Rangeland condition: its meaning and use

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Executive summary

Rangelands occur around the globe in areas to dry or with soils and topography unsuitable for broad-acre farming but fertile and wet enough for pastoralism. Because pastoralism may intensively utilise water and vegetation some rangeland areas can become damaged. This damage means that pastoralism itself ultimately suffers and damage causes conflicts with other land uses such as conserving biodiversity, hunting and gathering bush foods or firewood. Conflicts occur over these multiple uses because people with a sincere interest in rangelands (stakeholders) desire to maintain them in a state of good condition. We define rangelands in good condition as those systems having healthy (i) biophysical functions that include a high capacity to retain water, capture energy, produce biomass, cycle nutrients and provide habitats for diverse populations of native animals, plants and microorganisms, and (ii) socio-economic functions that adequately provide people with their material, cultural and spiritual needs. To maintain rangelands in good condition, these biophysical and socio-economic functions need to be measured and reported. This involves developing monitoring procedures and a system for reporting monitoring information to stakeholders.

In Australia, monitoring has historically been conducted by those responsible for maintaining healthy rangelands, typically State and Territory Government Department personnel in collaboration with local land managers and, in some cases, with regional catchment management authorities and natural resource management boards. Reporting rangeland monitoring information to stakeholders is facilitated by the Australian Collaborative Rangeland Information System (ACRIS), which was formed as a partnership between government agencies concerned with rangeland issues. State and Territory Departments provide rangeland monitoring data to the ACRIS, and assist with analysis, synthesis and reporting these data to stakeholders. Two national reports have been produced for stakeholders: *Rangelands – Tracking Changes, the Australian Collaborative Rangelands Information System* (in 2001), in which ACRIS was proposed, and *Rangelands 2008 – Taking the Pulse*. The ACRIS Management Committee commissioned this discussion paper to evaluate the way in which biophysical information on vegetation and soils was compiled in the latter report – biodiversity information is being evaluated in another paper – and to recommend possible ways to improve future reporting. This paper also reviews the literature describing the concept of rangeland condition and what it means according to different stakeholders' values and land use goals.

Major findings and recommendations

Concepts

1. Rangeland condition is commonly defined as a 'state of health' of an area and is viewed as analogous to human health. The health of a rangeland area is assessed by measuring attributes and indicators of its current functional state relative to an expected norm (a reference state). An indicator is a simple surrogate or index for a difficult to measure attribute.
   We recommend that:
   - the terms 'rangeland condition' and 'rangeland health' be viewed as equivalent; and
   - the condition or health of a rangeland area be assessed relative to a reference state. If reference sites within the area of interest are not available, then the characteristics of a highly functional state (healthy condition) should be defined hypothetically by stakeholders in a workshop setting and used as the reference state.
2. Statements assessing rangeland condition (or health) depend on values held by stakeholders. Given the same data on the functional state of a specified area, one stakeholder may judge the rangeland to be in good condition and another in poor condition. In other words, assessments of rangeland health or condition are related to purpose and are, in a sense, 'in the eye of the stakeholder' because each stakeholder group will evaluate rangeland health from their own point of view.
   We recommend that:
   - statements on rangeland condition clearly specify the stakeholder or stakeholder group making the assessment because different evaluations (and conflicts) are to be expected from different stakeholders;
   - participative approaches be used to resolve conflicting statements on rangeland health. Workshops allow participants to share their visions and goals in making their condition assessments and to learn from each other's experiences and land management choices; and

- participative approaches also be used when modifying or expanding monitoring programs to help select where and what to monitor because rangeland condition, as a 'functional state' or 'state of health' relative to a 'reference state', can be characterised by monitoring in different ways and by measuring different biophysical and socio-economic attributes and indicators.

Data and their interpretation

3. In addition to colour-coded maps and graphs such as 'time traces', we propose the use of colour-coded 'time-mark' graphs to interpret how a rangeland area is changing. These time-mark graphs are developed by positioning recently measured values for an indicator along a continuum (time-marks on an axis) relative to (between) maximum and minimum indicator values representing highly functional and totally dysfunctional reference states, respectively (see Section 4.6). Time-marks along this continuum are colour coded based on a matrix combining 'seasonal quality' (rows) and direction of change (increase, decrease or no change) for indicator values (columns). Reported change is the time period of interest relative to changes from prior periods or reference areas (see Box 2.1. 'Matrix: seasonal quality and direction of change' in *Rangelands 2008 – Taking the Pulse*). The time-mark data colour coded for seasonal quality, along with additional information on land management trends (e.g., stock number adjustments), assist with interpreting whether changes in rangeland condition are due to natural variations or management.

   We recommend that providers of monitoring information:

   - graph and report time-mark continuums for attributes and indicators where time-marks have been colour coded using the seasonal quality matrix approach used in *Rangelands 2008*;
   - report on whether changes in measured attributes or indicators are statistically significant using simple methods such as mean comparison tests (e.g., *t* statistics) and fitting of trend lines (e.g., linear regressions). Although more complex statistical models can be applied to monitoring data, their use is usually unnecessary to document significant changes; and
   - report whether available site-based monitoring data adequately samples the area of interest. For remote sensing-based monitoring data, which completely covers an area with pixel or grid-based data, information on

variability in the indicator data across all the pixels in the area should be reported.

Monitoring programs

4. Manuals for monitoring rangelands use similar steps and methods. The first step is to have stakeholders define their goals for monitoring. Second, guidelines are provided on selecting and establishing ground-based sites with the emphasis usually placed on monitoring for pastoral production purposes rather than on biodiversity conservation or other land use goals. Recent manuals included remote sensing-based monitoring methodologies because they help overcome the limitation that site-based monitoring can only feasibly cover a small total area.

   We recommend that:

   - ground-based monitoring sites be designed to accommodate monitoring for multiple purposes (e.g., pastoral production, biodiversity conservation) and be established to cover a wide range of locations in the rangeland area of interest. The latter is particularly important for reliably detecting changes in areas such critical habitats and 'nick-points' (i.e., areas where active rills and gullies are altering the hydrology and vegetation across the landscape); and
   - remote sensing-based methods be used to monitor those indicators that can be derived from satellite sensor data and that can be verified by ground-based data.

5. Rangeland monitoring datasets provided to ACRIS for the *Rangelands 2008 – Taking the Pulse* report had a number of strengths for detecting and reporting changes over time and space. Our evaluation of the 'Landscape function' and 'Sustainable management' themes in this report identified a notable weakness in reporting on indicators that could be readily compared across all rangeland jurisdictions. In particular, future ACRIS reports would be improved by having available robust and consistent landscape function data. We recognise that the monitoring programs of State/NT partners in ACRIS must meet individual jurisdictional requirements, but consistent national reporting of critical indicators such as functional state does require that core data about changes in vegetation and soils are available across the entire rangelands.

   We recommend that State/Territory providers of monitoring data aim to:

- explore ways of obtaining more robust and consistent data on landscape function indicators. For example, perennial ground cover vegetation (or lack of it) can be reliably derived from remote sensing-based data (except where tree/shrub canopies hide the ground surface from satellite sensors). This indicator strongly relates to a number of biophysical functions including how landscapes function to retain water and soils, and sustainably produce forage for stock.

We recommend that the ACRIS assist with:

- developing and testing remote sensing-based methods for acquiring landscape function data. For example, the landscape leakiness index has been developed to indicate changes in how well catchments are retaining water and soils. It is based on remotely sensed ground cover and digital elevation data. This index is currently being evaluated for catchments in the Burdekin but needs to be developed further and tested in other rangeland catchments (e.g., in regional pilot studies).

1. Introduction

Areas around the globe with arid, semi-arid and dry sub-humid climates, and where topography and soils are unsuitable for broad-acre farming, are generically referred to as rangelands because these areas are traditionally used for pastoralism (Harrington et al. 1984). Roughly half the globe's land surface is rangeland, about 67 million km^2 (WRI 1986). As a traditional and extensive land use, pastoralism tends to mask the many other more local intensive uses of rangelands such as mining and tourism (Williams et al. 1968). Protected areas within rangeland regions also serve to conserve biodiversity, and some areas are used for hunting and gathering of bush foods and firewood. Wildfires and intentional use of fire also affect rangelands. All these multiple uses affect rangelands and cause changes to various degrees and extents.

Detecting and understanding changes to rangelands caused by different land uses requires measuring and monitoring those attributes of rangelands defining how well they are functioning as ecosystems relative to what is expected from areas largely unaffected by land use (reference areas). In Australia, monitoring the functional status of rangelands is typically the responsibility of State and Territory Government land management department personnel. In some rangeland jurisdictions, this responsibility has now devolved to catchment management authorities or natural resource management boards. Rangeland monitoring is conducted in collaboration with local land managers. Large areas of Australia's rangelands are leased from governmental jurisdictions, and specific goals for managing leasehold lands are stated in leasehold agreements. The overall goal is to maintain rangelands in good condition (i.e., sustaining their basic biophysical and socio-economic functions relative to reference areas; Friedel et al. 2000, Whitehead et al. 2000, Pyke et al.

2002; also see Section 3 in this paper). Many management actions for rangelands in leasehold and in private ownership are regulated by State/Territory Government legislation on issues such as vegetation clearing, weed and feral animal control, water use, and mineral extraction (e.g., Neldner 2006).

To collate and synthesize rangeland monitoring information at national and regional scales and to report these integrated results to policy makers, land managers and others with an interest in rangelands (stakeholders), the Australian Collaborative Rangeland Information System (ACRIS) was formed as a partnership between those government organisations responsible for rangeland management. The ACRIS was proposed as part of the first major report for stakeholders: *Rangelands – Tracking Changes, the Australian Collaborative Rangelands Information System* (NLWRA 2001). It has subsequently produced *Rangelands 2008 – Taking the Pulse*, which brought together disparate data sets from 1992 to 2005 to report change for a number of biophysical and socio-economic themes (Bastin and ACRIS-MC 2008). The ACRIS Management Committee commissioned this paper to evaluate these reports, recommend possible ways to improve future reporting, and review the literature on what rangeland condition means to different stakeholders and how they assess condition relative to their values and land use goals. As might be expected, assessments of rangeland condition for the same area often markedly differ, and there is a need for an approach to help resolve conflicting statements about the condition of a rangeland area (e.g., Pringle et al. 2006). We recommend a participative approach (e.g., workshops) to help reconcile conflicting statements about the condition or health of a rangeland area (see Section 7.2).

Assessing changes in rangeland condition requires an understanding of rangelands as ecological and social systems – their 'states-and-transitions', stability, resilience – and what rangeland condition means in relation to these concepts (Friedel et al. 2000). These authors emphasize the need to understand the difficulties arising in detecting significant changes and trends given the great spatial and temporal variability inherent in rangelands. In this paper, we place rangeland condition or health within the context of current ecological and social concepts and, using ACRIS datasets, illustrate methods for detecting statistically significant changes and trends in the functional state of rangelands with respect to spatial and temporal variability. Finally, we recommend some ways of improving ACRIS data analysis and reporting of changes in functional state, which aim to help stakeholders evaluate rangeland condition.

Changes in rangeland condition have been described as 'desertification' when they are in a direction away from the positive values of pastoral production and biodiversity conservation. Here, we will not review the broad topic of rangeland desertification (about 300,000 'hits' on Google) because this has been done elsewhere (Reynolds and Stafford

Smith 2002). However, we note that many of the studies on desertification have been concerned with assessing the degree of desertification as a change in rangeland condition (Friedel et al. 2000).

2. Rangeland uses and stakeholders

As noted in the Introduction, rangelands are used by groups of people for many purposes. A few of these uses, among many, include people practicing traditional and commercial pastoralism, to those conserving landscapes and habitats for biota and tourism, to miners exploring for, and extracting, minerals, to Indigenous people practicing traditional customs and hunting, gathering and trading resources. In this paper we use the term 'stakeholder' to denote these groups of people who have an interest in the rangelands. This definition of stakeholder includes those groups of people who may not live within a rangeland area, but who have a concern for their condition or health including, for example, overseas tourists, members of global and national land conservation organisations, which typically are non-government organisations (NGOs), and governmental land managers and policy-makers who typically live in capital cities.

These stakeholder groups will have different visions and goals for the use of rangelands (e.g., Garnett et al. 2008). Stakeholder visions and goals for using rangelands are ideally defined in participatory settings, such as workshops. For example, a series of workshops were held to define stakeholder visions and goals for the use of rangelands in the tropical savannas of northern Australia (Whitehead et al. 2000). Although Aboriginal, conservation and pastoral stakeholder groups shared a similar general vision centred on having healthy country, their specific visions and objectives for use of these rangelands differed considerably, as might be expected. Because of their different visions and objectives, the expectation is that each stakeholder group will focus on different attributes and indicators being monitored to define the functional state of a rangeland system. Using the same reported information about the status of the system, each stakeholder group will evaluate its 'condition' or 'health' quite differently, resulting in conflicting statements which need to be resolved. We will return to these issues after considering what the term 'rangeland condition' means.

3. Rangeland condition

3.1. An analogy to human health

The term 'condition' in standard dictionaries means "state of being" or "health". In human health terms, poor health is a 'state of being' in reference to good health, which is typically assessed in terms of easily measured indicators such as body temperature, blood pressure, and resting heart rate. Rangeland condition is analogous. It is a human perception of the state of health of a rangeland area in reference to an area perceived to be in a state of good health – a reference or benchmark site (Friedel et al. 2000). This notion of assessing rangeland condition or health relative to a benchmark is not new, being applied in the 1940s to assessing changes in forage plant composition away from a theoretical 'climax community' composition (Humphrey 1949, Dyksterhuis 1949).

The state of the benchmark site, and other rangeland sites of interest, can be defined by a set of easily measured indicators related to, for example, production, conservation and aesthetic values (Keith and Gorrod 2006). Given such indicators, the state or condition of the rangeland site is judged by people (stakeholders) to be in a given state of health relative to the benchmark site. This health analogy is widely used, especially in the United States, and it has proven useful for talking about the state of rangelands. Some authors caution about "pushing the analogy too far" (Whitehead et al. 2000), and others (West *et al.* 1994) recommend against the use of the term 'health' because it is too value-laden and its meaning will change as society's views and values change. In this paper, we argue for the use of rangeland 'health' while acknowledging that assessing it is a value judgment.

3.2. In relation to landscape function and integrity

Rangeland health – the condition or state of the land – has been defined simply as "the status of the soil, water and biological resources in rangeland ecosystems" (Pyke et al. 2002, 2003). A more comprehensive definition of rangeland health is "the degree to which the integrity of the soil, vegetation, water and air, as well as the ecological processes of rangeland ecosystems, are balanced and sustained" with integrity meaning "the maintenance of the functional attributes characteristic of a locale, including normal variability" (SRM 1999), which is a concept that is known to be important in rangelands (West et al. 1994). Rangeland health viewed as functional integrity relates it to biodiversity across landscape scales, from local to regional (Ludwig et al. 2004), where simple indicators of the intactness of vegetation structure and function have been demonstrated as being significantly related to the diversity of birds, invertebrates and plants (Karfs and Fisher 2002, Landsberg et al. 2003, Fisher and Kutt 2006).

Rangeland condition or health is also linked to similar concepts embedded within the broader constructs of landscape function and health, where a landscape is an area of interconnected ecosystems (Turner et al. 2001). In Australia, healthy landscapes are viewed as having a number of important functional attributes (Tongway and Ludwig 1997a, Ludwig and Tongway 2000, Whitehead et al. 2000). These attributes include having the capacity to (i) maintain basic processes such as capturing energy, retaining water and cycling nutrients; (ii) provide habitats (food, shelter) for sustaining populations of all native plants, animals and microorganisms at appropriate scales in time and space; and (iii) provide people their cultural, spiritual, aesthetic and livelihood needs. In the USA similar landscape functional attributes are described (USGS 2002, NRCS 2003) including (i) site/soil stability, which is the capacity of a site (e.g., a rangeland watershed) to limit redistribution and loss of soil resources by wind and water; (ii) hydrologic function, which is the capacity of a site to capture, store and safely release water from rainfall, run-on and snowmelt (where relevant), to resist disturbances to this capacity, and to recover this capacity following degradation; and (iii) integrity of the biotic community, which is the capacity of a site to support communities given normal variability, to resist loss of this capacity due to disturbance, and to rapidly recover capacity following disturbances. Gorrod (2006) adds “the capacity of a site to provide habitat for all the indigenous species that may reasonably be expected to use it”. We will return to a discussion of indicators or surrogates for these functional attributes after we relate rangeland health to other ecological and socio-economic concepts.

3.3. In relation to stability, resilience and state-and-transition

Rangeland ecology and management is now based on a number of ecological and socio-economic concepts and paradigms. The concept of rangeland systems not being in one stable state of equilibrium (Wiens 1984, Westoby et al. 1989), but having multiple dynamic stable states (Holling 1973, Walker et al. 1981), is now widely accepted (Friedel et al. 1991,2000; Briske et al. 2003, 2005). Details are provided by these authors, but basically a rangeland system may occur in a number of different states within one stable state, with transitions between these states caused by disturbances (e.g., intense grazing), and it may also cross a threshold into another stable state and have multiple ‘states-and-transitions’ within this second stable state (Briske et al. 2005, see their Fig. 5). The concept is that management actions (e.g., reduced grazing) can usually reverse changes within each stable state, but not back across the threshold – major restorative actions are required.

These dynamics are also embedded within the concept of system resilience, a topic that has been widely explored for both ecological and socio-economic systems (e.g., Gunderson 2000, Fernandez et al. 2002, Gunderson and Holling 2002, Walker et al. 2006). Further, an understanding of these concepts can lead to improving rangeland health by repairing those that have been damaged (e.g., Noble et al. 1997, Whisenant 1999, McCullough and Musso 2004).

4. Monitoring the functional state of rangelands

Rangeland scientists around the world are working on ways to more effectively and efficiently stratify rangeland landscapes, measure their functional attributes (especially by using remote sensing), and evaluate their condition and trend relative to a greater variety of land use goals. Rangeland scientists are also exploring new ways to assist land managers improve the condition of landscapes.

Monitoring the functional state of rangelands, and evaluating their condition, occurs routinely in all States and Territories with rangelands in Australia. A number of procedures have been designed for monitoring rangelands and the development of new methods is an on-going process (e.g., Ludwig et al. 2007b). Rangeland monitoring protocols are merging globally as communications improve through the internet (e.g., www.srm.com, www.austrangesoc.com.au), international journals (e.g., *Rangeland Ecology & Management*, *The Rangeland Journal*), conferences (e.g., International Rangeland Congress) and exchange programs (e.g., Fulbright, Churchill, study leaves, sabbaticals).

As recently as the 1980s, rangeland monitoring procedures in different countries remained largely independent, as evident from the papers presented at an international workshop on evaluating grazing lands (Siderius 1984). Since then, a general protocol involving similar steps for monitoring rangelands has emerged (e.g., Tongway and Hindley 2004, Pellant et al. 2005, Herrick et al. 2005, Gibbons and Freudenberger 2006). Here in Section 4, we describe a seven-step procedure (Box 4.1) that builds on the protocols and steps described by these authors. We only briefly describe these seven steps, but then expand our discussion of three issues of particular interest to the ACRIS-MC. In Section 5 we discuss the issue of dealing with scale and spatial variability, especially in reference to using ground-based sites and remote sensing-based technologies. In Section 6 we illustrate the application of methods for detecting statistically significant changes and trends in the functional state of rangelands. Then in Section 7, we discuss the issue of how to resolve conflicting rangeland condition statements made by different stakeholders.

