Stewarding Soil: promoting soil quality to meet management objectives on California rangelands

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Abstract

Rangeland soils provide provisioning and regulating ecosystem services. All other rangeland sub-systems rely on soils. Net primary productivity, wildlife habitat, livestock forage, nutrient cycling, and available water are dependent upon the characteristics and condition of the soil. In this literature review, we articulate the relationships between soil attributes (expressed in terms of health and quality) and rangeland management objectives. Common objectives for California land managers include increased or improved; plant-available water, forage production, carbon sequestration, native species diversity, and vernal pool habitat conservation. We discuss the soil attributes that affect soil quality (function) for each objective. Management techniques to improve soil attributes, as well as management implications, are also discussed.

Contents

Introduction

California rangelands provide goods and services ranging from wildlife habitat and carbon sequestration to the production of food and fiber. In this literature review, we focus on the relationships between soil attributes and specific rangeland management objectives; plant-available water, forage production, carbon sequestration, native species diversity, and vernal pool habitat conservation.

Soil Health Versus Soil Quality

Provision of goods and services rests upon a variety of factors, one of the most critical of which is the condition of rangeland soil. The condition of the soil can be described using two terms: *soil health* and *soil quality*. Although often used interchangeably, soil health and soil quality actually refer to distinct aspects of a soil. *Soil health* refers to self-regulation, stability, resilience, and lack of stress symptoms in a soil considered as an ecosystem**.** It describes the biological integrity of the soil community, which depends on the balance between different organisms within the soil biota and the balance between those soil organisms and physical aspects of the soil environment. To interpret rangeland health overall, an interagency effort identified seventeen indicators that could be used in comparison to a reference state (Pellant et al. 2005). The indicators cover soil/site stability, hydrologic function, and biotic integrity. The soil attributes tend to emphasize physical properties indicative of erosion and compaction.

Soil quality describes the capacity of a soil to function as a plant growth media, a source of water and nutrients, a modifier of the atmosphere, an engineering medium, and a habitat for soil organisms (Brady and Weil 2008; Doran and Zeiss 2000). The Soil Science Society of America Ad Hoc Committee on soil quality defined soil quality as “the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation” (Karlen et al. 2008).

Approaches to Assess Soil Quality

Different approaches have been developed to assess soil quality. Some of these approaches are more appropriate to croplands, which have more intensive inputs and active management, than the wildland soils typical of rangeland settings. Nonetheless, several of these approaches are described below.

- RUSLE2 (Revised Universal Soil Loss Equation 2) is an equation that calculates expected erosion on a given soil based upon the management practices and other parameters. The USDA maintains a robust database containing values from across the country, such that users of the RUSLE2 need not perform soils testing on their own. Rather, as the manager goes through the RUSLE2 computer program, the appropriate data are added to the equation based upon answers to simple questions about climate, soil, topography, and management (USDA 2011). Although useful for calculating erosion risks, the RUSLE2 does not yield actual data about soil quality indicators such as soil organic matter, water holding capacity, or fertility. Other erosion prediction programs applied in wildland settings include WEPS (Wind Erosion Prediction

System), WEPP (Water Erosion Prediction Project) and RHEM (Rangeland Hydrology and Erosion Model), all projects funded by USDA.

- The Soil Conditioning Index (SCI) was adopted by the 2002 U.S. Farm Bill as a way of determining eligibility for the USDA's Conservation Security Program (CSP) and the Environmental Quality Incentives Program (EQIP). The SCI focuses only on soil organic matter as an indicator and is primarily centered on croplands. While soil organic matter is arguably one of the most comprehensive indicators of soil health because of the plethora of soil biological, physical, and chemical properties and processes it influences, SCI does not indicate soil degradation due to factors such as salinity or compaction.
- The Soil Management Assessment Framework (SMAF) is an expert system for soil quality (SQ) evaluation developed by USDA NRCS and ARS (Andrews et al. 2002). The first step in using the SMAF is indicator selection. Indicators are recommended based upon management objectives, current practices, and location (Karlen et al. 2008). For instance, a salinity test would probably not be very worthwhile to a dairy farmer in northern California's Humboldt County. However, salinity may be an important factor to a rancher in a more arid region, or to a manager trying to reintroduce native saline adapted plants.
- Another computer program for assessing soil quality is the Agroecosystem Performance Assessment Tool (AEPAT), which was designed to focus on assessing soil performance primarily in long-term agroecosystem experiments (e.g. a minimum of 20 years of high quality data) (Liebig et al. 2004).
- In 2007, the Cornell Soil Health Test program was initiated with four main goals: to facilitate education about soil health, to guide farmers and land managers in their selection of soil management practices, to provide monitoring for the NRCS and, finally, to indirectly increase land values by providing information regarding the soil's overall condition (Gugino et al. 2009).
- The NRCS provides information on testing kits and blank sheets for soil quality assessment (NRCS 2012; Soil Quality Institute Staff 2001).

Soil Quality Indicators

Most of the above approaches were initially oriented towards *cropland* soil quality. No matter which approach is used, soil quality is calculated through the assessment of one or more Soil Quality Indicators (SQI) and is based upon the management objective or desired use, such that the quality of a given soil may differ depending on the management goal (Brady and Weil 2008; Doran and Zeiss 2000).

SQI are specific soil properties that indicate rates of soil processes that can be interpreted with regards to implications for soil functions. In order to be useful as a SQI, a soil property must be easy and inexpensive to measure, well correlated with beneficial soil functions, and comprehensible and useful to land managers. To be fully useful for management, a SQI must elucidate ecosystem processes, meaning that it must do more than predict what will happen within a soil in response to management, it must shed light on why it happens (Brady and Weil 2008; Doran and Zeiss 2000; Karlen et al. 2008; NRCS 2009).

Herrick (2000), writing from a wildland perspective, discussed research issues to be addressed in adopting soil quality as an indicator of sustainable land management. These issues include: 1) causal relationships between soil quality and ecosystem functions, 2) predictive power of soil quality indicators to disturbance (including considerations of resilience and resistance), 3) land managers' access to monitoring systems and data, 4) integration of soil quality indicators with biophysical and socio-economic indicators, and 5) placement of soil quality indicators within a landscape context.

Finally, a SQI should be sensitive to variations in management. As such, SQI are categorized by management sensitivity. Indicators such as water or mineral nitrogen content are considered immediate SQIs as they are sensitive to management efforts that occur on a daily or routine basis. Intermediate indicators like soil organic carbon (SOC) or aggregation are sensitive to management actions that take place over several years. SQI such as slope, texture, and mineralogy are said to be permanent indicators because they are essentially insensitive to management over time. Table 1 displays commonly used SQIs organized by management sensitivity (Brady and Weil 2008; Doran and Zeiss 2000). More than 60 SQI have been suggested, although some are only useful for specific soil or management types. As of 2008, SMAF had scoring curves available for 12 SQIs (Karlen et al. 2008).

Table 1: Soil quality indicators by management sensitivity (Brady and Weil 2008).

Immediate	Intermediate	Permanent
Water content	Aggregation	Texture
pH	Soil organic carbon (SOC)	Slope
Bulk density	Soil organic matter (SOM)	Climate
Field soil respiration	Basal respiration	Restrictive layers
Mineral N	Specific respiration quotient	Mineralogy
Available K	Microbial biomass	Stoniness
Available P		Soil depth

SQI may also be categorized based upon how they affect soil function, or as biological, physical, or chemical indicators (Brady and Weil 2008; NRCS 2009). For some SQI, however, these categories are not clearly defined, as the given indicator may affect several soil functions. The Natural Resources Conservation Service (NRCS) further divides indicators into inherent and dynamic soil properties. Inherent properties are those derived from the soil forming factors such as climate, topography, parent material, biota, and time and are generally unresponsive to management, rather like the permanent indicators as described by Brady and Weil (2008). Dynamic indicators are those which reflect past or current management, like the immediate and intermediate indicators of Table 1 (Brady and Weil 2008; NRCS 2009).

Managing rangeland so as to improve or maintain soil quality is clearly a desirable goal. But different circumstances call for different measures: soil conditions deemed ideal for one particular set of management objectives may not be ideal for another set, and practices intended to affect the soil in a particular way may conflict with other management objectives. Therefore, in presenting the current state of knowledge of rangeland soil management, the remainder of this paper divides its discussions into sections focused around relatively specific soil-related management objectives and discusses how relevant practices may influence other aspects of the rangeland system. Some of these objectives may be seen as means for achieving other, more general objectives, but each represents a reasonably coherent segment of the literature. It is hoped that rangeland managers can incorporate this information into their broader management strategies as they see fit.

