent change	Net hectares		Percent change
		2014	2021–22	(2014 to 2021–22)	2014	2021–22	(2014 to 2021–22)
Weed management zones							
(i) Existing eradication zone	10525	388	594	53% ↑	6	23	306% ↑
(ii) Expanded eradication zone	74331	451	801	78% ↑	166	533	221% ↑
Park management zones							
(iii) Sandstone Plateau North	31563	856	1381	61% ↑	50	97	94% ↑
(iv) Sandstone Plateau South	20356	1388	1750	26% ↑	100	303	202% ↑
(v) Lowland North	20325	3556	5725	61% ↑	265	783	196% ↑
(vi) Lowland Central	15050	9663	12119	25% ↑	1411	2560	81% ↑
(vii) Lowland South	39419	1788	4169	133% ↑	33	242	646% ↑
(viii) Lowland West	17219	3000	4569	52% ↑	338	1469	335% ↑
TOTAL (iii to viii)	**143931**	**20250**	**29713**	**47% ↑**	**2196**	**5454**	**148% ↑**

[a] Gross infestation area was the summed area of all the grid cells containing *A. gayanus* within a defined area. The net infestation area was estimated by multiplying the number of grid cells counts in each cover class by the mid-point values of the cover class.

gayanus from 801 ha within the expanded eradication zone (Table 2), an area that includes 86% of visitor assets in the Park (Fig. 4c; Table S2). This management scenario resulted in 31% of avoided gross infestation due to the active management. However, again, *A. gayanus* spread outside of this eradication zone, resulted in an overall 12% increase in the gross infestation area in the total survey area (Table 3).

Achieving eradication in the much larger 'expanded *A. gayanus* eradication zone' is estimated to cost a total of AU$7,140,000 ± $64,800 over 10 years (Table 4). The initial *A. gayanus* eradication phase would cost AU$1,317,000 ± $5500 per year, over a 5-year work program (total of AU$6,585,000; Table 4). This equates to 7650 ± 29 labour hours per year. This assumed the majority of herbicide application would be ground-based, however, we note that some of the infestations in this extended zone are quite large and would be appropriate for aerial spraying which could reduce the time and costs of achieving the management outcome. Given the scope of work in this scenario, a full-time coordinator to provide planning support for the weed control contract team is required. Following 5-years of active *A. gayanus* eradication the weed management effort would shift to a 5-year maintenance phase. This includes an annual survey to identify any new *A. gayanus* incursions in the eradication zone and to deploy immediate ground control. Annual cost of this maintenance period includes aerial survey (AU$23,000 per year for eradication zones and adjacent surrounding area) and 467 ± 67 labour hours of ground control spraying (~AU$88,000 per year in labour equipment and chemicals). The total cost of the *A. gayanus* maintenance program is predicted to cost AU$111,000 ± $12,100 per year, over the 5-year program (total of AU$555,000 ± $49,400; Table 4). This maintenance program would have to continue in perpetuity to ensure eradication is maintained in the extended eradication zone.

3.3. The cost of not acting– comparison of eradication costs for scenario 2 based on 2014 and 2021–22 distribution

If intensive control was implemented in the existing eradication zone (scenario 2) in 2014 it would have cost AU$82,000 a year over the 5-year eradication phase (total cost AU$410,000; in 2021 dollars). However, no intensive eradication program was funded or implemented in this zone after the 2014 survey. As a result, *A. gayanus* has spread in the existing eradication zone and increased in cover (Table 2). It was estimated that the eradication phase for this same scenario in 2021–22 would now cost AU$165,000 a year over the 5-year work program (total cost AU$825,000; Table 4); more than double the original cost to achieve eradication.

3.4. Risk to visitor infrastructure assets

Under scenario 1 (no active management), no visitor assets are within the existing eradication zone, therefore all visitor assets are potentially at risk (Table S2). Under scenario 2 (existing eradication zone) *A. gayanus* is eradicated from the 10,525 ha zone. This would likely to protect 18% of visitor assets (including Buley Rockhole, Florence Falls; Fig. S4, Table S2), although 82% of visitor assets would continue to be at risk (including the Central Valley Campgrounds, Fig. S4, Table S2). Under scenario 3 (expanded eradication zone) *A. gayanus* is eradicated from the 74,331 ha zone. This would likely protect 86% of visitor assets (Fig. S4, Table S2). However, even with this level of intensive ground control, 14% of visitor assets would continue to be at risk (including the magnetic termite mounds), with *A. gayanus* infestations (Fig. S4).