Box 4.1. A seven-step procedure for monitoring rangelands

Step	Action
1	clearly define the goals and objectives for monitoring
2	establish the appropriate spatial and temporal scales for monitoring
3	select the appropriate attributes and indicators to measure at relevant scales
4	precisely measure attributes and indicators over the area of interest and repeat over time
5	rigorously analyse these data for significant differences over space and in time
6	report these analyses to the different stakeholder groups so that they can evaluate rangeland condition with respect to their goals and values
7	Periodically assess the value of reported indicators to stakeholders and continue to monitor the valued indicators to detect and assess any future changes in trends, especially following any rangeland management or policy actions taken by stakeholders

4.1. Step 1: Defining stakeholder goals for monitoring

This first step is perhaps the most crucial because the success or failure of any management action to improve rangeland condition can only be assessed against clearly defined goals and objectives (Brown et al. 1998, Friedel et al. 2000), which relate to the visions that stakeholders have for their use of rangelands. These visions vary, but different stakeholders also have many goals in common (Table 4.1), such as maintaining healthy savannas (Whitehead et al. 2000). Across the rangelands of Australia, specific goals and objectives that stakeholders have for achieving their visions varies with region and land use. A full discussion of their visions, goals and objectives are beyond the scope of this paper, but some examples are provided in the ACRIS Report “Rangelands 2008 – taking the pulse” (Bastin and ACRIS-MC 2008).

Table 4.1. Visions held by different stakeholder groups for use of rangelands based on a series of workshops on defining the health of the savannas and grasslands across northern Australia. Stakeholder groups are listed alphabetically.

Stakeholder group	Vision
Conservation	a diversity of landscapes and living things
Indigenous	clean country and healthy people
Mining	access to minerals and local labour
Pastoral	sustained production and land stewardship
Tourism	varied and intact landscape features

4.2. Step 2: Establishing spatial and temporal scales for monitoring

When specifying their visions, goals and objectives, individual stakeholders relate these to their specific rangeland area (scale) of interest, but groups of stakeholders operate across multiple scales (Table 4.2). For example, individual pastoralists are concerned with their livestock and will focus on managing paddocks within properties. Groups of stakeholders will focus on improving rangelands at paddock, property and regional (or catchment) scales; these groups include, for example, the Natural Resource Management (NRM) and Catchment Management bodies formed by Commonwealth and State/Territory Government initiatives.

Spatial scale. When selecting locations to establish site-based monitoring to achieve land management goals, the rangeland area of interest needs to be clearly defined because within a broader general area smaller specific areas (e.g., adjacent land units) can be in very different functional states and have different factors driving changes in these states (e.g., Pickup et al. 1994). Rangeland monitoring sites have typically been located for pastoral production goals (i) in preferred grazing areas, (ii) at a specified distance range from water, (iii) adjacent to tracks or fences, and (iv) on more stable mid-slopes. If goals include biodiversity conservation in addition to pastoral production, then monitoring sites need to include locations of critical habitat (e.g., Karfs and Fisher 2002). If goals include avoiding soil erosion or improving areas already eroding, then monitoring sites need to include areas prone to erosion or areas in the landscape where rill and gully-cutting are already affecting vegetation (referred to as ‘nick-points’; Pringle and Tinley 2003, Pringle et al. 2006). If remote sensing-based monitoring is being used to provide greater spatial coverage, then finer-scale (smaller-pixel) imagery may be needed to monitor smaller areas of critical habitat or landscapes with 'nick-points' and to compliment site-based monitoring.

Temporal scales. There is no doubt that long-term and frequent monitoring provides greater information for detecting and understanding changes in rangelands (see Section 6). However, there is always the trade-off between the desire for greater temporal (and spatial) coverage and costs. There are no simple answers to such cost-benefit trade-offs, but must relate to clearly defined goals for monitoring rangelands.

Establishing reference sites. To evaluate the contributions of natural versus management effects on changes on monitoring sites, reference sites or benchmark areas should available where management effects are minimal (e.g., Bastin et al. 2003). Benchmarks can include locations where grazing intensity is low such as areas remote from water and within long-established conservation reserves. However, such areas are not always available in rangeland regions. An alternative is to define hypothetical reference areas where stakeholders use their knowledge of a rangeland region to characterise a benchmark area in terms of being in a highly functional state as defined by biophysical attributes. This 'scenario analysis', expert knowledge approach has been widely used to characterise hypothetical rangeland regions (e.g., CAZR 2000, Stafford Smith 2000, Maru and Chewings 2008).

Table 4.2. Combinations of stakeholder groups and areas of interest across an increasing spatial scale. These combinations were defined in a series of workshops on defining the health of the savannas (Whitehead et al. 2000). Stakeholder groups are listed alphabetically. Codes for key attributes and indicators of functional status (e.g., C1, C2) are tabulated here, but these codes are defined and discussed in step 3.

	Area of interest (extent or scale)			
Stakeholder Group	Paddock/ Place	Property/ Clan Estate/ Protected Area	Bioregion Catchment Language Group	Rangeland-wide
Conservation	C1, C2, C3	C1, C2, C3	C1, C2, C3	C1, C2, C3
Indigenous	I1, I2, I3	I1, I2, I3, I4	I1, I2, I3, I4	I1, I2, I3, I4
Mining	M1	M1, M2	M1, M2	M1, M2
Pastoral	P1, P2	P1, P2	P1, P2	P1, P2
Tourism	T1	T1, T2	T1, T2	T1, T2

Various frameworks have proven useful for defining such hierarchical scales of interest. One of the most widely applied and useful frameworks is 'States-and-Transitions', which defines areas of vegetation as being in different 'states' that are linked by 'transitions', which are defined as the factors and processes causing changes in 'state' (Westoby et al. 1989). Since its inception in the late 1980s, the success of this framework has been attributed to its capacity to conceptualise landscape dynamics that are continuous or discontinuous and reversible or non-reversible (Briske et al. 2005).

The 'state-and-transition' framework has also been successful as an effective communication tool between different stakeholders, who also find it a useful planning tool by enabling them to set clearly defined land management targets (Bestelmeyer et al. 2003). For example, the VAST (Vegetation Assets, States and Transitions) framework has been applied in Australia to explore the impacts of human landscape modifications on land condition (Thackway and Lesslie 2006). This framework defines 'states' of land (e.g., from fully intact natural vegetation to highly modified farmlands), and the likely causes of change (transitions) between states (e.g., tree clearing, thinning). Areas of natural vegetation are used as reference areas or benchmarks against which changes due to land use and land management practices can be evaluated.

4.3. Step 3: Selecting appropriate attributes and indicators

It is crucial that the key stakeholders in the rangeland area of interest be involved in selecting the appropriate attributes and indicators to provide the information they require to assess condition. For example, the attributes and indicators considered important for assessing the health of Australia's savannas are listed in Table 4.3. This example lists those indicators that can be obtained by ground-based observations at smaller scales and by remote-based methods at larger scales including, for example, data obtained from satellite sensors, postal surveys, and interrogation of State/Territory or National databases.

The list of attributes and indicators in Table 4.3 is not intended to be exhaustive, but indicative of the participative approach for selecting indicators by diverse groups of stakeholders who worked together in workshop settings. This participatory approach has also been used in the Kalahari grazing lands of South Africa, where a participatory process involving pastoralists, extension workers and scientists was used to identify relevant indicators of rangeland condition (Reed and Dougill 2002). It was felt that this process provided a set of useful indicators, some having an 'early-warning' capacity. The participatory process also revealed some interesting, but expected, differences in stakeholder values. Pastoralists largely focussed on livestock and vegetation indicators whereas others were more likely to include soil, wildlife and socio-economic indicators.

Table 4.3. A list of the attributes and indicators considered vital for defining healthy landscapes by stakeholder groups operating across multiple scales in the savannas of northern Australia (simplified and adapted from Whitehead et al. 2000). These are the coded attributes and indicators used in Table 4.2 to illustrate how they apply across multiple scales.

Code	Attribute	Indicator
C1	Capacity to capture and retain water and soil resources, hence, promote growth in plant, animal and microbial populations	Perennial vegetation ground cover
C2	Provision of native populations with their habitat requirements, such as suitable soils, foraging and nesting sites, etc.	Complex (patchy) vegetation cover
C3	Exclusion or low populations of exotic weeds and feral animals, such as prickly acacia, camels, cats, foxes, donkeys, horses, etc.	Abundance and spread: weeds-ferals
I1	Reliable availability of basic resources to meet subsistence and trade needs, for example, clean water and healthy foods	Available bush foods and clean water
I2	Intact landscape elements with plants, animals and people in their customary places and in an undamaged state	Undamaged country: customary sites
I3	Facilities and services available to maintain the physical and mental health of people living on country	Quality of human health services
I4	Capacity to trade resources and maintain communications and ceremonies between neighbouring estates and language groups	Quality of transport services
M1	Access to explore for available mineral resources across the rangelands	Availability of exploration leases
M2	Capacity to extract mineral resources in remote localities by having available a local work force	Infrastructure and a source of labour
P1	Capacity to capture and retain resources (see C1)	Perennial vegetation ground cover
P2	Reliable supply of 3P forage plants: palatable, perennial and productive	Palatable forage cover (biomass)
P3	Low populations of exotic weeds and feral animals (see C3)	Abundance and Spread: weeds-ferals
T1	Maintained populations of native plants and animals (see C2)	Complex (patchy) vegetation cover
T2	Access to guides and rangers to explain landscapes to tourists	Source of labour

A number of the attributes listed in Table 4.3 relate to the amount of perennial vegetation forming ground cover as an indicator of the functional state of rangelands in terms of the capacity of the system to capture resources (water, soils, litter, seeds) and provide habitat for populations of plants, animals, and micro-organisms. The importance of ground vegetation and other surface obstructions such as soil crusts, logs and rocks for protecting soil surfaces and preventing excessive soil erosion on rangelands was documented over 20 y ago (Gifford 1984). Recently the importance of how protective cover is spatially configured within rangeland watersheds has been emphasized (e.g., Ludwig et al. 2005, Bartley et al. 2006).

4.4. Step 4: Measuring attributes and indicators

Numerous methods are available for measuring different attributes and indicators of the functional state of rangelands, and our intent here is not to repeat details on methods provided in available rangeland monitoring manuals (Box 4.2). In addition to these manuals, a number of universities offer classes to train students in methods of rangeland inventory and monitoring. For example, in the USA, Colorado State University, Kansas State University, New Mexico State University, Texas A&M University, Utah State University and others offer such classes. In Australia, the Gatton campus of the University of Queensland offers a class on “Rangeland Monitoring and Adaptive Management”.

Obviously, different methods of measuring attributes and indicators of the functional state of rangelands vary greatly in their ease of use, precision and accuracy (see definitions in Section 6.1). These characteristics have been well assessed for a number of methods (e.g., Holm et al. 1984, Friedel and Shaw 1987), but less so for others. A review of monitoring methods is beyond the scope of this paper, however, we note here that methods of rapid appraisal of indicators (e.g., visual estimates of ground cover), which serve to provide an efficient coverage of broad areas, may lack the precision and accuracy required to detect significant changes and trends in these indicators (Pickup et al. 1998b).

4.5. Step 5: Analysing attributes and indicators data

Rangeland monitoring data, such as that compiled in the ACRIS and reported in “Rangelands 2008 – taking the pulse” (Bastin and ACRIS-MC 2008), includes information for a number of different types of biophysical and socioeconomic indicators. There are questions in common to all these indicators: are they significantly changing over time and, if so, how (declining, increasing) and at what scale (e.g., over a paddock, a property, a region)? There are a number of statistical methods available for addressing these questions, which vary from simple *t* tests and linear regressions to more complex repeated measures analysis and multiple regressions. In Section 6, the application of these methods will be illustrated and discussed using datasets from the ACRIS.

Box 4.2. A selection of manuals for conducting rangeland monitoring

Title	Reference or source
A rancher's guide to monitoring rangelands	Sanders 2006
Assessment of Soil Condition of Tropical Grasslands	Tongway & Hindley 1995
Landscape function analysis: procedures for monitoring and assessing landscape with special reference to minesites and rangelands	Tongway & Hindley 2004
Monitoring manual for grassland, shrubland and savanna ecosystems. Volume I: Quick start	Herrick et al. 2005
Monitoring manual for grassland, shrubland and savanna ecosystems. Volume II: Design, supplementary methods and interpretation	Herrick et al. 2005
National range and pasture handbook, Revision 1. Chapter 4, Inventorying and monitoring grazing land resources.	NRCS 2003
Repairing damaged wildlands: a process-orientated, landscape-scale approach	Whisenant 1999

4.6. Step 6: Reporting results of analyses on indicators

Stakeholders require clear and concisely presented information on the state of rangeland indicators in order for them to evaluate the health or condition of the rangeland area of interest based on their visions and values for their use of the area. For example, in "Rangelands 2008 – taking the pulse" (Bastin and ACRIS-MC 2008), results on changes in rangeland attributes and indicators are grouped and reported by themes such as landscape function, sustainable management and biodiversity. Information is usefully presented in the form of maps and time-trace graphs. Information in maps is typically displayed at regional and sub-regional scales. Boundaries of regions are based on an interim bio-regionalisation of Australia into 85 bioregions and 403 sub-regions (IBRA 2008). The rangelands include 52 bioregions or parts of bioregions, as illustrated by a map in Chapter 2 of "Rangelands 2008 – taking the pulse". The Rangelands 2008 report extensively uses graphs to illustrate changes in indicators over time. Photographs are often used to illustrate changes documented in graphs and maps.

Map styles. A review of the different styles of maps used in "Rangelands 2008 – taking the pulse" to enhance the communication of information on changes in the functional state of Australia's rangelands to stakeholders will not be repeated here, except to note the usefulness of maps that illustrate types of change with colour-codes (Fig. 4.1). In the two example maps showing remote sensing-based changes in cover, over the monitoring period (1997-2000), colours are interpreted as: red = areas with low cover that declined in cover, yellow = areas with initial high cover that declined, blue = areas with low cover that increased in cover, cyan = areas with high cover that increased, and green = stable areas. The use of different colours to designate degrees and directions of change in space and time, and at different scales, has proven useful in studies assessing vegetation changes in farmlands and rangelands (e.g., Karfs et al. 2000, Newell et al. 2006, Wallace et al. 2006). We also note that the choice of colours used in these maps should consider colour-blind people.

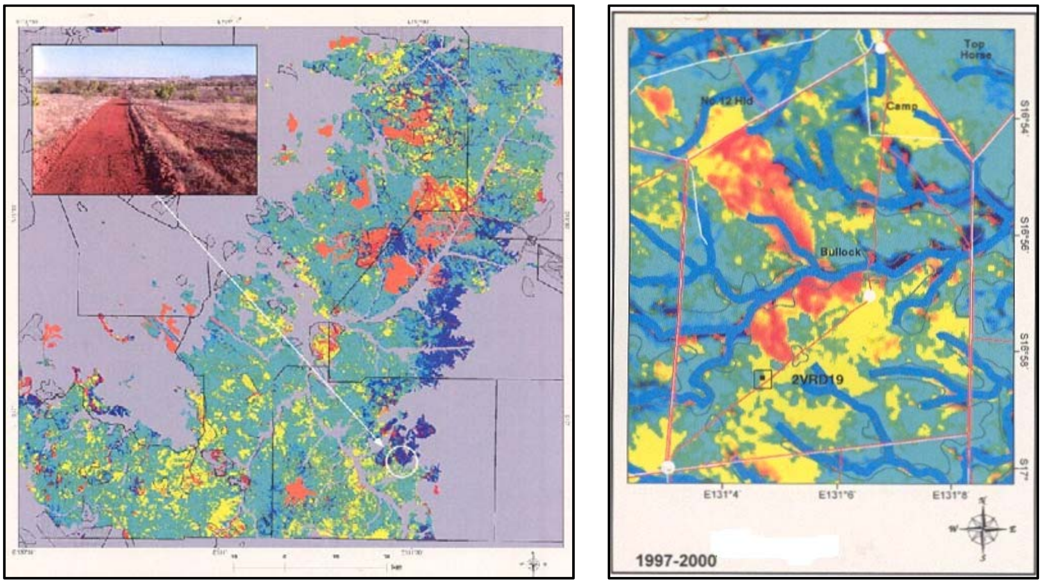


Figure 4.1. Examples of maps using colour-codes (see text) to illustrate changes in cover over rangeland areas at regional scales (left) and paddock scales (right). Details on methods of estimating cover and mapping changes are provided in Karfs et al. (2000), Wallace et al. (2004), Karfs and Trueman (2005).

Time trace graphs. Documents reporting on the functional state of rangeland systems also extensively uses graphs to illustrate changes in measured attributes and indicators (e.g., NLWRA 2001, Bastin and ACRIS-MC 2008). Different styles of graphs have been used, including simple line-scatter plots, bar charts and pie charts. Examples of these graphs will not be repeated here, except to illustrate one style of graph that lends itself to presenting information in a way that can be usefully interpreted, and then incorporated into a framework for evaluating rangeland condition or health. The style of graph in Fig. 4.2 illustrates spatial, temporal and co-variate information. The three panels illustrate: (A) the location and scale for the presented monitoring data (such maps are often presented as separate figures); (B) time-traces for changes in an indicator, here, mean ($\pm$ standard error) perennial vegetation ground cover for a set of monitoring sites versus that expected for a set of reference sites; and (C) the data for a factor that co-varies with cover (deviations in annual rainfall relative to the long-term median) that may be useful for interpreting (in part) a possible cause for the changes in the indicator (note that cover declined after or during years of lower rainfall on monitoring sites but changed little on reference sites, where cover gradually increased from 1993 to 2004). For this example graph, we used information available in two ACRIS reports (NLWRA 2001, Bastin and ACRIS-MC 2008) and from Karfs and Trueman (2005).

Time-mark continuum graphs. Time trace data for monitoring and reference sites, such as that illustrated in panel B of Figure 4.2, could be interpreted directly by stakeholders who wish to evaluate the condition or health of the rangeland area relative to their visions and values. However, to aid interpretations and evaluations of rangeland condition, we recommend that these time trace data be transferred to simple one-line continuum graphs (Fig. 4.3) that build on the conceptual framework that the functional state of a rangeland area of interest varies in time along a continuum from a fully functional state to a totally dysfunctional state (Tongway and Ludwig 1997b, Gibbons and Freudenberger 2006). The fully functional end of the continuum can be defined using information from reference sites. For example, the maximum perennial plant ground cover observed was about 90% (panel B of Fig. 4.2) based on sites in the region that had not been grazed by livestock for about 30 y (Bastin et al. 2003); this value of cover was used to define the 'fully functional' end of the continuum (Fig. 4.3A). The totally dysfunctional end of the continuum was defined theoretically as a landscape with no (0%) perennial plant ground cover; such landscapes would have little capacity to retain water and soil sediments in runoff or wind-driven soil particles. These extremes can be illustrated by including photographs taken at photo-points on monitoring and reference sites within the rangeland area (Fig. 4.3B and C). Along the continuum (Fig. 4.3A), the measured values for the cover indicator are noted as time-marks where, for example, the state of cover in 2004 (83%) which was measured after the growing season (the latest monitoring time), was approaching that expected for a fully functional system.

