Monitoring rangeland biodiversity: Plants as indicators

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Abstract As well as being important components of biodiversity in their own right, plants reflect the physical environment, are the primary target of many of the pressures acting on rangelands, and are relatively amenable to measurement. Hence, measurements based on plants have considerable potential to be efficient indicators of the response of rangeland biodiversity to land use. A recent report commissioned by the National Land and Water Resources Audit recommended a core set of 11 indicators, six of which relied on measurements of plants. These were trends in (i) the extent of clearing; (ii) the cover of native perennial ground-layer vegetation; (iii) the distribution and abundance of exotic plant species; (iv) the distribution and abundance of fire-sensitive species; (v) the distribution and abundance of grazing-sensitive species; and (vi) the distribution and abundance of listed threatened entities. Most indicated responses of plants to pressures acting on them. Only two (clearing and exotic plants) related to pressures. We recommend that the set be expanded to include two additional pressure indicators, one for grazing and another for fire, in recognition of their extent and potential influence on rangeland biodiversity. We also recommend that benchmark sites be included in all ground-based monitoring programmes to provide reference standards for those biotic indicators about which little is known. Assessments of the current state of knowledge about these indicators for two case-study regions, the Gascoyne–Murchison strategy area and Cape York Peninsula, have shown that it would be possible to monitor most of them directly at regional scales, but that current monitoring programmes fall short of achieving this.

Key words: benchmarks, Cape York Peninsula, clearing, fire, Gascoyne–Murchison strategy area, grazing.

INTRODUCTION

Plants as indicators of rangeland biodiversity

Biodiversity indicators are environmental or biotic attributes whose measurement signals the effects on biodiversity of pressures caused by human activities. For measurement, indicators need to be embedded in a well-developed interpretive framework and have meaning beyond the individual attributes being measured (Saunders et al. 1998). There are many attributes of plant assemblages, populations and species whose measurement can meet these requirements for rangeland biodiversity.

At the broadest scale, attributes of vegetation communities provide considerable information about the environment in which they occur. This is because vegetation communities reflect physical environmental variables, such as climate, weather, topography and soils. Their response to the physical environment flows on to influence biophysical variables, such as pasture biomass, biotic richness and flammability. Productivity, as monitored by greenness indices derived from remotely sensed data, is an example of one such vegetation attribute that is a powerful indicator of the state of the environment.

Plants are also useful indicators of the response of biodiversity to pressures associated with land use. This is especially so for rangelands, where plants are the primary target of many of the most potentially damaging pressures: grazing, clearing, weed invasions, inappropriate fire regimes and global change, which act most directly on the composition and structure of plant communities, and only secondarily on other elements of biodiversity (Woinarski 2001a). Indicators of response to pressure can be assessed in terms of vegetation communities (e.g. extent of clearance, trends in woody or ground-layer vegetation, wildfire scars) or individual plant species or populations (e.g. trends in threatened species or populations of grazing- or fire-sensitive plant species).

Plants are also important elements of biodiversity in their own right; hence, indicators of changes in the composition of vegetation communities or the relative abundance of their constituent plant species provide important primary information about these components of biodiversity. In addition, because plants are sensitive to many of the processes threatening rangeland biota, and are also relatively amenable to monitoring, plant indicators have considerable potential as...
surrogates for less amenable elements of biodiversity and/or for providing early warning of potentially catastrophic change. For example, Pringle (2001) showed that, in the north-eastern goldfields region of Western Australia, grazing impacts on floristic composition were considerably more widespread than impacts on landscape pattern and process, which were concentrated near watering points where livestock congregated. (It is noteworthy that changes in ant species were also concentrated near watering points, possibly because they were responding primarily to changes in landscape function.) Why is it, however, that many current monitoring programmes differentiate plant species perceived to be functionally important, such as pasture plants and woody vegetation, and neglect the substantial proportion of the rangeland flora that is not perennial (Fisher 2001; Whitehead 2001a)?

The most comprehensive monitoring programmes in Australian rangelands have a strong pastoral focus (Fisher 2001; Whitehead 2001a). Pastoral monitoring programmes concentrate on vegetation cover and perennial plant abundance. Short-lived plants are seldom included, except perhaps as a single functional group labelled ‘forbs’ or ‘ephemerals’. This is because (i) they are less important than perennials for tracking long-term change in pasture condition; (ii) their inclusion would be time-consuming because there are so many species involved; (iii) their identification requires specialist expertise; and (iv) their populations are spatially and temporally variable and hence difficult to record consistently under different environmental conditions. These reasons are undeniably valid in the context of pastoral productivity, but they are far less valid when the specific context is biodiversity.

Some of the more difficult plants – those that are short-lived and not well-known – might be particularly sensitive indicators of grazing impacts on rangeland biota. For example, when Landsberg et al. (2002) compared plant diversity patterns on pastoral lands with those on comparable undeveloped lands in north-western South Australia, they found two contrasting patterns among species that declined under grazing. The first group consisted of species known to be pastoral ‘decreasers’. These were species that declined where grazing impacts were concentrated near watering points. They tended to be palatable, drought-hardy perennial species. Although their abundance was clearly affected by grazing, healthy populations of these pastoral decreasers persisted in areas of paddocks where grazing was less intense. The second group of plant responders was unexpected; it consisted of a suite of species that were always rare in paddocks and were only ever abundant outside the pastoral lands, where livestock had never grazed. Many of the species in this more sensitive group were short-lived and little-known. Some were found only outside the pastoral lands.

Therefore, these short-lived, often-neglected plant species could be among the most sensitive indicators of grazing impact on biodiversity.

Plants are not sensitive to all threats, however. Several significant threats to rangeland biota, most notably introduced predators, act directly on fauna and only indirectly, if at all, on rangeland plants. Plant-related indicators have limited utility for monitoring the impacts of such threats, except by way of their influence on animal habitats. Thus, although a set of plant-related indicators should be an important component of any biodiversity monitoring framework for the rangelands, it will never be sufficient for all purposes.

**Building on the Audit report**

A great deal has already been said and published about potential indicators for monitoring rangeland biodiversity. Fortunately, much of it has been encapsulated in a recent major report for the National Land and Water Resources Audit by the ‘Tropical Savannas Cooperative Research Centre (Whitehead et al. 2001). The purpose of the report was to develop an analytical framework for monitoring aimed at State, Territory and Federal Governments. The framework (Whitehead 2001b) was accompanied by a draft manual for monitoring (Woinarski 2001b) and substantial reviews of the literature published on the following: changes in the status of rangeland biodiversity and threatening processes (Woinarski 2001a); pastoral monitoring programmes and their current and potential contribution to biodiversity monitoring (Fisher 2001); existing biodiversity monitoring programmes (Whitehead 2001a); and relevant international experience (Beggs 2001). Central to the framework is a set of 11 core indicators of the status of rangeland biodiversity and the processes that threaten its persistence (Whitehead 2001b; Table 1).

Our aim in the current paper was not to revisit ground already covered in this comprehensive report; instead, we planned to focus on the plant-related indicators it proposes, with the intention of identifying what we saw as some of the key issues that have hindered their adoption in the monitoring of rangeland biodiversity, and suggesting potential avenues for resolving these issues.

The Audit indicators consist of attributes that can be measured at a range of inventory scales to provide information about pressures on biodiversity, and the response of biodiversity to these and other pressures. Together they have the potential to provide comprehensive information on the response of plant biodiversity to land use. To achieve this potential, however, we suggest they need to be embedded in an appropriate monitoring and interpretative framework, which we
We also suggest expansion of the indicator set to include two important pressure indicators, and the formal inclusion of benchmark sites to provide points of reference for interpreting site-based monitoring data.

**Interpretative framework**

The different vegetation and plant attributes that can be used as biodiversity indicators illustrate the tension between a requirement for less-intensive but broad-scale monitoring to provide context, and highly intensive but local-scale monitoring to provide details of processes. We follow Noss (1990) in suggesting that the various ecological scales at which biodiversity can be described constitute a nested hierarchy, beginning with a coarse inventory of broad-scale indicators, then overlaying data on the distribution of pressures to identify biologically significant areas, formulating specific questions about what might be occurring in these areas, and designing and implementing specific studies to address these questions.

