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CHAPTER IV
Soil carbon sequestration in United States rangelands

Abstract
Rangelands are uncultivated lands that include grasslands, savannahs, steppes, shrub lands, deserts and tundra. The native vegetation on rangelands is predominantly grasses, forbs and shrubs (Kothmann, 1974). Rangelands cover 31 percent of the land surface area of the United States (Havstad *et al*., 2009), and up to half of the land surface area worldwide (Lund, 2007). Most land areas that are not developed, not cultivated, not forested and not solid rock or ice can be classified as rangelands. Because of their extent, a small change in soil carbon (C) stocks across rangeland ecosystems would have a large impact on greenhouse gas (GHG) accounts. There are 761 million acres of rangelands in the United States (Havstad *et al*., 2009), half of which are public lands in the West (Follett, Kimble and Lal, 2001). The primary activity focus on rangelands is grazing. Rangelands and grazing lands are two broadly overlapping categories. United States grazing lands, including managed pasturelands, have the potential to remove an additional 198 million tonnes of carbon dioxide (CO_2) from the atmosphere per year for 30 years (Follett, Kimble and Lal, 2001), when saturation is reached. This would offset 3.3 percent of United States CO_2 emissions from fossil fuels (EIA, 2009, and help protect rangeland soil quality for the future.

The past 20 years have seen a tremendous enhancement in the understanding of soil C, both its role in the global C cycle and the factors that influence its dynamics. Although soil organic carbon (SOC) has long been of interest to scientists, technical advisers and land managers as an indicator of soil health, the link between the C cycle and global climate change has provided increased impetus for quantification and, ultimately, management.

Even if atmospheric concentrations of GHGs were quickly stabilized, anthropogenic warming and sea levels would continue to rise for centuries

(IPCC, 2007a). Even the most drastic reductions in emissions of anthropogenic GHGs may not do enough, on their own, to preserve current environmental integrity for future generations. If the effects of global warming are to be kept to a minimum, C already emitted to the atmosphere as a result of human activities must be sequestered into stable forms.

Various strategies have been proposed, including the use of untested technologies requiring huge expenditures of energy and resources. For example, while geologic and deep ocean sequestration schemes have been proved to be physically possible, the economic, environmental and social costs associated with these technologies remain uncertain. For the immediate future, sequestration in terrestrial ecosystems via natural processes remains the most viable and ready to implement option, and one of the most cost-effective (DOE, 2009).

Soils hold over three times as much C as the atmosphere (Lehmann and Joseph, 2009), more than the Earth's vegetation and atmosphere combined, and have the capacity to hold much more (Lal, 2004). C stocks in terrestrial ecosystems have been greatly depleted since the beginning of the Industrial Revolution, with changes in land use and deforestation responsible for the emission of over 498 Gt of CO_2 to the atmosphere (IPCC, 2000), approximately half of which has been lost from soils (IPCC, 2000; Lal, 1999). Each tonne of C stored in soils removes or retains 3.67 tonnes of CO_2 from the atmosphere.

Soil C comprises SOC and soil inorganic carbon (SIC). SOC is a complex and dynamic group of compounds formed from C originally harvested from the atmosphere by plants. During photosynthesis, plants transform atmospheric C into the forms useful for energy and growth (Schlesinger, 1997). Organic C then cycles from the plant to the soil, where it becomes an important source of energy for the soil ecosystem, driving many other nutrient cycles. SIC is the result of mineral weathering and forms a small proportion of many productive soils. The focus of this paper is on SOC sequestration.

SOC makes up approximately 50 percent of all soil organic matter (SOM) (Wilke, 2005; Nelson and Sommers, 1982). SOM content is correlated with productivity and defines soil fertility and stability (Herrick and Wander, 1998). SOC and SOM buffer soil temperature, water quality, pH and hydrology (Pattanayak *et al.*, 2005; Evrendilek, Celik and Kilic, 2004). Increases in SOC and SOM lead to greater pore spaces and surface area within the soil, which subsequently retains more water and nutrients (Tisdall, Nelson and Beaton, 1985; Greenhalgh and Sauer, 2003). This factor is of critical importance in United States rangelands, most of which experience less than 600 mm precipitation per year. Higher soil C levels can reduce the impacts of drought and flood.

United States rangelands cover a vast area, comprise many different ecosystems and experience a wide range of environmental conditions. A protocol will reward landowners for changes in management practices or changes in C stocks. There are pros and cons associated with each approach. Where landowners and land managers have the ability to select which project actions to apply, these choices will be made with the goal of maximizing productivity and C sequestration according to local conditions. The ecological state of the landscape (Asner *et al*., 2003), its vegetation (Derner and Schuman, 2007) and land-use history all influence the effectiveness of different project actions.

Project actions for soil C sequestration, some of which require further research, include the following.

Changes in land use:

- conversion of abandoned and degraded cropland to grassland (Franzluebbers and Stuedemann, 2009)
- avoided conversion of rangeland to cropland or urban development (Causarano *et al*., 2008)

Changes in land management:

- Extensive management (i.e. does not require infrastructure development)
  - adjustments in stocking rates (Schuman *et al*., 1999; Conant and Paustian, 2002)
  - integrated nutrient management (FAO, 2008; Franzluebbers and Stuedemann, 2005, 2008)
  - introduction or reintroduction of grasses, legumes and shrubs on degraded lands (Schuman, Herrick and Janzen, 2001; Conant, Paustian and Elliott, 2001)
  - managing invasive species
- Intensive management (i.e. requires infrastructure development)
  - reseeding grassland species
  - addition of trees and shrubs for silvopastoralism (Sharrow, 1997; Nair, Kumar and Nair, 2009)
  - managing invasive shrubs and trees (Franzluebbers, Franzluebbers and Jawson, 2002)
  - riparian zone restoration

– introduction of black C (biochar) into soils (Lehmann and Joseph, 2009)

Rangeland ecosystems are complex systems involving different GHG fluxes. Changes in management that lead to increases in soil C stocks can in some cases lead to increased emissions of other GHGs, notably CH_4 and nitrous oxide. Management practices should be assessed to ensure that gains in soil C are not negated by increases in non-CO_2 GHGs.

There are two motivating factors likely to encourage landowners to adopt C sequestration practices. The first is the range of biophysical benefits – soil C is positively correlated with productivity such that as soil C increases, long-term soil productivity can be expected to increase under proper management. The second factor is increased financial benefit – landowners could benefit from revenues from the sale of emission reductions credits resulting from increased soil C sequestration. The existence of a comprehensive rangeland soil C protocol will allow increases in soil C storage to be converted to verified emissions reductions for use within an offset market, cap and trade system, or other regulatory framework or programme.

Environmental and financial benefits will result from C sequestration above that which would have occurred in the absence of the project. This additional sequestration will be achieved by the *transition* from one set of management practices to another, not by any set of management practices *per se*.

The many co-benefits associated with increasing levels of soil C suggest the prospect of win–win scenarios for landowners, climate change mitigation and ecosystem services. Optimizing uptake of sequestration activity depends on the design and implementation of the protocol, since it is here that incentives to implement changes in management practices will be generated.

When it comes to quantifying changes in soil C stocks, it is generally true that accuracy costs more, and that less expensive methods are less accurate. Extremes are not desirable: extreme data coarseness leads to low confidence in sequestration values and low market interest in credits generated; on the other hand, overly expensive quantification costs also lead to low uptake. Between these two extremes a balanced methodology will optimize adoption rates and environmental benefit.

There are many methods available for assessing changes in rangeland soil C stocks. Rather than tie a protocol to the limitations of one particular method, it is logical to combine the strengths of different methods into a single methodology, which may be updated as economics and technical advances allow. Potential elements of a final protocol include use of a performance standard, site-specific measurement, ecosystem modelling and remote sensing by satellite. It is important to achieve a balanced solution at a viable cost, and provide the economic and social incentives for adoption of enhanced management.

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CRITICAL TERMS DEFINED

For the purposes of this paper, a *methodology* is defined as an accredited means of scientifically quantifying changes in soil carbon (C) stocks within a GHG emissions reduction protocol. A *protocol* is the document that also includes all relevant rules, parameters and equations for the components of the credit accounting process – including deductions to be made from gross sequestration values. A *performance standard* is a methodology based to some degree on a number of standard assumptions, as opposed to a methodology largely reliant on site-specific measurements. By inference a performance standard is easier, faster, less expensive and less accurate than methodologies that rely on site-specific quantification.

Soils are often *C sinks*, and sometimes *C sources*. A sink absorbs more C than it emits; a source emits more C than it absorbs.

There is a difference between *soil C sequestration* and *soil C storage*. *Soil C sequestration* is the process whereby C is transferred from the atmosphere into soils. *Soil C storage* is the retention of sequestered C in the soil.

The term *soil C stocks* refers to the amount of C stored in the soil at any one time. Changes in stocks as a result of project activity are calculated as the difference between C stocks before and after that activity.