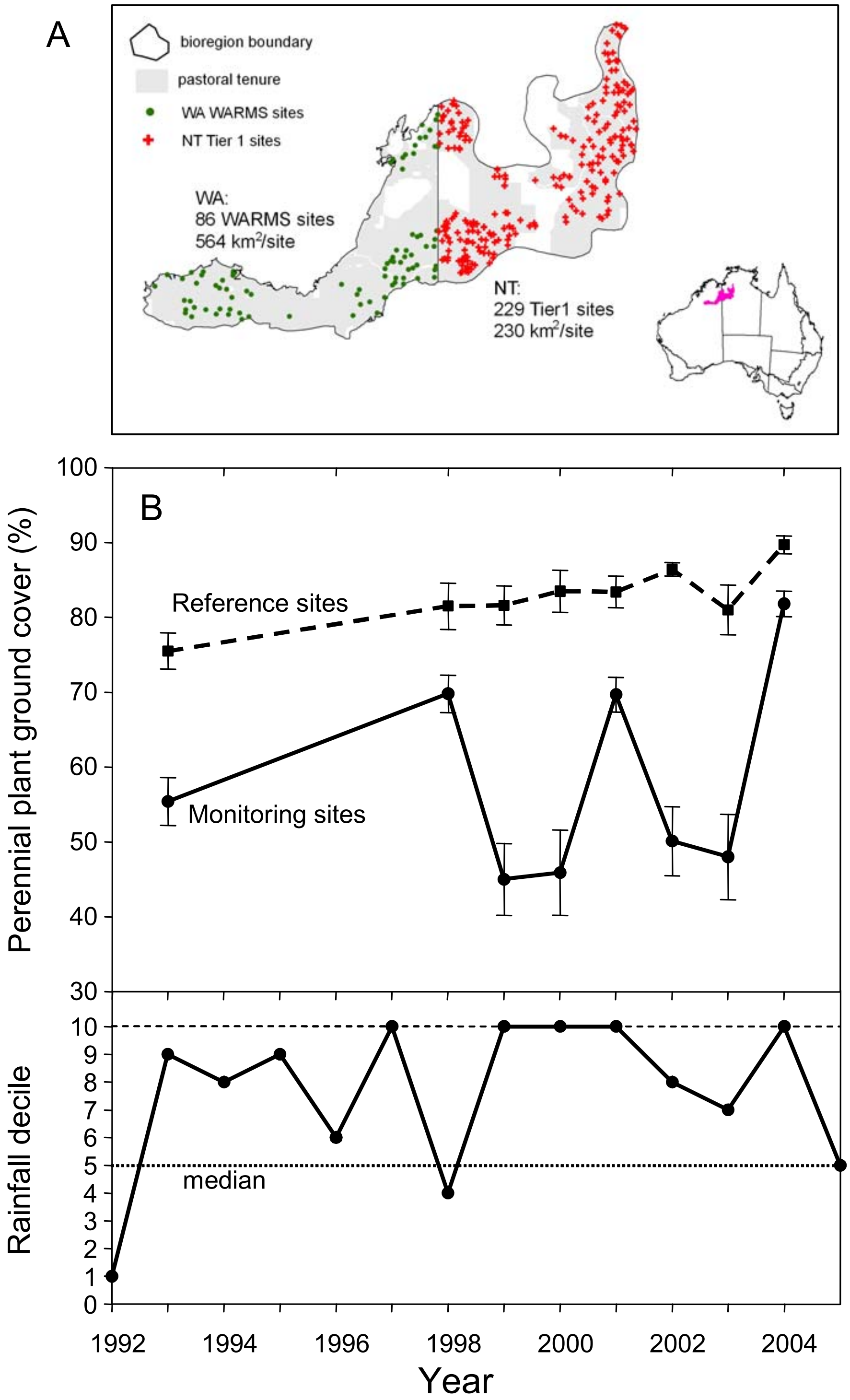


Figure 4.2. A style of graph useful for reporting rangeland monitoring information on (A) location of sites, (B) changes in an indicator on both monitoring and reference sites, and (C) data for an environmental factor (rainfall) that co-varies with the indicator.

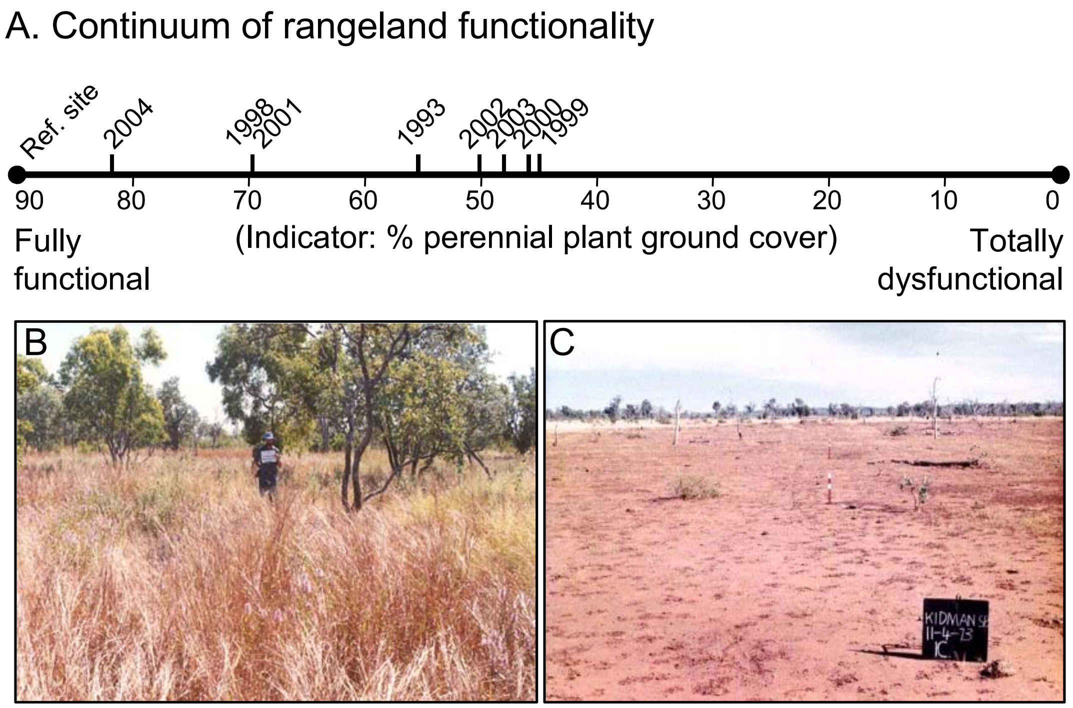


Figure 4.3. A. Time-marks along a rangeland functionality continuum based on the mean position of a plant cover indicator measured on sites monitored in 1993 and from 1998 to 2004. Photographs illustrate rangeland monitoring sites within the region that are typical of being in a (B) highly functional state and (C) dysfunctional state for perennial plant cover.

Changes in the position of time-marks for an indicator along its continuum (ranging from a fully functional value to a totally dysfunctional value; Fig. 4.3A) may be due to environmental or management effects and, most likely, their interaction. To help interpret changes, time-mark positions can be 'flagged' for environmental (e.g., 'seasonal quality') and management effects.

Time-mark continuums: colour coded for seasonal quality. For examining environmental effects, each time-mark can simply be flagged with a colour code to denote whether 'seasonal quality' was above average (blue), below average (red) or within a normal range (black) over the time period. Seasonal quality is a term used to define the climate of specified time periods relative to a long term record. It can be defined by various climate or climate-related products including rainfall, modelled pasture biomass and remote sensing-based vegetation greenness, and time periods can be varied to suit different regions and purposes (e.g., in the summer rainfall regions of northern Australia, the 'rainfall' year is typically defined by the year starting on April 1st but extending to March 31st in the following year; see details in Bastin and ACRIS-MC 2008, their Chapter 3, Climate Variability Theme).

Time marks colour coded for seasonal quality help illustrate how the amount of wet season rainfall prior to each assessment has affected the positioning of individual time marks on a continuum (Fig. 4.4). Note that no seasonal qualities were below average (red), only average (black) or above average (blue). For these example data, perennial plant covers varied from 45% to 83%, indicating medium to high functionalities over the monitoring period. The pattern of time-marks suggests considerable year-to-year 'noise'. Note how the 2001 time-mark position, which is towards the highly functional end of the continuum, may only reflect a period of above average seasonal quality because the two time-marks prior to 2001 (2000 and 1999) were more intermediate in position. Why was 1998 more functional than 1999, 2000, 2002 and 2003 when its seasonal quality was just average whereas these later years had above average seasonal qualities? An interpretation is that the positions for 1999, 2000, 2002 and 2003 may be more towards the dysfunctional end because grazing management effects were over-riding above-average seasonal qualities. What is the correct interpretation?

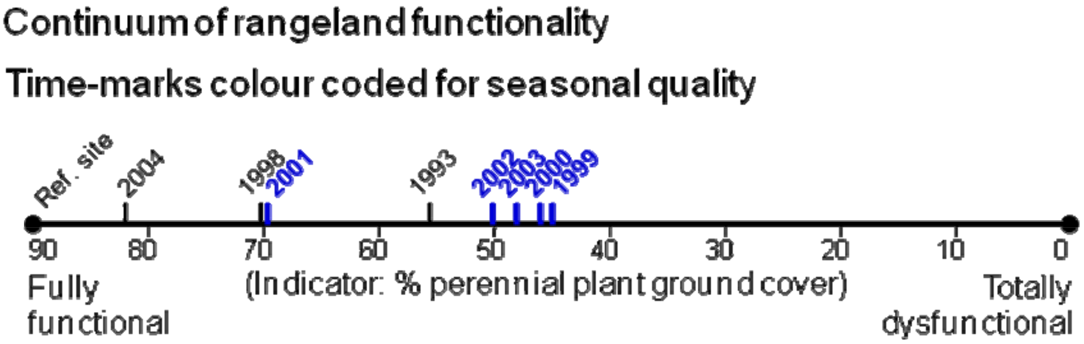


Fig. 4.4. An example of an indicator (perennial plant ground cover) with time-marks colour coded to denote seasonal quality (Blue = above average, Black = average).

Time-mark continuums: colour coded for management effects. To help interpret changes due to management effects, time-marks can also be colour coded in a different way to that described above based on a matrix combining 'seasonal quality' and direction of change (increase, decrease or no change) in values for the indicator. These values are means across the rangeland area of interest. Direction of change for each time period is relative to changes of more than 10% from prior periods. Changes are also evaluated relative to that expected based on values for reference areas which are assumed to be experiencing the same seasonal qualities as the monitoring sites. Time-marks along the continuum are colour coded as (see Box 2.2. 'Matrix: seasonal quality and direction of change' in *Rangelands 2008 – Taking the Pulse*):

Seasonal Quality	Direction of change in indicator		
	Decline	No change	Increase
Above average	XX	X	~
Average	X	~	√
Below average	~	√	√√

- red = a likely strong negative management effect because the indicator declined even though seasonal quality was above average,
- orange = a caution of a likely negative effect because either a decline occurred with an average seasonal quality or there was no change when seasonal quality was above average,
- bright green = a strong positive management effect because the indicator increased more than 10% even though seasonal quality was below average, and
- pale green = a likely positive management effect because either the indicator increased under average seasonal quality or remained unchanged when seasonal quality was below average.

This approach follows that illustrated for the Murchison Bioregion in Western Australia (Table 4.4) which is one of the 'focus' bioregions reported in "Rangelands 2008 - taking the pulse" (Bastin and ACRIS-MC 2008; see their Chapter 4, Focus bioregions).

For our example bioregion, time-marks were also colour-coded to denote when indicator values changed on monitoring sites relative to previous time periods and in relation to that expected based on values for reference areas (Fig. 4.5). Note that 1999 and 2002 are colour-coded red because the perennial plant ground cover index declined more than 10% relative to previous time periods and relative to that expected from reference site values, and this decline occurred even though seasonal quality was consistently above average for these two time periods; this infers a likely overall adverse effect of grazing management in the example bioregion. Also note in Fig. 4.5 that 2000 and 2003 are colour-coded yellow because mean values for the indicator remained little changed from their previous periods, 1999 and 2002, respectively. An increase would have been expected because seasonal quality was above average in 2000 and 2003; this signals a 'yellow caution' because any adjustments in grazing management applied in these periods did not increase perennial plant cover as might have been expected given above average seasonal qualities. No time-marks for plant growth periods were colour-coded bright green to infer positive management effects because seasonal qualities over the time periods evaluated in this bioregion (1993, and 1998-2004) were always average or above average. Although not applied here, time-marks could be varied (e.g., height of mark) to represent the amount of statistical variation in the mean value (e.g., standard error) for the indicator at each time period marked along the continuum.

Table 4.4. An example of using colour-codes to illustrate seasonally interpreted changes in a landscape function indicator (the Resource Capture Index) for WARMS sites in the Murchison bioregion in Western Australia (after Table 4.10 in Bastin and ACRIS-MC 2008). The red coloured cell denotes that the landscape function indicator declined when an increase would be expected given above-average seasonal quality; this infers a likely adverse grazing management effect. Orange signals a caution because a decline occurred when seasonal quality was average or there was no change when seasonal quality was above average. Bright and pale greens infer a likely positive management effect. In this example, 60% of WARMS monitoring sites showed a decline in the Resource Capture Index when seasonal quality was above average and 21% of sites showed an increase when seasonal quality was below average. The rationale and procedures used for the WARMS are described by Watson et al. (2007).

Seasonal quality	Number of sites	Decline. RCI < 0.90	No change. 0.90>= RCI <1.10	Increase. RCI >=1.10
Above average	94	60%	9%	32%
Average	141	55%	15%	30%
Below average	62	68%	11%	21%

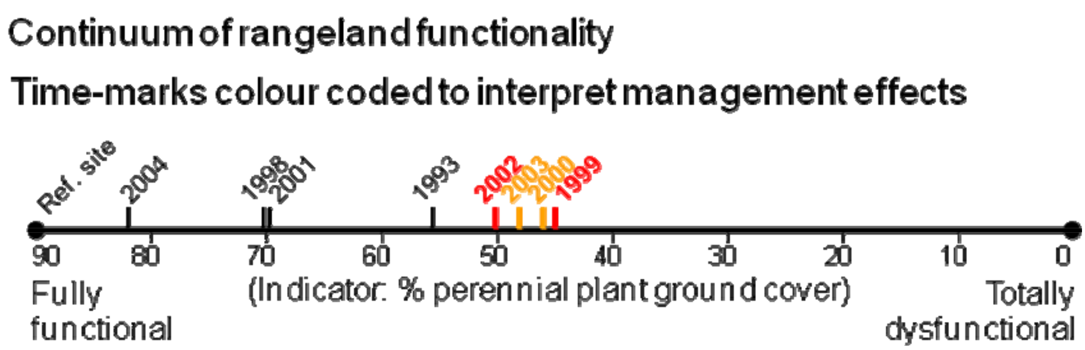


Fig. 4.5. For a landscape function indicator (perennial plant ground cover) time-marks along a continuum have been colour coded to help interpret management effects. Red time-marks infer a likely strong negative management effect and yellow implies a likely, but less strong, negative effect (see text for rationale of management effects).

In addition to being based on the concept of landscape functionality as a continuum, these time-mark graphs also build on other ecological concepts. For example, the functionality of arid and semi-arid landscapes in space and time as driven by rainfall events is encapsulated within the "trigger-transfer-reserve-pulse" framework (Ludwig et al. 1997). Shifts in time-marks along the continuum also reflect the resilience of a system, that is, its capacity to absorb and recover from disturbances without changing to a different state (Holling 1973). If disturbances trigger and drive a system into an alternative state, this can be described by using the "state-and-transition" framework (Westoby et al. 1989, Briske et al. 2003, 2005).

Here, we recommend using seasonal quality colour coded time-mark continuums for each attribute and indicator separately rather than attempting to combine them into various kinds of complex indices. Combining individual indices into complexes requires knowing how they are related such are whether their trajectories over time are additive. Such complexities have been reviewed by Gibbons and Freudenberger (2006), Tongway and Hindley (2004, see their Section on "Turning data into information") and Herrick et al. (2005, see their Chapter 17 on "Interpret results"). Although our recommended approach of examining individual indicators can result in a large chart with many colour-coded time-mark continuums, each continuum for an indicator represents readily interpretable information. Smaller charts are produced by combining attributes and indicators into fewer, but more complex indices, but the meaning of each index may be difficult to understand, which can limit their use by land managers (Andreasen et al. 2001). To reduce the size of seasonal quality coded time-mark continuum charts, rangeland attributes and indicators can be grouped into themes and presented as separate charts, or as partitions within larger charts.

As an example of a partitioned chart, Figure 4.6 illustrates time-mark continuums for three indicators grouped into two themes; these data are for the Ord Victoria Plains bioregion in northern Australia. The first time-mark continuum is for the landscape function indicator, perennial plant ground cover, which was described above. The second continuum is for an indicator of sustainable management (area burnt within the bioregion for the year). The reference for a fully functional state is based on information from a fenced exclosure site in the region that was ungrazed by livestock for over 30 years; this site has been judged, from an ecological view point, to be highly functional (Bastin et al. 2003). The dysfunctional end of this sustainable management continuum is less well defined and subject to debate. Here, we have hypothesised that if 50% of the bioregion is burnt every year this would lead to many highly dysfunctional landscapes, reflecting long-term unsustainable management in the region. Note that in 1998, 2003 and 2005 only about 5% of the bioregion burnt, which is likely to be sustainable, whereas in 1997, 2000, 2001, 2002 and 2004, over 20% of the region burnt, which would probably lead to dysfunctional systems in the long-term.

Another indicator in this sustainable management theme is the dust storm index, DSI_3 (McTainsh 1998, McTainsh et al. 2007), which indicates the yearly incidence of wind erosion across rangeland regions. Values for this indicator in the Ord Victoria Plains bioregion were consistently low (average about 1; Fig. 4.6), indicating that in general soil surface cover was high, as might be expected for this climatic region and given the wetter period this region experienced over the reporting period (bottom panel, Fig. 4.2). This low mean index value of about 1 is relative to maximum dust storm index values of 10 or more, which can occur in more arid regions with lower vegetation cover across the rangelands of Australia (e.g., maximum 'time-averaged' value of 8.44 between 1992 & 2005 for the Channel Country bioregion; Bastin and ACRIS-MC 2008, see their Chapter 3).

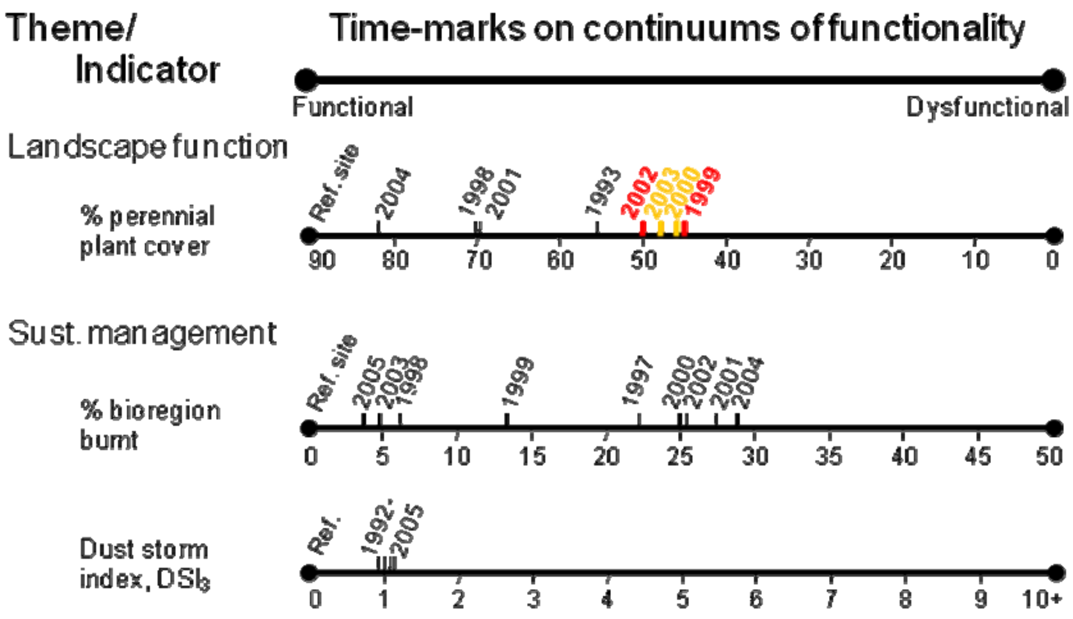


Figure 4.6. Time-marks along continuums of rangeland functionality for three indicators grouped within two themes. For demonstration purposes, the time-marks for the landscape function indicator have been colour coded as in Fig. 4.5.

Evaluating rangeland condition. These charts of time-mark data, along with maps and other types of graphs, provide the kinds of information required by stakeholders to evaluate the condition or health for a rangeland area of interest (e.g., a bioregion). Recall that different stakeholder groups have different visions and values for their use of a rangeland area, although many values are shared (Tables 4.1 & 4.2). This means that stakeholders will focus on different types of information on particular indicators of the functional state of the rangeland area of interest presented to them, for example, in maps and time-mark continuums.

