Impacts of Livestock Grazing on Soils and Recommendations for Management

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"The most important and most basic physical resource on rangelands is the soil. If excessive soil is lost, the potential of the site is changed." "Avoidance of accelerated erosion due to land management should be the underlying goal."

Society for Range Management, 1995.

"There is very little of the western range where, because of depletion of the plant cover by overgrazing, accelerated erosion has not destroyed a portion of the soil mantle and thus reduced the productivity of the site."

Ellison, 1960

Impacts of Grazing on Soils

Livestock grazing profoundly affects soils, as it affects other components of ecosystems. The impacts of livestock on soils have been studied throughout the West since the turn of the century. Livestock have been found to significantly alter almost every aspect of soil structure and function, including soil porosity, chemistry, microbiology, nutrient cycles, productivity, and erosion rates. Most studies have shown that livestock grazing increases soil compaction, erosion, and short-term nutrient availability, while it tends to reduce long-term soil nutrient and organic matter levels.

Soil structure: infiltration and compaction

Soil structure is basic to soil health and productivity. Soil structure is the arrangement of particles within the soil (Brady, 1984). Soil structure is an integrated description which includes soil porosity and the size and strength of soil aggregates. Soil structure controls the movement of air, water, roots and soil organisms into and through the soil. Structure is also the soil attribute most immediately affected by grazing. Grazing changes soil structure primarily by compaction. Compaction reduces water and air infiltration into the soil and restricts plant root growth both physically, by reducing the space available for root exploration (Tisdale et al., 1985), and biologically. In California, for example, studies have shown that low oxygen availability associated with soil compaction inhibits oak root growth, makes oaks more vulnerable to attack by fungal and other pathogens, and reduces oak survival (Costello et al., 1991). Similar responses have been observed for many other plant species in compacted soils (Tisdale et al., 1985). Finally, more water runs off and less water is absorbed when soils are compacted. This reduces water availability to plants throughout the growing season.

Cattle weigh 500 kg (1,100 lb) or more (Holecheck et al., 1995). The pressure on soil from moving cattle has been estimated by various researchers at between 1.7 and 4.2 kg/cm^2 (23.9 - 59.05 lb/in^2). By contrast a 68 kg (159lb) human exerts a static pressure of 0.4 kg/cm^2 (6.2 lb/in^2) (Abdel-Magid et al., 1987b; Ratliff, 1985). This intense and continual pressure from moving livestock easily compacts soil, particularly when the soil is wet and most vulnerable to compaction (Brady, 1984; Warren, 1987).

Studies on grazing and soil compaction generally find that exposure to livestock grazing compacts soil and that soil compaction increases with grazing intensity. This pattern is reflected in reviews of the scientific literature on the subject (Fleischner, 1994; Lauenroth et al., 1994; Kauffman and Krueger, 1984; Warren, 1987; Belsky and Blumenthal, 1995). Compaction is directly related to soil productivity because it reduces water and air movement into and through the soil and therefore reduces water and air availability to plant roots. Soil compaction also directly restricts root growth because compacted soils have fewer large pores and so there is little space for roots to enter (Tisdale et al., 1985).

Infiltration rate is the rate at which water enters the soil, as opposed to puddling or running off. Infiltration rate is often used as a measure of soil compaction. Heavy grazing has been shown to decrease infiltration in a number of soil types and geographic areas throughout the west, including northeastern Colorado (Rauzi and Smith, 1973), Wyoming (Abdel-Magid et al., 1987a,b), Texas (Pluhar et al., 1987; Warren et al., 1986), and Oregon (Bohn and Buckhouse, 1985). Abdel-Magid and coworkers (1987b), for example, found a 40% decrease in infiltration rate under simulated trampling, and this effect was observed whether the trampling was of short or long duration.

Soil bulk density is a more direct measure of soil compaction. A few studies have found no relationship between livestock stocking rate and bulk density (Warren et al., 1986c). Most studies, however, have found significant increases in bulk density (soil compaction) with grazing, particularly in finer textured soils and in surface soil layers (Abdel-Magid et al., 1987b; Orr, 1960; Bauer et al., 1987; Warren et al., 1986b; Firestone, 1995; Ellison, 1960). This effect is most pronounced with heavy grazing, but is also observed with moderate intensity grazing. Firestone (1995) observed a 13% increase in bulk density of grazed soils under oaks in California. Orr (1960) measured up to a 20% increase in bulk density in the top 4 inches of grazed South Dakota streambottom soils when compared with exclosures.

Erosion

Surface soil erosion has profound effects on soil productivity and ecosystem function. Nutrients, organic matter, microorganisms, soil fauna, and roots are all concentrated in the surface soil or soil A horizon (Brady, 1984). With mismanagement, an A horizon that took thousands of years to develop can be lost in a few years or decades. With loss of the A horizon, soils loose most of their productivity because they loose the nutrients, organic matter and associated water holding capacity that were concentrated there (Batie, 1984).

In addition, accelerated surface soil erosion causes serious water quality problems by adding sediment to creeks, lakes, and reservoirs (Batie, 1984). Researchers have frequently observed that all or part of public investment in reservoirs may be wasted due to accelerated sedimentation associated with grazing and other land management practices (Bari et al., 1995; Ellison, 1960).