Managing to Increase Plant-available Water

"Slowing down the water" constitutes the main goal of one of our favorite ranchers along California's Central Coast. In our predominantly Mediterranean climate, storing soil moisture from cool winters extends the growing season into the hot dry summers. Water limits plant productivity, therefore increasing and maintaining plant available water is a common management objective. Soil water can be expressed as water content (how much is in the soil) and water potential (the energy with which soil moisture is held to soil particles or released to plant roots). On croplands, soil water content, an indicator of soil water potential, is a SQI that may be immediately sensitive to management (e.g. Brady and Weil 2008). Soil water content can be increased through irrigation or conserved through crop management techniques such as row spacing and canopy closure. However, in most rangeland settings, irrigation is unfeasible due to constraints of terrain, water availability, or finances. Whether soil moisture derives from irrigation or precipitation, plant-available water is highly dependent upon infiltration and retention.

Infiltration

Infiltration is the movement of water into the soil surface through pores. When the infiltration rate is less than the precipitation rate, surface runoff occurs, resulting water loss that might otherwise enter the soil, along with possible erosion and sedimentation. Infiltration rates are soil-limited by aggregate stability, bulk density, and pore structure and continuity to the surface. Aggregate stability of surface crumb or granular structures is important because there are water- conducting macropores between intact peds. If aggregates with low stability (e.g. due to low organic matter content) are destroyed by raindrop impact or physical disruption, the soil will lose infiltration capacity. Physical soil crusting due to raindrop impact, dust deposition and/or sodium-induced clay dispersion also decreases infiltration (and seedling emergence). High bulk densities, concomitant with decreased pore space, caused by compactive forces can decrease infiltration and percolation though the soil profile. Macropores (>0.08 mm diameter) are responsible for rapid gravitational flow of water down the soil profile, conduct a greater percentage of water than micropores, and are formed between peds (soil structural units) or by the movement of plant roots and soil organisms (Angers and Caron 1998; Azooz and Arshad 1996; Brady and Weil 2008). Macropores formed by plants or animals are referred to as biopores and may be continuous for a meter or more, providing conduits directly from the soil surface to deep within the profile. Plants with deep tap roots provide excellent infiltration routes (Angers and Caron 1998).

There are several techniques that can increase infiltration. One set of strategies involves increasing the number of macropores in the soil. Planting deep-rooting plants, particularly those with taproots, can establish biopores that may increase infiltration. Although it is not a native, alfalfa is a well-studied example of a deep-rooting plant. Angers and Caron (1998) found that alfalfa (*Medicago sativa* L.) was ideal for promoting rapid infiltration. Alfalfa has particularly straight, large-diameter, long taproots. In one study, 27% of the macropores created by alfalfa extended to 50 cm below the soil surface. Another study using a dye tracer showed that the biopores left by decaying alfalfa roots conducted the tracer deeper than 55 cm. A third alfalfa study showed increases to infiltration rates with every continuous year under alfalfa. Preferential or gravitational flow under branching-type root systems, like those seen in perennial grasses, does occur and may or may not be significant on California's rangelands. *Nasella pulchra*, a native California perennial grass, has been shown to have deeper root mass than the exotic annuals *Trifolium hirtum* and *Brachypodium distachyon* (Holmes and Rice 1996), although differences in infiltration have not been reported.

A second set of strategies for increasing infiltration relies on retaining organic matter on (or adding it to) the soil surface. In a rangeland setting, this is most easily accomplished with maintenance of residual dry matter (RDM). RDM reduces the disturbance to the soil surface from direct raindrop impacts, which can dislodge topsoil and lead to surface sealing and erosion. In overland flow conditions, the RDM helps to slow down water movement and retain sediment particles, further promoting infiltration and resisting erosion (Bartolome et al. 2006; Mapfumo et al. 2000). RDM also reduces evaporation from the soil. Light to moderate grazing has been shown to leave sufficient RDM to prevent surface sealing and soften direct raindrop impact.

If surface sealing develops enough to limit infiltration, one possible remedy is to disrupt the sealing through tillage or other disturbance. Otherwise, tillage should be minimized to maintain continuous surface-venting pores and RDM.

Retention

Once moisture has infiltrated the soil, it percolates downward through the soil profile due to gravitational and matric forces. Some of this moisture ends up in the root zone, and some may drain deeper, beyond the reach of plants. Just as macropores were most important in transmitting water into the soil, micropores (<0.08 mm) are essential in retaining moisture through matric forces. The micropores form within soil aggregates (peds) and are generally responsible for horizontal water movement through the soil via capillary action (Angers and Caron 1998; Azooz and Arshad 1996; Brady and Weil 2008). The smaller the pores and soil particles are, the greater the ability of the soil to retain water. This explains why water is most strongly retained in clay soils.

However, this is where the difference between water retention (content) and water potential should be emphasized. The amount of soil water that is available for plant use, known as available water holding capacity (AWHC) is a product of a soil's ability to retain moisture. AWHC is the difference between field capacity, which is the amount of water in a freely drained soil two days after a thorough wetting, and the amount of moisture remaining in the soil (too tightly bound to soil particles to be available) when the plants growing in it reach the permanent wilting point. Further adjustments to AWHC are made for rock fragments and salinity (Brady and Weil 2008). It turns out that silty textured soils have the highest plant available water-holding. Sandy soils, dominated by macropores and low relative surface area, have rapid downward movement of water with little horizontal spread and have the lowest capacity to retain water. In conclusion, management practices don't change soil texture (a permanent SQI), but management practices may influence bulk density (an immediate SQI) and therefore porosity and pore size distribution.

Soil organic matter (SOM) is also important in improving soil water retention, as SOM has a greater capacity to hold water molecules than the mineral fraction of soil. SOM also plays a passive yet invaluable role in both infiltration and retention in its function as an adhesive in soil aggregates (Angers and Caron 1998; Azooz and Arshad 1996; Brady and Weil 2008).

In summary, a soil's ability to retain moisture—and by association its AWHC—is less sensitive to management actions than infiltration. This is mainly because moisture retention depends largely on soil characteristics classified as permanent (inherent). The rangeland manager cannot change the texture of a sandy soil. However, as noted above, SOM content has an important impact on moisture retention, and SOM is a soil quality that can be improved through appropriate management. Practices that tend to increase SOM will increase moisture retention and AWHC.

Adjusting grazing intensity to leave moderate amounts of RDM may be an effective way of increasing SOM over time and thereby improving water retention. In one grazing study, the increase in AWHC resulting from light to moderate grazing, as opposed to heavy grazing, was ascribed to an increase in organic matter inputs and incorporation (Mapfumo et al. 2000).

Managing for Forage Production

Whether managing for wildlife or livestock, increasing forage production is a common management goal. In California, forage production is most heavily influenced by four factors: precipitation, temperature, soil characteristics, and RDM (Bartolome et al. 2006). As precipitation, temperature and inherent soil characteristics cannot be influenced by management, efforts should be focused on RDM and on those soil characteristics that can be affected by management.

Soil Fertility

Fertility varies greatly across California's annual rangeland soils. However, three macronutrients have been found to be most important in influencing productivity. Nitrogen (N) is generally the most limiting nutrient, but phosphorus (P) and sulfur (S) may become secondary limiting factors in some soils. Soil amendments such as ammonium nitrate, urea, triple super phosphate, and elemental S have been shown to increase rangeland productivity (Brady and Weil 2008). The application of such fertilizers results in a temporary flush of productivity, which may last between 1 and 3 years, with productivity decreasing annually as the nutrients are removed from the system. Reapplication is necessary to maintain elevated productivity levels in lodgepole pine ecosystems in British Columbia (Wikeem et al.

1993). In some high rainfall areas of California and Oregon, with soils of pH 6.0 or lower molybdenum may be deficient (Jones 1974), interfering with legume nitrogen-fixation processes.

George and Davy (no date) provided a concise review of the classic research performed on California annual grasslands with respect to soil fertility and rangeland fertilization trials on 28 ranches in 20 counties during the 1950s and 1960s. Over several years of productivity measurements, P and S plus N applications produced more forage than N alone, probably due to better legume survival and N-fixation. These authors point out that "one of the most important benefits of N fertilization is that it can substantially increase production during the winter and early spring. This early feed is extremely valuable because it replaces expensive hay or other energy supplements for livestock." They warn against N-fertilization where rainfall exceeds 76 cm (30 inches) per year due to leaching risks and recommend annual legume seeding in its place.

Species composition is often altered as a result of nitrogen application. Nitrogen-fixing forb species are placed at a competitive disadvantage relative to more rapidly growing grasses, which can lead to the decrease or loss of clover and other leguminous species within a few years of fertilizer use (Wikeem et al. 1993). As a result, a single, unrepeated application of N can actually result in a net loss of nitrogen availability over time.

