

Applicability of Montreal Process Criterion 2 - productive capacity - to rangeland sustainability

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SUMMARY

Rangelands provide habitat for a wide array of plants and animals and forage for both domestic and wild herbivores. Estimating the cumulative area of rangeland in a country (Indicator 10) is complicated by how rangeland is defined and the scale at which range sites occur and are classified. Determining biomass production available for grazing (Indicator 11) including plantations and seedlings of native and exotic plants including cultivars (Indicator 12), and the amount of sustainable annual biomass removal (Indicator 13), and the removal of non-rangeland (\approx non-livestock) products (Indicator 14) is also difficult. Nevertheless, these indicators are just as applicable for monitoring sustainable management of rangelands as they are for forests. Productivity also varies with seral stage. Inclusion of multiple seral stages across the landscape generally increases the diversity of plant and animal species that occupy the landscape. Sometimes degraded landscapes require restorative practices; plant materials are available for rehabilitation plantings.

INTRODUCTION

The purpose of this paper is to assess how well Criterion 2, Maintenance of Productive Capacity, applies to rangelands. The June 1992 United Nations conference on Environment and Development, commonly called the Earth Summit, advanced the awareness of policy makers and scientists in both developed and developing countries about the importance of environmental sustainability for long-term economic development. The role of grazing animal production systems, both for domestic ungulates and wildlife, was a central part of that discussion.

Rangelands provide forage for domestic herbivores as well as for a wide array of wild

herbivores. A wide variety of other animals, ranging from insects to large carnivores, rely on these lands and their herbivores for habitat and an energy source. Expanding human populations are increasing the demand for agricultural products, including livestock products worldwide (Sere and Steinfeld, 1996). Food demands in developing countries have changed meat surpluses in the 1960s to a net deficit of nearly 2.5 million tons in 1987 (de Haan, 1991). However, increasing demand for forage to supply meat has started to cause cumulative losses in rangeland productivity, primarily because of erosion and desertification (Schlesinger et al., 1990; Heady

and Child, 1994). The importance of productive capacity of rangelands, therefore, is an essential element of sustainability at a national scale.

The traditional means of monitoring productivity on grazing lands have been well documented (Cook and Stubbendieck, 1986). A number of authors have correlated productivity with precipitation, evapotranspiration, temperature, and soils (Sims and Singh, 1978; Le Houerou *et al.*, 1988; Epstein *et al.*, 1997). This research has been based on evidence collected at the site level, which is not necessarily applicable when considering the measurement of sustainability at a national scale. Hierarchy theory states that ecosystem processes occurring at one level of a system are not indicative of system dynamics at a higher or lower scale (Allen and Starr, 1982). Nonetheless, valid syntheses have demonstrated broad relationships between abiotic factors and primary production (Webb *et al.*, 1983).

The Montreal Process conceived five indicators for Criterion 2, Maintenance of Productive Capacity, as it applies to temperate and boreal forests. They were numbered 10 through 14. As they apply to rangelands, the indicators would be: (10) Area of rangeland and net area of rangeland available for grazing; (11) total biomass of both palatable and unpalatable forage species on rangeland available for grazing; (12) area and production of native and exotic species; (13) annual removal of forage compared to the biomass determined to be sustainable; and (14) annual removal of non-forage rangeland products, e.g. non-forage plants or wildlife. We consider each of these indicators in turn.

The five indicators all refer to either area or biomass. In the following sections we evaluate their applicability to rangeland and rangeland products, principally forage. We also assess the status of data and protocols for monitoring these indicators.

Indicator 10: Area of rangeland and net area of rangeland available for grazing

Estimating the cumulative area of rangeland in a country is complicated by how rangeland is defined and the scale at which range sites occur and are classified. The former point is important because agencies responsible for national inventories of forests and rangelands can have different definitions of woodlands; some

inventories include woodlands in the rangeland base and some classify it as forested (Shiflet, 1980). Monitoring systems consequently have to keep track of how rangeland is classified in order to appraise the dynamics of rangeland area.

