Part 645 – National Range and Pasture Handbook

Subpart K – An Ecosystem View of Range and Pasture Soil Health

645.1101 Introduction: A Holistic View of Soil Health and Soil Quality on Range and Pasture

A. The purpose of this subpart is to provide information about

(1) The importance of ecosystem functions in determining overall health on range and pastureland;
(2) Concepts of soil quality and health and how they apply to rangeland and pasturelands;
(3) Discussions of carbon cycle and sequestration dynamics;
(4) Reviews of technologies and tools that can be used to assess overall range and pasture health;
(5) Identification of opportunities to provide information about the components of range and pasture health as an important part of grazing land management.

B. The terms rangeland and pastureland health are use in this document to encompass the full suite of ecosystem components, including soil health concepts.

C. A considerable amount of literature on the health of range, pasture, and soils has been published by the Soil Health Division (USDA-NRCS 2021a), other NRCS sources, university and government agency technical documents, books, journal articles, and the popular literature. A logical first step in becoming knowledgeable on the subject of health and the functionality of grazing lands is to understand that an environmental variable does not act alone. Grazingland plant communities are diverse and multivariate in nature, meaning that many variables (climate, geophysical factors, soils, plants, and biological components) all interact and act in concert within the ecosystem, which implies a holistic view to plant and soil health. Although soil health in particular is a key subject of importance in agriculture, addressing singular functions relative to soil, plants, and biotic components should be secondary.

D. In addition to climate, geophysical factors, soils, plants, and biological component interactions, the health of an ecosystem also includes ecological processes such as the hydrologic cycle, energy flow, the nutrient cycle, and the function of each within a natural range of variability. Since the complexity of environmental factors and ecological processes within grazing land systems are multivariate in nature, internal environmental processes are usually unique from one ecological site to another. The ecological site (ES) and its full range of dynamic attributes (see subpart B) is the foundation for NRCS conservationists in addressing range and soil health. The parameters of the ES are the key to addressing range and pastureland health. An ecological site is a conceptual classification of the landscape. It is a distinctive land unit based on a recurring landform with distinct soils (chemical, physical, and biological attributes), kinds and amounts of vegetation, hydrology, geology, climatic characteristics, inherent ecological resistance and resiliency, unique successional dynamics and pathways, natural disturbance regimes, geologic and evolutionary history including herbivore and other animal impacts, and response to management actions and natural disturbances. These discrete characteristics separate one ecological site from another. A range, pasture, and soil health assessment system must be based on a realistic baseline or standard as described in an Ecological Site Description (ESD) and requires a means of identifying current status and communicating the implications of departure from the standard. The ESD is the document that contains information about the individual ecological sites.

E. Direct quantitative measures of processes relating to biotic, soil dynamics, and hydrology are usually very time consuming, expensive, and may not be feasible for conservation planners and grazing land managers. It is important to point out that there are situations where detailed tests and analyses related to soil health are desirable and needed, especially in cropland settings (cash and specialty crops, etc.) (USDA-NRCS 2021a). For example, Cornell University has a comprehensive protocol: “Cornell Comprehensive Assessment of Soil Health (CASH).” Moebius-Clune (2016) provides a thorough examination of the physical, chemical, and biological indicators of soil health including soil organic matter, structural stability, microbial activity, diversity, and microbial and plant available nutrients. In addition to quantitative analyses of soil health, the In-Field Soil Health Assessment (Technical Note 450-06) is used to determine soil health using a mixture of quantitative and qualitative observations of biological and physical indicators. Observable biological and physical indicators can be used as indicators of the functional status of ecological processes (Printz et al. 2014; Brown and Herrick 2016; Pellant et al. 2020).

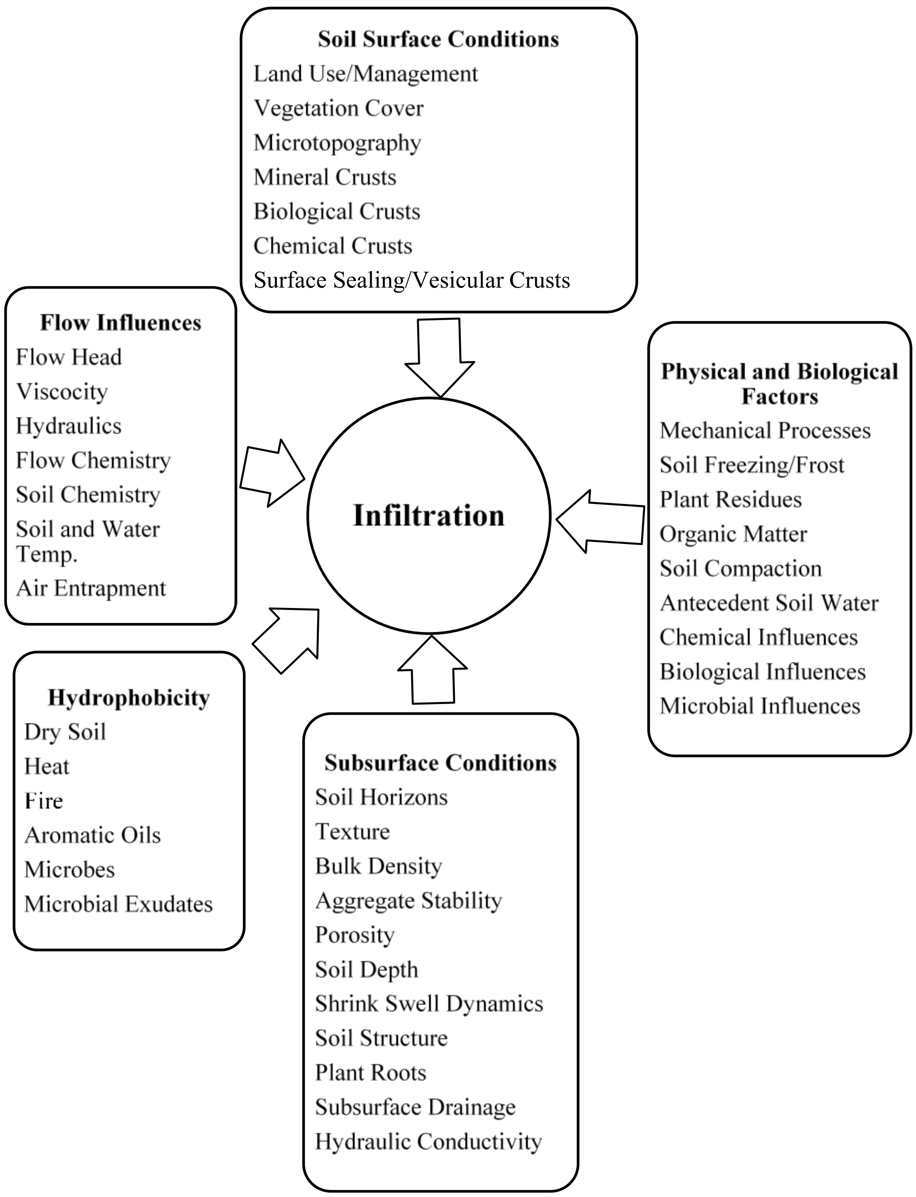
F. Some ecosystem processes or variables can be directly observed and measured, while other ecosystem processes or variables are inferred through other directly measured or observed characteristics. Statistically, variables that are not directly observed but are rather inferred (through a mathematical model) from other related variables are called latent or hidden variables. For example, a person’s overall health is a latent variable – there isn’t a single measurement of “health” that can be measured. Health is an abstract concept; however, measures of physical and chemical properties from our bodies, such as blood pressure, cholesterol level, weight, various measurements (waist, hips, chest), blood sugar, temperature, and a variety of other measurements are used holistically. The opposite of latent variables are manifest, observable, or indicator variables or factors that can be directly measured or observed. The same concept applies to rangeland and pastureland health. For example, measuring hydraulic properties such as infiltration capacity as a function of soil health in the field can be highly variable in space and time because there are many interacting factors (figure K-1). Infiltration capacity can be measured in the field with double ring infiltrometers, disc permeameter, Guelph Permeameter, and rainfall simulators. However, field infiltration measurements require specialized equipment, considerable time, and expense. Also, locating replicated samples, even within short distances on the same soil type, often results in high variability. Indicators such as level of compaction, amount of bare ground, plant cover, plant productivity, litter amount, lack of significant evidence of accelerated runoff and erosion, and nature of soil aggregates can be used to evaluate whether infiltration capacity is in sync within normal parameters for the site.

G. Measuring soil organic matter, chemical analyses, and macro and micro-nutrients can also be highly variable. Cornell University recommends sampling in about 10–15 locations throughout the garden or field, mixing the sub-samples, and obtaining a representative composite sample. In cases where erosion has occurred or significant changes in condition are apparent across the soil component or within the specific ecological site, stratification of the sample and sampling separate sites is warranted.

H. Evaluation of short-term grazing effects on range and pasture health may also not be readily detectable. Biomass, plant height, litter, plant cover, soil compaction, and keeping bare ground withing limits are correlated with short-term grazing effects and are also strong indicators of long-term grazing management. No single factor can assess grazing effects, although outcomes may be predictable and documented in state-and-transition models. Grazing impact has an immediate effect on the soil surface and plant growth and composition, which successively affects hydrology, energy and nutrient cycles, and erosion and sedimentation dynamics (figure K-2). Therefore, one measure cannot be used to evaluate grazing effects, thus the importance of a holistic approach to evaluating rangeland health.

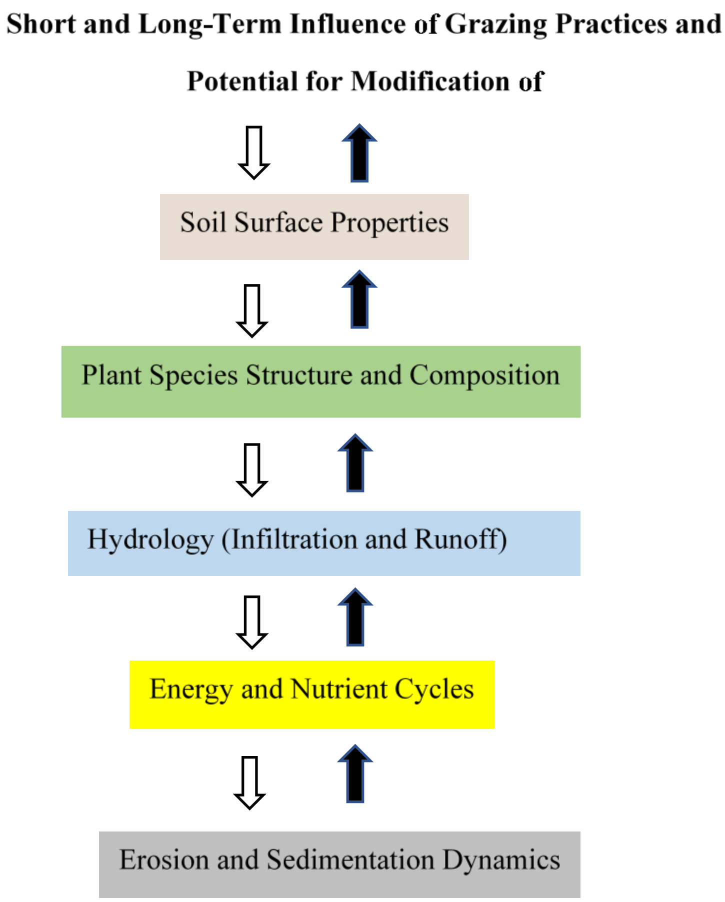
I. Various range and pasture health protocols rely on specific indicators whose characteristics (e.g., presence or absence, quantity, distribution) are used as an index of an attribute such as soil or site stability, hydrologic function, and biotic integrity. It is important to acknowledge that a single indicator or attribute cannot represent health on rangeland, pastureland, or cropland. Interpreting Indicators of Rangeland Health (IIRH) (Pellant et al. 2020), Pasture Condition Score Sheet (PCSS), relies on a suite of key indicators for an overall numerical score assessment; whereas Determining Indicators of Pasture Health (DIPH) (Spaeth 2020, see subpart E) evaluates three separate attributes (soil or site stability, hydrologic function, and biotic integrity). The IIRH protocol is the preferred tool to evaluate rangeland health, and the user may choose PCSS or DIPH for determining pasture health. The PCSS is generally a quick assessment, whereas DIPH provides an evaluation of soil and surface stability, hydrologic function, and biotic integrity.

Figure K-1. Interrelated factors associated with infiltration.



J. Interpreting Indicators of Rangeland Health lists appropriate applications and limitations (Pellant et al. 2020). For example, IIRH is determined with an appropriate reference sheet that is specific to the ecological site. If an ecological site has not been identified or written, then the procedure for Describing Indicators of Rangeland Health can be used (see Pellant et al. 2020). Interpreting Indicators of Rangeland Health is to be used by knowledgeable and experienced people with prior training (either a workshop or working closely with other users). The IIRH protocol is not to be used for identifying cause(s) of resource problems, and when used for monitoring or determining trends, quantitative data should also be used to support the indicator ratings (e.g., production, plant species ID and extent, invasive species ID and extent, and erosion and hydrology evaluations). The Rangeland Hydrology and Erosion Model can be used in conjunction with Soil and Surface Stability and Hydrologic Function (see subparts B and G). The DIPH tool can also be used with supporting information in the Ecological Site Description or be used as a stand-alone assessment with supporting documentation. The PCSS is designed to be used as a stand-alone tool.

Figure K-2. Model depicting influence of grazing practices on soil surface and subsequent results on plant communities, hydrology, energy and nutrient cycles, and erosion and sedimentation dynamics.



645.1102 Ecosystem Components of Range and Pastureland Health

A. Some of the major components related to range and pasture health are addressed in this section. Two terms, “ecosystem” and “plant community,” are reviewed because they are integral components to range and pasture health.

(1) An ecosystem is a community of plants, animals, and soil organisms and their environment combined as a functional system of complementary relationships and transmission of energy and matter (Whittaker 1975).

(2) A plant community is an assemblage of populations of plants that interact in a specific environment, forming a distinctive living system with inherent composition, structure, environmental relationships, historic development, and function. The function of plants

within the concepts of IIRH (Pellant et al. 2020) and DIPH (Spaeth 2020) are addressed principally as biotic integrity (tables K-1, K-2). The biotic community includes plants (vascular and nonvascular), animals, insects, and microorganisms occurring both above and below ground. Pellant et al. (2020) define biotic integrity as the “the capacity of the biotic community to support ecological processes within the natural range of variability expected for the site (ecological site associated plant community), to resist a loss in the capacity to support these processes, and to recover this capacity when losses do occur.”

B. In all terrestrial plant communities, both soil and surface stability and hydrologic function are interconnected with biotic components. Each dynamic affects and reacts to each other. As mentioned in the introduction, “many variables (climate, geophysical factors, soils, plants, and biological components) all interact and act in concert within the ecosystem.” For example, in IIRH, certain indicators overlap for all three assessments (table K-1, e.g., soil surface resistance to erosion, soil surface loss and degradation, and compaction layer). Litter cover and depth span both hydrologic function and biotic integrity (table K-1). Rills, water flow patterns, pedestals and terracettes, bare ground, and gullies encompass both soil and site stability and hydrologic function. Pellant et al. (2020) define soil or site stability as “the capacity of an area to limit redistribution and loss of soil resources (including nutrients and organic matter) by wind and water and to recover this capacity when a reduction does occur; and hydrologic function – the capacity of an area to capture, store, and safely release water from rainfall, run-on, and snowmelt (where relevant), to resist a reduction in this capacity, and to recover this capacity when a reduction does occur.”

Table K-1. Three attributes of rangeland health and 17 associated indictors. Indicators are arranged under the respective attributes. Some indicators span more than one attribute. (Pellant et al. 2020.)

<table>
<tr><th>Soil/Site Stability</th><th>Hydrologic Function</th><th>Biotic Integrity</th></tr>
<tr><td colspan="2">1. Rills</td><td>12. Functional/Structural Groups</td></tr>
<tr><td colspan="2">2. Water Flow Patterns</td><td>13. Dead or Dying Plants or Plant Parts</td></tr>
<tr><td colspan="2">3. Pedestals and/or Terracettes</td><td>15. Annual Production</td></tr>
<tr><td colspan="2">4. Bare Ground</td><td>16. Invasive Plants</td></tr>
<tr><td colspan="2">5. Gullies</td><td></td></tr>
<tr><td>6. Wind-Scoured and/or Depositional Areas</td><td colspan="2">14. Litter Cover and Depth</td></tr>
<tr><td>7. Litter Movement</td><td>10. Effects of Plant Community Composition and Distribution on Infiltration</td><td>17. Vigor with an Emphasis on Reproductive Capability of Perennial Plants</td></tr>
<tr><td colspan="3">8. Soil Surface Resistance to Erosion</td></tr>
<tr><td colspan="3">9. Soil Surface Loss and Degradation</td></tr>
<tr><td colspan="3">11. Compaction Layer</td></tr>
</table>

C. Table K-2 shows the commonality between IIRH and DIPH. There are common and unique indicators for DIPH because they represent specific characteristics of pasture environments. Unique indicators associated with DIPH include forage plant diversity, percent desirable forage plants, percent non-toxic legumes, based on adaptability with the ES and expected stand and longevity for the site, uniformity of use, and grazing and utilization. Seven livestock management factors are included in DIPH to focus on issues that are specific to livestock management. Although DIPH has over twice as many indicators as PCSS, certain indicators may not have issues, such as rill, wind, gully, and streambank erosion, and percent legumes; therefore, the field assessment process can proceed quickly. In field tests, comparison of time to complete PCSS and DIPH differ by not more than 10 minutes. The PCSS protocol derives a number score, where DIPH evaluates “preponderance of evidence” to determine the functional status of the three rangeland health attributes (see subpart E). The preponderance of evidence approach is used to select the appropriate departure category for each attribute and the overall decision for each of the three attributes. This assessment is based, in part, on where the majority of the indicators for each attribute fall under the five categories (none to slight, slight to moderate, moderate, moderate to extreme, and extreme to total).

Table K-2. Proposed Matrix for Determining Indicators of Pasture Health (DIPH). Comparison of indicators in rangeland health matrix and proposed matrix for Determining Indicators of Pasture Health. LMQF=Livestock Management Quality Factor.

Interpreting Indicators of Rangeland Health V 5		Determining Indicators of Pastureland Health	Assessment
1. Rills	SSS, HF	Erosion (Sheet and Rill)	SSS, HF
2. Water-flow Patterns	SSS, HF	Water-flow Patterns	SSS, HF
3. Pedestals and/or terracettes	SSS, HF	Pedestals and/or Terracettes	
4. Bare ground	SSS, HF	Bare Ground %	SSS, HF
5. Gullies	SSS, HF	Erosion (Gullies)	SSS, HF
6. Wind-scoured, blowouts, and/or deposition areas	SSS	Erosion (Wind)	SSS
		Erosion (Shoreline) if present	SSS, HF
7. Litter movement	SSS	Litter Movement	SSS, HF
8. Soil surface resistance to erosion	SSS, HF, BI		
		Live Plant Foliar Cover (hydrologic and erosion benefits)	SSS, HF
9. Soil surface loss and degradation	SSS, HF, BI	Soil Surface Loss or Degradation	SSS, HF, BI
10. Effects of plant community composition and distribution on infiltration and runoff	HF	Effects of Plant Community Composition and Distribution on Infiltration and Runoff	HF
11. Compaction layer	SSS, HF, BI	Compaction Layer	SSS, HF, BI
12. Functional/structural groups	BI		
		Forage Plant Diversity	BI, LMQF
		Percent Desirable Forage Plants (for identified livestock class)	LMQF
13. Dead or dying plants or plant parts	BI	Dead or Dying Plants or Plant Parts	BI
14. Litter cover and depth	HF, BI	Litter Cover and Depth	HF, BI
15. Annual production	BI	Annual Production	BI, LMQF
16. Invasive plants	BI	Invasive Plants	BI
17. Vigor with an emphasis on reproductive capability of perennial plants	BI	Plant Vigor with an emphasis on Reproductive Capability of Perennial Plants	BI
		Percent non-toxic Legumes (based on adaptability with Ecol. Site and/or what is expected stand and longevity for the site)	BI, LMQF
		Uniformity of Use	HF, BI, LMQF
		Grazing and Utilization	BI, SSS, HF, LMQF

D. If an ecological site does not exist or has not been completed, or the pasture state narrative in the ecological site description is not complete, the DIPH matrix can be used as a “stand-alone” document to determine indicator status. Likewise, if an ecological site is not available or does not exist, as is often the case on public lands, the protocol Describing Indicators of Rangeland Health can be used for rangeland health (see subpart E and Pellant et al. 2020). If repeated DIPH assessments are made on specific ecological sites, data can be collected to help develop the narrative for pasture groups and the ecological site description converted pasture state. In table K-1, several indicators can be evaluated with ecological aspects inherent to the ecological site. For example:

(1) Annual production capacity
(2) Percent non-toxic legumes (based on adaptability associated with ES or what is the expected stand for the site)
(3) Forage plant adaptability and projected diversity
(4) Litter amount and plant residue
(5) Erosion (sheet and rill)
(6) Erosion (gullies)
(7) Erosion (wind)
(8) Water flow patterns
(9) Percent bare ground
(10) Soil health attributes
(11) Dynamics of weeds and invasive plants

645.1103 Soil Health and Quality (selected excerpts included from Spaeth 2020)

A. Soil health and quality are not new ideas for sustainable agriculture; however, they have gained a resurgence among land users, especially on cropland. Many current popular publications and scientific studies published in peer-reviewed journals address new ideas and research on cover crops, applying manure and other soil amendments, practicing crop rotations, and using minimum tillage or no-till cropping systems. In 2013, the Food and Agriculture Organization of the United Nations (FAO) emphasized a greater focus on soil health (FAO-2013). FAO stated:

(1) "More attention to the health and management of the planet's soils will be needed to meet the challenge of feeding a growing world population . . . the importance of soil for food security should be obvious. From the origins of civilization in early farming communities up through today, we can see how societies have prospered thanks to healthy soils and declined when their lands became degraded or infertile . . ."
(2) "Healthy soil is not only the foundation of food production, but serves other functions . . . for example, soil is critical to the health of ground and surface waters and ecosystem health, and sequesters twice as much carbon as is found in the atmosphere . . . until recently, soils were the most overlooked and widely degraded natural resource. Today that state of affairs has at last begun to change, with World Soil Day poised to be recognized by the United Nations and a new, international Global Soil Partnership."

B. The resurgence of interest in soil health across all land uses is due to a variety of reasons:

(1) Exponential growth and interest in organic food products and farming ($43 B according to the Organic Trade Association's industry; Global sales of organic foods amounted to about 81.6 B U.S. dollars in 2015) (Statista 2018).
(2) The USDA reported 19,474 organic farms, ranches, and processing facilities in 2015, up more than five percent from the previous year and 250 percent from 2002, when record-keeping began. Throughout the world, there are more than 27,800 organic producers.
(3) Impetus from the FAO International Year of Soils (2015) by the United Nations Food and Agriculture Organization.
(4) Interest worldwide on landscape sustainability.
(5) Earth's population growth.

(6) Increasing recognition of degraded resources throughout the world.
(7) Interest in sustainable crop production systems that reduce tillage operations and wind and water erosion, use of no-till and reduced tillage to reduce soil disturbance.
(8) The correlation between soil health and human health.
(9) Greater affluence throughout the world and desire for higher protein diets.
(10) Desire to improve water quality.
(11) Maintaining productive farmland and rangelands for future generations.
(12) The economics of reducing fuel, soil amendments, and wear and tear on equipment.
(13) Improve and sustain crop yields and productivity.
(14) Maximize the local water budget, improve hydrologic function, and manage soil moisture.
(15) Enhancing wildlife habitat.
(16) New science and better information about the quantitative effects of soil health.

C. Conservationists and land managers should not lose sight of the functionality of the ecosystem in place. Soil health, plant health, soil and site stability, hydrologic function, biotic integrity, energy, and nutrient cycling, are all integral to the health of the plant community—one process does not exist or function without the other.