Integrated nutrient management is a recommended alternative to fertilizer application. It considers the physical, biological, and chemical processes within a system. Grazing is an important part of the nutrient cycle on many California rangelands and an important tool in integrated nutrient management. Grazing has been shown to increase soil N content, with moderate grazing having the largest increase. Mapfumo et al. (2000) reported significant increases in soil nitrogen over the pre-grazing values in Alberta for all grazing intensities, with moderate grazing having the most significant increase. Manure provides a source of organic matter and mineral nutrients and promotes internal nutrient cycling as herbivores process and redistribute nutrients removed from the soil by growing plants. The use of manure for nutrient enrichment is often less expensive than importing fertilizers to the system, and includes micronutrients not present in commercial fertilizers. The physical act of grazing has also been shown to stimulate plant growth, which may result in further nutrient cycling (Mapfumo et al. 2000).

Species composition is also influential in nutrient cycling and, by extension, soil fertility and productivity. Plants with nitrogen-fixing associations can increase the productivity of their neighbors over time, as they release excess nitrogen during growth or decomposition. Other plants with deep rooting systems may be adept at mining minerals from deep within the soil profile. In California oak woodlands, the soil beneath the tree canopies has been found to be significantly more fertile than that of adjacent grassland. Removal of the trees results in the loss of the island of fertility within 15 years (Bartolome et al. 2006; Dahlgren et al. 1997).

Residual Dry Matter

Residual dry matter (RDM) improves forage production in several ways. RDM acts as mulch, providing a source of soil organic matter and preventing erosion (Bartolome et al. 2006; Silver et al. 2010). During overland flow events, nutrients stored in soil particles are swept away as the top soil erodes. As RDM prevents this erosion and promotes infiltration, it also encourages nutrient retention and movement deeper into the soil profile. In areas with more than 15 inches of mean annual precipitation, RDM has a significant effect on California range productivity (Bartolome et al. 2006). Especially in the cases of plants with nitrogen fixing associations, RDM can act as a natural source of nutrients which are slowly released as the RDM is broken down. The presence of moderate amounts of RDM can also improve production by creating safe seed sites. Seeds are protected from graminivores and sheltered by the RDM. As the RDM reduces evaporation, it traps moisture in the top of the soil profile, which can increase germination. Young growth may also benefit from protection from herbivory and the elements.

The amount of RDM recommended depends largely upon the site, with sites at greater risk to erosion requiring more RDM. Erosion hazard increases with increasing slope, increasing rainfall, and/or decreasing woody cover. Tables 2-4 adapted from Bartolome et al. (2006) show RDM guidelines for three common California biomes. Dry Annual Grasslands are annual plant dominated, with the lowest average annual rainfall at less than 12 inches (see Table 2). Annual grassland/hardwood ranges have an annual understory with variable oak or shrub canopy and an average annual rainfall between 12 and 40 inches (see Table 3). Coastal prairies commonly support perennial grasses beneath a variable woody overstory with variable rainfall (see Table 4) (Bartolome et al. 2006).

Table 2: Minimum residual dry matter (RDM) guidelines for **Dry Annual Grassland** (lbs./acre dry weight) (Bartolome et al. 2006).

Percent Woody Cover	Percent Slope			
	0–10%	10–20%	20–40%	>40%
0–25	300	400	500	600
25-50	300	400	500	600
50-75	NA	NA	NA	NA
75-100	NA	NA	NA	NA

Table 3: Minimum residual dry matter (RDM) guidelines for **Annual Grassland/ Hardwood Range** (lbs./acre dry weight) (Bartolome et al. 2006).

Percent Woody Cover	Percent Slope			
	0–10%	10–20%	20–40%	>40%
0–25	500	600	700	800
25-50	400	500	600	700
50-75	200	300	400	500
75-100	100	200	250	300

Table 4: Minimum residual dry matter (RDM) guidelines for **Coastal Prairie** (lbs./acre dry weight) (Bartolome et al. 2006).

Percent Woody Cover	Percent Slope			
	0–10%	10–20%	20–40%	>40%
0–25	1,200	1,500	1,800	2,100
25-50	800	1,000	1,200	1,400
50-75	400	500	600	700
75-100	200	250	300	350

Although the benefits of maintaining recommended levels of RDM are many, excessive amounts of rank foliage can actually decrease productivity by shading out the new season's growth. Forb species are especially vulnerable to shading out by taller grass species. As such, tall stands of dense RDM can actually push a plant community composition shift in favor of faster growing grass species. Moderate grazing levels retain the suggested levels of RDM, while light and heavy will leave too much or too little (Bartolome et al. 2006).

Managing for Soil Organic Matter

Although both soil organic carbon (SOC) and soil organic matter (SOM) are used as intermediate indicators of soil quality, renewed interest in the function of carbon in soil, the role of soil carbon in the global carbon cycle and its potential to affect global climate change has led many land managers to focus on increasing SOC as a primary objective. In this section, we will give a brief background on soil organic matter, describe benefits that it confers to rangeland soils, carbon cycling in rangeland soils, carbon markets that my provide incentives for carbon management, and a wrap-up on rangeland management and soil organic matter.

Soil Organic Matter (SOM)

SOM is the organic fraction of the soil which includes plant and animal residues at various stages of decomposition, cells and tissues of soil organisms, and substances synthesized by the soil bacteria population (Brady and Weil 2008). SOM is the SQI most strongly correlated with productivity and is often used to define soil fertility and stability. SOM also plays an important role in determining the pH, cation exchange capacity (CEC), buffer capacity, as well as nutrient availability and cycling, especially the cycling of nitrogen, phosphorous, and sulfur (Conant et al. 2001; Fynn et al. 2009, Karlen et al. 2008). Soil carbon, a subset of soil organic matter, can be classified as either organic carbon (i.e. derived from living tissues) or inorganic carbon (i.e. derived from calcite in limestone) (see Figure 5). Soil organic carbon (SOC) formed from carbon dioxide harvested from the atmosphere by plants during photosynthesis. This newly formed organic carbon cycles from the plant to the soil, through decomposition above- and belowground, as well as root die-off and the release of carbohydrate-rich plant root exudates. Once in the soil, it acts as an energy source for the soil ecosystem and drives other nutrient cycles. Soil inorganic carbon (SIC) forms as the result of mineral weathering and is less responsive to management. SOC comprises approximately 58% of all soil organic matter (SOM), whereas SIC forms only a small portion of most productive soils. As such, SOC is the main focus of sequestration efforts. A third source, carbon from soil microbial biomass, composes between 1-3% of total soil carbon (Fynn et al. 2009).

Table 5: Types of soil carbon.

Soil Organic Carbon (SOC):	Soil Inorganic Carbon (SIC):
Formed originally during photosynthesis	Forms through mineral weathering
Comprises approximately 48-58% of all SOM	Small portion of productive soils, but common in arid zone soils or those derived from limestone
Intermediate management sensitivity responsive to management	Less responsive to management

A critical macronutrient for ecosystem functions, there is strong correlation between the SOC pool and soil fertility and forage production. SOC is found mostly in upper profile, with 64% of soil carbon in the top 50 cm of an average soil (Fynn et al. 2009). Silver et al. (2010) found that the top 20 cm held the most SOC in California rangelands. The location of SOC is important from both a sequestration and a soil fertility standpoint. Managers concerned with fertility and forage production must endeavor to prevent erosional loss of this important resource. Those managers seeking to sequester more carbon seek to increase the depth at which soil carbon is stored. SOC has been shown to have a positive correlation with increasing precipitation, clay, and iron, and to be negatively correlated with increasing temperature and bulk density (Fynn et al. 2009). It is thought that increased precipitation will result in increased plant productivity. Higher clay contents are thought to protect SOC somewhat from microbial decomposition. However, Silver et al. (2010) only observed a positive relationship with clay in their studies of California soils. SOC tends to be decomposed (to carbon dioxide) in well-aerated soils with other conditions that favor microbial decomposition. Litter quality (susceptibility to decomposition) also plays a role in rates of decomposition, with higher woody components (eg. Lignin) slowing down the process.

Benefits of Increased SOM

SOM has many intrinsic benefits that can assist in meeting management objectives. Fynn et al. (2009) cited numerous environmental co-benefits to increasing SOM content, including improved aeration, soil tilth, soil porosity, aggregation and aggregate stability. As was previously discussed in this paper, increasing aggregation increases the infiltration rates and water holding capacity of a soil and SOM is itself directly responsible for retaining soil moisture, such that increasing SOM actually increases drought resistance, especially in arid areas (Fynn et al. 2009; Silver et al. 2010). Increasing drought resistance “is of critical importance in a changing climate” (Fynn et al. 2009) and could prove immensely beneficial in California rangelands, where productivity is heavily dependent upon precipitation. SOM also plays a role in decreasing bulk density and preventing soil erosion and sedimentation. As an energy source for soil microorganisms, increasing SOM provides the nutrients necessary to support a more biologically diverse soil biota, which can in turn lead to more rapid nutrient cycling and improved availability for plants (Fynn et al. 2009).