Another source of confusion can be whether to incorporate pastureland and grazed cropland in a country's area of rangeland. Although they are not normally considered as rangeland, the two categories do affect forage supply at a national scale which ultimately can modify grazing use on native rangelands. Additionally, there is an indistinct partition between rangeland and pastureland in some regions. Pasturelands result from human activities, and can be distinguished as being planted with introduced forage species or commercial cultivars of native forage species. Furthermore, pasturelands ordinarily receive periodic renovation using cultural treatments such as tillage, fertilization or irrigation (Graetz, 1994). In the United States, grazed cropland reached an apex of 3.5 million ha in 1969 before starting a slow decline to approximately 25 million ha in the 1980s (Joyce, 1989). The latter area almost equals one-tenth of all US rangeland, not including the state of Alaska (US Department of Agriculture, 1989).

The scale of rangelands is consequential in countries monopolized by montane ecosystems where mountain grasslands and meadows tend to be interspersed with temperate forested lands. Plot-based national monitoring programmes generally cannot provide statistically valid estimates of the extent of rangelands where they do not constitute a dominant land use within a region (Nusser and Goebel, 1997). Riparian areas are very important sources of forage in montane zones, providing up to 80% of forage consumed; however, in the same regions, they can comprise less than 10% of the landscape (Roath and Krueger, 1982).

A US national forestland monitoring programme, Forest Inventory and Analysis (FIA), has historically focused upon timber resources and has been in place for 70 years in the United States (US Department of Agriculture, 1958). The National Forest Management Act (NFMA) of 1976 expanded this programme to cover all renewable natural resources, including rangeland resources, on both public and private lands (Powell *et al.*, 1994). Notwithstanding the broadened authority provided by NFMA, FIA has not yet incorporated all rangelands into its sampling framework. Another national survey, National Resource

Inventory (NRI), was commenced in 1982 to monitor all uses on non-federal US lands (Nusser and Goebel, 1997); however, its sampling population does not include federally-owned lands. Thus, from a national rangeland perspective, FIA and NRI are neither collectively exhaustive nor mutually exclusive.

Remote sensing provides an alternative for categorizing the area and status of rangeland in countries that do not have adequate ground-based monitoring systems (Tucker et al., 1985), although researchers have shown that spectral data alone can be inadequate for identifying land cover and land use within a large geographic area (Janseen et al., 1990). The spatial differentiation, or pixel size, of satellite data varies widely and could effect accuracy of rangeland classifications where different categories are interspersed (Campbell, 1996). The advanced high-resolution radiometer (AVHRR) has been used to monitor broad-scale vegetation patterns through a 'greenness index' (Loveland et al., 1991). AVHRR data, with a 1-km pixel size, have been shown to provide an acceptable perspective of ecological conditions across large areas (Singh and Saull, 1988). Thematic mapper satellites are providing finer and more accurate spatial resolution and thus more detailed spectral information than earlier space-borne sensors (Bullock et al., 1994).

Assessing the net area of rangeland available for grazing presents a problem at a national scale. Standards and guidelines have been devised for suitability criteria for domestic livestock on individual pastures, but their scale is inappropriate for estimating the availability of rangeland for grazing across large areas. One approach would be to base this aspect of Indicator 10, at least in part, upon societal restrictions incorporated into protected area designations. No algorithms exist for determining the area of rangeland in a country that is unavailable for wild herbivores.

Indicator 11: Total biomass from rangeland available for grazing

In the United States, rangelands can be grouped into the general types of pinyon-juniper, hardwood-shrub and hardwood-juniper, shrubland, shrub steppe, grasslands, and tundra. Some forests are also sometimes considered as part of the rangeland resource (Holechek et al., 1998).