D. Soil functions in various capacities:

(1) Sustains biological components, their activity, diversity, and ultimately productivity
(2) Regulates and partitions water as a function of the hydrologic cycle
(3) Serves as an environmental filter by buffering and degrading organic and inorganic matter
(4) Stores and cycles nutrients and other elements in the environment
(5) Provides the foundation and support for the infrastructure of civilization

E. The concepts of soil health and quality are often intertwined and used synonymously; however, soil quality is associated with a soil's natural composition, while soil health is related to the capacity and functionality of the soil. Soil quality relates to quantifiable natural properties that are inherent for a particular soil type; e.g., soil physical and chemical characteristics and historical soil-forming factors, which are fixed by nature. The USDA-NRCS defines soil health as "the capacity of the soil to function as a vital living ecosystem to sustain plants, animals, and humans" (USDA-NRCS 2021b). Six key soil physical and biological processes must function in a healthy soil (USDA-NRCS 2019):

(1) Organic matter dynamics and carbon sequestration
(2) Soil structural stability
(3) General microbial activity
(4) Carbon food source
(5) Bioavailable nitrogen
(6) Microbial community diversity

F. The United Nations states: "Healthy soils maintain a diverse community of soil organisms that help to control plant disease, insect and weed pests, form beneficial symbiotic associations with plant roots, recycle essential plant nutrients, improve soil structure with positive repercussions for soil water and nutrient holding capacity, and ultimately improve crop production" (FAO 2008).

G. The overall goal of conservation management practices for soil health are:

(1) Minimize soil disturbance and maintain natural soil quality factors
(2) Maximize soil cover
(3) Maximize biodiversity
(4) Maximize and enhance living roots—microorganisms are most active in and around plant roots, increase in diversity of beneficial microbia, and enhance the nutrient cycle in soils

H. In order to accomplish goals of improving soil health, an integrated approach is needed. For example, on cropland, cover crops, residue, and tillage management, reduced and no-till, and conservation crop rotation are some of the principal conservation management practices used in sustainable cropping systems and maintaining or building soil health. Likewise, on grazing lands, integrated approaches commonly include managed grazing, and the periodic use of prescribed burning, brush management, and herbaceous weed treatment, according to ecosystem dynamics and need. The objective of these practices is to maintain or restore plant communities. On native rangelands with representative levels of native species, maintaining these species is critical to biodiversity and long-term health and sustainability.

I. Significant correlations exist between livestock management (stocking rate), vegetation change, and soil function (Briske et al. 2011). As a consequence, other factors such as shrub invasion and increased woody species densities (Archer et al. 2001; Archer et al. 2011) and exotic plant species invasion (Sheley et al. 2011) are associated with land degrading processes. Inappropriate grazing management such as overutilization, unsustainable stocking rates, and continuous use are usually directly or indirectly linked to the initial stages of undesirable change. Although managing most rangeland systems involve inherent complexity, many soil and vegetation degrading factors can be linked in some fashion to poor grazing management (Brown and Herrick 2016). Early warning indicators of changes in range, pasture, and soil health are commonly associated with some aspect of livestock grazing management, which is expressed in vegetation changes (species composition, diversity, and production) (Briske et al. 2011). Fortuitously, improved managed grazing is frequently the most cost-effective means of avoiding soil health degradation or reversing a downward trend in soil health on rangelands (Brown and Herrick 2016).

J. Soils develop and evolve over long periods of time in conjunction with climate, geology, parent materials, vegetation, and local microbial and animal floras (Jenny 1941; 1961). “Soils—in the pedological [science that deals with formation, biotic and abiotic components and classification] sense—are born, mature, and die. They are born when sediments are deposited. They mature after sedimentation stops and soil horizons develop. And they die when erosion strips away the soil horizons and exposes underlying parent material” (USDA-NRCS 2016). Early Russian pedologists developed a metaphor called “soil memory.” Soils have a past history and so-called memory dictated by climate, vegetation, and land use. These properties are discernable by the properties and stratigraphy of the soil (scientific study of rock strata or layers, especially the distribution, deposition, correlation, and age of sedimentary rocks). All intact non-cultivated soils have a natural level of soil quality and health, which are both unique to that particular soil in its ecosystem setting.

K. Where soil organic matter may be naturally low (e.g., semiarid and arid environments), this property of soil quality is representative of that soil type, which was influenced by the historic soil-forming factors and the soil’s evolution over time. Naturally occurring pH of soils developing with forest vegetation are typically acid; however, a soil with high carbonates and limestone-derived parent material are typically alkaline. Both of these conditions are representative inherent soil quality factors and cannot be altered easily without high agricultural inputs. Natural levels of organic matter, pH, carbonates, sodium, sand, silt, clay, and soil structure constitute “natural soil quality.” Some of these physical and chemical soil attributes may not be conducive to agricultural or construction pursuits (or to one’s objectives). They represent the particular qualities of that soil. Because a natural soil may not have abundant organic matter, or a pH level that is desirable for a particular crop, or may be relatively low in essential plant nutrients, does not imply that the soil is “bad” or “unhealthy.” It just means that that soil may not meet specific needs or agricultural objectives; but regardless, it is healthy from a soil quality or naturalized point of view. The critical point regarding soil quality or health is that a reference condition is required for each soil type so that comparisons of existing plant and soil conditions can be made. Without a reference for each particular soil, soil quality or health determinations are not possible. This is an extremely important point: the soil component characteristics associated with the ecological site as defined in the ecological site description are the reference standard, as well as state changes and site thresholds caused by perturbations to the system.

645.1104 Soil Health Management on Rangeland and Pastureland

A. The management of rangeland ecosystems and subsequent effects on soil physical and chemical properties are dynamically different from cropland and other agricultural land uses, such as pastureland, that have a history of cultivation. Rangelands are largely managed as naturalized systems with minimal cultural inputs with multiple use objectives that may or may not include livestock grazing (see subpart A). Rangeland plant communities vary considerably across the continuum of rangeland ecosystems in the United States with respect to climate, geomorphology, soils, plant species composition and physiological traits, physical and chemical soil properties, and hydrology.

B. Pastoral systems that have a prior history of cultivation are typically seeded to adaptable forage species (introduced and native). Over time, any cultivated soil has a history of soil erosion and ultimately soil carbon losses. The dynamics of soil health on pasturelands can be significantly different than uncultivated rangelands, and recognition of these differences is critical to current and future management. Franzluebbers and Stuedemann (2013) state:

> Historic loss of soil organic C (SOC) with cultivation of land for crops has been dramatic. Estimates in different regions of the world range from 20 to 60 percent SOC loss when compared with native conditions (Giddens 1957, Mann 1986, Davidson and Ackerman 1993, VandenBygaart et al. 2003, Liebig et al. 2005, Ogle et al. 2005, Luo et al. 2010). In the Appalachian Piedmont region of the southeastern USA, topsoil loss from historic cultivation of a variety of crops, but primarily cotton (*Gossypium hirsutum* L.) and tobacco (*Nicotiana tabacum* L.) without sufficient conservation measures, has been estimated to be 20 cm (Trimble 1974).

C. Erosion is a natural process. Approximatively 80 percent of the world's land surface is vulnerable to geologic erosion (Thurow 1991). An extreme example in the United States is the Badlands National Park. Geologic erosion there has been active for the last 500,000 years (Stetler et al. 2011), and current average soil loss in the park is estimated at ~2.5 cm yr^{-1} (Stoffer 2003), approximately equivalent to 325 Mg ha^{-1} yr^{-1} (144.9 t ac^{-1} yr^{-1}). See exhibit K-1 below for examples of geologic soil loss (see subpart G for additional information on soil erosion).

Exhibit K-1. 1 mm of Soil Loss?

Several publications have provided estimates of average soil loss:

- 0.1 mm yr^{-1} for our most recent geologic epoch (Wilkinson and McElroy 2007)
- 0.021 mm yr^{-1} the average erosion rate at 21 meters per million years (m/m.y.) (Summerfield and Hulton 1994)
- Loess and glacial till areas of Iowa, United States, ~0.8–1.9 Mg ha^{-1} yr^{-1} (Ruhe and Daniels 1965, Walker 1966)
- On cultivated US croplands, average estimated erosion rates are about 600 m/m.y. (0.6 mm yr^{-1}) (USDA-NRCS 2000)

Equivalency of 1 mm of Soil Loss to Mg ha^{-1} and t ac^{-1}?

If 1 mm of soil erodes on a silt loam with a bulk density of 1.33 Mg m^3, what is the equivalent soil loss in Mg ha^{-1} and US tons ac^{-1}?

- For 1 mm soil depth = (1 m) (1 m) (0.001 m) = 0.001 m^3
- Weight of soil 1 m^2 at 1 mm depth = (0.001 m^3) (Bulk density 1.33 Mg m^3) = 0.0013 Mg m^3
- Weight of soil for 1 hectare at 1 mm depth = (100 m) (100 m) (0.0013 Mg m^3) = 13 Mg ha^{-1} (5.79 t ac^{-1})

D. For example, Bakker et al. (2004) find that crop productivity decreases 4.3–10.9 percent per 10 cm of soil loss. Soil erodibility is multivariate in nature, and many factors are involved such as intensity, duration, and amount of precipitation; topography – land slope and shape (linear, convex, con- cave) – types and amounts of vegetation; organic matter content and associated aggregate stability; soil particle size (texture); and nature of parent material (Butzer 1974).

E. In many natural terrestrial plant communities, a relative balance exists between weathering and erosion:

$$E = W + S$$

where E = erosion by runoff and mass movement, W = rate of weathering and soil formation, and S = soil wash and other incoming upslope colluvium (Bunting 1965). The soil building process continues where $E < W + S$; examples may be where convex slopes increase until an equilibrium is reached. Where ecological balance is disrupted, $E > W + S$. Erosion continues until a new equilibrium is reached or erosion continues to bare rock or other restrictive layer.

F. In the Great Plains of the United States, examples of organic matter depletion as a result of intense cultivation of native grasslands by pioneer farmers have reduced soil productivity by 71 percent over a 28-year period (Flach et al. 1997). The worst modern erosion event was the Dust Bowl of the Southern Great Plains of North America (western Kansas, southeastern Colorado, Oklahoma Panhandle, the northern two-thirds of the Texas Panhandle, and Northeastern New Mexico). Unfortunately, by the end of the Dust Bowl, many areas in the Midwestern Great Plains had cumulatively lost more than 75 percent of its topsoil from wind and water erosion (Hornbeck 2009). Many authors report that once cultivation begins by breaking up undisturbed native grassland and forest soils, about 20–50 percent of the original soil organic carbon is lost within 40–50 years (Campbell and Souster 1982; Mann 1986; Schimel 1995; Donigian et al. 1994; Rasmussen and Parton 1994; Houghton 1995) (figures K-3, K-4). After the decline of soil organic carbon, further changes are largely a function of active erosion and subsequent soil management (Rasmussen and Collins 1991).

Figure K-3. (a) Decline of SOM after cultivation in native tallgrass prairie with high historic organic matter content (7.5 percent). Graph shows progressive loss of organic matter in the upper 25.4 cm (10 in) after cultivation. If a field is seeded back to native grasses after 100 years, organic matter would slowly increase. **(b)** Development of organic matter over time, rate of accumulation is progressive until a constant level is reached—in equilibrium with climate, soil properties, plant community, and decomposition processes. (Spaeth 2020.)

(a) **(b)**

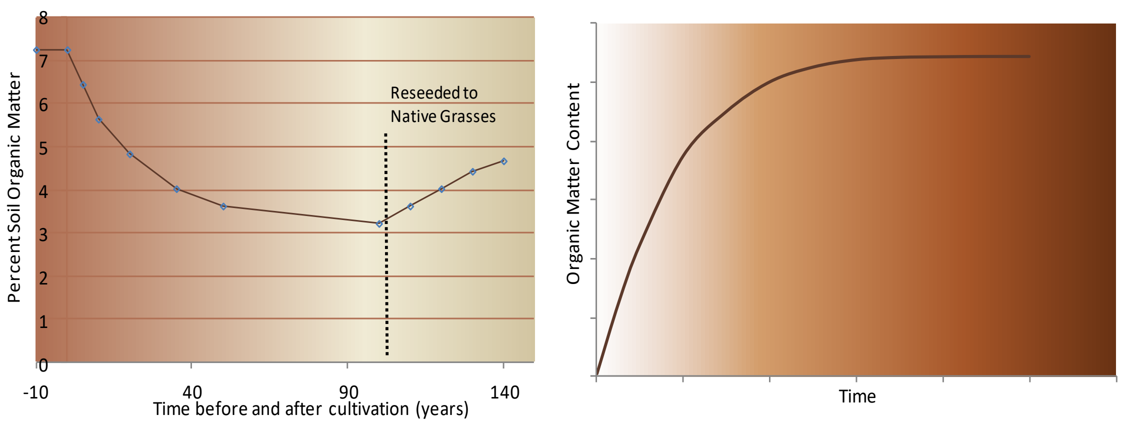


Figure K-4. Simulation of soil carbon changes from 1907 to 1990 for corn belt in central U.S. (Spaeth 2020, adapted from Donigian et al. 1994.)

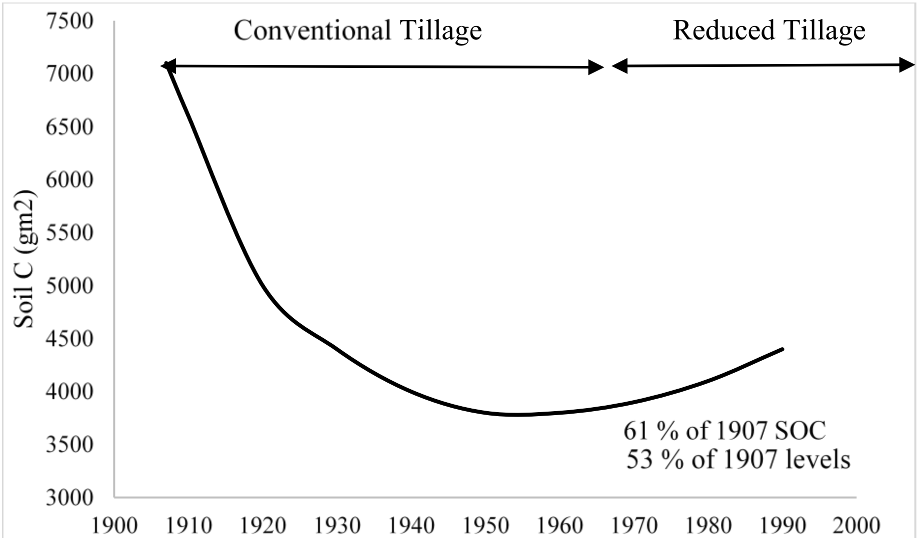


Exhibit K-2. (Spaeth 2020).

Ten Differences Between Rangeland and Cropland Factors that Influence Soil Health

Note: existing rangelands are typically not suitable to cultivation or have been re-established because of high erosivity and low crop potential.

- In contrast to croplands, which on the average either receive adequate rainfed precipitation or are irrigated, many arid and semiarid rangelands are characteristic of high inter- and intra-annual variability in timing and amount of precipitation, which is the key factor that drives ecological dynamics including hydrologic function, soil and site stability, biotic integrity, and response to management. Microbial populations that are active with carbon cycle “wax and wane” with available soil moisture content; therefore, many cropland soil health indicators are not applicable to rangeland. Rangeland and cropland are unique in their capacities, and evaluations of soil health should be based on their particular potentials and dynamics.
- Existing undisturbed rangeland soils are highly variable, ranging from shallow to very deep. Many highly productive croplands were highly productive rangelands (grasslands) with lower tolerable soil loss rates than cropland, especially on fragile arid and semi-arid rangelands. Much of the remaining uncultivated Great Plains prairies occur on soils with soil properties (shallow, steep, droughty, gravely and rocky, wet, saline, and high pH) that are unsuitable for crop production.
- Many rangeland soils in the southwest and western rangelands have lower soil organic matter content; however, the true prairie soils of the Great Plains with higher organic matter have greater soil organic carbon percentages for maintenance of sequestered carbon. Prairie grassland soils that were suitable for cultivation are now some of the most productive cropland soils.
- Soil organic carbon sequestration in arid and semiarid rangeland environments can be substantial because of carbon stocks from perennial root systems and large biomass turnover of above-ground plant litter. Rangeland ecosites with perennial shrubs and trees have substantial carbon storage above and below ground. Forests (1.2–1.4 Pg C yr^{-1}) and cropland (0.4–1.2 Pg C yr^{-1}) have the largest potentials for sequestering carbon, although grazing lands (range and pasturelands) can contribute up to 10 percent of the carbon sink capacity. On a global perspective, rangelands occupy about half of the world’s land area, 10 percent of the terrestrial biomass, and 10–-30 percent of the

soil organic carbon (Schlesinger 1997). An average estimate of globally sequestered soil carbon on rangelands is 0.5 Pg C yr^{-1} (Schlesinger 1997; Scurlock and Hall 1998).

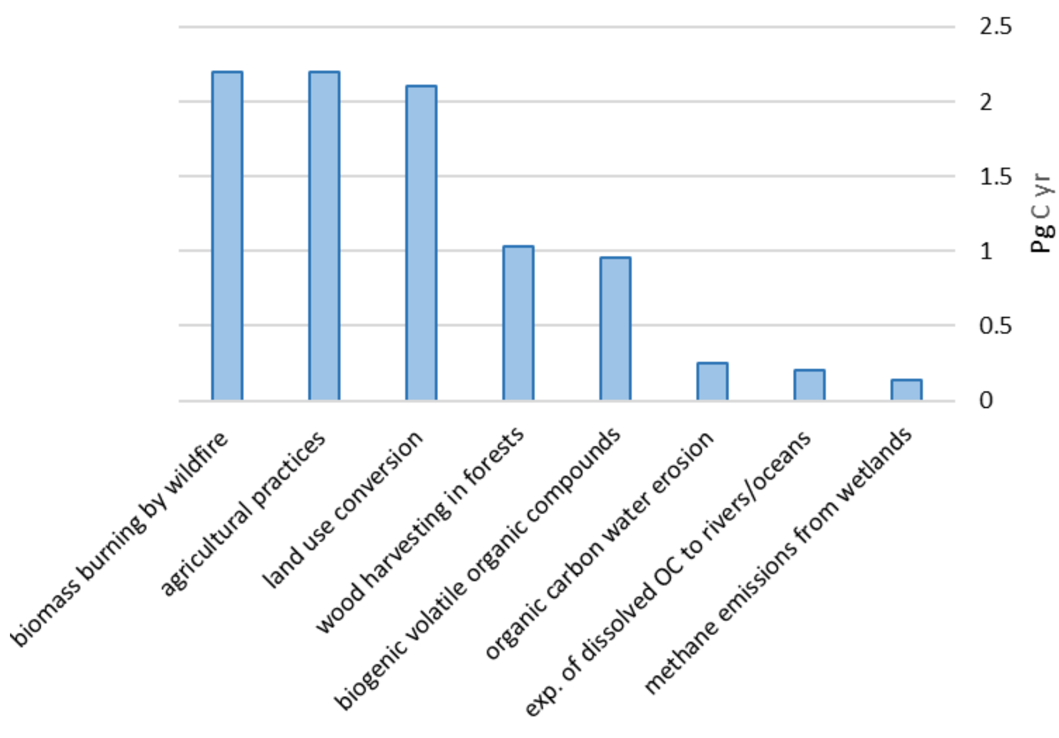
- Rangelands include many different types of vegetation life forms (trees, shrubs, forbs, grasses, and biotic crusts) and growth forms (bunch, sod, vine, stoloniferous, rhizomatous, tap and fibrous root systems) compared to monocultures or limited plant diversity in cropland settings. Carbon cycles, turnover, and sequestration comparisons between perennial rangeland vegetation and cropland are significantly different due to rooting depth dynamics and soil surface cultivation.
- Perennial vegetation on rangelands is typically associated with less soil disturbance than cultivated croplands (exceptions are no-till and reduced tillage). In heavier-textured rangeland soils, soil aggregates are maintained with more stable soil organic matter pools and microbial populations. Rangeland soil structure is typically not altered to the extent that may exist in recurring tillage operations with cropland agriculture. Also, cropland soils often have a plow layer or plowpan layer that persists after cultivation and may inhibit infiltration capacity and root development of plants.
- Soil heterogeneity is often more prevalent on existing rangeland because of geomorphic topographic influences that inhibit cultivation. This heterogeneity promotes plant species diversity and associated microbial mycorrhizal associations.
- Erosion control on rangeland is highly dependent on maintaining living plant foliar cover (> 60 percent cover), litter, and health root systems; whereas, on cropland, plant cover usually consists of annual plants that are harvested with various levels of remaining litter. On crop fields, residue management, crop and tillage management (reduced tillage and no-till, cover crops, strip cropping) and structural practices (terraces, waterways, diversions) are commonly used to manage runoff and erosion.
- Natural biological soil crusts can be prevalent in rangeland plant communities, especially in semiarid and arid shrub ecosites with plant interspaces. They provide soil surface stability against wind and water erosion and are active in nutrient cycling.
- Arid and semiarid rangelands have slower decomposition and carbon turnover rates because of more erratic climate and precipitation compared to temperate grasslands, savannahs, and shrublands. Cropland environments (rainfed and irrigated) supply more consistent moisture that is necessary to maintain microbial populations associated with nutrient cycling, decomposition, and organic matter turnover. Management associated with soil health and soil organic matter restoration may not be possible in many arid and semiarid rangeland plant communities due to inherent climate restrictions, original soil carbon dynamics, wind and water erosion, and changes from perennial to annual plant dominance. Rangelands dominated by annual exotic grasses and forbs are prone to increased wildfire and subsequent wind and water erosion, which creates an unstable environment to maintaining soil organic carbon levels. If major depreciating vegetative state changes have crossed critical thresholds, they may be permanent and irreversible. On croplands that still have good production potential (with adequate rainfall and/or irrigation), enhancement of soil organic matter and microbial mycorrhizae interactions are significantly more probable with the use of reduced and no-till practices and cover crops.

645.1105 Carbon Budgets and Balance in Terrestrial Ecosystems

A. Carbon is an essential element of all Earth's systems and the building block of all living organisms, a key foundational element of soil health. Carbon is fundamental for life because of its ability to combine with other important elements, such as oxygen, nitrogen, and phosphorus, and hydrogen to form organic molecules that are essential for cellular metabolism and reproduction (Bruhwiler et al. 2018). The carbon cycle is central to plant-soil interactions and ultimately soil health. On a weight basis, carbon ranks 19th in order of elemental abundance in Earth's crust (about 0.025 percent of the Earth's crust); however, carbon is a component of more compounds than all the other elements combined. Carbon is found throughout nature in vastly different forms: graphite is very soft, while diamond is one of the hardest substances found in nature. Chemically, carbon is a nonmetal and is a component of over 10 million different compounds, thousands of which are vital to life processes. When carbon combines with oxygen in differing amounts, hydrocarbons are formed (coal, petroleum, and natural gas). All organically derived substances contain the element carbon, and in soils, organic matter is about 58 percent carbon.

B. Atmospheric carbon represented as carbon dioxide (CO_2) and methane (CH_4) helps regulate the Earth's climate by trapping heat in the atmosphere (the greenhouse effect), and these and other greenhouse gases such as water vapor and nitrous oxide (N_2O) help maintain a habitable equilibrium (Ito 2019). The global carbon budget of terrestrial ecosystems is determined mainly by major flows of carbon dioxide (CO_2) from photosynthesis and respiration; however, various minor flows also influence global carbon stocks and their turnover (Ito 2019) (figures K-5, K-6). The carbon cycle and temporal fluxes (natural and anthropogenic) are the foundation to understanding the dynamics of how carbon is processed and compartmentalized into various pools and reservoirs. The carbon cycle involves four basic carbon reservoirs: the atmosphere, oceans, terrestrial biosphere, and fossil carbon. As seen in figure K-6, natural pools and rates from heterotrophs are much greater than anthropogenic-derived carbon from burning fossil fuels and other activities, and rising CO_2 in the atmosphere is due to an imbalance between anthropogenic production and CO_2 consumption (Kirchman 2010). The global estimate of burning of fossil fuels is 4–10 Pg yr^{-1}, and open burning produces 2–3 Pg C yr^{-1}. (Wiedinmyer and Neff 2007); however, soil organic carbon losses from erosion are estimated at 5.7 Pg C yr^{-1}, of which about 4 Pg C yr^{-1} is redistributed elsewhere on the landscape, 1.14 Pg C yr^{-1} is emitted to the atmosphere, and 0.57 Pg C yr^{-1} is transferred to the oceans (Lal 2003, 2008, 2010, 2018).