4. Discussion

Protected areas (PAs) are an essential tool for protecting biodiversity and functioning ecosystems against increasing threats (Pressey et al., 2015); including those posed by invasive alien plant species (Essl et al., 2020; Pyšek et al., 2020). Well-resourced and effective management of invasive alien plants in PAs is critical to mitigate their impacts (Genovesi and Monaco, 2013; Foxcroft et al., 2017; Cheney et al., 2018) and meet CBD global biodiversity framework targets for the protection of the PA ecosystems (Adams et al., 2019; Leadley et al., 2022). However, globally, studies have highlighted the substantial lack of resources for PAs to effectively manage threats (e.g. Adams et al., 2019; Coad et al., 2019; Craigie and Pressey, 2022; Cuthbert et al., 2022) and the poorer biodiversity outcomes that result from this resourcing shortfall (Adams et al., 2019; Craigie and Pressey, 2022). Here we demonstrate the substantial costs of not acting on the threat of invasive grasses, using a case study of the spread and management of ecosystem-transforming invasive grass, in an iconic PA in northern Australia; one that contains significant natural, cultural and social values. Our research shows that failing to fund and implement strategic weed management can lead to rapid expansion of invasive grass infestations, and cost of management can quickly escalate, as can the threat to high-value PA assets.

Our study demonstrates the importance of systematic landscape-scale surveys to assess the true scope of the problem and assess the risks to assets from alien plant invasion. The repeated surveys showed an increase in the gross infestation area of ~9,500 ha within a ~7-year timeframe across the survey area, and a 148% increase in net infestation area. This dramatic increase in a short timeframe is consistent with the