Reviews of the scientific literature consistently report that soil erosion increases with livestock grazing (Belsky and Blumenthal, 1995; Fleischner, 1995; Bari et al., 1995). Studies that have compared grazed areas with ungrazed exclosures have found greater sediment production rates under grazing in many plant community types (Bohn and Buckhouse, 1985; Ellison, 1960; Pluhar et al., 1987; Wood and Blackburn, 1981). As with compaction, sediment production is generally found to increase with grazing intensity (Beeskow et al., 1995; Bari et al., 1995; Thurow et al, 1986; Warren et al., 1986a; Warren et al., 1986b); although some researchers been unable to detect a relationship between grazing intensity and erosion (e.g. Warren et al., 1986c).

One study (Thurow et al., 1986) compared sediment production under heavy continuous grazing, moderate grazing, and livestock exclosures in several plant communities in Texas. They found increases in sediment production of between 460% to 2400%, depending on plant community type, under heavy grazing compared to exclosures. However, moderate grazing did not increase sediment production in this study.

Some researchers have correlated increased surface erosion with grazing-related changes in plant communities. Ellison (1960) cited a study of succession from undisturbed plant communities to weed fields following intensive grazing in the subalpine zone in Oregon's Blue Mountains. Erosion losses from undisturbed or lightly disturbed soils were estimated at about 2 ton/acre. As grazing damage increased to a transitional mixed grass and weed stage, erosion increased to between 212 and 606 ton/acre. In the most disturbed annual weed-dominated sites, sediment production rates were approximately 927 ton/acre. Beeskow and coworkers (1995) found similar trends. They described a sequence in which grasslands with very low erosion rates were transformed into shrub steppes with much higher erosion rates by intensive sheep grazing.

Bare soil

One site characteristic that is often used as an indicator of compaction and accelerated surface erosion, as well as ecological condition generally, is the amount of bare soil present on the site (Pluhar et al., 1987; Warren et al., 1986c; Menke et al., 1996; National Research Council, 1994). Vegetation protects the soil surface from the erosive forces of trampling, raindrop impact, overland flow, and wind. Vegetation and litter also buffer the soil from compaction. This is why the National Research Council (1994) has recommended that grazed areas cannot be classified as healthy if bare ground is apparent.

Several studies have found that the percentage of bare ground on a site increases with grazing, particularly at higher stocking densities (Schulz and Leninger, 1990; Warren et al., 1986c). One study (Naeth et al., 1991) found increases of between 270% and 470% bare ground in a grazed area compared to an ungrazed exclosure.

<u>Riparian and meadow soils</u>

Riparian and wetland soils are unique and very important components of California ecosystems**.** Riparian soils and vegetation provide irreplaceable habitat for aquatic plants and animals (Kauffman and Krueger, 1984; Naiman et al., 1993). Healthy riparian soils and vegetation also play important roles in maintaining water quality because both below- and aboveground vegetation act as filters for sediment and biological pollutants such as nutrients and microorganisms (Kauffman and Krueger, 1984; Kleinfelder, 1992; Clary et al., 1996). Many riparian soils are formed by the deposition of sediment during floods and the accumulation of undecomposed or partially decomposed organic materials under anaerobic conditions. Root densities of riparian species, such as *Carex nebrascensis* , in riparian soils are extremely high and these roots tend to make healthy riparian soils exceptionally resistant to compaction and other damage (Kauffman and Krueger, 1984; Kleinfelder et al., 1992; Manning et al., 1989).

Riparian areas, wetlands, and meadows are heavily impacted by livestock grazing. Livestock, like wildlife, are attracted by water and by the shade provided by riparian vegetation. Meadows, for example, make up only about 10% of the land area in the Sierra Nevada, but they make up a much larger proportion of the forage base (Ratliff, 1985). In the later parts of the grazing season, when upland vegetation is dry, livestock concentration in riparian zones increases because riparian zones contain forage that is still lush and palatable.

Because of the intense livestock use of riparian areas, meadows and wetlands, these areas absorb a disproportionate share of grazing damage, often with undesirable consequences. Both woody and herbaceous riparian vegetation is often overused by livestock. Trampling of streambanks damages root systems, weakens plant communities, and adds sediment to streams and other waters. As riparian soils and vegetation are damaged by grazing, their ability to trap sediment and build riparian soils is decreased. In addition, as the plant communities' ability to absorb the erosive force of water is reduced, channel downcutting or gullying may occur, lowering the water table, changing soil chemistry, and eroding soils (Hagberg, 1995; Chaney et al., 1993). Throughout California and the west, grazing-related gullying and stream downcutting have cut through organic soils that were laid down over centuries in meadows and wetlands. The process creates less productive dry meadows and weed fields (Hagberg, 1995; Cottam and Stewart, 1940; Odion et al., 1988). All of these impacts compromise the ability of the riparian areas to produce clean water and to provide habitat for native vegetation, fish and wildlife. (Kauffman and Krueger, 1984; Fleishner, 1995; Ohmart, 1996; General Accounting Office, 1988)

The connection between trampling and grazing livestock, stream channel erosion, and water table lowering has been known for decades (Sumner and Leonard, 1947; Cottam and Stewart, 1940). By the 1920s, domestic livestock numbers in Sierran meadows were already being reduced and efforts commenced to try to repair grazing-damaged meadow systems (Allen-Diaz, 1991). Discussion of this problem has intensified in recent years (National Research Council, 1994; Menke et al., 1996; Kattelmann and Embury, 1996; General Accounting Office, 1988). Overgrazing has been cited as second only to damming and other direct stream channel manipulation in causing degradation of riparian areas in the Sierra Nevada foothills (Kattelmann and Embury, 1996).