Pre-project C stocks are referred to as the *baseline*. The term baseline is also used to refer to the projected stocks or conditions *that would have been in place* in the absence of the project under a business as usual (BAU) scenario. We refer to the first definition as *pre-project baseline* and to the latter as *forward-looking baseline.*

Additionality refers to the concept that C sequestration achieved by project activity must be over and above any that would have occurred in the absence of the project, i.e. beyond BAU. Additionality must be proven for credits to be countable. For emissions reductions to be credited they must not be required as part of a regulatory framework and must not be double counted for any other reason.

Leakage is the concept that activity to reduce GHG emissions within project boundaries in some cases may force increased emissions outside project boundaries, thereby eliminating some of the achieved emissions reductions. For example, although converting usable cropland to rangeland may lead to increased C sequestration C sequestration within project boundaries, it could result in displaced crops being grown elsewhere.

Permanence refers to the idea that C sequestration achieved as a result of project activity must be secured over the lifetime of the credit. (In the first instance, permanence was an ecological concept.)

Strategies to address additionality, leakage and permanence are discussed below.

Successful project activity will lead to net reductions in GHG emissions on a project, regional and national basis. For this reason, soil C credits are henceforth referred to as *GHG emissions reductions credits*, or simply *emissions reductions credits*. In the context of this paper, all emissions reductions credits will be generated by soil C sequestration. It is also important to recognize the potential for climate change mitigation from woody biomass sequestration and reductions in non-CO_2 GHGs, within rangeland systems.

GHG emissions that may be affected by changes in management in rangeland ecosystems are CO_2, nitrous oxide (N_2O) and methane (CH_4). Over 100 years, the global warming potential (GWP) of CO_2 is 1, of CH_4 is 25 and of N_2O is 298 (IPCC, 2001a, 2007b). GWP values allow the net effect of changes in GHG budgets across different gases to be calculated, and for different scenarios to be compared. The resulting figures are given in $MTCO_2eq$, or metric tonnes of CO_2eq.

One credit represents one metric tonne of emissions reductions, in CO_2eq, achieved as a result of project activity, once verified according to the mechanisms specified in the relevant protocol, and issued by the operating registry.

DYNAMICS OF SOIL CARBON SEQUESTRATION

SOC is a dynamic group of compounds that have their origin in the photosynthetic activity of trees, grasses, shrubs, forbs and legumes. The C in these compounds cycles through solid forms back to the atmosphere at different rates, with turnover times ranging from months to hundreds of years (Davidson and Janssens, 2006; Six and Jastrow, 2002).

During photosynthesis, plants reduce C from its oxidized form into the organic forms useful for growth and energy storage (Schlesinger, 1997). Some of this C fixed from the atmosphere in time becomes soil C through the processes of above- and below-ground decomposition, root die-off, and the release of sap exudates from plant roots into the soil (exudates contain carbohydrates). Photosynthesis also provides the raw materials for indirect imports of C-rich material on to and into the soil, for example in the form of animal manure or compost.

Soil C includes SIC in the form of carbonates. SIC is the result of mineral weathering, and is less responsive to management than SOC, turning over much more slowly (Izaurralde, 2005). SIC content is low in many productive soils. Soil microbial biomass C forms 1–3 percent of total soil C.

SOC forms 48–58 percent of SOM (Wilke, 2005). SOM defines soil fertility and stability (Herrick and Wander, 1998). Most SOC is found in the top of the soil profile, as a result of the presence and influence of biotic processes there, with approximately 64 percent of soil C in the top 50 cm (Conant, Paustian and Elliott, 2001). Around 90 percent of C in rangeland systems is located in the soil (Schuman, Herrick and Janzen, 2001), as opposed to above-ground biomass.

SOC accumulation is positively correlated with precipitation and negatively correlated with temperature (Jones, 2007). The stock of soil organic C accumulation is highest in cool, wet conditions (Schlesinger, 1997) and lowest in deserts. The SOC content of rangeland soils varies from under one percent to over ten percent – even in drylands (Janzen, 2001). Soil C stocks are positively correlated with the presence of clay and iron, and negatively correlated with the bulk density of soil. (This factor also reflects the negative effect of compaction on productivity.)

The rate of C sequestration is determined by the net balance between C inputs and C outputs. C inputs and outputs are affected by management and by two biotic processes –production of organic matter in the soil 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 (Derner and Schuman, 2007; Jones, 2007; Post *et al*., 2001; Svejcar *et al*., 2008; Ingram *et al*., 2008).

Soil CO_2 is the main end product of the decay of SOC. Under aerobic conditions CO_2 is produced by respiration of bacteria and protozoa in the guts of insects, and bacteria and fungi in the soil. Soil CO_2 production accelerates with temperature and with exposure of SOM to air in pore spaces and on the surface of the soil. When decomposition and soil CO_2 production can be slowed, the net rate of soil C accumulation and storage may be increased.

Protection of soil carbon

There are three ways in which SOC and SOM can be protected from microbial metabolization or decomposition (Jastrow and Miller, 1998). Biochemical recalcitrance occurs because of the chemical characteristics of C substrates, and because substrates are consumed by microbes, remaining un-decayed compounds become progressively less decomposable. Chemical stabilization occurs with the bonding of positively charged actions associated with SOC to negatively charged iron and clay anions. Physical protection of SOM occurs within soil aggregates, held together by aggregate glues such as glomalin, a sticky substance produced by soil fungi that is 30–40 percent C by weight (Comis, 2002). SOC lower in the profile tends to be protected from microbial decomposition because of chemical stabilization. Physical protection can vary by depth and soil type (Del Galdo *et al*., 2003).

Carbon pools and carbon fractions

Researchers employ the concept of C pools to distinguish C that cycles at different rates in the ecosystem. C in each pool has a different turnover time or mean residence time (MRT). C pools are not distinct groups of C compounds, which are called *fractions*. There are two soil fractions, the light fraction and the heavy fraction, which are further classified and range from the *free light fraction* to the *heavy occluded fraction*.

Light fractions are composed of fresh plant materials that are subject to rapid decomposition, with turnover from a few months to a few years. Early changes in SOC resulting from management often occur in the small light fraction, which is known for its spatial and temporal variability. Because most of the turnover of SOM is in the light fractions, it is important to include this fraction within any chosen quantification methodology (Post *et al*., 2001). Accumulations of light fraction C can be quite large in permanently vegetated soils – i.e. forests and grasslands.

C in the heavy occluded fraction has an MRT from hundreds to over a thousand years. SOC and SOM in this fraction are less susceptible to decomposition than in the light fraction. The heavy fraction is composed of polysaccharides (sugars) and humic materials often stabilized in complexes with clay minerals and silt-sized particles (Schlesinger, 1997). One very chemically recalcitrant portion of the heavy fraction has turnover times of 1 500 to 3 500 years (Post *et al.*, 2001).

RANGELANDS IN THE WESTERN UNITED STATES[1]

Most rangelands in North America are in a region that experiences a continental climate (cold winters, warm wet spring and summer) with relatively uniform seasonal precipitation. This unique seasonal precipitation distribution governs the type and amount of plant production and C dynamics. Typically, soils gain C during periods of plant growth, while soils lose C during periods of dormancy. The length and severity (air temperatures) of dormant seasons can have an inordinate influence on C dynamics in Mediterranean systems compared with continental climates (Figure 1).

In the archetypical prairie rangelands of North America, soils are classified as mollisols (high organic matter formed from basic parent material over long periods in continental climates). These soils are relatively deep with high water-holding capacity and high levels of fertility. Mediterranean climate rangelands, on the other hand, are typically associated with more shallow and poorly developed soils (aridisols).

Major rangeland regions

The Great Plains

The physiography of the Great Plains consists of an enormous piedmont that flanks the eastern slope of the Rocky Mountains for a distance of several hundred miles. The climate is uniquely continental and is characterized by dominant north–south temperature and east–west precipitation gradients. Such climatic gradients and physiographic features define the province and ecological attributes of these ecosystems. The Northern Great Plains are vast grasslands occupying most of the states of North Dakota and South Dakota and substantial areas of Montana, northeastern Colorado and northern Nebraska. This region is generally flat to rolling, with features such as the Black Hills, badlands and rivers providing sharp breaks in the gentle topography. The influence of glaciation is very evident in the northeastern portion of the Northern Great Plains where, during the Pleistocene, continental glaciers moved south as far as the Missouri River. When they receded, the glaciers left behind millions of shallow depressions that are now wetlands, called prairie

[1] This section was extracted by Joel Brown from Havstad *et al.*, 2009

potholes. The Southern Plains are situated between the Rocky Mountains and the central lowlands and encompass portions of six states. Native vegetation is dominated by short and mid-height perennial grasses that evolved with natural disturbance regimes characterized by grazing, drought and fire.

Great Basin

The Great Basin has been defined in a variety of ways over the years. The two most common definitions are: (i) an area that is drained internally and has no outlet to the sea; or (ii) a floristically defined region that is characterized primarily by shrub-steppe (shrub/bunchgrass communities). The region designated as the Great Basin includes the area that is internally drained (hydrologic definition), but also includes additional areas of shrub-steppe to the north and east.