Many of the above benefits support other management objectives, therefore improving soil quality, and have a positive impact on overarching soil health. As such, increasing SOC may become a more desirable goal when these additional benefits are taken into account.

Carbon Cycling in Rangeland Soils

Carbon sequestration rates are determined by the net balance between inputs and outputs, which are in turn affected by management, production of organic matter by plants, and decomposition of organic matter by soil organisms. The biotic processes are strongly controlled by physical, chemical, and biological factors including biome, climate, soil moisture, nutrient availability, plant growth, and erosion. As a general rule, soils gain carbon during plant growth and lose carbon during dormancy (Fynn et al. 2009; Silver et al. 2010).

SOM and SOC can be retained within the soil through three naturally occurring soil processes. Biochemical recalcitrance occurs due to the chemical characteristics of carbon substrates and refers to the increasing resistance of soil carbon to microbial consumption over time. As substrates are consumed by microbes, remaining un-decayed compounds become progressively less decomposable (Fynn et al. 2009). These recalcitrant carbon compounds are often more prevalent in the subsurface soil horizons (Silver et al. 2010). Chemical stabilization of carbon occurs when cations associated with SOC bond to iron and clay anions. This is also generally found lower in soil profile where clay and ironmay have accumulated through soil-forming processes. Chemical stabilization may account for the high correlation between clay content and SOC (Fynn et al. 2009; Silver et al. 2010). SOC can also be physically protected within soil aggregates, which are frequently held together by ‘aggregate glues’ such as glomalin, a sticky substance produced by soil fungi that is 30-40% carbon by weight. Unlike the other two retention mechanisms, physical protection of SOC varies by depth and soil type with no trend yet established (Fynn et al. 2009; Angers and Caron 1998).

Marketing C Credits

In the US, grazing lands have the potential to remove and store an additional 198 million tons of CO_2 from the atmosphere each year for the next 30 years, enough to offset 3.3% of the US CO_2 emissions from fossil fuels. In California, the 13.6 million acres of California Annual Grassland have the potential to sequester a considerable amount of carbon (Fynn et al. 2009). This potential has led many to wonder if rangeland carbon credits could be used in carbon offset programs, similar to the way that carbon credits from timber can be sold.

Until 2010, the Chicago Climate Exchange (CCX) Rangeland Soil Carbon Offset Program looked like it might be just the market for rangeland carbon credits. Through the CCX, range managers could sell their carbon credits as part of a voluntary carbon offset program. Prices are highly variable, ranging from

$0.10 to $2.35 per credit in 2009, with an average of $1.13 per credit. The carbon credit sequestration rate for California annual rangelands was calculated at only 0.40 credits $ha^{-1}yr^{-1}$, resulting in a system that favored managers with large acreages and/or pastures that were so heavily degraded that they had a greater than average sequestration potential. Smaller operations were further hindered by a policy requiring offset projects sequestering less than 10,000 metric tons of CO_2 per year to be registered through an offset aggregator, a market intermediate which grouped together enough individuals to meet the offset minimums. Aggregators charged registration and trading fees of $0.15 and $0.05 per credit respectively, in addition to verification fees and aggregator fees, which range from 8 to 10% of the value of the credits. With so many fees, sequestering carbon for commercial reasons was not very profitable under the CCX system. In 2011, the CCX was assimilated into the Intercontinental Exchange (ICE), which does not appear to be continuing the rangeland soil carbon credit program. Although a reliable and profitable rangeland soil carbon credit program is not available for California rangeland owners and managers at this time, the growing demand for carbon credits and the increasing environmental consciousness of the public are fueling continued research into the verification systems and quantification techniques that would make such a program possible in the future.

Rangeland Management Effects on SOM

Since carbon sequestration is influenced both by management and by climatic factors, resource managers should select the techniques used to increase SOM and SOC scaled to their own local climate and existing management systems. Fynn et al. (2009) highlighted several potential actions to increase carbon sequestration in soils based upon either changes in land use or management. Restoration of abandoned or degraded cropland to grassland was shown to increase SOC content by over 3% annually (Conant et al. 2001).

Within an extensive management system, adjusting stocking rates such that grazing intensity is light to moderate and maintaining RDM within suggested minimums (Tables 3-5) may increase SOM and SOC over time (Conant et al. 2001; Fynn et al. 2009; Silver et al. 2010). Conant et al. (2001) reviewed carbon sequestration studies from 17 different countries and identified management efforts that influenced carbon sequestration. Moderate grazing was shown to increase carbon sequestration by an average of 2.9%, resulting in an average of 0.35 Mg C-ha-'-yr-' (Conant et al. 2001; Silver et al. 2010). In areas with a long history of grazing, adoption of moderate stocking rates lead to an average annual rate of soil carbon content increase of 7.7%. Moderate grazing was especially successful for increasing SOC in warm dry climates with high potential evapotranspiration (Conant et al. 2001). However, in a literature review of conservation effects Briske et al. (2011) report that five out of eight studies examined showed no response of SOM to stocking rates, one showed a decrease, and two showed an increase in SOC to increased stocking rate (relatively wet year in the Upper Great Plains). Milchunas and Lauenroth (1993) found that in 19 out of 34 comparisons, SOC decreased under grazed versus ungrazed settings (global comparisons). As Briske et al. (2011) state “the response of SOC to stocking rate is equivocal.”

Grazing has been shown to increase annual net primary productivity and root mass in some studies. Even in cases where grazing has reduced aboveground net primary productivity, SOC may still increase if the grazing pressure has increased root: shoot ratios or if the decrease in inputs in aboveground plant productivity are offset by manure inputs (Conant et al. 2001). On California rangelands, grazed sites had slightly more soil carbon at all modeled depths than ungrazed sites, but the differences were statistically insignificant at the level of resolution available for the Silver et al. (2010) study. It should also be noted that Silver et al. (2010) did not take management systems or grazing intensity into account, and that their grazed sites included different grazing pressures.

Intensive management techniques which may require inputs or changes to infrastructure include: integrated nutrient management, invasive species removal or control, introduction or reintroduction of desirable grass, forb, or shrub species, restoring riparian zones and introducing black carbon (biochar) or manure into soils as a carbon source (Fynn et al. 2009; Conant et al. 2001).

Adding trees or shrubs for silvopastoralism may also be an effective way to increase SOC. Deep rooting and woody plant species can increase carbon sequestration at a greater depth within the soil profile (Conant et al. 2001; Silver et al. 2010. In their study of California rangelands, Silver et al. (2010) found that the presence of woody plants increased soil carbon in the top meter of soil by 40 Mg ha^{-1} on average. Although woody species roots extended to a deeper depth, the SOC increases were seen throughout the profile, at sampling depths of 0–25 cm, 0–50 cm, and 0–100 cm.

On California oak woodlands, oak trees are an important source of SOC and an important factor in overall soil health. Oak understories have been found to have higher soil carbon and nutrient pools and lower bulk densities than the surrounding grasslands (Silver et al. 2010). Oak woodland ecosystems may also be better than open grasslands at retaining SOC over time due to more complete use of seasonally available water (Silver et al. 2010). The practice of removing oaks in favor of the short term increase in grass productivity has potentially decreased the carbon sequestration capacity of many California rangelands (Dahlgren et al. 1997). However, residual root matter from previously removed trees was found during soil sampling at some of the sites, and may have been contributing to SOC. Woody shrubs like the invasive native *Baccharis pilularis* also contribute to SOC. Over a 25 year study, *B. pilularis* invasion increased SOC, but that increase was paired with a 67% decrease in total aboveground net primary productivity (Silver et al. 2010). Given the reduction in productivity observed in shrub dominated systems, trees that allow for an herbaceous understory may be a more advantageous mechanism for incorporating SOC into the subhorizons.

Seeding perennials, legumes, and other deep-rooting herbaceous plants may also prove beneficial in increasing SOC. Conant et al. (2001) reported a 2.0% mean annual increase from sowing legume seeds, and a 1-2.3% mean annual increase from seeding deep-rooting grass species. However, it should be noted that some of the greatest increases in SOC have resulted from seeding deep-rooting African grasses (Conant et al 2001). Non-native perennials can provide the functional carbon sequestration benefits and may be more competitive under grazing and against annual invasives. Native perennials are often out-competed, especially when water is a limiting factor. However, non-native perennials may conflict with other management goals if a return to a native population is desired.

