CHAPTER

Soil carbon management

11

Luke Nave[a,*], Erika Marín-Spiotta[b], Todd Ontl[c], Matt Peters[d], Chris Swanston[c]

[a]*Biological Station and Department of Ecology and Evolutionary Biology, University of Michigan, Pellston, MI, United States,*

[b]*Department of Geography, University of Wisconsin, Madison, WI, United States,*

[c]*Northern Research Station, U.S. Department of Agriculture-Forest Service, Houghton, MI, United States,*

[d]*Northern Research Station, U.S. Department of Agriculture-Forest Service, Delaware, OH, United States*

[*]*Corresponding author.*

ABSTRACT

Forest soil organic carbon (SOC) is one of the largest terrestrial C pools, and it is not static. Soils are inherently dynamic; furthermore, global change and management can alter how much C is stored in a forest soil and how it is distributed across depths or functionally distinct pools. Over management-relevant timescales, forest soil C varies more across space—due to fundamental factors such as parent material, climate, and vegetation—than it does over time, except in cases of abuse (which deplete SOC). In this review, we discuss the stocks and management of forest SOC through the lens of 13 Global Ecological Zones (UN FAO climate-vegetation units) and 17 major soil types (IUSS WRB Reference Soil Groups). We use these spatial units to constrain generalizations derived from a wide-ranging literature survey and qualitative synthesis aimed at four objectives. For each ecological zone, we: (1) identify key SOC vulnerabilities in the context of global change and management, (2) discuss management strategies for minimizing SOC losses associated with those vulnerabilities, and (3) suggest opportunities for increasing SOC through management. As our fourth objective, within each ecological zone, we contextualize SOC management considerations according to the unique properties possessed by its most widespread soils, in order to add meaningful detail to this global review, anticipate where exceptions will occur, and provide a tractable basis for management intended to affect SOC. Key generalizations are: (1) SOC pools at or near the surface are vulnerable to disturbances that alter the processes responsible for maintenance of those pools; management that restores or maintains those processes can often reverse SOC losses over decadal timescales. (2) SOC in subsoils is typically less vulnerable to short-term losses, but the factors that promote this stability are difficult to manipulate in a positive direction through management. (3) SOC is one integral component of forest ecosystems, and as such is often not managed directly but through its interactions with other parts, such as the vegetation, according to constraints imparted by fundamental bottom-up factors (parent material and soil characteristics).

Why is forest soil carbon important?

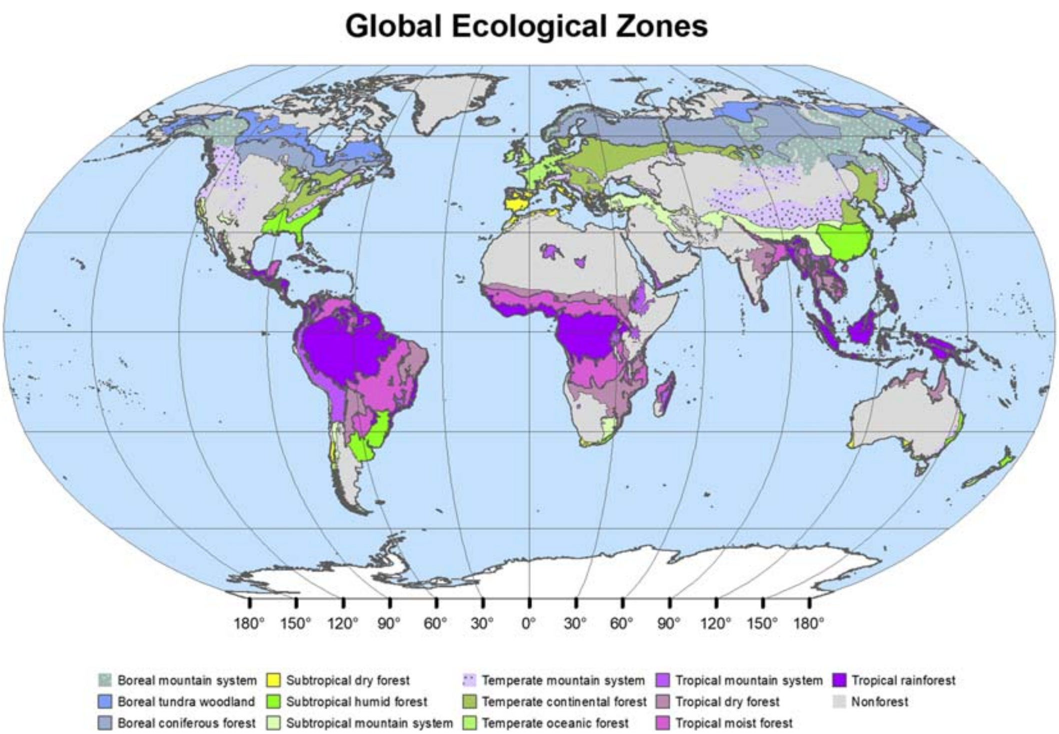
Soils comprise the largest terrestrial carbon (C) pool, holding ∼1500 Pg C. In forests, soils typically contain more than half, and often more than three-quarters, of the total C stock. Carbon held in soils tends to be much older (i.e., cycles more slowly) than C in other ecosystem pools, but it is not inert: C inputs to (e.g., dead roots, fungal and bacterial cells and residues, arthropod exoskeletons, leaf litter) and outputs from (e.g., via leaching, decomposition, and emission by fire) represent very large, nearly closely balanced fluxes that cycle globally significant quantities of C. Thus, C held in soils is critically important for its feedbacks to atmospheric CO_2 pollution and climate change, as well as a myriad of

Global Change and Forest Soils. https://doi.org/10.1016/B978-0-444-63998-1.00011-2

other forest ecosystem services tied to the soil organic matter (SOM) of which C is the principal constituent. Chief among these services are the retention and provision of plant-limiting nutrients, immobilization of toxic substances, mediation of hydrological processes that influence water quality and quantity, and provision of astounding biologic diversity that is the source of many human medicines. All of these ecosystem services make soil C important; in this chapter, we focus primarily on soil C as a foundational part of forest ecosystems, their internal functions and sensitivity to management and global change.

Scale and scope of this chapter

In order to be effective, any global assessment of forest ecosystems requires a spatial framework that balances broad generalization with meaningful specificity. In this chapter, we attempt to strike this balance by using a high-level ecoregional framework developed by the United Nations Food and Agricultural Organization: the Global Ecological Zone (GEZ) framework (Map 1; FAO, 2013). The GEZ framework uses a hierarchical approach to first stratify Earth by climate domains (tropical, subtropical, temperate, boreal, polar) and, within each domain, into GEZs defined by regional differences in



Map1. Global Ecological Zones (GEZs; FAO 2013). In this chapter, we place forest SOC management in a three-level spatial framework by exploring differences among domains (tropical, subtropical, temperate, and boreal), the 3-4 GEZs within each domain, and the most extensive Reference Soil Groups (RSGs) within each GEZ.

climate and/or vegetation composition and structure. The GEZ system was developed with expert input by Earth scientists from institutions worldwide, and while its exact borders sometimes differ from other large-scale spatial frameworks (e.g., regional or national maps), it has several advantages here. Most importantly, it is hierarchical in relation to increasingly localized ecoregional frameworks used in many countries. Our approach to discussing soil C management is similarly hierarchical: first, we briefly describe global-to-domain level considerations, and then move into detail by surveying literature from the individual GEZs within each domain. In some GEZs, we follow this hierarchical framework down to finer ecoregional levels, such as topographically or climatically distinct regional landscapes. However, because this chapter pertains as much to *soils* as it does to *forests*, the most specific spatial units that we consistently discuss are the Reference Soil Groups (RSGs; Table 11.1 and IUSS WRB Working Group, 2006) present in each GEZ. Similar to GEZs, RSGs are a working classification created by an international body. The 17 RSGs discussed in this chapter (32 are defined) are recognizable in the field based on diagnostic horizons, properties, and soil materials. Further, the horizons, properties and materials that define RSGs are critical factors influencing the amount, distribution, and forms of SOC present in a soil. As a result, RSGs can explain why the same management activity may produce different results across RSGs, or predict where existing variation in SOC stocks will likely be insensitive to management. Through this combination of GEZs (high-level climate and vegetation units) and RSGs (fundamental "soil types"), we work hierarchically downwards across the scales at which we can confidently present generalizations about forest soil C to researchers, managers, and policy makers concerned with specific geographies.

Having considered issues of scale, we next address the question of scope: what is forest soil C? In reality, soil C consists of a continuum of compounds differing in their origins, chemical and physical properties, cycling rates, and vulnerabilities to management and global change. We simplify this complexity, as we do with spatial scale, by using a categorical approach: we discuss *pools* of soil C, differentiated according to distinct chemical and physical properties. At the highest level of this distinction, we distinguish two pools: soil inorganic C versus soil organic C (SIC and SOC, respectively). Soil inorganic C is not usually considered a pool of direct interest to global change or management, is not inert, and represents a significant fraction of the C in some soils (Lal and Kimble, 2000; Sanderman, 2012; Tan et al., 2014). As the "skeleton" of the myriad organic compounds in soil, SOC is intimately associated with elements that are predominantly present in soil in organic forms, such as nitrogen (N) and phosphorus (P), as well as elements comprising the mineral matrix, such as silicon (Si), aluminum (Al), and iron (Fe) (Heckman et al., 2018; von Luetzow et al., 2008; Rasmussen et al., 2018). Researchers have long sought to physically and chemically fractionate SOC (or the SOM of which it is a major constituent) into functionally distinct pools that can be meaningfully interpreted, modeled, or used to assess effects of management and global change on soils. For years, the dominant paradigm that SOM consisted of chemically recalcitrant residues formed during the decay of plant litter guided these fractionation procedures (Adair et al., 2008; Sutton and Sposito, 2005). More recent research diverges from this top-down, litter decomposition-based view, arguing that the integration of microbial processes and organo-mineral interactions control SOC/SOM storage and turnover (Kallenbach et al., 2016; Lehmann and Kleber, 2015; Sollins et al., 1996). This paradigm shift is important because it drives, and is driven by, the methods used to measure SOC. In turn, our ability to detect SOC changes due to global change and management evolves with conceptual models and methods. Ultimately, it is important to remember that forest soil C is defined by operational methods, and it is the consistent implementation of standard methods that allows us to measure and report a common commodity in the context of global change and management.

Table 11.1 Key properties of Reference Soil Groups (RSGs) detailed in this chapter.

RSG	Description	Parent materials	Physiography
Acrisols	Subsoil enriched in low-activity clay, low in base cations and base saturation, high in Al	Acidic rocks and residuum; strongly weathered clays	Old, undulating to hilly landforms in warm, wet climates
Albeluvisols	Albic horizon protruding irregularly into clay-enriched subsoil	Glacial till; lacustrine, fluvial, or eolian deposits	Young, level to undulating landforms in temperate to boreal climates
Alisols	Subsoil enriched in high-activity clay, low base saturation, occasional albic horizon	Basic rocks, residuum, or unconsolidated materials	Hilly to undulating topography in wet, warm climates
Andosols	Very deep, porous soils with abundant, stable complexes of organic matter-metal hydroxides	Acidic to basic volcanic ejecta or silicon-rich materials	Undulating to mountainous topography in humid climates from tropics to arctic
Cambisols	Little altered from parent materials; subsoil has weak structural/physical/chemical development	Medium to fine-textured materials from any rock type	Level to mountainous terrain; all climates
Cryosols	Frost-affected soils with or without frozen water	Glacial till; eolian, alluvial, colluvial materials	Level to mountainous terrain in boreal to arctic climates
Ferralsols	Highly weathered soils high in Al/Fe hydroxides, modest contents of low-activity clays	Basic rocks; unconsolidated materials rich in silicon	Old level to undulating landforms in very wet, very warm climates
Histosols	Soil formed in organic matter	Organic (peat to muck) materials derived mostly from plant detritus	Level or depressional landforms, high water table or precipitation, from tropics to arctic
Leptosols	Thin, very rocky soils	Rock or unconsolidated coarse materials	Dissected topography, eroding areas; all climates
Lixisols	Subsoil enriched in low-activity clays, high base saturation	Unconsolidated, strongly weathered, fine-textured materials	Old erosional or depositional landforms; warm, seasonally dry climate
Luvisols	Subsoil enriched in high-activity clay, high base saturation	Glacial till; eolian, alluvial, colluvial materials	Level to undulating terrain; cool to warm climates
Kastanozems	Organic-rich topsoil over clay- or carbonate-rich subsoil (grassland-forest transitional soils)	Eolian deposits or other unconsolidated materials	Level to mountainous terrain in dry climates with wide temperature extremes
Nitisols	Deep clay soils with strong structure, high iron	Fine-textured weathering products of neutral to basic rocks	Level to hilly landforms in wet, very warm climates

Table 11.1 Key properties of Reference Soil Groups (RSGs) detailed in this chapter.—cont'd

RSG	Description	Parent materials	Physiography
Phaeozems	Organic-rich topsoil over clay- or carbonate-rich subsoil (grassland-forest transitional soils)	Glacial till; eolian or unconsolidated deposits of basic materials	Level to undulating terrain in humid continental climates
Podzols	Coarse soils with subsoils enriched in organic matter-metal-hydroxide complexes	Glacial till; alluvial or eolian deposits, weathering products of siliceous rocks	Level to hilly landforms in boreal or other humid climates
Regosols	Weakly developed soils	Unconsolidated fine materials	Depositional landforms in all climates
Vertisols	Soils comprised of shrink-swell clays that churn and mix during drying/re-wetting cycles	Sediments or rock weathering products high in shrink-swell clays	Level or depressional landforms in climates with distinct wet and dry seasons

From this point forward, we refer to forest soil C as a stock: a measured amount of SOC per unit area, at a given time. To make comparisons straightforward across large scales (Table 11.2), we report SOC stocks in Mg C ha^{-1}, to a standardized depth of 1 m, acknowledging that some soils are shallower and some are deeper. To provide greater detail regarding the vertical distribution of SOC, we recognize three distinct parts of forest soil profiles: (1) organic horizons, which typically hold the most recently deposited, fastest-cycling SOC, and which is the most responsive to disturbance and management; (2) surface mineral horizons, which hold a mixture of actively cycling, recently released detrital C that tends to be less sensitive to surface disturbance than organic horizons; (3) subsurface mineral soil horizons that are even more buffered from surface disturbances and typically hold C that is even older, more slowly cycling, and more intimately associated with minerals. Each of these three horizons possesses a continuum of SOC that cycles at different rates; the maximum level of detail we provide into this complexity is based upon density fractionation (Crow et al., 2007). The separation of soils into discrete fractions based on density has become a widely applied technique in forest soils, and is usually executed to yield three pools: (1) a free light fraction, made up of particulate, non-mineral associated SOM that originates from root, fungal or leaf detritus; (2) an occluded light fraction, which includes older, low-density particulate organic matter that is physically protected within soil aggregates; and (3) the heavy or high density fraction, which includes non-particulate organic material intimately associated with clay- or silt-sized minerals, and usually contains the oldest (as determined by ^{14}C) C in soil.

What controls variation in forest soil C?

There are two types of variation in forest SOC that challenge attempts to detect impacts of global change or management. First, temporal variation: soils (and thus SOC) are dynamic even without disturbances or management, albeit typically over long timescales. Second, three-dimensional spatial variation in most properties of forest soils (but particularly SOC) can create substantial differences in observations only meters apart. Together, large spatial variation, background temporal dynamics,

Table 11.2 Area, mean and standard deviation of the SOC stock (to 1 m) for the three most extensive Reference Soil Groups within the forest lands of each Global Ecological Zone.

Global Ecological Zone (GEZ)	Reference Soil Group (RSG)	Area (*10^6 km^2)	1 m SOC (Mg C ha^{-1})	σ (Mg C ha^{-1})
Tropical rain	Ferralsols	5.13	171	120
Forest	Acrisols	4.82	184	134
	Nitisols	0.49	466	250
Tropical moist	Ferralsols	2.99	112	53
Forest	Acrisols	1.76	140	71
	Lixisols	0.49	87	25
Tropical dry	Ferralsols	0.88	105	42
Forest	Lixisols	0.72	83	22
	Luvisols	0.57	91	41
Tropical mountain	Acrisols	0.60	209	86
Systems	Andosols	0.50	493	195
	Cambisols	0.48	223	135
Subtropical humid	Acrisols	1.58	146	78
Forest	Luvisols	0.45	142	105
	Alisols	0.37	181	69
Subtropical dry	Luvisols	0.24	147	44
Forest	Cambisols	0.17	184	64
	Leptosols	0.05	159	64
Subtropical	Luvisols	0.92	213	101
Mountain systems	Cambisols	0.34	207	88
	Leptosols	0.18	142	73
Temperate	Cambisols	0.54	287	178
Oceanic forest	Podzols	0.27	530	243
	Andosols	0.16	557	223
Temperate	Podzols	1.15	321	131
Continental forest	Luvisols	1.15	212	144
	Albeluvisols	1.13	351	163
Temperate	Cambisols	1.63	263	116
Mountain systems	Podzols	1.16	355	122
	Luvisols	0.56	185	68
Boreal coniferous	Cryosols	3.59	567	330
Forest	Podzols	2.93	430	196
	Albeluvisols	2.55	417	254
Boreal tundra	Cambisols	1.82	623	277
Woodland	Cryosols	1.04	626	252
	Histosols	1.03	1078	629

Table 11.2 Area, mean and standard deviation of the SOC stock (to 1 m) for the three most extensive Reference Soil Groups within the forest lands of each Global Ecological Zone.—cont'd

Global Ecological Zone (GEZ)	Reference Soil Group (RSG)	Area ($*10^6$ km^2)	1 m SOC (Mg C ha^{-1})	σ (Mg C ha^{-1})
Boreal mountain Systems	Cryosols	3.69	414	170
	Cambisols	2.67	385	157
	Podzols	1.83	401	153

Statistics were calculated using SoilGrids250m (Hengl et al., 2017), and GlobCover (forest land only; ESA and UCLouvain, 2010). See text for description of spatial units (GEZs and RSGs) and discussion of SOC management considerations.

and typically small impacts or slow responses to management or global change make it difficult to attribute and quantify changes in forest SOC. This persistent reality informs our approach to understanding variation in SOC.