Much of the Great Basin is in the Basin and Range Province, with isolated mountain ranges separated by valleys. The mountain ranges are a result of fault activity (the meeting of the Pacific and North American Plates), and generally have a north–south orientation. The Basin and Range geography results in rain shadows and steep elevation gradients, which create high temporal and spatial variability in both climate and vegetation.

Desert southwest

The desert rangelands in the southwestern United States are the driest, hottest and least productive rangelands in North America. Desert rangelands consist of three hot deserts: Chihuahuan, Sonoran and Mojave.

Most of the Chihuahuan Desert – the largest desert in North America covering more than 193 000 square miles – lies in Mexico. In the United States, it extends into parts of New Mexico, Texas and sections of southeastern Arizona. The Sonoran Desert covers 120 000 square miles in southwestern Arizona and southeastern California. The Mojave Desert, the smallest of the three hot deserts, is located in southeastern California and portions of Nevada, Arizona and Utah, and occupies more than 25 000 square miles. These three desert rangelands share a number of characteristics related to climate, vegetation and land-use dynamics related to human activities, yet differ in elevation, seasonality in rainfall and plant species composition.

Woodlands and forests

This region includes both the piñon-juniper woodlands and the widely dispersed forested lands of the western United States. Woodland vegetation is widely distributed in the West and is distinguished from more classically described forested land by the reduced height of the tree layer (30–50 ft). Of the western states with piñon-juniper vegetation, New Mexico has the largest area, and Idaho the smallest.

Forested lands regarded as rangeland have often been synonymous with forestland that is grazed by livestock. These lands, at least periodically, produce sufficient understorey vegetation suitable for forage that can be grazed without significantly impairing wood production and other forest values. These lands comprise nearly 20 percent of the total area grazed in the United States. Reflecting the diversity of ecosystems and western topography, these forested rangelands are interspersed with meadows, high elevation grasslands, riparian ecosystems and, often, with piñon-juniper woodlands at their lower elevation margins.

California annual grasslands

The California annual grasslands occupy about 13.6 million acres, primarily in the foothills of the Central Valley and in coastal valleys. This region has three major subtypes: inland valley grassland, coastal prairie and the coast range grassland.

The original dominants of the California grasslands were perennial grasses interspersed with native annual grasses and perennial herbs, probably with a higher proportion of annuals in drier areas. Conversion of this grassland to an ecosystem dominated by exotic annuals began with the introduction of livestock, cultivation and seed dispersal of Mediterranean-origin annual plants in the late eighteenth century. This introduction expanded dramatically with a series of severe droughts in the late 1800s. Plants from the Mediterranean region, mainly annual grasses, now dominate the valley grassland. The coastal prairie grassland retains a greater proportion of native species but has also been invaded by both perennial and annual plants from the Old World. The coast range grassland is characterized by some native perennials mixed with native and introduced annuals. The grasslands have been valued as a source of sustenance and homesteading, for livestock forage, as real estate, and increasingly for a diverse array of tangible and intangible services.

DEVELOPING A PERFORMANCE STANDARD

Harnessing different quantification methods

Each of the measurement, monitoring and verification (MMV) methods discussed below has strengths and weaknesses, in terms of factors such as cost, ease of use and suitability for a national emissions trading platform. Different methods tend to perform better at one scale (plot, field, landscape or region) than at others. Instead of being tied to the constraints of one method, a rangeland soil C protocol methodology may harness several methods in combination, with the goals of reducing transaction costs and achieving a balance between ease of use and scientific acceptability.

All methodologies, existing or potential, may be placed along a conceptual spectrum, with extreme ease of use (and data coarseness) at one end, and higher confidence levels (and expense) at the other. A methodology that is close to either end of the spectrum will not be popular among landowners or credit purchasers. A successful methodology lies somewhere between the two poles.

Rewarding changes in management practices or changes in stocks

When designing or selecting a methodology for rangelands, there are two core approaches to choose from: compensating landowners for verified changes in management practices or for achieved changes in C stocks. There are pros and cons associated with each approach. Hybrids between these two core approaches are also possible. (See Appendix 1 – Activity based and soil C measurement hybrids.)

The first core approach (Figure 2) is rewarding landowners for changes in practices. This is the simplest solution and probably comes with the lowest transaction costs; it could be a critical factor in increasing landowners' interest in such programmes. This approach implies use of a performance standard, with average values derived from established data sets, and could allow landowners to know how much they would be compensated for specific changes in management prior to their participation in the system.

Compensating for changes in practice would make it easy to restrict the number of project actions and contain the complexity of the programme. However, credit purchasers pay for emissions reductions, not changes in management. This discrepancy could be resolved firstly through a solid scientific basis for the assumptions embedded within the protocol, and secondly through financial means including the use of brokers, risk management tools and insurance.

The drawbacks of this approach are in issues of possible error and permanence. Risk of error can be estimated using modelling/measurement techniques, and reduced by increasing the spatial extent of lands within the system. Remaining unacceptable risk of error could then be absorbed by credit discounting until an acceptable threshold is reached. Permanence is discussed below.

The second core approach is to reward landowners for changes in C stocks. Options for assessing changes in these stocks are use of site-specific measurement, or a comparatively simpler performance standard based on established data sets, or a hybrid of the two (still called a performance standard). The discussion that follows focuses on the second core approach, rewarding changes in C stocks, because of the many potential methodologies falling under this category.

Providing compensation for changes in C stocks would allow for more project-specific accounting than an activity-based protocol; yet this may not be desirable in all scenarios, where increased transaction costs could not be defrayed from any extra revenue secured. Compensating for changes in C stocks could spur innovation among landowners and project developers, if these have some freedom to innovate. However, one perceived risk with this approach is that without a pre-defined list of project actions, transaction costs could escalate, if each project action needs to be assessed, and if there are significant costs associated with that assessment.

Under the umbrella of the second core approach, it will be helpful to compare the options of site-specific measurement and use of a performance standard (Figure 3). The primary benefit of site-specific measurement is accuracy; the primary drawbacks are likely higher transaction costs and reduced ease of use. These would translate to lower scalability and non-optimized rates of adoption and sequestration.

Features of a performance standard

A performance standard is simpler than quantifying soil C in every parcel of land, and would see standard metrics replace at least some project-specific measurement. Established databases and the published literature would be accessed to determine standard values for any or all of the following: (i) pre-project baseline soil C levels; (ii) BAU scenarios; and (iii) the effect of different management project actions on soil C stocks.

Any performance standard must be based on sound scientific correlations between changes in C stocks and surrogate variables or states that are easier to measure or document. An original performance standard may be developed to meet these needs, or an existing quantification system adapted. The primary benefit of using a performance standard is ease of use and cost-effectiveness; the primary potential trade-off is a loss of accuracy.

A hybrid option is also feasible, whereby a performance standard approach is used where there is greater confidence and homogeneity in the data, and site-specific measurement used where potential opportunities or a lack of data are most egregious. (This is akin to the process of parameterization, whereby sampling is used to decrease areas of greatest uncertainty associated with predictive models.) Another kind of hybrid could see a performance standard used to establish soil C baseline and BAU practices, and site-specific measurements for project-driven changes in soil C stocks.

Whatever form the quantification methodology takes, it should include a mechanism through which published literature can be used to help generate estimates of creditable tonnes of net emissions reductions; and benefit from future improvements in the accuracy and availability of data.

For quantification formats that follow the second core approach – compensating for changes in C stocks – specific increases in quantification expenditure will lead to breakthrough increases in confidence associated with the resultant credits. Where these points are matched by predicted elevated interest by landowners begins the area of optimized climate and socio-economic benefits (Figure 4). That area is bounded at the other extreme by the point at which high data-gathering costs cause adoption rates to drop away. Preferred protocol formats will fall within this area.

Key questions

There are several key questions to be addressed during the development of a rangeland soil C protocol. These are outlined below with some preliminary responses.

What is the framework for the application of the performance standard approach to rangelands?

There are two main ways to maximize soil C sequestration in rangelands: (i) maximize uptake during normal and above normal precipitation years; and (ii) minimize losses during drought. Both of these are best achieved by controlling harvest (this does not mean no harvest) to maximize productivity for the particular situation. Rangeland managers are skilled at doing just this. Evidence-based recommendations exist for grazing practices (stocking rate, distribution, etc.) and for virtually every type of vegetation. In addition, most of this information can be gathered via remote sensing and with a reasonable degree of reliability.

What are the key challenges that need to be overcome to develop a credible rangeland C performance standard?

These challenges are: establishing a reliable link among annual precipitation, stocking rate (harvest) and C dynamics that extends beyond the prairies/ prairie mollisol soils, where confidence levels are highest.

What would the reference practices be?

The primary reference practices would be: sustainable stocking rates, cattle distribution and season of use (proper grazing attributes). There are also opportunities to include brush control practices, reseeding (pasture lands), etc.