Forage production in forested areas is proportional to overstorey canopy cover (Mitchell and Bartling, 1991). Each rangeland type includes characteristic communities, often many communities (Kuchler, 1964; McArthur and Ott, 1996; Bailey, 1998). The total biomass available for grazing in the United States is unknown, but the range of forage production in rangeland types has been estimated. Using midpoint values for forage production and the area of that rangeland type, as estimated by US Department of Agriculture (1972, 1977) and reported by Holechek et al. (1998), there is an annual standing dry matter of forage production on US rangelands, excluding Alaska and Hawaii, of approximately 7.5×10^9 metric tons. Much of this is from forested lands. In general, certain grasslands (tallgrass prairie, California annual grassland) and forests (southern pine forest, eastern deciduous forest) are highest in herbage production, up to 3500-4000 kg/ha/yr, whereas some shrublands can produce less than 100 kg/ha/yr (Clary, 1989). The productive capacity of each site is dictated by many factors including soil fertility and mineral content, soil microflora, water availability, soil temperature, and solar irradiation (Black, 1968; Harner and Harper, 1973; Woodward et al., 1984; Trent et al., 1993; Pendleton and Warren, 1995). These factors vary in time and space. The variability in annual biomass production at a particular site is ordinarily greater by a factor of 1.5 than the variability in annual precipitation (Le Houerou et al., 1988). Two examples of temporal productivity variation in mesic and xeric ecosystems are provided by Mueggler and Stewart (1981) and Tew et al. (1997).

At any particular site, productivity can vary depending upon ecological condition or seral state. Often, but not always, late-seral sites have higher forage productivity than early seral sites (Uresk, 1990; Frost and Smith, 1991; Samuel and Hart, 1994). Differing seral conditions may be desired from a management standpoint depending upon management objectives (Uresk, 1990; Benkobi and Uresk, 1996). Some wildlife values are dependent upon a full range of seral stages to maintain productivity, density, and diversity (Rumble and Gobeille, 1985; Fritcher, 1998). Intermediate seral stages provide more forage for livestock production because of the abundance of palatable graminoids in most rangeland vegetation types (Uresk, 1990; Samuel

Table 1 Area of Bureau of Land Management rangelands in native perennial vegetation, seeded to exotic grasses¹ and dominated by annual grasses in the western United States (US Department of Interior, Bureau of Land Management, 1997)

State	Area (ha x 10 ³)		
	Native vegetation	Seeded grasses	Annual grasses
Arizona	4765	4.7	117.4
California	2464	27.9	614.2
Colorado	1892	99.6	0
Idaho	3645	644.2	290.5
Montana	3341	67.3	0
Nevada	12 120	363.2	0
New Mexico	4754	1.4	0
Oregon	5006	350.9	0
Utah	8110	284.3	0
Wyoming	4448	1.4	0
TOTAL	50 544	1845.0	1022.1

¹Nearly all of these lands were seeded to crested wheatgrass (*Agropyron cristatum*)

and Hart, 1994; Benkobi and Uresk, 1996). Other seral stages may be more important to different species of wildlife. Samuel and Hart (1994) and Benkobi and Uresk (1996) reported that biological diversity of plants was greater in early seral stage sites.

Currently, no protocols exist for defining or rating distributions of seral states across a landscape within a national monitoring system. Some landscape-level algorithms that describe patch size and shape, as well as boundaries and edges, may prove to be a useful way to summarize successional patterns within a region (Forman, 1995).

Indicator 12: Biomass of native and exotic rangeland species

As discussed under Indicator 11, the values for biomass production in the United States are not known with precision. This is also the case in other countries. However for the United States, the relative proportions of natural and introduced forage species may be inferred for the ten western states (Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, and Wyoming) having substantive areas under the administration of the US Department of the Interior, Bureau of Land Management (BLM) (Table 1). Ninety-seven percent of land in the BLM data base has a native vegetation. Most of

the approximately 7.5×10^9 metric tons of annual standing dry matter of forage production in the United States is undoubtedly comprised of native plants, although non-native weeds and seedings contribute to the total.