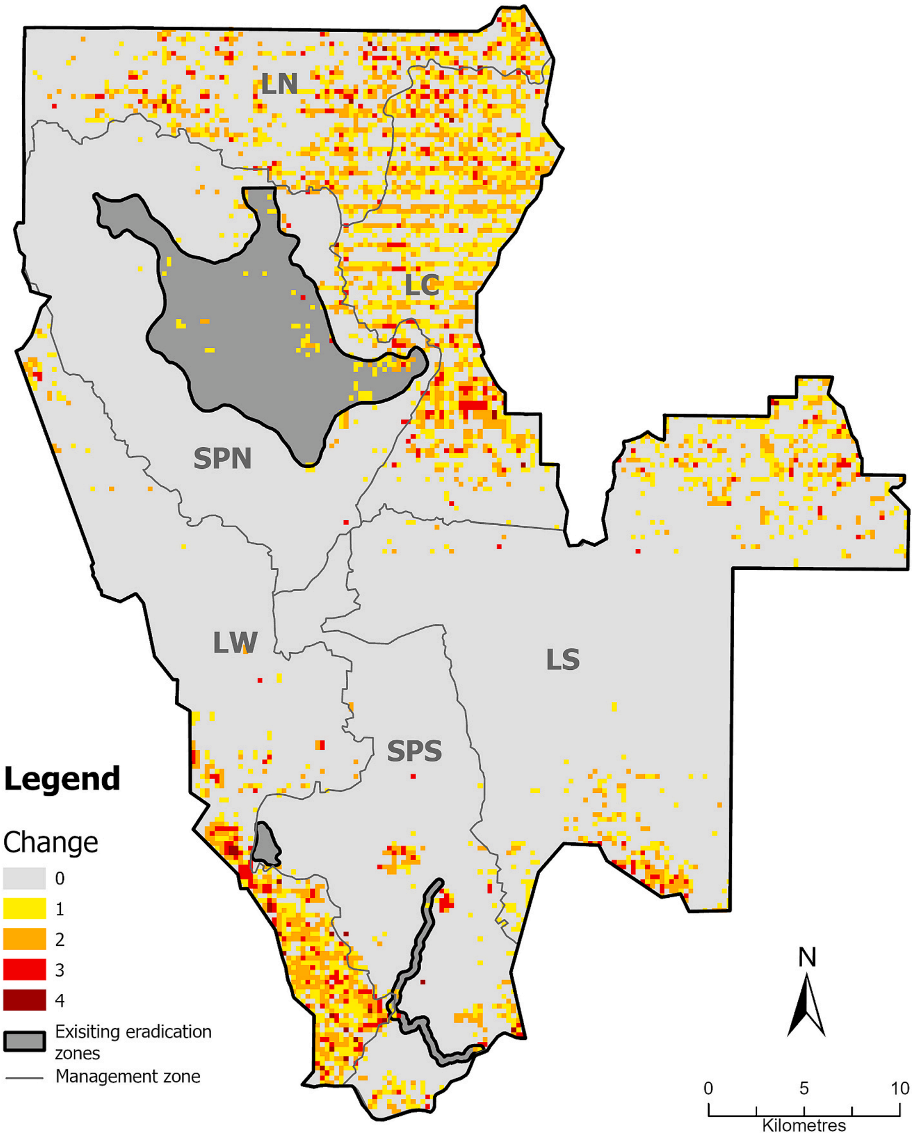


Fig. 3. Map showing increase in *Andropogon gayanus* (gamba grass) cover classes between 2014 and 2021–22 for each mapped grid cell in Litchfield National Park using four categories: grey = no change, yellow = increased by one cover class, orange = increased by two cover class, red = increased by 3 or 4 classes. (Cover classes 1: no *A. gayanus* recorded, 2: <1% cover, 3: 1–10% cover, 4: 10–50% cover, 5: >50% cover). See Tables 2 and 3 for supporting data.

invasion and spread of *A. gayanus* in the Park over two decades (Figs. 5 and 6), and to our knowledge this is the largest *A. gayanus* infestation in a national park in Australia. These results support the growing body of research showing the critical role of systematic, landscape-scale surveys, to enable PA mangers to accurately assess the true extent of alien plant invasions, and as a baseline for evidence-based weed management prioritisation, and monitoring programs (Cheney et al., 2018).

This survey data allowed us to quantify, for the first time, the environmental and financial costs of not acting on invasive grasses in PAs. Comparing the costs of achieving eradication in the small existing eradication zone in 2014 and 2021–22 allowed us to estimate the additional costs that would be incurred to achieve the goal, seven years after the initial management scenario was proposed. The delay has doubled the cost of achieving the Park's goal of local eradication in the high-value existing eradication zone. In addition to increased costs, the rapid expansion of *A. gayanus* highlights the potential impacts on other PA values. *A. gayanus* increased in all management zones in the Park over the last ~7 years, including those containing the highest biodiversity values (Ferdinands et al., 2016). These management zones provide habitat for endemic, vulnerable and threatened fauna species, such as the critically endangered northern quoll, the endangered northern brush-tailed phascogale, and brush-tailed rabbit-rat; as well as threatened species such as the partridge pigeon and black-footed tree-rat; and fire sensitive trees such as *Callitris* (Table 5). The rapid increase in cover of existing *A. gayanus* infestations is of particular concern, with a greater proportion of the Park's *A. gayanus* now in the two highest cover classes. Being a high-biomass grass, it is this cover that drives the transformation of invaded ecosystems (Rossiter et al., 2003; Setterfield et al., 2010). Previous studies have shown the ground layer become almost monocultures of tall grass, with an understorey of reduced biodiversity of 1–2 native plant species (Setterfield et al., 2010). This community dominance drives the documented increases in biomass/fuel loads, fire intensity (Rossiter et al., 2003; Setterfield et al., 2010, 2013) and negative impacts on native savanna ecosystems and ecosystem function (Rossiter et al., 2003; Rossiter-Rachor et al., 2008, 2009; Brooks et al., 2010).

Ecosystem collapse of invaded ecosystems is a potential endpoint if this invasion continues and further degrades native ecosystems. This scenario, where the invaded ecosystem has lost key defining features and functions (Bergstrom et al., 2021), has already been identified as a potential outcome of *A. gayanus* invasion (See Bergstrom et al., 2021