The "long list" of project actions with the potential to increase soil C in rangelands, and at various stages of research, include: conversion of abandoned and degraded cropland to grassland; avoided conversion of rangeland to cropland or urban development; adjustments in stocking rates; integrated nutrient management; introduction or reintroduction of grasses, legumes and shrubs on degraded lands; managing invasive species; reseeding grassland species; addition of trees and shrubs for silvopastoralism; managing invasive shrubs and trees; riparian zone restoration; and the introduction of biochar into soils.

Which of the project actions from this "long list" is ready and most appropriate for the first iteration of a new United States rangeland soil C protocol depends on the analysis conducted, and the specific factors arising, during the course of protocol development. Certainly there are sufficient data to include good grazing practices in the protocol. Other project actions should be examined when all regional and national data sets can be considered simultaneously.

How good are local and regional databases for specifying baselines?

The protocol development process requires existing databases to be synthesized. These are probably well suited to the task, but this question cannot be fully answered until compilation of databases has begun, and their applicability tested. This is especially true in consideration of the fact that most of this information is available in fine-grained Ecological Site Descriptions (ESDs).

A number of national coverage maps exist, with layered information levels. The most complete soil database in the world is represented by the STATSGO and SSURGO database, which covers close to 100 percent of the country, and is constructed and maintained by the USDA Natural Resources

Conservation Service (NRCS) as part of the national soil survey programme. This is already available in online format as the Web Soil Survey (USDA NRCS, 2009).

There are a host of attributes for each polygon, some relevant to soil C, some not. Also in existence are other databases of existing vegetation type, green material, etc. Most of these based on satellite imagery are available at 30 x 30 m resolution. There are some finer scale images available, but not without a tremendous amount of processing and interpretation.

A national spatial database for management is key, and is missing. While it is possible to reasonably construct a national map of C pools, or C potential, we cannot very well construct a national map, at anything other than a regional scale, of C levels. The regional scale is accurate at a large-scale inference because it pools all of the management attributes across a large area, but serves poorly at smaller scales. Although this national map does not yet exist, it can be built; there are some official efforts under way to do just this.

How can the high levels of variation within rangeland soil carbon systems, even at very fine levels of resolution, be credibly addressed in constructing a performance standard?

In general, this variation can be addressed in the same way that it is handled in any other system: smoothing out to the variation either by lengthening the time; or increasing the spatial extent to encompass more landscapes; or discounting the commodity to cover the risk associated with the known variability.

Most studies assessing the effects of management on rangeland soil C have found large variation across experimental sites or units. To make informed judgements relative to methodology design, an understanding of the spatial distribution of the variable (C) and its value is critical. The most fine-scaled groupings of soil properties on rangelands are soil map units (SMUs), associated with local soil surveys conducted in conformance with the National Cooperative Survey standards. These surveys represent the finest scale information available for land management decisions and inventories on a national basis. On rangelands, soils are grouped into the functional edaphic units known as ESDs. ESDs are agreed upon by all the federal land management agencies as the standardized carrier for soil and vegetation attributes organized into graphic models describing management options.

Predicting soil C dynamics requires not only a general knowledge of C sequestration processes, but also a site-specific knowledge of the effects of common management practices within the range of predictable climatic variability. That site-specific knowledge would have to include an accurate assessment of the current ecological state to reasonably predict outcomes of management initiatives. Performance standards would benefit by embedding ESDs as one of the primary causal layers or parameters within the global equation for credit quantification.

Public sector programmes

This paper focuses on the design of a protocol for private sector emissions trading, because that represents the most complex transactional environment. However, programmes to compensate for soil C sequestration are by no means limited to the private sector. Public sector programmes should also be considered, and most of the issues to be addressed are discussed in this paper. The bulk purchasing and risk-carrying powers of government agencies are important features that can keep transaction costs low and drive adoption.

Funding could come through state or federal programmes, such as the Conservation Reserve Program (CRP) or NRCS Environmental Quality Incentives Program (EQIP) grants. Emissions reductions in the agriculture sector may fit better into public sector programmes. These could further reduce transaction costs and increase financial yield to the landowner, leading to higher adoption rates and thus greater climate change mitigation. Options for public-private hybrids also exist, for example using public funding to establish baselines, above which producers could generate credits for private sale. It is not within the scope of this paper to consider these interesting options in greater detail.

Variables and parameters

The development of a performance standard for rangeland C sequestration should consider all variables that are easy to measure and that can serve as predictors of changes in C stocks through changes in activities. These variables include all typical characteristics of rangeland operations such as location, land area, topography, weather and climate, digital elevation map, management history, stocking rate, grazing method, soils map and profile descriptions from the USDA database, history of rangeland improvements and weed/brush control, fencing and stock water networks, rangeland site description and condition and indicators of BAU conditions. All of these variables could be input into an online automated expert system that could give an immediate preliminary assessment to the landowner or manager. The assessment would contain a list of the potentially creditable activities, their spatial extension and the potential value of the changes, in terms of C and money at current and projected prices.

Participating rangelands may be classified according to such variables in an attempt to match project lands as closely as possible to those represented in the scientific literature. This matching will reduce the costs associated with site-specific quantification. The more closely that the set of project-specific variables can be correlated to data in the literature, the lower the transaction costs will be, for two reasons: site-specific quantification costs will be reduced, and deductions from gross sequestration values to compensate for uncertainty (*conservatism*) will also be reduced.

A rangeland protocol represents the interface between ecosystem processes and credit accounting processes. Thus, there are two sets of factors to consider in compiling the protocol equation –ecological factors and accounting factors. The two sets of factors should match as closely as possible, that is to say, ecosystem drivers affecting soil C accumulation, and net GHG balance, should be represented as the parameters within the final accounting equation.

Since the ecosystem factors are pre-established, the accounting factors should be devised and matched against the most important of these ecosystem factors. To achieve this goal, the ecological factors could be prioritized and grouped according to their relative influence on soil C accumulation; then as the protocol or performance standard is developed, this prioritized list could serve as a reference to help determine which factors can be included as parameters within the final equation.

Careful thought should be given to the number of project actions to include within the protocol. Too few would represent missed opportunities to drive mitigation and adoption rates; too many would render the protocol unwieldy. It is not possible to predict accurately the effectiveness of a performance standard prior to its implementation (although public sector programmes may come with more predictability).

Recommendations

Along a continuum of potential protocol design formats, from the simplest and least costly activity-based approach at one end, to the most expensive measurement-based methodology at the other, there are certain locations where it will make the best economic sense to develop (or adapt) a methodology or performance standard.

These opportunities (see Figure 5) will occur at points on the continuum where data-gathering costs lead to non-linear breakthroughs in the predictive ability or quantitative accuracy of methodologies situated at those locations. These breakthrough points will represent higher efficacy of quantification dollars spent, and would lead to increased market confidence in the credits. This in turn would drive up both the credit sale price to buyers and adoption rates among landowners.

These breakthroughs will increase the height (revenue) between the credit sale price and the implementation+quantification costs needed to realize these opportunities. These points represent the greatest opportunities to maximize climate change mitigation, socio-economic and ecosystem benefits.

Therefore, a key recommendation is that, in the early stages of protocol development, a graph be plotted with data for each shortlisted protocol, and with different price opportunities quantified, as well as implementation costs and quantification costs. Those protocols suggesting the greatest net revenue to landowners and project developers should be scored for other factors such as barriers to entry and positive environmental impact, and then compared. In this way the contenders may be reduced to one or two best available options.

The recommended progression is the following:

- list viable protocol designs, discarding those unlikely to lead to optimized adoption rates and form a shortlist, including existing protocols and the best potential new formats;
- survey and conduct research to obtain the data needed for each protocol design to plot a graph for each approach;
- plot graphs and compare results;
- score remaining contenders for other factors;
- select the best option and proceed with protocol development.

(It is protocols and not methodologies that must be compared, since net revenue can only be calculated after all deductions have been made; and different protocols may contain different rules for these deductions.)

MEASUREMENT, MONITORING AND VERIFICATION METHODS

This section is most relevant to a quantification methodology that would compensate landowners for changes in C stocks, although it is also relevant to a performance standard that would compensate for changes in management, because the tools described here represent the means of obtaining the data in support of both methodology types.

Changes in ecosystem C stocks can be assessed either by measuring stocks at different times, or by quantifying the net rate of C flux into the system and multiplying by time. **Direct methods** measure soil C directly from a soil sample, either onsite or in the laboratory. **Indirect methods** are based on relationships between other predictor variables and C content, and require calibrations and modelling. Most, if not all, methods rely on some form of extrapolation of information from a small set of samples to the project or regional scale. All methods are ultimately based on data from samples.

Direct methods of quantifying changes in soil carbon stocks

The most established form of direct measurement is to extract and analyse **soil core samples** (Figure 6). The sample is combusted in the laboratory and analysed for C content. This process does not differentiate between SOC and SIC. When measurements of SOC only are needed, SIC must be excluded from the sample prior to analysis, by digestion with acid. Alternatively the Sherrod *et al.* (2002) method can be used to determine SIC and total soil C; deducting SIC from total C provides a quantification of SOC. Dry combustion is a very accurate and widely used technique, whereas wet combustion is older, less reliable and now rarely used.