Soil C is closely linked to many soil properties and processes, which makes conceptual models describing how soils form (Dokuchaev, 1879; Marbut, 1927; Jenny, 1941; Simonson, 1959) helpful for appreciating C storage and turnover in forest soils. Previous reviews of models (Bockheim et al., 2005; Richter, 2007; Richter and Yaalon, 2012) identify four general themes relevant to variation in forest SOC: (1) Soil properties develop through interactions among soil-forming factors [climate, organisms, topography, and parent material] operating through time. (2) Factors of soil formation set the conditions within which soil-forming processes operate. (3) Four processes operating within soils drive the development of soil horizons; these are gains, losses, transfers, and transformations of energy and matter. (4) Intrinsic feedbacks and thresholds operating within soils make them subject to change, even in the absence of external environmental changes. The soil-forming state-factor model has been used as a framework in several SOC reviews (Scull et al., 2003; Trumbore, 2009; Chaopricha and Marín-Spiotta, 2014), and individual studies often attempt to test it directly (*e.g.*, Schaetzl et al., 2015; Silva et al. 2007). Such studies often show the influence of a single factor, but they also acknowledge that in most cases, interactions among factors drive soil property development, including SOC stocks. Human actions, including management, have received explicit recognition as soil forming factors (Yaalon and Yaron, 1966; Richter and Yaalon, 2012; Smith et al., 2016), but except where land use change is intense and persistent, anthropogenic effects are typically less significant sources of variation than factors such as climate, soil texture or vegetation (Guo and Gifford, 2002; Laganiere et al., 2010; Nave et al., 2010; Mulder et al., 2015; Cook et al., 2016; Wan et al., 2018).

SOC vulnerability and management across Earth's forests

Management to achieve a defined goal—including the sustaining or increasing of SOC through intervention—is most clearly evaluated when it represents an alternative to a baseline condition. Many management studies show no net change in SOC, some show SOC loss, and a few show gains. Effects of management on forest SOC are discussed here in the context of SOC vulnerability to loss, which we consider a conservative approach to SOC stewardship, because it is easier to lose SOC than gain it. Indeed, given its many vulnerabilities, it is challenging enough to conserve SOC, much less actively increase it. Globally, forest SOC is vulnerable to climate change, land use and altered disturbance regimes; management offers tools and approaches for reducing vulnerability by avoiding or mitigating losses and, in some cases increasing SOC relative to baseline conditions. Because most management studies show no net change in SOC, our recommendations for SOC management are necessarily cautious, and two considerations are critical. First, throughout this section, we review and report on only *forest* SOC management. For example, for areas that are no longer forest, we only include literature on conversions back to forest. Further, for this review we ask "compared to what?" How the baseline condition is defined against which SOC change is expressed can profoundly alter the apparent effects of management. For example, reforestation of degraded lands—specifically defined as lands where uses have caused physical deterioration of soil structure, including substantial losses of organic matter—can produce apparently very large SOC gains compared to the degraded condition. As another example, for forests with regularly scheduled harvest interventions, the benefits of retaining harvest residues for SOC management are likely to be minor, while harvesting and utilizing the residues can result in other important benefits. In this section, we summarize the broadest (global to domain-level) patterns and organize more detailed information around Tables 11.2 and 11.3, which stratify global forests into the GEZ framework and report the physical properties and key vulnerabilities of distinct RSGs.

Global to domain-level patterns

In forests of the *tropical domain*, major sources of SOC vulnerability are forest degradation (e.g., due to overgrazing, fuelwood or high-grade selective harvesting), deforestation (including forest conversion to croplands or pastures), and native forest conversion to tree plantations. Forest zones within the *subtropical domain* face many similar global change and land use pressures as tropical forests, but also encompass the management activities practiced within the many short-rotation plantations of this domain, which is home to some of Earth's most commercially valuable forests. In both tropical and subtropical domains, mountainous zones, which are typically cooler and wetter, tend to have significantly greater SOC stocks (in terms of SOC density to 1 m depth) than other zones, although highly weathered deep soils under lowland forests can contain larger SOC stocks when extrapolated over their often deeper soil profiles. In *temperate forests*, key SOC vulnerabilities are associated with harvest practices (e.g., residue removals), and the importance of fire increases as climate dries, especially in seasonally dry zones. In the *boreal domain*, fire is the dominant source of forest SOC vulnerability, especially in the context of expanding mineralization seasons and accelerating thaw of soil permafrost. In each of these domains, from tropical to boreal, mountainous ecological zones contain steep altitudinal gradients and island effects, which compress spatial patterns of SOC variation and thus pose challenges to forest and SOC management. Similarly, wetland forests can occur from the tropics to the boreal forests, and are defined as having persistently saturated soil conditions, which can result in the formation of deep organic soils containing high densities of SOC. Carbon in these soils is highly vulnerable to management related changes to wetland hydrology, such as ditching and drainage, or vegetation conversions that alter the balance between precipitation and evapotranspiration and facilitate microbial decomposition. Importantly, we note that this review does not explicitly consider wetland forests (see Chapter 9), despite their often unique vegetation, hydrologic properties, management aspects, and typically very high SOC densities.

Beneath our generalizations about SOC at global to ecological domain levels exists critical site-to-site (and within-site) variation. Some of this variation can be explained by variation in the properties

Table 11.3 Key vulnerabilities, management strategies for minimizing SOC losses, and opportunities for increasing SOC through management, by Global Ecological Zone (GEZ).

FAO Global Ecological Zone	Key SOC vulnerabilities in context of global change/ management	Management strategies for minimizing SOC losses	Opportunities for increasing SOC through management	Key considerations
Tropical rain forest	Deforestation; native forest conversion to plantation; drought	Prevent grass invasions that inhibit reforestation; agroforestry	Active reforestation; facilitate N-fixer establishment; fertilization	Management effects vary by land use intensity and duration, and plantation spp.
Tropical moist deciduous forest	Deforestation; native forest conversion to plantation; drought	Fire management; prevent grass invasions that inhibit reforestation; agroforestry	Active reforestation; facilitate N-fixer establishment	
Tropical dry forest	Deforestation; forest degradation; fire; desertification	Fire management; prevent grass invasions that inhibit reforestation	Active reforestation; facilitate N-fixer establishment	Soils in different forest life zones differ in C stocks as a function of precipitation
Tropical mountain systems	Deforestation; native forest conversion to plantation; erosion	Landscape mosaic of forest fallows; selection of high-SOC species for plantation	Soil erosion control; deliberate reforestation	
Subtropical humid forest	Native forest conversion to plantation; short-rotation harvesting	Rotations must be long enough to recover SOC; mixed-species plantations	Reforestation; selection of high-SOC species or genetic families for plantation	
Subtropical dry forest	Overgrazing; drought; fire; regeneration failure	Thinning to decrease fuels; reforestation of burned-over areas; grazer density control	Retain residues during salvage logging; reforest degraded soils with high-SOC species	In reforestation consider density, fire tolerance, and feedbacks to fuel loads
Subtropical mountain systems	Deforestation for cropping or grazing; climate-land use interactions		In plantations, select high-SOC species, reforest eroded slopes	
Temperate oceanic forest	Conversion of native forest to plantation; wildfire in seasonally dry ecoregions	Fuel management; if native forest is converted, select high-SOC species. If cut, leave residues	Reduce extent of harvest area on a landscape level to allow long-term C	

Continued

Table 11.3 Key vulnerabilities, management strategies for minimizing SOC losses, and opportunities for increasing SOC through management, by Global Ecological Zone (GEZ).—cont'd

FAO Global Ecological Zone	Key SOC vulnerabilities in context of global change/ management	Management strategies for minimizing SOC losses	Opportunities for increasing SOC through management	Key considerations
		or compensate removals with fertilization	accumulation in soil and vegetation	
Temperate continental forest	Wildfire; harvest residue removal through burning or extraction	Manage fuels and use prescribed fire; reforest burned or harvested lands; retain harvest residues	N-fertilization; afforestation; reforestation of marginal croplands	Harvest usually impacts only O-horizons; residue = fuel in dry climates
Temperate mountain systems	Wildfire, erosion, and resulting forest-to non-forest conversion	Fuel management	Introduce/favor tree species that are faster-growing, more tolerant of heat or drought	Compound disturbances (e.g., salvage harvest after fire) can have additive impacts on SOC
Boreal coniferous forest	Wildfire and resulting transition to less productive forests	Decrease fire extent/severity by incorporating young stands into landscape mosaic; lighter harvest removals (thin vs. clearcut, retain residues and stumps)	Fell residual wood to drive paludification; transition from spruce to fast-growing aspen w/high SOC or fire-adapted pines with fast SOC recovery	Slow growth and large O-horizons mean C losses can be very large and slow to recover. Drainage and *Sphagnum* influence spatial variation in fire C losses.
Boreal tundra woodland	Fires; longer mineralization season; permafrost thaw	Forest tree expansion has variable effects on SOC	Fertilization improves tree growth/litter inputs and inhibits decomposition, which increases O horizons	
Boreal mountain systems	Thawing permafrost, insect outbreak – fire interactions		Encourage forest establishment on mountain tundra; reforest burned slopes	Chemically distinct SOC pools differ in vulnerability to loss upon thaw

See accompanying text and subsequent tables for further discussion, key considerations, and references.

that allow soils to be categorized into RSGs, which in turn provide a basis for more nuanced interpretations of SOC management. As we consider individual GEZs in the remainder of this chapter, we will use RSGs to inform our views on vulnerability and management. For example, distinct RSGs often co-occur at landscape levels and are indicative of important management considerations, e.g., differing mechanisms of SOM stabilization or the potential for hydrologic transport of groundwater C. As another example, some GEZs possess RSGs that are so different as to pose clearly different challenges and opportunities for SOC management. In all our discussions of RSGs, we attempt to use published literature specific to the GEZ under consideration to link SOC back to fundamental differences in soil properties and processes of formation.

Patterns within domains: tropical domain

Tropical rain forest and tropical moist deciduous forest zones

Within the tropical climate domain (all months without frost), rain and moist forest zones differ by the length of their dry seasons (≤3 months of dry weather for rainforest; 3–5 months for moist forest). Soil C stocks differ for tropical wet versus moist forest life zones (Marín-Spiotta and Sharma, 2013; Post et al., 1982), yet many of the key SOC vulnerabilities, management strategies for mitigating SOC losses, and opportunities for increasing SOC are similar. On a global basis, these zones are very closely co-distributed, and often intergrade rather than separate along abrupt boundaries (Staal et al., 2016). Total areas of the two zones are about 17 million km^2 for the rain forest, and 13 million km^2 for moist deciduous forest; globally, forest vegetation constitutes 60%–70% of the land cover in these zones.

Critical SOC vulnerabilities stem from deforestation for pasture or cropland use or mining (Alvarez-Berros and Aide, 2015; Cusack et al., 2016; Fearnside and Barbosa, 1998; Lugo and Brown, 1993), grass invasion in disturbed areas that inhibits forest re-establishment (Balch et al. 2015; Cusack et al., 2015), and native forest to plantation conversion (Chiti et al., 2014; Guillaume et al., 2015). Changes in drought frequency and magnitude, extreme rainfall events, and atmospheric N deposition are also expected to alter SOC dynamics in tropical rain forests (Cusack et al., 2016). Because SOC losses due to deforestation vary in magnitude (Ostertag et al. 2008) and can be reversed (Don et al., 2010), reforestation is generally an effective SOC loss mitigation strategy (Cusack et al., 2016; Silver et al., 2000). Accelerated rates of SOC recovery (Paul et al. 2002) or even net gains relative to native forest are possible when N-fixing tree species are planted (Macedo et al., 2008); fertilization (either with N or base cations) may also increase SOC, even in native forests (Cusack et al., 2011, 2018).

Land use change in tropical rain and moist forests has represented such a large share of global terrestrial C emissions to the atmosphere in recent decades (Houghton, 2003) that there has been considerable attention to the role of soils in this net flux. However, while land use effects on, and trajectories of, C in aboveground biomass are predictable, soils are much more difficult to generalize (Powers and Marín-Spiotta, 2017). Complicating matters, geographic bias in field study locations precludes extending the inferences from studies that do show net change in SOC with land use because these tend to be on sites with unique ecosystem factors (e.g., soil classification, climate) that are not widely represented across the tropical domain (Powers et al., 2011). This problem is exacerbated by the fact that soil properties and climate are important controls on SOC in their own right, as well as in interaction with land use and management (Marín-Spiotta and Sharma, 2013; Brown and Lugo, 2017; Weaver et al., 1987).

Table 11.4 Management considerations and supporting references for key RSGs discussed in the combined tropical rain + moist deciduous forest GEZs.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Ferralsols	42	SOC can turn over very rapidly, requiring sustained C inputs to maintain stocks	Hall et al. (2015); Hughes (1982); Johnson et al. (2011a,b); Marques et al. (2011); Wagai and Mayer (2007)
Acrisols	34	Clay stabilization imparts somewhat greater stability to SOC than the Al/Fe hydroxides of Ferralsols	Krull et al. (2002); Soares and Alleoni (2008)
Nitisols	3	High structural stability and SOC stabilization potential due to high clay content	IUSS WRB Working Group (2006)

For highly weathered soils of the wet and humid tropics, sorption to Al and Fe hydroxides or low-activity clays is the principal mechanism for SOC stabilization (Tables 11.1 and 11.4). Organic C that is associated with Al and Fe hydroxides or low-activity clays typically constitutes $\geq$80% of all SOC in these soils, and while this C typically has slower turnover rates than free particulate SOC, turnover can be surprisingly rapid — on the order of decades — especially following land use changes. This highlights the fact that for Ferralsols (the most extensive RSG in the tropical rain and moist forest zones), the maintenance of soil C stocks is strongly dependent on continuous inputs from the highly productive vegetation. This contrasts with Acrisols, the second most extensive soil across tropical rain and moist forests, which possess clay-rich subsoils that impart greater stability to adsorbed organic matter than do Al and Fe hydroxides. As a result, Acrisols tend to have higher C stocks (Table 11.2), and the presence of clay-stabilized C may mitigate C losses associated with rapid turnover in other SOC pools. Furthermore, although also strongly weathered, Acrisols still have some residual primary minerals, permitting further production of C-stabilizing clays and hydroxides within the soil profile.

Overall, SOC stocks and SOM stabilization mechanisms of Ferralsols and Acrisols suggest three principal constraints on SOC change in the context of management. First, rapid changes in SOC can occur with land use change; losses of relatively old, mineral-associated SOM and gains of fresh (light fraction) SOM may occur simultaneously, resulting in no apparent net change in bulk SOC stocks (Marín-Spiotta et al., 2009). Second, more significant clay-stabilization mechanisms may make Acrisols somewhat less vulnerable to rapid SOC losses due to land use or management changes, but slower to regain lost C against larger background levels; their higher primary mineral contents also likely make them less susceptible to severe degradation. Third, because vegetation productivity is often nutrient-limited (P, N, base cations), particularly after intensive agricultural or pasture use, fertilization to increase detrital C inputs to soil is an effective way to increase SOC in reforested soils. Between these and other soils that are widespread across tropical rain and moist forests, the relative sensitivity of vegetation and SOC to fertilization is probably inversely related to inherent fertility, such that the greatest results will be observed on Ferralsols, followed by Acrisols, followed by, e.g., Nitisols.

Likewise, if fertilization does increase primary production and C inputs to soil, the capacity for stabilizing some of that C on mineral surfaces is likely greatest for Nitisols, intermediate for Acrisols, and least for Ferralsols.

Tropical dry forest zone

Forests cover approximately one-half (~4.7 million km^2) of the tropical dry ecological zone, which has 5–8 dry months per year. While these forests share some of the key SOC vulnerabilities of the tropical rain and moist forest zones, such as forest degradation, deforestation, and grass invasion (Cusack et al., 2015; Griscom and Ashton, 2011, Freifelder et al., 1998) they are more susceptible to fire and savannization (Sivakumar 2007; Vargas et al., 2008). The most effective means of mitigating SOC losses in tropical dry forest are therefore the judicious management of land uses that can lead to forest degradation, such as grazing and fuelwood harvesting. In disturbed forests, preventing further deterioration of site productivity through fire control and the exclusion of grasses that increase fire susceptibility aids in maintaining forests, along with supplemental planting to maintain canopy cover (Griscom and Ashton, 2011). Where forests are lost, they can be challenging to recover due to interactions between drought and invasive grasses, but reforestation can be a successful way to increase SOC, and N-fertilization may also help (Silver et al., 2000; Marin Spiotta and Sharma, 2013; Cusack et al., 2015). Direct surface organic amendments (or, in a harvesting context, residue retention) can help retain soil water, which can enhance survivorship of planted trees during drought and in turn stand productivity.

Tropical dry forests are distributed across a wide range of soils and have the lowest SOC stocks in the tropical domain. Ferralsols are still most extensive (19% of forest area), and their management considerations are similar to the wetter tropical forest zones—namely, they are dependent on sustained C inputs to maintain SOC stocks, are prone to rapid SOC turnover when land use changes, and have less ability to recover from severe disturbances due to their advanced weathering state. Lixisols, Luvisols, and Vertisols are also extensive, and these soils differ substantially from Ferralsols in their properties and management (Tables 11.1 and 11.5). Lixisols and Luvisols have similar pedogenesis, possessing low-clay surface soils over high-clay subsoils. Because the topsoils of both of these RSGs have limited structural stability, physical disturbances can readily break down aggregates and release previously physically-protected SOC that cycles more slowly, and likely takes longer to recover, than the non-protected, free light fraction of SOC that responds most rapidly to changes in land management and use. The heavy clay Vertisols, with their extreme shrink-swell dynamics and vertical mixing, create a different set of considerations for SOC management. Although shrink-swell dynamics can inhibit reforestation after pasture abandonment in the dry tropics, the use of soil amendments and drought-tolerant plant species can increase reforestation success and also incorporate surface C inputs to considerable depths (Werden et al., 2017). Furthermore, while much of the research on soil amendments in Vertisols has been conducted in agricultural settings, the large area of these soils under tropical dry forests is worthy of attention as an opportunity for increased SOC sequestration, especially in the context of agroforestry (Srinivasarao et al., 2014).