Figure K-5. Ranking of minor carbon flows associated with land use change (data from Ito 2019).



C. Ito (2019) identified eight minor carbon flows associated with land-use change (figure K-5):

(1) Biomass burning by wildfire
(2) Emission of biogenic volatile organic compounds from plants
(3) Methane emissions from wetlands
(4) Methane oxidation in uplands

(5) Agricultural practices from cropping to harvesting
(6) Wood harvesting in forests
(7) Export of dissolved organic carbon by rivers to oceans
(8) Displacement and transport of soil particulate organic carbon water erosion

Figure K-6. The global carbon cycle with representative carbon pools and reservoirs that interact with Earth's atmosphere (units next to reservoir names are Pg C. Note: Pg = 10^{15} g; 10^6 g Mg megagram (ton); 10^{12} g Tg teragram; 10^{15} g Pg petagram). The soil reservoir contains about as much of the carbon reservoir as plants and the atmosphere combined. Carbon imbalance exists from the flow of burning fossil fuels, open burning (fire); and more carbon is emitted from the soil (62 Pg) than entering the soil (59–60 Pg) due to erosion and loss of organic matter. Land use changes have been estimated to produce 1.6–2 Pg C yr^{-1}, and the largest carbon reserve is in carbonate rocks (75 million Pg). Estimates are slightly different among authors (adaptations and data from Berner 1990; Schimel 1995; Batjes 1996; Falkowski et al. 2000; Pacala and Socolow 2004; Houghton 2007; Solomon 2007; Battin et al. 2009; Haddix et al. 2011; Pan et al. 2011; Lal 2018) (adapted from Spaeth 2020).

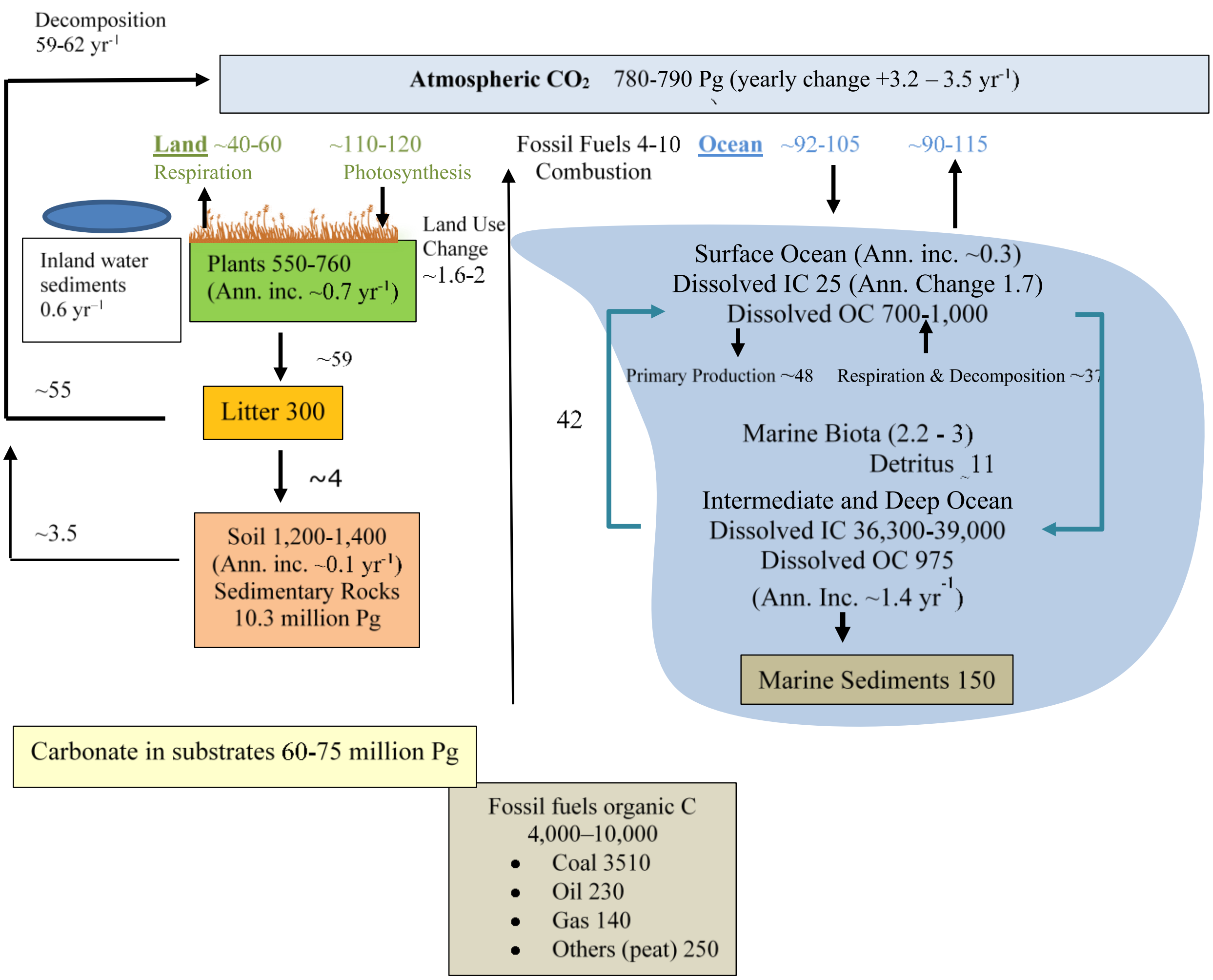
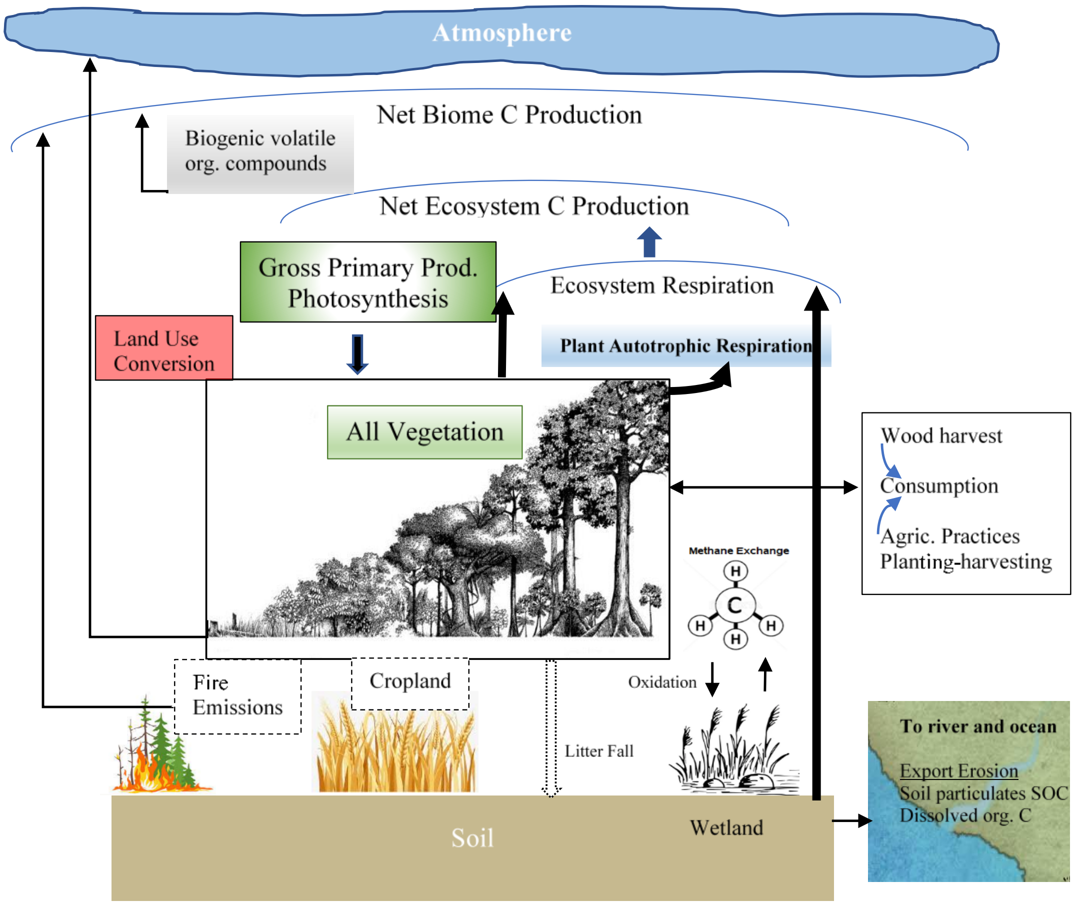


Figure K-7. Carbon budget with major and minor carbon flows. Thick lines=major C flows, thin lines=minor carbon flows.



645.1106 Soil Organic Matter

A. Unfortunately, there are many exaggerated and false narratives in the popular literature regarding organic matter (Spaeth 2020). Here we review some basic information with quotes relevant to soil organic matter. The major components of the soil environment are comprised of minerals, air, water, and living and dead organic matter (figure K-8). The amounts of each varies with soil type and management, but in general minerals comprise 45 percent, air and water 25 percent, and 1–5 percent is living or dead organic matter. Air and water change with soil moisture levels, while organic matter varies with soil type, climate, plant community, and management practices. Intrinsic physical, chemical, and biological properties in soils provide a basis for soil quality, while management affects plant growth, productivity, and soil carbon dynamics (figures K-7, K-9).

B. Soil organic matter represents the smallest fraction in soil but is one of the most important constituents as it is directly related to agricultural production. Organic matter is ~58 percent carbon and is derived from contributions of decomposed plant residues from leaf litter, crop residues, plant roots, root exudates, animals and feces, and decomposing remains of soil fauna and microbes. Carbon fixation from the atmosphere occurs through photosynthesis, and carbon content in the soil is dependent upon inputs from plants and microbes[1].

Figure K-8. Components of organic matter.

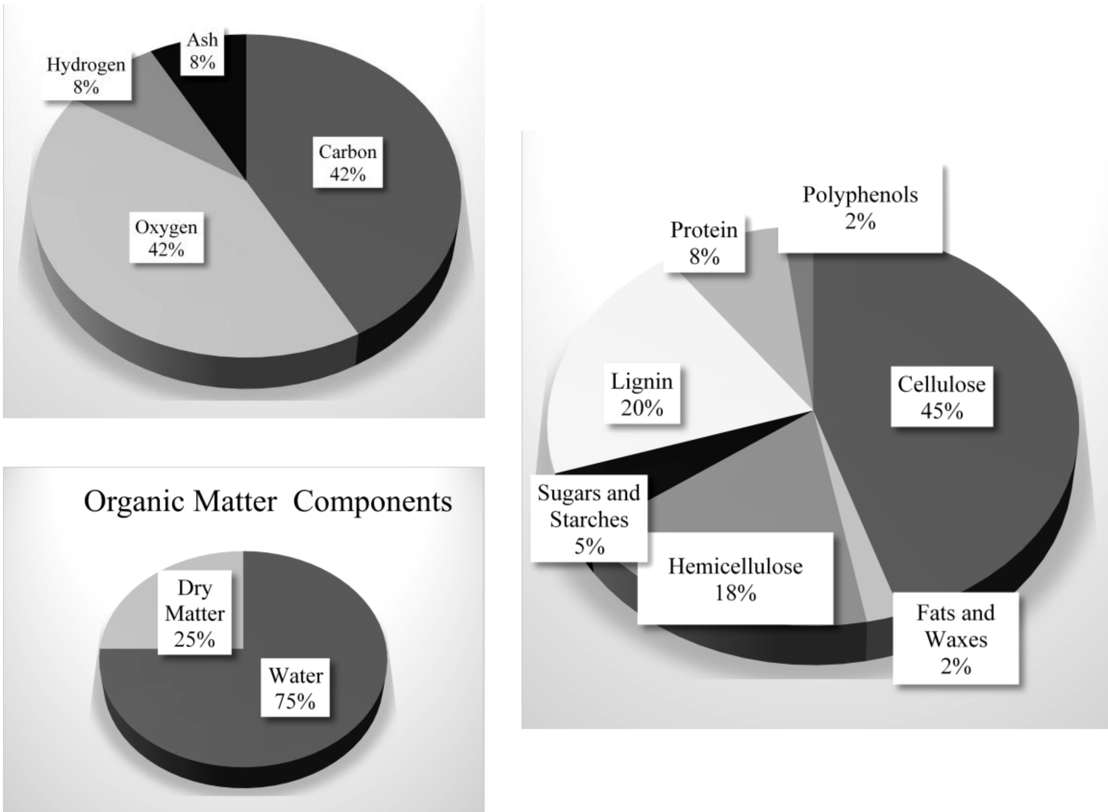


Figure K-9. Physical, chemical, and biologic properties associated with overall health of a plant community (adapted from Stirling et al. 2016).

Inherent Components of Soil	Plant and Management Practices	Soil and Plant Community Properties	
Minerals Soil Organisms Organic Matter Water Air	Tillage Fertilizer Inputs Soil Amendments Pesticides Residue Management Grazing Trafficking	Water Availability Acidity/Alkalinity Nutrient Availability Aeration Soil Borne Pathogens Weed Competition Compaction/Crusting Biodiversity	**Plant Growth and Productivity**

[1] Carbon dioxide (CO_2), the most common oxide of carbon, is a trace gas [also classified as a greenhouse gas; 0.04 percent (410 ppm and increasing by 0.5 percent per year) by volume compared to 280 ppm before the onset of the industrial revolution].

Exhibit K-3. Importance of Soil Organic Matter: Quotes (Spaeth 2020)

- “Soil organic matter is the main reservoir of SOC and soil organic nitrogen in rangelands and determines soil fertility, water retention, and soil structure” (Pineiro 2010).
- “Humus plays a leading part in the storage of energy of solar origin on the surface of the earth” (Waksman 1936).
- “Soil plays an essential role in the global carbon (C) cycle acting as both a source and sink of organic C . . . Soil contains three times more organic C than is contained in plants and the atmosphere, and while soil is classically viewed as a C sink on a global scale there is concern that climate and land use change will turn it into a C source” (Caruso et al. 2018).
- “When grassland or forest ecosystems are first converted to agriculture, multiple mechanisms result in SOM declines of between 20 percent and 70 percent. Two of the most important mechanisms are the reduction in organic matter inputs from roots following the replacement of perennial vegetation with annual crop species, and increases in microbial respiration when tillage breaks open soil aggregates exposing previously protected organic matter . . . Two of the overarching reasons why native terrestrial ecosystems have achieved greater soil organic matter levels than human agroecosystems are because they direct a greater percentage of productivity belowground in perennial roots, and they do not require frequent soil disturbance. A growing body of research including that presented in this review suggests that developing perennial grain agroecosystems may hold the greatest promise for agriculture to approach the SOM levels that accumulate in native ecosystems” (Crews and Rumsey 2017).
- “Carbon accumulation in temperate grasslands occurs mostly below ground. Past and current land use changes (e.g., conversion of arable land to grassland) as well as the agricultural management of grasslands affect the below-ground carbon stocks” (Soussana et al. 2004).
- “The amount of organic matter in soil depends on the input of organic material, its rate of decomposition, the rate at which existing soil organic matter is mineralized, soil texture, and climate. All four factors interact so that the amount of soil organic matter changes, often slowly, toward an equilibrium value specific to the soil type and farming system. For any one cropping system, the equilibrium level of soil organic matter in a clay soil will be larger than that in a sandy soil, and for any one soil type the value will be larger with permanent grass than with continuous arable cropping” (Johnston et al. 2009).
- “In rangeland ecosystems, above ground biomass provides carbon inputs to the soil over long periods of time. Rhizodeposition of below-ground carbon is important for carbon sequestration as carbon residence times are longer than above-ground biomass” (Cougnon et al. 2017; Sainju et al. 2017).
- “Grassland plant species allocate more below-ground carbon than cereal crops” (Pausch and Kuzyakov 2018).
- “Centuries before there was any science that acquainted people with the intricacies of plant nutrition, decaying organic matter, as in manure or other forms, was recognized as an effective agent in the nourishment of plants . . . Soil organic matter is one of our most important national resources: its unwise exploitation has been devastating, and it must be given its proper rank in any conservation policy as one of the major factors affecting the levels of crop production in the future” (Albrecht 1938).
- “Soil organic matter is a natural product resulting from microbial activity in the inorganic and/or organic soil environment. The amount and accumulation of soil organic matter are controlled by the composition and amounts of the plant residues, by climatic conditions, and soil texture. Other important factors are microbial activity, soil redox conditions, and other soil chemical and physical properties” (Haider and Guggenberger 2005).
- “In most soils, the percentage of soil organic matter is small, but its effects on soil function are profound. This ever-changing soil component exerts a dominant influence on many soil physical, chemical, and biological properties and ecosystem functions of soil” (Weil and Brady 2017).

- "Grassland soils can sequester more than 100 Mg ha^{-1} (40.5 t acre) of soil organic carbon and 10 MG ha^{-1} (4.05 t acre) of soil organic nitrogen in a surface meter of soil" (Milchunas and Lauenroth 1993; Derner et al. 2006; Pineiro et al. 2009).
- "The value of soil organic matter is more than just its ability to hold water and nutrients for plant growth. Its hidden value lies in its ability to regulate the environment, especially to mitigate the greenhouse effect" (Lal et al. 1998).
- "A fertile and healthy soil is the basis for healthy plants, animals, and humans. And SOM is the very foundation for healthy and productive soils. Understanding the role of organic matter in maintaining a healthy soil is essential for developing ecologically sound agricultural practices" (SARE 2012).
- "The value of SOM has long been recognized. From earliest times, its level in the soil was used as a general indicator of soil productivity . . . A major factor contributing to the level of SOM is annual input of plant residues . . . Under cultivated agriculture, crop residues serve as carbon inputs and thus influence both the level and the dynamics of organic matter in the soil" (Buyanovsky and Wagner 1997).
- "Soil organic matter is where soil carbon is stored and is directly derived from biomass of microbial communities in the soil (bacterial, fungal, and protozoan), as well as from plant roots and detritus, and biomass-containing amendments like manure, green manures, mulches, composts, and crop residues . . . OM in its various forms greatly impacts the physical, biological, and chemical properties of the soil. OM acts as a long-term sink, and as a slow-release pool for nutrients. It contributes to ion exchange capacity (nutrient storage), nutrient cycling, soil aggregation, and water holding capacity, and it provides nutrients and energy to the plant and soil microbial communities" (Moebius-Clune et al. 2016).
- "The weathering of minerals is an essential function in the development of soils. The second process of great significance to soil formation is the accumulation of organic matter, which tends toward an equilibrium level in well-developed soils. The level to which organic matter accumulates in the soil depends on the nature of the soil forming environment as it affects two opposing processes, namely: the addition of residues, principally from plants, and the decay of these residues by microbes and other soil-inhabiting organisms" (Hausenbuiller 1978).
- "Soil organic carbon is a complex of organic carbon compounds in the form of SOM. Soil organic matter includes everything in or on the soil that is of biological origin, irrespective of origin or state of decomposition (Baldock and Skjemstad 1999). It includes plant and animal remains in various states of decomposition, cells and tissues of soil organisms, and substances from plant roots and soil microbes. The ultimate product of the decomposition process is humus, an amorphous array of compounds highly resistant to further decomposition. Many organic compounds in the soil are intimately associated with inorganic soil particles. In agricultural soils, soil organic carbon content is usually less than 5 percent and decreases with soil depth (Baldock and Skjemstad 1999)" (citation from Xiao 2017).
- "Plant residues are the major source of carbon inputs in all terrestrial ecosystems" (Paustian et al. 1997).
- "Organic matter affects both the chemical and physical properties of the soil and its overall health. Properties influenced by organic matter include soil structure; moisture holding capacity; diversity and activity of soil organisms, both those that are beneficial and harmful to crop production; and nutrient availability. It also influences the effects of chemical amendments, fertilizers, pesticides, and herbicides" (Bot and Benites 2005).
- "Organic residues are carbon-containing compounds of biological origin. Decomposition is the breakdown of these complex organic materials into simpler components" (Franzluebbers 2005).

645.1107 Organic Matter Losses with the Advent of Agriculture and Land Use Practices

A. The terrestrial biosphere stores about two-thirds of the global carbon in the surface meter of the soil. There are five global carbon pools (Pg is a unit of mass equal to 1,000,000,000,000,000 (10^{15}) grams; 1Pg= 1,000,000,000 metric tons) (Berner 1990; Schimel 1995; Batjes 1996; Falkowski et al. 2000; Pacala and Socolow 2004; Houghton 2007; Canadell et al. 2007a, b; IPCC 2007; Battin et al. 2009; Haddix et al. 2011; Pan et al. 2011; Lal 2018):

(1) Oceanic 36,000–39,000 Pg, increasing at 2.3 Pg C yr-^{1}
(2) Fossil fuels 4,000–10,000 Pg, mined and combusted 8 Pg C yr^{-1}
(3) Pedologic 2,500 Pg @ 1 m soil depth
  (i) 1,500–1,550 Pg Soil organic carbon
  (ii) 950 Pg Soil inorganic carbon
(4) Atmospheric 780–790 Pg, increasing at 3.2–4 Pg C yr^{-1}
(5) Biotic pool 550–760 Pg live biomass
  (i) 60 Pg detritus material
  (ii) 300 Pg litter

B. Historical Record of Soil Carbon Loss

(1) Assuming that organized agriculture began about 10,000 years ago (Ruddiman 2003), more than 320 Pg carbon has been depleted by anthropogenic activities such as deforestation, biomass burning, cultivation, and drainage of peatlands. Of this amount, about 78±12 Pg of carbon has been lost from the soil (Lal 2010). Global conversion of 1.136 billion hectares (Bha) of forest and woodlands and 0.669 Bha of savannah and grasslands have occurred between 1700 and 1990 (Foley 2005). Since the 1850s, estimates between 124 and 158 Pg of soil carbon have been depleted by various land-use conversions (Canadell et al. 2007b). Lewis (2005) estimates that between the years 1750–2000, about 180 Pg carbon have been released to the atmosphere, 60 percent originating from the tropics.

(2) On a global scale, intensive cultivation has resulted in a soil carbon loss of about 55 Pg carbon – 25 percent of the original historic soil organic carbon levels (Cole et al. 1997; Lal et al. 1998; Six et al. 2006). The cumulative loss of soil carbon from cultivated soils (55 Pg) accounts for about seven percent of the current carbon in the atmosphere (Lal et al. 1998).

(3) Plant communities containing specific plant life and growth forms display inherent productivity levels, both above and below- ground, and therefore have different capabilities concerning inherent and potential sequestration of soil organic carbon. Plant productivity and subsequent carbon balance in plant communities are dependent upon temperature, timing, and amount of precipitation, both of which enhance or constrain the microbial process of decomposition. In arid plant communities, productivity and decomposition of organic matter are constrained and tenuous because of low annual precipitation, higher summer temperatures, and high occurrence of drought, wherein subhumid ecosystems, plant growth, and decomposition are more constant. Soil texture also plays an integral role in baseline carbon contents. Jobbagy et al. (2000) found that precipitation and climate were the best predictors of soil organic carbon in the upper 20 cm soil layer; however, clay content was the best predictor in deeper soil layers. Higher clay content may be responsible for slower soil organic carbon cycling due to clay particles and noncrystalline minerals that act to stabilize organic matter (Trumbore 2000). The soil orders Histosols, Mollisols, and Vertisols contain the highest soil organic carbon contents on rangelands (figure K-10); however, from a global perspective they do not comprise the highest global carbon percentages in the upper 1 m of soil (table K-3).