These data on indicators of either current functional state, or trend over time, for the rangeland area of interest will be evaluated by stakeholders relative to their land use goals, in a sense through a ‘values prism’, to arrive at an assessment of rangeland health or condition (Fig. 4.7). For example, if two different stakeholders are strongly focussed on the amount of perennial plant cover as an indicator of rangeland functionality, which is currently at an intermediate value along a continuum, then it would not be unusual to find that stakeholder one with an interest in the palatable (productive grazing) component may evaluate the current health of the rangeland to be in class B (‘good’) condition relative to their values for using this rangeland. For example, stakeholders in some rangeland areas have cleared trees and introduced buffel grass, which is highly productive and palatable to cattle (Friedel et al. 2006). In contrast, stakeholder two with a biodiversity conservation viewpoint may judge this same rangeland area in an intermediate functional state to be in only in class C (‘fair’) condition because buffel grass pastures are known to have low species diversity (Fairfax and Fensham 2000, Franks 2002, Grice 2006). In other words, rangeland condition is in the ‘eye-of-the-stakeholder’, and one person’s interpretation of good condition may be poor condition to another (Parkes and Lyons 2006). This notion is not a new concept for rangelands, being described by Ludwig and Freudenberger (1997; see their Fig. 10.2), by Reynolds and Stafford Smith (2002; see their Fig. 1.2), and recently by Gibbons and Freudenberger (2006; see their Fig. 1). Rangeland condition classes from A (best) to D (worst) are often described as rangelands in excellent, good, fair and poor condition, respectively (e.g., Lange et al. 1994, Friedel et al. 2000, Eyre et al. 2005, QDPI&F 2007).

In this framework (Fig. 4.7), we emphasise the importance of separating monitoring-based information on the **functional state** of a rangeland area from stakeholder’s value-based statements on **condition**, defined as a “state of health” (Macquarie Dictionary). We recommend directly linking the term rangeland condition with rangeland health, the latter now being widely used globally (e.g., DeSoyza et al. 2000, Whitehead et al. 2000, Pyke et al. 2002, McCullough and Musso 2004, Briske et al. 2005, Pellant et al. 2005).

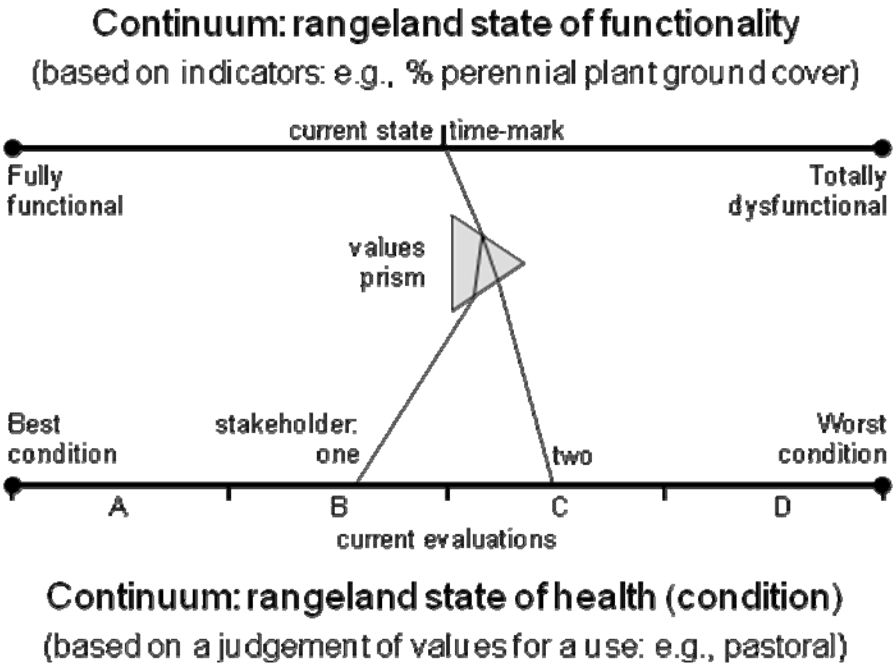


Figure 4.7. A framework for how an indicator of the functional state of a rangeland, currently positioned mid-way along a continuum, may lead (through a 'values prism; Gibbons and Freudenberger 2006) to different assessments of rangeland condition depending on how different stakeholders value this level of functionality relative to their specific land use goals.

4.7. Step 7: Assessing the value of indicators to stakeholders

As noted in the Introduction (Section 1), land uses and the kinds of stakeholders with an interest in the rangelands is continually changing. Therefore, in a rangeland monitoring program, it is important to periodically evaluate the usefulness of the different attributes and indicators currently being measured. Ideally, this evaluation of indicators would be conducted in a workshop setting where all stakeholders with an interest in the rangelands cooperatively examine the current set of indicators . Any proposed indicators to meet new needs would also be discussed in this setting. The aim of this cooperative workshop process would be for different stakeholders to learn from each other's experiences, understand each other's monitoring challenges, and respect each other's choices of indicators (Duff et al. 2008).

An evaluation of different attributes and indicators for monitoring rangelands can be based on a set of standard criteria, such as soundness, sensitivity, simplicity and generality, and a set of criteria specific to stakeholders with an interest in the rangelands, such as whether an indicator is highly valued and informative (Box 4.3). The definitions for the five standard criteria are based on those described in books and research journals on indicators of ecosystem health (e.g., Keddy et al. 1993, Walker et al. 1996, Harger and Meyer 1996, Fisher 1998, Bell and Morris 1999) and in rangeland applications (e.g., Whitehead et al. 2000, Smyth et al. 2003, Hunt et al. 2006). The rating of each indicator against these criteria can be kept general, such as using three (high, moderate, low) or two (yes, no) categories. There is no need to agonise or 'split-hairs' over precisely rating against more than three categories because the relative usefulness of different indicators soon emerges by looking at totals across multiple criteria, especially in a workshop setting with a larger number of participating stakeholders generating big totals.

Box 4.3. Definitions for eight criteria used to evaluate the usefulness of attributes and indicators used in reporting on the functional status of rangelands in Australia. The first four criteria are quite specific to rangeland users and the other four are standard criteria used globally (see text)

Name	Code*	Definition
Valued	Valu	Provides information valued by rangeland stakeholders
Informative	Info	Provides information easily communicated to rangeland stakeholders
Current	Curr	Indicator currently being monitored by rangeland stakeholders
Benchmark	Bnch	Indicator can be assessed relative to a benchmark or reference site
Soundness	Sndn	Indicator soundly connected to fundamental rangeland processes
Sensitivity	Sens	Indicator sensitive or robust for detecting changes in its measured values
Simplicity	Simp	Indicator relatively simple to measure objectively at a reasonable cost
Generality	Genr	Indicator measurable across a range of scales from local to regional

* code used in column headings in Table 4.4.

This principle of emergent usefulness of indicators can be illustrated by an example where two participants (GB, Manager of the ACRIS-MU, and JL, LASR Consulting) jointly evaluated 46 of the indicators used in the ACRIS report "Rangelands 2008 – taking the pulse" (Table 4.5). These 46 indicators were grouped into the 'Theme' and 'Information Product' categories used in the ACRIS 2008 report. In this example, we did not evaluate indicators in 'Biodiversity' and 'Socio-economic' Themes, as these indicators are being evaluated by 'Working Groups' established by the ACRIS Management Committee.

Inspection of Table 4.5 quite clearly suggests which of the 46 indicators were highly rated against the eight criteria (e.g., total 'ticks' = 15, or more), at least by these two participants. This indicator evaluation process would ideally be conducted in a workshop setting involving a wide range of current and potentially future rangeland stakeholders. In this example of an evaluation by two rangeland researchers of indicators used in the ACRIS 2008 report, indicators related to environmental factors such as rainfall and fire rated highly, and indicators related to the capacity of ground-layer vegetation to retain resources and provide forage, such as the cover, frequency or density of perennial grasses and shrubs, also rated high, as might be expected.

In this example (Table 4.5), the usefulness of reporting on indicators receiving, say, less than 20 ticks might be retained after evaluation by ACRIS relative to issues such as whether the indicators rated low because they were still being developed (e.g., the landscape leakiness index; Ludwig et al. 2007b) or because these indicators may only have regional importance but they are very important to stakeholders with an interest in those regions (e.g., kangaroo numbers rated lower overall but significantly contribute to total grazing pressure in some regions; Fisher et al. 2004). Other indicators may have rated low because data are currently limited in their spatial and temporal coverage (e.g., remotely sensed bare ground; Scarth et al. 2006) but these indicators may be worthy of ACRIS investment to increase their coverage. As noted above, this questioning would ideally be done in a workshop involving a number of stakeholders. The aim would be to select a set of indicators to be monitored for future reporting on the functional status of rangelands, which can then help stakeholders evaluate the health of areas of interest and assist them plan and manage actions to maintain or improve their rangelands.

Table 4.5. An evaluation* of 46 indicators (listed by Theme and Product) reported in Rangelands 2008 -- taking the pulse against eight criteria (see Box 4.3 for co

THEME	INFORMATION PRODU	INDEX	VALU	INFO	CURR	BNCH	SNDN	SENS	SIMP	GENR
Climate variability	Seasonal quality rain	1. Gridded recent rainfall	vvv	vvv	Y	Y	vvv	vvv	vvv	vvv
		2. Gridded long-term rainfall	vvv	vv	Y	Y	vv	vvv	vvv	vvv
	Seasonal quality prodn	3. Modelled recent plant prodn	vvv	vvv	Y	Y	vvv	vvv	vvv	vvv
		4. Modelled long-term plant prodn	vvv	vv	Y	Y	vv	vvv	vvv	vvv
Landscape function	Change in 'state'	5. Resource capture index	vv	vvv	Y	N	vvv	vvv	vvv	v
		6. Freq. peren. grasses (northern WA)	vv	vv	Y	N	v	vv	vvv	v
		7. Density shrubs (southern WA)	vv	vv	Y	N	vv	vvv	vvv	v
		8. Density shrubs (southern SA)	vv	vv	Y	Y	vv	vvv	vvv	v
		9. Cover prodn. loss graz. grad. index	v	vv	N	Y	vvv	vv	v	vvv
		10. Freq-cover peren. herbs index	v	v	N	N	v	v	vv	v
		11. Density-cover shrubland veg. index	v	v	N	N	v	v	vv	v
		12. Biomass-cover pasture spp. index	v	v	N	N	v	v	vv	v
		13. Cover prodn. loss graz. grad. index	v	vv	N	Y	vvv	vv	v	vvv
		14. Land cover change analysis	v	vv	N	N	vv	vv	vv	vvv
		15. Lndscp Func ratings by rapid survey	v	v	Y	N	v	vv	vvv	vv
		16. Modelled changes pasture growth	v	v	Y	N	v	vv	v	vv
		17. Remotely sensed changes bare grnd	v	v	N	Y	vv	vv	v	vvv
		18. Grnd-cover index (change analysis)	vv	vv	Y	Y	vv	vv	vv	vvv
		19. Landscape leakiness index	v	vvv	N	Y	vv	vvv	vv	vvv
		20. Mapped gross changes 1992-2005	vv	vvv	Y	N	vv	vv	v	v
		21. Mapped seasonal changes 1992-2005	vvv	vvv	Y	N	vv	vv	v	v
Sustainable management	Critical stock forage	22. Freq. decreaser grasses (northern WA)	vvv	vvv	Y	N	vvv	vvv	vvv	v
		23. Density decreaser shrubs (southern WA)	vvv	vvv	Y	N	vvv	vvv	vvv	v
		24. Density decreaser shrubs (southern SA)	vvv	vvv	Y	Y	vvv	vvv	vvv	v
		25. Freq. change 2P grasses 1992-2005	vvv	vvv	Y	N	vvv	vvv	vvv	v
		26.Comp. change 2P grasses time 1 - 2	vvv	vvv	Y	N	vvv	vv	vvv	v
		27. Freq. change 3P grasses time 1 - 1999	vvv	vvv	N	N	vvv	vvv	vvv	v
		28. Mapped gross change CSF (ex QLD)	vv	vvv	Y	N	vvv	vvv	vv	v
		29. Mapped seasonal change CSF (ex QLD)	vvv	vvv	Y	N	vvv	vvv	vv	v
		30. Mapped forage utilisation change QLD	vv	vv	N	Y	v	vv	v	vv
	Plant species richness	31. Number peren. plant species	vv	vv	Y	N	vv	v	vvv	v
		32. Number native herbaceous species	vv	vv	Y	N	vv	v	vvv	v
		33. Mapped gross change PSR (WA-NSW)	vv	vv	N	N	vv	v	vvv	v
		34. Mapped seasonal change PSR (WA-NSW)	vv	vv	N	N	vv	v	vvv	v
	Invasive weeds	35. Mapped distribution exotic weeds	vvv	vvv	N	Y	vv	vv	vv	v
		36. Mapped relative abundance exotic weeds	vvv	vvv	N	Y	vvv	vvv	v	v
Total grazing pressure	Livestock density	37. Mapped absolute DSE changes 1991-2005	vvv	vvv	Y	N	vvv	vv	v	v
		38. Mapped relative DSE changes 1991-2006	vvv	vvv	Y	Y	vvv	vvv	v	v
	Kangaroo density	39. Mapped absolute DSE changes 1993-2003	vvv	vvv	Y	N	vvv	vvv	vvv	vv
		40. Mapped relative DSE changes 1993-2004	vvv	vvv	Y	Y	vvv	vvv	vvv	vv
	Feral herbivores	41. Mapped distribution feral herbivores	vvv	vv	N	Y	vv	vv	vv	v
		42. Mapped relative abundance feral herbivores	vvv	vvv	N	Y	vvv	vvv	v	v
Lndscp func & Sust. manag.	Fire	43. Mapped yearly fire extent (% of IBRA)	vvv	vvv	Y	N	vv	vvv	vvv	vvv
		44. Mapped seasonal fire intensity (% of IBRA)	vv	vv	Y	N	vvv	vv	vvv	vvv
		45. Mapped fire freq. over 1997-2005	vvv	vvv	Y	N	vvv	v	vvv	vvv
	Dust	46. Mapped interpolated dust storm index	vv	vv	Y	N	vvv	vv	vv	v

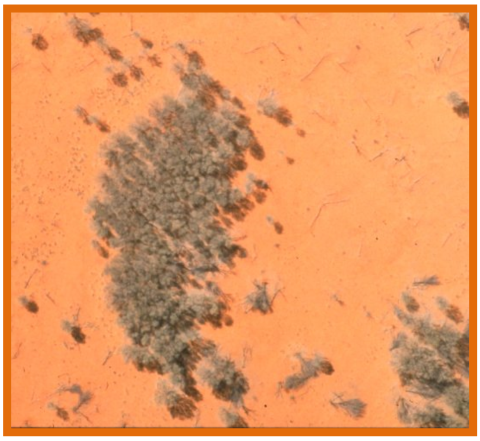
* jointly evaluated by GB (ACRIS-MU) & JL (LASR Consulting) where ratings are vvv = high, vv = moderate & v = low, and Y = yes & N = no. (in sums, Y = 2 vs, N = none)

** details on indicators in "Rangelands 2008 -- taking the pulse" (Bastin and ACRIS-MC 2008)

*** not included in ACRIS 2008 report but possibly for future reporting

5. Dealing with scale and variability in time and space

Rangelands are naturally highly variable because they mostly occupy vast areas within arid, semi-arid and seasonally dry sub-humid climates around the globe and this variability affects how we view changes across the rangelands (Friedel 1994).



5.1. Causes of variability in rangelands

In addition to climate, natural spatial variability is due to patterns in (i) geology where, for example, different rock types affect the colour and texture of surfaces; (ii) soils where different soil types have very different properties and colours, such as the contrast between grey, cracking, self-mulching clays and red, hard-setting, silty sands; and (iii) vegetation where different vegetation types vary from uniformly structured grasslands to heterogeneous mosaics of trees, shrubs and grasses of savannas and woodlands.

These natural spatial variations are often confounded or obscured by patterns of disturbance, such as the 'piosphere' effect where there is lower vegetation cover and higher exposed soil near stock watering points (Lange 1969). These effects lead to 'grazing gradients' where cover increases and bare soil decreases away from water points, which are most readily observed in large paddocks with dispersed waterpoints (i.e., > 100 km^2 and 3-4 waterpoints) with consistently high stock numbers (Pickup 1994). Disturbance patterns are also evident as 'erosion cell' mosaics due to selective grazing of preferred patches in the landscape (Pickup 1985, 1989). Selective grazing of preferred 'bands' of vegetation also enhances runoff-runon processes, which can lead to strongly banded vegetation (Ludwig and Tongway 1995).

Of course, all of these natural and modified spatial patterns have temporal variations that affect the surface reflectance data acquired by remote sensing-based monitoring. These variations are due to changes in the seasonal state of (i) vegetation, which varies intra-annually from highly green during active growth phases to very brown states of senescence and inter-annually with abundant growth during wet years but limited growth during droughts; (ii) soil surface colours, which tend to be darker when moist; and (iii) litter cover, which are lighter and in higher amounts following periods of vegetation senescence.

Ideally, temporal trends in rangelands are best evaluated over the long-term because of variations caused by fluctuations in climatic patterns (e.g., ENSO-driven rainfall) and disturbances (fire, grazing). However, shorter-term grazing trials are useful to establish quantitative relationships between stock performance and rangeland functionality. They often demonstrate the over-riding effects of seasonal trends. For example, the body condition of sheep being run in a trail to measure wool (Merino) and meat (Damara) production, and forage quality and quantity, markedly declined during a very dry period in Western Australia (Alchin et al. 2007).

5.2. Dealing with variability when monitoring rangelands

Spatial and temporal variability across the rangelands means that monitoring programs must deal with heterogeneity when measuring and reporting changes. A useful approach is to stratify or classify and map the rangelands into more homogeneous units. These units are defined as a hierarchy of scales depending on purposes defined by stakeholders. For example, at the national scale, a Government agency may wish to contrast various attributes of rangelands with other vegetation types (e.g., ESCAVI 2007), and Federal and State/Territory government agency partnerships may wish to contrast regions within the rangelands (e.g., Bastin and the ACRIS-MC 2008). At the regional scale, State/Territory rangeland management agencies, catchment management authorities and NRM boards may require land unit stratification down to the level of individual properties and paddocks, and at even finer scales, pastoralists may be concerned with units and sites within paddocks (Gibbons and Freudenberger 2006). When monitoring rangelands, stakeholders need to accurately define their area of interest so that providers of monitoring information can stratify and map the area into appropriate units.

A second approach is to explicitly incorporate spatial and temporal variability into the design of the monitoring program, and to build these programs on concepts such as rangeland stability, resilience and 'state-and-transition' (see Section 3.3). For example, the 'grazing gradient' method explicitly examines how vegetation cover varies spatially away from stock watering points and how this cover recovers over time (resilience), particularly after significant rainfall events (Pickup et al 1994). This resilience method is remote sensing-based and will be described in Section 6.4.4. Another remote sensing-based method that explicitly deals with spatial and temporal variability is 'land cover change analysis' (Wallace et al. 2004). Maps and time traces of changes in cover are analysed to identify which paddocks and properties within a rangeland area are improving, stable or declining in cover over time (Karfs et al. 2000). This method will also be described in Section 6.4.4.

6. Detecting significant changes

In this Section we address a critical issue in rangeland monitoring: how to detect and interpret significant changes in measures (indicators) of the functional state of rangelands so that stakeholders have robust information to evaluate rangeland condition or health. We examine this issue for two main types of monitoring data: (i) measures observed on ground-based sites, and (ii) measures derived from remote sensing-based technologies. These two sources of data will be addressed in sub-Sections 6.3 and 6.4, respectively. First, we define some terminology and list some assumptions relevant to both data types.

6.1. Terminology

Precision versus accuracy. These two terms subtly differ. Precision is how close repeated measurements on an object are to each other. For example, if three observers measured plant cover on the same monitoring site using the same method, the closeness of the three cover measurements reflects precision. Another example is where a pH metre is used three times to measure the pH of a soil sample and precision is the closeness of the three pH values. Accuracy is how close a measured value is to its true value, for example, how close the mean of the three cover measurements is to the true value of cover on the monitoring site. Bias is a description of how errors deviate from the true value, for example, are they consistently positive or negative.