Streambank erosion rates and resistance to compaction is often directly related to the species composition of the streambank vegetation. Carex *nebrascensis* dominated riparian communities, for example, have been found to be unusually resistant to compaction (Kleinfelder et al., 1992). In another study, streambanks dominated by Carex *nebrascensis* and *Juncus balticus* were resistant to erosion while streambanks dominated by non-native and upland species such as *Poa pratensis* showed higher erosion rates (Dunaway et al., 1994). Other researchers identify woody species, such as willows, as critical components of bank armoring vegetation (Chaney et al., 1993).

The structure of streambank vegetation is also important in capturing sediment and reducing streambank erosion. A recent study (Clary et al.,1996) examined the relationship between the height of vegetation with the ability of the vegetation to capture and retain sediment in simulated streams. They found that, in a single sediment deposition and flushing event, shorter vegetation heights (0.5 inches) were most effective in capturing sediment. However, longer vegetation stubble heights were more effective in retaining sediment. In addition, with repeated cycles of sediment deposition and flushing, the amount of sediment retained in the longest stubble heights (8 to 12 inches) continued to increase while the amount retained in shorter vegetation (up to 3 inches) leveled off.

<u>Soil fertility and nutrient cycles</u>

Grazing profoundly affects both soil fertility and soil chemistry. Grazing animals, through herbivory, digestion, and excretion, dramatically increase the decomposition rate and directly alter the amounts of nutrients stored in the soil, the spatial distribution of those nutrients, and the availability of those nutrients to plants. Grazing indirectly affects soil nutrients through its effects of plant species composition and soil structure. Grazing also appears to affect soil pH. Generally pH is significantly lower in grazed areas than in ungrazed areas (Ratliff, 1985; Firestone, 1995).

Total soil nutrients

There is some variability in the scientific literature regarding the nature of the impact of livestock grazing on the total amounts of various nutrients in the soil. The effects of grazing vary depending on the nutrient studied, the location of the study and the grazing management system. However, there is little disagreement among researchers that grazing significantly changes soil nutrient status (Pieper, 1994; Laurenroth et al., 1994).

There is no disagreement, however, that livestock remove many nutrients from the soil and ecosystem. Nutrients are removed as livestock consume plants and convert them into livestock biomass which is transported off site. Nutrients are also lost through increased erosion of nutrient rich surface soil, through accelerated decomposition of litter and organic matter, and through leaching. Some nutrients are returned to the ecosystem in feces and urine. Tiedemann and coworkers (1986) calculated a net loss of 3.2 kg nitrogen per acre under moderate grazing compared to ungrazed areas in the Pacific Northwest (cited in Pieper, 1994). Of that total, 0.9 kg nitrogen was estimated to be lost to volatilization during digestion, 2 kg was lost in livestock biomass, 0.7 kg was lost through increased erosion, and 0.22 kg was lost through increased leaching. Another study (Pieper, 1977; cited in Ratliff, 1985) estimated that 10.3% of nitrogen and 38.1% of phosphorous may be lost to the ecosystem under moderate grazing. Bauer and coworkers (1987) calculated that each 500 kg cow removes about 25 kg carbon and 4 kg nitrogen per hectare from grazed ecosystems.

Direct comparisons of grazed and ungrazed soils generally find that grazing reduces total soil nutrient levels. Comparing an 80 year old exclosure with heavily and moderately grazed pastures, researchers found significantly more total soil nitrogen in the exclosure than in the grazed areas. This nitrogen loss was observed down to 106.7 cm depth (Frank et al., 1995). Another study reports that soil nitrogen was reduced from 0.20% to 0.14% and soil carbon was reduced from 2.1% to 1.5% in heavily grazed soils compared with a 47 year old exclosure (Laurenroth et al., 1994). On the other hand, a study that examined 12 grazed and 12 ungrazed grasslands in North Dakota, found about 17% more total soil nitrogen in grazed areas. Carbon, however, showed the opposite trend, with grazed grasslands consistently showing lower total carbon levels than grazed areas (Bauer et al., 1987).

Spatial distribution

The spatial distribution of nutrients, particularly their horizontal distribution, is substantially altered by livestock use. Grazing animals remove plant biomass from certain areas, digest it, and redeposit it in other areas as concentrated urine and feces. Studies have found that grazing facilitates the development of a patchy distribution of soil nutrients in what would normally be more homogeneous surface soils and plant communities (Pieper, 1994). Mathews and coworkers (1994) found that because grazing cows spend much of the hot daylight hours under shade and around waterers, these areas experience substantially increased deposition of feces and urine. In that study, nitrogen, phosphorous, and potassium all accumulated in the third of the pastures closest to shade, water and supplemental feed. Each deposit of cow dung or urine acts as an island of nutrients in a pasture or meadow. Cow dung has been estimated to affect an area of about 2.6 sq. ft. (Ratliff, 1985).