Figure K-10. Average SOM content for selected soil orders on rangeland (USDA-NRCS soil data).

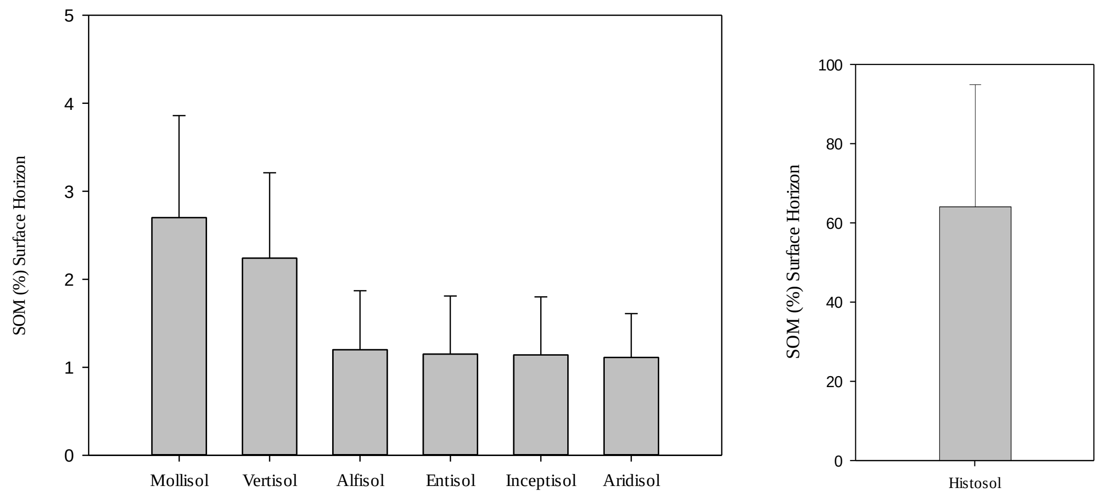


Table K-3. Organic and inorganic carbon mass in upper one meter of global soils. Note: inorganic matter is largely calcium carbonate in soils of arid regions. About 60–90 percent of carbon is in the upper one meter of the soil profile, although stored carbon is also significant below one meter in Histosols and Gelisols. (Data from Eswaran et al. 2000; Weil and Brady 2017.)

Soil Order	Global Area 10^3 km^2	Global carbon in upper 1 m (Pg)			Total C % of global soil
		Organic	Inorganic	Total	
Aridisols	15,699	59	456	515	20.6
Entisols	21,137	90	263	353	14.2
Gelisols	11,260	316	7	323	12.9
Mollisols	9,005	121	116	237	9.5
Inceptisols	12,863	190	34	224	9.0
Alfisols	12,620	158	43	201	8.0
Histosols	1,526	179	0	180	7.2
Ultisols	11,052	137	0	137	5.5

645.1108 Carbon Geochemical Cycle

A. Terrestrial ecosystems are a complex of biotic and abiotic components with various cycles that are active in all environments. Cropland settings, forest, rangeland, pastureland, and gardens all have particular features concerning biomass production, plant decay rates, soil-microorganism dynamics, and climatic effects (water and temperature are principal drivers of productivity in all plant communities). Organic matter gains and losses in the soil are dependent upon gains and losses of carbon. There are five biogeochemical cycles: carbon cycle, nitrogen cycle, oxygen cycle, phosphorus cycle, and the water cycle (see subpart G) that represent the flow of chemical elements between living organisms and the environment. Here we focus on the carbon cycle as it is an integral link to plant productivity on all land uses. The carbon cycle is of primary importance to soil health and is closely tied to energy flow and all the other cycles via living organisms. The carbon cycle involves the assimilation of CO_2 by plants and animals and microbial tissues, all of which release O_2 by respiration.

B. Essentially, soil organic carbon originates from atmospheric CO_2, processed by plants during photosynthesis. Herbivores and secondary animal predators process this carbon in the food cycle, which in part is ultimately returned to the soil via microbial decomposition. The amount of soil carbon that soil can eventually accumulate is a balance of carbon inputs and carbon of organic material. Soil organic matter content is dependent upon climate and climatic conditions, soil type, parent materials, physiographic influences, and how the land is used (wildlands or domestically grazed). Decomposition involves both abiotic and biotic processes. Abiotic processes include weathering, freeze/thaw cycles, alternating drying and wetting, and UV photooxidation. Microorganisms decompose soil organic matter by cellular metabolism and extracellular excretions of exoenzymes. Although microbial biomass represents a small fraction (1–5 percent) of the total carbon, nitrogen, phosphorus, and sulfur pools, microorganisms are necessary for the decomposition process to proceed (Balota et al. 2003). Bacteria and fungi provide more than 95 percent of the biotic contribution to organic matter decomposition (Persson et al. 1980). Since microorganisms are living components and have high surface-to-volume ratios, they respond more quickly to changes in the soil environment than chemical alteration of soil organic matter and other soil physical and chemical properties. Microbial function and action is a key factor in diagnosing and monitoring soil health.

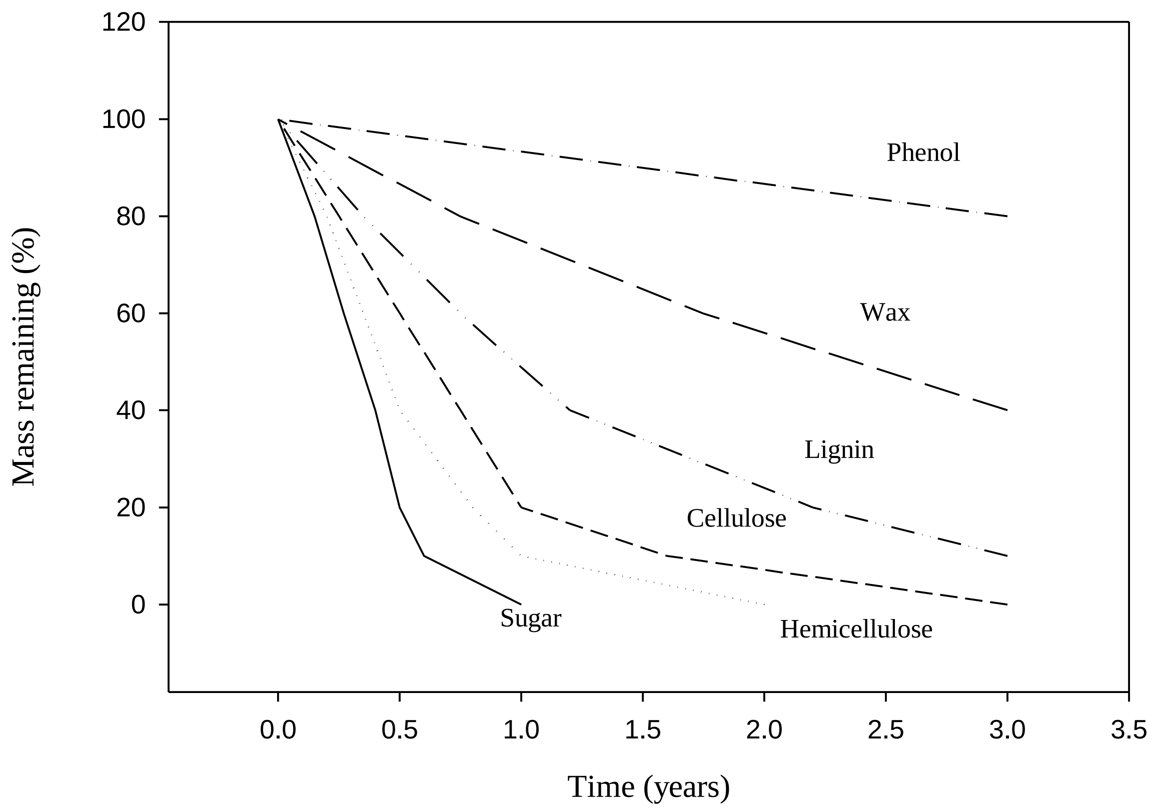
C. Decomposition via microbial action begins on standing live vegetation as different fungal species colonize different parts of plants. When moisture is adequate, fungi can colonize the culms of grasses. Some fungal species inhabit various internode and culm distances from the ground (microenvironments are more humid near the ground). Nutritional differences of the upper and lower internodes may also influence fungal flora (Hudson and Webster 1958). Fungi also can infect above-ground pine needles five to six months before needle fall (Burges 1963). In deciduous trees, leaf miners can damage the palisade layers of leaves, which instigates microbial attack of attached leaves. Most of the decomposition phase occurs when organic material is in contact with the soil.

D. A myriad of organisms (insects, earthworms, mites, millipedes, centipedes – detritivores and saprophages, etc.) fragment the debris, open plant cuticles, exposing parenchyma cells to microbial invasion, and consume raw material, which through excreta, is again acted upon by microorganisms. The physical action of these macro feeders can expose leaf area 15 times the original leaf size (Ghilarov 1970). This initial stage of decomposition by the macro feeders assimilates about 10 percent of the plant debris (mostly easily digested proteins and carbohydrates), thus allowing most of the material to pass through the gut. Plant debris, once in contact with the soil, is acted upon by bacteria, fungi, yeasts, and actinomycetes that feed on the material at rates determined by temperature and moisture. Decomposition is accelerated at high temperatures (35°C; 95°F), provided that sufficient moisture and oxygen are available for decomposition. Loceya and Lennona (2016) report that the Earth could contain nearly one trillion microbial species, only one-thousandth of one percent which are now identified. Oxygen can be a limiting factor in decomposition, as it is limited in oxygen-poor environments (Sierra et al. 2017). Figure K-8 shows some of the basic components of organic matter: dry matter-water content, elemental composition, and biologically derived components. These components in organic matter break down at different rates. Fungal mycelium and non-spore-forming bacteria quickly utilize carbohydrates (sugars) and simple proteins in the organic material. The composition and breakdown of organic compounds vary, from rapid to slow rates (figure K-11):

(1) Sugars, and simple proteins
(2) More complex proteins
(3) Hemicellulose
(4) cellulose
(5) Lignin and fats

E. The portion of organic matter that is not mineralized is humus, the stable fraction of organic matter. There are four basic pools of soil organic matter, each with variable turnover times: plant residues, particulate organic carbon, humus carbon, and recalcitrant organic carbon (figure K-13). These organic matter pools vary in chemical composition, decomposition rates, and carbon and nutrient cycles. The response of these carbon pools to management is critical to soil function and health. The labile pool which turns over relatively rapidly (weeks to < 5 years), results from the addition of fresh residues such as plant roots and living organisms, while resistant residues, which are physically or chemically protected, are slower to turn over (20–40 years). The protected humus and charcoal components make up the stable soil organic pool, which can take hundreds to thousands of years to turn over.

Figure K-11. Theoretical rates of decomposition for individual components of plant cells (Spaeth 2020 – source adapted from Minderman 1968, Franzluebbers 2005).



F. Bacteria and fungi provide more than 95 percent of the biotic contribution to organic matter decomposition (Persson et al. 1980). Bacteria and fungi are short lived and are consumed by other microbial organisms as they expire. As this process continues, nutrients are immobilized in microbial tissue, and upon death, nutrients are released, or mineralized-nutrients (nitrogen, phosphorous, sulfur, etc.) are again available for use by microbes and primary producers[2]. As bacteria decompose plant residues, they use low-molecular-weight compounds from plant biomass (nucleic acids, lipids, proteins, and carbohydrates) to develop their own biomass (Miltner and Bombach 2012). Enzymatic biochemical processes of soil microorganisms reduce decomposed organic matter into mineral compounds that may be utilized by plant roots. One important point that is often not recognized is that, as plant residues are being decomposed by microorganisms in a yearly cycle and become part of

[2] Primary producers in the ecosystem are organisms that produce biomass from inorganic compounds (autotrophs). Autotrophs are photosynthetically active organisms (plants, certain algae, and photosynthetic bacteria, cyanobacteria, and other unicellular organisms).

the soil, a considerable amount of the original carbon (65–85 percent of what was part of plant litter and animal detritus) is released back into the atmosphere as CO_2 via respiration (figure K-12). Only about 15–35 percent of this remaining carbon may remain in the soil as either live biomass (~2–5 percent), non-living labile carbon compounds (~3–10 percent), and stabilized humus (~10–30 percent) (Weil and Brady 2017).

Figure K-12. Carbon dioxide respiration as a consequence of the breakdown of carbon-based organic residues. Example of an annual cycle where organic matter is incorporated into the soil and 65–85 percent of the biomass is released as CO_2. Less than one-third of the original carbon remains in the soil as microbial biomass, non-living labile carbon, and humus (Spaeth 2020 – adapted from Weil and Brady 2017).

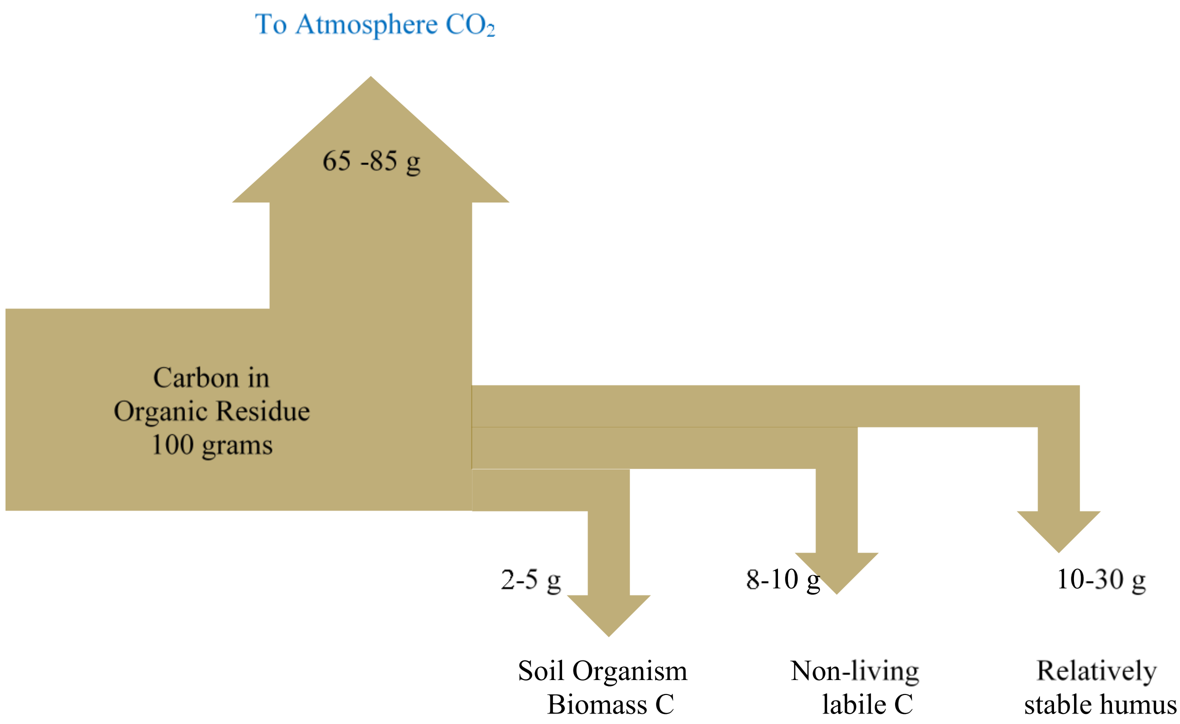


Figure K-13. Decomposition or turnover rates of four organic carbon pools (Spaeth 2020).

Crop Residues	Particulate Organic Carbon	Carbon-Humus	Recalcitrant Organic Carbon
labile	labile	resistant	inert
weeks to years	years to decades	decades to centuries	centuries to millennia

G. Soil respiration is an important indicator of microbial activity, soil organic matter cycling, and decomposition dynamics. Temperature, moisture levels, pH, salinity, porosity, and soil physical factors such as soil texture, bulk density, and aggregate stability can impact soil respiration. For example, low porosity relates to lower soil respiration rates. For every 10°C (18°F) increase in soil temperatures, microbial respiration more than doubles to a maximum of 35 to 40°C (95 to 104°F) (USDA-NRCS 2014). As CO_2 is released from the soil via microbial decomposition of soil organic matter, the rates of respiration can be an indicator of soil health trends. For example, high respiration rates in cropland settings may be an indicator of excessive tillage or other factors that degrade soil health (USDA-NRCS 2014). Soil respiration can also be an indicator of organic matter nutrients being converted to usable forms, such as phosphate as PO_4, nitrate-nitrogen as NO_3, and sulfate as

SO_4. Climate plays an important part in soil respiration on rangelands; compaction from livestock (increasing bulk density and lowering porosity) most commonly affects soil respiration.

H. Stabilization and the accumulation of organic matter in the soil are regulated by the soil environment and microbial activity. In almost every aspect of our terrestrial ecosystem, especially the soil environment, organic matter is involved in the chemical and physical aspects in one way or another. The ramifications are complex and beyond the scope of this discussion. One important soil physical factor, soil aggregate stability, is especially affected by particulate organic matter (plant tissues and cell wall materials) that were inaccessible to microbial action, where microaggregates form around them, together with microbial exudates. These hydrophobic (water repellent) biomolecules of either plant origin or byproducts of microbial synthesis (bacterial polysaccharides, fungal glycoproteins (glomalin)) are important constituents in stabilizing soil aggregates, which are an important aspect of hydrologic function and indicator of soil health. Polarity zones formed by microbial oxidation of these biomolecules allow the carbon to be stabilized and bonded to soil mineral surfaces, forming stabilized aggregates. Many other compounds are associated with soil aggregate stability: waxes, humic acids, aliphatic carbon, hydrophobins (fungal proteins), fatty acids, fulvic acids, and many other extracellular enzymes and polysaccharides. Cultivation, grazing hoof impact, foot traffic, intense fire effects, and erosion can cause degradation of soil aggregates. On grazing lands, managed grazing can be used to mitigate the effects of soil aggregate disturbance and breakdown. Stocking rates and timing of grazing with consideration to soil texture and soil stability dynamics are important variables to consider. For example, heavy stocking on heavier textured soils in early spring during wet periods will result in compaction and have a significant effect on soil aggregates, which will have an effect on infiltration and subsequent runoff and erosion risks.

I. Conceptual Flow of Soil Organic Matter and Associated Nutrients

(1) Several iterations of the organic matter cycle are presented because many examples are available in the literature. Although basic patterns are similar, each includes different details (figures K-14, K-15, K-16, K-17). In figure K-14, a simple representation of the organic matter cycle is given. Figure K-15 shows plant litter carbon dynamics in a grassland soil (Mollisol). Figure K-16 provides an example of microbial metabolic processes associated with carbon cycling. Figure K-17 shows content and turnover of organic dry matter in the tallgrass prairie with calculations of soil organic matter turnover. Figures K-18, K-19, and K-20 provide estimates for soil organic carbon sequestration after long-term grass establishment on farmed soils in central Texas. Figure K-21 shows soil C dynamics for long-term studies in the United Kingdom for permanent grass, seeded grass, long-term arable, and plowed grassland. Figures K-24 and K-25 show soil organic carbon dynamics in rangeland plant communities with disturbances (including grazing).

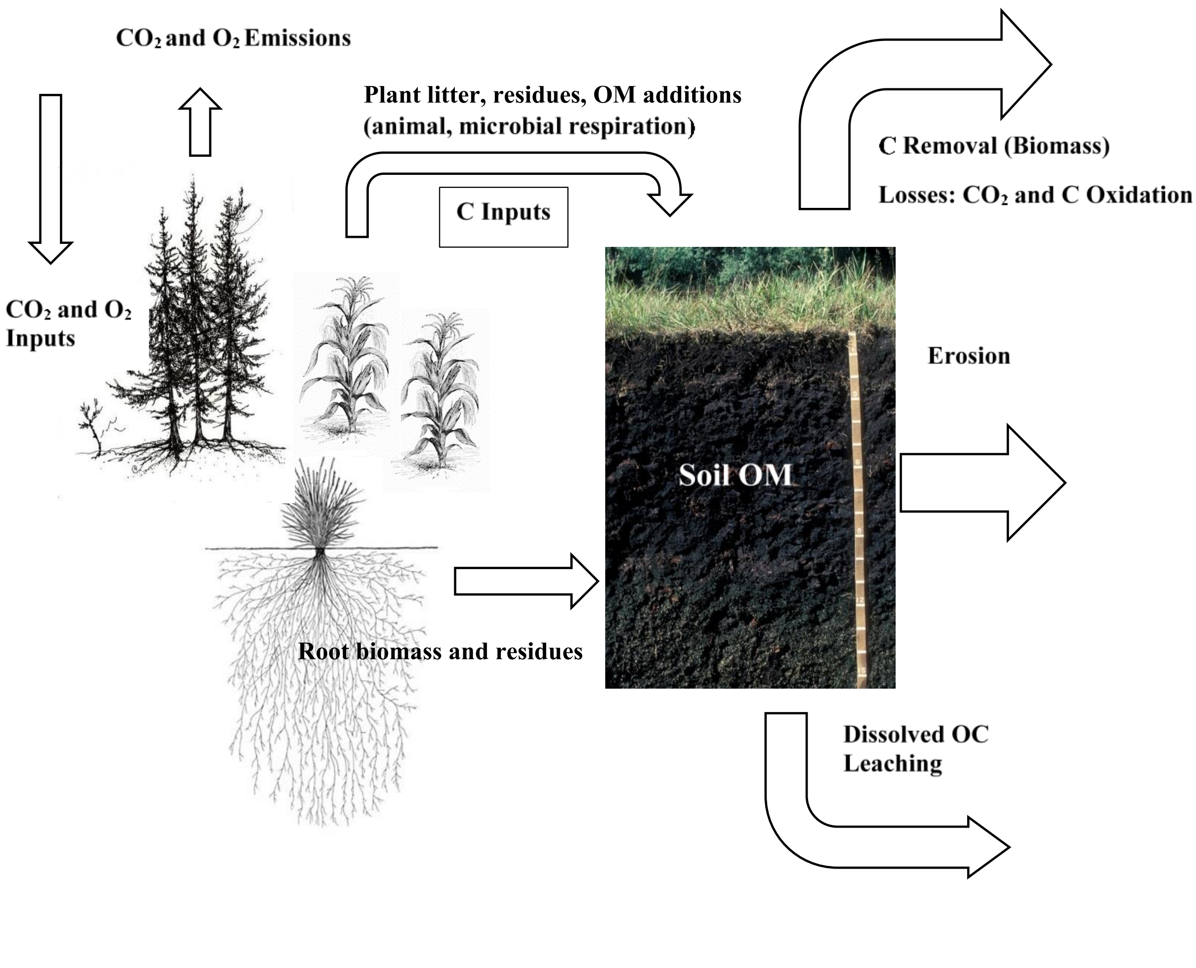
(2) Two figures are included showing carbon balance in cropland. These examples are included to show the progression and reality of increasing or enhancing soil organic matter in short time periods (see figures K-22, K-23).

(3) One important point regarding soil organic matter on stable rangelands: The annual increase in organic matter fluctuates between positive and negative values, but the long-term average is approximately a zero gain (Larcher 1983). Many studies document that organic matter contents of undisturbed soils (under natural vegetation) are in equilibrium with biological and biochemical properties (Dzurec et al. 1985; Trasar-Cepeda et al. 1998; Potter et al. 1999; McLauchlan et al. 2006; Zhang et al. 2007; Guilherme et al. 2009; Johnston et al. 2009; Johnston 2011; Crews and Rumsey 2014). There is a biochemical balance in undisturbed soils, and when soils are disturbed or subject to stress conditions, this balance is disrupted (Guilherme et al. 2009). Crews and Rumsey (2014) state: “Grassland restorations have been shown to sequester soil organic carbon (SOC) at rates of ~0.5 Mg C Ha^{-1} yr^{-1} averaged over several decades (Potter et al. 1999 – a century) through increased below-ground primary

productivity and stabilization of soil organic matter (Burke et al. 1995, Potter et al. 1999, Conant et al. 2001, McLauchlan et al. 2006, Kucharik 2007, Matamala et al. 2008)."