When considered alone, direct determination of SOC by dry combustion is generally expensive in relation to the required number of samples and expected revenues from C credits. Costs have two components: sampling and sample handling, and laboratory analysis. Costs of laboratory analysis range from USD15 to USD35 per sample. Costs of obtaining and handling the samples can vary widely, depending on site remoteness and accessibility, and on who performs the sampling.

Sampling costs can be reduced by ***stratification***. This is a means of improving the efficiency of sampling by subdividing the area to be measured into regions (strata) that are relatively homogeneous in the characteristics that are being measured, in this case, characteristics that affect stocks and fluxes of C. Stratification attempts to maximize variation among strata and minimize variation within strata, because only the variation within strata contributes to the variance for the whole population estimates (Thompson, 1992). Stratification allows optimal allocation of sampling effort to the different strata to minimize the cost for a given level of precision. In general, more samples are allocated to strata that are more variable, larger and more cheaply sampled.

While stratified sampling is appropriate for high variation systems such as rangelands, in order to minimize variation within individual strata, variation among strata remains, contributing to variation of overall estimates.

Spectral analysis technologies (***LIBS***, ***MIRS*** and ***NIRS***) are non-destructive, require no reagents, and are easily adaptable to automated and *in situ* measurements (Izaurralde, 2005). All spectral measurements require field calibrations requiring sampling and analyses using established methods, such as dry combustion (Chatterjee *et al*., 2009).

LIBS (Laser-induced Breakdown Spectroscopy) uses a high-energy laser to create a plasma of the ionized elements in the sample. The light from the plasma is resolved spectrographically and integrated to give concentrations of each element in the sample. Currently, C from SOC and SIC is not directly discernible, but methods are being developed to create this capability. LIBS allows for the simultaneous analysis of many elements, not just C. LIBS has a detection limit of 300 Mg C/kg with precision of 4–5 percent, and an accuracy ranging from 3–14 percent (Izaurralde, 2005).

MIRS (Mid-InfraRed Spectroscopy) is a stationary device that analyses core samples on-site, and was originally developed to measure protein content in forages. MIRS can differentiate between SOC and SIC, and is best applied with other methodological tools. McCarty *et al*. (2002) found that MIRS yielded better spectral information than NIRS and was a better predictor of total C and carbonate.

NIRS (Near InfraRed Spectroscopy) is a simple, rapid way to assess SOC, widely used to characterize chemical compounds. Less accurate than MIRS, it was originally found to underpredict SOM concentrations at the high end of the scale (McCarty *et al*., 2002).

The ***eddy flux*** *or* ***eddy covariance (EC) method*** performs practically continuous measurements of net CO_2 fluxes between the ecosystem and the atmosphere. Multiple micrometeorological variables are measured simultaneously. Fluxes are integrated over time to obtain yearly estimates of net change in C. The method has the advantage of providing abundant information for modelling of C fluxes on the basis of weather and vegetation measurements. An EC system, usually referred to as a "tower," measures fluxes representative of an area of approximately 1 ha.

The EC method has disadvantages. It only measures CO_2 flux and thus it would not detect other potential additions or losses of C such as erosion and exportation/importation of crops, residues and soil amendments. Moreover, the method is not stock-specific and is sensitive to changes in non-creditable stocks such as standing herbage mass. EC systems are labour intensive and tend to give poor measurements when the air is still. Data require lots of processing. This method is not applicable at a project level, but can be used as a basis for regional measurements to create and back up a performance standard.

Indirect methods of quantifying changes in soil carbon stocks

Indirect methods can be subdivided into two types. First, C stocks can be predicted by using models. These models are given the sequence of values of factors that affect C stocks, such as weather, vegetation type and grazing regime, and they provide predictions or estimations of C stocks. Second, C stocks and changes can be estimated by using statistical relationships "calibrated" with previously obtained data. These relationships or equations use values of variables that are more easily or cheaply measured than C to estimate C. Input variables can be quantitative, such as amount of radiation reflected by soils and vegetation in each of several spectral bands, or qualitative, such as soil series.

Models

Ecosystem models used to quantify soil C stocks or changes therein are known as *process-based* or *mechanistic models* (Figure 7). These use an understanding of ecological processes and the factors influencing these processes either to forecast or enhance past data sets under different management and environmental regimes. Such process-based models also have a critical role in translating data to project-scale landscapes (Post *et al.*, 2001). Such models are needed to quantify changes in rangeland C stocks because they provide estimates of changes in ecosystem C storage under varying management regimes and over different time periods.

Models include CENTURY, DNDC, COMET-VR, DAYCENT and EPIC. CENTURY appears quite popular for research purposes, and has been in use for three decades. DNDC is a well-known GHG model that also models soil C. COMET-VR and DAYCENT are variants of CENTURY. Of the three models in the CENTURY family, only COMET-VR can model GHGs other than CO_2. DNDC and COMET-VR can predict CH_4 and N_2O fluxes.

DNDC (DeNitrification-DeComposition) is a process-oriented simulation model of soil C and nitrogen (N) biogeochemistry that models GHGs. At the core of DNDC is a soil biogeochemistry model describing C and N transport and transformation as driven by a series of soil environment factors such as temperature, moisture, redox potential, pH and substrate concentration gradients (Li *et al*., 2003). The model recognizes four major SOC pools: plant residue or litter, microbial biomass, humads (active humus) and passive humus. DNDC also contains submodels for soil climate, decomposition, nitrification, denitrification and fermentation (Li *et al*., 2003).

The following three models are closely related:

CENTURY simulates dynamics of C, N and phosphorus in grassland, forest, savannah and crop systems (Metherell *et al*., 1993; Parton *et al*., 1993) CENTURY has submodels for plant production, nutrient cycling, water flow and SOM (Parton *et al*. 2005). The major input variables include soil texture, bulk density, soil hydric status, soil depth, soil field capacity, wilting point, location and climate data. CENTURY's plant production and water flow models use monthly timesteps; the nutrient cycling and SOM submodels use weekly ones (Parton *et al*., 2005).

DAYCENT is a modified version of CENTURY running on a daily timestep (Parton *et al*., 2005). DAYCENT simulates crop production, soil organic-matter changes, and C, N, nitrous oxides (NOx), and CH_4 fluxes from weather, soil-texture class, and land-use inputs (Parton *et al*., 2005).

COMET-VR runs a on a monthly timestep and has a graphical user interface. COMET-VR provides some estimation of energy use and N_2O emissions (the former from direct-measurement data and the latter from DAYCENT model output); and generates an estimate of uncertainty based on published data on the practices in question (Paustian *et al*., 2009).

In the analysis conducted to date, the CENTURY model (and its variants) has proved to be a very good method of estimating C flux on rangelands.

Models must be tested prior to implementation since they need to be calibrated to each site for which they are used. A preliminary run produces output data that are checked against data obtained from an alternate source; typically, the modelled data set is compared with an actual data set derived from field measurements. Discrepancies allow the model to be corrected and refined. When the model produces output that is within an acceptable margin of error, the model is considered calibrated and can be reliably used under the conditions or geographic region for which it was tested.

Models are not static, but are regularly recalibrated and improved. As information of the site improves and technology advances, the model can become more robust. Each model has strengths and weaknesses under particular circumstances, such as the physical and biological conditions of the region under study; the amount of experimental experience incorporated within the models; richness of climate; and land-use and geographic information available for the analysis (Post *et al*., 2001).

The most effective means for improving model performance is parameterization, which is the process of identifying specific areas in the existing data sets used by predictive models where a lack of information is most significantly decreasing confidence, and then collecting information representative of those areas. This involves both the collection of data for existing variables as well as investigating the influence of new variables. It is the fastest way to improve soil C databases and is straightforward to accomplish.

Model structure and algorithms can always be improved.

Other indirect methods

There are other indirect methods available in addition to ecosystem models.

Remote sensing uses satellite or airborne sensors to gather data. They measure reflected radiation in a few bands of wavelength. These measurements can then be calibrated to various characteristics of the landscape by using direct C measurements. Because of the repetitive nature of image acquisition, remote sensing provides information on landscape and vegetation changes through time (Post *et al*., 2001).

Land-use history and databases are valuable in allowing the placement of current soil C levels within a historical trajectory of declining or increasing stocks. In addition, databases can allow the correlation of land-use history with enduring "signatures" that remain, for example, within the composition of microbial communities and the balance of various isotopes. Understanding these correlations can strengthen and refine models. Various databases, such as SSURGO (Soil Survey Geographic), are available through USDA NRCS, and local agencies. Land-use and land-cover databases can also be developed from remotely sensed data (Post *et al.*, 2001).

PROJECT BOUNDARIES

There are two kinds of project boundaries: physical boundaries and GHG boundaries. Physical project boundaries are defined as the area of land on which project activity occurs. This must be clearly delineated, preferably with geographic information system (GIS) or global positioning system (GPS) coordinates. Physical boundaries will also help determine the precise extent of GHG boundaries, since all changes in GHG fluxes occurring within physical boundaries will fall within GHG boundaries. Special care must be taken in the case of aggregated project activity if overlapping ownership by different parties occurs within the project's physical boundaries. Landowner responsibilities fall to the aggregator in such cases. Assessments of baseline and additionality must match this area on a *wall-to-wall* basis.