In a three year study, SOC increased slightly under perennials but was reduced under annuals (Mapfumo et al. 2000), although the brevity of the study and the cultivation of the annuals may have affected the results. In rangeland conditions without cultivation, there would be no mechanical breakdown of the RDM from annual plants, which could lead to slower decomposition of organic carbon from plant matter by microbes.

In certain situations, annual grasslands have been shown to sequester a great deal of carbon. SOC pools in the top 100 cm of California rangelands were slightly larger than the average SOC pools for rangelands and pastures in the rest of the continental United States. When rangelands with similar mean annual temperature and precipitation were compared, California annual rangelands had more SOC than samples from the same depth in the Great Plains under perennial grasses. This could possibly be explained by the timing of California precipitation. Precipitation events in the late winter and spring are utilized by rapidly growing annuals. Warm, dry summers inhibit decomposition of the accumulated biomass. Some annual plants have as much as 30% of their root system at depths 30 cm beneath the soil surface (Silver et al. 2010).

Additional Soil-related Management Issues

Native Species Diversity

Although native species diversity is a common management goal, and research has been conducted on managing conditions to promote specific native species (Bartolome et al. 2004; Jackson and Bartolome 2002; Kimball and Schiffman 2003), relatively little published work is available on stewarding soil quality for native plants in general. To some degree, the soil attributes desired vary with the needs of individual species. As a general rule, native plants are adapted to the attributes of their local soils, and the disappearance of these natives is generally more a factor of being out-competed by faster growing or more prolific exotic plants than of inadequate soil quality.

In certain situations, plants that are dependent upon specific soil attributes may be positive or negatively impacted by management of soil. In vernal pool systems, for instance, vegetation is completely dependent upon the existence and maintenance of a impermeable subsurface feature within the soil (Barry 1998; Ferren et al. 1998; Marty 2005). While breaking up that impermeable layer to improve percolation would help other plants grow, it would destroy endemic vernal habitat.

Many natives are adapted to low soil fertility and can grow in soils that non-adapted, non-natives cannot tolerate. In these situations, the application of fertilizers, which is usually thought to improve soil quality, can actually open the system up for invasion and ultimately result in the loss of those native species (Pickart 2009; Weiss 1999). Research in Alameda and Contra Costa Counties demonstrated that phosphorus (P) was the most important element in determining the presence of the native bunchgrass *Nasella pulchra* (purple needlegrass). *N. pulchra* was most abundant on sites with less than 6 ppm P (Rao 2011). In the case of native dune plants, instability and low fertility in their growth substrate is exactly the quality required. In Humboldt County, yellow bush lupine (*Lupinus arboreus*) was introduced to stabilize dunes. The addition of nitrogen from this leguminous shrub has catalyzed the invasion of other non-natives. As the amount of biomass and SOM increases with each new invader, the sandy substrate becomes more and more stable and organic rich, which makes it increasingly susceptible to invasion, and uninhabitable to native dune species (Pickart 2009).

Similarly, serpentine-adapted native plants are able to thrive in soils that are generally considered to be of low quality for other management objectives like forage production. Serpentine soils are characterized by high magnesium and low calcium levels, often contain pockets of heavy metals, and are generally thin, rocky soil substrates with low fertility (Brady and Weil 2008; Weiss 1999). Although these characteristics make them ill-suited to many agricultural practices, these sites are of high quality for preserving the native vegetation that is adapted to these conditions (Fenn et al. 2010; Weiss 1999). While non-native, invasive plants have come to dominate most of California's grasslands, serpentine soils are generally resistant to these invaders. However, the addition of nitrogen to these systems can open them up for invasion by making these naturally low fertility soils rich enough for invasive habitation (Fenn et al. 2003; Fenn et al. 2010; Weiss 1999).

Atmospheric Nitrogen Deposition

Nitrogen can enter a system through wet or dry deposition, in which nitrogen from nitrogen dioxide (NO_2), nitric acid vapor (HNO_3), ammonia (NH_3), particulate nitrate (pNO_3^-), and particulate ammonium (pNH_4^+) can enter a terrestrial system, either through stomatal uptake by plants or through surface deposition (Weiss 1999). This phenomenon is most prevalent in areas that are downwind of large urban pollution sources (Fenn et al. 2003; Fenn et al. 2010; Weiss 1999), in sites near large point sources of nitrogen (e.g. a coal-burning power plant or an industrial complex), or in regions with a mixture of urban, mobile, and agricultural sources (Fenn et al. 2003). Vehicles are an important source of NH_3 and nitrogen oxides (Fenn et al. 2003; Fenn et al. 2010).

This deposition results in chronic, low-level fertilization that can ultimately increase nitrogen levels and plant growth over time (Fenn et al. 2003; Fenn et al. 2010; Weiss 1999). In native soils, this increase in fertility changes the soil chemistry and increases the nitrogen available for plants. When nitrogen availability is no longer limiting, plant growth increases (Weiss 1999). Although increasing plant productivity is a management goal for many land stewards, increased productivity has been shown to be negatively correlated to biodiversity in delicate, nitrogen limiting systems (Fenn et al. 2003; Fenn et al. 2010; Weiss 1999).

In a study in the San Francisco Bay Area, Weiss (1999) concluded that nitrogen deposition from air pollution was actually increasing the soil nitrogen enough to open serpentine soils to invasion by non-native grasses, particularly by *Lolium multiflorum* (Fenn et al. 2010). Weiss (1999) measured the nitrogen deposition in different serpentine soil communities around the southern San Francisco Bay Area using ion exchange resin bags. Weiss estimated that 10–15 kg N ha^{-1} yr^{-1} is deposited on inland sites such as Silver Creek and Kirby Canyon, which are south of San Jose and downwind of major pollution sources. These sites had significantly higher nitrogen deposition than the Jasper Ridge Biological Preserve site, which is located on the San Francisco Peninsula. As most of the winds reaching the Jasper Ridge Biological Preserve site come from the Pacific Ocean and pass over the relatively undeveloped Santa Cruz Mountains, it receives less nitrogen deposition from pollution, approximately 4–6 kg N ha^{-1} yr^{-1} (Weiss 1999). *Lolium* and other invasive grasses such as *Avena* and *Bromus* species out compete the endemic serpentine forbs. These grasses grow taller than native forbs, restricting their access to sunlight. The thick thatch of RDM left by undisturbed annual grasses can prevent forb emergence in subsequent growing seasons (Weiss 1999). In the 1999 study by Weiss, the invasion of *Lolium* resulted in greatly reduced populations of the key native forb *Plantago erecta*. As grasses invaded, the native forbs became restricted to shallow soils directly adjacent to serpentine rock outcrops (Weiss 1999). Grass invasion was more pronounced on sites without any cattle grazing.

Nitrogen deposition is becoming increasingly influential in California rangelands. An estimated 13% of California receives nitrogen deposition in excess of 10 kg N ha^{-1} yr^{-1} (Fenn et al. 2010). Forests in the more exposed regions of southern California experience the highest N deposition in North America (30 to over 70 kg N ha^{-1} yr^{-1}) (Fenn et al. 2003; Fenn et al. 2010; Weiss 1999). In regions of high nitrogen deposition in the Los Angeles Basin, soil base saturation and pH have decreased significantly in forests and chaparral sites as a result of the additional nitrogen (Fenn et al. 2010). In the Los Angeles air basin, deposition rates may vary from 25 to 45 kg ha^{-1} yr^{-1} in a normal year, but may actually double in foggy years, when wet deposition becomes a major contributor (Fenn et al. 2003). Nitrate leaching into streamwater may also occur as a result of nitrogen deposition (Fenn et al. 2003; Fenn et al. 2010). Affected vegetation communities include coastal sage scrub, mixed conifer forests, chaparral, oak woodlands, grasslands, pinyon-juniper, and desert ecosystems (Fenn et al. 2010).

Coastal sage scrub is subject to nitrogen deposition levels of approximately 20 kg ha^{-1} yr^{-1} (Fenn et al. 2010). This deposition increases exotic invasion, which simultaneously reduces biodiversity and increases the fuel load for fires. This ultimately creates a self-perpetuating cycle, in which nitrogen deposition allows invasion of non-native annual grasses, which increase the fire frequency, which in turn prevents the reestablishment or regrowth of native shrubs and forbs, allowing for further invasion and loss of native biodiversity (Fenn et al 2010). Elevated nitrogen levels also have been shown to result in reduced colonization and diversity of arbuscular mycorrhizal fungi, which are important symbionts in coastal sage scrub communities. The critical load (CL) for native species cover and forb richness was determined to be around 10 kg N ha^{-1} yr^{-1} in coastal sage scrub communities.