Nutrient availability

Although nutrient availability is more important to ecosystem function than total soil nutrient levels, the effects of livestock grazing on soil nutrient availability has not been as well studied. The presence of a nutrient in the soil does not mean that that nutrient is available to plants. Vast amounts of nitrogen, phosphorous, carbon and other nutrients are stored in relatively unavailable forms in vegetation, litter, and in soil organic matter. It is only when vegetation and soil organic matter are broken down by decomposition that nutrients are released into the soil solution and become available to plants and other organisms.

Livestock grazing increases the rate at which vegetation and soil organic matter are decomposed (Laurenroth et al., 1994). Vegetation grazed by livestock is rapidly decomposed during digestion and many nutrients are returned to the soil in readily available forms in feces and urine. Therefore grazing generally increases short-term soil nutrient availability. Afzal and Adams (1992) treated soil with cattle dung and simulated cattle urine. They found that the available nitrogen concentration in the soil was increased significantly under both types of excreta. A study in a California oak woodland (Dahlgren and Singer, 1991) found higher levels of highly available and mobile nitrate-nitrogen under oak canopies in grazed areas as opposed to ungrazed areas, although the difference was not statistically significant. Another California oak woodland study similarly found significantly greater levels of available nitrogen (ammonium and nitrate) under oaks in a grazed grassland than in an adjacent exclosure (Firestone, 1995).

Ecosystem impacts

Nutrient availability controls many aspects of ecosystem function. Obviously it affects the productivity of the system, at least in the short term, since soil nutrients, particularly nitrogen are often limiting for plant growth. However, increased nutrient availability also often facilitates invasion of plant communities by weedy plant species (Burke and Grime, 1996; Vinton and Burke, 1995). In addition, when nutrients, particularly nitrate, are free in the soil solution they are much more liable to be leached from the system in ground or surface water or lost through volatilization or other processes (Barber, 1984). The combination of nutrient losses through increased nutrient availability, and nutrient losses in livestock biomass described above, is likely to compromise the long term productivity of grazed ecosystems.

<u>Soil organic matter and litter</u>

Organic matter improves soil structure, water holding capacity, and infiltration (Brady, 1984). Soil organic matter and litter are important repositories for soil nutrients (Tisdale et al., 1985). They also help the soil to absorb and water and so increase water availability to plants throughout the growing season (Roberson, et al., 1991). Litter and soil organic matter also help soils to resist erosion, compaction and deformation (Batie, 1984; Ellison, 1960; Ratliff, 1985;

Tisdale et al., 1985). Thus for hydrologic benefit managing for litter (and soil organic matter) accumulation may be as important as management for increasing live plant cover (Naeth et al., 1991).

Grazed areas tend to have lower litter levels, and consequently lower soil organic matter levels, than ungrazed exclosures. This is to be expected because of the large amount of plant biomass removed by livestock. In Colorado, 30 year old exclosures had two times the litter of grazed areas, as well as more bare soil (Schulz and Lenninger, 1990). Another study compared an 80 year old exclosure with heavily and moderately grazed treatments. They found half the mass of decomposed litter in the grazed treatments (Frank et al., 1995). Other studies have found similar effects of grazing on both litter and soil organic matter (Naeth et al., 1991; Williams and Quinton, 1995).

<u>Conclusion</u>

This brief review shows that grazing significantly affects the structure, composition, fertility, chemistry, and functioning of soils, frequently in ways that compromise both short- and long-term productivity. It is essential that any management plan or environmental analysis for livestock grazing management include strong, effective, and specific measures to prevent these effects or to repair soil damage where it has already occurred. Soils must be thoroughly evaluated for grazing damage in order to develop grazing management programs that will promote soil health. The next section will discuss some approaches to making this evaluation.

Analysis of Impacts

In NEPA documents or any other land management analyses, measurable ecological indicators must be used to analyze the impacts of past and proposed management to soils and other resources. These indicators should also be used to disclose the results of environmental analyses to the public and to decisionmakers. Quantitative or objective ecological indicators provide an objective, consistent measure of resource condition. They also provide a clear and understandable basis for evaluation of the impacts of alternative management proposals by the public and decisionmakers. Ecological indicators should be used to report the current state of the resources in the planning area, and to quantitatively summarize predictions for the future state of resources under various planning alternatives.

Many indicators are available to evaluate soil quality and the impacts of management (Doran et al., 1994). Discussions of the most sensitive and useful indicators are available in the scientific literature and in land management agency guidebooks (e.g. National Research Council, 1994; USDA Forest Service 1994; Doran et al., 1994). There is remarkable consistency among these sources regarding the most useful potential indicators of soil health.

The review of scientific literature above suggests several specific soil quality indicators which are both relatively easy to measure and have been demonstrated to be sensitive to grazing impacts. These include indicators of soil structure changes such as bare soil, bulk density and infiltration rate which appear to be sensitive to grazing intensity across a wide range of soil and plant community types (Rauzi and Smith, 1973; Abdel-Magid et al., 1987; Warren et al., 1986; Warren, 1987, Kauffman and Krueger, 1984); indicators of changes in soil nutrient status such as changes in pH, or available nitrogen or oxygen (Ratliff, 1985; Tiedemann et al., 1986; Costello et al., 1991); indicators of accelerated erosion such as pedestals, rills, and bare soil (Batie, 1984; National Research Council, 1994), and indicators of damage to riparian soils such as bare or damaged streambanks (Sumner and Leonard, 1947; Cottam and Stewart, 1940; Kauffman and Krueger, 1984) and replacement of bank armoring hydrophilic vegetation with upland species (Kleinfelder et al, 1992; Chaney et al., 1993).