(i) In central Texas tall grass prairie, Potter et al. (1999) showed that agricultural practices reduced soil organic carbon 30 to 43 percent in the surface 60 cm of heavy clay soils. Restoring previously tilled (six to 60 yrs.) soils to tall grasses resulted in an increase in soil organic carbon in the surface 60 cm at a mean rate of 0.447 Mg C ha^{-1} yr^{-1} (0.199 t ac^{-1} yr^{-1}) (figs. K-18, K-19, and K-20). Using the bulk density values in the study (Clay, bulk density=1.250 Mg m^3), the equivalent percent soil organic carbon increase at 0.447 Mg C ha^{-1} yr^{-1} equals about 0.006 percent increase per year (figure K-18). Thus, the estimated time to restore soil organic carbon to comparative levels of native prairie from cropped lands seeded to grass ranged from 100 to 175 years. On farmed soils (> 20 yrs.) in Minnesota, soil organic carbon on restored 40-year grass stands (mixtures of C3 and C4 species) in the top 10 cm of soil accumulated at a constant rate of 0.62 Mg ha^{-1} yr^{-1}, regardless of whether the vegetation type was dominated by C3 or C4 grasses (Mclauchlan et al. 2006). They estimated that 55–75 years would be required to achieve soil organic carbon equivalents with unplowed native prairie sites.

Figure K-14. Input and output of soil organic matter. Balance can be sustainable or unsustainable, depending on disturbances (natural and anthropogenic), and land use and management. Management in terrestrial plant communities, especially those concerned with soil health, should focus on inputs and minimize outputs. (Spaeth 2020 – adapted from Weil and Brady 2017.)



(ii) Johnston et al. (2009) point out that it is "not always appreciated that soil organic matter changes toward an equilibrium level in any farming system, and the level will vary with a number of factors . . . Existing evidence shows that the amount of organic matter in soils depends on: 1) the input of organic material and its rate of oxidation, 2) the rate at which existing soil organic matter decomposes, 3) soil texture; and 4) climate conditions. . . For any single cropping system, the equilibrium level of soil organic matter in a clay soil will be larger than in a sandy soil, and for any single soil type the equilibrium level will be larger under permanent grassland than under continuous arable cropping."

(4) In Liang et al. (2017) (figure 13) a "conceptual scheme" shows related pathways and effects of fungi and bacteria growth, metabolism, and death in a terrestrial carbon cycle. The primary inputs of carbon are achieved via two pathways: *in vivo* turnover (inside the living organism) and *ex vivo* (outside the organism) modification driven by microbial catabolism (the breakdown of complex molecules and compounds to form simpler ones, together with the release of energy), and/or anabolism (creation of other molecules that catabolism breaks down). The authors use the microbial carbon pump concept as a sequestration system during *in vivo* turnover, resulting in anabolism-induced necromass, the persistent carbon pool. The authors stress that much more work is needed to understand microbial contribution to the terrestrial carbon cycle.

Figure K-15. Plant Litter Carbon Dynamics in a Grassland Soil (Mollisol) (20 cm soil depth). Units for carbon pools expressed as (kg C m^{-2}) and annual allocations (kg C m^{-2} yr^{-1}). At 20 cm soil depth, total carbon content is 10.4 kg C m^{-2} (104 Mg ha^{-1}; 46.4 t ac^{-1}); 84 percent of carbon respired (Spaeth 2020 – adapted from Schlesinger 1977).

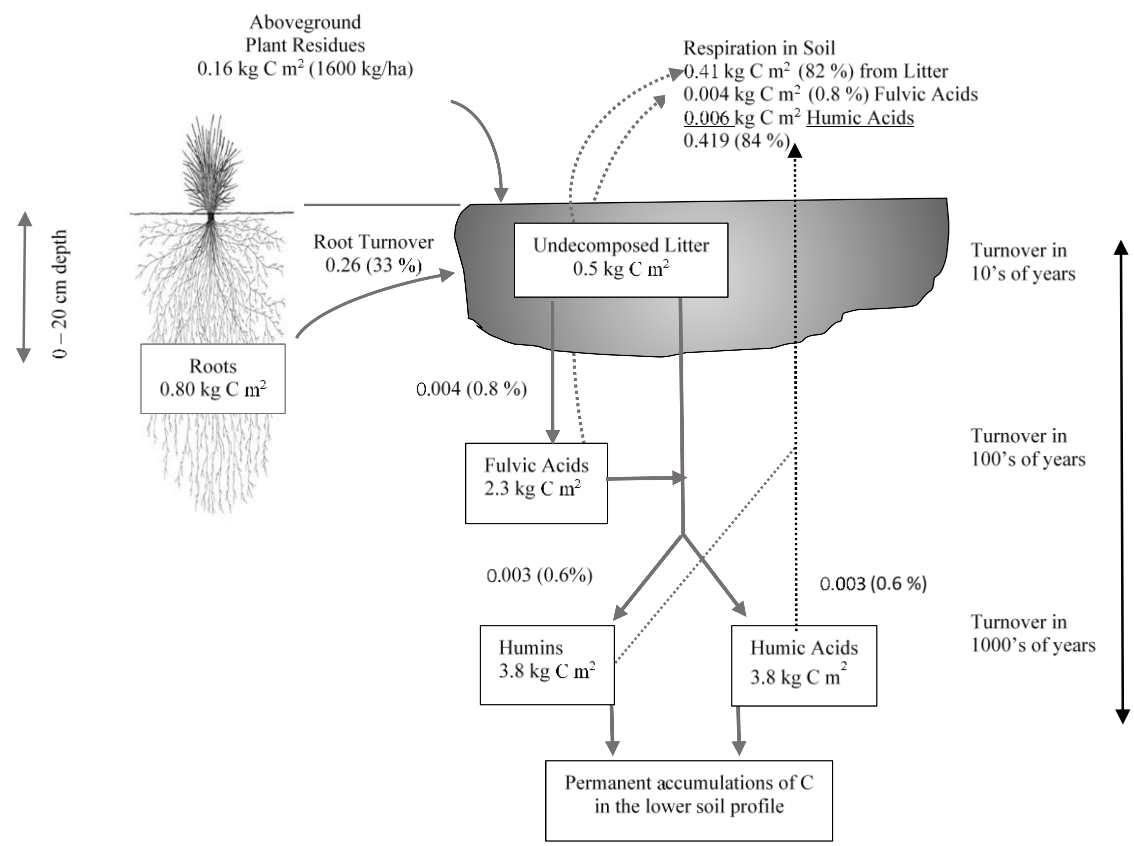


Figure K-16. Microbial metabolic processes associated with carbon cycling (Spaeth 2020 – adapted from Liang et al. 2014).

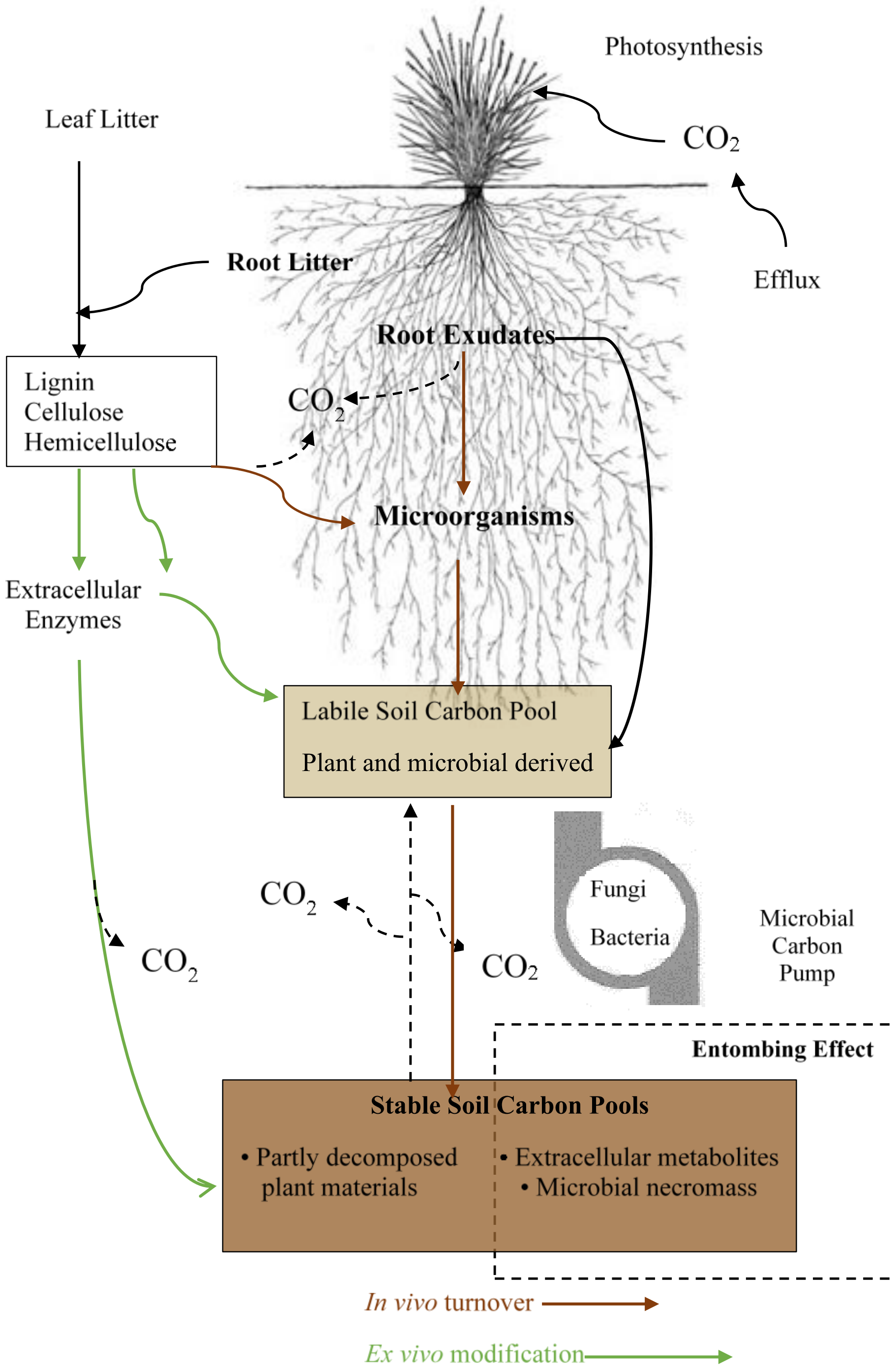
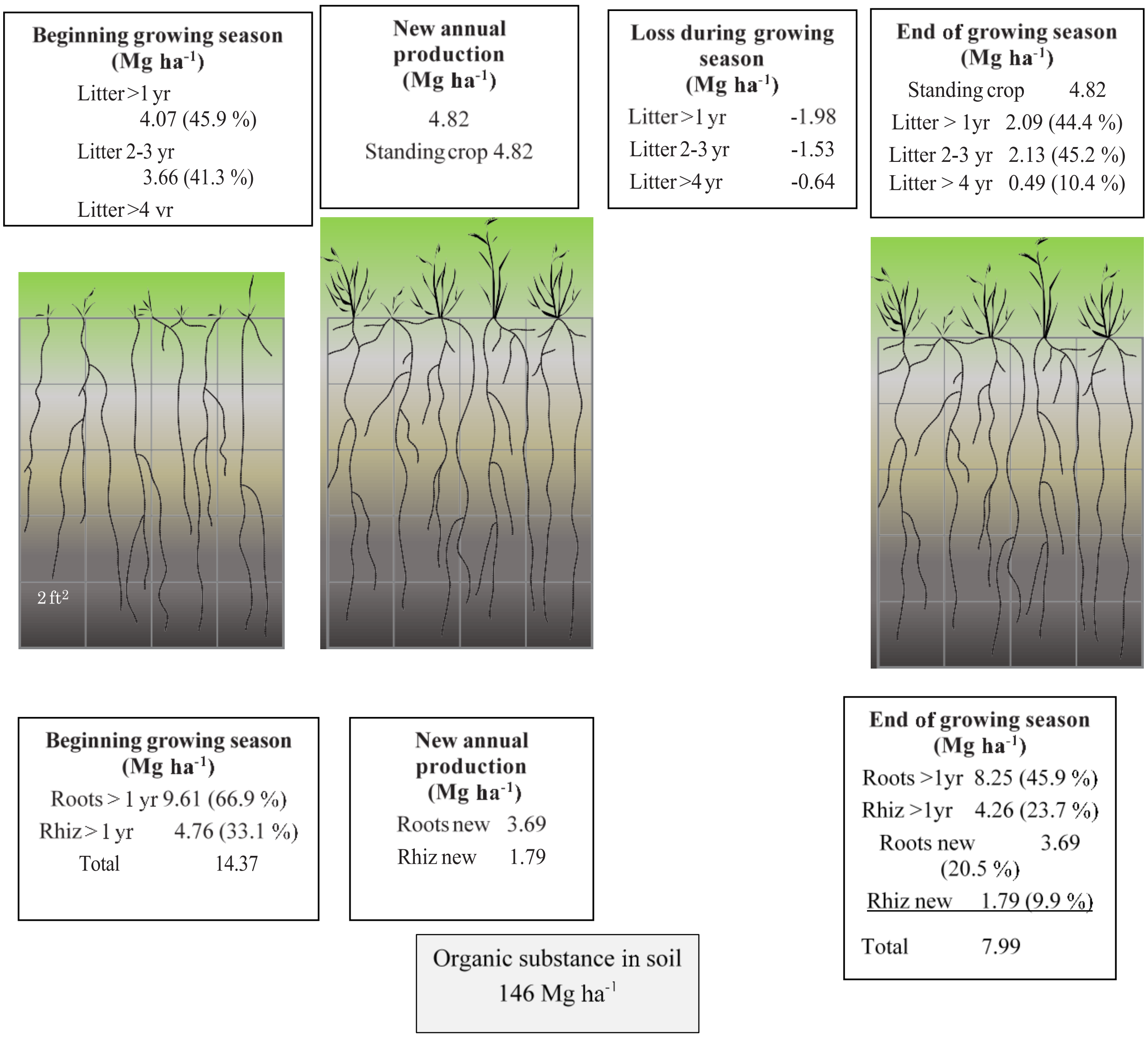


Figure K-17. Content and turnover of organic dry matter in the tallgrass prairie. Site Data: No grazing of large herbivores, only from insects and rodents. During the growing season, the vegetation grows rapidly and at the same time, parts of the shoots and roots die off or are eaten. The annual increase in organic matter fluctuates between positive and negative values, but the long-term average is approximately a zero gain (Larcher 1983 – adapted from Spaeth 2020; data from Kucera et al. 1967; Larcher 1983).



Carbon Balance in Tall Grass Prairie (1 year)

Weight of soil kg ha @ 0.15 m soil depth
(100m) (100m) (0.15m) = 1500 m^3 ha^{-1}
Weight of soil = (1,500 m^3 ha^{-1}) (Bulk density 1.5 Mg m^3) = 2,250 Mg ha^{-1} = 2,250,000 kg ha^{-1}
SOC = 1.75% therefore total wt. of SOC per ha = (2,250,000 kg ha^{-1}) (0.0175 % SOC) = 39,375 kg C ha^{-1}
SOC Carbon to nitrogen ratio = 12:1; 39,375/12 = 3,281.3 kg N ha^{-1}
SOC Carbon to phosphorous ratio = 50:1; 39,375/50 = 787.5 kg P ha^{-1}
End of year above ground production 4.82 Mg ha^{-1} or 2.15 t ac^{-1} (Kucera et al. 1967)
Root mass (8.25 Mg ha^{-1}) and rhizomes (4.25 Mg ha^{-1}) end of year = 12.51 Mg ha^{-1} (Kucera et al. 1967)
Root decomposition for Big Bluestem (Weaver and Darland 1947)

Root Size	**Pct Decomposition 1 yr**
Coarse 87%	55.7%
Medium 10%	33.2%
Fine Roots 3%	11.1%
Weighted average = 52.1%	

Root Biomass remaining after decomposition (12.51 Mg ha^{-1}) (0.521%) = 6.52 Mg ha^{-1}
Above Ground Biomass 4.82 Mg ha^{-1}
Litter Mg ha^{-1} = 2.09 Mg ha^{-1} (Kucera et al. 1967)
Total Biomass Current Yrs. Growth, Litter, Roots = 13.43 Mg ha^{-1}

C content biomass ~ 40% (13.43 Mg ha^{-1}) (0.4) = 5.37 Mg ha^{-1} = 5,371.68 kg ha^{-1}
Wt. of SOC and Total Prairie Site Veg OC = 39,375 kg ha^{-1} + 5,371.68 kg ha^{-1} = 44,746.68 kg ha^{-1}
Percent C retained in decomposed residue considered as humus (10-25%) used 20% = (5,371.68) (0.20) = 1,074.34 kg ha^{-1}
SOC + vegetation and roots 44,746.68 + 1,074.34 = 45,821.02 kg ha^{-1}
Increase of SOC from original soil carbon (1.75% SOC)
(45,821.02 kg ha^{-1}) (1.75%) / 39,375 kg C ha^{-1} = 2.04%, a gain of 0.29% in 1 yr.
Equivalent Soil Organic Matter start (1.75%) (1.72) = 3.01%, End of year SOM = (2.04%) (1.72) = 3.5%

Summary: This example represents a stable undisturbed tall grass prairie site. The annual increase in organic matter fluctuates between large positive and negative values, but the long-term average is approximately a zero gain (Larcher 1983). Removing corn stover not only reduces carbon return to soil but also nitrogen.

J. Three locations in central Texas (Temple, Burleson, and Riesel) were evaluated for soil organic carbon sequestration after long-term grass establishment on farmed soils (adapted from Potter et al. 1999). A cropland example is given to show lag in litter breakdown and amount of CO_2 released. Treatments were long-term farmed cropland (> 100 yrs.), pristine never-tilled native prairie, and sites seeded to grass (Temple seeded site established six yrs.; Burleson est. 26 yrs.; and Reisel est. 60 yrs). Soil organic carbon sequestration rate was determined to be 0.447 Mg ha^{-1} yr^{-1} (approx. 0.006 percent yr.) (see calculation below). Soils at all three sites were Vertisols.

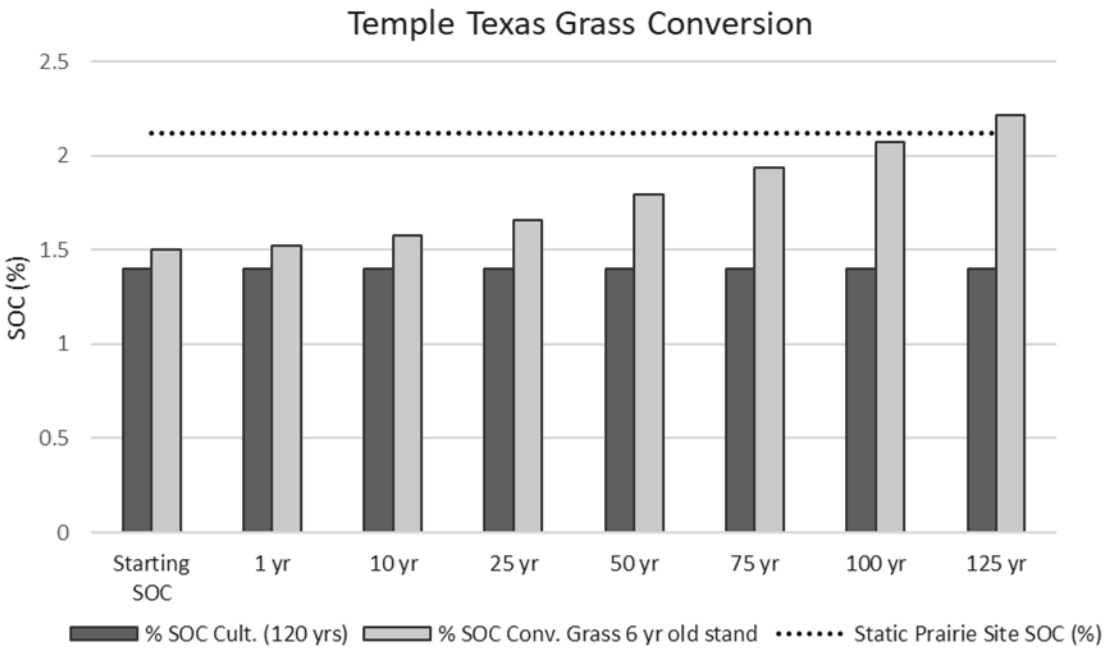
(1) Temple site:

(i) Site conditions

- Farmed 120 yrs., Clay, bulk density=1.250 Mg m^3 @ 0.6 meters soil depth
- (100 m) (100m) (0.6 m) = 6,000 m^3
- (6,000 m^3) (bulk density 1.250 Mg m^3) = 7,500 Mg ha^{-1} (soil wt.)
- (sequestration rate 0.447 Mg ha^{-1} yr^{-1}) (100) / 7,500 Mg ha^{-1} = 0.006 percent SOC yr^{-1}
- wt. of soil 7,500 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 1.4 percent.
- Native prairie was indiangrass (*Sorghastrum nutans*), little bluestem (*Schizachyrium scoparium*), and Johnsongrass (*Sorghum halepense*), with some giant ragweed (*Ambrosia trifida*). Clay, bulk density = 1.178 Mg m^3 @ 0.6 m soil depth, wt. of soil 7,068 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 2.12 percent.
- Seeded grass was switchgrass (*Panicum virgatum*) 6 yr. stand, bulk density = 1.34 Mg m^3 @ 0.6 m soil depth, wt. of soil 8,064 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 1.50 percent.

(ii) Figure K-18 shows step increases of soil organic carbon sequestration in a converted six-yr-old stand of switchgrass at Temple, TX (1.50 percent soil organic matter). The farmed site static level for soil organic carbon = 1.4 percent, and the native prairie static soil organic carbon level was 2.12 percent. Soil organic carbon was reduced 66 percent by agricultural cropping practices in the surface 60 cm of heavy clay soils. Returning the previously tilled soils to seeded grass resulted in an increase in soil organic carbon in the surface 60 cm at a mean rate of 0.447 Mg C ha^{-1} yr^{-1}. The length of time to reach static soil organic carbon levels to that of the native prairie would require 100 to 125 yrs.

Figure K-18. Change in Soil Organic Carbon over time on a Temple, Texas, site.