GHG boundaries include all fluxes of all GHGs affected by project activity, including leakage. Net credit quantification will include gross C sequestration in soils, avoided emissions from the ecosystem, project-associated emissions and any other significant project-driven GHG emissions reductions. Complex GHG interactions can occur within rangeland ecosystems, with or without the presence of project activity. Regional modelling and/or surveying the available scientific literature can help provide emissions factors in this regard.

For example, changes in stocking rate will lead to increases or decreases in net CH_4 emissions from livestock. Importing feed from beyond project boundaries involves increased use of fossil fuels. Using fertilizers on pasture lands is likely to lead to little or no net benefit in terms of the GHG balance, in large measure because the GWP of N_2O is so high.

BASELINE

The term *baseline* has two related meanings. First, baseline is a quantified value of C stocks before any changes in management or environmental conditions occur. Second, within the context of project credit accounting, baseline is an extrapolated value for C levels *as they would have been* in the absence of project activity, under BAU. Both are key metrics. The first helps in the calculation of the second. The second metric is used to quantify additionality and net emissions reductions generated by project activity.

To reflect the mitigation effect of project activity accurately, the forward-looking baseline should be quantified over the lifetime of the associated credits. Where data and modelling reveal positive net GHG emissions (source activity) under BAU, additionality may be achieved by implementing practices that decrease or eliminate source activity or turn the source into a sink.

It is important to consider that when BAU shows declining C stocks, projects that stop the decline (i.e. maintain the current stocks) can be credited for the otherwise expected loss. In the forestry C arena, these projects are known as Reducing Emissions from Deforestation and Forest Degradation (REDD) projects, which are relatively new. The concept of REDD is significant because it can be applicable to rangelands that are subject to destruction or disappearance through development and urbanization. The net C effects of preserving rangelands against urban development are not well known and should be studied further. Since REDD is by now well established for forests, applying it to grasslands is a feasible progression.

Rangelands in the United States contain a very high degree of spatial and temporal variation. The baseline should therefore be established regionally according to the best available resources, including USDA NRCS databases (such as SSURGO), local land-use history, ecosystem modelling, soil archives, remotely sensed imagery and associated data processing and, where necessary, discrete soil sample measurements.

For the purposes of establishing the baseline – and relevant to other areas of protocol development – boundaries between different regions must be defined. These will be determined using environmental criteria, data availability factors and economic factors around quantification. The availability of data-gathering technologies, techniques and databases will also be relevant. For example, remote sensing technologies may reduce the costs of mapping the different regions but natural biome and ecosystem boundaries will strongly influence the extent of each region and suggest natural boundaries.

ADDITIONALITY

The term *additionality*, like that of baseline, has two related definitions. The *concept* of additionality is that in order to attract compensation, emissions reductions must be in addition to what *would* have occurred under BAU. The *quantification* of additionality represents the credits that have been generated by project activity that can be transacted. Additionality is calculated against the forward-looking baseline. The concept and method of quantifying additionality are closely related to the concept and method of quantifying baseline.

Additionality is calculated as post-project C stocks less the forward-looking baseline, less deductions for leakage and *risk of reversal* (the permanence factor), and less project-generated (non-ecosystem) GHG emissions.

There are two broad approaches to establishing additionality: project-based additionality testing and use of a performance standard. Project-based testing evaluates projects on a case-by-case basis. Commonly used project-based tests (Stockholm Environment Institute, 2009) include the following.

- Legal and Regulatory Additionality Test: the project activity must not be legally mandated within a compliance system.

- Financial Test: the project is only viable if it is not profitable without revenue from emissions reductions.
- Barriers Test: the project is only additional if there are barriers that would prevent its implementation under BAU, regardless of profitability.
- Common Practice Test: the project is only viable if it employs practices not already in common use.

In the context of land-use emissions reductions, *legal and regulatory additionality* is the approach usually discussed.

Under most performance standards, determination of baseline and additionality is not sought on a project-specific basis. Instead, regional or ecosystem benchmarks are established, based on approximate or aggregated data (Stockholm Environment Institute, 2009). Benchmarks bring simplicity but the risk of inaccuracy. Ensuring purchaser confidence and real emissions reductions are critical factors.

When landowners and project developers select management practices, they are typically guided by economic factors. Practices that offer the greatest net financial return will be the most attractive. The gross revenue generated through C credits will be principally determined by the degree of additionality that each project action, or combined suite of actions, represents. The degree of additionality represented by various project actions will be determined by factors specific to project activity, factors relating to the baseline and the influence of local environmental factors, such as precipitation, soil type and land use.

Rules and the way they are applied must lead to accurate quantitative (metric) and qualitative (subjective) assessment of mitigation benefits as a result of project activity. The main challenge associated with quantifying additionality comes in determining what would occur in the absence of the project. How is this to be accurately assessed? The additionality rules of various emissions trading platforms have attracted criticism for lack of clarity, over-reliance on subjective assessment of what would have occurred in the absence of the project, and an apparent incompatibility with market dynamics. Such subjectivity, however, may be inescapable if a balance is to be achieved between the integrity of credits and not deterring investment with unworkable rules (Meyers, 1999).

Additionality poses a significant problem, particularly for rangelands, because it does not reward good land stewards who, in spite of greater costs or simply because of more altruistic land management objectives, have already achieved saturation of C stocks. Seen from a slightly different perspective, because of additionality, operations that depleted their soil C stocks prior to the trading system start date would be rewarded for their unsustainable practices because it would be easier for them to pass the additionality test and to sequester more C above the baseline.

Therefore, relevant to the concept of additionality is the idea of rewarding early adopters, parties who have acted as voluntary pioneers, often losing money in the process. In theory, such action could also be used to promote best practices and encourage future innovation. Options here include payments to offset losses, bonus credits provided by a buffer pool and non-financial rewards. The active engagement of stakeholders over this issue will ultimately ensure a higher level of industry participation.

LEAKAGE

Leakage occurs when "a carbon sequestration activity on one piece of land inadvertently, directly, or indirectly, triggers an activity which counteracts the carbon effects of the initial activity" (IPCC, 2001b). Most instances of leakage have a negative effect on the assessment of project benefit. Positive leakage occurs when management practices promote *reductions* in GHG emissions beyond project boundaries (Murray and Sohngen, 2004). Negative leakage is further categorized as either *market leakage* or *activity-shifting leakage*. Market leakage refers to increased GHG emissions outside project boundaries, resulting from substitution of goods lost as a result of project activity, when an established C market is impacted. Activity-shifting leakage occurs when activities that would occur within project boundaries under BAU are displaced beyond the project boundaries.

Landowners and project developers seek to minimize lost revenue resulting from leakage, but up to a certain threshold these emissions may feasibly go uncounted. Rangeland soil C projects may encounter less leakage than a proportion of afforestation/reforestation projects – because the land remains in production – provided that services provided by the rangelands in question are maintained or increased as a result of project activity (FAO, 2009).

For rangeland soil C projects, leakage potential exists from land that is set aside for project activity. Most of the research into soil C leakage has analysed not rangelands but cropping systems, assessing changes associated with tillage and fertilizer practices, and land retirement. This research therefore helps inform the following discussion. In addition, several strategies to assess and mitigate leakage have been developed for afforestation/reforestation projects that may be applicable for rangeland soil C projects (FAO, 2009).

Leakage from conservation projects

Leakage can occur if under project activity lands used for grazing are no longer used for this purpose. Much of the research on leakage has focused on converting cropland or forests, not open range, to habitat preserves (although grazing can have a positive effect on habitat and biodiversity). For example, such studies suggest that leakage (measured in tonnes of CO_2eq) associated with C sequestration in agricultural soils would range from less than 10 percent for working lands to 20 percent for retired land; whereas leakage associated with forest conservation could reach as high as 90 percent (CBO, 2007).

The Conservation Reserve Program (CRP) is a federal programme that retires highly erodible land from production, whether cropping or grazing. A study of cropland retired in the central United States under CRP found that for each 100 acres retired, 20 acres of non-cropland were converted to cropland in the same region (Wu, 2000), representing a secondary loss of land of 20 percent. It should be noted that lands retired from cropland to rangeland use tend to be marginally productive, and so have a lesser effect on commodity supply and leakage than more productive lands.

Wu (2000) did not examine C leakage directly. Carbon leakage is not proportional to secondary loss of land because the land entering or leaving the production base has differing potential to sequester C (Murray and Sohngen, 2004). While the research discussed above may provide some evidence of activity shifting in the agriculture sector, little empirical work has been conducted to estimate C leakage from NRCS programmes (Murray, Sohngen and Ross, 2007).

Estimating local leakage separately could assist project designers to mitigate it, since local leakage is more likely to be within their control than distant leakage. The state of the art method in market leakage estimation uses aggregated data (regional and national) either in statistical or simulation models. There are many models and data sets available that factor in market phenomena, policy impacts and leakage analysis at the county, regional or national level. However, separating market leakage into local and distant varieties is challenging because it is difficult to identify how changes in one parcel affect the management of neighbouring parcels. National and transboundary leakage quantification may be addressed through monitoring key indicators and using standard risk coefficients (Watson, 2000).