In US areas where plantations or seedings have occurred, many exotic species have been used. In the drier rangelands of the West, seedings of exotic species were a standard operating procedure during the mid-twentieth century (Plummer et al., 1955). The seeding procedure was to mix several species on lands degraded by overgrazing or wildfire or to enhance herbaceous plant productivity after clearing woody vegetation (Plummer et al, 1968). Usually, these seed mixes were dominated by exotic perennial grasses, but the current impetus is to limit the mixture to native plants (Roundy, 1996; Richards et al., 1998; McArthur and Young, 1999).

Cold desert rangelands in the western United States have had substantial areas seeded to wheatgrasses of the Graminae (grass family) tribe Triticeae (Holechek, 1981; Johnson, 1986; Asay, 1995). Holechek (1981) estimated that some 8×10^6 ha had been seeded to crested wheatgrass (*Agropyron cristatum*) alone; Sharp (1986) provided a lower estimate of 5×10^6 ha. Specific figures from a BLM data base by state show 1.8×10^6 ha of their lands have been seeded to crested wheatgrass. Substantial area of cold desert rangelands have been planted to other exotic forage plants, including those in the tribe Triticeae

(Interagency Committee, 1951; McArthur, 1988; Barnes *et al.*, 1995). In the hot deserts of the US southwest, exotic love grasses (*Eragrostis*), especially Lehmann love grass (*E. lehmanniana*), have become naturalized on thousands of ha (Cox *et al.*, 1990; Roundy and Biedenbender, 1995). Other exotic grasses and legumes are important pastureland plants that are also seeded in rangeland settings throughout the United States, e.g. tall fescue (*Festuca arundinacea*) and lespedezas (*Kummerowia* spp.) in the southeastern United States (McGraw and Hoveland, 1995; Sleper and Buckner, 1995). In fact, grasses and legumes are widely seeded on rangelands and pasturelands and around the world irrespective of their point of origin (McArthur, 1988; Barnes *et al.*, 1995). Thus, productivity of native and exotic rangeland species in the United States and other countries is not known, especially as productivity characterized by origin of the species. Monitoring efforts in the future may need to consider place of origin of the species if this information is deemed important.

Indicator 13: Annual removal of rangeland biomass compared to rangeland biomass determined to be sustainable

The removal of biomass from rangelands in a manner sustainable to productivity and stability requires an understanding of the community dynamics. Such an understanding is difficult given the vagaries of rangeland productivity (see discussion under Indicator 11) and basic changes in our understanding of successional dynamics.

Recent ecological theory has disputed the traditional Clementsian successional paradigm, especially for the arid and semi-arid landscapes typical of many rangelands. Instead of considering mono- or polyclimax vegetation communities on rangelands as the highest stable, steady states, many ecologists are now of the opinion that there are possibly multiple steady states. Furthermore, many ecologists believe that successional trajectories have probabilistic rather than pre-determined courses, and that association of particular plant species is much more random than previously thought (West, 1988; Smith, 1978; Westoby *et al.*, 1989; Friedel, 1991; Laycock, 1991; Johnson and Mayeux, 1992; Joyce, 1993; Tausch

et al., 1993). The major issue is linear succession to a single stable 'climax' versus plant successional trajectories to a variety of states.

At the site level, removal of forage biomass by grazing animals in relation to total biomass is termed utilization. Utilization information is considered a management tool useful for assessing livestock distribution and interpreting other longer-term data. Utilization is not, however, a substitute for long-term vegetation or soils data, nor is it well correlated with attributes associated with rangeland sustainability like riparian function or trends in secondary succession (Sanders, 1998). Utilization statistics, generally expressed as a percentage of total biomass present, do not lend themselves to aggregation across broader scales.

Nationally, removal of rangeland vegetation may best be expressed by other appropriate indicators. For example, the effects of overgrazing over a period of time will be manifested in Indicator 2 (extent of area by vegetation type and successional stage), Indicator 5 (fragmentation of vegetation types) and Indicator 18 (area with significant soil erosion). The range science literature is replete with mechanisms and examples of consequences of overgrazing (Ellison, 1960; Milchunas and Lauenroth, 1993).