A recent Soil Science Society of America publication brought together the observations of many experts regarding soil quality indicators (Doran et al., 1994). Their recommendations for useful indicators included many of these same parameters: soil bulk density, infiltration rate, organic matter, as well as soil aggregate stability, rooting depth, and the type and number of soil microorganisms and soil fauna.

The land management agencies also have produced proposals for soil quality indicators. The 1993 Region 5 Forest Service Draft Soil Quality Standards (USDA Forest Service, 1993), for example, recommend bulk density, surface organic matter, soil organic matter, and soil hydrologic function (infiltration and surface runoff) as suitable measures of soil health. The Rocky Mountain Region Forest Service Rangeland Analysis Guidebook (USDA Forest Service, 1994) presents a "Rangeland Health Evaluation Matrix" (Table 1, below) based on the recommendations of the National Research Council (1994). This matrix suggests that indicators of healthy rangeland soils should include the presence or absence of an unfragmented A horizon, pedestals, rills or gullies, scouring or sheet erosion, litter distribution, and rooting depth.

A consensus also exists regarding soil health indicators for riparian soils. Most researchers concur that soil cover, particularly cover by plant species producing large amounts of strong roots, such as some *Carex* and *Salix* species, is essential for protecting soil health (Kauffman and Krueger, 1984; Kleinfelder, 1992; Chaney et al., 1993). The height of riparian vegetation also plays a role in controlling streambank erosion and sediment capture (Clary et al., 1996). The land and resource management agencies provide ample direction for measuring riparian impacts (Bureau of Land Management, 1995; California Rivers Assessment, 1996; USDA Forest Service, 1992; Environmental Protection Agency, 1993). These sources all recommend percent vegetation cover, bank stability, and overhanging streambanks be among the indicators of streambank soil health and stability.

To summarize the recommendations of the sources reviewed above, the indicators of soil health that appear to be most useful include:

Erosion indicators:
- Presence of rills, gullies and pedestals
- % cover by bare soil

Compaction indicators
- Comparisons of infiltration rates between grazed and ungrazed reference areas.
- Comparisons of soil bulk density between grazed and ungrazed areas.
- Comparisons of rooting depth between grazed and ungrazed areas.

Indicators of soil nutrient status
- Comparisons of *available* nutrients, particularly nitrogen, between grazed and ungrazed areas.
- Comparisons of soil litter and organic matter levels between grazed and ungrazed areas.

Indicators of riparian soil integrity
- % cover by native, hydrophilic, bank armoring vegetation, such as *Carex nebracensis* and *Salix* sp., on streambanks. This should be compared with ungrazed reference sites.
- % cover by bare soil on streambanks
- % cover by livestock trampling damage on streambanks
- the height of vegetation on streambanks at the end of the grazing season
- presence of gullies
- comparisons of the percentage of overhanging banks between grazed and ungrazed areas.
- Depth to water table

Methods for measuring these parameters are available in the literature cited above, from the Natural Resources Conservation Service, in land management agency guidebooks (Bureau of Land Management, 1995; California Rivers

Assessment, 1996; USDA Forest Service, 1992; Environmental Protection Agency, 1993), and in Soil Science Society of America publications (e.g. Klute, 1986).

These indicators are well established in the scientific literature and in land management agency procedures. Any NEPA analysis or other resource analysis should include, at minimum, the current measurements of these indicators and should forecast the probable range of values for these indicators under each alternative.

Reference areas

In order to provide a frame of reference for measurements of soil health in grazed areas, it is critical that NEPA documents provide information on the undisturbed or pre-European values of ecological indicators. This information can be obtained from undisturbed reference sites in similar ecosystems, such as National Parks, or from records of historical conditions. This comparison provides the public and decisionmakers with a basis for assess the current and future health of the ecosystem under different management schemes.

Representative ungrazed vegetation reference plots should be placed or identified in appropriate areas to demonstrate the potential ungrazed soil and vegetation conditions such as soil bulk density, cover by litter and vegetation, and rooting depth. The use of reference areas has been recommended by a variety of land management agency directives, including Region 5 range specialists (Stokke et al., 1994), the Region 5 Ecosystem Management Handbook (Manley et al., 1994), as well as ecologists (Bock et al., 1993).

These indicators should also be used to develop quantitative Desired Condition definitions and ecological standards that will trigger changes in management (see below) (Manley et al., 1995; Doran and Parkin, 1994; Interagency Ecosystem Management Task Force, 1995; Ecological Society of America, 1995; Williams and Marcot, 1991).

Evaluation of Significance

After soil health indicators have been identified and measured, and after a forecast is made regarding the probable future values of these indicators under each alternative, NEPA documents must also evaluate which of the probable future environmental impacts of each alternative are environmentally significant. In order to do this, standards, or acceptable values, must be defined for each envionrmental indicator.