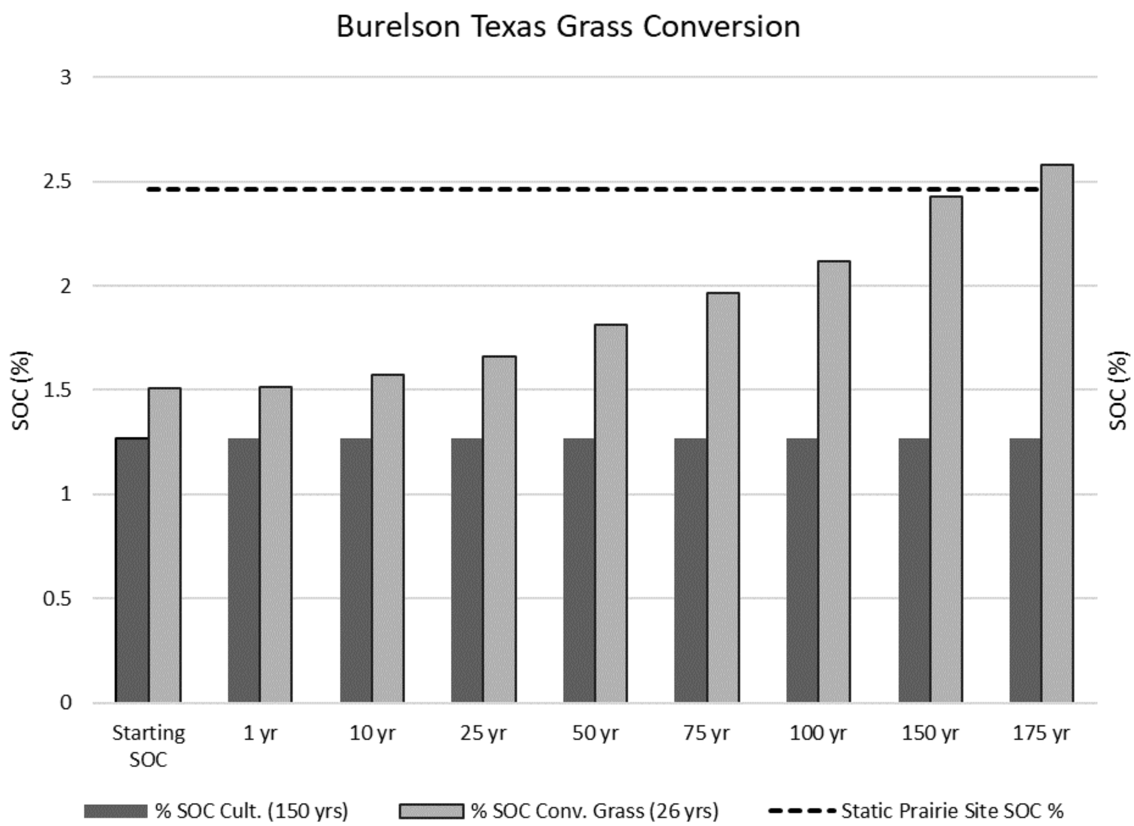
(2) Burleson site:

(i) Site conditions

- Farmed 120 yrs., clay, bulk density=1.317 Mg m^3 @ 0.6 meters soil depth, wt. of soil 7,902 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 1.27 percent.
- Native prairie was indiangrass and little bluestem (*Schizachyrium scoparium*), which may have been overgrazed in the past. Clay, bulk density = 1.278 Mg m^3 @ 0.6 m soil depth, wt. of soil 7,668 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 2.464 percent.
- Seeded grass was indiangrass, little bluestem, switchgrass, big bluestem (*Andropogon gerardii*), and sideoats grama *(Bouteloua curtipendula)* 26 yr. stand, bulk density = 1.22 Mg m^3 @ 0.6 m soil depth, wt. of soil 7,320 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 1.51 percent.

(ii) Figure K-19 shows step increases of soil organic carbon sequestration in a converted 26-yr-old stand of native grasses at Burleson, TX (1.51 percent soil organic matter). The farmed site static level for soil organic carbon = 1.27 percent, and the native prairie static soil organic carbon level was 2.464 percent. Soil organic carbon was reduced 51.5 percent by agricultural cropping practices in the surface 60 cm of heavy clay soils. Returning the previously tilled soils to seeded grass resulted in an increase in soil organic carbon in the surface 60 cm at a mean rate of 0.447 Mg C ha^{-1} yr^{-1}. The length of time to reach static soil organic carbon levels to that of the native prairie would require 150 to 175 yrs.

Figure K-19. Change in Soil Organic Carbon over time on a Burleson, Texas, site.



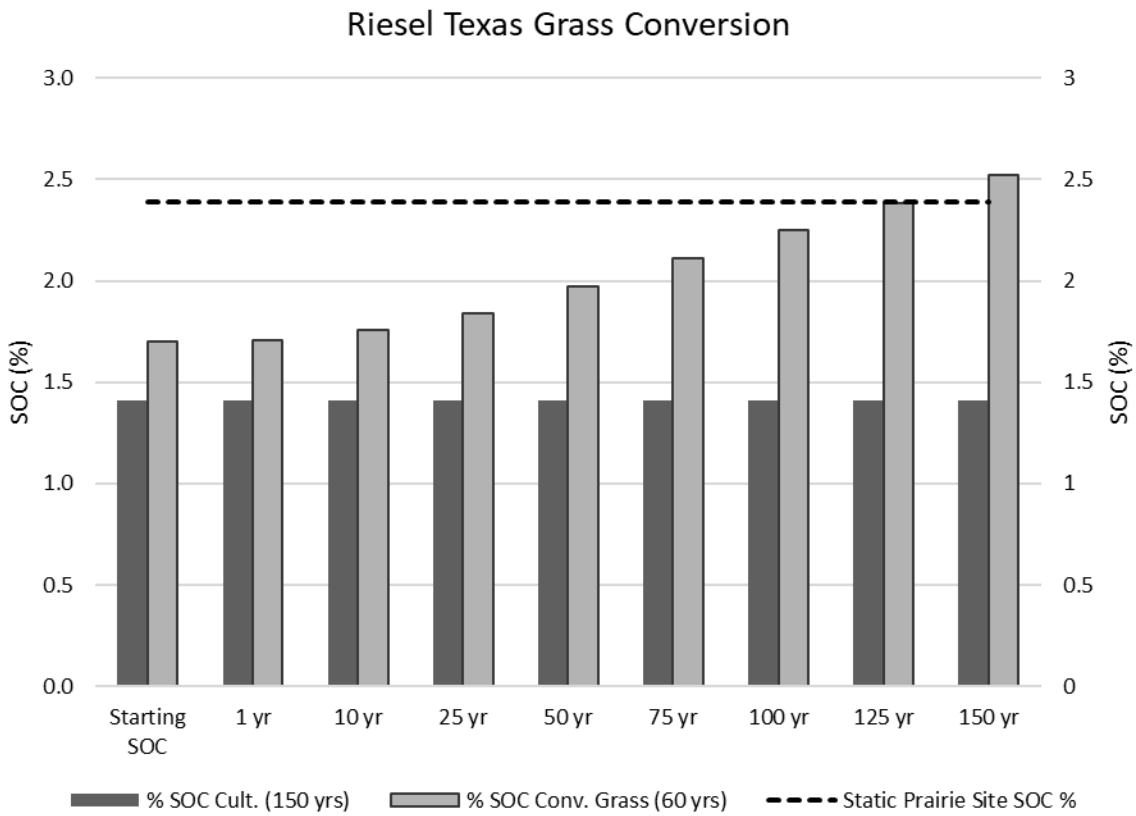
(3) Riesel site:

(i) Site conditions

- Farmed 120 yrs., clay, bulk density= 1.457 Mg m^3 @ 0.6 meters soil depth, wt. of soil 8,742 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 1.4 percent.
- Native prairie was King Ranch bluestem (*Bothriochloa ischaemum*) and little bluestem (*Schizachyrium scoparium*), with a strong influence of giant ragweed (*Ambrosia trifida*). Clay, bulk density = 1.22 Mg m^3 @ 0.6 m soil depth, wt. of soil 7,320 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 2.39 percent.
- Seeded grass was King Ranch bluestem *(Bothriochloa ischaemum*), little bluestem, indiangrass (*Sorghastrum nutans*), and switch grass (*Panicum virgatum*) 60 yr. stand, bulk density = 1.37 Mg m^3 @ 0.6 m soil depth, wt. of soil 8,220 Mg ha^{-1} @ 0.6 m, percentage SOC @ 0.6 m = 1.7 percent.

(ii) Figure K-20 shows step increases of soil organic carbon sequestration in a converted 60-yr-old stand of native grasses at Riesel, TX (1.7 percent soil organic matter). The farmed site static level for soil organic carbon = 1.4 percent, the native prairie static soil organic carbon level was 2.39 percent. Soil organic carbon was reduced 58.6 percent by agricultural cropping practices in the surface 60 cm of heavy clay soils. Returning the previously tilled soils to seeded grass resulted in an increase in soil organic carbon in the surface 60 cm at a mean rate of 0.447 Mg C ha^{-1} yr^{-1}. The length of time to reach static soil organic carbon levels to that of the native prairie would require 125–150 yrs.

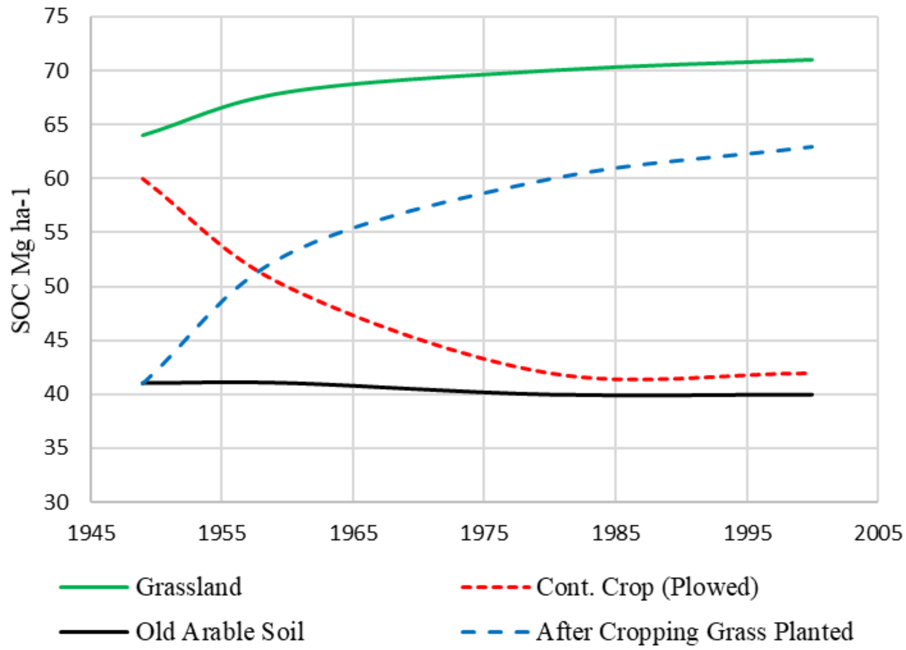
Figure K-20. Change in Soil Organic Carbon over time on a Riesel, Texas, site.



(4) In conclusion, in central Texas, agricultural practices reduced soil organic carbon 30 to 43 percent in the surface 60 cm of heavy clay soils (Potter et al. 1999). Restoring previously tilled (six to 60 yrs.) soils to grass resulted in an increase in SOC in the surface 60 cm at a mean rate of 0.447 Mg C ha^{-1} yr^{-1} (0.199 t ac^{-1} yr^{1}). In three tall grass locations in Texas, estimated time to restore soil organic carbon of cropped lands seeded to grass ranged from 100 to 175 years.

K. Figure K-21 shows the changes in organic C ha^{-1} for permanent grass, old arable field, continuous plowed crop, and reseeded grass for a period of about 50 years (Johnston et al. 2009). On the field with old arable cropping, organic C remained essentially constant but declined steadily where the old grassland soil was plowed. On the undisturbed permanent grass field, organic C slowly increased toward a new equilibrium level as a result of more intensive management and increased N applications that increased above-ground yields, resulting in greater root growth and decomposition that increased organic matter inputs. On the old arable soil sown to grass, the amount of C increased slowly; but after about 50 years it was still lower than the permanent grass field.

Figure K-21. Changes in SOC (Mg ha^{-1}) in surface 23 cm silty clay loam (Rothamsted United Kingdom 1949–2002). Four treatments represent old grassland, plowed and cropped, old arable crop field, and field sown to grass (adapted from Johnston et al. 2009).



L. Corn stover harvest removes carbon that potentially could be recycled and incorporated into soil organic carbon pools. However, decomposition of crop residue by soil microbes with associated large carbon loss as CO_2 is not commonly recognized. Figure K-22 shows decomposition rates and cumulative CO_2 loss to the atmosphere at one to eight years. In the long term, 85 percent of the original corn stover biomass is lost as atmospheric CO_2. Removing corn stover not only reduces carbon return to soil but also nitrogen.

Figure K-22. Effect on Soil Carbon with Corn Stover Removal Over Time. What is the Effect on Soil Carbon with Stover Removal? (Spaeth 2020 – adapted from Sawyer and Mallarino 2007).

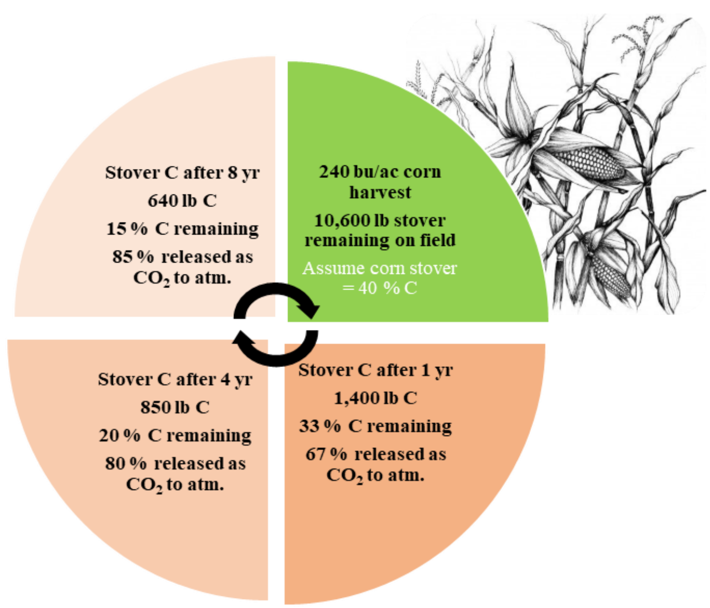


Figure K-23. Example of annual carbon balance in Iowa corn field. Units in parentheses are kg ha^{-1} (Spaeth 2020).

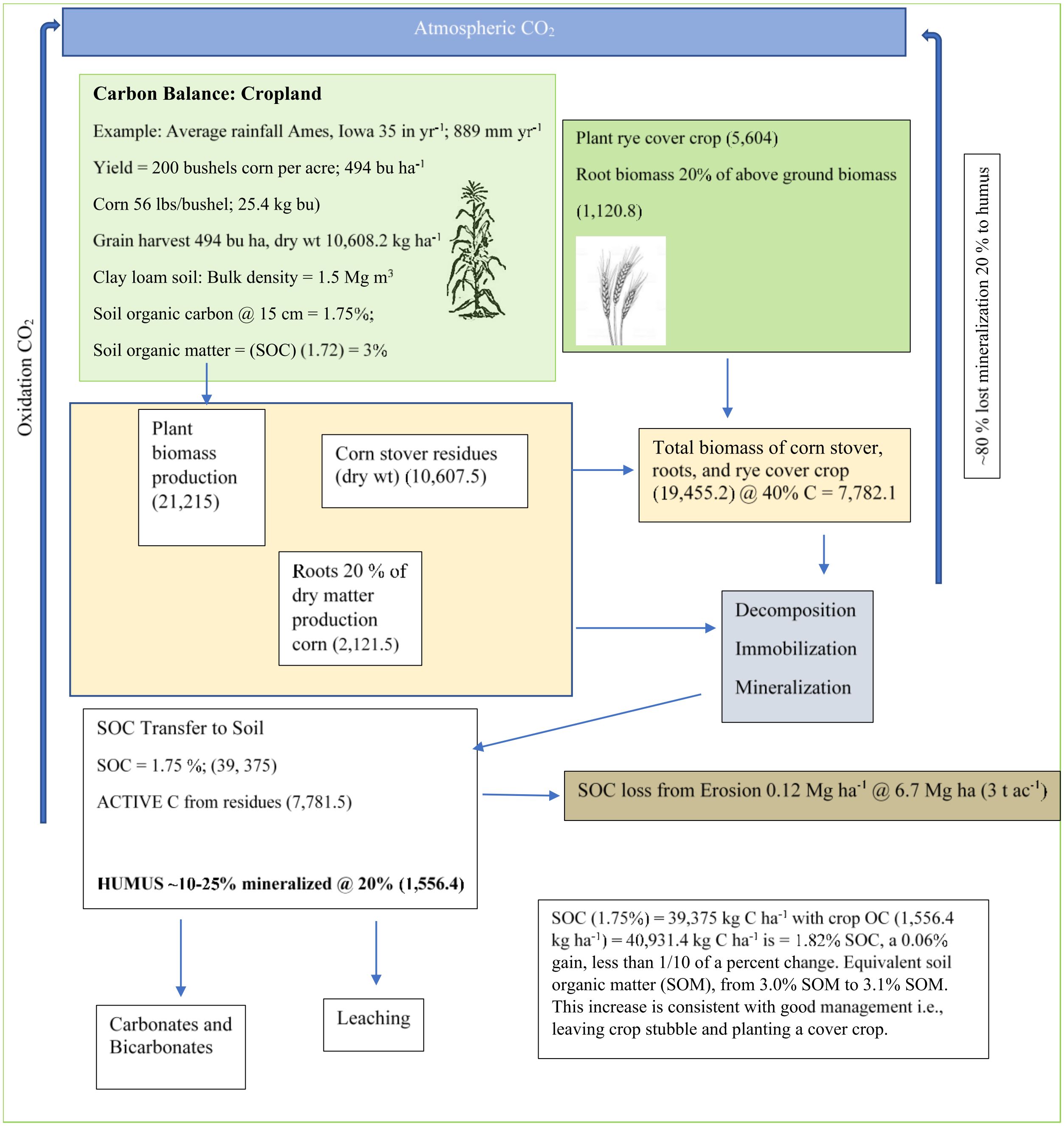
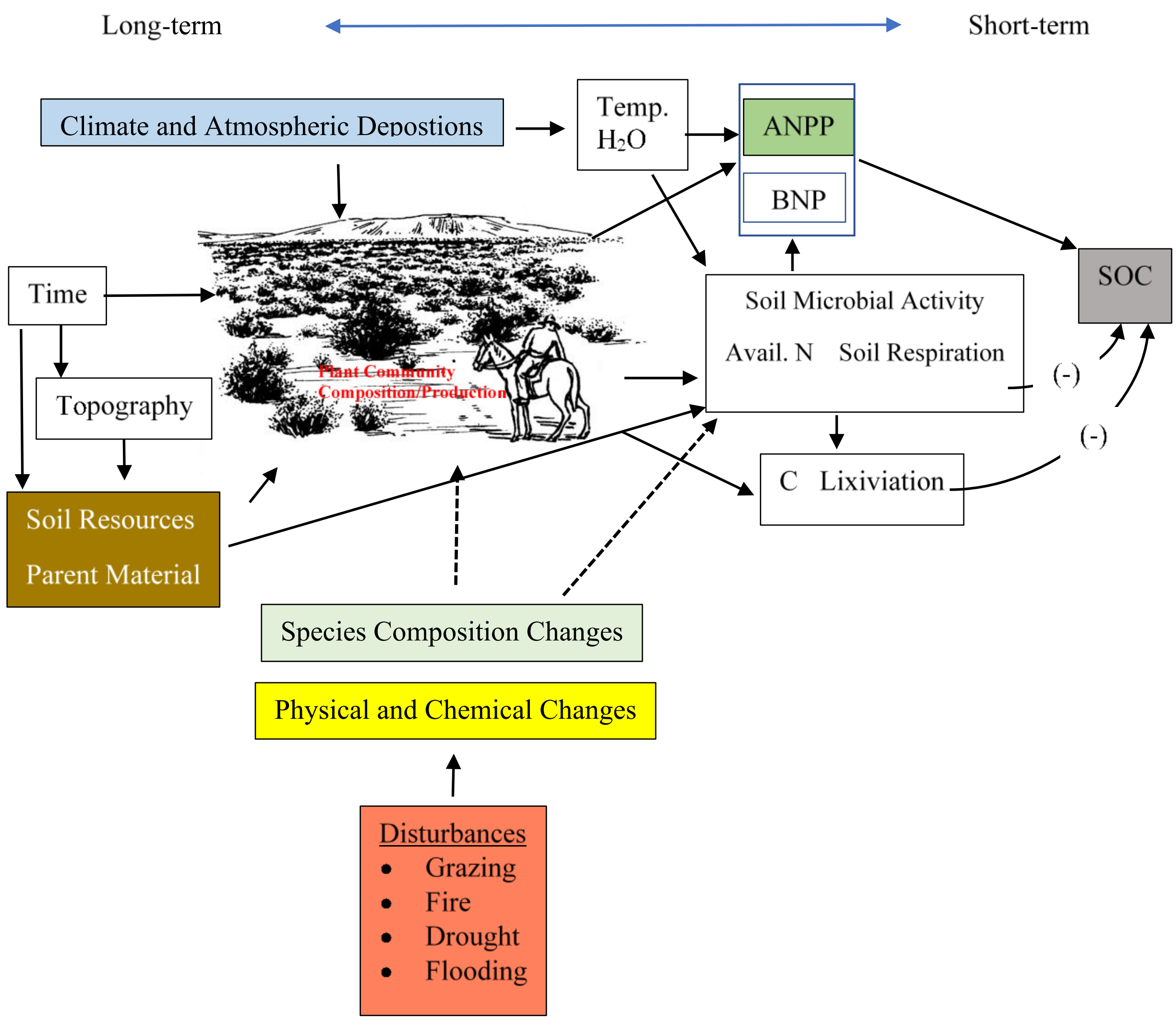


Figure K-24. Soil organic carbon (SOC) dynamics in rangeland plant communities with disturbance interactions. Dashed line shows effects of disturbances (including grazing), ANPP = above ground net primary productivity, BNPP = below ground net primary productivity. Lixiviation is the process of separating soluble from insoluble substances by dissolving the former in water (adapted from Pineiro et al. 2010).

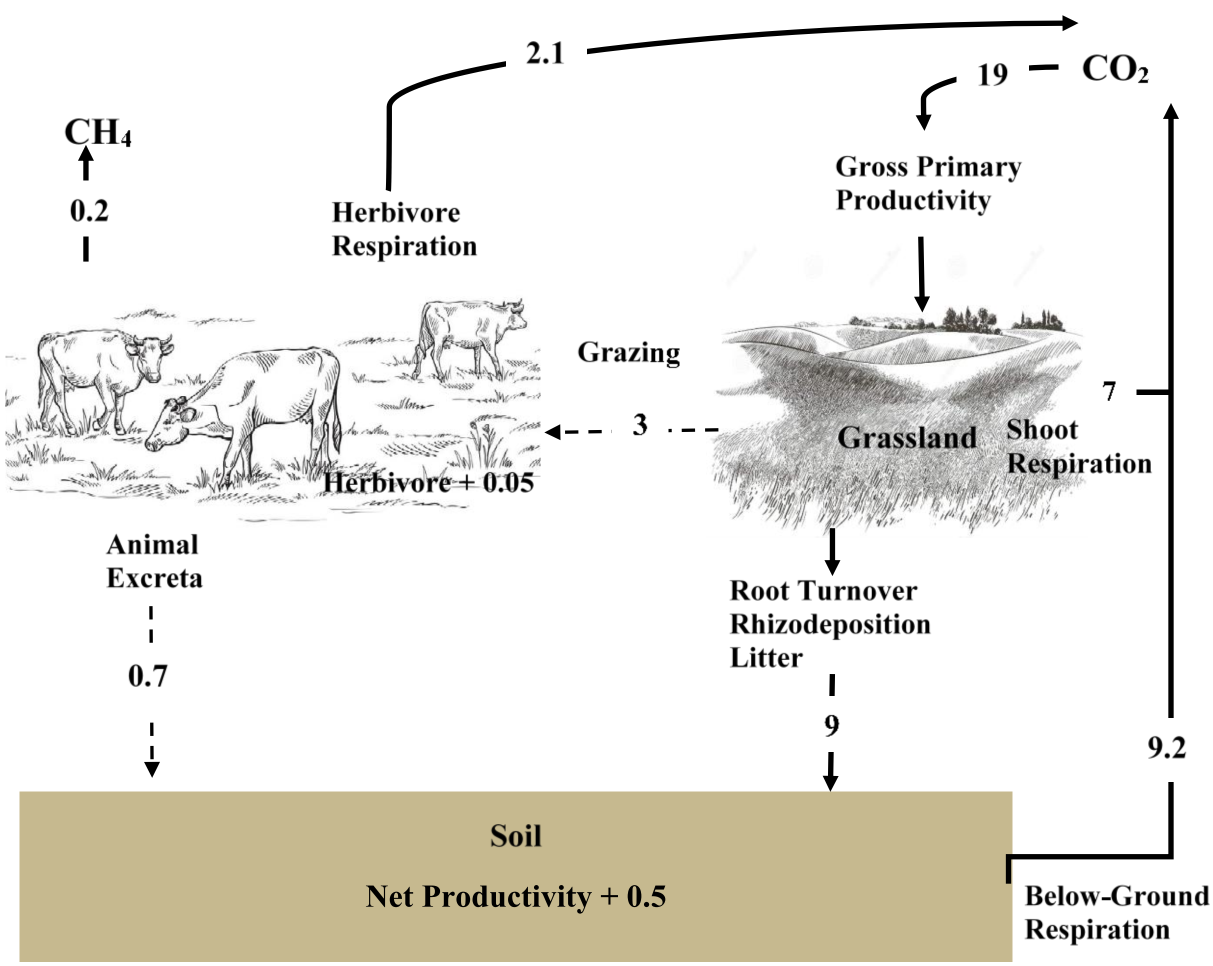


645.1109 Grazing Effects on Soil Organic Carbon

A. Rangeland plant communities are diverse and include grasses (bunch- and/or sod-forming spp.), forbs, shrubs, and trees; plants with different root morphologies; and mixtures of C3 (cool season spp.), C4 (warm season spp.), and CAM (succulents) plants. Included in the mix of complex environmental interactions are perturbations such as fire, herbivory, and a propensity to drought. All of these factors influence the soil organic carbon cycle. Soil organic matter is the main reservoir of soil organic carbon and soil organic nitrogen in rangeland ecosystems and plant communities and affects the three rangeland health attributes (soil and surface stability, hydrologic function, and biotic integrity). Range and pasturelands make up about 10–30 percent of the Earth's terrestrial capacity (Shuman and Derner 2004), and management practices such as grazing affect soil organic carbon because plant community production and composition are dynamic throughout the season and from year-to-year. On rangelands, carbon cycling and sequestration dynamics are quite complex, and estimation of rates and amounts are systematically more difficult than for cultivated croplands (Schuman et al. 2002) because of more heterogeneous edaphic characteristics, wide daily temperature fluctuations, intermittent precipitation, diverse vegetation life and growth forms, plant composition, productivity, root-shoot ratios, different rooting depths, herbivore use, and imposed disturbance and management practices.