Holecheck *et al.* (1998) summarized the relationship of biomass production and removal and sustainability at a site or management unit level:

- (1) The four basic components are proper stocking rate, proper timing of use, proper distribution, and proper grazing system. Proper stocking is the most important.
- (2) Lighter grazing intensities are necessary for sustainability in arid as compared to humid rangelands. Areas with long growing seasons, high amounts of growing season precipitation, deep soils, and flat terrain can withstand heavier grazing intensities than the contrasting situations.
- (3) Maintenance of desirable vegetation is ordinarily best for both short- and long-term returns (financial and ecological).
- (4) Conservative stocking rates is a low cost and low risk approach in improving forage production on most degraded ranges. However in some cases manipulative tools, e.g. fire, herbicides, seeding, may be required.

Indicator 14: Annual removal of non-rangeland products compared to the level determined to be sustainable

Traditionally, livestock grazing has been the primary use and focus of rangelands. However, in the United States, Hart (1996) points out that a smaller and smaller proportion of the population make their living from agriculture.

Rangelands, broadly defined, are a principal component of wildlife habitat and serve as full or part-time habitats to some 500 000-700 000 pronghorn antelope, 10 000 000 deer (half of the total population), 400 000 elk (nearly the total population), all of the big horn sheep and mountain goats, and 80% of the moose in the United States (Hart, 1996). There are many more species of smaller animals. Hart (1996) reported some 330 threatened or endangered animal species on US rangelands. Watts *et al.* (1989), Stacy (1995) and several contributors in Krausman (1996) document large numbers of mammals, birds, reptiles, and arthropods on rangelands.

Rangelands are a source of plant materials for restoring damaged or degraded sites by seed collection from native stands, providing the source materials for cultivar development, ornamental plant development and use, and for industrial and medicinal products (Carlson and McArthur, 1985; McArthur, 1988; Meyer and Kitchen, 1995).

CONCLUSIONS

Documenting and monitoring for Montreal Process Indicators 10 through 14 is difficult and not being adequately accomplished in the United States or other countries. Long-term data are limited for rangelands. Steady states may exist for many years before changes occur in species composition (Samuel and Hart, 1994). Techniques to measure successional changes have not been sensitive enough to detect change in plant composition unless a major change has occurred (Uresk, 1990). An understanding of plant successional processes is fundamental to predicting responses to disturbances, i.e. grazing, fire, drought, and insect and disease irruptions. Future climates and other disturbance issues

including CO₂ elevation and species introduction may drive vegetation composition away from present vegetation composition on various scales (Polly, 1997). However, even with these uncertainties, maintaining the health of rangelands so that the land, biological diversity, and its inherent productivity can be sustained would be, it seems to us, the principal management objective for rangelands.

Use of plant materials for rehabilitation or increasing rangeland production should be approached with care. When native, site-indigenous plant materials will serve to meet management objectives, they should be maintained and managed (or protected), as the case may be. This action keeps open the greatest number of future options for sustainable management at a local level. In other cases, when severe disturbance requires seeding to protect the soil resource and restore productivity or management objectives for that land are for different uses, plant materials that can enhance productivity, heal or protect the land, or restore it to a former or future desired condition, there is a wide array of plant materials available (Carlson and McArthur, 1985; McArthur, 1988; Johnson *et al.*, 1990; Ray and Harms, 1994; Meyer and Kitchen, 1995).

Weed-infested rangelands are a particular concern. Controlling the spread of weeds and restoring weed-infested lands to health and productivity is a difficult challenge that may require step-wise action including protecting native rangelands with buffers and rehabilitating disturbed lands in stages until later seral communities can be established (Pellant, 1990; Monsen and Kitchen, 1994; Haferkamp *et al.*, 1997).

Mined land reclamation may require intensive effort to restore suitable productivity and ground cover. The results may be quite different from the pre-mining vegetation (Lang, 1982).

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