Recent advances in statistical techniques, such as spatial econometrics, may allow leakage to be estimated at a fairly disaggregated level. Such estimations, however, often require a large amount of primary data, which it may be impractical to collect in the case of many rangeland soil C projects. Local leakage may be best handled through project and contract design, by extending the C accounting boundary beyond the boundaries of the project. This will allow any localized shifting of activity in response to the project to be covered in the project accounting system and not generate unaccounted leakage locally (Murray and Sohngen, 2004).

PERMANENCE

Permanence refers to the stable retention of newly sequestered C for the duration of the project contract. Usually the period is 100 years. This means that if a credited tonne of CO_2 is released back to the atmosphere before this period is complete, the credit loses all or part of its value. Securing achieved mitigation benefits in terrestrial ecosystems requires addressing the risk of reversal. This is because land-use projects are considered to be more susceptible to natural disaster than other project categories, and to changes in either landownership or management practices. Any of these may affect the permanence of C stored in soils. The risk of non-permanence is much lower when adoption of soil C sequestration practices also leads to more sustainable or profitable farming systems (FAO, 2009), or is embedded within system-wide GHG emissions reductions transitions.

C crediting policy must include a mechanism for handling permanence to ensure that payments for C sequestration are not under- or overvalued. If a programme makes per tonne payments equal to the value of permanent sequestration, overpayments would occur if changes in land-use or management practices re-released C back to the atmosphere, unless payments are adjusted for these releases (Lewandrowski *et al*., 2004).

The solution has been broadly identified, in the sense that liability provisions will be required in any sequestration crediting. However, a single instrument is yet to receive universal acceptance (Rose, 2008). Suggestions include having projects run in perpetuity, debits for all releases, project replacement and partial or initially delayed credits. Permanence may also be addressed through various internal and external risk reduction approaches, including good practice management systems, project diversification, self-insurance reserves, standard insurance services, involvement of local stakeholders and regional C pools (Watson, 2000).

The mechanisms that have received most attention include creating a buffer, comprehensive accounting, *ex ante* discounting and temporary crediting/leasing.

Creating a buffer

The Voluntary Carbon Standard (VCS) aims to remove the risk to permanence by using a *buffer pool*. Every project undergoes a risk assessment to determine how many credits from the project will be contributed to the buffer pool account. The intention is to ensure that credits are fungible, so that if the project collapses, the buffer account can fill the credit gap, and the credit can be traded interchangeably with any other VCS credit. Remaining questions around this approach include the necessary size of the account and how it would actually work in practice (Rose, 2008). The Climate Action Reserve also uses a buffer pool within its Forestry Protocol 3.0 (CAR, 2009).

Comprehensive accounting

This method balances debits and credits as they occur over time, and is consistent with national GHG accounting practices as currently used by Annex 1 countries (IPCC, 1996). It can be based on changes in C stocks or average storage over a specific time. C stocks are measured at regular time intervals and credits are quantified accordingly. Given the frequency with which credits and debits may be exchanged, an average storage approach has been suggested to credit the average amount of C stored by a project over an extended period of time, smoothing out temporary stock fluctuations (Schroeder, 1992). One of the downsides of comprehensive accounting is the high amount of MMV required (Murray, Sohngen and Ross, 2007).

Ex ante discounting

This approach accounts for the possibility of loss by reducing the number of credits from the outset, based on the expectation of reversal. If it is expected that sequestered C may be released in the future, the expected amount and timing of this release is estimated and values adjusted accordingly. Standard financial discounting methods are used to calculate the equivalence of any delayed releases in proportion to the permanent emissions reductions for which they are being traded (Murray, Sohngen and Ross, 2007). Net C sequestration values are based on assumptions of the permanence of storage, rather than observed outcomes. This simple formula allows for easy implementation of this approach; the trade-off is a potential lack of accuracy.

Temporary crediting/leasing

Based on the idea that practices may only yield temporary reductions in atmospheric CO_2, this approach places a finite life on the credit. Reversal risk is addressed by treating the credit as if it must be redeemed in the future. Credits could carry expiration dates, at which time they would have to be regenerated by continuing the sequestration project, establishing a new project, or otherwise achieving a permanent reduction in emissions. A high amount of MMV is needed, but this approach would allow for upfront payments and may encourage uptake by landowners. Temporary credit leasing is not a popular option with some project developers, however, who consider it unrealistic and not suited to real market dynamics.

Some project developers are willing to use temporary crediting/leasing, while others are not. Buyers/lessors are seeing this as a purchase versus lease economic decision, subject to clarity on the rules, which will not arrive until after legislation has passed and rule-making is complete. Either way, liability for reversals has to be addressed.

Increased productivity provided by more sustainably managed rangelands also provides certain disincentives to reversal, although this will vary case by case.

OWNERSHIP

Ownership refers to the issue of who has legal claims to the land used for project activity, and what the process is for addressing all claimants, in order to avert litigation. Ownership of credits usually resides with the landowner, unless otherwise specified in the project design and contract. In the case of soil C sequestration and other GHG emissions reduction activities on rangelands, varying land and livestock ownership and management scenarios could create different credit ownership scenarios. The combinations include the following:

- land and livestock ownership are the same;
- land and livestock have different private ownership;
- land ownership by a land trust and private livestock ownership;
- private land with easement (e.g. land trust) with private livestock ownership;
- public land agency permits ranching on state or federal lands (livestock are privately owned, but the land is publicly owned and maintained by the rancher);
- livestock have access to both private and public land;
- public funds are used for management practices that yield C benefits;
- changes in ownership of the land and/or the livestock over time;
- an agency seeks to reclaim mineral rights on privately owned land.

A rangeland protocol should specify which party will own the credits. In case of controversy, there are ways to prevent and resolve potential disputes, including: establishing a contract with interested parties; including relevant information within the documentation when buying, selling, or leasing land; or involving a third party verifier to facilitate the process. Ownership of credits on leased land should be subject to private contracts between the landowner and rancher.

Only private land ownership is considered within the scope of this paper. A host of other issues and potential solutions arise for project activity on federal, state and other publicly owned lands. These will be important to address if the 262 million acres of publicly owned grazing lands in the West become available for C sequestration project activity.

ENVIRONMENTAL CO-BENEFITS

Sequestering C in rangeland soils brings about a number of positive environmental outcomes, or co-benefits, beyond offsetting GHG emissions, including its effects on soil quality – a term used to describe the fitness of soils to perform particular ecosystem functions by Weil and Magdoff (2004). SOC is a critical macronutrient in soils that supports a host of ecosystem functions. Increasing SOM content improves aeration and soil tilth, and decreases bulk density by increasing soil porosity. SOM plays an important role in determining soil chemical properties including pH, nutrient availability and cycling, cation exchange capacity and buffer capacity (Tisdale, Nelson and Beaton, 1985; Evrendilek, Celik and Kilic, 2004). Soil aggregation and aggregate stability are also improved by increased SOM accumulation (Gollany *et al*., 1992; Tisdall, 1994).

Changes in agricultural practices that increase C sequestration can also improve water quality (Greenhalgh and Sauer, 2003; Pattanayak *et al.*, 2005). Increased SOM content improves water infiltration and water-holding capacity of soils (Tisdale, Nelson and Beaton, 1985; Greenhalgh and Sauer, 2003). Water quality is further enhanced by an associated reduction in soil erosion and sedimentation (Zebarth *et al*., 1999; Celik, 2005). Increasing SOM is an effective method for increasing drought resistance in arid areas (Overstreet and DeJong-Huges, 2009), by increasing the soil's ability to retain water that falls on it and passes through it. This is of critical importance in a changing climate, and where the economic viability of ranching operations may already be in question.

Improvements in soil water quality and availability can increase productivity (Mader *et al*., 2002; Huston and Marland, 2003). There is also a strong correlation between the size of the SOC pool and both soil physical fertility and forage production (Mader *et al*., 2002; Blair *et al*., 2006). Soil management affects biodiversity and ecosystem functioning (Huston and Marland, 2003). Soils with higher organic matter can support a more diverse array of soil micro-organisms (Lal *et al*., 2007; Evrendilek, Celik and Kilic, 2004).

Soil management methods that increase C inputs to the soil, such as manuring, are often observed to enhance microbial biomass, populations and activities (Acea and Carballas, 1999; Ritz, Wheatley and Griffiths, 1997; Witter, Martensson and Garcia, 1993). The long-term use of manure also supplies large amounts of readily available C, resulting in a more diverse and dynamic microbial system compared with inorganically fertilized soil (Peacock *et al*., 2001). Biodiversity of soil fauna and flora are strongly correlated with soil quality and its functions (Bohlen, Edwards and Edwards, 1995; Huston and Marland, 2003).