B. Figure K-25 depicts a carbon cycle scenario of intensive continuous grazing (60 percent) utilization on grassland (Soussana et al. 2004). Frequency and intensity of grazing affects carbon balance. In this example, Sousanna et al. (2004) state that “The largest part of the ingested carbon is digestible (up to 75 percent for highly digestible forages) and, hence, is respired shortly after intake. Only a small fraction of the ingested carbon is accumulated in the body of domestic herbivores or is exported as milk. Large herbivores, such as cows, respire approximately one tonne of carbon per year. Additional carbon losses (0.5 percent of the digestible carbon) occur through methane emissions from the enteric fermentation. The non-digestible carbon (25±40 percent of the intake according to the digestibility of the grazed herbage) is returned to the pasture in excreta (mainly as faeces).”

Figure K-25. Carbon cycling in grazed grassland. Numerical carbon fluxes (Mg C ha^{-1} yr^{-1}). Example represents continuously intensive grazing (adapted from Soussana et al. 2004).



C. The structural components of organic matter are soil organic carbon and nitrogen, characterized as carbon:nitrogen (C:N) ratios. Soil organic matter is also the primary source of nitrogen (12:1), phosphorus (50:1), and sulfur (70:1). The C:N ratio of the organic material is critical to the amount of nitrogen required by microbes for decomposition. If the C:N ratio is above 25:1, soil microbes will mine soluble nitrogen from the soil (Spaeth 2020). Mineralization and immobilization are important concepts related to C:N relationships and organic matter. Mineralization is a biological process where organic substances are converted to inorganic substances by soil microorganisms, with end products that include CO_2, H_2O, and nutrients. Carbon-nitrogen ratios less than 25:1 (25 parts carbon to 1 part nitrogen) have sufficient nitrogen for the microorganisms involved in decomposition (mineralization), as well as providing some excess inorganic nitrogen to plants. Immobilization is the reverse of mineralization and involves the incorporation of mineralized inorganic compounds into organic molecules within living cells of organisms. All living things require nitrogen; therefore, microorganisms in the soil compete with plants for nitrogen. During immobilization, nitrate and ammonium are taken up by soil organisms and therefore become unavailable to plants (Johnson et al. 2005).

D. Soil organic matter C:N ratios can shift with grazing or other disturbances. Soil organic nitrogen dynamics frequently constrain carbon dynamics and soil organic carbon accumulation in soils (Wedin 1995; Pineiro et al. 2006, 2009; Harpole et al. 2007). For example, the C:N ratios of plant materials and their respective litter dynamics are quite varied [fertilized Kentucky bluegrass (*Poa pratensis*) 20:1; Bermuda grass (*Cynodon dactylon*) hay 49:1; timothy grass (*Phleum pratense*) hay 58:1; blue grama (*Bouteloua gracilis*) 20:1 rangeland; rye cover crop 65:1; corn stover 60:1; wheat straw 80:1]. Soil organic nitrogen is not the only constraint to soil organic matter accumulation, as water availability, soil moisture, and carbon uptake (via photosynthesis and net primary production) can be limiting factors for soil organic matter accumulation and sequestration, especially in arid sites (Burke et al. 1998).

E. Grazing frequency and intensity affects soils, hydrology, plant composition, and productivity of grazing lands (McNaughton 1985; Sala 1988; Thurow 1991; Milchunas and Lauenroth 1993; Altesor et al. 1998; Schuman et al. 1999; Spaeth 2020). Effects can be assessed in three attributes: soil and surface stability, hydrology, and biotic components (Pellant et al. 2020). Understanding and communicating the effects of grazing on specific rangeland plant communities is important during conservation planning and implementation. For example, how does grazing affect soils (bulk density, soil organic carbon balance and cycling, aggregate stability, nutrient cycles), hydrologic implications associated with runoff and erosion, and the plant community (composition and trajectories of states and phases, and inherent thresholds at the ecological site level)? Ecological Site Descriptions, Rangeland Health worksheets, and State-and-transition models are all helpful in providing information and assisting rangeland users. However, some uncertainties are not predictable – climatic effects can change and alter what was once a predictable outcome to uncertain shifts in plant composition (both in the short and long term). For conservation planners, this is an important point that needs to be emphasized with land-users and land managers.

F. It is difficult to synthesize generalizations about the effects of grazing on soil organic carbon on grazing lands (Milchunas and Lauenroth 1993; Schuman et al. 2005; Derner and Schuman 2007; Pineiro et al. 2010). Managed grazing can avoid or alleviate serious soil disturbance factors. However, changes in vegetation composition in response to grazing can alter soil carbon (Pineiro et al. 2010; Hewins et al. 2018). In addition, soil microbial shifts also occur during drought, which can result in losses of soil carbon (Ingram et al. 2004). Pineiro et al. (2010) reviewed 20 articles with 67 comparisons of grazed and ungrazed sites. They found that soil organic carbon decreased or remained unchanged with contrasting grazing conditions with varied grassland temperature and precipitation gradients. Their conclusions were that grazing effects are complex in their effects on soil organic carbon. There is inconclusive evidence about how grazing affects the distribution and maintenance of soil carbon in different rangeland ecosystems. Since rangelands are so diverse with respect to vegetation composition, climate, and soils, deriving a general conclusion about grazing effects and systems on soil organic carbon cycles and carbon sequestration is unrealistic and unlikely. The dynamics of carbon balance and cycling can be documented at the ecological site level; however, the effects of grazing in the short-term cannot be reliably determined. In addition, Schuman et al. (2005) point out that the combination of severe drought and heavy grazing can result in significant losses of soil organic carbon that was previously stored under normal production levels. Rangelands then shift from sequestering carbon to releasing CO_2 to the atmosphere (Balogh et al. 2005a, b).

G. Numerous studies show varying effects of grazing on carbon balance:

(1) Grassland soils can store more than 100 and 10 tons per hectare of SOC and SON, respectively, in the surface meter (Jobbagy and Jackson 2000), and grazing can increase, decrease, or maintain unaltered the size of both pools (Milchunas and Lauenroth 1993; Derner et al. 2006; Pineiro et al. 2009, 2010).

(2) Soil organic content was reduced on grazed native grasslands compared to ungrazed grasslands (Bauer et al. 1987).

(3) In central Texas tall grass prairie, soil organic carbon in long term cropped fields in the surface 120 cm was 25 to 43 percent of the original prairie. In fields seeded to grasses (6 to 60 yrs. establishment), the estimated carbon sequestration rate was 0.447 Mg C ha^{-1} yr^{-1}. Based on this rate, it would take 100 to 185 years for the carbon pool to be equivalent to that of the undisturbed prairie (Potter et al. 1999).

(4) Pineiro et al. (2010) reviewed 20 articles with 67 comparisons of grazed and ungrazed sites and summarized some general patterns:

(i) Root biomass (a primary control of SOC formation) was lower at sites with intermediate precipitation (400 mm to 850 mm), but higher in grazed compared to ungrazed counterparts at the driest and wettest sites

(ii) Soil organic matter C:N ratios frequently increased under grazed conditions, suggesting potential N limitations for soil organic matter formation with grazing

(iii) Soil organic carbon decreased or remained unchanged with contrasting grazing conditions with varied grassland temperature and precipitation gradients

(iv) Soil bulk density either increased or did not change in grazed sites.

(5) Heavily grazed fescue grasslands in Alberta, Canada [0.2 ha per animal unit month (AUM^{-1})], had less soil organic matter compared to grazing at 0.8 ha per AUM^{-1} (Johnston et al. 1971).

(6) Sheep grazing on native *Stipa-Bouteloua* prairie at 2.5 ha AUM^{-1} showed increased soil carbon compared to grazing at 1.7 ha AUM^{-1} (Smoliak 1986).

(7) In northern mixed-grass prairie, North Dakota, moderate grazed treatments contained 17 percent less soil carbon compared to ungrazed exclosures. Heavily grazed treatments did not result in less soil carbon compared to the exclosure. The authors surmised that an increase in blue grama (*Bouteloua gracilis*), a grass species with a heavy dense root system, may be responsible for maintaining soil carbon levels equal to the exclosure (Frank et al. 1995).

(8) In meadow bromegrass (*Bromus riparius*) pastures in Alberta, Canada, "heavy and medium grazing intensities produced 83 and 90 percent as much above-ground dry matter and 87 and 90 percent above-ground carbon as the light intensity . . . heavy grazing reduced the contribution of vegetative dry matter in vitro digestible organic matter, carbon and nitrogen to the residual 41, 50, 36, and 52 percent of that for light grazing . . . estimated fecal carbon inputs were 68, 51, and 42 percent of all carbon inputs for heavy, medium, and light grazing, respectively" (Baron et al. 2002).

(9) Derner and Schuman (2007) stated: "although there was no statistical relationship between change in soil carbon with longevity of the grazing management practice in native rangelands

of the North American Great Plains, the general trend seems to suggest a decrease in carbon sequestration with longevity of the grazing management practice across stocking rates."

(10) In northern mixed-grass prairie, Schuman et al. (1999) found that light or heavy grazing resulted in higher carbon sequestration compared to nongrazed exclosures.

(11) Eighty-one years of moderate and heavy grazing in northern mixed-grass prairie showed increases of 19 and 34 percent of soil carbon at 0–5 cm and 5–15 cm soil depth (Wienhold et al. 2001).

(12) Manley et al. (1995) found that short-duration rotational grazing, rotationally deferred grazing, and continuous season-long grazing at heavy stocking rates did not affect carbon sequestration rates.

(13) In shortgrass steppe in northeastern Colorado, Derner et al. (1997) found increased soil carbon storage in grazed (1983 g m^2; 0.41 lb ft^2) compared to ungrazed areas (1321 g m^2; 0.27 lb ft^2) at 0–15 cm (0–5.9 in) soil depth, while no differences were found at the 15–30 cm (5.9–11.8 in) soil depth.

(14) During a 12-year period in "coastal" Bermuda grass (*Cynodon dactylon*)/tall fescue (*Lolium arundinaceum*) paddocks, annual soil organic carbon change at 0–90 cm soil depth was as follows: low grazing pressure (1.17 Mg C ha^{-1} $year^{-1}$; 0.52 t C ac^{-1} yr^{-1}) was greater than unharvested grass (0.64 Mg C ha^{-1} $year^{-1}$; 0.29 t C ac^{-1} yr^{-1}) but nearly equal to high grazing pressure (0.51 Mg C ha^{-1} $year^{-1}$; 0.23 t C ac^{-1} yr^{-1}). Hayed paddocks showed the lowest annual rate of soil organic carbon change (0.22 Mg C ha^{-1} $year^{-1}$; 0.1 t C ac^{-1} yr^{-1}) (Franzluebbers and Stuedemann 2009).

(15) During a five-year evaluation of soil organic carbon sequestration in Bermuda grass (*Cynodon dactylon*) pasture, sequestration in the surface 6 cm was 1.4 Mg C ha^{-1} $year^{-1}$ (0.62 t C ac^{-1} yr^{-1}) when grazed in summer by cattle, 0.65 Mg C ha^{-1} year−1 (0.29 t C ac^{-1} yr^{-1}) when ungrazed, and 0.29 Mg C ha^{-1} $year^{-1}$ (0.13 t C ac^{-1} yr^{-1}) when hayed (Franzluebbers et al. 2001).

(16) Hewins et al. (2018) evaluated 108 pairs of long-term grazed and ungrazed study sites in dry mixed-grass prairie, central parkland, foothills fescue, and montane and upper foothills representing six distinct climate subregions across 5.7 M ha^{-1} (14 M ac^{-1}) of Alberta, Canada. Their findings found that moderate grazing increased soil organic carbon by 12 percent in the upper 15 cm (5.9 in) of soil, and soil organic carbon concentrations in deeper mineral soil layers were associated more with regional climate and increase from dry to mesic subregions. They concluded that "longterm livestock grazing may enhance soil organic carbon concentrations in shallow mineral soil and affirm that climate rather than grazing is the key modulator of soil carbon storage across northern grasslands."

(17) Mountain meadows grazed for 1–3 months by sheep and cattle in the Medicine Bow National Forest in Wyoming showed higher soil organic carbon in grazed compared to ungrazed treatments at 0–7.5 cm (0–3 in) soil depth (Povirk 1999).

(18) "Proper grazing management has been estimated to increase soil carbon storage on US rangelands from 0.1 to 0.3 Mg C ha^{-1}year-1 (0.044 – 0.133 t C ac^{-1} yr^{-1}) and new grasslands have been shown to store as much as 0.6 Mg C $ha^{-1}$$year^{-1}$ (0.26 t C ac^{-1} yr^{-1})" (Schuman et al. 2002).

645.1110 Conclusions

A. The objective of this chapter is to present basic information and facts about soil organic carbon and matter dynamics on grazing lands. Soil organic matter is a key concept to overall ecosystem function and health and integral to soil health. Understanding basic information about the carbon cycle is necessary to have accurate and meaningful discussions on range, pasture, and soil health with NRCS customers. Since individual range and pasture sites (fields) are representative of particular plant communities and on a larger scale, an ecosystem, it is necessary to focus on all the attributes that are connected to "overall health." During the past three decades, the rangeland health assessment tool (Pellant et al. 2020) has a well-documented history and has continually been refined. Soil health is a major theme of IIRH and DIPH and is assessed by indicator variables (see subpart E for examples of assessments). More recently, the concepts of IIRH (Pellant et al. 2020) have been applied to pastureland (DIPH; Spaeth 2020). Both assessment tools have been sufficiently tested to provide a basis for assessing three overall attributes (biotic integrity, soil and site stability, and hydrologic function) of the plant community and ecological sites. The Pasture Condition Scoresheet is useful for a quick overall assessment of overall pasture health but does not provide a specific breakdown of the three attributes associated with IIRH and DIPH.

B. In conclusion, in determining range and pasture health with supporting soil health principles, three protocols are available: IIRH, PCSS, and DIPH. For IIRH, PCSS, and DIPH, quantitative data can be used in determining indicators (see discussion in this document and subparts B and E).

645.1111 References

A. Altesor, A.I., E. Di Landro, H. May, and E. Ezcurra. 1998. Long-term species change in a Uruguayan grassland. Journal of Vegetation Science 9: 173–180.

B. Albrecht, W.A. 1938. Loss of soil organic matter and its restoration. In Soils and Men, Yearbook of agriculture 1938. eds. H.G. Knight et al. p. 347–376. Washington D.C. United States Department of Agriculture.

C. Archer, S., T.W. Boutton, and K.A. Hibbard. 2001. Trees in grasslands: biogeochemical consequences of woody plant expansion. In Global biogeochemical cycles in the climate systems. ed. D. Schulze, M. Heimann, S. Harrison, E. Holland, J. Lloyd, I.C. Prentice, and D. Schimel, p. 115–130. San Diego, California: Academic Press.

D. Archer, S.R., K.W. Davies, T.E. Fulbright, K.C. McDaniel, B.P. Wilcox, and K.I. Predick. 2011. Brush management as a rangeland conservation strategy: a critical evaluation. In Conservation benefits of rangeland practices: assessment, recommendations, and knowledge gaps, ed. D.D. Briske, 105–170. Washington, D.C: USDA-NRCS.

E. Bakker, M.M., G. Grovers, and D.A. Rounsevell. 2004. The crop productivity-erosion relationship: An analysis based on experimental work. Catena 57: 55–76.

F. Baldock, J.A., and J.O. Skjemstad. 1999. Organic soil C/soil organic matter. In Soil Analysis: An interpretation manual. eds. K.I. Prveril, L.A. Sparrow. L.A., and D.J. Reuter. CSIRO Publishing: Collingwood, Victoria 159–170.

G. Balogh, J., S. Fóti, A. Juhász, S. Czóbel, Z. Nagy, and Z. Tuba. 2005a. Seasonal CO2- exchange variations of a temperate semi-desert grassland in Hungary. Photosynthetica 43: 107–110.

H. Balogh, J., S. Czóbel, S. Fóti, Z. Nagy, O. Szirmai, E. Péli, and Z. Tuba. 2005b. The influence of drought on carbon balance in loess grassland. Cereal Research Communications 33: 149–152.

I. Balota, E.L., A. Colozzi-Filho, D.S. Andrade, and R.P. Dick. 2003. Microbial biomass in soils under and crop rotation systems. Biological Fertility Soils 38: 15–20.

J. Baron, V., S.E. Mapfumo, A.C. Dick, M.A. Naeth, E.K. Okine, and D.S. Chanasyk. 2002. Grazing intensity impacts on pasture carbon and nitrogen flow. Journal of Range Management 55: 535–541.

K. Bauer, A., C.V. Cole, and A.L. Black. 1987. Soil property comparisons in virgin grasslands between grazed and nongrazed management systems. Soil Science Society of America Journal 51.1: 176–182.

L. Batjes, N.H. 1996. Total carbon and nitrogen in the soils of the world. European Journal of Soil Science 47: 151–163.

M. Battin, T.J., S. Luyssaert, L.A. Kaplan, A.K. Aufdenkampe, A. Richter and L.J. Tranvik. 2009. Nature Geoscience 2: 598–600.

N. Bauer, A., C.V. Cole, and A.L. Black. 1987. Soil property comparisons in virgin grasslands between grazed and nongrazed management systems. Soil Science Society of America Journal 51: 176–182.

O. Berner, R.A. 1990. Atmospheric carbon dioxide levels over Phanerozoic time. Science 249, 1382.

P. Bot, A., and J. Benites. 2005. The importance of soil organic matter. Key to drought-resistant soil and sustained food production. Rome, Italy, FAO Land and Plant Nutrition Management Service. Food and Agriculture Organization of the United Nations.

Q. Briske, D.D., J.D. Derner, D.G. Milchunas, and K.W. Tate. 2011. An evidence-based assessment of prescribed grazing practices. Conservation benefits of rangeland practices: assessment, recommendations, and knowledge gaps. Washington, DC, USA: USDA-NRCS, pp. 21–74.

R. Brown, J.R., and J.E. Herrick. 2016. Making soil health a part of rangeland management. Journal of Soil and Water Conservation 71: 55A–60A.

S. Bruhwiler, L., A.M. Michalak, R. Birdsey, J.B. Fisher, R.A. Houghton, D.N. Huntzinger, and J.B. Miller. 2018. Chapter 1: Overview of the global carbon cycle. In Second State of the Carbon Cycle Report (SOCCR2): A Sustained Assessment Report [Cavallaro, N., G. Shrestha, R. Birdsey, M.A. Mayes, R.G. Najjar, S.C. Reed, P. Romero-Lankao, and Z. Zhu (eds.)]. U.S. Global Change Research Program, Washington, DC, USA. pp. 42–70, https://doi.org/10.7930/SOCCR2.2018.Ch1.

T. Bunting, B.T. 1965. The geography of soil. Chicago: Aldine.

U. Burges, A. 1963. The microbiology of a podzol profile. In Soil organisms. eds. J. Doeksen and J. van der Drift, 151–157. Amsterdam: North Holland Publ. Co.

V. Burke, I.C., W.K. Lauenroth, M.A. Vinton, P.B. Hook, R.H. Kelly, H.E. Epstein, M.R. Aguiar, M.D. Robles, M.O Aguilera, K.L. Murphy, and R.A. Gill. 1998. Plant-soil interactions in temperate grasslands. In Plant-induced soil changes: Processes and feedbacks (pp. 121–143). Springer, Dordrecht.

W. Butzer, K.W. 1974. Accelerated soil erosion: A problem of man-land relationships. In Perspectives on environment, ed. R. Manners and M.W. Mikesell. Washington D.C.: Association of American Geographers, Pub No. 13.

X. Buyanovsky, G.A., and G.H. Wagner. 1997. Crop residue input to soil organic matter in Sanborn Field. In Soil organic matter in in temperate ecosystems: Long-term experiments in North America. ed. E.A. Paul p. 73–84. Boca Raton, Florida: CRC Press.

Y. Campbell, C.A., and W. Souster. 1982. Loss of organic matter and potentially mineralizable N from Saskatchewan soils due to cropping. Canadian Journal of Soil Science 62: 651–656.

Z. Canadell, J.G., M.U.F. Kirschbaum, W.A. Kurz, M.J. Sanz, B. Schlamadinger, and Y. Yamagata. 2007a. Factoring out natural and indirect human effects on terrestrial carbon sources and sinks. Environmental Science and Policy 10: 370–384.

AA. Canadell, J.G., D.E. Pataki, R. Gifford, R.A. Houghton, Y. Luo, M.R. Raupach, and W. Steffen. 2007b. Saturation of the terrestrial carbon sink. In Terrestrial ecosystems in a changing world p. 59–78. Berlin: Springer.

AB. Caruso, T., F.T. De Vries, R.D. Bardgett, and J. Lehmann. 2018. Soil organic carbon dynamics matching ecological equilibrium theory. Ecology and Evolution 8: 11169–11178.

AC. Conant, R.T., K. Paustian, and E.T. Elliott. 2001. Grassland management and conversion into grassland: effects on soil carbon. Ecological applications 11.2: 343–355.

AD. Cole, C.V., J. Duxbury, J. Freney, O. Heinemeyer, K. Minami, A. Mosier, K. Paustian, N. Rosenberg, N. Sampson, D. Sauerbeck, and Q. Zhao. 1997. Global estimates of potential mitigation of greenhouse gas emissions by agriculture. Nutrient cycling in agroecosystems 49: 221–228.

AE. Cougnon M., T. De Swaef, P. Lootens, J. Baert, P. De Frenne, and R. Shahidi. 2017. In situ quantification of forage grass root biomass, distribution, and diameter classes under two N fertilisation rates. Plant Soil 411: 409–422.

AF. Crews, T.E., and B.E. Rumsey. 2014. What agriculture can learn from native ecosystems in building soil organic matter: A review. Sustainability 9.4: 1–18.

AG. Davidson, E.A., and I.L. Ackerman. 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20: 161–193.

AH. Derner, J.D., D.D. Briske, and T.W. Boutton. 1997. Does grazing mediate soil carbon and nitrogen accumulation beneath C4, perennial grasses along an environmental gradient? Plant and Soil 191: 47–156.