Tropical mountain systems

Globally, approximately one-third of the 6.6 million km^2 of tropical mountain systems is forested, and these systems are closely co-distributed with other ecological zones of the tropical climate domain. In areas of high precipitation, tropical mountain systems share some sources of SOC vulnerability with

Table 11.5 Management considerations and supporting references for key RSGs discussed in the tropical dry forest GEZ.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Lixisols	15	Topsoils have limited structural stability, making aggregate-protected SOC vulnerable to physical disturbance	Garcia-Oliva et al. (2006)
Luvisols	12	Physical vulnerability is similar to Lixisols, but high-activity subsoil clays may promote stronger sorption of SOC	Fonte et al. (2010); IUSS WRB Working Group (2006)
Vertisols	11	High content of heavy clay and vertical mixing may promote deep incorporation and clay stabilization of surface organic inputs	Hua et al. (2014); Leinweber et al. (1999); Quin et al. (2014)

wet tropical forest zones. Namely, native forest conversion to plantation, grazing, or cultivation are often associated with decreased SOC (de Blecourt et al., 2013; Girmay et al., 2008; Tesfaye et al., 2016), although reforestation can partly reverse SOC losses due to forest land use changes on some soils (Cusack et al., 2016; Don et al., 2010; Lugo and Brown, 1993). Tropical mountain systems are unrivaled in the tropical domain in their degree of heterogeneity in climate, landforms, soils, and vegetation types (Ponette-Gonzalez et al., 2014), however, and therefore possess unique SOC vulnerability and management considerations. Steep slopes make erosion a key source of SOC vulnerability (Walker and Shiels, 2008), and climate change-vegetation-SOC interactions in these highly heterogeneous zones are complex (Alvarez-Arteaga et al., 2013; Martin and Fahey, 2014). Due to the high degree of spatial variability in physiographic factors, ecosystems, and soils, landscape approaches to management (e.g., delineating and prescribing management based on soil-topographic units) are useful for prioritizing areas for intervention or protection (Jeyanny et al., 2013).

Many of the RSGs that are extensive in adjacent ecological zones are also extensive in tropical mountain systems, including Acrisols (21% of forest area in tropical mountain systems), Ferralsols and Luvisols (13% each). However, tropical mountain systems also have large areas of soils that are distinct from other tropical zones. These are the Andosols, which form in volcanic parent materials, and Cambisols, which form in relatively young alluvial or colluvial deposits common in erosional-depositional montane settings. Compared to the older landscapes and soils of non-mountainous tropical zones, these mountain system soils are often relatively young. Andosols are rapidly weathered, forming abundant short-range-order minerals and stable organo-mineral complexes, whereas Cambisols show little parent material alteration and thus have very low contents of the illuvial clays or Al and Fe hydroxides that provide stable binding sites for SOM (Table 11.1). Soil C management considerations of Ferralsols and Acrisols in mountain systems are consistent with other tropical zones (e.g.,

Table 11.6 Management considerations and supporting references for key RSGs discussed in the tropical mountain systems GEZ.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Andosols	18	High porosity and abundant Al/Fe hydroxides promote rapid accumulation of, and impart stability to SOC	Baisden et al. (2013); Batjes (2008); Lemenih et al. (2005)
Cambisols	17	Low initial contents of clay and metal hydroxides limits capacity for C stabilization	Heitkotter et al. (2017); Johnson et al. (2015)

Table 11.4; although their stocks are considerably higher-see Table 11.2), while the management considerations of Andosols and Cambisols are as different as their physical properties (Table 11.6). Most importantly, due to their high capacity to rapidly accumulate organo-metal SOM, Andosols hold much more SOC than other soils in tropical mountain landscapes. Detecting management effects on bulk SOC stocks against such a large background and spatial variability is very difficult (Homann et al., 2008), requiring methods that isolate distinct C pools to detect change (Crow et al., 2015). In contrast, because subsoil C in Cambisols is of limited stability due to a lack of metals and clays for stabilization, management activities to promote deep rooting and accelerate weathering may increase SOC accumulation in the long term. In terms of active management, this likely means reforestation, supplemental planting, or plantation establishment using tree species with significant belowground C allocation to mycorrhizae and root exudates (Aoki et al., 2012; Camenzind et al., 2018). Similarly, if maintenance of existing SOC stocks in native forests is the management goal, ecosystems with large absolute quantities (or large proportions) of ecosystem C held in soil may be prioritized for protection or minimal-impact management (Manrique et al., 2011).

Patterns within domains: subtropical domain

Subtropical humid forest zone

Forests cover approximately one-half (3.1 million km^2) of the subtropical humid ecological zone, and these forests are some of the most productive and economically valuable on Earth. The climate of this ecological zone is favorable, with no dry season and $\geq$8 months per year with a mean temperature greater than 10°. However, while their climate makes these forests ideally suited to fiber and fuel production, their global distribution on geologically old parent materials and highly weathered soils poses some challenges to SOC management. Specifically, the key sources of SOC vulnerability in this zone derive from the potential of intensive forest management to deplete SOM or nutrients either directly through physical impacts, or via feedbacks with decreased vegetative production. Native forest conversion to plantations and short-rotation harvesting are often associated with SOC decreases (Chen et al., 2004, 2016; Fan et al., 2016; He et al., 2008; Lu et al., 2015; Lyu et al., 2017), but these impacts are not universal (Samuelson et al., 2017) and retaining harvest residues (Butnor et al., 2012) or ameliorating site impacts through fertilization (Vance, 2000) can mitigate or even prevent losses.

Furthermore, longer rotation lengths can protect against small SOC declines that may otherwise become compounded over multiple entry cycles (Chen and Xu 2005, 2014), and mixed-species plantations (Wei and Blanco, 2014), or plantings of distinct genetic families with higher belowground C allocation may increase SOC stocks (Fox, 2000; Sarkhot et al., 2008).

Subtropical humid forests have played a leading role in the “green revolution” in forestry. Scientists and managers have learned to incorporate site-level factors and vulnerabilities into a very active approach to plantation management that sustains high levels of biomass output and long-term site productivity (Fox et al., 2007; Nichols et al., 2010; Vance et al., 2010). Critically, many of these site factors are soil properties that either mediate the effects of, or exert greater control over SOC than the management itself (Johnson et al., 2002; Lupi et al., 2017; Maier et al., 2012; Schoenholtz et al. 2000).

In the subtropical humid zone, Acrisols, Luvisols, and Alisols are the most extensive RSGs; other widespread soils include Cambisols and Phaeozems. Acrisols are the most vulnerable of these soils because their physical and chemical properties make them prone to declines in fertility and forest production, and less able to retain or gain SOC than other RSGs (Tables 11.1 and 11.7). Specifically, the low-activity clays of Acrisols (as compared to Luvisols and Alisols), low levels of minerals that weather to clay (as compared to Cambisols), and low levels of organic matter (compared to Phaeozems) render Acrisols the soils that require the greatest care in management (or the most intensive inputs) to sustain or increase SOC. In terms of inputs, N addition can increase SOC stocks, but the limited clay- or organo-mineral stabilization potential of these soils limits their capacity to retain soluble organic amendments. Phaeozems imply an interesting scenario for SOC management in the context of climate and vegetation change. These high-SOC soils have formed in regions where forest and grassland vegetation have been in tension for millennia and may possess SOM-stabilizing processes typically attributed to one or the other (but not both) of these vegetation types. That is, while physical aggregation has been viewed as more important in grasslands, and organo-metal (Al, Fe) complexation

Table 11.7 Management considerations and supporting references for key RSGs discussed in the subtropical humid forest GEZ.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Acrisols	50	N-fixers or fertilization can restore diminished productivity, but soluble organic amendments are poorly retained	Pegoraro et al. (2010), 2016; Zhang et al. (2014)
Alisols	12	Higher-activity clays and mineral surfaces retain more SOC than Acrisols, but more susceptible to erosion	Ma et al., 2015; Wu et al. 2017
Phaeozems	4	Multiple processes stabilize C, promoting larger SOC stocks than co-occurring soils	Barthold et al. (2013); Helfrich et al. (2010); Rodionov et al. (1998); Schoening et al. (2005)

more important in forests—research shows that organo-metal stabilization occurs in, and stabilizes large amounts of SOC in Phaeozems currently supporting grassland (Masiello et al., 2004). This is a reminder that the management of SOC cannot necessarily be achieved through the management of vegetation, at least on the timescales of the latter.

Subtropical dry forest zone

Forests cover about one-quarter (0.6 million km^2) of the subtropical dry zone, which carries a different set of climate and land use pressures and SOC vulnerabilities along with its dry climate and generally lower vegetation productivity. Here, grazing and changes to forest structure and soils that can result from it—including grass invasions, regeneration failure, and fire—are key drivers of SOC vulnerability (Abril et al., 2005; Bonino, 2006; Conti et al., 2016; Potes et al., 2012; Wolfe and Van Bloem, 2011). The proactive approach to avoiding SOC losses due to these drivers is to manage grazer density in order to avoid direct, physical impacts to SOC and indirect feedbacks through vegetation change (Assefa et al., 2017). Reforestation—whether as supplemental planting in disturbed forests, or new forest establishment where it has been lost—is effective for increasing SOC, which increases with tree density (Oubrahim et al., 2016). In the subtropical dry zone, the seasonally dry climate exacerbates the challenge of re-establishing forests when they are lost. However, because the risk of extreme fire is related to the density of seasonally dry vegetation and fuels and interactions with drought and insect pests (Anderegg et al., 2015; Kolb et al., 2016), overstocking can be a source of vulnerability, so active management to hold fuel loads at acceptably low levels mitigates fire related risk of SOC loss (Ruiz-Peinado et al., 2013, 2017). Where fires occur, salvage logging is a common practice, and in these cases, retention of residues is recommended in order to avoid exacerbating fire-driven SOC losses (Santana et al., 2016). When plantations are established in the subtropical dry zone, it is often after deforestation has occurred and is practiced as a remediation technique following the intervening land use (e.g., grazing) or disturbance (e.g., fire). In these cases, plantation forestry is an active alternative to natural regeneration, which can fail for various reasons, and is known to increase SOC. Species selection matters, as with other zones, because associated traits such as fire tolerance, N-fixation, fast growth, deep rooting, and high root to shoot ratios, can affect whether plantings succeed, SOC increases or decreases (Guedes et al., 2016; Hoogmoed et al., 2012, 2014; Maraseni et al., 2008; Richards et al., 2007).

Globally, Luvisols and Cambisols are the most extensive soils in the subtropical dry zone, but easily degraded Leptosols and Lixisols are also extensive (Tables 11.1 and 11.2). The principal distinction between Luvisols and Cambisols that informs their SOC management is how their differing development influences their capacity for subsoil SOM stabilization (Table 11.8). Specifically, Luvisols, with their subsoils of high-activity clays and considerable amount of remaining primary minerals that will weather to clay, have greater capacity to stabilize SOM and buffer it from short-term changes, which are often notable for Cambisols. Lixisols are essentially older and more strongly weathered Luvisols; their surface soils are vulnerable to structural deterioration and the resulting loss of occluded OM, while their subsoils are characterized by low-activity clays with less capacity for organo-mineral stabilization. Therefore, Lixisols are a case for conservative management in that they can readily lose, but not easily regain, SOC. Where degraded or converted to other land uses, a combination of reforestation (or agroforestry) and physical protection of the soil surface (e.g., amendments) is likely the most effective means to mitigate the loss or promote the recovery of lost SOC.

Table 11.8 Management considerations and supporting references for key RSGs discussed in the subtropical dry forest GEZ.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Luvisols	37	Amendments or retained residues may increase SOC due to the stabilization capacity of high-activity clays	Noble et al. 2000; Thomas et al. 2007
Cambisols	25	SOC can turn over quickly; surface residues and pyrogenic C do not persist due to limited SOM-stabilizing clays and metals	Jimenez-Gonzalez et al. (2016); Maria de la Rosa et al. (2018)
Lixisols	7	Low clay activity of subsoil drives loss of SOC during deforestation; agroforestry can mitigate losses	Beedy et al. 2010; Dawoe et al. 2014; Garland et al. 2018

Subtropical mountain systems

Subtropical mountain systems cover ~6.5 million km^2 and are on a global basis about one-third forested. This GEZ includes a wide range of forest and soil types because its wide, steep altitudinal gradients, often spanning diverse parent materials, confer dramatically different climatic conditions over compressed areas. At one extreme, i.e., low elevations with ocean-moderated climates, these forests are warm, wet, productive, and dominated by subtropical tree taxa growing on mature landforms and soils. At the other extreme, subtropical mountain systems can host temperate or even boreal (often coniferous) tree taxa growing on high-elevation terrestrial "islands," where landforms may have been recently deglaciated and soils are young. Because altitude plays such a strong role in influencing the spatial distribution of forest types and SOC stocks in subtropical mountain zones, climate change interactions with other ecosystem properties and sources of SOC vulnerability are critical (Li et al., 2016; Singh et al., 2011). For example, even baseline SOC stocks have shifted with climate change in recent decades, though rates differ across landforms and land uses (Martin et al., 2010). In this zone, as with others in the subtropical domain, deforestation is the principal direct source of SOC vulnerability (Dorji et al., 2014; Falahatkar et al., 2014; Yimer et al., 2007). Losses may be mitigated if partial forest cover is retained during the land use change, either by retaining trees on grazed parcels, or retaining forests in a rangeland-forest mosaic (Renison et al., 2009). Nonetheless, across this ecological zone, deforestation is occurring to open land for grazing and cultivation, and these conversions show a generally consistent signal of decreased SOC from the forest condition, especially when the land use change is to cultivation. On the other hand, reforestation, forest-to-plantation conversion, or species management within existing forestlands can be used to select for species that are associated with high rates of C uptake and SOC accumulation (Jiang et al., 2011; Diaz-Pines et al. 2015; Schomakers et al., 2017). Appropriate species and management strategies vary depending on soils, however, of which subtropical mountain systems have a wide variety. As in the subtropical dry forest zone, Luvisols (46%) and Cambisols (17%) predominate, but Leptosols, Kastanozems, and Regosols are also common.

Management of SOC on Luvisols and Cambisols in subtropical mountain systems share similar considerations with these RSGs in other ecological zones, as influenced by their divergent physical factors (e.g., high vs. low capacity, respectively, for SOC stabilization on clay minerals). The other RSGs that are widespread in forests of subtropical mountain systems warrant special discussion (Table 11.9). Watershed management and slope stabilization are important to SOC management in subtropical mountain systems because of steep slopes and extensive areas of easily eroded soils (Leptosols and Regosols). Leptosols, which are rocky, shallow, and typically steep, and Regosols, which are weakly-developed soils often found forming in the alluvial deposits below eroding Leptosols, are physically vulnerable soils that tend to have very low and superficial SOC stocks (Ozsoy and Aksoy, 2011; Sevink et al., 1998). If subjected to erosion, such as after fire or forest clearing, the limited organic matter in these soils can be lost off site and negative feedbacks to soil stability and forest re-establishment can follow. However, the erosional-depositional processes common in humid, mountainous systems can bury and stabilize old soil C, and weathering processes in new alluvial deposits can create new mineral surfaces for SOM stabilization quite rapidly (Morisada et al., 2004; Portes et al., 2016; Turk et al., 2008). Where these soils have been destabilized, re-establishing forest vegetation and litter (or other detrital) cover is the most effective management strategy for protecting soils from continued erosion. Kastanozems are noteworthy among soils of the GEZ because they developed historically under grassland vegetation. The presence of forest on these soils represents climate and disturbance regimes that have favored forest expansion in recent centuries, but are likely to deteriorate as the climate warms and dries and soils on the forest-grassland continuum transition toward RSGs that hold less SOC (Barthold et al., 2013; Dimeyeva et al., 2016; Krupenikov et al., 2011). At present, the organic-rich Kastanozems have inherently higher productivity than Leptosols and Regosols, but their SOC is similarly vulnerable to erosion in steep montane settings.

Table 11.9 Management considerations and supporting references for key RSGs discussed in the subtropical mountain systems GEZ.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Leptosols	9	Water-stable aggregates protect SOC and erosion, but are vulnerable to intensive agroforestry, deforestation, fire	Ingelmo et al. (2001); Kizilkaya and Dengiz (2010); Le Bissonnais et al. (2018); Martin et al. (2012)
Kastanozems	7	SOC in steeply sloping soils can be protected by mulching and organic amendments	Baptista et al. (2015)
Regosols	5	Site access for reforestation and slope stabilization is difficult but necessary to mitigate erosion, SOC losses, and feedbacks	Hua et al. (2016); Fernandez-Romero et al. (2014); Lado et al. (2016)

Patterns within domains: temperate domain

Temperate oceanic forest zone

The temperate oceanic forest GEZ occupies just over 3 million km^2, and is approximately one-third forested on a global basis. Ecosystems in this GEZ, which is characterized by an oceanic climate with the coldest month's mean temperature remaining above freezing, are most extensive in western Europe, but are also distributed across maritime areas of western North America, southwestern South America and southeastern Oceania. Forests that have developed in the moderate climate and long growing season of this GEZ are dominated by large-stature, long-lived taxa (*Nothofagus, Picea, Pseudotsuga*), and many grow on highly fertile soils. As a result, SOC stocks are larger in this zone than in others of the temperate domain (Table 11.2). These forests are therefore economically valuable and active management and plantations are common. However, key SOC vulnerabilities in this zone include the conversion of native forests into plantations (Prietzel and Bachmann, 2012; Turner and Lambert, 2000), and, in ecoregions with seasonally dry climates, wildfires capable of consuming significant amounts of O-horizon and even mineral soil C (Homann et al., 2015). The capacity of these forest ecosystems to accumulate large C stocks in soil, live biomass and woody debris, suggests maintaining non-managed forests on some fraction of large landscapes may be optimal with respect to C storage (Homann et al., 2005; Matsuzaki et al., 2013; Schulp et al., 2008). Where active management is practiced, SOC losses may be mitigated by selection of plantation species with high SOC density (Oostra et al., 2006; Schulp et al., 2008), grown over longer rotations than typically practiced (Gray and Whittier, 2014). Retaining harvest residues may help maintain O-horizon and mineral soil C pools; if these pools are diminished, their resulting nutrient removals may be mitigated with fertilization to maintain vegetation productivity and feedbacks to SOC (Hopmans and Elms, 2009; Jones et al., 2011). However, large trees produce significant quantities of harvest residues which, if not managed (e.g., chopped and scattered) can act as ladder fuels in wildfires. The diversity of soils, forests, and even genotypes of economically important tree species in the temperate oceanic forest zone have spurred close connections between research and management that optimizes practices to the unique properties of individual sites (Chappell et al., 1991; Turner et al., 2001). Broad generalizations based upon RSGs may therefore be useful in some situations, especially if site-specific information is limited, but do not diminish the ability for even intensively managed forests to maintain their SOC in comparison to the non-managed forests that are often considered the baseline (Bravo-Oviedo et al., 2015; Jurevics et al., 2016).