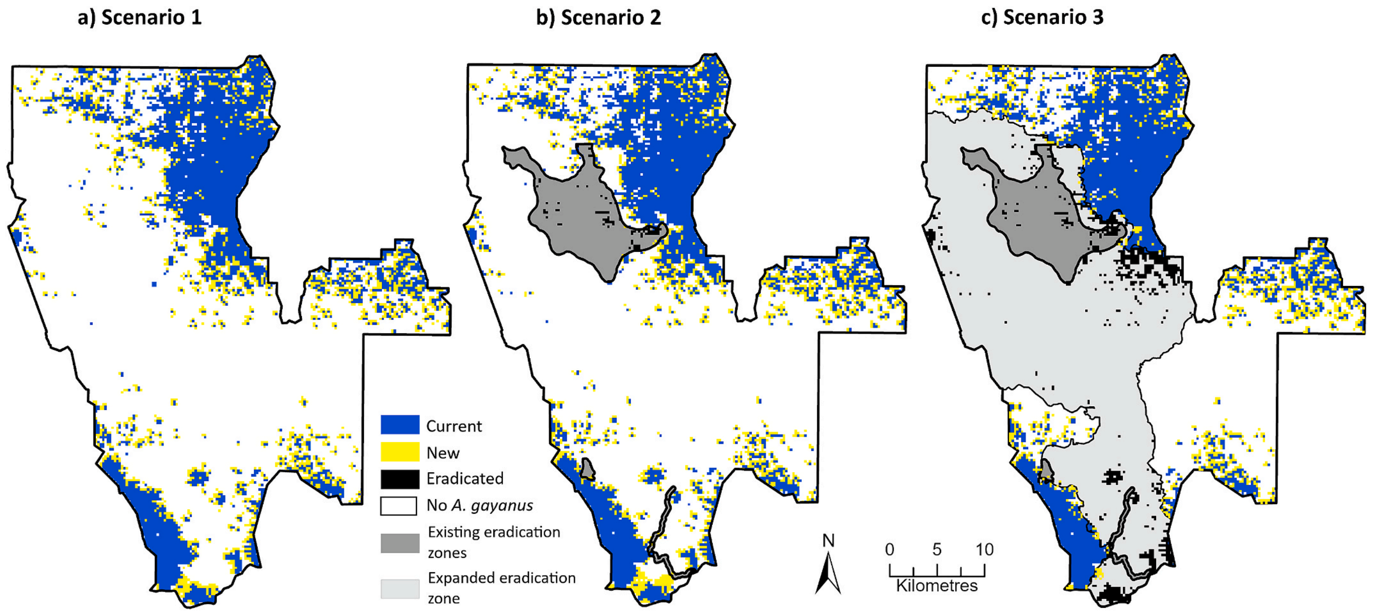


Fig. 4. Modelled *A. gayanus* (gamba grass) distribution in Litchfield National Park (Australia) showing predicted distribution after 10 years of: (a) 'No active management' (scenario 1), (b) active management within '*existing* eradication zone' (scenario 2) and (c) 10 years of management within '*expanded* eradication zone' (scenario 3). Blue = current *A. gayanus* infestations (as of 2021–22 survey), yellow = new infestations after 10 years of management (2031–32), black = eradicated infestation due to active management action.

Table 3

Model outcomes for three *A. gayanus* management scenarios (each run 100 times) for 10 years based on the 2021–22 distribution (Fig. 2b). Results are for the total survey area (143,931 ha) after a 10 year period and show: (i) the predicted percent of surveyed grid cells containing *A. gayanus*, (ii) the predicted gross infestation area (ha), with the percentage increase after 10 years in brackets; (iii) the predicted avoided gross infestation area as a result of management, when compared to scenario 1 (no active management); (iv) the predicted net infestation area, with the predicted percentage increase after 10 years in brackets; and (v) avoided net infestation area due to management (when completed with Scenario 1- no management). Data from the 2014 and 2021–22 surveys provided for comparison.

	2014	2021–22	2031–32		
			Scenario 1	**Scenario 2**	**Scenario 3**
			No active management	Existing eradication zone	Expanded eradication zone
(i) % of surveyed grid cells containing *A. gayanus*	14%	21%	29.4%	28.6%	23.1%
(ii) Gross infestation (ha)[a]	**20250**	**29713**	**42388** (43% ↑)	**41231** (39% ↑)	**33225** (12% ↑)
(iii) Avoided gross infestation (ha) compared to Scenario 1	–	–	–	1157 (4%)	8006 (31%)
(iv) Net infestation (ha)[a]	**2196**	**5454**	**19826** (263% ↑)	**19444** (256% ↑)	**16578** (204% ↑)
(v) Avoided net infestation (ha) compared to Scenario 1	–	–	–	382 (7%)	8006 (60%)

[a] Gross infestation area was the summed area of all the grid cells containing *A. gayanus* within a defined area. The net infestation area was estimated by multiplying the number of grid cells counts in each cover class by the mid-point values of the cover class.