The plant-related indicators identified in the Audit report span the extremes of this hierarchy, but provide incomplete information on the middle level (Table 1). Indicators 2 and 4 (clearing and ground cover) are essentially broad-scale indicators amenable to coarse-scale inventory. The other indicators are mostly fine-scale, high-resolution indicators that provide insight into what is occurring in relatively small areas. Although indicators 2 and 5 (clearing and exotic plants) provide some information on the distribution of pressures, we suggest this middle layer of the hierarchy needs to be expanded to include the two most pervasive pressures on many rangeland ecosystems.

**Additional indicators of pressure**

Many studies have shown that grazing by livestock (primarily sheep and cattle) and feral animals (particularly rabbits, goats, and pigs) is the greatest threat to rangeland plants of conservation concern (Woinarski 2001a). Changed fire regimes also pose significant threats to some plant communities, particularly in the tropical savannas (Russell-Smith et al. 2001). We suggest that the set of Audit indicators would benefit from expansion to include explicit means of monitoring these important pressures.

Like clearing, fire regimes can be monitored directly, so there is no need for surrogate indicators of this pressure. However, grazing pressure is far less amenable to direct monitoring, although it is an even more ubiquitous pressure on rangeland biodiversity. Grazing density (number of grazing animals per management unit) can and should be monitored, but it can provide a false impression of grazing pressure on individual vegetation types. This is because the distribution of grazing pressure is highly variable in space (depending at least as much on the preferences of grazing animals as their total numbers) and time (with grazing preferences affected by both seasonal conditions and previous grazing history). Because livestock need to drink regularly, watering places are one of the major influences that give spatial expression to grazing behaviour in arid rangelands (Lange 1969) and become foci for grazing and other impacts on biodiversity (James et al. 1999). Hence, distance from sources of water has become widely used as a surrogate for grazing pressure in observational studies (Pringle 2001; Landsberg et al. 2002; Pringle & Landsberg 2004) and national assessments (Landsberg & Gillieson 1996; Morgan 2001). It is a coarse-scale indicator, however, and many other factors need to be taken into account when considering how grazing pressure is likely to be distributed at paddock to regional scales (Cridland & Stafford Smith 1993; Pringle 2001; Pringle & Landsberg 2004).

**Benchmark sites**

The pluralistic nature of biodiversity adds another dimension of difficulty to interpreting data about...
indicators. There is increasing recognition that biodiversity cannot be adequately described by simple algebraic indices (e.g. richness, evenness and diversity) that treat all species as equivalent (Cousins 1991; Landsberg et al. 1999a). There is less agreement, however, about alternatives. Angermeier and Karr (1994) argued convincingly that biological integrity – the degree of intactness of a system relative to a benchmark state – would provide a more effective primary management goal than diversity. By definition, naturally evolved assemblages possess integrity, but random assemblages do not. Adding exotic species from distant populations might increase local diversity, but it reduces integrity. Furthermore, many changes in diversity can be evaluated objectively only on the basis of changes in integrity. For example, continuous grazing of a system evolved under intermittent or light grazing could increase local diversity, but eliminate unique entities. Although such a change can be interpreted as either a gain or a loss in diversity, there is no ambiguity about integrity, which is clearly reduced. Unfortunately, there are very few ecosystems for which the structural, functional and compositional elements of biodiversity have been sufficiently well described to define their benchmark state. This is especially true for many rangeland environments where even basic inventory data are lacking (Morgan 2001). The incorporation of benchmark sites into biodiversity monitoring programmes would go a long way toward rectifying this lack, as well as providing a means for evaluating the effects of management and standardizing reporting of site-based monitoring data (Appendix I).

**Testing Plant Indicators in Case-Study Regions**

The discussion that follows focuses on testing the applicability of the Audit’s plant-related indicators (Table 1), supplemented by additional pressure indicators, in two contrasting case-study regions: the Gascoyne–Murchison strategy area and the Cape York Peninsula bioregion. Both regions were referred to in the Audit report (Crowley & Fisher 2001; Hopkins & Watson 2001) and share many of the features that typify Australian rangelands (relative remoteness, low population density and extensive land use), while providing contrasts in climate, vegetation, land use, pastoral management and processes threatening biodiversity (Table 2).

### Table 2. Characteristics of case-study regions (Curry et al. 1994; Neldner and Clarkson 1995; Sattler and Williams 1999; Crowley and Fisher 2001; Hopkins and Watson 2001; Woinarski 2001a; Australian Natural Resources Atlas 2003a,b)

<table>
<thead>
<tr>
<th>Regional characteristics</th>
<th>Cape York Peninsula bioregion</th>
<th>Gascoyne–Murchison strategy area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Location</td>
<td>Far northern Queensland</td>
<td>Western–central Western Australia</td>
</tr>
<tr>
<td>Area</td>
<td>137 200 km²</td>
<td>Approximately 340 000 km²</td>
</tr>
<tr>
<td>Information base</td>
<td>Cape York Peninsula Land Use Study</td>
<td>Regional range surveys, land-system mapping and biogeographical survey</td>
</tr>
<tr>
<td></td>
<td>Vegetation mapping at 1:250 000</td>
<td>Approximately 460 rangeland monitoring sites; vegetation mapping at 1:250 000</td>
</tr>
<tr>
<td>Planning context</td>
<td>Cape York NHT Plan (Commonwealth and State government funded)</td>
<td>Gascoyne–Murchison strategy (Commonwealth and State government funded)</td>
</tr>
<tr>
<td>Climate</td>
<td>Monsoonal (&lt;1000 mm rainfall/year) 60% Reliable intense wet season; predictable but extended dry season</td>
<td>Arid and semi-arid (approximately 200 mm average rainfall/year) Winter dominant but unreliable rainfall; summer rains highly variable</td>
</tr>
<tr>
<td>Vegetation</td>
<td>Highly diverse flora (&gt;3000 spp.) Eucalypt woodlands 64% Melaleuca woodlands 14% Rainforest 6% (rich in endemics) Heathlands 3% Grasslands 6%</td>
<td>Highly diverse flora (&gt;2000 spp.) Mostly shrublands or low woodlands naturally depauperate in perennial grasses other than spinifex (Triodia spp.) Mulga (Acacia aneura) woodland and scrubs dominate the hardpan plains and ranges, with chenopod shrublands prominent on calcareous soils, floodplains and salt lakes</td>
</tr>
<tr>
<td>Population</td>
<td>&gt;18 000 people, of whom approximately 60% are indigenous, approximately 70% live in towns and approximately 4% (700 people) live on pastoral properties</td>
<td>Major population centres in the Murchison bioregion are Meekatharra (population approximately 2000), Leonora (population unknown), Cue (population approximately 400) and Mount Magnet (population 717) Population in the upper Gascoyne local government area is approximately 360 and there are no main centres</td>
</tr>
</tbody>
</table>
Approach

There are three aspects to using indicators to assess biodiversity: identification, implementation and interpretation. By identification, we mean determining which specific indicator components are likely to respond to different pressures. For example, eucalypt woodlands of monsoonal Australia have been considered highly stable and largely insensitive to changing fire regimes, whereas grasslands are rapidly contracting worldwide (Bowman 1988; Crowley & Garnett 1998). Monitoring widespread eucalypt woodlands might be less informative about the impact of changed fire regimes than monitoring the less-extensive grasslands. Identification of informative taxa and vegetation characteristics has been patchy through the rangelands. We describe current knowledge of potential indicators and how this could be supplemented.

Once appropriate indicator components are identified, a system of collecting and collating data about them must be implemented. Schemes of information collection are in place for some indicators, but are patchy. Most rangeland areas suffer from logistical barriers to regular monitoring. However, some creative solutions are available. We explore the adequacy of the current programmes and make suggestions for new initiatives. However, interpretation of the collected data is fraught with difficulties. Most of the data in the following discussion have been collected for purposes other than monitoring biodiversity, and there are many differences in methodologies. Hence, apart from stressing the value of benchmark sites, we make few recommendations regarding how an interpretative framework should be constituted, and merely flag it as an essential aspect of ensuring that the collection of indicator data achieves its primary purpose of informing about the status and trend of biodiversity.