Soil physical and chemical properties are an important basis for management decisions, and these vary considerably between the three most widespread soils in this zone (Tables 11.1 and 11.10): Cambisols, Podzols, and Andosols. Cambisols, because of their incipient development, have lower contents of Al and Fe hydroxides and therefore less capacity to stabilize SOC (Schoening and Kogel knabner 2006; Spielvogel et al., 2008). As a result, subsoil C in Cambisols is largely root-derived compared to Podzols, which develop their larger, more stable subsoil SOC stocks via vertical translocation of organo-mineral complexes (Rumpel et al., 2002). This difference highlights why Cambisols may be the RSG with the greatest potential for positive SOC responses in this GEZ. Specifically, where Cambisols are found in parent materials high in plant nutrients, active management of fast-growing species may promote deep rooting and accelerated mineral weathering that creates SOM-stabilizing clays and metal hydroxides. In contrast, the Podzols and Andosols of this GEZ possess much larger SOC stocks than in other zones, arguing for management approaches that aim to maintain SOC stocks rather than

Table 11.10 Management considerations and supporting references for key RSGs discussed in the temperate oceanic forest GEZ.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Cambisols	48	Residue retention can minimize compaction on fine-textured soils and may promote SOC accumulation	da Costa et al. (2016); Klaes et al. (2016)
Podzols	21	Maintenance of SOC depends upon stability of climate, absence of disturbance, continued C inputs via detritus and soluble organic matter	Pengerud et al. (2014); Sanborn et al. (2011)
Andosols	15	Organomineral and aggregate-bound C pools are large and resistant to disturbance; physical structure is resilient to compaction	McNabb and Boersma (1993); Panichini et al. (2017)

attempt to increase them (a questionable proposition). In a sense, Podzols are the logical conclusion of soil development and SOC accumulation the cool, wet climate of this GEZ. Their high SOC densities result from a large pool of physically vulnerable C in fire-vulnerable surface detritus (the O horizon), some of which is transferred into subsoils where C is more stable but nonetheless vulnerable to climate change and disturbance (Pengerud et al., 2014; Sanborn et al., 2011). Podzols and Andosols share the metal hydroxide mechanism of SOM stabilization, but Andosols have several differences with implications for their SOC management. Namely, Andosols tend to be deeper soils, have greater porosity and rootability, lower bulk density, and possess well-developed hierarchical aggregates that provide an additional mechanism of C stabilization (Asano and Wagai, 2014; Parfitt, 2009; Takahashi and Dahlgren, 2016). As a result, physical properties and SOC stocks on Andosols are probably less susceptible to deleterious management impacts than on Podzols.

Temperate continental forest zone

The temperate continental zone, which is slightly less than one-half forested, covers nearly 12 million km^2 and as a result spans a very wide range of climates, forest, and soil types. It is likely the best-studied zone in many aspects, including SOC management. Similar to temperate oceanic forests, key SOC vulnerabilities include wildfires and harvest residue removals (e.g., by slash burning or "whole tree" biomass harvesting). Negative impacts of residue removal are not the norm (Vance, 2018), but some reviews indicate that removing residues sometimes causes SOC losses, and a prudent approach is therefore to retain them to ensure that C stocks and the other ecosystem services they support are maintained (Janowiak and Webster, 2017; Thiffault et al., 2011). Nonetheless, harvest residues are potential fuel in many temperate continental forests, so managing them (e.g., chip and scatter, broadcast burn) may mitigate against larger problems (and SOC losses) occurring due to uncontrolled high-severity fires.

In general, prescribed fires and low-severity wildfires generally cause smaller SOC losses than high-severity wildfires, justifying proactive burning or allowing low-severity fires to burn as a way to manage SOC (Bennett et al., 2017; Jandl, 2007; Nave et al., 2011). Over large areas and long timescales, fuel reduction treatments (with or without prescribed fire) have uncertain effects on the C balance of fire-prone ecosystems and soils (Restaino and Peterson, 2013), but that is small comfort to individual forest owners and managers concerned with protecting SOC on specific lands. This highlights the importance of site- and project-specific plans for management involving harvest residues and fuels.

Unlike residue and fuel management, reforestation is an activity that has generally very consistent and positive impacts on SOC, across a range of spatial levels and project sizes. Whether after harvesting, fire, or other disturbances or intervening land uses, reforestation and afforestation increase SOC stocks in temperate continental forests over decadal timescales (Cunningham et al., 2015; Laganiere et al. 2010; Nave et al., 2013, 2018a; 2019). Where tree establishment is on previously non-forested lands, afforestation has the dual benefit of increasing SOC density and increasing forest area. Additions of N to forests are another way to increase SOC stocks in the temperate continental GEZ. Increases vary depending on many factors, including the form and timescale over which N is input, but commercial-scale fertilizer applications, inclusion or retention of N-fixing vegetation, and high rates of atmospheric N deposition are all associated with SOC increases (Hoogmoed et al., 2014; Johnson and Curtis, 2001; Nave et al., 2009). However, over-addition of N can cause its own set of problems (e.g., soil greenhouse gas emissions, eutrophication of water bodies), and may not be the factor that controls primary production in many ecosystems or C storage in many soils. Thus, these general patterns should be informed by site-level constraints and knowledge of the unique properties of their soils.

The most widespread soils in the temperate continental zone are a relatively even mix of Podzols, Luvisols, and Albeluvisols. Podzols and Luvisols differ substantially in many ways, including parent material, morphology, dominant soil-forming processes and mechanisms of SOM stabilization (Table 11.2; Sanborn et al., 2011; Lavkulich and Arocena, 2011). However, they are frequent associates at landscape to hillslope levels, where they are hydrologically coupled and even intergrade morphologically as Albeluvisols. Thus, despite differences in properties that inform their physical vulnerabilities (e.g., to compaction) and SOC management (e.g., Al-hydroxide vs. clay mineral C stabilization), these soils are justifiably considered as a coupled system (Table 11.11). In other words, management occurring on one of these soils may produce impacts on another. For example, organo-mineral C that is released from an upland Podzol subsoil following harvesting may be laterally transported in groundwater through a closely associated hillslope Albeluvisol, and ultimately stabilized on clay minerals in a low-lying Luvisol. This potential raises an important issue: attempts to quantify management impacts on SOC, typically as a stock change on the site where the management occurs, may be missing farther-reaching responses and therefore under-estimating effects. Where these soils are present as discrete units, differences in their properties do afford some clearer distinctions in terms of SOC management. For example, SOC in the coarse-textured, drought-prone Podzols is more vulnerable to loss via fires, and severe fires may render reforestation difficult. In contrast, the clayey Luvisols, which can be wet to the point of developing stagnic conditions, may afford opportunities to gain SOC (or mitigate its loss in a warming, drying climate) through hydrologic management that maintains saturated conditions that drive organic matter persistence. As another example, to the degree that plant production and detrital inputs are responsible for C stocks in these soils (e.g., the O horizons), the acidic, low-fertility Podzols may show stronger N-fertilization responses than Luvisols, where many dominant tree taxa are calciphilic and limited by base cations.

Table 11.11 Management considerations and supporting references for key RSGs discussed in the Temperate Continental Forest GEZ.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Podzols	22	O horizon and organo-mineral subsoil C may be destabilized by harvesting or climate change	Rothstein et al. (2018); Ussiri and Johnson (2007)
Luvisols	22	SOC is overall less vulnerable than Podzols due to thinner O horizons and subsoil C stabilization via clay minerals	Lavkulich and Arocena (2011)
Albeluvisols	21	Destabilized SOC may be hydrologically transported across soil-landscape units	Bockheim (2003), 2011; Gillin et al. (2015)

Temperate mountain systems

Temperate mountain systems cover approximately 5 million km^2 on Earth, and are overall about 40% forested. Similar to mountain systems of other domains, temperate mountain systems have highly heterogeneous topography, parent materials, climate, soils, and forests. This variability is enhanced because portions of most temperate mountain systems were glaciated until the recent past; thus, even within localized areas, soils may span orders of magnitude in their ages, SOC stocks and associated forest cover types (Whelan and Bach, 2017; Zhou et al. 2016). Despite their uniquely high spatial heterogeneity, temperate mountain forests share fire as a major source of SOC vulnerability with the other temperate forest GEZs. However, post-fire erosion is a more significant consideration in mountains due to steep topography and orographically-enhanced precipitation (Baird et al., 1999; Bormann et al., 2008; Cui et al., 2014). Severe or repeated fires can cause drastic changes in vegetation, such as forest to meadow (Buma and Wessman, 2013; Fletcher et al., 2014; Kashian et al., 2006). Because of the potential for significant declines in SOC and forest area, stem density and fuel management treatments such as thinning with harvest residue removals may avert or mitigate SOC losses that occur with severe wildfires (Agee and Skinner, 2005; Page-Dumroese and Jurgensen, 2006). As with any system, context, site specific details and tactics matter, as do the nature, number, sequence and timing of disturbances-all sources of variation that are highlighted by the variability in SOC responses to harvesting and fuel management (Buma et al., 2014; Esquilin et al., 2008; Jang et al., 2016; Kranabetter and Macadam, 2007; Switzer et al., 2012). As the fuel loading consequences of long-term fire suppression, lack of management, and increasing wildfire threat due to climate change continue to undergo positive feedbacks in temperate mountain systems, decreases in SOC are likely (Kashian et al., 2013; Littell et al., 2009). In the context of dramatically shifting baselines (e.g., vegetation cover, SOC stocks, and "normal" fire seasons), climate-adaptive forest management strategies may help maintain (or mitigate the loss) of SOC. Specific approaches can include assisted migration of adventive species that have evolved under hotter, drier climates, individual genotypes or families from widely distributed species, or upslope treeline migration. These approaches can increase forest structural complexity, SOC, and their resilience to disturbances and climate change (Bojko and Kabala, 2017; Chmura et al., 2011; Hof et al., 2017). In this regard, the significant variation in climate, physiography, and tree genoytpes in temperate mountain systems confers an advantage on climate change-informed forest management, because the performance of distinct lines and populations of key species has long been empirically tested across a range of conditions (Campbell, 1991). Looking to the future, seed zone guidelines such as those provided by Campbell (1991) and others may become very useful for plans involving assisted migration of genotypes, families, or species.

Soils of temperate mountain systems include Cambisols (38%), Podzols (27%), and Luvisols (13%). Detail on these soils from previous sections will not be duplicated here. Rather, three specific cases furnish examples of how highly compressed spatial variation and interactions between topography, climate, vegetation, and soil in temperate mountains can be used to inform SOC management. First, higher elevations and shaded slope aspects may represent physiographic refugia for protecting high-SOC, climate-threatened soils such as Podzols (Nauman et al., 2015). Because of the strong role that snowmelt and acidic vegetation inputs play in driving SOC maintenance in the coarse-textured Podzols, this may be an RSG whose SOC is readily managed by manipulating forest cover types or establishing keystone tree species. Second—and in contrast– the finer-textured Luvisols, with their higher water holding capacity and inherent fertility, may be suitable for more aggressive management tactics (e.g., harvest residue removals, forest floor scalping) that can exert negative impacts on soil and plant productivity on more drought-prone or low-fertility soils (Page-Dumroese et al., 2010). Third, comparing SOC stocks in temperate mountains (Table 11.2) to RSGs that formed during historic grassland-forest tension in this GEZ (Chernozems, 177 Mg C ha^{-1}; Kastanozems, 141 Mg C ha^{-1}) suggest potential for major SOC losses if forest RSGs are converted to grassland RSGs through increasingly frequent, extensive and severe fires. Thus, Chernozems and Kastozems may be priority areas for active fuel management, fire control, and reforestation in order to maintain forests and their higher SOC stocks.

Patterns within domains: boreal domain

Boreal coniferous forest zone

At nearly 19 million km^2, the vast boreal coniferous forest zone is over three-quarters forested. Like other GEZs, SOC in this zone is fire-vulnerable (Harden et al. 2000). In boreal forest, severe fires not only emit large amounts of C from biomass and soils to the atmosphere, but can convert closed-canopy black spruce forests with massive moss, woody debris, and SOC stocks to forest types that are less productive and hold less SOC, such as open-canopy spruce-lichen woodlands (Bastianelli et al., 2017; Bond-Lamberty et al., 2007). Because high-severity fires can be very extensive in this GEZ, one option to mitigate assumed, future fire-induced SOC losses is to use harvesting to create young, low-fuel stands in landscapes dominated by older, high-fuel stands where the former may act as fire-breaks (Boulanger et al., 2017). On the other hand, because vegetation grows slowly, O-horizons hold much of the SOC, and C cycles slowly, C removed from these ecosystems takes a long time to replace (Prescott et al., 2000). Thus, considerable research has been devoted to whether SOC stocks are generally susceptible to loss as a function of harvest removals. Research suggests there may be less SOC loss from thinning than clearcutting (Kishchuk et al., 2016; Kreutzweiser et al., 2008), and that avoiding removal of residues and stumps minimizes SOC loss (Clarke et al., 2015; Jiang et al., 2002; Pare et al., 2002; Walmsley and Godbold, 2010). Opportunities to increase SOC mostly pertain to active interventions in areas that have already burned, where the default management is often "hands-off."

In such settings, felling residual (burned and unburned) snags directly augments O horizons with inputs of slowly decomposing woody material, and can feed back to accelerated O-horizon accumulation via paludification (Hagemann et al., 2010; Jacobs et al., 2015). In the context of climate change, managing vegetation—whether following fire, or in its absence—to transition from slow-growing coniferous taxa with high SOC density (e.g., *Picea* spp.) to faster-growing taxa with reasonably large SOC stocks (e.g., *Populus tremuloides*, Luckai et al., 2012) or more fire-tolerant species with fast rates of postfire SOC recovery (e.g., *Pinus banksiana*, Norris et al., 2009) may, in the long term, have a net positive effect on SOC stocks compared to no intervention. Species selections aimed at increasing SOC stocks may be more realistic in northern European boreal coniferous forests, where management is more widespread, but evidence there is ambivalent regarding the influence of species on SOC stocks (Berg et al., 2009; Stendahl et al., 2010).

As in other GEZs, SOC stocks vary widely in the boreal coniferous forest, most notably with hydrology, bryophyte cover, and soil characteristics (Harden et al., 1997; Kranabetter, 2009; Shetler et al., 2008). The most extensive RSGs are Cryosols (25%), Podzols (20%), Albeluvisols (18%), Cambisols (16%), and Histosols (11%). The extensive area and very high SOC density of Cryosols is alarming in the face of a warming climate and the vulnerability of permafrost SOC (Schuur et al., 2015). Rising temperatures and the additional warming to which Earth is committed due to past and present CO_2 emissions suggest that some share of the SOC in Crysols will not remain frozen. Thawing of Cryosols and emission of permafrost C could conceivably be delayed in key areas by felling snags and trees (as mentioned above for promoting paludification), insulating the soil with deep organic detritus and possibly increasing snowpack accumulation and albedo (Gray 1975; Troendle and King, 1985). That said, current SOC stocks in this RSG (Table 11.2) suggest that transitions from Cryosols to Albeluvisols, Podzols, or Cambisols would maintain SOC stocks at 75%–85% their current values. Furthermore, where saturation persists (or expands as permafrost melts) and inhibits decomposition, SOC may even increase: on average, Histosols have 65% higher SOC densities than Cryosols in the boreal coniferous forest zone. While this speculation is based on assumptions that are themselves sensitive to changing climate, it highlights three management activities that could conceivably maintain SOC stocks in key areas. First, maintain inputs of organic matter to the soil surface, e.g., through harvest residue retention, as to keep soils as cold and wet as possible. Second, where possible, manage hydrology directly, as through impoundments, to maintain or increase saturation. Third, where critical soil transitions occur—especially on formerly frozen upland soils—utilize active management to minimize fires, reforest after disturbance, and select for the southernmost species that will tolerate current and future disturbance regimes for a given soil texture and drainage condition. For the latter of these three activities, this likely means more nutrient-demanding taxa such as *P. tremuloides and P. tremula* (and, perhaps *Betula* spp. currently more common to the south) on finer-textured, more cation-rich Albeluvisols, and southern genotypes of *P. banksiana* and *P. sylvestris* on coarser and more acidic soils likely to develop into Podzols.

Boreal tundra woodland

The boreal tundra woodland ecological zone (10 million km^2), which is about one-half forested, spans a range of woody plant densities from shrub tundra to woodland. This GEZ has received much attention for SOC vulnerability to permafrost thaw due to warmer soil temperatures, longer mineralization seasons, and feedbacks involving wildfires and hydrology (Hobbie et al., 2000; O'Donnell et al., 2011; Wilmking et al., 2006). As forests expand into tundra during permafrost thaw, detrital inputs to soil may increase, but changes in SOC may occur in either direction (Hartley et al., 2012; Parker et al., 2015; Steltzer, 2004). In contrast to this ambiguity, it is clear that N inputs have positive effects on SOC. In northern Europe, where a larger fraction of boreal tundra woodlands are managed or subject to atmospheric N deposition, N inputs can improve tree growth for economic purposes, while increasing detritus inputs, slowing decomposition, and increasing C storage in O-horizons (Berg, 2000; Hyvonen et al., 2007; Makipaa, 1995; Prescott, 2010). If active management follows forest expansion into this GEZ, it is likely possible to increase SOC storage through N-fertilization. However, C accumulation in surface detritus and litter is not without risk; these materials are fuels and fire management will therefore be critical to realize the potential C gains associated with fertilization.