In mixed conifer forests, oak woodlands, and chaparral communities, increased nitrogen has been shown to result in changes in ectomycorrhizal fungal and epiphytic lichen communities, understory biodiversity, soil acidification, and increases in bark beetle activity, which in turn increases associated tree mortality. Nitrogen levels exceeding the CL of 17 kg N ha^{-1} yr^{-1} can also decrease fine root biomass in Ponderosa pine by 26% and cause nitrate leaching into streamwater to exceed critical threshold levels (0.2 mg NO_3-N L^{-1}) (Fenn et al. 2010), impacting water quality (Fenn et al. 2003). The CL for oak woodland water quality is even lower than that of mixed conifers, with 14 kg ha^{-1} yr^{-1} for catchments 30 ha and greater, and deposition rates of 10 kg N ha^{-1} yr^{-1} causing elevated nitrate leaching in catchments of 4 to 10 ha (Fenn et al. 2010).

Most California grasslands have already been heavily invaded by exotic annual grasses. Nitrogen deposition threatens the remaining native-dominated grasslands through opening them up to nitrogen-dependent invasives (Fenn et al. 2010; Weiss 1999). The CL for grassland invasion by exotic annual grasses is 5 kg N ha^{-1} yr^{-1} as dry deposition or 6 kg N ha^{-1} yr^{-1} including 1 kg N ha^{-1} yr^{-1} from wet and particulate deposition. It should be noted that this CL was developed based upon the serpentine grasslands in which Weiss (1999) conducted his research (Fenn et al. 2010).

Annual grass invasion in desert ecosystems increases the fuel load, dramatically increasing the frequency and intensity of fires. As native species are generally not fire-adapted, this creates a cycle that perpetuates invasion. This unfortunate phenomenon has been observed in the California Mojave and

Sonoran deserts (Fenn et al. 2010). Desert critical loads were established from fertilization experiments at Joshua Tree National Park, and differed by ecological site. Sites characterized by sandy soil with little rock cover were most susceptible to invasive grass colonization at lower levels of added nitrogen. Native vegetation on these sites is creosote scrub, but the addition of nitrogen can lead to colonization by annual grasses, especially in a wet year. In a year with above average precipitation, the CL for a California desert creosote scrub community is 8 kg ha^{-1} yr^{-1} (Fenn et al. 2010). Pinyon-juniper woodlands are at greater fire risk than creosote scrub systems, due to higher quantities of woody fuel and higher mean annual precipitation, which allows for more annual plant growth and higher fuel loads. As such, the CL and the fire stabilization load for pinyon-juniper were lower than in creosote scrub. The use of a modeling program, DayCent, enabled researchers to estimate a critical range for these two desert ecosystems, based upon variation in soil type and precipitation. For creosote scrub, the CL ranges from 3.2 to 9.3 kg N ha^{-1} yr^{-1}, while that for pinyon-juniper is 3.0 to 6.3 kg N ha^{-1} yr^{-1} (Fenn et al. 2010).

Table 6 lists the critical load (CL), or the amount of nitrogen deposition which causes a change in an attribute of interest within that community, for each of these ecosystem types (Fenn et al. 2010). Each of the seven ecosystems studied by Fenn et al. (2010) has at least some portion of their area (29 to 54%) in exceedance of the CLs (Table 6). This results in a cumulative area of 35% of the land area covered by these vegetation types, or 99,639 km^2 throughout the state of California (Fenn et al. 2010).

Table 6: Critical loads [of N} for seven California rangeland ecosystems (adapted from Fenn et al. 2010).

Ecosystem Type	Critical Load(s)	% Exceeding CL
Coastal sage scrub	10 kg N ha^{-1} yr^{-1}	53.7
Mixed conifer forests	17 kg N ha^{-1} yr^{-1}	28.7
Chaparral	10 to 14 kg N ha^{-1} yr^{-1} (water quality), 5.5 kg N ha^{-1} yr^{-1} (lichen)	52.9 (lichen)
Oak woodlands	10 to 14 kg N ha^{-1} yr^{-1} (water quality), 5.5 kg N ha^{-1} yr^{-1} (lichen)	41.2 (lichen)
Grasslands	5 kg N ha^{-1} yr^{-1}	43.6
Pinyon-juniper	3.0 to 6.3 kg N ha^{-1} yr^{-1}	30.5
Desert scrub	3.2 to 9.3 kg N ha^{-1} yr^{-1}	31.2

Most California native plants are adapted to what the Natural Resources Conservation Service (NRCS) defines as inherent soil quality, or the sum of the properties that developed in that soil as the soil was formed over geologic time (NRCS 2009). Thus, the most effective method of maintaining soil quality for natives is essentially to preserve the native soil condition. In degraded areas, species specific restoration responses may be possible (Ferren et al. 1998; Pickart 2009; Weiss 1999). In many cases, soil quality is not the limiting factor in native species survival; rather, invasive species competition or development is the main source of habitat loss.

Atmospheric nitrogen deposition is becoming increasingly influential in loss of native species diversity through invasive competition (Fenn et al. 2010; Weiss 1999). As Table 7 shows, deposition levels vary by location (Fenn et al. 2003; Weiss 1999). As such, managers should take into consideration their proximity to deposition sources and local or regional deposition levels and the CL (Table 6) for the vegetation community of interest to determine the extent to which nitrogen deposition may be a management concern.

Table 7: Atmospheric nitrogen deposition levels by location (adapted from Fenn et al. 2003; Weiss 1999).

Location	Nitrogen deposition (kg N ha^{-1} yr^{-1})
San Bernardino Mountains	45
Los Angeles Basin	24-45
Bakersfield	10-20
Sacramento	10-20
Fremont	6
Santa Barbara	6
Gasquet	1
Sequoia National Park	1
Yosemite National Park	1

Depending on the species involved, targeted grazing may be a useful tool in managing interspecies competition (Bartolome et al. 2004; Jackson and Bartolome 2002; Kimball and Schiffman 2003; Weiss 1999). In a study of native perennials on a California coast range grassland, *Nassella pulchra* was found to increase under grazing pressure, as grazing removed competing invasive annuals, with spring restricted-season grazing providing an opportunity to allow for *N. pulchra* expansion. Conversely, *Nassella lepida* increased with grazing removal, while *Danthonia californica* showed no response to grazing (Bartolome et al. 2004). In the serpentine plant communities of the southern San Francisco Bay Area, dry nitrogen deposition, largely from air pollution, has increased the soil fertility such that non-native plants are able to invade (Weiss 1999). The removal of cattle grazing resulted in an increase of non-native invasive grasses, which greatly reduced the native forb populations. Cattle selectively graze invasive grasses over native forbs. As such, moderate cattle grazing can be utilized to control invasive grasses in serpentine soil communities. Cattle can be used to maintain low sward heights, allowing forbs to be more competitive for sunlight. They also trample or eat grass thatch, breaking up the physical barrier it can create. Although nitrogen deposition from pollution is beyond the control of an individual land manager, managers can manage the invasive grasses so that native grasses can remain competitive despite the additional nitrogen. Weiss saw success with both summer-fall and winter-spring grazing.

Carefully monitored cattle grazing at moderate stocking rates can be used to remove unwanted invasive grasses, allowing native plants to remain competitive (Weiss 1999).

However, this utilizes grazing as a method to remove competing exotics, not to cause any direct improvement to soil quality. Similarly, herbicides, prescribed burning, and reseeding desirable plants have all been utilized in attempt to combat noxious weeds, but none of those methods endeavor to do so through direct manipulation of soil quality. Bartolome et al. (2004) emphasized the variable nature of native plant responses to grazing and burning techniques, and cited climatic and local factors as those most influential in species composition, saying:

> The most important characteristics for any restoration management scheme in California grasslands are flexibility and opportunism. Flexible schemes will adapt to changing conditions and improved understanding of system response to the environment and management. Opportunism implies that researchers and managers will embrace and learn from the nonequilibrium, variable nature of this dynamic ecosystem (Bartolome et al. 2004).

Conserving Vernal Pools

Vernal pools are oases of native vegetation in modern California rangelands, which have been almost universally dominated by non-native invasive annual grasses. The unique hydrologic qualities that create vernal pools maintain a wealth of native biodiversity; of the 200 identified vernal plant species, 90% were natives (Barry 1998). More than 100 vascular plant and 34 crustacean species are endemic to California vernal pools (Marty 2005). Vernal pool plants are adapted to seasonal hydrologic extremes that range from inundation in the winter and spring to complete desiccation by summer and fall (Barry 1998; Marty 2005). Invasives lack the adaptations to survive such extremes, and are unable to compete with native vernal vegetation. As such, maintaining the native biodiversity in vernal pools depends upon maintaining the hydrologic extremes, which are a direct result of certain soil conditions (Barry 1998).