The order we discuss the indicators differs from that in Table 1, because we have separated indicators relat-
ing mostly to pressures from those relating more to the response of biodiversity to those pressures.

**Pressure indicators**

**Grazing distribution (recommended additional indicator 1)**

**Domestic livestock** Spatially referenced data on the location of water points, fences and other infrastructure are available across most of the Gascoyne–Murchison strategy area (Curry et al. 1994; Watson & Thomas 2002); these provide much of the information necessary for modelling the spatial distribution of grazing pressure from livestock, particularly in landscapes that are widespread and uniform (or regularly patterned), where distance-from-water is a reliable surrogate for grazing pressure (Cridland & Stafford Smith 1993; Pringle 2001; Pringle & Landsberg 2004). However, many other factors related to landscape heterogeneity need to be taken into account to achieve paddock-scale resolution of actual grazing distribution (Pringle & Tinley 2001; Pringle & Landsberg 2004).

There is much less development of infrastructure on pastoral leases on Cape York Peninsula (Table 2), and fewer spatially referenced data are available on the location of the infrastructure that does exist. But even if such data were available, prospects for modelling the spatial distribution of grazing by livestock in the environments of the Peninsula are far less promising than they are for more arid rangelands such as the Gascoyne-Murchison area. In more arid rangeland areas, the activities of livestock and feral grazing animals are strongly constrained by their need to drink and by the relative scarcity of drinking water. This is not the case in Cape York Peninsula and many other parts of the monsoon tropics, where permanent and semi-permanent natural surface waters, and the grazing they support, are far more widespread. Although localized areas of heavy grazing pressure do occur near infrastructure foci such as yards, water and supplement points, they also occur in many other parts of the landscape, particularly at creek and river frontages (Ash et al. 1997), resulting in far more complex, and hence less predictable, patterns of distribution.

**Non-domestic grazing animals** Feral grazing animals are present in most rangeland regions in sufficient abundance to place local biodiversity under pressure (Wilson et al. 1992; Woinarski 2001a). High populations of native kangaroos also add to the total grazing pressure in some regions, particularly where they have benefited from the provision of artificial sources of drinking water and control of dingoes (James et al. 1999). Absolute numbers of these animals are seldom known, but their relative abundance in regions has been estimated by experts (Wilson et al. 1992; Morgan 2001), and their relative abundance within regions can usually be inferred from what is known of their habitat preferences.

In the Gascoyne–Murchison area, the most potentially damaging non-domestic grazers are feral goats, rabbits and native kangaroos (Wilson et al. 1992; Woinarski 2001a; Table 2), all of which can move freely through fences, and none of which is as strongly influenced by the location of drinking water as are domestic livestock. The most damaging may be feral goats, which are widespread and increasing in density in the region (Morgan 2001), and which are also able to graze in steep, rocky areas that previously provided a refuge from grazing for plants of restricted habitat (Woinarski 2001a).

The main grazing pressures from non-domestic animals on Cape York Peninsula come from feral cattle and pigs (Wilson et al. 1992; Woinarski 2001a; Table 2). Feral cattle are widespread, even in National Parks, which are generally unfenced and have a history of use as pastoral leases prior to their acquisition. Feral pigs are far more abundant, however, with an estimated population of 2.5 million, or 18 per square kilometre (Anonymous, date unknown). They are most abundant in floodplain and seasonal wetland environments, where their rooting activities cause substantial physical damage, but they also graze in native grasslands, where their impact on preferred species can be severe (G. Crowley, unpubl. data, 2003). Partial mapping of their regional distribution and abundance has been undertaken to assist in strategic control programmes (Anonymous, date unknown).

Environmental damage by feral rabbits, goats and pigs is nationally recognized as a key threatening process to Australian biodiversity; consequently, all three groups of animals are or will be subjects of nationally coordinated threat abatement plans, which have a strong monitoring component (Environment Australia 2003a).

**Fire regimes (recommended additional indicator 2)**

Allan et al. (2001) recently reviewed current programmes for monitoring fire regimes in northern Australia, which include ground-based measurements, aerial photographs and satellite technology. They found that ground-based programmes have been established since 1998 in Queensland and since 1993 in Western Australia, but are generally restricted to particular locations (e.g. national parks), issues (e.g. woody vegetation change) or opportunities (e.g. a particularly intense fire). They also report that satellite technology is now widely used to provide regular, broad-scale monitoring of the distribution of current fires (hotspots) and fire histories (fire scars as they accumulate during the course of a year). Fire history maps have been available for all of northern Australia since 1997, and for some regions since 1993. Satellite
data on hotspots and fire histories are readily available for both case-study regions (Hopkins & Watson 2001; Cape York Peninsula Development Association 2002).

Although there was far less documentation of fire regimes before satellite data became available, longer-term fire histories have been compiled for Cape York Peninsula from accounts of European explorers and from pastoralists’ diaries (Crowley & Garnett 2000). They indicate a shift in patterns of fire use since pastoral settlement, from fires being lit throughout the dry season (May–October) by Aboriginal people, to most burning being confined to the early dry season (May–August) since pastoral settlement.

**Trends in the extent of clearing of native vegetation (Audit indicator 2)**

The clearing of native woody vegetation is an issue of national concern and it is the focus of a collaborative, nationally coordinated, State-run monitoring programme (Barson et al. 2000). However, most States and Territories focus their monitoring on the intensively used agricultural zones that were once heavily wooded, and exclude large areas of extensively used rangelands where woody vegetation is naturally more open, and land cover is assumed to be relatively intact.

The exception is Queensland, which has established that clearing is also taking place in many rangeland areas and has expanded its monitoring area accordingly (Barson et al. 2000).

As a result, much of the Cape York Peninsula bioregion is now included in the Queensland Statewide Landcover and Tree Study (SLATS; Department of Natural Resources and Mines 2003). Prior to SLATS, the perception of clearing on Cape York Peninsula was that little had occurred, but it posed a threat to three of the region’s smallest but most fertile regional ecosystems (Neldner 1999). Summaries of more recent data collected by the Environmental Protection Agency, which include non-woody communities and woody communities with less than 20% crown cover (Wilson et al. 2002), confirm both assessments: little clearing has occurred (99% of the bioregion has not been cleared) and the limited clearing that has occurred has been concentrated in fertile regional ecosystems of limited extent.

Existing Statewide landcover data cannot be used to monitor clearing rates in the Gascoyne–Murchison strategy area, because Western Australia’s current landcover protocols do not provide data at an appropriate canopy-cover resolution. If it were available, it would likely show very little clearing has occurred, and hence that there has been little landscape-scale replacement of biodiversity. However, the possibility remains that any clearing that has occurred (e.g. that associated with irrigated horticulture) could have targeted ecosystem types of limited extent; hence, there could be loss of distinctive biodiversity, as appears to have occurred at Cape York Peninsula.

Remote sensing using Land Cover Change Analysis (LCCA) has been undertaken in a number of Western Australian rangeland regions, including two Landsat scenes around Leonora and Carnarvon in the Gascoyne–Murchison strategy area (Watson et al. 2001). Although outputs from the technique could be used to report on trends in clearing and cover, no systematic reporting has yet been undertaken in the Gascoyne–Murchison strategy area.

**Exotic plant species (Audit indicator 5)**

The National Land and Water Resources Audit assessment of landscape health (Morgan 2001) is the first Australia-wide, regional-scale assessment of the current status (distribution and density) of all 20 exotic plant species identified as Weeds of National Significance (WONS). Assessments were conducted at the scale of bioregion (Queensland) or subregion (all other jurisdictions), and based mainly on expert knowledge and simple ranked scores (occasional, common or abundant). The report indicates significant weed presence in both case-study regions, but particularly in Cape York Peninsula (Table 3). The maps produced by Morgan (2001) indicate the only WONS occurring in the Gascoyne–Murchison strategy region are buffel grass (mainly *Cenchrus ciliaris*; widespread, ranging from common to abundant) and mesquite (*Prosopis* spp.; occasional or localized). The only species in either region for which changes are reported by Morgan (2001) are buffel grass, which is reported as becoming increasingly extensive and increasing in density, and both rubber vine and mesquite, which, like most northern woody WONS, are reported as generally increasing in extent and density (Morgan 2001).