The most widespread soils in the boreal tundra woodland are Cambisols, Cryosols, and Histosols, where the SOC densities of these widely distributed soils are 10%—30% higher than boreal coniferous forests and 70%—130% than temperate continental forests (Table 11.2). Superficially, these three soils are quite distinct (Table 11.1), but in boreal landscapes they often inter-grade based on their depth to permafrost and amounts of organic versus mineral materials. The presence of mineral soil materials is a critical factor for SOC management in these soils because it can determine whether SOC that was previously frozen or saturated has secondary stability via clay minerals, Al and Fe hydroxides, or Ca bridging (Table 11.12; Tarnocai and Bockheim, 2011; Shaw et al., 2008). In many cases, physical changes due to warming and drying (e.g., drainage, subsidence) can be sufficient to shift soils currently classified as Cryosols or Histosols to Cambisols, the latter of which are frequent associates on warmer, drier, or higher topographic positions in Cryosol or Histosol-dominated landscapes. The extensive area of Cambisols in this GEZ indicates the very wide range of parent materials, landforms, vegetation types, and SOC stocks associated with these soils, highlighting the difficulty of generalizing at broad scales. In ecoregions, landscapes, or landforms where Cambisols are reasonably accessible, freely drained, and capable of supporting forest, they afford the widest latitude in SOC management options because of their physical characteristics. Because their limited degree of development in many places results from low vegetation productivity and short thaw seasons, practices that increase forest stocking

Table 11.12 Management considerations and supporting references for key RSGs discussed in the Boreal tundra woodland GEZ.

Reference Soil Group (RSG)	Percent of forest area	Management considerations	References
Cambisols	38	Burning, site-dependent warming and drying may increase C transfer to and stabilization in subsoils over long timescales	Smith et al. (2011); Shaw et al. (2008)
Cryosols	22	Mineral soils have C stabilization mechanisms that organic Cryosols lack; accessibility for management is likely to decline with thawing	Tarnocai and Bockheim (2011)
Histosols	21	Saturated conditions are critical to protect SOC from mineralization and fire	Hugelius et al. (2014)

and production rates (e.g., supplemental planting, fertilization) on climatically favorable sites may in the long run lead to SOC gains as detrital inputs accumulate and pedogenesis advances. The very high SOC densities of Histosols in the boreal tundra woodland (nearly double those of Cryosols) reflects the difficulty of constraining their distribution and depth (Hugelius et al., 2014), but suggests that thawing alone does not mean the release of all C held in Cryosols.

Boreal mountain systems

Boreal mountain systems (14 million km^2 and about two-thirds forested) represent an extreme tension zone in the context of climate change-they combine the vulnerability of permafrost SOC with strong localized contrasts in landforms, climate, vegetation, fire, and hydrology, all of which strongly affect SOC stocks and distributions (Johnson et al., 2011a,b). Because these mountain systems intergrade with temperate mountain systems, they share several similar sources of vulnerability; most notably, insect outbreak and fire interactions, which will likely continue to advance northwards as climates warm (Kurz et al., 2013). Similarly, the tundra-to-forest transition that is occurring laterally in boreal tundra woodlands is also occurring with altitude in boreal mountain systems. This transition clearly increases C storage in aboveground vegetation, but because its impacts on SOC are difficult to generalize (Kammer et al., 2009; Kirdyanov et al., 2012), it is not clear whether assisted upslope treeline migration can be used to increase SOC. However, because the intensification of wildfires now occurring in temperate mountain systems will likely advance into the boreal domain, reforestation will become an important means of mitigating SOC losses due to fire. Where possible, selecting species with climate-fire interactions in mind will help ensure rapid and robust reforestation, in order to more quickly restore forest cover in what will likely be an increasingly challenging environment for forest establishment as climate change progresses (Cai et al., 2013).

The most widespread soils in boreal mountain systems are similar to other boreal zones, including Cryosols (39%), Cambisols (29%), and Podzols (20%), but there are relatively fewer Histosols (9%) and a notable presence of Andosols (2%). In some ways, boreal mountain systems are similar to temperate mountain systems in terms of SOC management, most notably in that topography may be a useful tool for maintaining soils with large, vulnerable SOC stocks in the face of climate change (e.g., Cryosols) on north slopes and high elevations (Bockheim and Munroe, 2014; Dymov et al., 2013). On the other hand, while management might conceivably be used to promote SOC gains in Histosols through forest or hydrologic management (e.g., via tree felling or impoundment) in more low-lying, low-relief boreal zones, steep topography evidently limits Histosol development in boreal mountain systems and indicates why such activities are not favorable here. The Andosols of boreal mountain systems, while low in proportion, still occupy >200,000 km^2; given their greater extent and very high productivity in temperate zones to the south, the potential appears to exist for more intensive management to follow the warming climate northwards.

Concluding remarks

In this chapter, we assessed forest SOC management across discrete ecological zones and soil types. This survey was not comprehensive, but intended to identify broad generalizable patterns while recognizing important site and landscape-level variability. This review also was intended to be an inclusive and defensible representation of management activities that affect forest SOC. Recognizing that soils are diverse and offer many exceptions, we offer three generalizations that guide our understanding of vulnerability of SOC to global change, and how management may aid in maintaining or enhancing SOC:

- Where soils store large fractions of C in O horizons or topsoils, disturbances that alter the processes maintaining those stocks (e.g. detrital inputs, saturated or frozen conditions) increase SOC vulnerability; management that restores or maintains those processes may have a positive effect on SOC storage. At best, that positive effect may be a net gain, but mere recovery of lost SOC is the more realistic (and nonetheless challenging) target.
- Carbon held in subsoils may be less vulnerable to short-term losses (e.g. by wildfire or erosion) due to physical isolation, aggregation, or organo-mineral association, but when it is negatively impacted by disturbances, management options for recovery are few, and likely slow to materialize.
- Given that SOC is only one of the many components that comprise the integrated unit that we call a forest ecosystem, soils and C stocks are not managed in isolation of other ecosystem components. Managing SOC may be best achieved by managing other factors, such as vegetation, according to the limits imposed by still other factors such as climate and landform. Viewed as a part of the whole, forest soil C benefits become one outcome of management activities directed at the entire system, which may take decades to appreciate. Likewise, management to influence SOC can have unintended or undesirable impacts on other components, and is thus best considered in context.

Ultimately, there is a clear need for more research—especially on operational treatments and controlled experiments—to fill spatial and conceptual gaps in our understanding, and to extend the timescales over which we measure changes in forest SOC. Timescale becomes all the more important when considering that forests, soils, research agendas, researchers and forest managers all change, and at different rates. Research agendas change quickly, and with them the terminology, context and justification for studies of forest soil C. Thorough reviews (e.g., McFee and Kelly, 1995) and seminal works (e.g., Romell and Heiberg, 1931; Lutz and Chandler, 1946) as relevant today as ever, become forgotten as new agency directives and funding sources evolve. Researchers and managers with experience in forest SOC leave the field or retire; those most persistent have careers long enough to witness a full cycle of change in forest vegetation following a management intervention at a site they know well. Soils, however, have histories that vastly exceed those of agencies, people, and trees—they are recorders. Soils present evidence of events that have occurred over much longer timescales than do other parts of forest ecosystems, such as the vegetation with which people are so often concerned. Soils are just as readily modified by global change and direct intervention, but their changes are more difficult to observe and may take longer to manifest than what we observe aboveground. Similarly, when abused, soils take long to recover, especially at the depths where the most stable, oldest SOC is typically sequestered. In sum, these realities influence not only our attempts, but our methods and abilities to detect management impacts on forest soils, and assess their role in longer-term biosphere C sequestration.

References

Abril, A., Barttfeld, P., Bucher, E.H., 2005. The effect of fire and overgrazing, disturbs on soil carbon balance in the Dry Chaco forest. Forest Ecology and Management 206, 399–405.

Adair, E.C., Parton, W.J., Del Grosso, S.J., Silver, W.L., Harmon, M.E., Hall, S.A., Burke, I.C., Hart, S.C., 2008. Simple three-pool model accurately describes patterns of long-term litter decomposition in diverse climates. Global Change Biology 14, 2636–2660.

Agee, J.K., Skinner, C.N., 2005. Basic principles of forest fuel reduction treatments. Forest Ecology and Management 211, 83−96.

Alvarez-Arteaga, G., Calderon, N.E.G., Krasilnikov, P., Garcia-Oliva, F., 2013. Carbon storage in montane cloud forests in Sierra Norte of Oaxaca, Mexico. Agrociencia 47, 171−180.

Alvarez-Berros, N.L., Aide, T.M., 2015. Global demand for gold is another threat for tropical forests. Environmental Research Letters 10, 014006, 2014.

Anderegg, W.R.L., Hicke, J.A., Fisher, R.A., Allen, C.D., Aukema, J., Bentz, B., Hood, S., Lichstein, J.W., Macalady, A.K., McDowell, N., Pan, Y.D., Raffa, K., Sala, A., Shaw, J.D., Stephenson, N.L., Tague, C., Zeppel, M., 2015. Tree mortality from drought, insects, and their interactions in a changing climate. New Phytologist 208, 674−683.

Aoki, M., Fujii, K., Kitayama, K., 2012. Environmental control of root exudation of low-molecular weight organic acids in tropical rainforests. Ecosystems 15, 1194−1203.

Asano, M., Wagai, R., 2014. Evidence of aggregate hierarchy at micro- to submicron scales in an allophanic Andisol. Geoderma 216, 62−74.

Assefa, D., Rewald, B., Sanden, H., Rosinger, C., Abiyu, A., Yitaferu, B., Godbold, D.L., 2017. Deforestation and land use strongly effect soil organic carbon and nitrogen stock in Northwest Ethiopia. Catena 153, 89−99.

Balch, J.K., Brando, P.M., Nepstad, D.C., Coe, M.T., Silverio, D., Massad, T.J., Davidson, E.A., Lefebvre, P., Oliveira-Santos, C., Rocha, W., Cury, R.T.S., Parsons, A., Carvalho, K.S., 2015. The Susceptibility of Southeastern Amazon Forests to Fire: Insights from a Large-Scale Burn Experiment. Bioscience 65, 893−905.

Baird, M., Zabowski, D., Everett, R.L., 1999. Wildfire effects on carbon and nitrogen in inland coniferous forests. Plant and Soil 209, 233−243.

Baisden, W.T., Parfitt, R.L., Ross, C., Schipper, L.A., Canessa, S., 2013. Evaluating 50 years of time-series soil radiocarbon data: towards routine calculation of robust C residence times. Biogeochemistry 112, 129−137.

Baptista, I., Ritsema, C., Querido, A., Ferreira, A.D., Geissen, V., 2015. Improving rainwater-use in Cabo Verde drylands by reducing runoff and erosion. Geoderma 237, 283−297.

Barthold, F.K., Wiesmeier, M., Breuer, L., Frede, H.G., Wu, J., Blank, F.B., 2013. Land use and climate control the spatial distribution of soil types in the grasslands of Inner Mongolia. Journal of Arid Environments 88, 194−205.

Bastianelli, C., Ali, A.A., Beguin, J., Bergeron, Y., Grondin, P., Hely, C., Pare, D., 2017. Boreal coniferous forest density leads to significant variations in soil physical and geochemical properties. Biogeosciences 14, 3445−3459.

Batjes, N.H., 2008. Mapping soil carbon stocks of Central Africa using SOTER. Geoderma 146, 58−65.

Beedy, T.L., Snapp, S.S., Akinnifesi, F.K., Sileshi, G.W., 2010. Impact of Gliricidia sepium intercropping on soil organic matter fractions in a maize-based cropping system. Agriculture Ecosystems & Environment 138, 139−146.

Bennett, L.T., Bruce, M.J., Machunter, J., Kohout, M., Krishnaraj, S.J., Aponte, C., 2017. Assessing fire impacts on the carbon stability of fire-tolerant forests. Ecological Applications 27, 2497−2513.

Berg, B., 2000. Litter decomposition and organic matter turnover in northern forest soils. Forest Ecology and Management 133, 13−22.

Berg, B., Johansson, M.B., Nilsson, A., Gundersen, P., Norell, L., 2009. Sequestration of carbon in the humus layer of Swedish forests − direct measurements. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 39, 962−975.

Bockheim, J.G., 2003. Genesis of bisequal soils on acidic drift in the upper Great Lakes region, USA. Soil Science Society of America Journal 67, 612−619.

Bockheim, J., 2011. Distribution and genesis of ortstein and placic horizons in soils of the USA: a review. Soil Science Society of America Journal 75, 994−1005.

Bockheim, J.G., Munroe, J.S., 2014. Organic carbon pools and genesis of alpine soils with permafrost: A review. Arctic Antarctic and Alpine Research 46, 987−1006.

Bockheim, J.G., Gennadiyev, A.N., Hammer, R.D., Tandarich, J.P., 2005. Historical development of key concepts in pedology. Geoderma 124, 23−36.

Bojko, O., Kabala, C., 2017. Organic carbon pools in mountain soils−sources of variability and predicted changes in relation to climate and land use changes. Catena 149, 209−220.

Bond-Lamberty, B., Peckham, S.D., Ahl, D.E., Gower, S.T., 2007. Fire as the dominant driver of central Canadian boreal forest carbon balance. Nature 450, 89-+.

Bonino, E.E., 2006. Changes in carbon pools associated with a land-use gradient in the Dry Chaco, Argentina. Forest Ecology and Management 223, 183−189.

Bormann, B.T., Homann, P.S., Darbyshire, R.L., Morrissette, B.A., 2008. Intense forest wildfire sharply reduces mineral soil C and N: the first direct evidence. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 38, 2771−2783.

Boulanger, Y., Girardin, M., Bernier, P.Y., Gauthier, S., Beaudoin, A., Guindon, L., 2017. Changes in mean forest age in Canada's forests could limit future increases in area burned but compromise potential harvestable conifer volumes. Canadian Journal of Forest Research 47, 755−764.

Bravo-Oviedo, A., Ruiz-Peinado, R., Modrego, P., Alonso, R., Montero, G., 2015. Forest thinning impact on carbon stock and soil condition in Southern European populations of P. *sylvestris* L. Forest Ecology and Management 357, 259−267.

Brown, S., Lugo, A.E., 2017. Trailblazing the carbon cycle of tropical forests from Puerto Rico. Forests 8.

Buma, B., Wessman, C.A., 2013. Forest resilience, climate change, and opportunities for adaptation: a specific case of a general problem. Forest Ecology and Management 306, 216−225.

Buma, B., Poore, R.E., Wessman, C.A., 2014. Disturbances, their interactions, and cumulative effects on carbon and charcoal stocks in a forested ecosystem. Ecosystems 17, 947−959.

Butnor, J.R., Johnsen, K.H., Sanchez, F.G., Nelson, C.D., 2012. Impacts of pine species, stump removal, cultivation, and fertilization on soil properties half a century after planting. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 42, 675−685.

Cai, W.H., Yang, J., Liu, Z.H., Hu, Y.M., Weisberg, P.J., 2013. Post-fire tree recruitment of a boreal larch forest in Northeast China. Forest Ecology and Management 307, 20−29.

Camenzind, T., Hattenschwiler, S., Treseder, K.K., Lehmann, A., Rillig, M.C., 2018. Nutrient limitation of soil microbial processes in tropical forests. Ecological Monographs 88, 4−21.

Campbell, R.K., 1991. Soils, seed-zone maps, and physiography−guidelines for seed transfer of douglas-fir in southwestern Oregon. Forest Science 37, 973−986.

Chaopricha, N.T., Marin-Spiotta, E., 2014. Soil burial contributes to deep soil organic carbon storage. Soil Biology and Biochemistry 69, 251−264.

Chappell, H.N., Cole, D.W., Gessel, S.P., Walker, R.B., 1991. Forest fertilization research and practice in the pacific-northwest. Fertilizer Research 27, 129−140.

Chen, C.R., Xu, Z.H., 2005. Soil carbon and nitrogen pools and microbial properties in a 6-year-old slash pine plantation of subtropical Australia: impacts of harvest residue management. Forest Ecology and Management 206, 237−247.

Chen, C.R., Xu, Z.H., Mathers, N.J., 2004. Soil carbon pools in adjacent natural and plantation forests of subtropical Australia. Soil Science Society of America Journal 68, 282−291.

Chen, L.C., Wang, H., Yu, X., Zhang, W.D., Lu, X.T., Wang, S.L., 2014. Recovery time of soil carbon pools of conversional Chinese fir plantations from broadleaved forests in subtropical regions, China. The Science of the Total Environment 587, 296−304.

Chiti, T., Grieco, E., Perugini, L., Rey, A., Valentini, R., 2014. Effect of the replacement of tropical forests with tree plantations on soil organic carbon levels in the Jomoro district, Ghana. Plant and Soil 375, 47−59.

Chmura, D.J., Anderson, P.D., Howe, G.T., Harrington, C.A., Halofsky, J.E., Peterson, D.L., Shaw, D.C., St Clair, J.B., 2011. Forest responses to climate change in the northwestern United States: ecophysiological foundations for adaptive management. Forest Ecology and Management 261, 1121−1142.

Clarke, N., Gundersen, P., Jonsson-Belyazid, U., Kjonaas, O.J., Persson, T., Sigurdsson, B.D., Stupak, I., Vesterdal, L., 2015. Influence of different tree-harvesting intensities on forest soil carbon stocks in boreal and northern temperate forest ecosystems. Forest Ecology and Management 351, 9−19.

Conti, G., Kowaljow, E., Baptist, F., Rumpel, C., Cuchietti, A., Harguindeguy, N.P., Diaz, S., 2016. Altered soil carbon dynamics under different land-use regimes in subtropical seasonally-dry forests of central Argentina. Plant and Soil 403, 375−387.

Cook, R.L., Binkley, D., Stape, J.L., 2016. Eucalyptus plantation effects on soil carbon after 20 years and three rotations in Brazil. Forest Ecology and Management 359, 92−98.

Crow, S.E., Swanston, C.W., Lajtha, K., Brooks, J.R., Keirstead, H., 2007. Density fractionation of forest soils: methodological questions and interpretation of incubation results and turnover time in an ecosystem context. Biogeochemistry 85, 69−90.

Crow, S.E., Reeves, M., Schubert, O.S., Sierra, C.A., 2015. Optimization of method to quantify soil organic matter dynamics and carbon sequestration potential in volcanic ash soils. Biogeochemistry 123, 27−47.

Cui, X.Y., Gao, F., Song, J.F., Sang, Y., Sun, J.B., Di, X.Y., 2014. Changes in soil total organic carbon after an experimental fire in a cold temperate coniferous forest: a sequenced monitoring approach. Geoderma 226, 260−269.