Standards are "acceptable" values of ecological indicators. Failure to meet standards should induce immediate changes in management to improve ecosystem health. The selection of standards should be based on ecological thresholds and on desired condition. Ecological thresholds represent risk to the integrity of the ecosystem. Once a threshold, such as loss of the A horizon, has been crossed, it becomes much more difficult, even impossible, to restore the ecosystem to its previous state. Standards should be values of ecological indicators which suggest that a threshold is being approached. Failure to meet a standard, therefore, should trigger immediate management changes. It is not adequate to wait until a threshold is reached or crossed to alter management, because by then restoration may be impossible (Milton et al., 1994; Friedel, 1991). Standards must trigger management changes while the ecosystem is still well on the healthy side of the threshold. For example, in the Rangeland Health Evaluation Matrix (Table 1), appropriate standards would be the "healthy" levels of each indicator.

An interdisciplinary team including a soil scientist, botanist, plant ecologist, range management specialist, hydrologist, and other specialists should determine the standards for each indicator based on the best scientific information for each ecosystem. One criterion for identifying a suitable standard should be that if that standard is met or exceeded, ecosystem health should improve over time. For example, a standard of 10% maximum streambank trampling would not be suitable if other indicators of ecosystem health, such as water quality, native vegetation cover, or water table depth, remain at unacceptable levels while that standard is being maintained. If maintenance of a standard for a particular ecological indicator fails to improve ecosystem health, as measured by improvement in other indicators, the standard for the indicator should be adjusted. Standards should be based on the best available information in the scientific literature. The Rangeland Health Evaluation Matrix (Table 1) (National Research Council, 1994) provides one example that has already been adopted by at least one region of the Forest Service (USDA Forest Service, 1994).

Table 1. Rangeland Health Evaluation Matrix (National Research Council, 1994)

Indicator	Healthy	At Risk	Unhealthy
	Soil stability and watershed function		
Soil A horizon	Present and unfragmented	Present but fragmented distribution developing	Absent or present only in association with prominent plants or other obstructions.

Pedestalling	None	Pedestals present, but on mature plants only; no roots exposed	Most plants and rocks pedestaled, roots exposed
Rills and gullies	Absent or with blunted or muted features	Small, embryonic, and not connected into a dendritic pattern	Well defined, actively expanding, dendritic pattern established
Scouring or sheet erosion	None visible	Patches of bare soil or scours developing	Bare areas and scours well developed and contiguous
Sedimentation or dunes	No visible soil deposition	Soil accumulating around plants or small obstructions	Soil accumulating in large barren deposits or dunes or behind large obstructions
	<u>Distribution of nutrient cycling and energy flow</u>		
Distribution of plants	Plants well distributed across site	Plant distribution becoming fragmented	Plants clumped, often in association with prominent individuals, large bare areas between clumps.
Litter distribution and incorporation	Uniform across site	Becoming associated with prominent plants or other obstructions	Litter largely absent
Root distribution	Community structure results in rooting throughout available soil profile	Roots are absent from portions of the available soil profile	Roots only present in one portion of the available soil profile
Distribution of photosynthesis	Photosynthetic activity occurs throughout the period suitable for plant growth	Most photosynthetic activity occurs during one portion of plant growth period	Little or no photosynthetic activity on location during most of the period suitable for plant growth
	<u>Recovery mechanisms</u>		
Age-class distribution	Distribution reflects all species and age classes	Seedlings and young plants missing	Primarily old or deteriorating plants present

Plant Vigor	Plants display normal growth form	Plants developing abnormal growth form	Most plants in abnormal growth form
Germination	Microsites suitable for germination present and well distributed	Developing crusts, soil movement, or other factors degrading microsites; crusts are fragile	Soil movement of crusting sufficient to inhibit most germination and seedling establishment

The Region 5 Forest Service Draft Soil Management Handbook (USDA Forest Service, 1993) also presents some critical values or standards for soil health evaluation. However, these may not be appropriate for grazing. The Draft Soil Quality Standards appear to have been developed primarily to protect soils from logging damage. Logging is a one-time activity and soils have an opportunity to recover for years or sometimes decades afterward. This is very different from a continuous activity such as grazing where there is little or no opportunity for soil recovery. This is no doubt one reason that the Draft Standards are still be reviewed for adequacy by the Forest Service (Rob Griffiths, Soil Scientist, USFS Pacific Southwest Region, San Francisco, pers. commun.). Moreover, the Soil Management Handbook, does not provide the scientific basis for the values chosen as Soil Quality Standards. The Rangeland Health Evaluation Matrix, on the other hand, was designed by a National Research Council panel of range scientists and references the scientific studies upon which it is based.

Most of the standards and indicators in Table 1 are subjective or qualitative measures. The literature review above suggests several quantitative indicators that should also be considered for NEPA analyses. We suggest some appropriate standards for these indicators.

- Indicator: stability of soil surface aggregates compared to ungrazed or undisturbed site (Warren, 1987; Warren et al., 1986b; Wood and Blackburn, 1981).
  Standard: Soil surface aggregate stability should be equivalent to similar ungrazed sites.

- Indicator: % cover by bare soil
  Standard: bare soil should cover less than 5% of soil surface

- Indicator: Rooting Depth
  Standard: Depth to common roots. This is the depth at which root density is found to meet the definition of "common" of the Natural Resources Conservation Service (personal commun., Desi Zamudio, Soil Scientist, Toiyabe NF Sparks, NV). This depth should be equivalent for grazed sites

and reference areas.