'Abrupt ecosystem collapse profile'- Supplementary data, pp 15–17). Our study demonstrates that *A. gayanus* cover is far more extensive than in 2014. Urgent management action is needed to reduce the spread and cover of *A. gayanus* in the Park reduce the risk of *A. gayanus*-mediated ecosystem collapse.

Increased fire risk and extreme *A. gayanus* fire behaviour could impact tourism. The impact of *A. gayanus* on fire risk is well documented for this region (Setterfield et al., 2010, 2013), and further supports the well-established impact of high-biomass invasive grasses altering fire regimes globally (Brooks et al., 2004; Gaertner et al., 2014; Fusco et al., 2019; Kerns et al., 2020). However, we are unaware of studies that have demonstrated the follow-on impact of these changes in fire regimes on tourism. Recent extreme *A. gayanus* fire behaviour in Litchfield has demonstrated the increasing risk to park visitors: extreme *A. gayanus* wildfires at night resulting in night evacuations of campgrounds (NT Gamba Grass WAC, 2021); day time evacuations and closure of the campgrounds, and the and the full closure of the Park on *A. gayanus*-mediated 'Catastrophic' fire danger days (ABC, 2020). Having to refund tourists for campground closures could impact on the overall NT Parks visitor revenue, estimated at ~AU$16 million for 2021–22 (NTG, 2022). For example, in August 2022 the newly opened Central Valley Campgrounds, which cost AU$17 million (NTG, 2019) were evacuated and closed for a period of time due to the risk posed by a high-intensity *A. gayanus* fuelled wildfire (Supplementary Fig. S5).

Weed management prioritisation tools can support strategic invasive grass weed management. We applied our existing weed management prioritisation tool to evaluate management options, and demonstrate that managers can easily identify alternatives that might better reduce the risk to assets and meet budget constraints. Application of such structured decision making with decision support tools ensures that managers understand the option of each budgetary decision, and that *not* investing in, and increasing weed control activity, is an important decision with consequences for weed spread, fire risk and the loss of values.

Without action, our model predicts that the *A. gayanus* infestation will rapidly worsen over the next decade. Under this scenario, we predict that within 10 years *A. gayanus* cover will increase by almost 13,000 ha, covering ~30% of the surveyed area and impact on all key visitor assets in the Park. We provided two possible alternative scenarios. One scenario in the small existing eradication zone is likely to

Table 4
Modelled management costs (AU$) for *A. gayanus* management scenario 2 (existing eradication zone) and 3 (expanded eradication zone). Costs and labour hours are presented as average (± standard deviation) for: (a) Average annual cost for eradication phase (years 1–5) and the average number of labour hours per year spent on ground control; and (b) Average annual cost for ongoing maintenance phase and average labour hours per year spent on ground control and (c) Total costs. All costs are AU$ 2021 dollars, and (± Standard deviation).

	Scenario 2 Existing eradication zone	Scenario 3 Expanded eradication zone
a) **Eradication costs (Years 1–5)**		
Average annual cost, AU$	$165,000 ($2500)	$1,317,000 ($5500)
Average annual labour, hrs	830 (15)	7650 (29)
b) **Maintenance costs (ongoing)**		
Average annual costs, AU $	$29,000 ($5400)	$111,000 ($12,100)
Average annual labour, hrs	115 (30)	467 (67)
c) **Total costs (AU$)**		
Sum of 5 years eradication	$825,000 ($16,000)	$6,585,000 ($22,300)
Sum of 5 years maintenance	$145,000 ($23,00)	$555,000 ($49,400)
Total for 10 years	**$970,000** ($29,600)	**$7,140,000** ($64,800)

Costs include equipment, herbicide, aerial spraying, and annual aerial survey of the eradication zone, labour (including on-costs), and planning support. All labour is carried out by weed control contractors, including the aerial survey.

protect 18% of key visitor assets in the Park, although invasion will impact many others. We also demonstrated a second, more intensive management scenario which 86% visitor assets in the Park are maintained/protected. However, even in this best case (scenario 3), the PA values of the Park are significantly reduced with ~23% of the surveyed area invaded by *A. gayanus* grass, with the Lowlands become increasingly degraded. This highlights the complexity of managing the Park into the future to achieve its primary intended goal of conservation, and prioritisation and short- and long-term goal setting is required.