In the Gascoyne–Murchison strategy area, exotic plant species are assessed in the sites included in the Western Australian Rangeland Monitoring System (WARMS), but not with sufficient spatial coverage for mapping purposes (Hopkins & Watson 2001). Hence, detection of regional trends, particularly for species not identified as WONS but potentially threatening to regional biodiversity, remains problematic. Fortunately, few exotic species were recorded in the condition survey of the Murchison River catchment within the Gascoyne–Murchison strategy area (Curry et al. 1994), so they might pose less of a threat to biodiversity than environmental weeds currently pose for Cape York Peninsula.

Detailed weed monitoring was undertaken during the Cape York Weeds and Feral Animals Project between 1999 and 2003 (Anonymous, date unknown; Anonymous 2003). Of 37 significant weeds on Cape York Peninsula, 24 were thoroughly mapped (including six WONS), seven were partially mapped, and six were
Table 3. Significant environmental weeds present on rangelands in CYP (Crowley & Fisher 2001; Anonymous, date unknown, 2003), the threats they pose (Smith 2002; Institute of Pacific Islands Forestry 2003) and the extent to which they have been mapped (Anonymous, date unknown, 2003)

<table>
<thead>
<tr>
<th>Common name</th>
<th>Scientific name</th>
<th>Mapped</th>
<th>Distribution and threat</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gamba grass</td>
<td><em>Andropogon gayanus</em></td>
<td>Yes</td>
<td>Present in Bamaga, Weipa, Cooktown and Laura districts; can invade native grasslands and alter fire regimes causing serious damage to native woody species and other biota; Lockhart River to Temple Bay; and Newcastle Bay; WONS; dispersed by sea from populations further south; invades Melaleuca wetlands</td>
</tr>
<tr>
<td>Pond apple</td>
<td><em>Annona glabra</em></td>
<td>Yes</td>
<td>Eastern coastal areas from Cedar Bay to Cape Melville; dispersal by sea from populations further south; invades Melaleuca wetlands</td>
</tr>
<tr>
<td>Elephant creeper</td>
<td><em>Argyreia nervosa</em></td>
<td>Yes</td>
<td>Roadside infestations</td>
</tr>
<tr>
<td>Barleria</td>
<td><em>Barleria prionitis</em></td>
<td>Yes</td>
<td>Small populations in south-eastern CYP; ornamental weed that colonizes disturbed areas</td>
</tr>
<tr>
<td>Bauhinia</td>
<td><em>Bauhinia monandra</em></td>
<td>Yes</td>
<td>South-eastern CYP, Lakefield NP, Coen region; invades water courses in low rainfall areas</td>
</tr>
<tr>
<td>Mother of millions</td>
<td><em>Bryophyllum spp.</em></td>
<td>Partly</td>
<td>South-eastern CYP; agricultural weed; smothers native vegetation</td>
</tr>
<tr>
<td>Calopogonium</td>
<td><em>Calopogonium mucunoides</em></td>
<td>No</td>
<td>South-eastern CYP; agricultural weed; smothers native vegetation</td>
</tr>
<tr>
<td>Mossman River grass</td>
<td><em>Cenchrus echinatus</em></td>
<td>Partly</td>
<td>Associated with disturbance and coastlines; spread by animals Common along many river systems in central and southern CYP; WONS; smothers vegetation, replacing native species, particularly in heavily grazed areas fringing river systems</td>
</tr>
<tr>
<td>Rubber vine</td>
<td><em>Cryptostegia grandiflora</em></td>
<td>Partly</td>
<td>Present in central and southern CYP; WONS; has ability spread rapidly across flood plains and form dense stands, which smothers native vegetation and along riverbanks and floodplains</td>
</tr>
<tr>
<td>Navua sedge</td>
<td><em>Cyperus aromaticus</em></td>
<td>Yes</td>
<td>Small isolated populations in south-eastern and central CYP; weed of improved pastures and fertile soils</td>
</tr>
<tr>
<td>Water hyacinth</td>
<td><em>Eichhornia spp.</em></td>
<td>No</td>
<td>Present in restricted wetland areas in southern and western CYP; potential to spread in seasonal floodwaters</td>
</tr>
<tr>
<td>Hymenachne</td>
<td><em>Hymenachne amplexicaulis</em></td>
<td>Yes</td>
<td>Present in central and southern CYP; WONS; has ability spread rapidly across flood plains and form dense stands, which smothers native vegetation and along riverbanks and floodplains</td>
</tr>
<tr>
<td>Knob weed</td>
<td><em>Hyptis capitata</em></td>
<td>Yes</td>
<td>South-eastern CYP; agricultural weed of high rainfall areas Widespread and common throughout CYP; particularly in grazed and disturbed areas; forms dense thickets particularly on floodplain margins</td>
</tr>
<tr>
<td>Hyptis, Horehound</td>
<td><em>Hyptis suaveolens</em></td>
<td>No</td>
<td>Widespread and common throughout CYP, particularly in grazed and disturbed areas; forms dense thickets particularly on floodplain margins</td>
</tr>
<tr>
<td>Bellyache bush</td>
<td><em>Jatropha gossypijfolia</em></td>
<td>Yes</td>
<td>Palmer River area; replaces native vegetation</td>
</tr>
<tr>
<td>Lantana</td>
<td><em>Lantana camara</em></td>
<td>Yes</td>
<td>High rainfall areas in eastern CYP; WONS; forms dense thickets; replaces native vegetation; poisons stock</td>
</tr>
<tr>
<td>Lion's tail</td>
<td><em>Leonotis nepetafolia</em></td>
<td>Yes</td>
<td>Restricted locations in Lakefield NP and Bamaga district; has ability to develop large colonies that displace native species, particularly along riverbanks and floodplains</td>
</tr>
<tr>
<td>Leucaena</td>
<td><em>Leucaena leucocephala</em></td>
<td>Partly</td>
<td>Large areas in Weipa area; elsewhere around towns and along roadsides; highly invasive; replaces native vegetation in savannas</td>
</tr>
<tr>
<td>Giant sensitive plant</td>
<td><em>Minosa diplotochica</em></td>
<td>Yes</td>
<td>South-eastern CYP; pasture weed of high fertility soils in humid areas</td>
</tr>
<tr>
<td>Parkinsonia</td>
<td><em>Parkinsonia aculeata</em></td>
<td>Yes</td>
<td>Two small populations in south-western CYP; WONS; invades low rainfall areas, along river courses and floodplains</td>
</tr>
<tr>
<td>Parthenium weed</td>
<td><em>Parthenium hysterophorus</em></td>
<td>Yes</td>
<td>Scattered locations, mainly in south-eastern CYP; WONS; can invade all grazed agricultural and savannah landscapes</td>
</tr>
<tr>
<td>Praxelis</td>
<td><em>Praxelis clematidea</em></td>
<td>Yes</td>
<td>South-eastern CYP; invades pasture</td>
</tr>
<tr>
<td>Castor oil plant</td>
<td><em>Ricinus communis</em></td>
<td>Yes</td>
<td>Pormpuraaw township; forms thickets that replace other vegetation</td>
</tr>
<tr>
<td>Salvinia</td>
<td><em>Salvinia molesta</em></td>
<td>Yes</td>
<td>Laura River catchment; WONS; potential to spread in seasonal floodwaters and cover fresh to brackish lakes, dams and slow-moving waterways</td>
</tr>
<tr>
<td>Candlebush</td>
<td><em>Senna alata</em></td>
<td>Yes</td>
<td>Bamaga district; forms thickets in open areas</td>
</tr>
<tr>
<td>Sicklepod</td>
<td><em>Senna obtusifolia</em></td>
<td>Yes</td>
<td>Common in south-eastern CYP in disturbed areas with isolated populations in northern and central CYP; often in dense stands along flood plains</td>
</tr>
<tr>
<td>Sida</td>
<td><em>Sida acuta</em></td>
<td>Partly</td>
<td>Scattered infestations; associated with disturbance, particularly overgrazing</td>
</tr>
</tbody>
</table>
not mapped (Table 3). In most cases, mapping was combined with control measures and education of landholders. However, the programme was not cheap; its total cost was nearly $4.7 million, of which more than $3.6 million was provided directly from National Heritage Trust funds. Only some of these resources were used for monitoring exotic plants, with much of the project effort directed toward other components of the project, including community involvement, education and training, and pest control (pest animals as well as plants). The various activities are interlinked, however. For example, education and training are necessary precursors to community involvement in weed detection and monitoring.