In rangeland monitoring, accuracy is desired, but true values for measured indicators are usually unknown, hence, the errors and any bias of methods used to measure indicators is also unknown (Elzinga et al. 2001). Given these unknowns, what can be done to be as accurate and unbiased as possible? Using a precise instrument or method helps, but accuracy can only be evaluated by using standards. For example, a pH metre will typically be highly precise (provides close repeated measurements). If the instrument is calibrated for accuracy against standard solutions of known pH, then one can reasonably assume that if this calibrated metre is used to measure the pH of a soil sample, then the measured value should be unbiased and accurate.

This same principle applies to other measured attributes such as cover. If cover on a site is very carefully and painstakingly measured, say by charting individual plants and summing individual covers to obtain a total cover, then this cover value can be taken as a standard against which people using a more rapid cover assessment method (e.g., a visual estimate) can be calibrated. Training against standards helps to improve precision (repeatability) and accuracy (closeness to the standard) by reducing bias (deviations from the standard).

If a method for measuring an attribute can reasonably be assumed to be precise and unbiased, then it can also be assumed (from the central limit theorem) that making additional measurements on the attribute will result in an average value (arithmetic mean) that is quite accurate (Quinn & Keough 2002). A sampling statistic defined as the standard deviation of the mean (often referred to as standard error) provides an estimate of the variability about a mean assuming repeated sampling to estimate means. Taking a larger number of measurements (sample size) will reduce the estimated standard error about the mean.

Hypothesis testing and power. In classical statistics, the testing of hypotheses concerning, say differences in two means obtained from sampling two populations, involves two steps. First, a statistical null hypothesis is stated, typically specifying that no difference in the two means is expected. Second, a test statistic is chosen to evaluate this null hypothesis, for example, the t statistic. This evaluation of the null hypothesis involves setting a low probability (e.g., $P = 0.05$) to avoid making a false conclusion that a difference in means exists when in fact there may not be a difference (i.e., a Type I error; Box 6.1). There is also a probability for reaching a false conclusion that no difference in means exists when in fact there is real difference (a Type II error). Related to this Type II error is 'power', which is the probability of making a correct conclusion that there is a real difference (Power = 1 – Type II error).

Power is an important notion in rangeland monitoring because we desire to be correct when concluding that a significant change has occurred, for example, in mean values for an indicator of rangeland condition over time (trend) or between monitoring sites and reference sites at a specified point in time (improvement or decline). The number of sites on which an indicator is measured (i.e., sample size) strongly influences the value of the test statistic used to evaluate the probability of correctly detecting a real change (i.e., power). This leads to power analysis or an examination of the adequacy (size) of sampling required to detect a specified level of change at a nominated level of power (Quinn and Keough, 2002, see their Chapter 7). We illustrate calculations for evaluating adequacy of sample size in Section 6.3.

Box 6.1. Brief definitions of statistical error types and power

Type I Error – when the decision is to reject the null hypothesis (e.g., that two means do not differ) when in fact this decision is false. This decision is based on the probability of obtaining (or exceeding) a value for the test statistic (e.g., a t value) from the set of observed data. The probability is usually set at a low level (e.g., $P = 0.05$) to reduce the chance of reaching a false decision.

Type II Error – when the decision is to accept the null hypothesis of, say, no difference in two means, when in fact there is a real difference. This decision is also based on setting a low probability (e.g., $P = 0.10$) relative to obtaining or exceeding a test statistic from the dataset.

The probabilities for making Type I and Type II errors do not necessarily need to be set at the same level because a Type I Error can only occur if the null hypothesis is true and a Type II Error can only occur when the null hypothesis is false. However, in most rangeland monitoring data analyses, the probabilities for Type I and Type II errors are usually set at the same level (e.g., $P = 0.10$) because it is just as important to confidently detect a real change as it is to avoid making a false decision (Watson 1998, Herrick et al. 2005).

Power – when the decision is to reject the null hypothesis because there is a real difference in, say two means, and this decision is correct. The probability for power is equal to one minus the probability set for making a Type II Error.

For a thorough discussion of these definitions see Chapters 3 and 7 in Quinn and Keough (2002).

Statistical and ecological significance. If a test statistic (e.g., t value) and a reasonable probability for power (e.g., $P = 0.10$) point to a decision that a real difference, say in two means, exists, then this difference is interpreted as being statistically significant. If sample sizes are large, the test statistic may suggest that a relatively small difference in means is significant at the specified level of probability, but a question arises: is this small difference really meaningful from an ecological point of view?

It can be reasonably argued that ecological significance may have nothing to do with statistical significance (Quinn and Keough, 2002, see their Box 3.4). Whether a statistical finding that an indicator has changed over time, say between a monitoring site and a reference site, is ecologically significant depends on the questions being addressed. These questions often relate to the size of the change in an indicator logically expected given the spatial and temporal variability over the rangeland area of interest. In evaluating adequacy of sample size in Section 6.3, we emphasise the need to set a meaningful level of change expected in an indicator, and that this change must be specified in absolute terms (i.e., a specified amount of change in the units of measurement).

6.2. Assumptions

In the following Section 6.3, which illustrates methods of detecting significant changes in an indicator (mean differences and trends) over time based on data from rangeland monitoring sites, we make a number of assumptions. Our aim in making these assumptions is to simplify the examples we present, however, we quickly add that these assumptions need to be carefully considered when designing rangeland monitoring programs and evaluating monitoring data information systems such as ACRIS.

First, we assume that the attribute or indicator is being measured with sufficient precision and accuracy so that, if differences are detected, they reflect real changes, not differences due to low measurement precision and accuracy. Here, we assume that jurisdictions conducting rangeland monitoring will have evaluated the types of measurement errors affecting the precision and accuracy of their methods, and will be aware of strengths and weaknesses of different methods. A number of studies evaluating precision and accuracy of different monitoring methods have been conducted in the rangelands of Australia (e.g., Holm et al. 1984, Friedel & Shaw 1987) and overseas (e.g., Risser 1984, Bonham 1989, BLM 1996, NRCS 2003, Herrick et al. 2005).

Second, indicators being measured on monitoring sites are also being observed on comparable reference sites (similar in climate and land type, that is, topography, vegetation and soils). This is important so that any changes on the monitoring sites can be evaluated relative to changes on the reference sites. Evaluating rangeland condition relative to a benchmark or baseline is recommended because it provides a reference point for changes occurring on monitoring sites. It is appreciated that for many rangeland areas, benchmark or reference sites that reflect natural variations in space and time do not exist, and in some cases it may be useful to define a hypothetical reference state for assessing condition (see Section 4.1.2).

Third, in addition to measuring indicators on monitoring and reference sites, we assume that data is also being collected, or is available, on factors that help to explain why changes and variations have occurred. If a significant difference or change over time (trend) has been detected, what information is available to interpret this trend? Is the difference due to disturbance factors such as livestock or feral animal grazing or is it due to natural variations in time and space such as patterns of rainfall. In other words, additional (multifactor) data is assumed to be available to statistically partition source of variation to help interpret differences, both from statistical and ecological viewpoints.

Fourth, in the following Section 6.3, we apply a number of relatively simple statistical methods including *t*-tests, analysis of variance and one-factor linear regression analysis to detect changes in an indicator. Although more complex statistical analyses can be applied to monitoring data, such as multifactor analysis of variance, analysis of co-variance, multiple regression and non-linear regression, we only use simple analyses because we have intentionally kept the questions being addressed on detecting significant changes straightforward – *t*-tests, analysis of variance and simple linear regressions are sufficient. We illustrate some statistical calculations but assume that the reader has a basic understanding of statistics so that, if desired, details on how to apply statistical methods can be obtained from standard statistical references such as Quinn and Keough (2002).

6.3. Detecting changes for indicators measured on field sites

In this Section, we address a general set of questions where the aim is to detect significant differences in an indicator at two points in time for a rangeland area being monitored with a set of field sites, and in comparison to a set of reference field sites. Of interest is the effect of spatial heterogeneity and related scale (extent of area) effects on the ability to detect significant changes for the rangeland area under consideration. We compare differences in means at two points in time for three scenarios where monitoring sites are located on a pastoral property or set of properties: (i) with only one relatively homogeneous land type; (ii) with only one, but a more heterogeneous, land type; and (iii) with a mix of land types within an entire sub-IBRA region. The expectation is that there will be notable effects due to differences in the spatial heterogeneity of land types and due to the greater variability inherent at broader sub-IBRA scales.

We address three key questions that relate to (i) are there statistically significant differences in means based on standard *t*-tests; (ii) what amount of change in a mean measured at, say, an initial time period is considered important (e.g., 10% change); and (iii) what level of sampling (number of sites) is required to detect this % change? We address these questions using an ACRIS dataset provided by a jurisdiction (here anonymous) where an indicator of landscape functionality (the cover of perennial ground vegetation, primarily grasses, hereafter simply referred to as cover) was measured on a large number of monitoring and reference sites located on multiple land types within a sub-IBRA. For these sites, a seasonal quality index has been derived (details in Bastin and ACRIS-MC 2008) and will be used here to note if any statistically significant changes detected might be due to this environmental factor (also see Section 4.6).

6.3.1. Change between two points in time: monitoring sites only

One relatively homogeneous land type. To illustrate fine scale (spatial variability) effects we first tested for statistical differences in the cover at two points in time within a relatively small area with only one land type that tends to be spatially homogeneous. We used two example datasets from this small area. The first example uses a fixed set of sites (n = 13) where cover was measured on the same set of sites at an initial time and at a later time (after 11 y), that is, observations were paired. This first example used a paired *t*-test, and the mean cover significantly increased 22.6% from the initial time to the later time (Table 6.1).

The second example dataset is where, after measuring an initial set of sites within the land unit (n = 18), additional monitoring sites were established and measured, adding to the initial set (n = 41); this is a typical situation in rangeland monitoring. In this case, where site numbers increase, hence are not paired, the mean comparison between the initial time and the later time is based on a group mean *t*-test using separate variances (not pooled) because of differing sample sizes and standard deviations (Quinn and Keough 2002). For this dataset, mean cover significantly increased by 23.8% (Table 6.1). This increase in cover can be explained, in part, by the seasonal quality before the monitoring period which was notably below average so that cover was low in the initial period.

Table 6.1. For a relatively homogeneous land type, statistical tests of mean differences in an indicator (perennial vegetation cover – %) between an initial time and a later time for a fixed set of sites (paired) and for a variable number of sites (grouped).

	Sample size (n)		Means						Sample size
Dataset	Initial	Later	Initial	Later	Difference	*t*-value	df	*P*	OK
Sites paired	13	13	64.7	87.3	22.6	5.2	12	<0.001	Yes[1]
Sites grouped	18	41	62.6	86.4	23.8	5.9	20	<0.001	Yes[2]

[1] for detecting a 20% change in the initial mean for the dataset where sites are paired (Box 6.2).

[2] for detecting a 20% change in the initial mean for the dataset where sites are in groups (Box 6.3).

Sampling 13 paired sites on this relatively homogeneous land unit was adequate to detect a 20% change in cover from the initial time period (Table 6.1), based on a desired ‘Power’ of 90%, that is, the probability of reporting a real change is 0.90 (Box 6.2). The initial sample size of 18 grouped sites is also adequate to detect a 20% change in cover given the set of sampling parameters (Box 6.3).

Box 6.2. Example calculations for determining the adequacy of sample size, ***n****, for detecting a specified change in mean cover measured at an initial (first) monitoring time, where the dataset is a fixed set of sampling units and where mean differences between two time periods (e.g., first to last) are examined with a paired *t*-test (Herrick et al. 2005, see their Appendix C, pgs 165-168).

The equation to apply is $n^* = \dfrac{(S_{dif})^2\,(Z_\alpha + Z_\beta)^2}{(MDC)^2}$,

where S_{dif} is the standard deviation of the difference between the paired samples,

Z_α is the Z-coefficient for a Type I error, which is the probability of reporting a change when there has been no change,

Z_β is the Z-coefficient for 'Power', which is the probability of reporting that a change has occurred and there really has been a change (Power is equal to 1 minus a Type II error, which is the probability of reporting no change when in fact there has been a change), and

MDC is the minimum detectable change, which must be expressed in absolute terms. For example, using the initial perennial vegetation cover of 64.7% for paired sites reported in Table 6.1, and if you want to detect a 20% change in cover from this initial measurement, then MDC = (0.20 x 64.7) = 12.94.

For these paired site data, the S_{dif} was equal to 15.7. If we assume an acceptable Type I error of 0.10 and a Power of 0.90, the respective Z- coefficients are $Z_\alpha = 1.64$ and Z_β is 1.28 (from a table of standard normal deviates), then by applying the equation and rounding up:

$$n^* = \frac{(15.7)^2\,(1.64 + 1.28)^2}{(12.94)^2} = 13,$$

which equals the actual sample size (13), suggesting sampling was just adequate.

However, if you want to detect smaller change, say a difference in the initial mean of a 10% change in cover, then MDC = (0.10 x 64.7) = 6.47, and

$$n^* = \frac{(15.7)^2\,(1.64 + 1.28)^2}{(6.47)^2} = 51,$$

which suggests that a set of 51 sampling sites would need to be established and measured to detect a 10% change in the initial cover measurement.

Box 6.3. Example calculations for determining the adequacy of sample size, ***n****, for detecting a change in mean cover measured at an initial (first) monitoring time, where the dataset has a variable number of sampling units and where mean differences between two time periods are examined with two-sample *t*-test.

The equation to apply is $n^* = \dfrac{(S_{max})^2\,(Z_\alpha + Z_\beta)^2}{(\mathrm{MDC})^2}$,

where S_{max} is the largest standard deviation of the two standard deviations in the dataset (16.3, and 7.4 for the initial sample and the later sample, respectively); this is the most conservative approach (a less conservative method would be to 'pool' the two standard deviations).

Z_α and Z_β are defined as in Box 6.2, and MDC = 12.52 for this example using an initial perennial vegetation cover of 62.6% and a desired detectable change of 20% in this mean: MDC = (0.20 x 62.6) = 12.52, then

$$n^* = \frac{(16.3)^2\,(1.64 + 1.28)^2}{(12.52)^2} = 15,$$

suggesting that the set of 18 monitoring sites established at the time of the initial sampling was an adequate sample size to detect a 20% change in the mean of 62.6% cover.

However, if the aim was to detect a smaller change in this initial mean of, say 10%, then MDC = (0.10 x 62.6) = 6.26, and

$$n^* = \frac{(16.3)^2\,(1.64 + 1.28)^2}{(6.26)^2} = 58,$$

which suggests that the set of 18 sites established at the initial monitoring time was inadequate to reliably detect a 10% change in the mean cover observed at this time.

If the future aim is to detect a 10% change in perennial vegetation cover from the later sampling period, which had a mean value of 86.4% with a standard deviation of only 7.4, then

$$n^* = \frac{(7.4)^2\,(1.64 + 1.28)^2}{(8.64)^2} = 7,$$

which suggests that by increasing the number of monitoring sites to 41 this number is now quite adequate to detect in the future a 10% change in the mean cover measured at the later sampling period.

One relatively heterogeneous land type. The second example is also for only one land type that occupies a relatively small area, but this land unit tends to be more spatially heterogeneous. We tested for statistical differences in the cover indicator between an initial time and a later time (after 11 y) using paired and grouped datasets, as in the first example. For the dataset, where observations were paired and mean covers at the two times were compared with a paired *t*-test, cover increased 39.0%, with significance $P < 0.05$ (Table 6.2). For the dataset with differing site numbers (grouped sites), where mean covers were compared with a two-sample *t*-test, cover increased by 29.7% and this difference was significant at $P < 0.05$. In this second case the number of sites measured at the later time (n = 2) actually decreased from the initial time (n = 5). This occurred because of logistical constraints where not all the sites located within the land unit were measured during the second time period; this is a common situation in rangeland monitoring.

As for the previous example, the significant increase in cover can be explained by patterns in the quality of growing seasons prior to and during the monitoring period. A drier period with well below average seasons preceded the initial sampling, and this effect carried into this period, which also had a below average seasonal quality.

Table 6.2. For a more heterogeneous land unit, statistical tests of mean differences in cover between an initial time and a later time for a set of paired sites and for a set of grouped sites. Adequacy of sample size calculations follow the examples in Box 6.2 and Box 6.3.

	Sample size (n)		Means						Sample size
Dataset	Initial	Later	Initial	Later	Difference	t-value	df	*P*	OK
Sites paired	2	2	46.5	85.5	39.0	13.0	1	0.049	Yes
Sites grouped	5	2	55.8	85.5	29.7	3.8	5	0.015	No

In this example for a more heterogeneous land unit, the set of 2 paired sites initially observed was just adequate (***n**** = 2) to detect a 20% change in the mean of 46.5% cover (Table 6.2). This result was perhaps unexpected, but given that the estimated variability in the paired mean difference, S_{dif}, was only 4.24, following calculations in Box 1 suggested that a sample size of only 2 was required to detect this 20% level of change. A sample size of 2 sites per land unit is considered an absolute minimum in rangeland monitoring (Herrick et al. 2005).

For the dataset of 5 sites observed as a group at the initial sampling time, the variability was higher (S_{max} = 14.13), and 5 sites were not adequate (calculations as in Box 6.3) suggesting that a sample size of 14 sites would have been required to detect a 20% change in the mean cover of 55.8% at a 'power' level of 0.90. A statistically significant increase in mean cover to 85.5% was detected (P = 0.015), but this probability was due to the fact that the two-sample *t*-test also uses the variance for the later sampling time, which was considerably smaller (S_{max} = 6.36). Here, to calculate adequacy of sampling to detect a 20% change in mean cover for the first sample, we applied the more conservative approach and used the largest standard deviation of the two samples observed (S_{max} = 14.13) rather than a 'pooled' standard deviation or the smaller standard deviation of 6.36.

Multiple land types across a heterogeneous sub-IBRA region. The third example illustrates calculations for detecting changes in mean cover between two monitoring times across the spatially heterogeneous mix of multiple land types found in a large sub-IBRA region. For paired and grouped datasets, two questions were addressed: (i) are there significant changes in mean covers, and (ii) are sample sizes adequate to detect 20% changes from the initial means?

For both the paired and grouped datasets, we detected a significant increase in mean covers (Table 6.3), and sample sizes were adequate in both cases. Again, as for the two previous examples of smaller rangeland areas, the significant increase in cover at sub-IBRA (i.e. regional) level can be attributed, at least in part, to changes in seasonal quality. An extended period of below average seasonal quality occurred prior to the initial monitoring period, leading to below average initial covers.

Table 6.3. For a mix of land types across a sub-IBRA region, statistical tests of mean differences in an indicator of the state of landscape functionality (% perennial vegetation as ground cover) between an initial time and a later time (after 11 y) for a fixed set of sites (paired) and for a variable number of sites (grouped). Adequacy of sample size calculations as in Boxes 6.2 and 6.3.

	Sample size (n)		Means						Sample size
Dataset	Initial	Later	Initial	Later	Difference	*t*-value	df	*P*	OK
Sites paired	22	22	64.2	85.5	21.3	6.3	21	<0.001	Yes
Sites grouped	42	75	55.4	81.8	26.4	7.3	64	<0.001	Yes

Note that the number of paired sites sampled at the two time periods (n = 22) was adequate (***n**** = 14) and was considerably less than the number of grouped sites sampled initially (n = 42) and later (n = 75) (Table 6.3). Only 22 of the 42 sites observed at the initial time were observed again at the later time period in this example ACRIS data, although they were observed during other time periods. This staggered sampling of sites across time (and space) is a typical situation in rangeland monitoring.

Although the dataset for this large sub-IBRA region included a mix of land units, and the standard deviation of the 42 observations for the initial time period was relatively large (S_{max} = 20.6), the relatively large sample size of 42 was adequate (n^* = 30) to detect a 20% change in the initial mean cover of 64.2% (Table 6.3). However, we found that the sample size of 42 was not adequate to detect a smaller 10% change (n^* = 118). In contrast, the sample size of 75 sites as a group was very adequate for detecting a 10% change in the mean cover for the later time period (n^* = 27) when cover was consistently higher (81.8%) and the standard deviation of cover observations was lower (S_{max} = 14.5).