Local eradication of *A. gayanus* within the proposed expanded zone is economically feasible, but is likely approaching the upper limits of feasibility of eradication. With a net area of 533 ha, this *A. gayanus* area is within the size range of documented successful weed eradication case studies (net areas ranging between 0.04 and 2480 ha; Parkes and Panetta, 2009). While there has been considerable debate in the literature about the upper limits to the feasibility of weed eradication (see Simberloff, 2013), we believe the local eradication of *A. gayanus* is feasible as: (i) it does not form persistent seedbanks (1–2 year seed longevity; Setterfield et al., 2004; Flores et al., 2005; Bebawi et al., 2018), (ii) it is highly detectable in the dry/control season (Petty et al., 2012; Shendryk et al., 2020); (ii) it can be effectively controlled with relatively inexpensive herbicide (glyphosate; Brooks et al., 2006); (iv) our survey has delimited the current extent of *A. gayanus* in the Park before an intensive local eradication program has commenced and (v) the costed management scenarios and extensive ecological research on *A. gayanus* support optimal management (Setterfield et al., 2018). As a consequence of this research Park Managers have recently secured Australian Government funding to undertake eradication in priority areas of the Park, based on the two active management scenarios presented in this study. The data and costed scenarios have enabled park managers to assess the scale of the current threat to the Park, and prioritise *A. gayanus* control to protect high value assets (J. Vea, Parks & Wildlife Commission NT, pers comm). The local eradication program is planned to commence in 2023. However, local eradication will only be possible with improved weed management governance, guiding evidence-based funding for weed planning, control and monitoring programs.

4.1. Time to act: meeting the challenge of invasive grasses in PAs over the next decade

Litchfield is a case study that exemplifies the enormous challenges and risks posed by invasive grasses to PAs globally, and the need for prioritised management. The rapid spread by this invasive grass emphasises the threat that invasive alien species pose to biodiversity in PAs, and supports the growing literature repeatedly reporting this outcome for PAs globally (Pyšek et al., 2020; Shackleton et al., 2020). It also highlights the need for adequate PA management, if countries are to meet the CBD post-2020 global biodiversity framework targets (Adams et al., 2019; Essl et al., 2020; Leadley et al., 2022). At a time when counties are heavily investing in purchasing new land, and expanding the PA network to meet the required 30% of land protected targets, better resourcing and managing existing PAs first would likely have better outcomes for conserving biodiversity (Adams et al., 2019).

Our results point to three fundamental changes needed to the current approach to invasive grass management in PAs by governments, policymakers, planners and managers if they are to meet this growing challenge over the next decade:

- Firstly, improved invasive grass distribution data is needed to enable evidence-based decision making. Baseline distribution data is essential for assessing the scope of the problem, developing optimal control strategies and accessing resources need (Foxcroft et al., 2013b; Cheney et al., 2018). Put simply, PA managers cannot