Vernal pools form in depressions in the landscape that are poorly drained due to a hard subsurface layer, usually of clay, that prevents water from infiltrating very deeply into the soil (Barry 1998; Ferren et al. 1998). Water-holding capacity is crucial as well. The balance between water-holding capacity and infiltration is delicate. If infiltration improves and water moves more rapidly into the soil, grassland species can invade the vernal pool. At least two weeks of standing water are required to prevent grassland plant invasion, although most vernal pool species preferred longer inundation periods. However, if the water-holding capacity becomes too great and inundation persists long enough, the vernal pool can become susceptible to invasion by marsh species (Barry 1998).

Many factors can negatively impact the delicate hydrology of a vernal pool system, including threats like development for urban or agricultural use. In Santa Barbara California, 90% of the vernal pools have been lost to military, urban, university, and agricultural development (Ferren et al. 1998). Sacramento County has seen similar losses, with over one third of the vernal pools lost over the past 20 years (Marty 2005). However, one of the greatest threats to vernal pool ecosystems is something usually associated positively with water systems – the complete exclusion of grazing (Barry 1998; Marty 2005). California rangelands may have had a long history of grazing dating back into the Pleistocene Epoch and continuing more recently with tule elk and pronghorn (Marty 2005). Although vernal pool natives may not have evolved with grazing pressures from domesticated livestock, they have been exposed to grazing in some form for millennia. For the past 150 years alone, the Sacramento Valley has been grazed extensively by both sheep and cattle, but as management practices change, cattle are being removed from vernal pool systems, either seasonally due to seasonal grazing, or permanently due to new fencing systems (Barry 1998). This removal has had unfortunate and unforeseen consequences for vernal pool ecosystems. As Jaymee Marty (2005) of the Nature Conservancy explained, “A long history of grazing coupled with the altered plant community yields a system that is now adapted to the changes brought about by cattle and one that becomes quickly degraded when cattle are removed.”

In a study on the long-term effects of grazing removal on the Vina Plains in Tehama County, California, vegetation surveys of three sites were compared. Site A had been excluded completely from grazing pressure for the past 15 years. The vernal pool in site B was divided by a fence, such that most part of the pond had grazing access, while the ungrazed side had also been untouched by cattle for the last 15 years. Table 8 summarizes the plants found in each of the three pools. Sites A and B Ungrazed both had abrupt boundaries between native and non-native vegetation, with rank exotic species matter standing right up to the edge of the pool. Site B Grazed had an indistinct boundary, with native plants mingling amongst the exotics and clover up until the outermost band of native *Limnanthes douglasii* ssp. *rosea*. The bands of native vegetation were much wider on the grazed side of the pool at Site B, and it was estimated that the ungrazed side had lost 12 ft. of the vernal pool footprint over the 15 years of rest.

Table 8: Vernal pool vegetation from three sites in the Vina Plains, Tehama County (Adapted from Barry 1998).

Pool Location	Vegetation	Site A Ungrazed	Site B Grazed	Site B Ungrazed
Edge	*Limnanthes douglasii ssp. rosea,* Rosy Meadowfoam	<5 plants	10-15 ft. wide band	< 1 ft. wide band
	Lasthenia fremontii, Fremont's Goldfields	1-2 plants	10-20 ft. wide band	1-3 ft. wide band
	Layia fremontii, Fremont's Tidy-tips	1-2 plants	None	None
Center	*Eryngium vaseyi,* Vasey's Coyote-thistle	35%	Scattered around edges, <2%	Scattered around edges, <2%
	Plagiobothrys sp., Popcorn-flower	25%	Present in hoof prints	None
	Psilocarphus brevissimus, Wooly Marbles	Scarce	Present in hoof prints	None
	Navarretia leucocephala, Navarretia	Scarce	Present in hoof prints	None
	Downingia bicornuta, Two-horned Downingia	Scarce	30%	30%
	Eleocharis acicularis, Dwarf Spikerush	None	45%	45%
Rim	RDM	950 lb./acre Taeniatherum caput-medusae, Medusahead	500 lb./acre Native vernal pool plants, exotic annuals, clover	2*,800 lb./acre* *Lolium multiflorum*

As the results of the Vina Plains study show, complete removal of grazing from a vernal pool ecosystem gives invasive annuals a competitive advantage.

In a Nature Conservancy study of 72 vernal pools in the California Central Valley, plant species cover and diversity, soil compaction, pool inundation period (hydroperiod), and aquatic invertebrate diversity were compared across four grazing treatments: ungrazed (UG), released from grazing; dry season grazed (DG), grazed from October through November; wet season grazed (WG), from December to mid-April; continuously grazed (CG), control, grazed from October through June as had been done for the past 100 years. Over the entire three year period, CG pools had the highest relative cover of native species. By the third year of the study, native cover in CG treatments was 80% higher in the edge zone and 160% higher in the upland zone than the UG treatments. Exotic grass cover was continuously higher in the UG treatment. In 2001, 2002, and 2003 the UG treatment had 67, 130, and 88% more exotic cover in the pool zones than the CG treatments. There was also a shift in the plant community composition as grasses increasingly out-competed the forbs with removal of grazing pressure. By the final year of the study, the grass to forb ratio was four times higher in UG than CG pools and two to three times higher in UG and DG than CG in upland areas. Native species richness decreased by an average of 25% on the edge and upland zones of the UG pools, whereas native plant species richness either increased or remained constant in all of the other treatments. The inundation periods were also altered by the end of the three year grazing period. In 2003, the final year of the study, CG pools were inundated an average of 49 more days than the UG pools and 24 days later than the WG pools. The CG pools dried less than once during the variable rain season, whereas all other treatments dried down twice. This change in the hydroperiod seems to have had a direct impact on the endemic vernal pool invertebrates (Marty 2005).

Table 9 shows the average inundation period and the average number of invertebrate taxa for each grazing treatment. CG had both the longest hydroperiod and the greatest number of distinct invertebrate taxa. This suggests that continuous grazing pressure was maintained the hydrologic integrity and water holding capacity of the vernal pool system, allowing for more inundated days in which invertebrates could complete their life cycles. As the number of wet days decreases and the number of dry-downs increases in the UG treatment, some of the more long-lived invertebrates have been lost as they no longer have time to complete their life cycles. Soil compaction was significantly lower in UG than CG pools. However, Marty could not determine the effect that soil compaction had upon the water holding capacity based upon the pattern of water depths in her study. The primary cause of the decreasing hydroperiod with removal of grazing may be an increase in evapotranspiration rates resulting from the increase in non-native vegetation. Studies in the Great Plains have documented increased evapotranspiration rates in ungrazed compared to grazed grasslands, while still more studies have documented the negative impacts of overabundant vegetation on stream and wetland hydrology (Marty 2005).

Table 9: Average hydroperiod and number of invertebrate taxa by treatment for 2003 (Marty 2005).

Treatment	Inundation Period (days)	Invertebrate Taxa (number)
Continuous Grazed	115 (± 9)	14 (± 0.50)
Dry Grazed	78 (± 7)	12 (± 0.72)
Wet grazed	65 (± 8)	11 (± 0.62)
Ungrazed	65 (± 8)	10 (± 0.80)

Livestock grazing is a powerful management tool for maintaining vernal pool health and species richness (Barry 1998; Marty 2005). As awareness and protection of vernal pools increases, the threats from development decrease, but vernal pools and their endemic plant and invertebrate species are still threatened by the non-native invasive plants that comprise most of California's rangeland vegetation. As the majority of these exotics are grasses and the majority of the vernal pool native endemics are forbs, cattle can provide particularly useful at removing or maintaining the invasive matter with minimal disturbance to the native plants as cattle selectively choose grasses over forbs when grazing (Marty 2005). Although cattle may stand or step in pools, this is not necessarily a negative impact. Recall from Table 6 that two native endemics, Wooly Marbles (*Psilocarphus brevissimus*) and Navarretia (*Navarretia leucocephala*) were scarce in ungrazed Site A and completely absent from Site B ungrazed. Both of those species were present in the grazed part of Site B, but only in the hoof prints left by grazing cattle (Barry 1998). These hoof prints create microdepressions which act as miniature pools within the vernal pools, retaining deeper water longer than the surrounding pool area and providing additional habitat. It is also possible that the compaction from cattle activity within the pool may contribute to the water holding capacity and the impermeability of the subsoil hardpan (Marty 2005). If ample water is provided by the land manager, water consumption by livestock may be minimized. The positive hydrologic benefits of reduced evapotranspiration from reducing the exotic vegetation load seem to outweigh any water consumption by cattle in the studies sited (Barry 1998; Marty 2005).