Response indicators

Cover of native perennial ground-layer vegetation
(Audit indicator 4)

Remote sensing is routinely used to assess cover of ground-layer vegetation in several rangeland jurisdictions, particularly in South Australia and the Northern Territory, but not in either case-study region. Although the Department of Natural Resources and Mines Climate Impacts and Natural Resource Systems team in Queensland is working on a statewide project to monitor total ground cover using satellite imagery, it is still in the developmental stage (Taufee et al. 2002). The two main operational methods used in other jurisdictions are the grazing gradient approach (Pickup et al. 1998), which has most applicability to arid regions with widely spaced watering points, and LCCA (Karfs 2001), which is used mainly in savanna regions.

An early version of the grazing-gradient approach was tested in the Murchison and Carnarvon regions of Western Australia (Cridland & Stafford Smith 1993), but no consistent grazing-induced change could be discerned. The authors attributed this to the diversity of land systems and the number of watering points per paddock. Some experimental development of the LCCA technique has also been undertaken in the Gascoyne–Murchison strategy area (for the two Landsat scenes near Leonora and Carnarvon); the results were promising but there has been no follow up as yet (Watson et al. 2001).

Neither approach has been tested in the Cape York Peninsula bioregion. The LCCA technique could be applicable, given it has been adopted for routine monitoring in parts of the tropical savannas of the Northern Territory (Karfs 2001). To date, however, it has been most successful for detecting trends in relatively uniform grassland environments, and less successful for trend detection in more variable land system types, such as those that cover the majority of Cape York Peninsula (Karfs 2001). The extent of fire scarring on Cape York Peninsula (>70% of the total area burnt during 2002; P. Thomson, pers. comm., 2002) could further complicate the potential for using remotely sensed assessment to detect relatively subtle, grazing-related shifts in ground cover in this region. However, by undertaking the assessment very early in the dry season, effects of fire scars could be minimized.

We are not aware of any routinely collected ground-based data that could be used to map regional trends in vegetation cover in either case-study region. However, relative cover change has been estimated for the Gascoyne–Murchison strategy area, from ground-based data collected under the statewide WARMS. Ground cover was not assessed directly, but an index of shrub cover was estimated from field measurements of canopy width (Watson & Thomas 2002). Because

### Table 3. (continued)

<table>
<thead>
<tr>
<th>Common name</th>
<th>Scientific name</th>
<th>Mapped</th>
<th>Distribution and threat</th>
</tr>
</thead>
<tbody>
<tr>
<td>Singapore daisy</td>
<td>Sphagneticola triobata</td>
<td>Partly</td>
<td>South-eastern and isolated outbreak in central CYP; garden escapee; smothers riversides</td>
</tr>
<tr>
<td>Giant rat's tail grass</td>
<td>Sporobolus pyramidalis</td>
<td>Yes</td>
<td>Palmerville; replaces native grasses, particularly in association with overgrazing</td>
</tr>
<tr>
<td>Snakeweed</td>
<td>Stachytarpheta spp.</td>
<td>Yes</td>
<td>Weed of wetter areas</td>
</tr>
<tr>
<td>Grader grass</td>
<td>Themeda quadrivalvis</td>
<td>No</td>
<td>Prominent along roadsides, heavily grazed and disturbed areas in much of CYP; can invade native grassland and seriously reduce diversity</td>
</tr>
<tr>
<td>Thunbergia</td>
<td>Thunbergia grandiflora</td>
<td>Yes</td>
<td>Cooktown area; smothers native vegetation</td>
</tr>
<tr>
<td>Caltrop</td>
<td>Tribulus terrestris</td>
<td>Yes</td>
<td>Lakeland area; can be spread by road works</td>
</tr>
<tr>
<td>Chinese burr</td>
<td>Triumfetta rhomboidea</td>
<td>No</td>
<td>South-eastern CYP; weed of disturbed areas</td>
</tr>
<tr>
<td>Vetiver grass</td>
<td>Vetiveria spp.</td>
<td>No</td>
<td>Planted to stabilize edges of road cuttings; declared plant in Cook Shire; limited potential for spread as reproductively sterile</td>
</tr>
<tr>
<td>Noogoora burr</td>
<td>Xanthium occidentale</td>
<td>Yes</td>
<td>South-eastern CYP; spreads along waterways</td>
</tr>
<tr>
<td>Chinee apple</td>
<td>Ziziphus mauritiana</td>
<td>Partly</td>
<td>Western and central CYP; invades savannas</td>
</tr>
</tbody>
</table>

CYP, Cape York Peninsula; NP, national park; WONS, weed of national significance.
much of the Gascoyne–Murchison strategy area is
chenopod shrub-steppe, and shrubs dominate the
perennial ground layer at most sites (650 of 686),
shrub cover was seen as a reasonable surrogate for the
perennial ground-layer vegetation. The estimated
shrub cover was greater than the actual shrub cover
because it was based on maximum canopy width and
assumed 100% foliage cover. Hence, the cover esti-
mates were used to assess relative change between two
monitoring dates, rather than to make inferences about
absolute cover on any particular date. Watson and
Thomas (2002) used these estimates to assess relative
cover change for 223 WARMS sites that had been
assessed twice – once between 1993 and 1997 and a
second time between 1999 and 2001. Estimated
canopy area increased during the interval on 96% of
sites, with an average increase in canopy area per site of
81%. The increase was consistent across different
vegetation groups and was attributed to unusually high
rainfall during the assessment period. No benchmark
sites were included in the WARMS programme, so it
was not possible to determine whether grazing
management was also influential.

Just as the lack of benchmark sites in the WARMS
programme means there is little ecological justification
for using the data to make inferences about the impact
of grazing on biodiversity, the relatively low density of
sample sites means there is no statistical justification for
using the data for regional mapping. Although the
WARMS sampling effort was moderately intensive,
with around 460 sites in the combined Gascoyne and
Murchison bioregions, it equates to site densities of
only 1 site per 1000 km² (both estimates from table 5
in Fisher 2001), a density much too low to provide
sufficient statistical rigour for making inferences about
spatial coverage (Webster & Oliver 1990). The data are
useful for compiling regional narratives, however, and
also for cross-regional comparisons of selected subsets of
data.

Despite the shortcomings of the WARMS pro-
grame for biodiversity monitoring, it far exceeds
any routine ground-based monitoring currently being
undertaken on Cape York Peninsula. The existing
Queensland pasture monitoring programmes, QGraze
and Traps, do not include any sites on Cape York
Peninsula (Fisher 2001). Queensland Parks and
Wildlife Service has recently established a small
number of sites where plants that dominate the
ground cover are assessed (approximately 25 by 2001;
Crowley & Fisher 2001; and another 30 sites in 2002;
G. Crowley, unpubl. data, 2003), with all sites being
measured between two and four times. The first sites
were specifically situated to measure changes in the
grassland habitat of the golden-shouldered parrot
(Psephotus chrysopterygius) on a single property, rather
than to provide a basis for regional mapping. The
second group of sites covers a wider range of habitats,
but all are on the same property. Crowley and Fisher
(2001) suggested that reassessment of approximately
700 of the inventory sites used for mapping of the
region’s vegetation could provide a statistically
adequate sample for mapping vegetation change, but
no repeat measurements have been undertaken to
date. Locational information for most of these sites is
too inaccurate for within-site comparisons to be made
(J. R. Clarkson, pers. comm., 2003). However,
comparison of all sites within a regional ecosystem
should provide useful information on trends.