These findings emphasise the importance of spatial heterogeneity across a region and how it varies with time, and the need to have a relatively large set of sites to observe over the region to reliably detect changes over the range of variation that might be expected, especially if the desire is to detect smaller absolute changes in means (e.g., 10%). Sample sizes also need to be large if the level of 'power' is increased from the 0.90 probability of reporting that a real change has occurred (recall that power is equal to 1 minus a Type II error, which is the probability of reporting no change when in fact there has been a change). When the level of power is increased the probability of making a Type I error (reporting a change when there has been no change) is usually also set to a stricter level. For example, here we set the probabilities of Type I and Type II errors at 0.10 (power = 0.90), where from a table of normal deviates Z_α and Z_β were 1.64 and 1.28, respectively. If we had set probabilities of Types I & II errors to 0.05 (power = 0.95), then Z_α and Z_β would be 1.96 and 1.64, respectively. These values enter the numerator of the equations used to calculate adequacy of sampling (see Box 6.2 and Box 6.3), hence, they increase the number of samples required to detect a specified level of absolute change in a mean (e.g., 20% cover). These results demonstrate the importance of setting both the level of power and expected change in a mean on the adequacy of sample size; these effects are also discussed by Price et al. (2003) and Mac Nally et al. (2004).

6.3.2. Change between two points in time: monitoring sites versus reference sites

In rangeland monitoring and assessment of condition, there is a need to detect significant differences in an indicator being measured on monitoring sites compared to that on reference sites within a rangeland area of interest. These differences can be examined at one or more points in time. Here, we simply illustrate calculations for detecting mean differences within the example ACRIS region at two time periods, but example calculations can readily be extended to more than two time periods.

For the three examples illustrating effects of spatial heterogeneity and scale, we now compare the effects found for sets of monitoring sites with those for equivalent sets of reference sites. Two questions apply: (i) are mean covers significantly different between monitoring sites and reference sites at the initial time of observation, and (ii) at the later time? In this example ACRIS data, reference sites are defined as those field sites that have maintained a high cover over all years of observation, especially in those periods when seasonal quality was below average.

For monitoring site versus reference site comparisons, data were in groups, so two-sample *t*-tests were applied at initial and later time periods. Although a two factor (site type, time) analysis of variance could also be applied, here we used the simpler individual one-factor comparisons.

Sample adequacy was examined in Section 6.3.1 for monitoring sites, and will not be repeated here, or applied to reference sites (typically fewer in number or do not occur in many rangeland areas).

One relatively homogeneous land type. For this case, we found that reference sites were significantly higher in mean cover than monitoring sites at both sampling periods (Table 6.4). Although the mean difference in cover at the later sampling time was only 4.6%, this small difference was statistically significant at $P < 0.01$ because the variability among reference site observations was small (S_{max} = 2.2). Whether this small mean difference is ecologically significant is open for debate.

One relatively heterogeneous land type. For the second example, where one land unit has been observed on only a few sites, we found that there were no significant differences in mean cover between the set of monitoring sites and the set of reference sites at the initial time period and at the later time period (Table 6.4). This result was expected because earlier analyses suggested that sampling was only just adequate to detect a 20% change in the mean cover for the monitoring sites at the initial time period (see Table 6.2).

Multiple land types across a heterogeneous sub-IBRA region. For the third example, which compares monitoring sites with reference sites across a larger more heterogeneous region, we found mean covers differed significantly at the initial time period and at the later time period (Table 6.4). This finding was expected because sampling was more than adequate to detect a 20% change in the initial mean cover for the set of monitoring sites (refer back to Table 6.3).

Table 6.4. For three example datasets representing different levels of heterogeneity and scale (extent of rangeland area), an indicator of landscape functionality (% perennial ground cover) is explored for statistical differences in mean covers between data from a set of monitoring sites and a set of reference sites. Mean differences were tested for an initial time period and a later time (after 11 y).

Dataset	Sample Time	Mean % Cover (n = sample size) Monit. (n)	Refer. (n)	Difference	*t*-value	df*	*P*
One land type	Initial	62.5 (18)	80.0 (5)	17.5	3.8	20	<0.01
(homogeneous)	Later	86.4 (41)	91.0 (5)	4.6	3.0	18	<0.01
One land type	Initial	55.8 (5)	69.0 (2)	13.2	1.4	3	0.26
(heterogeneous)	Later	85.5 (2)	76.0 (2)	-9.5	1.4	2	0.29
Multiple land types	Initial	55.4 (42)	75.5 (15)	20.1	5.1	52	<0.001
(heterogeneous)	Later	81.8 (75)	89.7 (11)	7.9	3.9	60	<0.001

* These two-sample *t*-tests were based on separate variances and averaged degrees of freedom (df) were rounded to the nearest whole integer.

The number of reference sites located within a rangeland area, whether a small area of one land unit or an entire sub-IBRA region, is typically much lower than the number of monitoring sites within that same area. For example, 75 monitoring sites were observed within the sub-IBRA region at the later time period but only 11 reference sites were identified and observed in the region. Although not included here, it is possible to calculate statistical differences in an indicator between a set of many monitoring sites and only one reference site (Herrick et al. 2005, see their Appendix C).

6.3.3. Changes over multiple time periods (trend): monitoring and reference sites

When monitoring in a rangeland area has occurred over a considerable period of time, of interest are questions about whether there are significant trends in an indicator over this time period, and if so, what factors might explain these changes? Here, we explore some methods for addressing the question about changes over time, including analysis of variance (one-way and repeated-measures) and simple linear regression analysis. As before, we also illustrate spatial variability effects by comparing results for monitoring sites located within a relatively small, homogeneous area of rangeland with those for monitoring sites located across a larger, more heterogeneous region (sub-IBRA). Again we use the ACRIS data provided by the anonymous jurisdiction, but now expand these data to include four sampling periods, which were observed 5, 8 and 11 y after the initial (0 y) sampling time. As in previous examples, these data may be paired (same set of sites repeatedly observed over the four time periods), or data may be grouped (numbers of sites varied over the four times). In the paired case, repeated measures analysis of variance was applied to test for significant changes in means and for significant trends. In the grouped case, single-factor (one-way) analysis of variance was applied to test for significant mean differences over time. Regression analyses was used to explore for trends in the grouped data.

One relatively homogeneous land type. For this example, where cover was repeatedly measured on the same set of monitoring sites (paired data), repeated measures analysis of variance suggested that cover significantly differed over time (Table 6.5), and that this increase in cover fit a linear trend ($P < 0.001$). These findings were confirmed by an analysis of variance of the set of grouped data where site numbers (n) differed over the four times. For these grouped monitoring site data, linear regression suggested a highly significant increasing trend ($r^2 = 0.24$, $P < 0.001$; Fig. 6.1A). The linear regression coefficient b was 2.07, which defines linear change in % cover per year over the 11 y of monitoring. The trend in cover for the reference sites (n = 5) across the four times was also positive ($b = 0.95$; Fig. 6.1A), and this linear trend-line was also significant, but less so ($P < 0.05$). Although there was some convergence over the period of monitoring, the slopes of these two regression lines do not differ significantly ($P = 0.16$) based on an equality of regression coefficients t test (Box 6.4).

Table 6.5. For three example datasets for monitoring sites, representing different levels of heterogeneity and scale (extent of area), cover was explored for statistical differences in means estimated at four times over a 11 y period for a set of fixed sites (repeatedly measured) and for a variable number of sites (grouped).

		Mean % Cover (n = sample size) at Sampling Times						
Dataset	Type	One (n)	Two (n)	Three (n)	Four (n)	F	df*	*P*
One land unit	Repeated	64.7 (13)	75.2 (13)	73.4 (13)	87.3 (13)	9.3	3/36	<0.001
(homogeneous)	Grouped	62.5 (18)	73.6 (19)	74.2 (42)	86.4 (41)	14.1	3/116	<0.001
One land unit	Repeated	46.5 (2)	70.0 (2)	76.5 (2)	85.5 (2)	29.6	3/3	<0.01
(heterogeneous)	Grouped	55.8 (5)	70.0 (2)	76.5 (2)	85.5 (2)	4.4	3/7	0.06
Many land units	Repeated	64.2 (22)	75.5 (22)	68.1 (22)	85.5 (22)	12.3	3/63	<0.001
(heterogeneous)	Grouped	55.4 (42)	69.8 (30)	69.7 (75)	81.8 (75)	20.6	3/218	<0.001

* F-value degrees of freedom are numerator df (number of groups – 1) over denominator df (error).

One relatively heterogeneous land type. For this second example and using the repeated monitoring site data, we found a significant difference in mean covers ($P < 0.01$; Table 6.5), even though there were only two sites consistently monitored over the four sampling times. The repeated measures analysis of variance design is a very powerful statistical analysis when there is a consistent difference and a strong linear trend across the monitoring site data (Quinn and Keough 2002), as in this ACRIS data example (Fig. 6.1B). Using the grouped dataset and one-way analysis of variance, means across years were not quite significantly different at the $P_{.05}$ level ($P = 0.06$). However, the linearly increasing trend in cover over the 11 y was highly significant by linear regression of these monitoring site data ($b_1 = 2.67$, $r^2 = 0.63$, $P < 0.01$).

In contrast, for the reference site data set, there were no significant mean differences or linear trends in cover over time for the grouped data ($b_2 = 0.61$, $r^2 = 0.10$, $P = 0.43$) (Fig. 6.1B), or for the repeated dataset. The regression slopes for the monitoring and reference site grouped data were not significantly different at the $P_{.05}$ level, but were significantly different at the $P_{.10}$ level ($P = 0.06$ for $t = 2.03$ at 15 df; calculations as in Box 6.4). For the repeated reference dataset, the variance and regression analyses had low degrees of freedom because, as for monitoring sites (Table 6.5), only two reference sites were consistently observed over the four time periods.

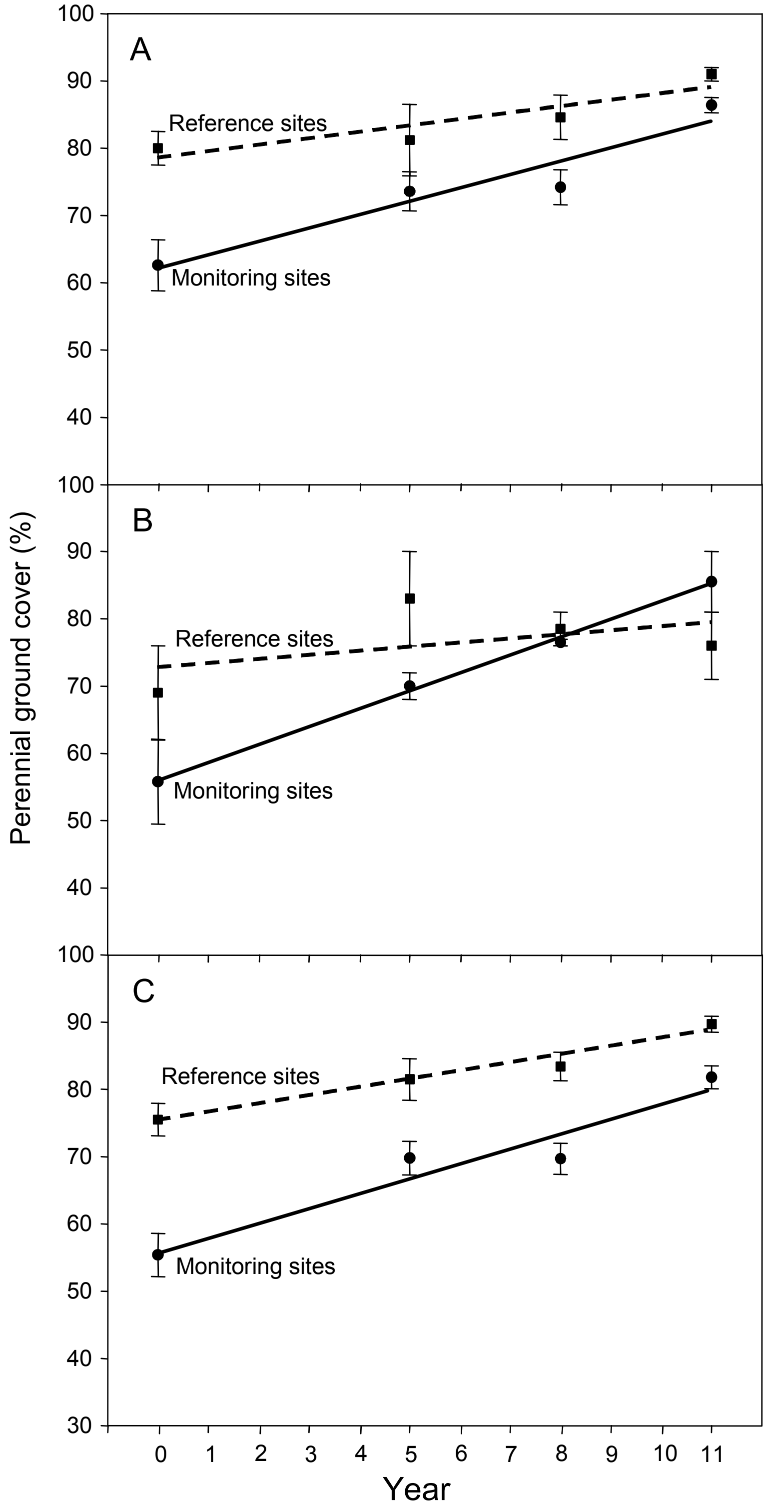


Figure 6.1. Trends in perennial cover monitored 4 times over 11 years on reference sites and monitoring sites for (A) one relatively homogeneous land unit, (B) one more heterogeneous land unit, and (C) multiple land units across a heterogeneous sub-IBRA region.

Box 6.4. Example calculations for testing the equality of two regression coefficients, which is a test that the two regression lines have the same slope, not that they are the same line.

We illustrate calculations for testing the equality of slopes using a simple ***t*** test, but in practice it is perhaps more efficient to use a statistical package that calculates interactions between factors (e.g., type of site, years of monitoring) using general linear modelling procedures.

The equation to apply is $$\boldsymbol{t} = \frac{(b_1 - b_2)}{\sqrt{[(\Sigma x_1^2 + \Sigma x_2^2)/(\Sigma x_1^2 * \Sigma x_2^2)] * [(\acute{s}_1^2 + \acute{s}_2^2)/ df]}},$$

where b_1 and b_2 are the two slopes being compared,

Σx_1^2 and Σx_2^2 are the sums of squares for the predictor variables x_1 and x_2 used in computing the two linear regressions for the response variables y_1 and y_2, respectively,

$\acute{s}_1^2$ and $\acute{s}_2^2$ are the unexplained sums of squares for these two linear regressions, and

df is the degrees of freedom, which is the sum of the df for the two sample sizes, $(n_1 - 2) + (n_2 - 2)$.

The right-hand term in square-brackets in the denominator is the weighted average of the unexplained sums of squares for the two linear regressions.

The right-hand and left-hand terms in the denominator are multiplied and the square-root of this value is calculated.

The ***t*** value obtained is compared to probabilities in a table of critical values for the ***t*** distribution.

An example calculation for comparing the two slopes for cover regressed on years for monitoring sites and reference sites for the relatively homogeneous land unit (Fig. 6.1A) uses the following data:

(i) monitoring and reference site regression slopes are $b_1 = 2.07$ and $b_2 = 0.95$;

(ii) there are 120 monitoring sites and 20 reference sites, hence df = (120 - 2) + (20 - 2) = 136;

(iii) based on unexplained sums of squares for the two linear regressions and df, the right-hand term in the denominator = (22,502.7 + 978.5)/136 = 172.6;

(iv) based on sums of squares for the predictor variable (year of sampling: 0, 5, 8 & 11) for the two regressions, the left-hand term in the denominator = (1,641.3 + 330.0)/(1,641.3 * 330.0) = 0.00364; hence

$$\boldsymbol{t} = \frac{(2.07 - 0.95)}{\sqrt{[0.00364 * 172.6]}} = 1.41, \text{ and with 136 df, } P = 0.16,$$

suggesting that the two regression slopes are not significantly different at the $P = 0.05$ level.

Multiple land types across a heterogeneous sub-IBRA region. For this third example, we found means in cover on monitoring sites differed significantly for the four time periods (Table 6.5). These differences were found in both the repeated and grouped datasets, as tested by repeated measures and one-way analysis of variance, respectively. The trend over time on monitoring sites was significantly positive and linear (b = 2.25, r^2 = 0.20, P < 0.001) (Fig. 6.1C). The trend for the reference sites was also positive (b = 1.22) and significantly linear (r^2 = 0.31, P < 0.001). These two trend lines (regressions) converged somewhat after 11 y, but slopes did not significantly differ at the $P_{.05}$ level (P = 0.10 for t = 1.67 at 266 df; calculations as in Box 6.4).

6.3.4. Reflections on ground-based monitoring

Selected indicator. In Sections 6.3.1, 6.3.2 and 6.3.3, we illustrated the application of different statistical methods for detecting significant changes in means of perennial ground cover, and in trends over time for these means. These methods would apply to other quantitative attributes and indicators measured on rangeland monitoring sites. The measure, perennial ground cover, was selected because it forms a major part of the rangeland monitoring data provided ACRIS by its State and Territory partners. Ground cover serves as an indicator of landscape function, which can be observed on field sites and can also be derived in the form of various indices from remote sensing-based spectral data (see Section 6.4.1). Specific site-based data for cover were chosen to illustrate the effects of spatial heterogeneity and scale (extent) on the ability of statistical methods to detect significant changes in cover; these effects are inherent in rangeland monitoring data.

Non-linear trends. In Section 6.3.3, we illustrated examples of changes in perennial ground cover over time that tended to have linear trends, which were adequately fit by linear regressions (Fig. 6.1). However, rangeland indicators can have non-linear trends, which overall may be declining, increasing or even fluctuating up and down. As examples of non-linear increasing trends, it has been observed that recovery of soil surface condition and vegetation indicators from a moderately damaged state trend slowly upward and then level off (Fig. 6.2A) whereas recovery of these indicators from a highly damaged state trend slowly upward at first, then trend upward more rapidly before levelling off (Fig. 6.2B) (e.g., Tongway and Hindley 2004, Ludwig et al. 1999). These non-linear trends can be described by a simple 'exponent rise to a maximum' equation and by a 'sigmoid' equation, respectively. These particular equations were chosen because their parameter values provide useful ecological information (Tongway and Hindley 2000, 2004). For example, the parameter sum (Y_0 + a) estimates the upper asymptote for the sigmoid curve, which provides information on the maximum perennial ground cover expected for the rangeland unit of interest given its climate, topography and soils.

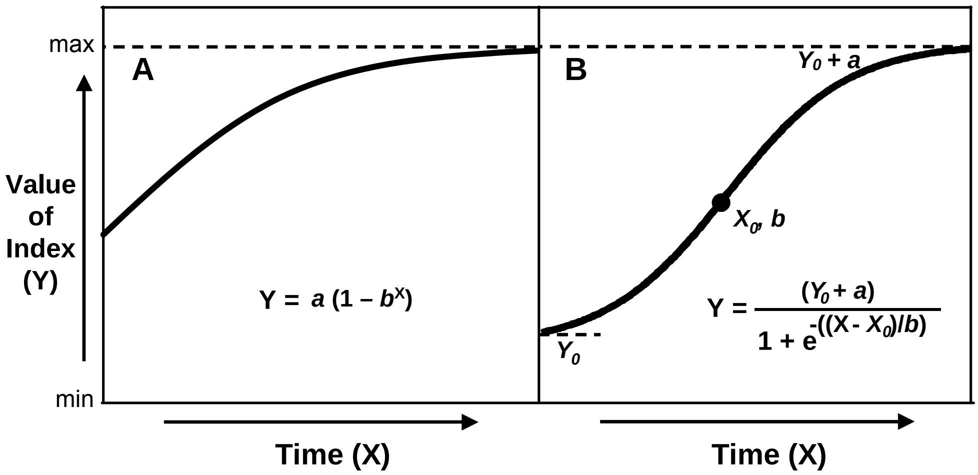


Figure 6.2. Typical non-linear trends observed for recovery of landscape function indicators (e.g., ground cover) from a state of (A) moderate damage where the index starts at an intermediate value and recovery is fit by an equation for a simple exponent rise to a maximum, and (B) heavy damage where the index starts at a low value and recovery is fit by a four parameter sigmoid equation. Equation parameters *a*, *b*, Y_0 and X_0 are optimised to fit data using non-linear curve fitting programs.