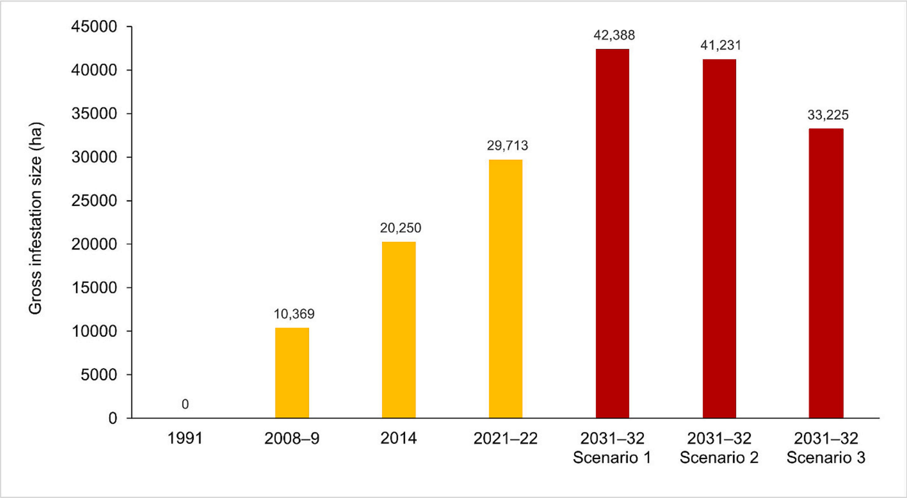


Fig. 5. Thirty-year time series of *A. gayanus* (gamba grass) distribution in Litchfield National Park (Australia), and the predicted distribution of *A. gayanus* over the next decade. Orange = the gross infestation (ha) of *A. gayanus* in 1991, 2008–9, 2014, and 2021–22; red = the predicted gross infestation (ha) of *A. gayanus* by 2031–32 after 10 years of management of one of three management scenarios (a) scenario 1- no active management, (b) scenario 2- eradication in existing eradication zone, (c) scenario 3- eradication in expanded eradication zone. Source: 1991 as stated in the Litchfield Management Plan (NTG, 1992) and aerial surveys in 2008–9, 2014, and 2021–22.

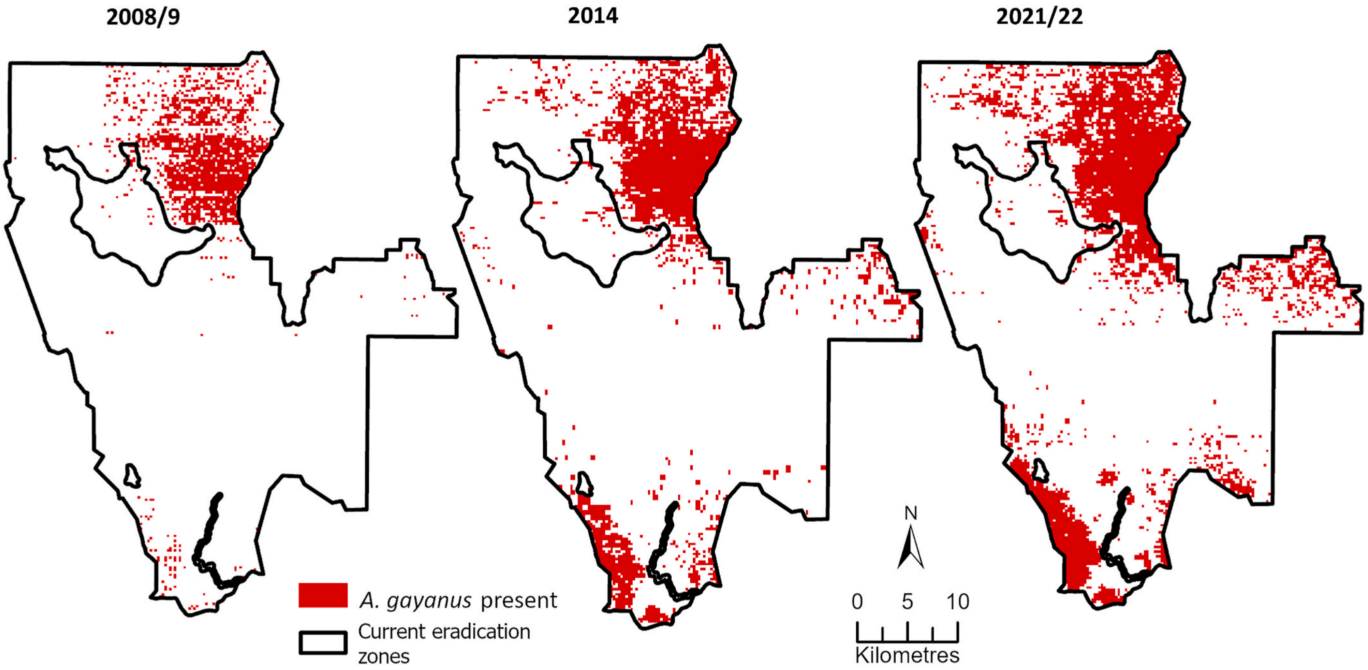


Fig. 6. Time series of the extent of *A. gayanus* (gamba grass) within Litchfield National Park in (a) 2008–9, (b) 2014, and (c) 2021–22. White = no *A. gayanus* recorded, red = *A. gayanus* present. Source: aerial surveys in 2008–9, 2014, and 2021–22.