AI. Derner, J., T. Boutton, and D. Briske. 2006. Grazing and ecosystem carbon storage in the North American Great Plains. Plant and Soil 280: 77–90.

AJ. Derner, J.D., and G.E. Schuman. 2007. Carbon sequestration and rangelands: a synthesis of land management and precipitation effects. Journal of Soil and Water Conservation 62: 7–85.

AK. Donigian, A.S., T.O. Barnwell, R.B. Jackson, A.S. Patwardhan, and K.B. Weinrich. 1994. Assessment of alternative management practices and policies affecting soil carbon in agroecosystems of the central United States (No. PB-94-189420/XAB). AQUA TERRA Consultants, Mountain View, California.

AL. Dzurec, R.S., T.W. Boutton, M.M. Caldwell, and B.N. Smith. 1985. Carbon isotope ratios of soil organic matter and their use in assessing community composition changes in Curlew Valley, Utah. Oecologia 66: 17–24.

AM. Eswaran, H., P.F. Reich, J.M. Kimble, F.H. Beinroth, E. Padamnabhan, and P. Moncharoen. 2000. Global carbon stocks. In Global climate change and pedogenic carbonates, eds. R. Lal, J.M. Kimble, H. Eswaran, and B.A. Stewart, 15–25. Boca Raton, FL: CRC/Lewis Publishers.

AN. Falkowski, P., R.J. Scholes, E.E.A. Boyle, E.E.A., J. Canadell, D. Canfield, J. Elser, N. Gruber, K. Hibbard, P. Högberg, S. Linder, and R.T. Mackenzie. 2000. The global carbon cycle: a test of our knowledge of earth as a system. Science 290: 291–296.

AO. Foley, J.A. 2005. Global consequences of land use. Science 309: 570–574.

AP. Food and Agriculture Organization (FAO). 2008. Soil health definitions. http://www.fao.org/agriculture/crops/thematic-sitemap/theme/spi/soil-biodiversity/the-nature-of-soil/what-is-a-healthy-soil/en/

AQ. Food and Agriculture Organization (FAO). 2013. http://www.fao.org/news/story/en/item/209369/icode/

AR. Flach, K.W., T.O. Barnwell Jr, and P. Crosson. 1997. Impacts of agriculture on atmospheric carbon dioxide. In Soil organic matter in temperate agroecosystems. Long-term experiments in North America eds. E.A. Paul, K. Paustian, E.T. Elliot, and C.V. Cole. pp. 3–13. Boca Raton, Florida.

AS. Frank, A.B., D.I. Tanaka, I. Hoffmann, and R.F. Follett. 1995. Soil carbon and nitrogen of the Northern Great Plains grasslands as influenced by long-term grazing. Journal of Range Management 48: 470–474.

AT. Franzluebbers, A.J., J.A. Stuedemann, and S.R. Wilkinson. 2001. Bermudagrass management in the Southern Piedmont USA. I. Soil and surface residue carbon and sulfur. Soil Science Society of America 65: 834–841.

AU. Franzluebbers, A.J. 2005. Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. Soil and Tillage Research 83: 120–147.

AV. Franzluebbers, A.J., and J.A. Stuedemann. 2009. Soil-profile organic carbon and total nitrogen during 12 years of pasture management in the Southern Piedmont USA. Agriculture, Ecosystems & Environment 129: 28–36.

AW. Ghilarov, M.S. 1970. Regularities in Adaptions of Arthropods to the Terrestrial Life. Moscow, Nauka Publishing House.

AX. Giddens, J. 1957. Rate of loss of carbon from Georgia soils. Soil Science Society of America Journal 21: 513–515.

AY. Guilherme M.C., D.D. Myrold, and P.J. Bottomley. 2009. A soil quality index based on the equilibrium between soil organic matter and biochemical properties of undisturbed coniferous forest soils of the Pacific Northwest. Soil Biology and Biochemistry 41: 822–830.

AZ. Haddix, M.L., A.F. Plante, R.T. Conant, J. Six, J.M. Steinweg, K. Magrini-Bair, R.A. Drijber, S.J. Morris, and E.A. Paul. 2011. The role of soil characteristics on temperature sensitivity of soil organic matter. Soil Science Society of America Journal 75: 56–68.

BA. Haider, K.M., and G. Guggenberger. 2005. Organic matter: genesis and formation. In Encyclopedia of the soils in the environment. Vol 3. ed. D. Hillel, 93–101. Oxford, England. Elsevier.

BB. Harpole, W.S., D.L. Potts, and K.N. Suding. 2007. Ecosystem responses to water and nitrogen amendment in a California grassland. Global Change Biology 13: 2341–2348.

BC. Hausenbuiller, R.L. 1978. Soil science principles and practices. Dubuque, Iowa: W.C. Brown Co.

BD. Hewins, D.B., M.P. Lyseng, D.F. Schoderbek, M. Alexander, W.D. Willms, C.N. Carlyle, S.X. Chang, and E.W. Bork. 2018. Grazing and climate effects on soil organic carbon concentration and particle-size association in northern grasslands. Scientific Reports 8 (1): 1336.

BE. Hornbeck, R. 2009. The enduring impact of the American dust bowl: Short and long-run adjustments to environmental catastrophe. Working paper 15605. Cambridge: National Bureau of Economic Research.

BF. Houghton, R.A. 1995. Changes in storage of terrestrial carbon since 1850. In Soils and global change. Eds. R. Lal, J. Kimble, E. Levine, and B.A. Stewart. Boca Raton, Florida: CRC/Lewis Publishers.

BG. Houghton, R.A. 2007. Balancing the Global Carbon Budget. Annual Review of Earth and Planetary Sciences 35: 313–347.

BH. Hudson, H.J., and J. Webster. 1958. Succession of fungi on decaying stems of Agropyron repens. Transactions of the British Mycological Society 41: 165–177.

BI. Ingram. L.J., G.E. Schuman, P.D. Stahl, J.M. Welker. G.E. Vanice, and K. Gunjegunte. 2004. The influence of grazing on microbial activity in a northern mixed-grass prairie. Agronomy Abstracts, American Society of Agronomy. Madison, Wisconsin.

BJ. IPCC, 2007. Climate change 2007: The physical science basis. Summary for policymakers. Intergovernmental Panel on Climate Change, United Nations.

BK. Ito. 2019. Disequilibrium of terrestrial ecosystem CO2 budget caused by disturbance-induced emissions and non-CO2 carbon export flows: a global model assessment. Earth System Dynamics 10: 685–709.

BL. Jenny, H. 1941. Factors of Soil Formation: A System of Quantitative Pedology, NY: McGraw-Hill.

BM. Jenny, H. 1961. Derivation of state factor equations of soils and ecosystems. Soil Science Society of America Journal 25: 385–388.

BN. Jobbagy, E.G., and R.B. Jackson. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications 10: 423–436.

BO. Johnson, C., G. Albrecht, Q. Ketterings, J. Beckman, and K. Stockin. 2005. Nitrogen basics-the nitrogen cycle. Cornell University fact sheet 2. Cornell: Cornell University Cooperative Extension.

BP. Johnston, A., J.F. Dormaarand, and S. Smoliak. 1971. Long-term grazing effects on fescue grassland soils. Journal of Range Management 185–188.

BQ. Johnston, A.E., Poulton, P.R. and Coleman, K., 2009. Soil organic matter: its importance in sustainable agriculture and carbon dioxide fluxes. Advances in agronomy 101: 1–57.

BR. Johnston, J. 2011. Soil organic matter changes towards and equilibrium level appropriate to the soil and cropping system. Better Crops 95, 2pp.

BS. Kirchman, D.L., ed. 2010. Microbial ecology of the oceans 36. New York: Wiley and Sons.

BT. Kucera C.L., L.C. Dahlman, and M.R. Koelling. 1967. Total net productivity and turnover on an energy basis for tallgrass prairie. Ecology 48 pp.

BU. Kucharik, C.J. 2007. Impact of prairie age and soil order on carbon and nitrogen sequestration. Soil Science Society of America Journal 71: 430–441.

BV. Lal, R. 1998. Soil erosion impact on agronomic productivity and environment quality. Critical reviews in plant sciences 17: 319–464.

BW. Lal, R. 2003. Soil erosion and the global carbon budget. Environment International 29: 437–450.

BX. Lal, R. 2008. Carbon sequestration. Philosophical Transactions of the Royal Society 363: 815–831.

BY. Lal, R. 2010. Managing Soils and Ecosystems for Mitigating Anthropogenic Carbon Emissions and Advancing Global Food Security. BioScience 60: 708–721.

BZ. Lal, R. 2018. Accelerated soil erosion as a source of atmospheric CO_2. Soil and Tillage Research 188: 35–40.

CA. Larcher, W. 1983. Physiological Plant Ecology. Berlin, Springer-Verlag.

CB. Lewis, S.L. 2005. Tropical forests and atmospheric CO2: Current conditions and future scenarios. In H.J. Schellnhuber ed. Avoiding Dangerous Climate Change. p. 147–153, Cambridge University Press.

CC. Liang, C., J.P. Schimel, and J.D. Jastrow. 2017. The importance of anabolism in microbial control over soil carbon storage. Nature Microbiology 2: 1–6.

CD. Liebig, M.A., H.A. Johnson, J.D. Hanson, and A.B. Frank. 2005. Soil carbon under switchgrass stands and cultivated cropland. Biomass and bioenergy 28: 347–354.

CE. Loceya, K.J., and J.T. Lennona. 2016. Scaling laws predict global microbial diversity. PNAS DOI: 10.1073/pnas.1521291113.

CF. Luo, Z., E. Wang, and O.J. Sun. 2010. Can no-tillage stimulate carbon sequestration in agricultural soils? A meta-analysis of paired experiments. Agriculture, ecosystems and environment 139: 224–231.

CG. Manley, J.T., G.E. Schuman, J.D. Reeder, and R.H. Hart. 1995. Rangeland soil carbon and nitrogen responses to grazing. Journal of Soil and Water Conservation 50: 294–298.

CH. Mann, L.K. 1986. Changes in soil C storage after cultivation. Soil Science 142: 279–288.

CI. Matamala, R., J.D. Jastrow, R.M. Miller, and C.T. Garten. 2008. Temporal changes in C and N stocks of restored prairie: implications for C sequestration strategies. Ecological Applications 18: 1470–1488.

CJ. McLauchlan, K. 2006. The nature and longevity of agricultural impacts on soil carbon and nutrients: a review. Ecosystems 9: 1364–1382.

CK. McNaughton, S.J. 1985. Ecology of a grazing ecosystem: the Serengeti. Ecological Monographs 55: 259–294.

CL. Milchunas, D.G., and W.K. Lauenroth. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. Ecological Monographs 63: 327–366.

CM. Miltner, A., P. Bombach, B. Schmidt-Brücken, and M. Kästner. 2012. SOM genesis: microbial biomass as a significant source. Biogeochemistry 111: 41–55.

CN. Minderman, G. 1968. Addition, decomposition and accumulation of organic matter in forests. Journal of Ecology 56: 355–362.

CO. Moebius-Clune, B.N., D.J. Moebius-Clune, B.K. Gugino, O.J. Idowu, R.R. Schindelbeck, A.J. Ristow, H.M. van Es, J.E. Thies, H.A. Shayler, M.B. McBride, K.S.M Kurtz, D.W. Wolfe, and G.S. Abawi. 2016. Comprehensive Assessment of Soil Health – The Cornell Framework, Edition 3.2, Cornell University, Geneva, NY.

CP. Ogle, S.M., F.J. Breidt, and K. Paustian. 2005. Agricultural management impacts on soil organic carbon storage under moist and dry climatic conditions of temperate and tropical regions. Biogeochemistry 72: 87–121.

CQ. Pacala, S., and R. Socolow. 2004. Stabilization wedges: solving the climate problem for the next 50 years with current technologies. Science 305: 968–972.

CR. Pan, Y., R.A. Birdsey, J. Fang, R. Houghton, P.E. Kauppi, W.A. Kurz, O.L. Phillips, A. Shvidenko, S.L. Lewis, J.G. Canadell, and P. Ciais. 2011. A large and persistent carbon sink in the world's forests. Science 333: 988–993.

CS. Pausch J., and Y. Kuzyakov. 2018. Carbon input by roots into the soil: Quantification of rhizodeposition from root to ecosystem scale. Global Change Biology 24: 1–12.

CT. Paustian, K, H.P. Collins, and E.A. Paul. 1997. Management controls on soil carbon. In Soil organic matter in temperate agroecosystems. eds. E.A. Paul, E.T. Elliott, K. Paustian, and C.V. Cole, 15–49. New York: CRC Press.

CU. Pellant, M., P.L. Shaver, D.A. Pyke, J.E. Herrick, N. Lepak, G. Riegel, E. Kachergis, B.A. Newingham, D. Toledo, and F.E. Busby. 2020. Interpreting Indicators of Rangeland Health, Version

5. Tech Ref 1734-6. U.S. Department of the Interior, Bureau of Land Management, National Operations Center, Denver, CO.

CV. Persson, T., E. Baath, M. Clarholm, H. Lundkvist, B.E. Söderström, and B. Sohlenius. 1980. Trophic Structure, Biomass Dynamics and Carbon Metabolism of Soil Organisms in a Scots Pine Forest. Ecological Bulletins 32: 419–459.

CW. Pineiro, G., J.M. Paruelo, and M. Oesterheld. 2006. Potential long-term impacts of livestock introduction on carbon and nitrogen cycling in grasslands of southern South America. Global Change Biology 12: 1267–1284.

CX. Pineiro, G., J.M. Paruelo, E.G. Jobbagy, R.D. Jackson, and M. Oesterheld. 2009. Grazing effects on belowground C and N stocks along a network of cattle exclosures in temperate and subtropical grasslands of South America. Global Biogeochemical Cycles 23: GB2003.

CY. Pineiro, G. J.M. Paruelo, M. Oesterheld, and E.G. Jobbagy. 2010. Pathways of grazing effects on soil organic carbon and nitrogen. Rangeland Ecology and Management 63: 109–119.

CZ. Potter, K.N., H.A. Tobert, H.B. Johnson, and C.R. Tischler. 1999. Carbon storage after long-term grass establishment on degraded soils. Soil Science 164: 718–725.

DA. Povirk, K. 1999. Carbon and nitrogen dynamics of an alpine grassland: effects of grazing history and experimental warming on CO2 flux and soil properties. MS thesis, Laramie, Wyoming: Dept. Renewable Resources.

DB. Printz, J.L., D. Toledo, and S.C. Boltz. 2014. Rangeland health assessment: The key to understanding and assessing rangeland soil health in the Northern Great Plains. Journal of Soil and Water Conservation 69: 73A–77A.

DC. Rasmussen, P.E., and H.P. Collins. 1991. Long-term impacts of tillage, fertilizer and crop residue on soil organic matter in temperate semi-arid regions. Advances Agronomy 45: 93–134.

DD. Rasmussen, P.E., and W.J. Parton. 1994. Long-term effects of residue management in wheat-fallow. I. Inputs, yield and soil organic matter. Soil Science Society America Journal 58: 523–530.

DE. Ruhe, R.V., and R.B. Daniels. 1965. Landscape erosion-geologic and historic. Journal of Soil and Water Conservation 20: 52–57.

DF. Ruddiman, W.F. 2003. The anthropogenic greenhouse era began thousands of years ago. Climatic Change 61: 261–293.

DG. Sainju U.M., B.L. Allen, A.W. Lenssen, and R.P. Ghimire. 2017. Root biomass, root/shoot ratio, and soil water content under perennial grasses with different nitrogen rates. Field Crops Research 210: 183–191.

DH. Sala, O. E. 1988. The effect of herbivory on vegetation structure. In: M.J.A. Werger, P.J.M. van der Aart, H.J. During, and J.T.A. Verboeven (eds.). Plant form and vegetation structure. The Hague, Netherlands: SPB Academic Publishing. p. 317–330.

DI. Sawyer, J.E. and A.P. Mallarino. 2007. Carbon and nitrogen cycling with corn biomass harvest. Integrated Crop Management News.

DJ. SARE. 2012. Why soil organic matter is so important. Sustainable agriculture research and education. https://www.sare.org/Learning-Center/Books/Building-Soils-for-Better-Crops-3rd-Edition/Text-Version/Organic-Matter-What-It-Is-and-Why-It-s-So-Important/Why-Soil-Organic-Matter-Is-So-Important.

DK. Schimel, D.S. 1995. Terrestrial ecosystems and the carbon cycle. Global change biology 1: 77–91.

DL. Schlesinger, W.H. 1977. Carbon Balance in Terrestrial Detritus. Annual Review of Ecology and Systematics 8: 51–81.

DM. Schlesinger, W.H. 1997. Biogeochemistry: An Analysis of Global Change. 2nd edition. San Diego: Academic Press 558 p.

DN. Schuman, G.E., J.D. Reeder, J.T., Manley, R.H. Hart, and W.A. Manley. 1999. Impact of grazing management on the carbon and nitrogen balance of a mixed-grass rangeland. Ecological applications 9: 65–71.

DO. Schuman, G.E., H.H. Janzen, and J.E. Herrick. 2002. Soil carbon dynamics and potential carbon sequestration by rangelands. Environmental Pollution 116: 391–396.

DP. Schuman, G.E., L.I. Ingram, P.D. Stahl, and G.E. Vance. 2005. Dynamics of long-term carbon sequestration on rangelands in the western U.S.A. In XX International Grassland Congress. 26 June 1 July 2005. Dublin, Ireland. Eds. E.P. O'Mara, R.J. Wilkins, L. Mannetje, D.K. Lovett, P.A.M. Rogers and T.M. Boland. 590. The Netherlands, Wagenmen Academic Publishers.

DQ. Schuman, G.E. and Derner, J.D., 2004, June. Carbon sequestration by rangelands: Management Effects and potential. In CDROM Proceedings of the Western Regional Cooperative Soil Survey Conference (pp. 13–17).

DR. Scurlock, J.M.O., and D.O. Hall. 1998. The global carbon sink: a grassland perspective. Global Change Biology 4: 229–233.

DS. Sheley, R.L., J.J. James, M.J. Rinella, D. Blumenthal, and J.M. DiTomaso. 2011. Invasive plant management on anticipated conservation benefits: a scientific assessment. In Conservation Benefits of Rangeland Practices: Assessment, Recommendations, and Knowledge Gaps, ed, D.D. Briske, 291–336. Washington, DC: USDA Natural Resources Conservation Service.

DT. Sierra, C., S. Malghani, and H.W. Loescher. 2017. Interactions among temperature, moisture, and oxygen concentrations in controlling decomposition rates in a boreal forest soil. Biogeosciences 14: 703–710.

DU. Six, J., S.D. Frey, R.K. Thiet, and K.M. Batten. 2006. Bacterial and fungal contributions to carbon sequestration in agroecosystems. Soil Science Society of America Journal 70: 555–569.

DV. Smoliak, S. 1986. Influence of climatic conditions on production of Stipa-Bouteloua prairie over a 50-year period. Journal of Range Management 39: 100–103.

DW. Soloman 2007. Carbon dioxide storage: Geological security and environmental issues–Case study on the sleipner gas field in Norway. Bellona report 128.

DX. Soussana, J.F., P. Poiseau, N. Vuichard, E. Ceschia, J. Balesdent, T. Chevallier, and D. Arrouays. 2004. Carbon cycling and sequestration opportunities in temperate grasslands. Soil use and management 20: 219–230.

DY. Spaeth, K.E. 2020. Soil Health on the Farm, Ranch, and in the Garden. Cham, Switzerland, Springer International Publishing.

DZ. Statista, 2018. Statista, https://www.statista.com/statistics/.

EA. Stetler, L.D., R. Benton, and M. Weiler. 2011. Erosion rates from Badlands National Park. In International Symposium on Erosion and Landscape Evolution (ISELE), 18–21 September 2011, Anchorage, Alaska (p. 26). American Society of Agricultural and Biological Engineers.

EB. Stirling, G., H. Helen, T. Pattison, and M. Stirling. 2016. Introduction: Soil health, soil biology and sustainable agriculture and evidence-based information Soil health. In: Soil Biology, Soil borne diseases and Sustainable Agriculture, 1st ed. ed. P. Storer, Queensland Australia: Csiro publishing p. 1–5.

EC. Stoffer, P.W., 2003. Geology of Badlands National Park: a preliminary report (pp. 1–63). US Department of the Interior, US Geological Survey.

ED. Summerfield, M.A., and N.J. Hulton. 1994. Natural controls of fluvial denudation rates in major world drainage basins: Journal of Geophysical Research 99: 13,871–13,883.

EE. Thurow, T.L. 1991. Hydrology and erosion. p. 141–159. In: R.K. Heitschmidt and J.W. Stuth (eds.). Grazing management: an ecological perspective. Portland, Oregon: Timber Press.

EF. Trasar-Cepeda, C., C. Leiros, F. Gil-Sotres, and S. Seoane. 1998. Towards a biochemical quality index for soils: an expression relating several biological and biochemical properties. Biology and Fertility of Soils 26: 100–106.

EG. Trimble, S. W. 1974. Man-induced soil erosion on the southern Piedmont. Soil Conservation Society of America.

EH. Trumbore, S.E. 2000. Age of soil organic matter and soil respiration: radiocarbon constraints on belowground C dynamics. Ecological Applications. 10: 399–411.

EI. USDA-NRCS. 2014. https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_051573.pdf

EJ. USDA-NRCS. 2016. PA#2205, 2017. Soil Planner. Washington, DC.

EK. USDA-NRCS. 2019. Soil health. https://www.nrcs.usda.gov/wps/portal/nrcs/main/soils/health/

EL. USDA-NRCS. 2000. U.S. Department of Agriculture Natural Resources Division Resources Assessment Division Washington, DC.

EM. USDA-NRCS. 2021a. https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/health/assessment/?cid=nrcs142p2_053873

EN. USDA-NRCS. 2021b. https://www.nrcs.usda.gov/wps/portal/nrcs/main/soils/health/

EO. VandenBygaart, A.J., E.G. Gregorich, and D.A. Angers. 2003. Influence of agricultural management on soil organic carbon: A compendium and assessment of Canadian studies. Canadian Journal of Soil Science 83: 363–380.

EP. Waksman, S.A. 1936. Humus origin, chemical composition, and importance in nature. Baltimore, Maryland: Williams and Wilkins Co.

EQ. Walker, P.H. 1966. Postglacial environments in relation to landscape and soils on the Cary Drift (Research bulletin 549) (p. 549). Ames, IA: Agriculture and Home Economics Experiment Station, Iowa State University.

ER. Weaver, J.E., and R.W. Darland. 1947. A method of measuring vigor of range grasses. Ecology 28: 146–162.

ES. Wedin, D.A., L.L Tieszen, B. Dewey, and J. Pastor. 1995. Carbon isotope dynamics during grass decomposition and soil organic matter formation. Ecology 76: 1383–1392.

ET. Wiedinmyer, C., and J.C. Neff. 2007. Estimates of CO2 from fires in the United States: Implications for carbon management. Carbon Balance and Management 2: 12p.

EU. Wienhold, B.J., J.R. Hendrickson, and J.F. Karn. 2001. Pasture management influences on soil properties in the northern Great Plains. Journal of Soil and Water Conservation 56: 27–31.

EV. Weil, R.R., and N.C. Brady. 2017. The nature and properties of soils. NY, Pearson.

EW. Whittaker, R.H. 1975. Communities and ecosystems 2nd ed. New York: McMillan Publ. Co.

EX. Wilkinson, B.H., and B.J. McElroy. 2007. The impact of humans on continental erosion and sedimentation. Geological Society of America Bulletin 119: 140–56.

EY. Xiao, C. 2017. Soil organic matter storage (sequestration) principles and management. Potential role for recycled organic materials in agricultural soils of Washington State Dept. Of Ecology Pub. No. 15–07–005.

EZ. Zhang, Y., T. YanHong, J. Jie, and Y. YongHai. 2007. Characterizing the dynamics of soil organic carbon in grasslands on the Qinghai-Tibetan Plateau. Science in China Series D: Earth Sciences 50: 113–120.