Transformed indicators. We applied simple statistical methods directly to perennial ground cover data. An alternative would be to transform ground cover by expressing its values relative to an environmental factor such as the amount of rainfall in a specified period prior to the time ground cover was observed. These transformed values of ground cover per unit rainfall essentially reflect 'rain use efficiency', which has been demonstrated as a useful response variable in arid and semi-arid rangelands (e.g., Le Houerou 1984, Ash et al. 1997).

Modelled indicators. Another approach would be to use models to predict variations in plant cover (or its related variable, biomass) over time from not just recent rainfall but from other explanatory factors or driving variables such as temperature and incoming solar radiation. Such predictive models can be empirical, such as multiple (many factor) regression models, or they can be mechanistic, such as computer simulation models that incorporate environmental factors as drivers of the biological processes that produce plant biomass and cover (e.g., AussieGRASS; Carter et al. 2000, 2002; Hall et al. 2001). Models applicable to rangelands have been reviewed by Stafford Smith (1988) and the NLWRA (2004).

6.4. Detecting changes in indicators acquired by remote sensing

As a tool for monitoring and assessing the condition of rangelands, remote sensing has a number of useful applications. These applications have been reviewed elsewhere (e.g., McVicar and Jupp 1998, Pickup et al. 1998b, Wallace et al. 2004, Bastin et al. 2006, 2008), and will not be repeated here except to briefly discuss three particularly important uses of remote sensing: (i) to assist in stratifying rangelands into relatively similar units for designing monitoring programs, (ii) to identify rangeland areas undergoing rapid changes to assist in where to establish monitoring sites, and (iii) to monitor indicators to detect changes in the functional state of rangelands relative to baselines and reference areas. However, before addressing these three issues, we review some commonly used indices derived from the spectral reflectance data acquired from satellites, although this review will be brief because details are provided elsewhere (e.g., Karfs et al. 2000, see their Appendix 3; McVicar et al. 2002, 2003; Bastin et al. 2006).

6.4.1. Indices for monitoring rangelands by remote sensing

For passive remote sensing, sensors on board satellites measure and transmit to receiving stations on earth the amount of electromagnetic radiation being reflected off the earth's surface. These acquired reflectance data are in discrete electromagnetic wavelengths (multispectral), typically in the visible and near-infrared wavelengths. For example, the four bands of wavelengths commonly used from sensors on-board the Landsat Thematic Mapper (TM) satellite include blue (0.425-0.525 microns), green (0.525-0.6 microns), red (0.6-0.7 microns) and near infrared (0.775-0.9 microns). The amount of reflectance in these spectral bands is measured and recorded as a number between 0 (no reflectance) to 255 (maximum reflectance). Each measurement corresponds to an area on the surface, which depends on the resolution of the sensor on a given satellite and its position in space (Table 6.6). For example, sensors on Landsat-TM record data in a grid of cells (pixels) where each pixel has a spatial resolution of 30-m directly below the satellite sensor (nadir).

For monitoring rangelands, the reflectance data acquired by these sensors is used to compute various indices related to surface attributes. For example, the Red band can be used to estimate an index for vegetation cover (Table 6.7) because low spectral values for Red relate to higher amounts of vegetation cover and high values correspond to low vegetation cover or bare soil (Karfs et al. 2000). Values for the Red band can be scaled to a proportional cover where zero equates to bare ground and one to the highest cover for the rangeland area of interest. An application of this vegetation cover index to rangelands in South Australia suggested that this simple vegetation cover index (VCI) is robust for actively growing and dry vegetation, and for brown, reddish-brown or light-coloured soils (Bastin et al. 2006), but may be less robust for grey or dark-coloured soils.

Table 6.6. Characteristics of archived data from satellites and sensors commonly used to monitor rangelands. Satellites are listed from largest to smallest pixel size (i.e., lowest to highest spatial resolution). Although sensors on-board these satellites are multispectral, only those spectral bands typically used to compute indices for rangelands are listed.

Satellites (first year)	Sensor name	Pixel size	Spectra colour	Band wavelength	Frequency (repeat cycle)
NOAA[1] (1978)	AVHRR	1100 m	Red	580-680 nm	12 hours
			NIR[5]	725-1100 nm	
NASA[2] (2000)	MODIS	250 m	Red	600-700 nm	2 days
			NIR	700-1100 nm	
Landsat[3] (1972)	MSS	80 m	Green	500-600 nm	18 days
			Red	600-700 nm	
			NIR	700-800 nm	
Landsat[3] (1983)	TM[4]	30 m	Blue	450-520 nm	16 days
			Green	520-600 nm	
			Red	630-690 nm	
			NIR	769-900 nm	

[1] A series of USA National Oceanic and Atmospheric Administration satellites with Advanced Very High Resolution Radiometer sensors (AVHRR) on-board.

[2] Two USA National Aeronautics and Space Administration satellites, Terra and Aqua, carry MODerate resolution Imaging Spectro-radiometer (MODIS) sensors, which also acquire data at 500 m and 1000 m pixel sizes. The coarser resolution sensors acquire data across a larger range of bandwidths.

[3] A series of USA NASA satellites where Landsats 1-3 carried a Multi-Spectral Sensor and Landsats 4 and 5 carried the Thematic Mapper sensor.

[4] The TM sensor on-board the Landsat 7 satellite, launched in 1999, was enhanced to include a number of other spectral bands, hence this sensor is often referred to as ETM.

[5] NIR = near infrared wavelengths

The vegetation greenness index, NDVI (Table 6.7), has been extensively applied to rangelands (Cridland and Fitzgerald 2001), with some examples described below. NDVI has also been widely applied to other vegetation types (Cridland 2000, McVicar et al. 2003, ESCAVI 2007). However, it appears to be most reliable where vegetation cover is higher, unlike most rangelands, and where there is a desire to monitor 'greenness' in response to recent rains (McVicar and Jupp 1998). In rangeland situations, where cover is typically lower and where longer term effects are of interest, soil adjusted indices such as $MSAVI_2$ and PD54 (Table 6.7) is more useful (e.g., Bastin et al. 1995, 2006).

Table 6.7. Indices of earth surface attributes derived from archived remote sensing data commonly used to monitor rangelands. Detailed descriptions and examples of applications are referenced.

Index Name	Brief description	Example Applications
VCI	Vegetation cover index based on the Red band scaled from 0 (bare soil) to 1 (highest cover)	Karfs et al. 2000 Bastin et al. 2006
GCI	Ground cover index based on multiple regression of bare soil cover onto Landsat TM spectral bands	Scarth et al. 2006
NDVI	Normalized difference vegetation index based on a ratio of Red and NIR bands	McVicar & Jupp 1998 Cridland 2000
$MSAVI_2$	2^{nd} Modified soil-adjusted vegetation index, which is aimed to improve NDVI for low covers	Huete 1988 Qi et al. 1994
PD54	Perpendicular distance from bare soil index based on Red (5) and Green (4) bands of Landsat MSS	Pickup et al. 1993 Bastin et al. 1993a,b

6.4.2. Stratifying rangelands into monitoring units

Images acquired by airborne and satellite sensors have been widely used to assist in stratifying and mapping landscapes into units with similar attributes (e.g., Wallace et al. 2006). Land unit maps are important tools for communications among stakeholders monitoring and managing rangelands. Land unit stratification and mapping can be conducted at a range of scales to serve the different purposes of stakeholders. For example, State/Territory Government land management agencies typically operate at broad regional scales (e.g., Neldner 2006), whereas individual rangeland enterprise managers operate at finer property, paddock, landscape unit and monitoring site scales (Gibbons and Freudenberger 2006).

For national reporting, for example, Australia's rangelands have been stratified into broad bioregions with sub-regions mapped within these bioregions (IBRA 2008). Also at a broad scale, vegetation types occurring in the tropical savannas and rangelands of northern

Australia have been mapped (Fox et al. 2001), which are available as two map sheets at 1:2,000,000 scale and also digitally at 1:1,000,000 scale. To help identify and conserve remnant habitats, these broad savanna vegetation types have also been mapped into regional ecosystem units at 1:50,000 and 1:100,000 scales (e.g., Neldner et al. 2005).

Ground-based pasture sites and biodiversity survey sites are usually located in stratified landscape units within paddocks, and this design has been referred to as 'point' monitoring (Fig. 6.3A). Specific rangeland monitoring designs vary with each State/Territory in Australia, as described in Appendix 1 of Bastin and ACRIS-MC (2008). Remote sensing-based monitoring has the advantage of providing complete spatial coverage of the rangeland area of interest, rather than a sampling, and has been referred to as 'population' monitoring (Fig. 6.3B). As illustrated in this figure, the rangeland area being monitored can be stratified relative to distance from water (as a surrogate of grazing distribution or pressure – grazing gradients) and used to explore for spatial pattern using Landsat or airborne imagery; this approach is referred to as 'pattern' monitoring (Fig. 6.3C) (Bastin et al. 1993a,b; Pickup et al. 1998a). An example of this pattern-based monitoring is described in Section 6.4.4. Rangeland areas being monitored by remote sensing can also be very broadly mapped into land units at region, sub-region and catchment scales where lower-resolution assessments are usually conducted using the spectral data in the large pixels of coarse-grained satellite imagery (e.g., Cridland 2000). An example of this application is described below in Section 6.4.3.

Remote sensing-based mapping and analysis of images that have been archived over time have proven useful for identifying areas that are apparently changing, which can then be targeted for on-ground monitoring and assessments. For example, analysis of maps produced by remote sensing-based imagery have been used at broad regional scales to identify 'unstable' rangeland areas, These areas are identified by specific changes in 'green flush after rainfall' vegetation indices (the normalised difference vegetation index, NDVI; Cridland 2000, Cridland and Fitzgerald 2001). These changes in 'greenness after rainfall' NDVI analyses take advantage of archived images from NOAA satellites with Advanced Very High Resolution Radiometer (AVHRR) sensors. Pre-1991 archived images are standardised to 5-km pixels and archived images since 1991 are at a 1.1-km pixel resolution. Map sequences can be examined at regional scale for areas of abnormal or unexpected changes in 'greenness', which are areas that can then be targeted by ground-based or high-resolution remote sensing-based monitoring and assessments.

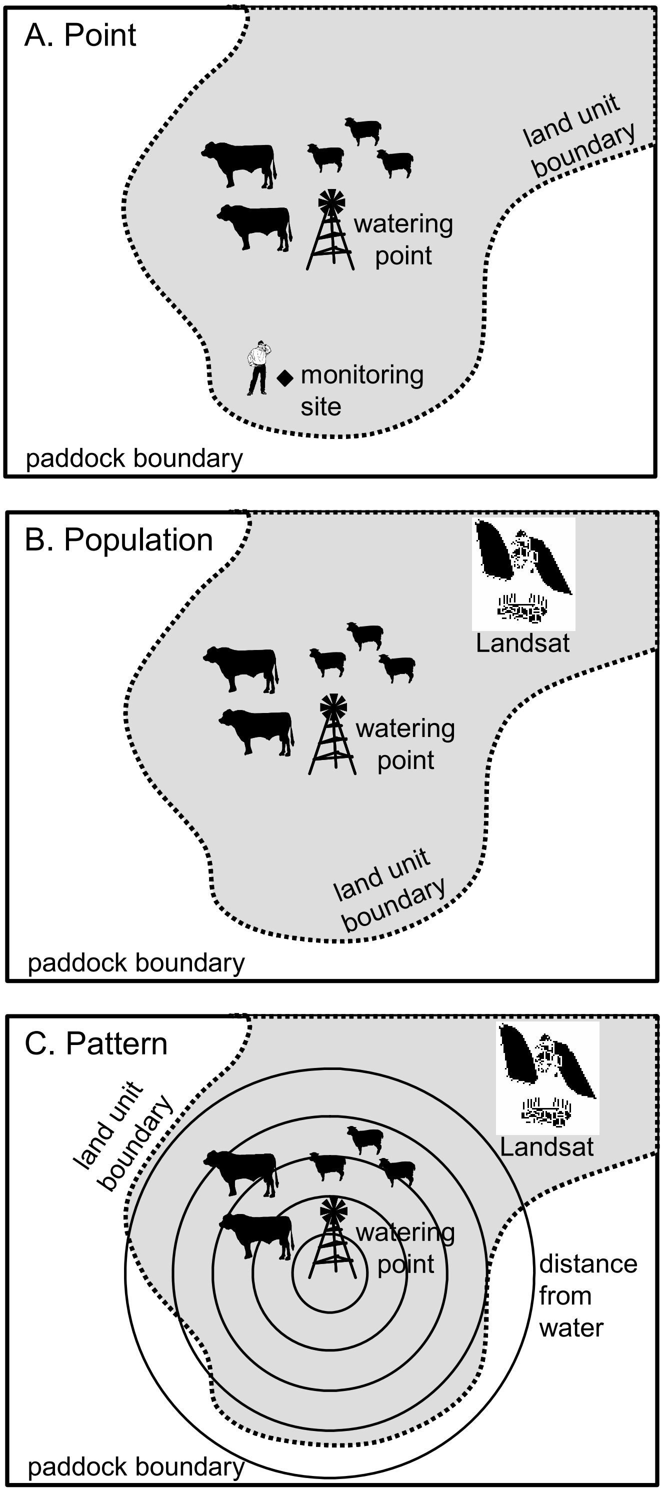


Figure 6.3. Rangeland monitoring approaches for assessing a landscape unit (shaded) within a grazed paddock: (A) on-ground, site-based ('point') measurements of attributes and indicators, (B) remote sensing-based assessments of spectral signature-based indicators for all the pixels ('population') within the unit, and (C) remote sensing based assessment of indicator shifts with distance from water ('pattern') within the unit. Point, population and pattern terminology follows Pickup et al. (1998b) and Bastin et al. (2008).

6.4.3. Identifying rangeland areas to monitor

Archives of Landsat Multi-Spectral Scanner (MSS) and Thematic Mapper (TM) imagery for the rangelands of Australia go back to about 1980, and have also been used to map and identify areas undergoing unexpected changes in vegetation cover. These 'Land Cover Change Analyses' (LCCA) have helped target areas for field-based assessments and for planning management actions in the rangelands of Western Australia and the Northern Territory (Karfs et al. 2000, Wallace et al. 2006).

Recent airborne surveys of rangelands have been compared with archived aerial photographs to help identify and assess landscapes where changes have occurred that drastically altered hydrological processes, which resulted in altered states of landscape function and vegetation patterns (Pringle and Tinley 2003). These analyses identified 'nick-points' in landscapes that required field-based assessments and monitoring. These nick-points and altered landscapes are often missed when pastoral monitoring programs are established in a rangeland area (Pringle et al. 2006), which leads to conflicting evaluations of rangeland condition (see Section 7).

6.4.4. Monitoring and detecting changes in rangeland indicators

Monitoring changes. As noted above, remote sensing-based data back to the 1980s are provided by archives of NOAA AVHRR and Landsat MSS-TM imagery, and these archived data have been used to derive indicators such as vegetation 'greenness after rainfall' and vegetation cover. These indicators of how rangeland areas change and respond to rainfall can be analysed to detect significant trends through time, ideally in reference to benchmarks or baselines. For example, Cridland (2000) has provided a continental-wide perspective of NDVI 'greenness' for each rangeland bioregion in Australia. Cridland's analyses provided yearly pixel-based regional maps of minimum, maximum and average 'green flush after rainfall' values, which were colour-coded to highlight patterns in the degree of change in NDVI. These analyses also provided maps of significant deviations in 'green flush after rainfall' relative to the long-term average for each rangeland bioregion. These longer-term, coarse-grained, regional scale maps of vegetation 'greenness' provide useful historical context, but are less useful for indicating changes in the amounts of perennial vegetation cover at finer paddock scales, which are important for detecting changes due to grazing managements (Bastin et al. 1995).

Another application of remote sensing-based imagery is land cover change analysis (LCCA), which is based on archived time series of Landsat MSS/TM and MODIS imagery (e.g., Bastin et al. 2006, Wallace et al. 2006). Relative to AVHRR-based NDVI analyses, LCCA represents a finer-scale procedure for rangeland monitoring and detection of trends because of the higher pixel resolution of MODIS (250 m) and Landsat (30-m) (Table 6.6). In the cattle producing rangelands of northern Australia, the LCCA procedure has focused on monitoring and mapping using a cover index (Table 6.7), where imagery is utilised from the late dry season when primarily perennial vegetation cover persists (Karfs et al. 2000, Wallace et al. 2004, 2006). These LCCA also examined deviations from longer-term baselines or reference areas, where these areas were defined by sites located in paddocks with low cattle grazing, which consistently had high perennial ground cover regardless of seasonal quality. These reference areas are also assessed as being in good condition from a pastoral viewpoint. In these northern rangelands, LCCA has proven useful in detecting trends away from baselines using colour-coded maps to highlight grazing areas within regions where seasonally adjusted trends have been above or below expected baselines (e.g., Karfs et al. 2000).

An example of the application of LCCA for sheep producing rangelands in southern Australia is described by Bastin et al. (2006). Initially, longer-term analyses of changes to provide a historical (30+ y) context were provided participating wool growers using Landsat-derived maps of vegetation cover and NOAA AVHRR-derived maps of vegetation greenness or NDVI (see Tables 6.6 and 6.7). Then, MODIS imagery, which has been archived and available free of direct costs since late 2000, was used to derive VCI and $MSAVI_2$ indices and maps of seasonal and yearly changes. These LCCA were largely focussed on paddock and property scales to assist individual producers with stocking rate decisions, but regional analyses were also conducted to provide producers with a feeling for how they were tracking compared to regional benchmarking.

A remote sensing-based methodology that explicitly monitors changes in the functional state of a rangeland area relative to a benchmark is the grazing gradient method (Bastin et al. 1993a; Pickup and Chewings 1994; Pickup et al. 1994). This method is based on the ecological concept of resilience, which is “a measure of the persistence of systems and their ability to absorb change and disturbance and still maintain the same relationships between system variables” (Holling 1973). In rangelands subject to grazing, the amount of ground cover (or standing biomass) will decline to relatively low levels during dry periods with the lowest amounts occurring near watering points (Fig. 6.4). After significant rains, ground cover increases to a higher level, reflecting the resilience of the system (Bastin et al. 1996). If the amount of cover near the watering point recovers to the same level (dotted line) as the area far from water (the benchmark level), the assumption is that there has been little or no permanent grazing effect because the system is fully resilient. However, if the level of cover observed near and trending away from water remains below the benchmark level or reference line (i.e., trend analysis; Pickup et al 1998a), then this departure is likely to reflect long-term or relatively permanent effects of grazing on the vegetation.

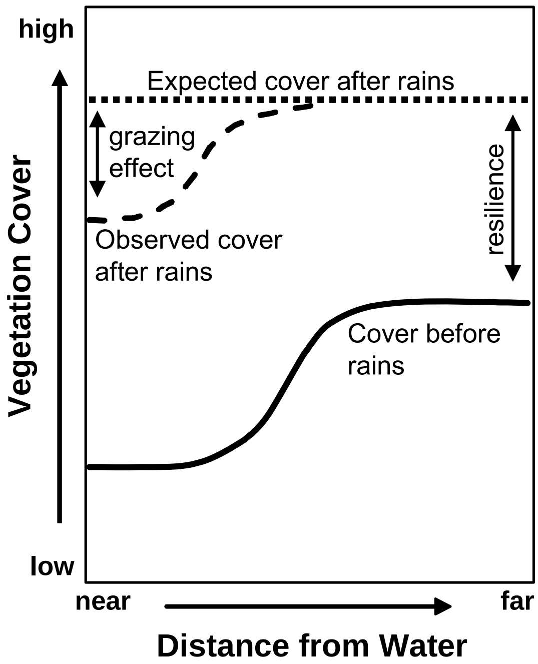


Figure 6.4. A framework for the grazing gradient method where the resilience of vegetation cover near stock watering points indicates the effects of long-term grazing (modified from Fig. 1a in Bastin & Ludwig 2006).