Vernal pool restoration or creation is another option for areas in which pools have become too invaded or otherwise degraded for grazing alone to mitigate. In the Santa Barbara area of southern California, the California State Coastal Conservancy, the Isla Vista Recreation and Park District, the staff of the Museum of Systematics and Ecology at the University of California Santa Barbara, and the County of Santa Barbara implemented the Del Sol Vernal Pools Enhancement Plan in 1986. The Enhancement Plan focused on three phases of action involving 16 pools: first, to enhance or restore existing degraded vernal pools; second, to re-create vernal pools where they occurred historically; and third, to create new vernal pools from upland habitat at Del Sol Open Space and Vernal Pool Reserve (Ferren et al. 1998). In Phase 1 (1986), one pool was protected from vehicular access with a post barrier and given an additional soil berm to reduce drainage, two large pools were restored by removing fill and debris, and six new small ponds were created on upland sites. Of the newly created ponds, three were inoculated with material from two natural pools a mile from the restoration site and the other three were left uninoculated as a control group. In 1991 Phase 2 began with the re-creation or re-establishment of three pools that had been previously illuminated by tillage. A skip-loader and backhoe were used to prepare the sites, which were inoculated with seed bank material from one of the previously enhanced pools and with additional seeds from locally rare plants. Seeds were spread across the newly graded sites, raked into the soil, and then compacted with a roller. In Phase 3, ten years after the initiation of Phase 1, an additional large pool was re-created, the three pools re-created in Phase 2 and one restored pool from Phase 1 were enlarged.

After three years, enhanced and restored pools exhibited hydrology similar to natural vernal pools, whereas created pools failed to exhibit vernal pool hydrology. However, drought conditions could have affected Phase I results, exhibited by shorter duration and less extent of flooding than expected. Vegetation species richness and zonation of Phase 1 pools resembled those of natural pools. However, annual vernal pool plants dominated cover in restored and created pools, such that restored and created pools did not resemble natural pools in plant species cover and abundance (Ferren et al. 1998). Therefore, the expected species were present and in the same zones within the pool systems, but the ratios and percentages of species were not the same as those observed in natural pools. Using inoculum from natural vernal pools successfully produced vernal pool plants and invertebrate animals in restored and created pools. The inoculated created pools had a significantly greater ostracod population and lower turbidity than created pools without inoculum. Native avian, amphibian, and invertebrate sightings suggest that the restored and inoculated created pools fulfilled similar ecosystem functions to natural pools. Finally, enhanced, restored, and inoculated created pools were self-sustaining during the first three years (Ferren et al. 1998).

After the sixth year, annual plants were dominating the plant cover during year-one of re-created Phase 2 pools, but perennial plants were becoming increasingly dominant in five and six year restored and inoculated-created pools from Phase 1, although the distribution and abundance was still not exactly that of natural pools. Inoculated pools had higher species richness and cover than uninoculated pools (Ferren et al 1998). By the sixth year of the project, the pools were providing habitat for a variety of waterfowl, with large restored pools supporting the greatest numbers and most diverse complement of birds. Enhanced and natural pools tended to have more submergent and emergent vegetation which attracted more ducks and dowitchers. After desiccation, the pools were still utilized by avian species that expanded their use from other areas on the reserve (Ferren et al. 1998).

After ten years, analysis of plant distributions within and around the pools revealed that the vegetation of manipulated pools, with the exception of created-uninoculated pools, was within the range of variation found in natural pools in the area (Ferren et al 1998). The success of the Del Sol Vernal Pools Enhancement Plan shows that vernal pools can restored to ecosystem functionality similar to natural pools within a decade or less, depending on the level of existing degradation. Vernal pools can even be successfully created, as long as these new pools are inoculated with material from existing pools.

Discussion

Although there may be as many as 60 SQI (Karlen et al. 2008), the most critical indicators for each of the management objectives addressed in this paper are summarized in Table 10. It should be noted that many of the SQI overlap, with SOC and SOM being the most common SQI of interest. As such, these may be good indicators to focus upon in soil testing, or to direct management efforts towards.

Table 10: Soil quality indicators of greatest interest by management objective.

Water Infiltration/ Retention	Forage Production	Soil Organic Matter	Native Species	Vernal Pools
↑ SOC*	Mineral N	↑ SOC*	Mineral N	Restrictive layers
↑ SOM*	Available P	↑ SOM*	Variable	↑ Bulk Density
↑ Aggregation	Available S			Texture (Clay)
↓ Bulk density	↑ SOC*			Mineralogy
	↑ SOM*			Slope

*causes ↑ Aggregation and ↓ Bulk Density

Vernal pools and water infiltration both have bulk density as a SQI of interest, yet soil test results for this indicator would be interpreted differently based upon management goal. High bulk density at shallow depths is essential for vernal pool hydrology, but low bulk density is necessary for infiltration. This is a good example of the variable definition of soil quality dependent upon objective; a high quality soil for vernal pool habitat would be a low quality soil for infiltration. The SQI required for native species depend upon the species of management interest. However, most native species would do well in a healthy, functional soil. Unfortunately, such soils are also perfectly viable habitat for highly competitive invasives.

Table 11 summarizes the different management techniques mentioned in this paper with reference to their applicable objectives. This serves to illustrate the practices that promote multiple objectives. Carbon sequestration, forage production, and water infiltration/yield are all highly compatible, and can be improved through light to moderate stocking rates, maintaining recommended amounts of RDM, and establishment of perennials. In many cases, management techniques that are recommended for one objective have neutral effects on others. Invasive species removal, while not specifically mentioned by the sources reviewed as a method for increasing water infiltration or yield, is generally not detrimental. Removing invasives may reduce some of the competition for plant available water, similar to the way removal of invasives around vernal pools increased the length of the hydrocycle (Marty 2005). Native species have differing soil quality requirements depending upon the natives in question. Therefore, many of these management techniques have impacts that vary based upon the species.

Table 11: Management techniques by objective.

Technique	Sequester Carbon	Forage Production	Water Infiltration/Yield	Vernal Pools	Native Species
Light – moderate stocking rates	Yes?	Yes	Yes	Variable	Variable
Maintain RDM at optimal levels	Yes	Yes	Yes	No	Variable
Remove invasive spp.	Yes	Yes	Yes	Yes	Yes
Plant perennials	Yes	Yes	Yes	Variable	Variable
Integrated nutrient management	Yes	Yes			
Riparian restoration	Yes		Yes	N/A	Yes
Black carbon / compost	Yes				
Convert cropland to grassland	Yes				Yes
Silvopastoralism	Yes	Yes			Variable
Minimize tillage	Yes		Yes	Yes	Variable
Reseeding Natives		Yes	Yes	Yes	Yes
Targeted grazing				Yes	Variable
Plant legumes	Variable	Yes	Variable	No	Variable

Many of the management objectives benefit from properly managed grazing. Light to moderate stocking rates are recommended for increasing water infiltration and yield, forage production, and carbon sequestration (Conant et al. 2001; Fynn et al. 2009; Mapfumo et al. 2000; Silver et al. 2010).

However, many different studies use different rates for light and moderate stocking. As a general rule, stocking rates calculated to leave the suggested amounts of RDM for a given area are considered to be moderate (Bartolome et al. 2006). Holechek and Galt (2000) defined parameters for grazing intensity as a percentage of utilization of perennial grasses (see Table 12) with utilization measured at the end of the growing season.

Table 12: Grazing intensity by utilization (Holechek and Galt 2000).

Grazing Intensity	**Utilization (Percent)**
Light to non-use	0-30
Conservative	31-40
Moderate	41-50
Heavy	51-60
Severe	>61

Soil quality describes the capacity of a given soil to support a given management objective and is calculated based upon soil quality indicators. Comprehensive land management should take into consideration soil quality for any given management objective. Within the objectives reviewed in this paper, water infiltration and yield, forage production, soil organic matter, native species diversity, and vernal pool habitat conservation, there was a great deal of overlap in stewardship techniques. As such, it is possible to promote multiple uses and improve soil quality for multiple objectives while simultaneously improving overall soil health.

Future Questions

The elephant in the room is the need to understand soil quality indicators, what they indicate, and how rangeland management influences stewardship objectives on a landscape basis. The most prudent approach is to: 1) support rangeland research stations and reserves where long-term research can be undertaken and 2) support local land managers who accurately document local conditions, management activities and responses. Soil quality must be studied in a *local* context over *adequate time periods* that can capture favorable and unfavorable precipitation years. Dr. Phil R. Ogden, at the University of Arizona, always used to say "study one ranch at a time." Soils may take decades to adjust to either climate change or management changes. One rancher on the Central Coast aspires to "slow down water and create beauty." Another, on the North Coast, involves undergraduates in a study to determine soil organic matter in continuously grazed versus intensively managed paddocks. The generations of managers on the ground will ultimately do the most meaningful research for their own particular place on the planet.

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