Cunningham, S.C., Cavagnaro, T.R., Mac Nally, R., Paul, K.I., Baker, P.J., Beringer, J., Thomson, J.R., Thompson, R.M., 2015. Reforestation with native mixed-species plantings in a temperate continental climate effectively sequesters and stabilizes carbon within decades. Global Change Biology 21, 1552−1566.

Cusack, D.F., Silver, W.L., Torn, M.S., McDowell, W.H., 2011. Effects of nitrogen additions on above- and belowground carbon dynamics in two tropical forests. Biogeochemistry 104, 203−225.

Cusack, D.F., Lee, J.K., McCleery, T.L., LeCroy, C.S., 2015. Exotic grasses and nitrate enrichment alter soil carbon cycling along an urban-rural tropical forest gradient. Global Change Biology 21, 4481−4496.

Cusack, D.F., Karpman, J., Ashdown, D., Cao, Q., Ciochina, M., Halterman, S., Lydon, S., Neupane, A., 2016. Global change effects on humid tropical forests: evidence for biogeochemical and biodiversity shifts at an ecosystem scale. Reviews of Geophysics 54, 523−610.

Cusack, D.F., Markesteijn, L., Condit, R., Lewis, O.T., Turner, B.L., 2018. Soil carbon stocks across tropical forests of Panama regulated by base cation effects on fine roots. Biogeochemistry 137, 253−266.

da Costa, A., Albuquerque, J.A., da Costa, A., Warmling, M.T., Magro, B.A., 2016. Pine harvest impact on soil structure of a Dystric Cambisol (humic). Revista Brasileira de Ciência do Solo 40.

de Blecourt, M., Brumme, R., Xu, J.C., Corre, M.D., Veldkamp, E., 2013. Soil carbon stocks decrease following conversion of secondary forests to rubber (*Hevea brasiliensis*) plantations. PLoS One 8.

de la Rosa, J.M., Rosado, M., Paneque, M., Miller, A.Z., Knicker, H., 2018. Effects of aging under field conditions on biochar structure and composition: implications for biochar stability in soils. The Science of the Total Environment 613, 969−976.

Dawoe, E.K., Quashie-Sam, J.S., Oppong, S.K., 2014. Effect of land-use conversion from forest to cocoa agroforest on soil characteristics and quality of a Ferric Lixisol in lowland humid Ghana. Agroforestry Systems 88, 87−99.

Diaz-Pines, E., Rubio, A., Van Miegroet, H., Montes, F., Benito, M., 2015. Does tree species composition control soil organic carbon pools in Mediterranean mountain forests? Forest Ecology and Management 262, 1895−1904.

Dimeyeva, L.A., Sitpayeva, G.T., Ussen, K., Orlovsky, L., Ablaikhanov, E., Islamgulova, A.F., Zhang, Y., Zhang, J., Suleimenova, N.S., 2016. Meadow vegetation the Zhetysu Alatau Mountains. Applied Ecology and Environmental Research 14, 375–398.

Dokuchaev, V.V., 1879. Tchernozéme (terre noire) de la Russie d'Europe (Chernozem [black earth] of European Russia [four lectures of 1877]). Société Impériale Libre ÉconomiqueTrénke and Fusnot, St. Pétersbourg, 66 pp.

Don, A., Schumacher, J., Freibauer, A., 2010. Impact of tropical land-use change on soil organic carbon stocks—a meta-analysis. Global Change Biology 17, 1658–1670.

Dorji, T., Odeh, I.O.A., Field, D.J., Baillie, I.C., 2014. Digital soil mapping of soil organic carbon stocks under different land use and land cover types in montane ecosystems, Eastern Himalayas. Forest Ecology and Management 318, 91–102.

Dymov, A.A., Zhangurov, E.V., Startsev, V.V., 2013. Soils of the northern part of the Subpolar Urals: morphology, physicochemical properties, and carbon and nitrogen pools. Eurasian Soil Science 46, 459–467.

Esquilin, A.E.J., Stromberger, M.E., Shepperd, W.D., 2008. Soil scarification and wildfire interactions and effects on microbial communities and carbon. Soil Science Society of America Journal 72, 111–118.

European Space Agency and Université Catholique de Louvain, 2010. GlobCover 2009: Global Land Cover Map. http://due.esrin.esa.int/page_globcover.php.

Falahatkar, S., Hosseini, S.M., Mahiny, A., Ayoubi, S., Wang, S.Q., 2014. Soil organic carbon stock as affected by land use/cover changes in the humid region of northern Iran. Journal of Mountain Science 11, 507–518.

Fan, S.H., Guan, F.Y., Xu, X.L., Forrester, D.I., Ma, W., Tang, X.L., 2016. Ecosystem carbon stock loss after land use change in subtropical forests in China. Forests 7.

Fearnside, P.M., Barbosa, R.I., 1998. Soil carbon changes from conversion of forest to pasture in Brazilian Amazonia. Forest Ecology and Management 108, 147–166.

Fernandez-Romero, M.L., Lozano-Garcia, B., Parras-Alcantara, L., 2014. Topography and land use change effects on the soil organic carbon stock of forest soils in Mediterranean natural areas. Agriculture, Ecosystems and Environment 195, 1–9.

Fletcher, M.S., Wolfe, B.B., Whitlock, C., Pompeani, D.P., Heijnis, H., Haberle, S.G., Gadd, P.S., Bowman, D., 2014. The legacy of mid-Holocene fire on a Tasmanian montane landscape. Journal of Biogeography 41, 476–488.

Fonte, S.J., Barrios, E., Six, J., 2010. Earthworms, soil fertility and aggregate-associated soil organic matter dynamics in the Quesungual agroforestry system. Geoderma 155, 320–328.

Food and Agriculture Organization of the United Nations, 2013. Global Ecological Zones, second ed. Digital Map. http://ref.data.fao.org/map?entryId=2fb209d0-fd34-4e5e-a3d8-a13c241eb61b&tab=metadata.

Fox, T.R., 2000. Sustained productivity in intensively managed forest plantations. Forest Ecology and Management 138, 187–202.

Fox, T.R., Jokela, E.J., Allen, H.L., 2007. The development of pine plantation silviculture in the southern United States. Journal of Forestry 105, 337–347.

Freifelder, R.R., Vitousek, P.M., D'Antonio, C.M., 1998. Microclimate change and effect on fire following forest-grass conversion in seasonally dry tropical woodland. Biotropica 30, 286–297.

Garcia-Oliva, F., Lancho, J.F.G., Montano, N.M., Islas, P., 2006. Soil carbon and nitrogen dynamics followed by a forest-to-pasture conversion in western Mexico. Agroforestry Systems 66, 93–100.

Garland, G., Bunemann, E.K., Oberson, A., Frossard, E., Snapp, S., Chikowo, R., Six, J., 2018. Phosphorus cycling within soil aggregate fractions of a highly weathered tropical soil: A conceptual model. Soil Biology & Biochemistry 116, 91–98.

Gray, H.C., 1975. Airflow patterns and snow accumulation in a forest clearing. In: Proceedings of the 43rd Western Snow Conference. Western Snow Conference, Coronado, CA, pp. 106–113.

Gillin, C.P., Bailey, S.W., McGuire, K.J., Gannon, J.P., 2015. Mapping of hydropedologic spatial patterns in a steep headwater catchment. Soil Science Society of America Journal 79, 440−453.

Gray, A.N., Whittier, T.R., 2014. Carbon stocks and changes on Pacific Northwest national forests and the role of disturbance, management, and growth. Forest Ecology and Management 328, 167−178.

Girmay, G., Singh, B.R., Mitiku, H., Borresen, T., Lal, R., 2008. Carbon stocks in Ethiopian soils in relation to land use and soil management. Land Degradation and Development 19, 351−367.

Griscom, H.P., Ashton, M.S., 2011. Restoration of dry tropical forests in Central America: a review of pattern and process. Forest Ecology and Management 261, 1564−1579.

Guedes, B.S., Olsson, B.A., Karltun, E., 2016. Effects of 34-year-old Pinus taeda and Eucalyptus grandis plantations on soil carbon and nutrient status in former Miombo forest soils. Global Ecology and Conservation 8, 190−202.

Guillaume, T., Damris, M., Kuzyakov, Y., 2015. Losses of soil carbon by converting tropical forest to plantations: erosion and decomposition estimated by delta C-13. Global Change Biology 21, 3548−3560.

Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: A meta analysis. Global Change Biology 8, 345−360.

Hagemann, U., Moroni, M.T., Gleissner, J., Makeschin, F., 2010. Accumulation and preservation of dead wood upon burial by bryophytes. Ecosystems 13, 600−611.

Hall, S.J., McNicol, G., Natake, T., Silver, W.L., 2015. Large fluxes and rapid turnover of mineral-associated carbon across topographic gradients in a humid tropical forest: insights from paired C-14 analysis. Biogeosciences 12, 2471−2487.

Harden, J.W., O'Neill, K.P., Trumbore, S.E., Veldhuis, H., Stocks, B.J., 1997. Moss and soil contributions to the annual net carbon flux of a maturing boreal forest. Journal of Geophysical Research-Atmospheres 102, 28805−28816.

Harden, J.W., Trumbore, S.E., Stocks, B.J., Hirsch, A., Gower, S.T., O'Neill, K.P., Kasischke, E.S., 2000. The role of fire in the boreal carbon budget. Global Change Biology 6, 174−184.

Hartley, I.P., Garnett, M.H., Sommerkorn, M., Hopkins, D.W., Fletcher, B.J., Sloan, V.L., Phoenix, G.K., Wookey, P.A., 2012. A potential loss of carbon associated with greater plant growth in the European Arctic. Nature Climate Change 2, 875−879.

He, Y., Xu, Z.H., Chen, C.R., Burton, J., Ma, Q., Ge, Y., Xu, J.M., 2008. Using light fraction and macroaggregate associated organic matters as early indicators for management-induced changes in soil chemical and biological properties in adjacent native and plantation forests of subtropical Australia. Geoderma 147, 116−125.

Heckman, K., Lawrence, C.R., Harden, J.W., 2018. A sequential selective dissolution method to quantify storage and stability of organic carbon associated with Al and Fe hydroxide phases. Geoderma 312, 24−35.

Heitkotter, J., Heinze, S., Marschner, B., 2017. Relevance of substrate quality and nutrients for microbial C-turnover in top-and subsoil of a Dystric Cambisol. Geoderma 302, 89−99.

Helfrich, M., Flessa, H., Dreves, A., Ludwig, B., 2010. Is thermal oxidation at different temperatures suitable to isolate soil organic carbon fractions with different turnover? Journal of Plant Nutrition and Soil Science 173, 61−66.

Hengl, T., de Jesus, J.M., Heuvelink, G.B.M., Gonzalez, M.R., Kilibarda, M., Blagotic, A., Shangguan, W., Wright, M.N., Geng, X.Y., Bauer-Marschallinger, B., Guevara, M.A., Vargas, R., MacMillan, R.A., Batjes, N.H., Leenaars, J.G.B., Ribeiro, E., Wheeler, I., Mantel, S., Kempen, B., 2017. SoilGrids250m: Global gridded soil information based on machine learning. PLoS One 12.

Hobbie, S.E., Schimel, J.P., Trumbore, S.E., Randerson, J.R., 2000. Controls over carbon storage and turnover in high-latitude soils. Global Change Biology 6, 196−210.

Hof, A.R., Dymond, C.C., Mladenoff, D.J., 2017. Climate change mitigation through adaptation: the effectiveness of forest diversification by novel tree planting regimes. Ecosphere 8.

Homann, P.S., Harmon, M., Remillard, S., Smithwick, E.A.H., 2005. What the soil reveals: potential total ecosystem C stores of the Pacific Northwest region, USA. Forest Ecology and Management 220, 270–283.
Homann, P.S., Bormann, B.T., Boyle, J.R., Darbyshire, R.L., Bigley, R., 2008. Soil C and N minimum detectable changes and treatment differences in a multi-treatment forest experiment. Forest Ecology and Management 255, 1724–1734.
Homann, P.S., Bormann, B.T., Morrissette, B.A., Darbyshire, R.L., 2015. Post wildfire soil trajectory linked to prefire ecosystem structure in douglas-fir forest. Ecosystems 18, 260–273.
Hoogmoed, M., Cunningham, S.C., Thomson, J.R., Baker, P.J., Beringer, J., Cavagnaro, T.R., 2012. Does afforestation of pastures increase sequestration of soil carbon in Mediterranean climates? Agriculture, Ecosystems and Environment 159, 176–183.
Hoogmoed, M., Cunningham, S.C., Baker, P.J., Beringer, J., Cavagnaro, T.R., 2014. Is there more soil carbon under nitrogen-fixing trees than under non-nitrogen-fixing trees in mixed-species restoration plantings? Agriculture, Ecosystems and Environment 188, 80–84.
Hopmans, P., Elms, S.R., 2009. Changes in total carbon and nutrients in soil profiles and accumulation in biomass after a 30-year rotation of Pinus radiata on podzolized sands: impacts of intensive harvesting on soil resources. Forest Ecology and Management 258, 2183–2193.
Houghton, R.A., 2003. Revised estimates of the annual net flux of carbon to the atmosphere from changes in land use and land management 1850-2000. Tellus Series B Chemical and Physical Meteorology 55, 378–390.
Hua, K.K., Wang, D.Z., Guo, X.S., Guo, Z.B., 2014. Carbon sequestration efficiency of organic amendments in a long-term experiment on a vertisol in Huang-Huai-Hai plain, China. PLoS One 9.
Hua, K.K., Zhu, B., Wang, X.G., Tian, L.L., 2016. Forms and fluxes of soil organic carbon transport via overland flow, interflow, and soil erosion. Soil Science Society of America Journal 80, 1011–1019.
Hugelius, G., Strauss, J., Zubrzycki, S., Harden, J.W., Schuur, E.A.G., Ping, C.L., Schirrmeister, L., Grosse, G., Michaelson, G.J., Koven, C.D., O'Donnell, J.A., Elberling, B., Mishra, U., Camill, P., Yu, Z., Palmtag, J., Kuhry, P., 2014. Estimated stocks of circumpolar permafrost carbon with quantified uncertainty ranges and identified data gaps. Biogeosciences 11, 6573–6593.
Hughes, J.C., 1982. High-Gradient magnetic separation of some soil clays from Nigeria, Brazil and Colombia .1. The interrelationships of iron and aluminum extracted by acid ammonium oxalate and carbon. Journal of Soil Science 33, 509–519.
Hyvonen, R., Agren, G.I., Linder, S., Persson, T., Cotrufo, M.F., Ekblad, A., Freeman, M., Grelle, A., Janssens, I.A., Jarvis, P.G., Kellomaki, S., Lindroth, A., Loustau, D., Lundmark, T., Norby, R.J., Oren, R., Pilegaard, K., Ryan, M.G., Sigurdsson, B.D., Stromgren, M., van Oijen, M., Wallin, G., 2007. The likely impact of elevated [CO_2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. New Phytologist 173, 463–480.
Ingelmo, F., Molina, M.J., Ruiz, M., 2001. Water stable aggregates in simulated eroded soils affected by fire intensity. In: Villacampa, Y., Brebbia, C.A., Uso, J.L. (Eds.), Ecosystems and Sustainable Development Iii, pp. 311–320.
International Union of Soil Science Working Group WRB, 2006. World Reference Base for Soil Resources 2006. World Soil Resources Reports No. 103. FAO, Rome.
Jacobs, J., Work, T., Pare, D., Bergeron, Y., 2015. Paludification of boreal soils reduces wood decomposition rates and increases wood-based carbon storage. Ecosphere 6.
Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson, D.W., Minkkinen, K., Byrne, K.A., 2007. How strongly can forest management influence soil carbon sequestration? Geoderma 137, 253–268.
Jang, W., Page-Dumroese, D.S., Keyes, C.R., 2016. Long-term soil changes from forest harvesting and residue management in the northern rocky mountains. Soil Science Society of America Journal 80, 727–741.

Janowiak, M.K., Webster, C.R., 2017. Promoting ecological sustainability in woody biomass harvesting. Journal of Forestry 108, 16–23.

Jenny, H., 1941. Factors of Soil Formation: A System of Quantitative Pedology. McGraw-Hill, New York, NY p. 281.

Jeyanny, V., Balasundram, S.K., Husni, M.H.A., Rasidah, K.W., Arifin, A., 2013. Spatial variability of selected forest soil properties related to carbon management in tropical lowland and montane forests. Journal of Tropical Forest Science 25, 577–591.

Jiang, H., Apps, M.J., Peng, C.H., Zhang, Y.L., Liu, J.X., 2002. Modelling the influence of harvesting on Chinese boreal forest carbon dynamics. Forest Ecology and Management 169, 65–82.

Jiang, P.K., Meng, C.F., Zhou, G.M., Xu, Q.F., 2011. Comparative study of carbon storage in different forest stands in subtropical China. The Botanical Review 77, 242–251.

Jimenez-Gonzalez, M.A., De la Rosa, J.M., Jimenez-Morillo, N.T., Almendros, G., Gonzalez-Perez, J.A., Knicker, H., 2016. Post-fire recovery of soil organic matter in a Cambisol from typical Mediterranean forest in Southwestern Spain. The Science of the Total Environment 572, 1414–1421.

Johnson, D.W., Curtis, P.S., 2001. Effects of forest management on soil C and N storage: meta analysis. Forest Ecology and Management 140, 227–238.

Johnson, D.W., Knoepp, J.D., Swank, W.T., Shan, J., Morris, L.A., Van Lear, D.H., Kapeluck, P.R., 2002. Effects of forest management on soil carbon: results of some long-term resampling studies. Environmental Pollution 116, S201–S208.

Johnson, K.D., Harden, J., McGuire, A.D., Bliss, N.B., Bockheim, J.G., Clark, M., Nettleton-Hollingsworth, T., Jorgenson, M.T., Kane, E.S., Mack, M., O'Donnell, J., Ping, C.L., Schuur, E.A.G., Turetsky, M.R., Valentine, D.W., 2011a. Soil carbon distribution in Alaska in relation to soil-forming factors. Geoderma 167–68, 71–84.

Johnson, K.D., Scatena, F.N., Silver, W.L., 2011b. Atypical soil carbon distribution across a tropical steepland forest catena. Catena 87, 391–397.

Johnson, A.H., Xing, H.X., Scatena, F.N., 2015. Controls on soil carbon stocks in el Yunque National Forest, Puerto Rico. Soil Science Society of America Journal 79, 294–304.