The grazing gradient method uses time-series analyses of archived Landsat MSS and TM imagery to derive a suitable index of vegetation cover (e.g. PD54; Bastin et al. 1993b, Pickup et al. 1993, 1994) (Table 6.7). Patterns in the cover index with distance from stock watering points were examined using pixels within concentric circular areas (Fig. 6.3C). These patterns are then interpreted relative to the grazing gradient resilience framework (Fig. 6.4) to assess any changes due to grazing effects and to evaluate rangeland condition. The grazing gradient approach has been tested and proven useful for assessing a number of different rangeland areas (Bastin et al. 1993a, 1996, 1998; Pickup and Chewings 1994; Pickup et al. 1994, 1998a; McGregor et al. 1999; Brook and Fleming 2001), being most useful for large paddocks typical of arid and semi-arid rangeland regions where livestock are dependent on point sources of water and where paddocks are large enough to detect distinctive grazing gradients.

Detecting changes. Detecting statistically significant changes in data for indicators derived from remote sensing imagery can use the same statistical methods illustrated in Section 6.3, but their application requires a different viewpoint and terminology from that used to detect significant differences and trends in site-based data. Because remote sensing-based, spectral signature-derived values for indicators of the functional state of rangelands, such as perennial vegetation cover (Table 6.7), are estimated for every pixel within a rangeland area of interest, sampling is complete (i.e., data represents a population; Fig. 6.3B). In this figure, for example, the mean vegetation cover for a landscape unit within a paddock at a given time period would be a fixed population mean parameter, μ. This population parameter differs conceptually from the sampling statistic, $\hat{y}$, which is used to estimate this parameter based on a sample drawn (randomly) from the population, where $\hat{y}$ varies from sample to sample (see Chapter 2 in Quinn and Keough 2002). This viewpoint on the population mean parameter also applies to the variance and standard deviation parameters for the complete population of observations.

This same population parameter viewpoint and terminology also apply to a mean for an indicator obtained for a number of landscape units assessed across a rangeland paddock, property or broad region. Statistical tests can still be applied to detect for statistically significant differences in (i) population means between two points in time, (ii) between population means for monitoring units and reference areas, and for (iii) trends in population means over time, as illustrated in Section 6.3. These tests will not be repeated here, except to note that they are now based on fixed or known mean and variance population parameters (e.g., population-based t statistics) or on trend parameters (e.g., population-based regression slopes), not variable sample statistics, which are used to estimate and draw inferences about population parameters (Quinn and Keough 2002).

6.4.5. Reflections on remote sensing-based monitoring

Matching methods to questions. Indicators such as NDVI that reflect 'greenness after rains', which are derived from NOAA AVHRR satellites having broad and frequent coverage, have proven useful for applications such as the identification of general regions that are (i) undergoing rapid changes to help target areas for rangeland monitoring (Cridland and Fitzgerald 2001) and (ii) having a long run of poor seasons to designate regions with exceptional droughts (McVicar and Jupp 1998, Holm et al. 2003). Because archives of NOAA AVHRR date are available from the late 1970s (Table 6.6), AVHRR-based NDVI has also proven useful for providing a historical context for more recent MODIS-based rangeland monitoring (e.g., Bastin et al. 2006). However, there is the temptation to apply broad-scale remote sensing-based monitoring to questions about changes in the functional state of rangelands when the low rigour of answers to such question using coarse-grained imagery and low-resolution analyses are not fully appreciated (Ludwig et al. 2007c).

For example, monitoring the health of a rangeland watershed by remotely detecting the size and configuration of patches of bare soil, which is a critical attribute for indicating the capacity of a watershed to retain water and soil (Bartley et al. 2006, Ludwig et al. 2005, 2007a), has generated statistically significant regressions of mean bare patch size onto percent bare ground derived from AVHRR imagery using data from 129 sites in south-central New Mexico (DeSoyza et al. 2000); these sites were part of a State-wide analysis of land cover changes (Minor et al. 1999). However, the variation in bare patch size accounted for by this remotely-sensed cover attribute was only 11%, far too low in rigour to confidently estimate this important indicator of landscape function.

This example illustrates the importance of addressing questions of rangeland condition or health with a method applicable to the scale of the problem (Bastin and Ludwig 2006, Ludwig 2007, Ludwig et al. 2007c, Pringle et al. 2006). AVHRR imagery is useful for broad-scale mapping, for identifying general rangeland areas with potential problems, and for providing a longer-term broad context for more recent MODIS time-series rangeland monitoring (Bastin et al. 2006). However, finer-scale imagery such as Landsat (30-m pixels), and even finer-scale imagery such as Quickbird (2.4-m pixels), is required to detect and interpret smaller areas with critical problems (Bastin and Ludwig 2006, Wallace et al. 2006, Ludwig et al. 2007c), although costs of fine-scale imagery can be prohibitive. Some critical problem areas can most effectively be identified by surveillance from aircraft and subsequent investigations on the ground, such as identifying 'nick-points' in a watershed where small areas of active erosion can initiate long-term and profound changes to the hydrology and ecology of the broader watershed (Pringle and Tinley 2003, Pringle et al. 2006).

Matching method to scale. As noted above, monitoring based on AVHRR NDVI usually lacks the resolution required to detect paddock scale changes in rangeland condition due to shifts in rangeland management (Bastin and Ludwig 2006). Therefore, the causes of rangeland changes as monitored by coarse-grained NDVI are not readily separated into disturbance (e.g., grazing) effects and natural (e.g., climate fluctuation) effects (Bastin et al. 2008).

Analysis of changes at the paddock scale are inherent in the grazing gradient method, but studies have demonstrated that the grazing gradient method is most useful in arid and semi-arid rangelands where grazing management is extensive (e.g., the Barkly Tableland; Pickup et al. 1994, Bastin et al. 1996, Brook and Fleming 2001). In these rangelands paddocks are typically large so that areas exist within the paddock that are well away from water (e.g., > 8 km), hence, subject to little grazing; these distant areas can then serve as important reference areas to establish benchmark levels of expected levels of cover (Fig. 6.4). The grazing gradient method also inherently separates short-term grazing effects from long-term grazing management and natural climatic variability effects (Bastin et al. 1996, Pickup et al. 1998a). For example, if palatable perennial vegetation does not recover near a stock watering point after rains, but only ephemeral vegetation, this indicates a long-term grazing effect. Although the vegetation resilience-based grazing gradient method has proven useful for assessing arid rangelands, it also has a number of constraints (see Bastin et al. 1999, 2002, 2008; Pickup et al. 2000), such as being difficult to communicate to stakeholders and being of limited use in wetter rangelands where paddocks are smaller and grazing effects near water points are less apparent.

The LCCA method has proven useful for detecting changes at the paddock scale in northern Australia (Karfs et al. 2000), southern Australia (Bastin et al. 2006) and Western Australia (Wallace et al. 2006). By using time-series analyses of MODIS and Landsat MSS and TM spectral data to estimate the mix of ephemeral and perennial herbs and woody shrubs and trees in a paddock, LCCA can infer likely changes due to grazing management, such as shrub thickening, versus those due to climatic fluctuations (Wallace et al. 2004, 2006; Bastin et al. 2006). The mix of vegetation for heavily grazed areas can be compared to similar landscapes with little or no grazing (reference areas). If LCCA indicates a loss of perennial forage and a gain of unpalatable shrubs and/or short-lived ephemerals in a grazed paddock, this infers a long-term grazing management effect. LCCA has also proven useful for (i) monitoring pastures in rangelands where tree canopy cover does not substantially 'hide' forage in the ground layer, (ii) detecting changes at paddock, property and catchment scales, and (iii) monitoring relative changes over time rather than precisely measuring vegetation or bare soil cover at any point in time (Karfs et al. 2000, Bastin et al. 2006, Scarth et al. 2006, Wallace et al. 2006).

Recent advances in remote sensing. Improvements continue in remote sensing technologies, including the types of sensors on satellites (e.g., hyper-spectral, very high resolution), modes of data acquisition and storage, and the power of image processing. Computing packages, such as RANGES (Qi et al. 2002), have been developed to assist with the routine processing of remote sensing imagery. Perhaps most important, have been recent reductions in the costs of archived and newly acquired imagery. For example, archived MODIS imagery back to late 2000 is freely available for downloading from websites such as http://modis.gsfc.nasa.gov/ (accessed 12 Apr 2008).

These technological and economic advances will help overcome one of the main limitations of using remote sensing for monitoring rangelands, which is that the types of indicators that can be derived from remote sensing-based spectral-signature data are very limited compared to what can be measured on ground-based sites. For example, 17 ground-based indicators were measured on 11 sites in southeast Arizona (Buono et al. 2005), but only three indicators were considered measurable using Landsat TM imagery. These three indicators were canopy cover, plant biomass and mesquite composition, which were used as proxies for bare soil (site stability), annual production (site potential) and species invasions (weediness), respectively. Although no significant differences (± 95% C.I.) were found between these three remotely-sensed and ground-based indicators, these authors caution that additional studies were needed to establish their relationships.

Advances continue in establishing robust relationships between indices derived from the spectral-signature data of sensors carried on satellites and ecologically meaningful measures on the ground (e.g., Wallace et al. 2004, 2006; Bastin et al. 2008). These advances are occurring at multiple scales (Ludwig 2007), and include (i) broad-scale on-ground applications of NOAA AVHRR-based ‘greenness’ indicators (Lu et al. 2003), (ii) moderate-scale applications of NASA MODIS to derive vegetation cover indices for monitoring changes on wool producing rangelands in southern Australia (Bastin et al. 2006) and beef producing rangelands in northern Australia (pers. comm., Kate Richardson, Department of Natural Resources, Environment and the Arts, Northern Territory), (iii) fine-scale links of Landsat TM-based landscape function indicators to detect grazing-induced changes in biodiversity (Karfs and Fisher 2002), and (iv) very fine-scale studies using Quickbird imagery (2.4-m pixels) to derive ground cover values to validate a new landscape function indicator, the landscape leakiness index (Ludwig et al. 2007b), which is being applied at the sub-catchment scale (pers. comm., Gary Bastin, CSIRO’s Water for Healthy Country Flagship Program).

7. Evaluating the condition of rangelands

Issues on evaluating rangeland condition or health can be viewed as having two main components: first, how do individual stakeholders assess the information available on the functional state of the rangeland area of interest and judge its condition relative to their visions and values, and second, if the area of rangeland is being evaluated by multiple stakeholders having different visions and values, how to best resolve any conflicting statements on the condition or health of the rangeland. We discuss these two issues separately.

7.1. Assessing rangeland condition

We emphasised early on in Section 4 (step 1, rangeland monitoring) that evaluations of rangeland condition are "in the eye of the stakeholder", because stakeholders assess or judge information on the functional state of rangelands relative to their visions and goals, in other words, they evaluate condition through their own 'values prism' (see Fig. 4.6). Information on the functional state of a rangeland is provided by those measuring attributes and indicators on monitoring sites, or as acquired by remote sensing (Sections 5 & 6).

An issue for providers of information on the state of a rangeland is how to encourage stakeholders to broadly assess all the data available rather than narrowly focus on a few specific indicators of prime interest to them. In Section 4.6 on reporting results of analyses on indicators we recommended that information providers present data on each indicator, and for environmental factors such as rainfall that co-vary with these indicators, in the form of colour-coded maps and simple 'time-trace' graphs (see Fig. 4.2). Then, we suggested that these time-trace graphs for each indicator be converted to colour coded time-mark graphs where values for the indicator, coded for seasonal quality and management effects at different time periods, are positioned as marks along a continuum ranging from a maximum value representing a fully functional rangeland system to a minimum value for a totally dysfunctional system (see Fig. 4.3). For final presentation to stakeholders, we then recommended that the colour-coded time-mark continuum graphs for different indicators be synthesized into one graph with a consistent style (see Fig. 4.4). If the number of indicators is large, they can be grouped into themes and presented as a set of graphs. We recommended this synthetic style of presenting data for multiple indicators because it leads stakeholders to viewing a wealth of information on many different attributes and indicators rather than focussing on one piece of information. Although each stakeholder will still assess rangeland condition or health somewhat differently depending on their values, viewing a broader range of information will help them appreciate rangelands as functional systems serving multiple purposes; this appreciation can help resolve multiple use conflicts.

7.2. Resolving conflicting evaluations of rangeland condition

As noted earlier, statements issued on the condition of rangelands are often conflicting and require resolution. In some cases, statements are strongly conflicting and resolution has been required at the highest levels of government. For example, in the United States, two conflicting reports on the condition of public (leasehold) rangelands administered by the Bureau of Land Management (BLM) were so different that they had to be reconciled at the US House of Representatives level (GAO 1991). One report, issued in 1989 by the US National Wildlife Federation and the Natural Resources Defense Council (NRDC 1989), stated that most of BLM' rangelands, which includes about 69 million ha (170 million acres), were in unsatisfactory condition, whereas a report at about the same time (BLM 1990), issued by BLM concluded that its rangelands were improving and in better condition than ever before. The GAO reconciled these conflicting reports by concluding that "the conclusions are not necessarily inconsistent with each other and can be attributed more to the context in which the available data on rangeland conditions were interpreted and presented than to differences in the data themselves." In other words, two different stakeholders in BLM's public rangelands evaluated rangeland condition from their own perspectives and value systems, another illustration of how rangeland condition is 'in the eye-of-the-stakeholder'.

In Australia, conflicting evaluations on the condition of native vegetation, in general, not just rangelands, is a concern of the Executive Steering Committee for Australian Vegetation Information (ESCAVI). To help resolve conflicts, ESCAVI has produced an "interim approach paper" to generate discussion on ways to improve national assessments of the condition of native vegetation by using better indicators (Parkes and Lyon 2006, ESCAVI 2007).

In the Gascoyne-Murchison region of Western Australia, conflicting assessments of rangeland condition appear to have arisen largely due to a narrower perspective using site-based monitoring plots compared to broader assessments conducted by aircraft-based landscape-scale surveys (Pringle et al. 2006). Both assessments were based on data collected in the late 1990s and early 2000s within the same general region. However, the site-based pasture monitoring was often conducted on more stable mid-slope areas, which usually miss those areas actively eroding or degrading. Pasture monitoring reported positive changes in perennial plants (Watson 1998, Watson and Thomas 2003) whereas aircraft-based assessments reported areas of desiccation and vegetation change (Pringel et al. 2006). Subsequent ground-based surveys of these latter areas suggested that landscape desiccation was due to hydrological changes at critical points in the landscape (i.e., rill and gully cutting; referred to as 'nick-points') (Pringle and Tinley 2003).

The issue is, how to best resolve conflicting statements on rangeland condition? We recommend a participative approach based on a wide body of theory on conflict resolution (e.g., Wall and Callister 1995) and participative approaches (e.g., Connor 2001), and on our own experiences with rangeland workshops (e.g., Smyth et al. 2003). Within a workshop setting and using role-playing scenarios (Fig. 7.1), participants share each other's visions and values, learn from other's experiences, and understand other challenges, all of which builds trust. By this participative process, different stakeholders embrace their differences and develop respect for each other's choices when dealing with management problems and when assessing the condition of a rangeland.

8. Recommendations

8.1. Rangeland condition: conflicting statements

It is not unusual to read conflicting statements about the condition or health of a rangeland area. In some cases these conflicts are because of differences in the spatial scale of reporting, for example, one statement may apply to a pastoral property or conservation park within a region whereas another statement may refer to overall condition within the region. Conflicts may also arise due to differences in the time period covered by different statements. However, in most cases, conflicting statements on the condition of a rangeland area are due to a lack of appreciation that assessments of rangeland health are value statements prepared by different users and, therefore, statements are going to differ depending on the specific visions and goals of each user.

Differences naturally occur even if all users are provided exactly the same information about the past and current functional states of the rangeland area being evaluated. We recommend that information providers and rangeland users:

- adopt the perspective that rangeland health is "in the eye of the stakeholder" where statements on condition will naturally differ because each stakeholder will evaluate the same set of measured attributes and indicators defining the functional state of a rangeland area from their own specific point of view; and
- apply a participative approach, such as a workshop setting, to resolve conflicting statements, where participants can share and learn from one another's experiences, understand each other's challenges, and develop trust and respect for each other's differences and choices.

8.2. Reporting changes

To evaluate health, stakeholders require robust information characterising the functional state of the rangeland area of interest. People providing rangeland monitoring data to the

ACRIS typically measure a number of functional attributes. When reporting changes in these attributes and indicators we recommend that information providers:

- use simple graphical procedures, such as positioning mean values for an indicator as time-marks along a continuous gradient of values (a continuum) ranging from a minimum to a maximum value. These min and max values should, ideally, be based on reference (benchmark) areas, but where this is not feasible, then end-points for the continuum can be defined hypothetically. For example, we know a hillslope with 0% perennial plant cover will retain a minimum amount of water during rain storm events relative to a maximum for a hillslope with 100% cover (or the highest cover expected for the climate, soils and topographic setting). Then mean values for perennial plant cover, measured at different points in time (including an initial time) on monitoring sites over a rangeland area, can be graphed and positioned as time-marks relative to these min and max values. We propose that time-marks be colour coded to infer management effects based on the matrix approach where colours relate to combinations of seasonal quality and direction of change in indicators (see Box 2.2. 'Matrix: seasonal quality and direction of change' in *Rangelands 2008 – Taking the Pulse*). These colour-coded time-mark data provide indicator by indicator information and a simple continuum perspective (examples in Section 4.6) that can be readily evaluated by stakeholders to assess the condition of a rangeland area relative to their goals and land use purposes.
- place a priority on monitoring a set of indicators that can be readily compared across Australia's rangelands, ideally those that can be derived by remote sensing-based methods (see below). The usefulness of indicators should be periodically evaluated by a participatory process where stakeholders meet in a workshop setting to examine how different attributes and indicators rate against a set of defined criteria (an example using eight criteria was provided in Section 7).

8.3. Detecting significant changes in rangelands

Stakeholders also require robust information on the functional state of their rangeland area of interest, that is, they need to have confidence that the changes being reported represent real or statistically significant differences, while being aware, of course, that statistical significance does not necessarily equate to ecological significance. First, the attributes and indicators being monitored should, ideally, lend themselves to being reliably measured so that any differences detected are not due to imprecise or inaccurate measurements. Second, attributes and indicators should lend themselves to being analysed statistically for detecting any significant changes in the state of the rangeland system. We recommend that information providers:

- use relatively straightforward statistical analyses to explore for differences in means between areas being monitored at two different points in time (e.g., *t*-tests), and for different directional trends over time or space (e.g., simple regressions). Analyses to determine the adequacy of sample sizes for detecting change are also needed. Example applications of these analyses are provided in Section 6.
- include in their monitoring and statistical analyses, where feasible, ancillary data on factors that help interpret whether changes in attributes and indicators are due to natural factors, such as differences in seasonal quality, or due to differences in rangeland management.

8.4. Methods for monitoring rangelands

Monitoring data on the functional state of rangeland systems can be acquired using a variety of methods, including ground-based and remote sensing-based approaches. The wider aim is to provide information so that the interests of a diverse group of stakeholders are met, which means that monitoring information must cover a broad range of spatial extents and time-frames. We recommend that rangeland information providers:

- use a combination of site-based and remote sensing-based data to report changes in the functional state of rangelands. In Australia, this is currently being done for the ACRIS, but the emphasis has largely been on site-based pastoral monitoring data. In the future reporting will require a greater use of remote sensing-based information, especially as remote technologies improve and costs decline. The advantages and limitations of site-based and remote sensing-based methods for monitoring rangelands are discussed in Section 4.
- include, where feasible, information for primary production and habitat conservation indicators that have been acquired collaboratively on the same set of monitoring/survey sites to achieve a value-adding of information for stakeholders.

9. Acknowledgments

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