One such study of eastern-central Cape York
Peninsula indicated that management-induced cover
change may be extensive. This study compared 64 sites
surveyed in 1964 and 1993, and identified several
species (including Themeda triandra, Sorghum plumo-
sum, Heteropogon spp.) and communities (grasslands)
that appeared particularly susceptible to changing land
management (Crowley & Garnett 1998).

Fire-sensitive plant species and communities
(Audit indicator 6)

Fires are uncommon throughout much of the
Gascoyne–Murchison strategy area, except in the
limited areas of spinifex grasslands, and we have no
information about the distribution or environmental
correlates of fire-sensitive vegetation types.

Fires are widespread and frequent throughout Cape
York Peninsula, where fire is recognized as an im-
portant influence on biodiversity (Crowley & Fisher 2001;
Cape York Peninsula Development Association 2002).
Interpretation of what constitutes fire sensitivity is
complicated, however, because much of the region is
naturally fire-prone; hence, plant species and com-
munities respond to fire regimes in ways that are not
adequately described in terms of simple presence or
absence of wildfire. Fire-response vegetation groupings
are being developed by Queensland Parks and Wildlife
Service as part of their ongoing fire management
planning (G. Crowley, unpubl. data, 2003). Based on
regional ecosystem units, such classification is made on
a property by property basis. Detailed ground-truthing
of mapped fire-sensitive units and observations of their
responses to contemporary fire regimes are required to
improve this system. Lists of fire-sensitive plant taxa
are yet to be compiled.

It should be possible, however, to make inferences
about likely impacts of current fire regimes on savanna
communities more generally, using a combination of
existing mapping (e.g. Fox et al. 2001), documented
responses of major savanna vegetation types to fire
regimes (Russell-Smith et al. 2001) and satellite-based
monitoring of fire regimes (Allan et al. 2001).

Two contrasting types of fire-sensitive plant com-
munities are currently of greatest concern (Russell-
Smith et al. 2001), and should therefore be a priority
for such analysis. The first is vegetation types that contain many obligate seeder species that can be eliminated by too frequent fires. Examples include riverine gallery forest and heathlands of rocky rangelands. Because they tend to occur in relatively small patches or narrow linear strips that cannot be resolved at current mapping scales (1:250 000 or coarser), these vegetation types are often not individually mapped, making precise location difficult. They are, however, frequently listed as a minor component of the vegetation in the polygons that are mapped. The second is vegetation types where woody thickening and shrub invasion occur if fires are too infrequent, or not sufficiently intense. Examples include grazed mesic savannas, and grasslands associated with seasonally inundated wetlands (Crowley & Garnett 1998). Although some of these vegetation types are wide spread, many individual grassland patches are too small to be included on currently available maps, even as a minor component in vegetation polygons.

In conclusion, although vegetation and fire regime mapping are moderately well advanced in many savanna regions, it is seldom at sufficiently high resolution to locate small areas of fire-sensitive vegetation accurately. Hence considerable on-ground work will usually be required to assess the relationship between vegetation types and fire regimes, particularly when patches with different fire requirements occur in close proximity.

Grazing-sensitive plants (Audit indicator 7)

This indicator remains one of the most problematic for reporting at regional scales, because there are inherent constraints to making cross-regional comparisons from data on individual indicator species. There are two reasons for this. One is that the distribution of many rangeland plant species, particularly in the arid and semiarid zones, is localized (restricted extent) or sparse (widespread, but with low local abundance). This is illustrated by the findings of Landsberg et al. (in press) on eight grazing gradients located across the central and southern rangelands of Australia. Of the 466 ground-layer species identified, 72% were found on only one gradient and none at all were found on all eight. Even within individual gradients, 45% of species were restricted to just one or two of the six sites that constituted a gradient and 46% were locally uncommon at every site where they were found. Hence, few plants in these rangelands are likely to be sufficiently wide spread or common to have potential as indicators in spatially or environmentally disparate locations. This might not be such a problem in the tropical savannas, where a suite of grazing sensitive plants might be more widespread.

The second inherent problem likely to be common to all rangeland types is that many plant species show differential responses to grazing depending on the nature of the grazing pressure and the plant assemblage in which they occur. This occurs because palatability to grazing animals is context-sensitive, and so too is the ability of plant species to tolerate grazing (Landsberg et al. 1999b). For example, in the grazing gradient study by Landsberg et al. (in press), two relatively wide spread and common grasses, Aristida contorta and Eragrostis dielsii showed mixed responses to grazing; on mulga gradients, where more palatable grass species were common, both tended to increase in response to grazing, whereas on chenopod gradients, where no grass species were common, both species declined with grazing.

Attempts to identify indicator response types, rather than species, for monitoring grazing-induced change have met with limited success. For example, in a study in south-western Queensland, species with ‘large, erect tussocks branching above ground’ showed potential as response types indicative of light grazing in a mulga grassland community, and species with ‘small, sprawling basal tussocks’ showed potential as indicators of heavy grazing; however, no comparable trait combinations were identified for a gidgee shrubland community in the same region (Landsberg et al. 1999b). One of the most widespread grass species showing this decreaser syndrome is also one of the most consistent in its response: Themeda triandra was once widely distributed in semiarid, temperate and tropical rangelands in Australia and southern Africa; it generally decreases in abundance in response to moderate to heavy grazing pressure in most environments where it occurs. Localized responses are known for many other pasture species in particular environments, and these are widely promoted as indicators of pasture condition in those environments (Mitchell & Wilcox 1994; Partridge 1999). The problems with using these species as indicators of biodiversity condition are twofold. First, because they are useful pasture species they are almost invariably relatively insensitive to grazing – by the time known pasture indicators are in decline many other less well-known species may be lost from the local species pool (Landsberg et al. 2002). Second, their local abundance depends as much on site potential as on the cumulative effects of grazing; hence, without benchmark sites, the information provided by monitoring their abundance is likely to be difficult to interpret and insensitive to all but gross trends.

What this means is that detailed monitoring against benchmark sites is required (i) to identify the local pool of grazing-sensitive species; and (ii) to separate grazing-related trends in their abundance from natural variation. Woinarski (2001a) suggests this should be done for one or several nominated species or species groups for each environment of interest. However, to achieve (i), we suggest that it would be more efficient and effective initially to monitor as wide a range of...
candidate species as possible in order to maximize the prospects of identifying the most grazing-sensitive responders.

It might be possible, with sufficiently intensive ground-based monitoring of both treatment plots and adequate controls or benchmark sites, to derive regionally comparable measures (e.g. proportion of the regional species pool in decline). However, the intensity of sampling that would be needed to do this for each of the main vegetation types in a region is likely to limit the number of regions and/or vegetation types in which it could be conducted. Development of regional narratives linked to locality-specific information might be a more realistic reporting aim for monitoring this indicator. Current monitoring sites would need to be supplemented with benchmark sites to achieve even this rather narrow objective. Linking locality-specific monitoring like this with aerial-based monitoring of grazing pressure could prove more informative and cost-effective than attempting to achieve spurious cross-regional comparability in trends shown by individual species.

Watson and Thomas (2002) used ground-based data from the WARMS in the Gascoyne–Murchison strategy area to explore the utility of pastorally defined plant response groups. They used published sources and local knowledge to assign the 71 perennial species monitored on WARMS sites to four categories based on postulated responses to grazing: decreasers (33% of species), intermediates (44% of species), increasers (13% of species) and unassigned species (10% of species). They found little difference among categories in population growth rate, canopy growth and frequency of occurrence, and they attributed this to the overall improvement in the condition of most sites over the assessment period (which was one of unusually high rainfall). The only substantial difference within response groups was for canopy size, where the overall increase was less for decreaser species than for the others. The authors suggested that, although this might reflect a differential response as a result of grazing sensitivity, it was not sufficient to influence other population parameters. However, they also noted that the allocation of species to grazing response groups was difficult and this is currently under review.