- Indicator: Soil compaction
  Standard: Soil bulk density should be equivalent (+/- 5%) between grazed sites and reference areas.

- Indicator: Streambank trampling
  Standard: Streambank trampling should be equivalent between grazed sites and reference areas.

Desired Condition

It is also important to identify values for the Desired Condition (DC) for each indicator so that progress towards complete ecological recovery and sustainability can be measured. Values of indicators at DC should be defined as the values those indicators would have on a similar ungrazed or undisturbed site. Information on potential ungrazed ecological structure and species composition may be gathered by investigating reference areas, similar plant communities in National Parks or other protected areas, by consulting key researchers, and by inspecting historical records from the last century (Ellison, 1960; Manley, 1995; Stokke et al., 1994, Bock et al., 1993). Information on the qualitative and quantitative values of ecological indicators at DC should be included in the NEPA document.

Subjective vs. objective rangeland health evaluations

Wherever subjective or qualitative measures are used to evaluate soil health, they should be verified and calibrated with objective, quantitative measures of some subsample of the planning area. The Rangeland Health Evaluation Matrix (Table 1) contains many examples of subjective or qualitative indicators of soil health. For example if a subjective evaluation is made that only patches of bare soil are present, then some transects or other quantitative methods should be used to demonstrate objectively what percentage of the planning area is in fact bare soil. The quantitative measures do not have be performed throughout the planning area. Rather they should be performed on a random subsample of the planning area in order to ensure that the subjective evaluations are being done correctly. The information from both quantitative and qualitative measures should be included in NEPA documents so that public and decisionmakers will be able to properly evaluate the quality of the data used in NEPA documents.

In addition, annual photographs should be used to document progress towards standards. Photographs should be carefully standardized following an accepted published protocol (e.g. EPA, 1993; Kinney and Clary, 1994). Where available, photographs showing current range condition should be included in NEPA documents.

Mitigation

The NEPA document should clearly identify what actions will be taken to improve ecosystem health when standards are not being met. The preceding literature review suggests several appropriate measures to take when soil and ecosystem health is compromised by poor livestock grazing management. The major point that emerges is that increasing grazing intensity leads to increasing soil compaction and erosion, and reduced infiltration rates. It therefore follows that decreases in grazing intensity will reduce or reverse these impacts when they occur.

Many researchers suggest complete rest as the most effective and rapid method to repair grazing damage to soils and other resources, particularly in damaged riparian areas (Fleishner, 1995; Ratliff,1985; Clary and Webster, 1989; Sumner and Leonard, 1947; Chaney et al., 1993). Elmore and Kauffman (1994) note that livestock exclusion has consistently resulted in the most dramatic and rapid rates of riparian ecosystem recovery. These authors also note, however, that "simply excluding the riparian area into a riparian exclosure does not address the needs of the upland vegetation or the overall condition of the watershed". Unless a landscape-level approach is taken, important ecological linkages between uplands and riparian cannot be restored and riparian recovery will likely be limited. This may mean that livestock exclusion must be extended to an entire watershed.

For soils outside of riparian areas, the literature review above shows that ungrazed exclosures consistently display improved soil health over grazed areas. However, it also shows that in some cases, light or moderate intensity grazing may also be able successfully maintain soil health.

Some authors suggest that conservative grazing use and other changes in management can be very effective in protecting soil health, while maintaining some grazing use. Kauffman and Krueger (1984) have stated that herding on a somewhat daily basis has been successful in limiting number of livestock in riparian areas and improving upland utilization. Ratliff (1985) suggests several measures to deal with trampling damage in Sierran meadows including: (1) adjusting use, particularly of high elevation meadows and soft meadow edges, to periods when the soil is firm. (2) locating salt grounds well away from meadows (3) routing trails to keep livestock and people off meadows, and (4) closing (fencing) sensitive sites to livestock grazing and people. For riparian systems, Clary and Webster (1989) recommend 4-6 inch minimum stubble height remain at the end of the grazing season to protect soils in most healthy riparian areas. They also recommended setting minimum stubble heights greater than six inches for critical fisheries, easily eroded streambanks, or unhealthy riparian areas (such as those not meeting standards, or those "functioning at risk"). However, Elmore and Kauffman (1994) report 3 major short-comings of grazing strategies that fail: (1) cookbook approach, no recognition of complexities or heterogeneity of riparian zones (2) many strategies do not consider woody vegetation, streambank integrity, or riparian function (3) many strategies were developed to maximize livestock production rather than to protect other resources. Any mitigation or recovery plan must have resource condition as its first priority, rather than the continuation of historical grazing use.

Adaptive management

Certainly, changes in grazing management, particularly rest, have been shown to be very effective in controlling soil compaction, infiltration and erosion. The most effective approach to soil or ecosystem protection, therefore, is to link grazing management directly to soil or ecosystem health through monitoring and adaptive management. Below, we propose an adaptive management system for adjusting grazing management to protect soil and ecosystem health when soil health standards are not being met. Adaptive management has been repeatedly recommended as a cornerstone of ecosystem management and of sustainable management generally by the agencies (e.g. Interagency Ecosystem Management Task Force, 1995; Manley et al., 1995; Chaney et al., 1993) and by the scientific community (Ecological Society of America, 1995; Grumbine, 1994).