Table 5
Species of conservation significance found in Litchfield National Park in the Sandstone Plateau and Lowland management zones.

Scientific Name	Common Name	Northern Territory Conservation Status	Australian Conservation Status
Sandstone Plateaus *(These management zones are included in scenario 2 and 3 eradication zone)*			
Conilurus penicillatus	Brush-tailed rabbit-rat	EN	VU
Cycas armstrongii	Cycad	VU	
Dasyurus hallucatus	Northern quoll	CR	EN
Hipposideros inornata	Arnhem leaf-nosed bat	VU	
Petrogale concinna	Nabarlek	VU	
Phascogale pirata	Northern brush-tailed phascogale	EN	VU
Varanus mertensi	Mertens' water monitor	VU	
Lowlands *(These management zones are not included in either the Scenario 2 or 3 eradication zone)*			
Antechinus bellus	Fawn antechinus	EN	
Erythrotriorchis radiatus	Red goshawk	VU	VU
Geophaps smithii	Partridge pigeon	VU	VU
Mesembriomys gouldii	Black-footed tree-rat	VU	
Rattus tunneyi	Pale field-rat	VU	
Varanus panoptes	Floodplain monitor	VU	

CR – Critically Endangered; EN – Endangered; VU – Vulnerable.
Source: Litchfield National Park Plan of Management (NTG, 2017) and Threatened Species of the Northern Territory (NTG https://nt.gov.au/environment/animals/threatened-animals).

adequately manage what they don't adequately know. Invasive grass surveys need to be frequent, landscape-scale, and collected in a way that allows feedback of this spatial data into decision-making frameworks. This includes mapping before- and after-control to enable evaluation of program effectiveness. This is a major gap in many existing invasive grass management programs.

- Secondly, invasive grass control programs need clearly defined goals and outcomes, to ensure meaningful biodiversity outcomes. This includes defining outcomes both in terms of level of threat reduction, and the state of biodiversity. This will enable mangers to better define the specific goals/targets, and the activities and resources required to achieve these outcomes (refer to examples of 'results chains' PA planning tools; Margoluis et al., 2013; Coad et al., 2015; Pressey et al., 2015)
- Thirdly, adequate resourcing needs to be allocated to PAs to enable adequate invasive grass management. This study serves as a cautionary case study of the significant environmental and financial costs of not acting on the threat of invasive grasses. To address this challenge there is an urgent need for more funding for invasive grass management, and that funding needs to directed to strategic planning, control, and monitoring programs in order to better protect PA values in the long term..

5. Conclusions

Globally, there are many well-documented examples of the impact of invasive grasses on PA values, including reduced biodiversity, and alterations to fire regimes. However, there is often a mismatch between the level of threat posed by invasive grasses and the urgency of management response in PAs. Our study, for the first time, quantified the financial costs of not acting early on invasive grasses in PAs, and the costs that could have be avoided by earlier action. Our case study of Litchfield National Park showed that in the absence of well-funded and strategic management, the ecosystem-transforming *A. gayanus* rapidly expanded, more than doubling the cost of achieving the Park's weed management goals and to support development of optimal management and monitoring scenarios, and to enable an understanding of the resources needed. Our modelling predicted that without intensive management, this infestation will continue to rapidly spread over the next decade, and will impact the majority of Park assets. We showed the value of landscape-scale systematic mapping to provide a baseline to assess the scope of invasive species problem, and support development of optimal management and monitoring scenarios, and understand the resources needed. Our study highlights the urgent need for significant investment to address this serious threat of invasive grasses to PAs across northern Australia. It also highlights the potential costs and risks of any further delays.

Credit author statement

Natalie Rossiter-Rachor: Conceptualization; Funding acquisition; Methodology; Investigation; Writing – original draft; Writing – review & editing. **Vanessa Adams**: Conceptualization; Methodology; Software; Writing – original draft; Writing – review & editing. **Caroline Canham**: Data curation; Visualization; Writing – review & editing. **Dan Dixon**: Data curation; Software; Visualization; Writing – review & editing. **Thorsteinn Cameron** Software; Visualization. **Samantha Setterfield**: Conceptualization; Funding acquisition; Methodology; Writing – original draft; Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2022.116785.

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