The study in eastern-central Cape York Peninsula (referred to under indicator 4, ground cover) identified Themeda triandra, Sorghum plumosum and Heteropogon spp. as potential grazing indicators in these tropical savannas (Crowley & Garnett 1998). Chrysopogon fallax is another species that could be an indicator of grazing pressure in the region, although the data are equivocal. Although vast areas of Cape York Peninsula were mapped as Chrysopogon fallax-dominated pastures in the 1960s (Galloway et al. 1970), it is seldom dominant in pastures today. Rather than indicating a decline, however, Crowley and Garnett (1998) suggested it might indicate an artefact of extrapolating from one mapping unit to another.

Listed threatened plant species and communities (Audit indicator 10)

Commonwealth regulation of environmental issues has been greatly expanded with the introduction of the Environment Protection and Biodiversity Conservation (EPBC) Act 1999, which places special emphasis on listed threatened entities (species and communities) as matters of national environmental significance (Environment Australia 2002a). Therefore, their inclusion as biodiversity indicators is important within both regulatory and planning contexts. The value of lists of threatened species as indicators is, however, limited by bias in the types and locations of the species included in them, and the bureaucratic nature of the listing process, which ensures that the lists are very slow to change (Possingham et al. 2002). Furthermore, changes to the list are usually a result of changes in knowledge (resulting from inventory surveys) rather than changes in actual status, particularly for plant species; and last, bulked indices of numbers of threatened entities per site or region are ambiguous as indicators of biodiversity status. For example, increases in numbers of listed species in an area could indicate an improvement in their status (identification of populations not previously reported from the area) or deterioration (additions of local species to the threatened lists).

At the national level, nominations to add species or communities to or to subtract species or communities from threatened lists are assessed by the Threatened Species Scientific Committee, which uses IUCN criteria for guidance in determining conservation status and provides detailed justifications for all determinations (IUCN 2001; Environment Australia 2002b). However, the vast majority of currently listed entities were assessed before the EPBC Act commenced and no documentary evidence is available describing the rationale for their listing. For example, approximately 90% of the plant species currently listed as endangered were transferred from pre-EPBC Act lists (Environment Australia 2002c). Recent work on the listed plants of Cape York Peninsula illustrates many sources of potential bias or error in current lists of threatened plant species at State and Commonwealth level (J. Landsberg & J. Clarkson, unpubl. data, 2003). These include the following:

1. Discrepancies between State and Commonwealth lists, mainly as a result of poor information flow.
2. Overrepresentation of some taxa, particularly orchids, palms and cycads, in both lists (Table 4).
3. Selective listing of some species on the basis of restricted distributions, without evidence of threat-
4. Listing of species thought to be uncommon on the basis of herbarium records, but shown to be common and/or widespread when searched for.

Although the first three problems could eventually be resolved if old lists are reviewed, and species assessed under new procedures come to dominate the lists, the last problem is symptomatic of a broader issue relating to the types of entities most likely to be listed. Most plant species tend to be assessed against distributional, rather than population criteria. For example, the two IUCN criteria most applicable for assessing plant species are criterion B1(ii) (small distribution coupled with inferred continuing decline) and criterion D1 (extremely restricted distribution) (Gardenvors 2001). Hence, the plant species most likely to be listed are those that are localized in their distribution. Sparsely distributed species (those that are widespread but have low local abundance everywhere) are unlikely to be listed because detection of their rarity is only apparent from population data, and these are seldom available. Yet many rangeland plants are sparsely distributed (Landsberg et al. in press). Although such plants may be highly susceptible to widespread threats, such as grazing and inappropriate fire regimes, because the plants are themselves widespread, they are unlikely to be listed; hence, changes to their status are unlikely to be reflected in lists of threatened species.

Listing of threatened ecological communities is a much newer, and hence less tested, endeavour. As far as we are aware, Australia is the only country to include threatened ecological communities in its formal listing process. Although many States and Territories identify and list threatened ecological communities, they do this at a range of scales and using a variety of different criteria. Currently there are more than 580 threatened ecological communities recognized by New South Wales, Victoria, Western Australia, Queensland, Tasmania and the Australian Capital Territory; whereas the Northern Territory and South Australia do not maintain lists of threatened ecological communities (Environment Australia 2003b).

Threatened ecological communities achieved particular prominence when the Commonwealth EPBC Act came into force in 2000 (Environment Australia 2002a). This is because threatened ecological communities are identified as a matter of national environmental significance under the EPBC Act’s assessment and approval provisions. As a consequence, any action likely to have a significant impact on a listed threatened ecological community requires approval from the Commonwealth Environment Minister. In addition, the Commonwealth Threatened Species Scientific Committee is striving for a unified approach to defining ecological communities that is consistent across the States and Territories (Environment Australia 2003a), with obvious benefits for national reporting of regional conservation status.

As yet, however, there are still relatively few ecological communities formally listed as nationally threatened. Criteria for listing communities can be based on declining distribution or on declining condition (Environment Australia 2002d). For those communities listed on the basis of declining distribution (usually in the form of clearing), list status essentially duplicates information provided under indicator 2. This is the case for most of the currently listed communities. However, for many rangeland communities, listing on the basis of declining condition has potential to be far more informative. Several of the communities listed since the inception of the EPBC Act fall into this category; hence, monitoring their status and trend could provide a useful indication of the status of the biota they include. Examples of such communities include the community of native species dependent on the natural discharge of groundwater from the Great Artesian Basin, and bluegrass (*Dichanthium* spp.).

### Table 4. Representation of families and life forms of plant species occurring on Cape York Peninsula and included on State and/or Commonwealth lists of threatened species, compared with the total flora of the bioregion (J. Landsberg, unpubl. data, 2003)

<table>
<thead>
<tr>
<th></th>
<th>Listed species (%)</th>
<th>All species recorded in bioregion (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(n = 69)</td>
<td>(n = 3338)</td>
</tr>
<tr>
<td>Orchids (Orchidaceae)</td>
<td>30</td>
<td>5</td>
</tr>
<tr>
<td>Palms (Arecaceae)</td>
<td>6</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Grasses (Poaceae)</td>
<td>7</td>
<td>9</td>
</tr>
<tr>
<td>Cycads (Cycadaceae)</td>
<td>6</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Other trees</td>
<td>10</td>
<td>ND</td>
</tr>
<tr>
<td>Other shrubs</td>
<td>17</td>
<td>ND</td>
</tr>
<tr>
<td>Other vines, creepers and twiners</td>
<td>12</td>
<td>ND</td>
</tr>
<tr>
<td>Other epiphytes</td>
<td>1</td>
<td>ND</td>
</tr>
<tr>
<td>Other herbs</td>
<td>4</td>
<td>ND</td>
</tr>
<tr>
<td>Ferns</td>
<td>6</td>
<td>ND</td>
</tr>
</tbody>
</table>

ND, no data.
dominant grasslands of the Brigalow Belt bioregions (Environment Australia 2002e).

There are no nationally listed ecological communities in either case-study region, however, or in most other rangeland bioregions.

Benchmark sites

Given the importance of biodiversity benchmark sites for distinguishing between seasonal fluctuations and management effects, and providing a mechanism for standardizing cross-regional reporting (Appendix I), what are the prospects for locating benchmark sites in the case-study regions?

Although the WARMS includes a far more extensive system of ground-based monitoring sites than most, it does not include benchmark sites. Instead, most sites are deliberately located in the zone of greatest anticipated grazing-induced change, usually between 1.5 and 3.0 km from livestock watering points. In the Gascoyne–Murchison strategy area, only 4% of the 686 WARMS sites are in the lightly grazed zone more than 5 km from water (Watson & Thomas 2002). Watson and Thomas (2002) have identified the following gaps in need of consideration, were WARMS to be expanded for more general environmental monitoring: over representation of pastorally productive areas and under representation of areas considered pastorally robust or of low pastoral value; restriction to land held under pastoral tenure; exclusion of vegetation types of potential importance for regional biodiversity (e.g. ephemeral wetlands, riparian areas); and low spatial coverage. We suggest the lack of benchmark sites against which to standardize measures of biodiversity indicators is an additional gap reducing the utility of WARMS sites for monitoring biodiversity.