Jones, H.S., Beets, P.N., Kimberley, M.O., Garrett, L.G., 2011. Harvest residue management and fertilisation effects on soil carbon and nitrogen in a 15-year-old Pinus radiata plantation forest. Forest Ecology and Management 262, 339–347.

Jurevics, A., Peichl, M., Olsson, B.A., Stromgren, M., Egnell, G., 2016. Slash and stump harvest have no general impact on soil and tree biomass C pools after 32-39 years. Forest Ecology and Management 371, 33–41.

Kallenbach, C.M., Frey, S.D., Grandy, A.S., 2016. Direct evidence for microbial-derived soil organic matter formation and its ecophysiological controls. Nature Communications 7.

Kammer, A., Hagedorn, F., Shevchenko, I., Leifeld, J., Guggenberger, G., Goryacheva, T., Rigling, A., Moiseev, P., 2009. Treeline shifts in the Ural mountains affect soil organic matter dynamics. Global Change Biology 15, 1570–1583.

Kashian, D.M., Romme, W.H., Tinker, D.B., Turner, M.G., Ryan, M.G., 2006. Carbon storage on landscapes with stand-replacing fires. BioScience 56, 598–606.

Kashian, D.M., Romme, W.H., Tinker, D.B., Turner, M.G., Ryan, M.G., 2013. Postfire changes in forest carbon storage over a 300-year chronosequence of *Pinus contorta*-dominated forests. Ecological Monographs 83, 49–66.

Kirdyanov, A.V., Hagedorn, F., Knorre, A.A., Fedotova, E.V., Vaganov, E.A., Naurzbaev, M.M., Moiseev, P.A., Rigling, A., 2012. 20th century tree-line advance and vegetation changes along an altitudinal transect in the Putorana Mountains, northern Siberia. Boreas 41, 56–67.

Kishchuk, B.E., Morris, D.M., Lorente, M., Keddy, T., Sidders, D., Quideau, S., Thiffault, E., Kwiaton, M., Maynard, D., 2016. Disturbance intensity and dominant cover type influence rate of boreal soil carbon change: a Canadian multi-regional analysis. Forest Ecology and Management 381, 48–62.

Kizilkaya, R., Dengiz, O., 2010. Variation of land use and land cover effects on some soil physico-chemical characteristics and soil enzyme activity. Zemdirbyste-Agriculture 97, 15–24.
Klaes, B., Struck, J., Schneider, R., Schuler, G., 2016. Middle-term effects after timber harvesting with heavy machinery on a fine-textured forest soil. European Journal of Forest Research 135, 1083–1095.
Kolb, T.E., Fettig, C.J., Ayres, M.P., Bentz, B.J., Hicke, J.A., Mathiasen, R., Stewart, J.E., Weed, A.S., 2016. Observed and anticipated impacts of drought on forest insects and diseases in the United States. Forest Ecology and Management 380, 321–334.
Kranabetter, J.M., 2009. Site carbon storage along productivity gradients of a late-seral southern boreal forest. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 39, 1053–1060.
Kranabetter, J.M., Macadam, A.M., 2007. Changes in carbon storage of broadcast burn plantations over 20 years. Canadian Journal of Soil Science 87, 93–102.
Kreutzweiser, D.P., Hazlett, P.W., Gunn, J.M., 2008. Logging impacts on the biogeochemistry of boreal forest soils and nutrient export to aquatic systems: a review. Environmental Reviews 16, 157–179.
Krull, E.S., Bestland, E.A., Gates, W.P., 2002. Soil organic matter decomposition and turnover in a tropical Ultisol: evidence from delta C-13, delta N-15 and geochemistry. Radiocarbon 44, 93–112.
Krupenikov, I.A., Boincean, B.P., Dent, D., Krupenikov, I.A., Boincean, B.P., Dent, D., 2011. The Chernozem Family.
Kurz, W.A., Shaw, C.H., Boisvenue, C., Stinson, G., Metsaranta, J., Leckie, D., Dyk, A., Smyth, C., Neilson, E.T., 2013. Carbon in Canada's boreal forest—a synthesis. Environmental Reviews 21, 260–292.
Lado, M., Inbar, A., Sternberg, M., Ben-Hur, M., 2016. Effectiveness of granular polyacrylamide to reduce soil erosion during consecutive rainstorms in a Calcic Regosol exposed to different fire conditions. Land Degradation and Development 27, 1453–1462.
Laganiere, J., Angers, D.A., Pare, D., 2010. Carbon accumulation in agricultural soils after afforestation: a meta-analysis. Global Change Biology 16, 439–453.
Lal, R., Kimble, J.M., 2000. Pedogenic carbonates and the global carbon cycle. In: Global Climate Change and Pedogenic Carbonates. Lewis Publishers Inc, Boca Raton, pp. 1–14.
Lavkulich, L.M., Arocena, J.M., 2011. Luvisolic soils of Canada: genesis, distribution, and classification. Canadian Journal of Soil Science 91, 781–806.
Le Bissonnais, Y., Prieto, I., Roumet, C., Nespoulous, J., Metayer, J., Huon, S., Villatoro, M., Stokes, A., 2018. Soil aggregate stability in Mediterranean and tropical agro-ecosystems: effect of plant roots and soil characteristics. Plant and Soil 424, 303–317.
Lehmann, J., Kleber, M., 2015. The contentious nature of soil organic matter. Nature 528, 60–68.
Leinweber, P., Schulten, H.R., Jancke, H., 1999. New evidence for the molecular composition of soil organic matter in vertisols. Soil Science 164, 857–870.
Lemenih, M., Karltun, E., Olsson, M., 2005. Soil organic matter dynamics after deforestation along a farm field chronosequence in southern highlands of Ethiopia. Agriculture, Ecosystems and Environment 109, 9–19.
Li, L.G., Vogel, J., He, Z.L., Zou, X.M., Ruan, H.H., Huang, W., Wang, J.S., Bianchi, T.S., 2016. Association of soil aggregation with the distribution and quality of organic carbon in soil along an elevation gradient on Wuyi mountain in China. PLoS One 11.
Littell, J.S., McKenzie, D., Peterson, D.L., Westerling, A.L., 2009. Climate and wildfire area burned in western U. S. ecoprovinces, 1916-2003. Ecological Applications 19, 1003–1021.
Lu, M.K., Xie, J.S., Wang, C., Guo, J.F., Wang, M.H., Liu, X.F., Chen, Y.M., Chen, G.S., Yang, Y.S., 2015. Forest conversion stimulated deep soil C losses and decreased C recalcitrance through priming effect in subtropical China. Biology and Fertility of Soils 51, 857–867.
Luckai, N., Larocque, G.R., Archambault, L., Pare, D., Boutin, R., Groot, A., 2012. Using the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) to examine the impact of harvest and fire on

carbon dynamics in selected forest types of the Canadian Boreal shield. The Forestry Chronicle 88, 426–438.
Lugo, A.E., Brown, S., 1993. Management of tropical soils as sinks or sources of atmospheric carbon. Plant and Soil 149, 27–41.
Lupi, A.M., Boca, T., Garcia, M.D., Diaz, D., Ingaramo, L., 2017. Management of residues from *Eucalyptus grandis* Hill ex Maiden harvesting in contrasting sites. Ciência Florestal 27, 767–782.
Lutz, H.J., Chandler, R.F., 1946. Forest Soils. J. Wiley and Sons, Inc, New York.
Lyu, M.K., Xie, J.S., Ukonmaanaho, L., Jiang, M.H., Li, Y.Q., Chen, Y.M., Yang, Z.J., Zhou, Y.X., Lin, W.S., Yang, Y.S., 2017. Land use change exerts a strong impact on deep soil C stabilization in subtropical forests. Journal of Soils and Sediments 17, 2305–2317.
Ma, R.M., Cai, C.F., Li, Z.X., Wang, J.G., Xiao, T.Q., Peng, G.Y., Yang, W., 2015. Evaluation of soil aggregate microstructure and stability under wetting and drying cycles in two Ultisols using synchrotron-based X-ray micro-computed tomography. Soil & Tillage Research 149, 1–11.
Macedo, M.O., Resende, A.S., Garcia, P.C., Boddey, R.M., Jantalia, C.P., Urquiaga, S., Campello, E.F.C., Franco, A.A., 2008. Changes in soil C and N stocks and nutrient dynamics 13 years after recovery of degraded land using leguminous nitrogen-fixing trees. Forest Ecology and Management 255, 1516–1524.
Maier, C.A., Johnsen, K.H., Dougherty, P., McInnis, D., Anderson, P., Patterson, S., 2012. Effect of harvest residue management on tree productivity and carbon pools during early stand development in a loblolly pine plantation. Forest Science 58, 430–445.
Makipaa, R., 1995. Effect of nitrogen input on carbon accumulation of boreal forest soils and ground vegetation. Forest Ecology and Management 79, 217–226.
Manrique, S., Franco, J., Nunez, V., Seghezzo, L., 2011. Potential of native forests for the mitigation of greenhouse gases in Salta, Argentina. Biomass and Bioenergy 35, 2184–2193.
Maraseni, T.N., Mathers, N.J., Harms, B., Cockfield, G., Apan, A., Maroulis, J., 2008. Comparing and predicting soil carbon quantities under different land-use systems on the Red Ferrosol soils of southeast Queensland. Journal of Soil and Water Conservation 63, 250–256.
Marbut, C.F., 1927. In: Glinka, K.D. (Ed.), The Great Soil Groups of the World and Their Development. Edwards Bros., Ann Arbor, MI. Transl.
Marin-Spiotta, E., Sharma, S., 2013. Carbon storage in successional and plantation forest soils: a tropical analysis. Global Ecology and Biogeography 22, 105–117.
Marin-Spiotta, E., Silver, W.L., Swanston, C.W., Ostertag, R., 2009. Soil organic matter dynamics during 80 years of reforestation of tropical pastures. Global Change Biology 15, 1584–1597.
Marques, F.A., Calegari, M.R., Vidal-Torrado, P., Buurman, P., 2011. Relationship between soil oxidizable carbon and physical, chemical and mineralogical properties of umbric Ferralsols. Revista Brasileira de Ciência do Solo 35, 25–40.
Martin, P.H., Fahey, T.J., 2014. Mesoclimatic patterns shape the striking vegetation mosaic in the Cordillera Central, Dominican Republic. Arctic Antarctic and Alpine Research 46, 755–765.
Martin, D., Lal, T., Sachdev, C.B., Sharma, J.P., 2010. Soil organic carbon storage changes with climate change, landform and land use conditions in Garhwal hills of the Indian Himalayan mountains. Agriculture, Ecosystems and Environment 138, 64–73.
Martin, A., Diaz-Ravina, M., Carballas, T., 2012. Short- and medium-term evolution of soil properties in Atlantic forest ecosystems affected by wildfires. Land Degradation and Development 23, 427–439.
Masiello, C.A., Chadwick, O.A., Southon, J., Torn, M.S., Harden, J.W., 2004. Weathering controls on mechanisms of carbon storage in grassland soils. Global Biogeochemical Cycles 18.
Matsuzaki, E., Sanborn, P., Fredeen, A.L., Shaw, C.H., Hawkins, C., 2013. Carbon stocks in managed and unmanaged old-growth western redcedar and western hemlock stands of Canada’s inland temperate rainforests. Forest Ecology and Management 297, 108–119.

McFee, W.W., Kelly, J.M. (Eds.), 1995. Carbon Forms and Functions in Forest Soils. SSSA, Madison, WI. https://doi.org/10.2136/1995.

McNabb, D.H., Boersma, L., 1993. Evaluation of the relationship between compressibility and shear-strength of andisols. Soil Science Society of America Journal 57, 923–929.

Morisada, K., Ono, K., Kanomata, H., 2004. Organic carbon stock in forest soils in Japan. Geoderma 119, 21–32.

Mulder, V.L., Lacoste, M., Martin, M.P., Richer-de-Forges, A., Arrouays, D., 2015. Understanding large-extent controls of soil organic carbon storage in relation to soil depth and soil-landscape systems. Global Biogeochemical Cycles 29, 1210–1229.

Nauman, T.W., Thompson, J.A., Teets, S.J., Dilliplane, T.A., Bell, J.W., Connolly, S.J., Liebermann, H.J., Yoast, K.M., 2015. Ghosts of the forest: mapping pedomemory to guide forest restoration. Geoderma 247, 51–64.

Nave, L.E., Vance, E.D., Swanston, C.W., Curtis, P.S., 2009. Impacts of elevated N inputs on north temperate forest soil C storage, C/N, and net N-mineralization. Geoderma 153, 231–240.

Nave, L.E., Vance, E.D., Swanston, C.W., Curtis, P.S., 2010. Harvest impacts on soil carbon storage in temperate forests. Forest Ecology and Management 259, 857–866.

Nave, L.E., Vance, E.D., Swanston, C.W., Curtis, P.S., 2011. Fire effects on temperate forest soil C and N storage. Ecological Applications 21, 1189–1201.

Nave, L.E., Swanston, C.W., Mishra, U., Nadelhoffer, K.J., 2013. Afforestation effects on soil carbon storage in the United States: a synthesis. Soil Science Society of America Journal 77, 1035–1047.

Nave, L.E., Domke, G.M., Hofmeister, K.L., Mishra, U., Perry, C.H., Walters, B.F., Swanston, C.W., 2018. Reforestation can sequester two petagrams of carbon in US topsoils in a century. Proceedings of the National Academy of Sciences of the United States of America 115, 2776–2781.

Nave, L.E., Walters, B.F., Hofmeister, K.L., Perry, C.H., Mishra, U., Domke, G.M., Swanston, C.W., 2019. The role of reforestation in carbon sequestration. New Forests 50, 115–137.

Nichols, J.D., Smith, R.G.B., Grant, J., Glencross, K., 2010. Subtropical eucalypt plantations in eastern Australia. Australian Forestry 73, 53–62.

Noble, A.D., Gillman, G.P., Ruaysoongnern, S., 2000. A cation exchange index for assessing degradation of acid soil by further acidification under permanent agriculture in the tropics. European Journal of Soil Science 51, 233–243.

Norris, C.E., Quideau, S.A., Bhatti, J.S., Wasylishen, R.E., MacKenzie, M.D., 2009. Influence of fire and harvest on soil organic carbon in jack pine sites. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 39, 642–654.

O'Donnell, J.A., Harden, J.W., McGuire, A.D., Kanevskiy, M.Z., Jorgenson, M.T., Xu, X.M., 2011. The effect of fire and permafrost interactions on soil carbon accumulation in an upland black spruce ecosystem of interior Alaska: implications for post-thaw carbon loss. Global Change Biology 17, 1461–1474.

Oostra, S., Majdi, H., Olsson, M., 2006. Impact of tree species on soil carbon stocks and soil acidity in southern Sweden. Scandinavian Journal of Forest Research 21, 364–371.

Ostertag, R., Marin-Spiotta, E., Silver, W.L., Schulten, J., 2008. Litterfall and decomposition in relation to soil carbon pools along a secondary forest chronosequence in Puerto Rico. Ecosystems 11, 701–714.

Oubrahim, H., Boulmane, M., Bakker, M.R., Augusto, L., Halim, M., 2016. Carbon storage in degraded cork oak (*Quercus suber*) forests on flat lowlands in Morocco. iForest: Biogeosciences and Forestry 9, 125–137.

Ozsoy, G., Aksoy, E., 2011. Genesis and classification of entisols in mediterranean climate in northwest of Turkey. Journal of Food Agriculture and Environment 9, 998–1004.

Page-Dumroese, D.S., Jurgensen, M.F., 2006. Soil carbon and nitrogen pools in mid- to late-successional forest stands of the northwestern United States: potential impact of fire. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 36, 2270–2284.

Page-Dumroese, D.S., Jurgensen, M., Terry, T., 2010. Maintaining Soil Productivity during Forest or Biomass-to-Energy Thinning Harvests in the Western United States. Western Journal of Applied Forestry 25, 5–11.

Panichini, M., Neculman, R., Godoy, R., Arancibia-Miranda, N., Matus, F., 2017. Understanding carbon storage in volcanic soils under selectively logged temperate rainforests. Geoderma 302, 76–88.

Pare, D., Rochon, P., Brais, S., 2002. Assessing the geochemical balance of managed boreal forests. Ecological Indicators 1, 293–311.

Parfitt, R.L., 2009. Allophane and imogolite: role in soil biogeochemical processes. Clay Minerals 44, 135–155.

Parker, T.C., Subke, J.A., Wookey, P.A., 2015. Rapid carbon turnover beneath shrub and tree vegetation is associated with low soil carbon stocks at a subarctic treeline. Global Change Biology 21, 2070–2081.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K., 2002. Change in soil carbon following afforestation. Forest Ecology and Management 168, 241–257.

Pegoraro, R.F., da Silva, I.R., de Novais, R.F., de Barros, N.F., Fonseca, S., 2010. Stock of carbon and nitrogen in organic matter fractions of soil cultivated with eucalypts in conventional and fertirrigated system. Ciência Rural 40, 302–309.

Pegoraro, R.F., da Silva, I.R., de Novais, R.F., de Barros, N.F., Cantarutti, R.B., Fonseca, S., 2016. The N-15 Natural Abundance and Nitrogen Forms in Ultisol Cultivated with Eucalyptus and Acacia. Ciencia Florestal 26, 295–U343.

Pengerud, A., Johnsen, L.K., Mulder, J., Strand, L.T., 2014. Potential adsorption of dissolved organic matter in poorly podzolised, high-latitude soils. Geoderma 226, 39–46.

Ponette-Gonzalez, A.G., Marin-Spiotta, E., Brauman, K.A., Farley, K.A., Weathers, K.C., Young, K.R., 2014. Hydrologic connectivity in the high-elevation tropics: heterogeneous responses to land change. BioScience 64, 92–104.

Portes, R.D., Spinola, D.N., Reis, J.S., Ker, J.C., da Costa, L.M., Fernandes, E.I., Kuhn, P., Schaefer, C., 2016. Pedogenesis across a climatic gradient in tropical high mountains, Cordillera Blanca–Peruvian Andes. Catena 147, 441–452.

Post, W.M., Emanuel, W.R., Zinke, P.J., Stangenberger, A.G., 1982. Soil carbon pools and world life zones. Nature 298, 156–159.