A framework for adaptive management is displayed below. This framework presents

1. a clear description of specific actions that will be taken,
2. a specific timeline showing when such actions will be taken, and
3. specific criteria for taking action (i.e. failure to meet standards).

This type of clear statement of what, when, and why management actions will be taken has been long requested both by the ranching community as well as by the environmental community (Dan Macon, California Cattlemen, pers. commun.; also see letter to Bertha Gillam from California Cattlemen, CNPS, California Mule Deer, and California Trout, enclosed).

Adaptive management model

1. Monitoring for ecosystem health indicators should be performed and reported every one to three years.

2. If monitoring data indicates that the condition any resource is not meeting ecological standards (see examples in section on Evaluation of Significance above), and if there is evidence that grazing impacts are causing or contributing to this unsatisfactory condition, then grazing management will be adjusted before the following grazing season.

   Adjustments shall be determined by an interdisciplinary team including specialists with relevant expertise. Adjustments shall be designed to show rapid, substantive and measurable progress towards desired conditions.

Adjustments should include but need not be limited to:

a. reductions in season of use by a minimum of 20% in the affected area. Season of use changes should be designed to improve ecological condition, OR
b. reductions in allowed utilization by a minimum of 20% in the affected area, OR
c. a combination of changes in season of use and utilization.

3. If after two years of altered management, resource condition still does not meet standards, and if there is evidence that the problem continues to be related to grazing impacts, then management will be further adjusted as in (2) above.

We recommend that the Forest Service include in NEPA documents, at both the Forest and the Allotment level, this kind of clear and specific statement of how management will be adjusted to ensure the health of the resource.

Monitoring and Enforcement

The NEPA document should also describe comprehensive monitoring plans, with timelines, to ensure that ecological indicators, compliance with standards, and progress towards DC, are measured and reported regularly and accurately.

<u>Monitoring schedules</u>

Monitoring for compliance with utilization, trampling and season of use standards should be performed and reported annually. Monitoring for ecological condition should be performed and reported at a minimum of every three years because this is the interval over which many management-induced changes in range condition can be detected (Sierra Nevada Ecosystem Project, 1996). Some ecological indicators, such as streambank vegetation cover and cover by bare soil, should be monitored annually because they are easy to monitor and respond rapidly to management.

Quantitative vs. qualitative monitoring

As discussed above, NEPA documents should disclose specific indicators of soil health and general ecological condition and should discuss the scientific literature that forms the basis for the selection of each indicator. Ecological indicators may be qualitative (subjective), but should be quantitative (objective) where possible. Qualitative measures should be used for routine annual monitoring of ecological condition only if quantitative measure is demonstrated to be unfeasible. If a qualitative ecological indicator, such as the BLM's definition of "proper functioning condition" for riparian areas (California Rivers Assessment, 1996; BLM, 1995), is used for routine annual monitoring, it should be supplemented by quantitative standards which will be measured periodically (minimum of every three years) to verify and calibrate monitoring results for the indicator. Quantitative indicators should also be used for more detailed site examinations if qualitative measurements show that a problem exists. We present some suitable examples of both qualitative and quantitative indicators, above.

Annual photographs should also be used to document progress towards standards. Photographs should be carefully standardized following an accepted published protocol (e.g. EPA, 1993; Kinney and Clary, 1994).

As noted above, information from both quantitative and qualitative monitoring and analyses should be included in NEPA documents so that public and decisionmakers will be able to properly evaluate the quality of the data in NEPA documents.

Conclusions and Recommendations

Several points emerge from the scientific literature on grazing and soil health and from the literature on ecosystem management.

1. All NEPA analyses should include objective or quantitative information on current soil health and its relationship to past livestock grazing management. Aspects of soil health that must be addressed include soil compaction, erosion, nutrient levels, organic matter, and litter.

2. Information in NEPA documents should include quantitative and qualitative measurements of the soil health indicators, such as those presented in the Analysis of Impacts sections above.

3. All NEPA analyses should also include predictions of future soil health under each management alternative. These predictions should also include an estimate of the probable range of values for soil health indicators under each alternative.

4. Ungrazed reference areas should be identified in National Parks or other suitable areas or established within allotments. These should be used for comparisons of ecosystem health indicators between grazed and ungrazed areas.

5. NEPA analyses should disclose information on the values of soil health indicators in ungrazed ecosystems. This information may come from reference areas or from the scientific literature.

6. Standards or acceptable values should be set by an interdisciplinary team for each soil health indicator. The NEPA document should include information from the scientific literature showing the basis for each standard and soil health indicator.

7. Desired condition should also be set for each soil health indicator based on the value of that indicator in an ungrazed or undisturbed system.

8. Monitoring protocols, with monitoring schedules, for each indicator should be included in the NEPA document. Monitoring for ecological condition should be performed and reported at minimum every three years.

9. Monitoring methods should include both qualitative techniques, for some routine monitoring, and quantitative techniques, for periodic verification and calibration of qualitative results and for in depth investigations of site conditions.

10. Whenever monitoring shows that standards are not being met, management should be adjusted immediately, before the following grazing season, to protect resources.

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