Fortunately, there appear to be reasonable prospects for locating benchmark sites in the Gascoyne–Murchison strategy area, at least for the Murchison catchment, where lightly grazed reference areas have been identified for all but one major vegetation type (Curry et al. 1994). The Murchison report also indicates a moderate number of areas greater than 5 km from water within particular vegetation types, where grazing impacts are expected to be relatively light (but see Landsberg et al. 2002 for limitations of this approach).

Locating benchmarks in Cape York Peninsula will be more difficult, however, because the main pressures on biodiversity – inappropriate fire regimes, weed incursions, feral cattle and pigs – are extremely widespread and largely independent of land use or tenure boundaries. Although complete eradication of feral animals and weeds from all areas of the Peninsula is not a realistic management aim, eradication in some areas of especially high conservation value is being attempted (G. Crowley, unpubl. data, 2003). If studies show that uncontrolled grazing and other pressures have major impacts on threatened biota, they could provide impetus for implementing other targeted eradication programmes. Should such programmes be undertaken, there could be major benefits for the monitoring of regional biodiversity if potential benchmark sites were included.

CONCLUSIONS

Assessments of the current state of knowledge of the plant-related indicators proposed in the Audit report (Table 1) show that it would be possible to monitor many of them directly at regional scales, and to use the results to assess national status and trend in plant diversity. However, the only monitoring programmes that currently provide that level of information are the statewide landcover and associated regional ecosystem survey and mapping programmes in Queensland. The WARMS provides an extensive network of ground-monitoring sites that can be used for making inferences about some of the indicators. However, significant gaps in the system limit the utility of the sites for making inferences about biodiversity. Lists of threatened plant species and communities are maintained at national and State scales, and could also be incorporated into biodiversity monitoring programmes without collecting additional data. Unfortunately, their utility for making inferences about biodiversity status and trend is limited by the nature of the listing processes.

Current monitoring programmes do not provide sufficient information about most of the ground-based indicators to allow statistically meaningful extrapolation for regional reporting of their condition. Impediments are partly to do with resourcing (implementation and sustained funding of new programmes would be required in most cases, and/or considerable supplementation and ongoing funding of existing ground-based programmes); partly to do with technological limitations (remotely sensed assessment of ground cover might require further technical development before it could be applied across the diversity of rangeland environments); and partly to do with knowledge limitations, particularly about how to link ground-based, species-level information to regional trends in biodiversity status.

Two additions to the framework recommended in the Audit report would go some way toward assisting implementation of many of the ground-based indicators:

1. Indicators of all the main pressures acting on rangeland biota should be explicitly included in the core set to provide a spatial and environmental context for selecting where to locate ground-based monitoring sites. This means including indicators.
of grazing pressure and fire regimes that can be monitored at sufficiently broad scales to provide a context for identifying meaningful locations for case studies and other forms of ground-based monitoring.

2. For ground-based indicators, there should be an explicit requirement that trends are assessed relative to benchmark sites, as outlined in Appendix I. The only exceptions should be for those indicators where reference conditions are implicit in the indicator (e.g. the implied benchmark states for clearing and exotic plants are no clearing and no weeds).

With these additions, the plant-related indicators in Table I provide a workable set that has potential for providing valuable information about the regional status of rangeland biodiversity. We reiterate, however, the importance of using the indicators at appropriate scales within a nested hierarchy, as suggested by Noss (1990). From the present review of the state of knowledge of the indicators in the case-study regions, we suggest the following:

1. Indicators 4 (ground cover) and 10 (listed threatened entities) might be most appropriate for coarse-scale inventory.

2. Indicators 2 (clearing) and 5 (exotics), plus the additional indicators of fire regimes and grazing pressure might be most appropriate as overlays to identify specific areas where biodiversity might be at risk.

3. Indicators 6 (fire-sensitive plants) and 7 (grazing-sensitive plants) might be most appropriate for high-resolution studies to address questions about condition and trend in identified priority areas.

REFERENCES


APPENDIX I

Biodiversity benchmarks

What are benchmarks?

The Australian Concise Oxford Dictionary (1987) defines a benchmark as ‘a criterion or point of reference’. In ecology, benchmark sites are often used to establish reference conditions for complex or highly variable ecosystem properties. In the USA, for example, Brinson and Rheinhardt (1996) used wetlands chosen for their high level of sustainable functioning as benchmark sites to set standards for the restoration of damaged wetlands, whereas Fulé et al. (1997) used pre-settlement fire regimes and forest structure at benchmark sites to establish reference conditions against which to assess the impact of fire exclusion on Pinus ponderosa forests.

Why use benchmarks for monitoring biodiversity?

Biodiversity is, by definition, highly diverse. As an ecosystem property it is also inherently complex because it is multifaceted. Hence, site-specific reference conditions for biodiversity indicators are frequently poorly understood, and are sometimes unknown. In such cases, benchmark sites are essential for determining locally appropriate reference standards. In the specific context of monitoring rangeland plant diversity, benchmark sites can serve three related purposes:

1. They can provide a source of background information about variation in environments where natural history knowledge of many indicator taxa is rudimentary and seasonal fluctuations in abundance can be extreme.

2. Because of this they can provide local points of reference against which to evaluate the effects of specific management actions on the indicators being monitored. This can be for simple demonstration purposes (showing local managers what the system can look like) or for calculation of indices of change.

3. Site-specific data on indicator trends can be standardized against the reference conditions established at comparable benchmark sites to allow meaningful comparisons among ecosystems, landscapes and regions with few biotic elements in common.

What selection criteria are appropriate for identifying biodiversity benchmarks?

Worldwide, many geographically restricted native species are declining because of rapid change, and being replaced by a much smaller number of expanding species that thrive in altered environments (McKinney & Lockwood 1999). Hence, native species that have evolved in situ and are not coping well with change are the most vulnerable and least replaceable elements of global biodiversity. It is important, therefore, that biodiversity benchmarks provide points of reference for monitoring their status and trend. However, how to choose such benchmarks is a subject of some controversy. Some scientists have recommended using naturalness as the primary criterion, with natural defined to mean ‘without human influence’ (Hunter 1996). Others (e.g. Haila 1997) have argued that it is neither possible, nor especially helpful to separate people from nature. Haila (1997) suggests we should instead focus deliberately on those factors we know are likely to cause harm to things we want to protect. In the specific context of rangeland plant diversity, a focus on potentially harmful factors makes selecting biodiversity benchmarks relatively simple – conceptually at least. Many studies have shown that grazing by livestock and feral grazing animals is the primary threat to rangeland plants of conservation concern (Woinarski 2001a). Logically, therefore, benchmark sites should be located in areas that have been minimally grazed by livestock and feral grazing animals, while still being representative of the ecosystem of interest. They should also be chosen where other potentially threatening factors, such as changed fire regimes, clearing or weed incursions (Woinarski 2001a), have had minimal influence.

Best available is much better than nothing

Nowhere in the Australian rangelands is totally free of impacts from threatening factors, nor is their influence evenly spread (Morgan 2001). Within any plant community or ecosystem, sites that have been least impacted by local threats (i.e. sites in the best relative condition) are likely to capture more of the local biodiversity than sites that have been more heavily modified (Gibbons et al. 2002). Thus, they should constitute the best available local benchmarks.

Using benchmarks for regional-scale assessments

Parkes et al. (2003) have developed an operational approach using benchmarks for regional-scale biodiversity assessment. Although it has been developed in agricultural landscapes in Victoria, their ‘habitat hectares approach’ has potential for similar scales of operation in rangeland environments. It is based on explicit comparisons between existing vegetation features and those in benchmark sites representing the average characteristics of mature stands of the same
vegetation type in the best available condition. The features assessed are chosen to provide an integrated view of habitat suitability for all local native species that might reasonably be expected to use a site. In fragmented agricultural landscapes in Victoria they include site factors such as numbers and health of large trees, understorey cover and life forms, and lack of weeds; and landscape factors such as patch size and connectivity to other native vegetation in the neighbourhood. Although different features would be needed to characterize habitat quality in rangeland landscapes, a range of contenders has been identified in recent studies (e.g. Martin & Green 2002) and monitoring guides (e.g. Department of Natural Resources 1999).