Potes, M.D., Dick, D.P., Santana, G.S., Tomazi, M., Bayer, C., 2012. Soil organic matter in fire-affected pastures and in an Araucaria forest in South-Brazilian Leptosols. Pesquisa Agropecuaria Brasileira 47, 707–715.

Powers, J.S., Marin-Spiotta, E., 2017. Ecosystem processes and biogeochemical cycles in secondary tropical forest succession. Annual Review of Ecology, Evolution and Systematics 48, 497–519.

Powers, J.S., Corre, M.D., Twine, T.E., Veldkamp, E., 2011. Geographic bias of field observations of soil carbon stocks with tropical land-use changes precludes spatial extrapolation. Proceedings of the National Academy of Sciences of the United States of America 108, 6318–6322.

Prescott, C.E., 2010. Litter decomposition: what controls it and how can we alter it to sequester more carbon in forest soils? Biogeochemistry 101, 133–149.

Prescott, C.E., Maynard, D.G., Laiho, R., 2000. Humus in northern forests: friend or foe? Forest Ecology and Management 133, 23–36.

Prietzel, J., Bachmann, S., 2012. Changes in soil organic C and N stocks after forest transformation from Norway spruce and Scots pine into Douglas- fir, Douglas- fir/spruce, or European beech stands at different sites in Southern Germany. Forest Ecology and Management 269, 134–148.

Quin, P.R., Cowie, A.L., Flavel, R.J., Keen, B.P., Macdonald, L.M., Morris, S.G., Singh, B.P., Young, I.M., Van Zwieten, L., 2014. Oil mallee biochar improves soil structural properties-A study with x-ray micro-CT. Agriculture, Ecosystems and Environment 191, 142–149.

Rasmussen, C., Heckman, K., Wieder, W.R., Keiluweit, M., Lawrence, C.R., Berhe, A.A., Blankinship, J.C., Crow, S.E., Druhan, J.L., Pries, C.E.H., Marin-Spiotta, E., Plante, A.F., Schadel, C., Schimel, J.P., Sierra, C.A.,

Thompson, A., Wagai, R., 2018. Beyond clay: towards an improved set of variables for predicting soil organic matter content. Biogeochemistry 137, 297–306.
Renison, D., Hensen, I., Suarez, R., Cingolani, A.M., Marcora, P., Giorgis, M.A., 2009. Soil conservation in Polylepis mountain forests of Central Argentina: is livestock reducing our natural capital? Austral Ecology 35, 435–443.
Restaino, J.C., Peterson, D.L., 2013. Wildfire and fuel treatment effects on forest carbon dynamics in the western United States. Forest Ecology and Management 303, 46–60.
Richards, A.E., Dalal, R.C., Schmidt, S., 2007. Soil carbon turnover and sequestration in native subtropical tree plantations. Soil Biology and Biochemistry 39, 2078–2090.
Richter, D.D., 2007. Humanity's transformation of Earth's soil: pedology's new frontier. Soil Science 172, 957–967.
Richter, D.D., Yaalon, D.H., 2012. "The changing model of soil" revisited. Soil Science Society of America Journal 76, 766–778.
Rodionov, A., Amelung, W., Urusevskaja, I., Zech, W., 1998. Relationships between climatic factors and C, N pools in particle-size fractions of steppe soils, Russia. Zeitschrift Fur Pflanzenernahrung Und Bodenkunde 161, 563–569.
Rommel, L.G., Heiberg, S.O., 1931. Types of humus layer in the forests of the North Eastern United States. Ecology 12, 567–579.
Rothstein, D.E., Toosi, E.R., Schaetzl, R.J., Grandy, A.S., 2018. Translocation of carbon from surface organic horizons to the subsoil in coarse-textured spodosols: implications for deep soil C dynamics. Soil Science Society of America Journal 82, 969–982.
Ruiz-Peinado, R., Bravo-Oviedo, A., Lopez-Senespleda, E., Montero, G., Rio, M., 2013. Do thinnings influence biomass and soil carbon stocks in Mediterranean maritime pinewoods? European Journal of Forest Research 132, 253–262.
Ruiz-Peinado, R., Bravo-Oviedo, A., Lopez-Senespleda, E., Bravo, F., del Rio, M., 2017. Forest management and carbon sequestration in the Mediterranean region: a review. Forest Systems 26.
Rumpel, C., Kogel-Knabner, I., Bruhn, F., 2002. Vertical distribution, age, and chemical composition of organic, carbon in two forest soils of different pedogenesis. Organic Geochemistry 33, 1131–1142.
Samuelson, L.J., Stokes, T.A., Butnor, J.R., Johnsen, K.H., Gonzalez-Benecke, C.A., Martin, T.A., Cropper, W.P., Anderson, P.H., Ramirez, M.R., Lewis, J.C., 2017. Ecosystem carbon density and allocation across a chronosequence of longleaf pine forests. Ecological Applications 27, 244–259.
Sanborn, P., Lamontagne, L., Hendershot, W., 2011. Podzolic soils of Canada: genesis, distribution, and classification. Canadian Journal of Soil Science 91, 843–880.
Sanderman, J., 2012. Can management induced changes in the carbonate system drive soil carbon sequestration? A review with particular focus on Australia. Agriculture, Ecosystems and Environment 155, 70–77.
Santana, V.M., Gonzalez-Pelayo, O., Maia, P.A.A., Varela, M.E., Valdecantos, A., Vallejo, V.R., Keizer, J.J., 2016. Effects of fire recurrence and different salvage logging techniques on carbon storage in *Pinus pinaster* forests from northern Portugal. European Journal of Forest Research 135, 1107–1117.
Sarkhot, D.V., Jokela, E.J., Comerford, N.B., 2008. Surface soil carbon size-density fractions altered by loblolly pine families and forest management intensity for a Spodosol in the southeastern US. Plant and Soil 307, 99–111.
Schaetzl, R.J., Luehmann, M.D., Rothstein, D., 2015. Pulses of podzolization: the relative importance of spring snowmelt, summer storms, and fall rains on spodosol development. Soil Science Society of America Journal 79, 117–131.
Schoenholtz, S.H., Van Miegroet, H., Burger, J.A., 2000. A review of chemical and physical properties as indicators of forest soil quality: challenges and opportunities. Forest Ecology and Management 138, 335–356.
Schoening, I., Kogel-Knabner, I., 2006. Chemical composition of young and old carbon pools throughout Cambisol and Luvisol profiles under forests. Soil Biology and Biochemistry 38, 2411–2424.

Schoening, I., Knicker, H., Kogel-Knabner, I., 2005. Intimate association between O/N-alkyl carbon and iron oxides in clay fractions of forest soils. Organic Geochemistry 36, 1378−1390.

Schomakers, J., Jien, S.H., Lee, T.Y., Huang, J.C., Hseu, Z.Y., Lin, Z.L., Lee, L.C., Hein, T., Mentler, A., Zehetner, F., 2017. Soil and biomass carbon re-accumulation after landslide disturbances. Geomorphology 288, 164−174.

Schulp, C.J.E., Nabulars, G.J., Verburg, P.H., de Waal, R.W., 2008. Effect of tree species on carbon stocks in forest floor and mineral soil and implications for soil carbon inventories. Forest Ecology and Management 256, 482−490.

Schuur, E.A.G., McGuire, A.D., Schadel, C., Grosse, G., Harden, J.W., Hayes, D.J., Hugelius, G., Koven, C.D., Kuhry, P., Lawrence, D.M., Natali, S.M., Olefeldt, D., Romanovsky, V.E., Schaefer, K., Turetsky, M.R., Treat, C.C., Vonk, J.E., 2015. Climate change and the permafrost carbon feedback. Nature 520, 171−179.

Scull, P., Franklin, J., Chadwick, O.A., McArthur, D., 2003. Predictive soil mapping: a review. Progress in Physical Geography 27, 171−197.

Sevink, J., Verstraten, J.M., Jongejans, J., 1998. The relevance of humus forms for land degradation in Mediterranean mountainous areas. Geomorphology 23, 285−292.

Shaw, C.H., Banfield, E., Kurz, W.A., 2008. Stratifying soils into pedogenically similar categories for modeling forest soil carbon. Canadian Journal of Soil Science 88, 501−516.

Shetler, G., Turetsky, M.R., Kane, E.S., Kasischke, E., 2008. Sphagnum mosses limit total carbon consumption during fire in Alaskan black spruce forests. Canadian Journal of Forest Research 38, 2328−2336.

Silva, A.C., Torrado, P.V., Perez, M.G., Martin, L., Vasquez, F.M., 2007. Relationship between soil organic matter and slope steepness in a toposequence of Oxisol in southern Minas Gerais, Brazil. Revista Brasileira de Ciência do Solo 31, 1059−1068.

Silver, W.L., Ostertag, R., Lugo, A.E., 2000. The potential for carbon sequestration through reforestation of abandoned tropical agricultural and pasture lands. Restoration Ecology 8, 394−407.

Simonson, R.W., 1959. Outline of a generalized theory of soil genesis. Soil Science Society of America Proceedings 23, 152−156.

Singh, S.K., Pandey, C.B., Sidhu, G.S., Sarkar, D., Sagar, R., 2011. Concentration and stock of carbon in the soils affected by land uses and climates in the western Himalaya, India. Catena 87, 78−89.

Sivakumar, M.V.K., 2007. Interactions between climate and desertification. Agricultural and Forest Meteorology 142, 143−155.

Smith, C.A.S., Webb, K.T., Kenney, E., Anderson, A., Kroetsch, D., 2011. Brunisolic soils of Canada: genesis, distribution, and classification. Canadian Journal of Soil Science 91, 695−717.

Smith, P., House, J.I., Bustamante, M., Sobocka, J., Harper, R., Pan, G.X., West, P.C., Clark, J.M., Adhya, T., Rumpel, C., Paustian, K., Kuikman, P., Cotrufo, M.F., Elliott, J.A., McDowell, R., Griffiths, R.I., Asakawa, S., Bondeau, A., Jain, A.K., Meersmans, J., Pugh, T.A.M., 2016. Global change pressures on soils from land use and management. Global Change Biology 22, 1008−1028.

Soares, M.R., Alleoni, L.R.F., 2008. Contribution of soil organic carbon to the ion exchange capacity of tropical soils. Journal of Sustainable Agriculture 32, 439−462.

Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. Geoderma 74, 65−105.

Spielvogel, S., Prietzel, J., Kogel-Knabner, I., 2008. Soil organic matter stabilization in acidic forest soils is preferential and soil type-specific. European Journal of Soil Science 59, 674−692.

Srinivasarao, C., Lal, R., Kundu, S., Babu, M., Venkateswarlu, B., Singh, A.K., 2014. Soil carbon sequestration in rainfed production systems in the semiarid tropics of India. The Science of the Total Environment 487, 587−603.

Staal, A., Dekker, S.C., Xu, C., van Nes, E.H., 2016. Bistability, spatial interaction, and the distribution of tropical forests and savannas. Ecosystems 19, 1080−1091.

Steltzer, H., 2004. Soil carbon sequestration with forest expansion in an arctic forest-tundra landscape. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 34, 1538−1542.
Stendahl, J., Johansson, M.B., Eriksson, E., Nilsson, A., Langvall, O., 2010. Soil organic carbon in Swedish spruce and pine forests−differences in stock levels and regional patterns. Silva Fennica 44, 5−21.
Sutton, R., Sposito, G., 2005. Molecular structure in soil humic substances: the new view. Environmental Science and Technology 39, 9009−9015.
Switzer, J.M., Hope, G.D., Grayston, S.J., Prescott, C.E., 2012. Changes in soil chemical and biological properties after thinning and prescribed fire for ecosystem restoration in a Rocky Mountain Douglas-fir forest. Forest Ecology and Management 275, 1−13.
Takahashi, T., Dahlgren, R.A., 2016. Nature, properties and function of aluminum-humus complexes in volcanic soils. Geoderma 263, 110−121.
Tan, W.F., Zhang, R., Cao, H., Huang, C.Q., Yang, Q.K., Wang, M.K., Koopal, L.K., 2014. Soil inorganic carbon stock under different soil types and land uses on the Loess Plateau region of China. Catena 121, 22−30.
Tarnocai, C., Bockheim, J.G., 2011. Cryosolic soils of Canada: genesis, distribution, and classification. Canadian Journal of Soil Science 91, 749−762.
Tesfaye, M.A., Bravo, F., Ruiz-Peinado, R., Pando, V., Bravo-Oviedo, A., 2016. Impact of changes in land use, species and elevation on soil organic carbon and total nitrogen in Ethiopian Central Highlands. Geoderma 261, 70−79.
Thiffault, E., Hannam, K.D., Pare, D., Titus, B.D., Hazlett, P.W., Maynard, D.G., Brais, S., 2011. Effects of forest biomass harvesting on soil productivity in boreal and temperate forests−a review. Environmental Reviews 19, 278−309.
Thomas, G.A., Dalal, R.C., Standley, J., 2007. No-till effects on organic matter, pH, cation exchange capacity and nutrient distribution in a Luvisol in the semi-arid subtropics. Soil & Tillage Research 94, 295−304.
Troendle, C.A., King, R.M., 1985. The effect of timber harvest on the fool creek watershed, 30 years later. Water Resources Research 21, 1915−1922.
Trumbore, S., 2009. Radiocarbon and Soil Carbon Dynamics. Annual Review of Earth and Planetary Sciences 37, 47−66.
Turk, J.K., Goforth, B.R., Graham, R.C., Kendrick, K.J., 2008. Soil morphology of a debris flow chronosequence in a coniferous forest, southern California, USA. Geoderma 146, 157−165.
Turner, J., Lambert, M., 2000. Change in organic carbon in forest plantation soils in eastern Australia. Forest Ecology and Management 133, 231−247.
Turner, J., Lambert, M.J., Hopmans, P., McGrath, J., 2001. Site variation in *Pinus radiata* plantations and implications for site specific management. New Forests 21, 249−282.
Ussiri, D.A.N., Johnson, C.E., 2007. Organic matter composition and dynamics in a northern hardwood forest ecosystem 15 years after clear-cutting. Forest Ecology and Management 240, 131−142.
Vance, E.D., 2000. Agricultural site productivity: principles derived from long-term experiments and their implications for intensively managed forests. Forest Ecology and Management 138, 369−396.
Vance, E.D., Maguire, D.A., Zalesny, R.S., 2010. Research strategies for increasing productivity of intensively managed forest plantations. Journal of Forestry 108, 183−192.
Vance, E.D., 2018. Conclusions and caveats from studies of managed forest carbon budgets. Forest Ecology and Management 427, 350−354.
Vargas, R., Allen, M.F., Allen, E.B., 2008. Biomass and carbon accumulation in a fire chronosequence of a seasonally dry tropical forest. Global Change Biology 14, 109−124.
von Luetzow, M., Kogel-Knabner, I., Ludwig, B., Matzner, E., Flessa, H., Ekschmitt, K., Guggenberger, G., Marschner, B., Kalbitz, K., 2008. Stabilization mechanisms of organic matter in four temperate soils: development and application of a conceptual model. Journal of Plant Nutrition and Soil Science-Zeitschrift Fur Pflanzenernahrung Und Bodenkunde 171, 111−124.

Wagai, R., Mayer, L.M., 2007. Sorptive stabilization of organic matter in soils by hydrous iron oxides. Geochimica et Cosmochimica Acta 71, 25–35.

Walker, L.R., Shiels, A.B., 2008. Post-disturbance erosion impacts carbon fluxes and plant succession on recent tropical landslides. Plant and Soil 313, 205–216.

Walmsley, J.D., Godbold, D.L., 2010. Stump harvesting for bioenergy–a review of the environmental impacts. Forestry 83, 17–38.

Wan, X.H., Xiao, L.J., Vadeboncoeur, M.A., Johnson, C.E., Huang, Z.Q., 2018. Response of mineral soil carbon storage to harvest residue retention depends on soil texture: a meta-analysis. Forest Ecology and Management 408, 9–15.

Weaver, P.L., Birdsey, R.A., Lugo, A.E., 1987. Soil organic-matter in secondary forests of puerto-rico. Biotropica 19, 17–23.

Wei, X.H., Blanco, J.A., 2014. Significant increase in ecosystem C can Be achieved with sustainable forest management in subtropical plantation forests. PLoS One 9.

Werden, L.K., Alvarado, P., Zarges, S., Calderon, M.E., Schilling, E.M., Gutierrez, M., Powers, J.S., 2017. Using soil amendments and plant functional traits to select native tropical dry forest species for the restoration of degraded Vertisols. Journal of Applied Ecology 55, 1019–1028.

Whelan, P., Bach, A.J., 2017. Retreating glaciers, incipient soils, emerging forests: 100 Years of landscape change on mount baker, Washington, USA. Annals of the Association of American Geographers 107, 336–349.

Wilmking, M., Harden, J., Tape, K., 2006. Effect of tree line advance on carbon storage in NW Alaska. Journal of Geophysical Research-Biogeosciences 111.

Wolfe, B.T., Van Bloem, S.J., 2011. Subtropical dry forest regeneration in grass-invaded areas of Puerto Rico: understanding why Leucaena leucocephala dominates and native species fail. Forest Ecology and Management 267, 253–261.

Wu, X.L., Wei, Y.J., Wang, J.G., Wang, D., She, L., Wang, J., Cai, C.F., 2017. Effects of soil physicochemical properties on aggregate stability along a weathering gradient. Catena 156, 205–215.

Yaalon, D.H., Yaron, B., 1966. Framework for man-made soil changes–an outline of metapedogenesis. Soil Science 102, 272-&.

Yimer, F., Ledin, S., Abdelkadir, A., 2007. Changes in soil organic carbon and total nitrogen contents in three adjacent land use types in the Bale Mountains, south-eastern highlands of Ethiopia. Forest Ecology and Management 242, 337–342.

Zhang, Q.C., Hou, C.P., Liang, Y.Y., Feng, Y., 2014. Dissolved organic matter release and retention in ultisols in relation to land use patterns. Chemosphere 107, 432–438.

Zhou, J., Bing, H.J., Wu, Y.H., Yang, Z.J., Wang, J.P., Sun, H.Y., Luo, J., Liang, J.H., 2016. Rapid weathering processes of a 120-year-old chronosequence in the Hailuogou Glacier foreland, Mt. Gongga, SW China. Geoderma 267, 78–91.