Management practices to increase soil C sequestration may in some cases have a negative environmental impact. For example, the addition of animal manure to the soil can alter plant community composition by modifying competitive interactions between plant species. In addition, uncomposted manure may introduce seeds of invasive species or have a detrimental effect on water quality, depending on factors such as manure concentration and type, application method, location and timing, and precipitation patterns.

Methods used to sequester C in soils include increasing C inputs to the soil through changes in production or allocation by fertilization, irrigation, sowing legumes or more productive grass species, or by improving grazing management (Paustian *et al*., 1997; Conant, Paustian and Elliott, 2001). Practices such as N fertilization (on pasturelands) could lead to leaching, and increases in N_2O emissions that offset the benefits of C sequestration (Conant *et al*., 2005).

Preservation and restoration of woodlands and trees at lower densities within rangeland landscapes can provide significant soil C benefits, and other benefits associated with those. Forage quality and quantity under California oaks have been found to be significantly greater than for areas where oaks have been removed (Dahlgren, Singer and Huang, 1997; Camping *et al*., 2002). Soil C levels under some California oak species can be higher per unit area than in the trees themselves (Gaman, 2008). Grazing can deter invasive weeds, shrubs and trees (e.g. Franzluebbers, Franzluebbers and Jawson, 2002), often with positive effects on avian habitat.

Because of the many functions performed by soil C and the degraded status of many soils, there is a high potential for positive environmental impacts as a result of the implementation of rangeland soil C projects. Most changes in rangeland management that are intended to increase C sequestration represent a shift towards more sustainable management practices. However, each practice needs to be assessed for any potential negative impacts.

MARKET INTEREST

A robust rangeland methodology should be cost-effective, transparent and provide real benefits in the forms of GHG emissions reductions and more resilient rangeland ecosystems. The ultimate economics of this methodology, however, will not be known until actual development begins.

The Waxman-Markey ACES Bill – the American Clean Energy and Security Act 2009 – that has passed in House and has not, at the time of writing, passed in the Senate, is designed to reduce national GHG emissions by 80 percent against 2005 levels by 2050. The passage of such a bill, promoting a national cap and trade system, would increase demand for the development of land-based C sequestration and the necessary methodologies, spurring faster and greater increases in the prices for precompliance, and then compliance, and emissions reductions credits. In Europe, the size of the compliance market proved to be eight times that of the precompliance market. In the United States, the consensus in 2008 was that there were not enough quality credits available to meet demand even from voluntary and precompliance markets (Barbour and Philpott, 2008).

"Carbon federalism" is in effect what sees regions and states acting as laboratories for C regulation and creating momentum for federal legislation (Berendt, 2008). Under California's Assembly Bill 32, among the country's leading climate change legislation, 85 percent of emissions will be capped. Under the Western Climate Initiative (WCI), comprising 11 partner and 14 observer states and provinces from Nova Scotia to Mexico, including California, up to 49 percent of reductions may initially be achieved through offsets (CARB, 2008).

It has been predicted that after the climate talks in Copenhagen in December 2009, prices for C (not CO_2) in the United States will reach USD73 per tonne (Point Carbon, 2009). Investors acquainted with terrestrial C through forestry credits are becoming aware of soil C sequestration. The quality of these credits and actual potential of this opportunity depend upon the quality of the associated methodology and the confidence it attracts.

The term *slippage* is sometimes used to refer to deductions from revenue resulting from costs associated with a particular GHG emissions reductions typology. From the investor's or project developer's perspective, AFOLU (Agriculture, Forest and Land Use) typologies come with several drawbacks: lower returns, more slippage – from buffers, leakage, verification costs and project costs – low near-term yield and the risk of liability with respect to permanence.

Therefore, activation on the open market of the mitigation potential represented by rangelands is likely to require price signals that are significantly higher than those that have been offered on the Chicago Climate Exchange (CCX, 2009), or alternatively through a public sector programme.

Voluntary emissions reductions have traded as private sales via the Climate Action Reserve for significantly more than USD10 per tonne.

SUMMARY

If the effects of global warming are to be minimized, C already emitted to the atmosphere must be sequestered into stable forms. Soil C sequestration appears to be one of the most cost-effective ways of achieving this. Rangelands cover 31 percent of the land surface area of the United States and grazing is the chief activity on these lands, with the potential to mitigate at least 3.3 percent of United States CO_2 emissions from the combustion of fossil fuels, every year for 30 years or more, until saturation is reached.

Project actions with the potential to increase soil C in rangelands include: conversion of abandoned and degraded cropland to grassland; avoided conversion of rangeland to cropland or urban development; adjustments in stocking rates; integrated nutrient management; introduction or reintroduction of grasses, legumes and shrubs on degraded lands; managing invasive species; reseeding grassland species; addition of trees and shrubs for silvopastoralism; managing invasive shrubs and trees; riparian zone restoration; and the introduction of biochar into soils.

Soil organic C forms 50 percent of soil organic matter and is a critical macronutrient in soil ecosystems, driving many other nutrient cycles. Each new tonne of soil C represents the removal of 3.67 tonnes of CO_2 from the atmosphere. Soils hold over three times as much C as the atmosphere and because of historic depletion have the capacity to store much more. The unique role of C in the soil system offers the potential for win–win scenarios for climate change mitigation, the environment, project developers and landowners. Activating this potential depends largely on the methodology or performance standard employed.

Within a protocol, an existing method of quantifying soil C may be used or several different methods may be harnessed into a combination methodology. Either way, a balance must be achieved between ease of use and accuracy. A balanced methodology will lead to the optimization of the potential for additional soil C sequestration in United States rangelands. An understanding of ecosystem states and the varying responses of different soil landscapes to the same changes in management will be critical to the development of an accurate and efficient methodology or performance standard.

Landowners will be compensated for changes in management or for quantified increases in soil C stocks. There are pros and cons associated with each approach. A hybrid is also possible. If changes in management are to be rewarded, close correlations must be established between those changes and the effect they have on C stocks. If changes in stocks are to be quantified, this may occur on a per project basis or according to regional and ecosystem benchmarks.

Our analysis focuses largely on options for methodologies that compensate for achieved changes in C stocks. In this regard, a performance standard can be used – its complexity depends on how it is designed – or alternatively a more site-specific methodology, which would almost certainly be more costly and difficult to apply. Along a conceptual continuum of all potential quantification methodologies, critical opportunities exist where non-linear breakthroughs occur in quantification efficiency. These locations on the graph are the natural places to develop new (or adapt and use existing) methodologies. These points offer the best potential for elevated adoption rates, climate change mitigation, socio-economic and ecosystem benefits.

The protocol development process will benefit early on from an analysis of these points, matched to the shortlisted protocol options. In fact, if the goal is to maximize any or all of the above benefits, such analysis could be used as the primary basis for selection of the final soil C and credit quantification format.

Although this paper focuses on the dynamics of private sector trading systems, public sector programmes are also an important option, and are likely to come with reduced transaction costs. Arguably, the buying power and risk-carrying power of government agencies may achieve results beyond the reach of the more heterogeneous private sector.

Permanence, or the risk of reversal, is a major issue that needs to be addressed within a rangeland soil C protocol. Broadly, the solution of discounting has been agreed upon and has been tested in the context of forestry C sequestration; within this umbrella, there are a number of different instruments available, with none yet receiving universal acceptance.

Demand for high-quality rangeland soil C credits is likely to be high within a compliance system such as federal cap and trade or other programme, provided that risks to the private or public sector are addressed through measures such as conservative discounting and buffer pools.

The unique benefits to the environment and producers associated with increasing soil C stocks in United States rangelands should provide the necessary impetus to overcome hurdles on the path to protocol development. Some solutions may only become apparent once the process has begun.

APPENDIX 1

Activity-based and soil carbon measurement hybrids

Hybrids blending activity-based performance standards and site-specific measurement are also possible. For example, regional baselines could be established from existing databases and the published literature; thereafter, post-project soil C levels could be quantified on a site-specific basis.

Alternatively, a practice-based performance standard could include an opt-out option whereby landowners or project developers pay extra to have post-project soil C quantified, when they are confident of significant gains above benchmarks. This would allow new project actions to be included within global protocol activity without being written into the core protocol at its inception. This format would prevent the protocol from becoming burdened by a proliferation of project actions, while still encouraging innovation and optimization of climate change mitigation and other benefits.

New project actions would need to be assessed for their net effect on GHG emissions. However, once a project action has been added to the record, other landowners could implement it without a repetition of the primary assessment; indeed, range managers could reference a growing (online) database of project actions that have been admitted in this way. Thus, they would have an effective soil C management tool to use when developing global ranch management plans. Potential interactive effects of different project actions on net GHG flux will also need to be assessed.

Within this format, the one-time assessment costs needed to register a new project action could either be covered by early adopters or subsidized from a slightly augmented buffer pool (primarily established to address permanence, leakage and margin of error). Enough unique measurements could in time inform new benchmarks.

The added attractions of this format would be in allowing landowners to interact with the system as they choose; and that the community of landowners would decide which project actions would be added to the protocol. A natural selection of additional project actions would occur, with the protocol growing organically and efficiently, with some reduction in administrative costs.

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