UNEP

CBD

Convention on
Biological Diversity

Distr.
GENERAL

UNEP/CBD/COP/11/INF/25
13 September 2012

ORIGINAL: ENGLISH

CONFERENCE OF THE PARTIES TO THE
CONVENTION ON BIOLOGICAL DIVERSITY
Eleventh meeting
Hyderabad, India, 8-19 October 2012
Item 11.3 of the provisional agenda*

ORGANIC CARBON STOCKS AND THE CONSERVATION AND SUSTAINABLE USE OF BIODIVERSITY

Note by the Executive Secretary

INTRODUCTION

1. The Conference of the Parties, in paragraph 9 (b) of decision X/33, requested the Executive Secretary to collaborate with relevant international organizations to collect scientific knowledge and case-studies and identify knowledge gaps on the links between biodiversity conservation and sustainable use and organic carbon stock conservation and restoration, and make the results available to Parties through the clearing-house mechanism.

2. In response to this request, the Secretariat compiled and analysed scientific information and case-studies on the links between biological diversity and organic carbon stocks. Information on biodiversity and carbon storage and sequestration in forests can be found in CBD Technical Series No. 59 (SCBD 2011),[1] while information on peatlands can be found in the assessment on peatlands, biodiversity and climate change by Parish et al. (2008).[2] Therefore, the present note focuses on non-forest and non-peatland ecosystems.

3. This note draws upon recent workshops and literature reviews conducted by the German Federal Agency for Nature Conservation (Bundesamt für Naturschutz, BfN) and the United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) on the subject of ecosystem-based mitigation in non-forest, non-peat ecosystems (Epple 2012) and the Cambridge Conservation Initiative/UNEP-WCMC workshop on biodiversity and payments for soil carbon sequestration (Cambridge Conservation Initiative 2011).

4. Drawing on this literature, this note provides a brief review of:

(a) The state of research and knowledge gaps in the understanding of soil carbon dynamics;

---

* UNEP/CBD/COP/11/1.

[1] http://www.cbd.int/doc/publications/cbd-ts-59-en.pdf.

[2] http://www.wetlands.org/WatchRead/Currentpublications/tabid/56/mod/1570/articleType/ArticleView/articleId/2029/Global-Peatland-Assessment.aspx.

(b) The importance of biodiversity and ecosystems in the global carbon cycle and the global-scale ecosystem services of carbon storage and sequestration provided by the biosphere, and some updated global estimates of ecosystem carbon storage;

(c) The implications of a better understanding of the size and distribution of global soil carbon pools, with a focus on several biomes; and

(d) The relevance of soil carbon to the Aichi Biodiversity Targets of the Strategic Plan for Biodiversity 2011-2020.

I. BACKGROUND

5. The Second Ad Hoc Technical Expert Group on Biodiversity and Climate Change (AHTEG) assessed the linkages between biodiversity and climate change (SCBD 2009). The work of the AHTEG was supported by a comprehensive literature review undertaken by UNEP-WCMC (Campbell et al. 2009). In this review and report, the analysis of the relationship between climate change mitigation and biodiversity also considered the links between organic carbon stock conservation and restoration and the conservation and sustainable use of biological diversity.

6. Some of the AHTEG report’s key messages regarding the links between organic carbon stocks and biological diversity (SCBD 2009) are:

(a) Conserving natural terrestrial, freshwater and marine ecosystems and restoring degraded ecosystems (including their genetic and species diversity) is essential for the overall goals of the United Nations Framework Convention on Climate Change (UNFCCC) because ecosystems play a key role in the global carbon cycle and in adaptation to climate change, while also providing a wide range of ecosystem services that are essential for human well-being and for the achievement of the Millennium Development Goals;

(b) About 2,500 Gt C is stored in terrestrial ecosystems, ~ 38,000 Gt C is stored in the oceans (37,000 Gt in deep oceans, i.e., layers that will only feed back to atmospheric processes over very long time scales, and ~ 1,000 Gt in the upper layer), compared to approximately 750 Gt C in the atmosphere. An average of ~160 Gt C cycles naturally between the biosphere (in both ocean and terrestrial ecosystems) and the atmosphere. Thus, small changes in ocean and terrestrial sources and sinks can have large implications for atmospheric CO_2 levels;

(c) Ecosystems are generally more carbon dense and biologically diverse in their natural state, and the degradation of many ecosystems is significantly reducing their carbon storage and sequestration capacity, leading to increases in emissions of greenhouse gases and loss of biodiversity at the genetic, species and ecosystem level;

(d) There is a range of activities in the agricultural sector, including conservation tillage and other means of sustainable cropland management, sustainable livestock management, and agroforestry systems, that can result in the maintenance and potential increase of current carbon stocks and the conservation and sustainable use of biodiversity;

(e) Policies that integrate and promote the conservation and enhanced sequestration of soil carbon, including in peatlands and other wetlands as well as in grasslands and savannahs, can contribute to climate change mitigation and be beneficial for biodiversity and ecosystem services.

7. Since the publication of the AHTEG report, there has been increased consideration of the importance of carbon stored in soils. Although the majority of carbon finance for organic carbon stock conservation and restoration is focused on above-ground biomass, and in large majority on tropical forests

through REDD+, the increasing, although still uncertain, consideration of soil carbon stocks and sequestration potential has important implications for ecosystem-based approaches to mitigation and the linkages between biodiversity and climate change mitigation.

8. The consideration of soil carbon as well as biomass, and recent scientific research on soil carbon stocks and biosequestration rates, have identified ecosystem types with high carbon density that were not previously recognized, and have generated a greater understanding of the mitigation potential of non-forest ecosystems, which store a large proportion of carbon in soils.

II. SOIL CARBON DYNAMICS

9. In absolute terms, soil carbon stocks are much larger than carbon sequestered in biomass (Lal 2004). Overall fluxes between the atmosphere and soils are an order of magnitude larger than anthropogenic emissions (IPCC 2000). However, nearly all climate change mitigation projects and policies focus on above-ground carbon and forest biomass carbon, in particular. While there is historical and scientific rationale for this focus, the generally weaker understanding of soil carbon dynamics, and the difficulty in soil carbon measurement, also contribute to the above-ground biomass focus (Trumper et al. 2009; Epple 2012). Nevertheless, recent research has informed the understanding of soil carbon dynamics and, combined with improved modelling approaches, allows for a greater consideration of soil carbon in climate change mitigation. Soil carbon can be examined from two angles: (i) stocks of carbon in soils and (ii) active sequestration of additional carbon into soils.

10. Carbon in soils can be divided into two major pools: organic carbon (SOC) and inorganic carbon (SIC) (Lal 2009):

(a) Organic carbon is derived from organic matter and is also more important in soil fertility and will be the focus of the balance of this note;

(b) Inorganic carbon can be classified into two types: (i) carbonates derived from weathering of rocks (lithogenic) and (ii) carbonates derived from the direct absorption of carbon dioxide into the soils (pedogenic). Soil inorganic carbon (SIC) sequestration rates are generally an order of magnitude lower than those of soil organic carbon (SOC), but soil inorganic carbon can be a significant carbon pool and has been estimated as high as 930-1738 Gt C globally, with significant concentrations in arid regions and in degraded ecosystems (Lal 2009). However, the soil inorganic pool is relatively stable, and is thought not to be a net sink nor to be strongly affected by land management and therefore not as relevant to climate change mitigation (Walcott et al. 2009). Recent research however points to SIC sequestration in certain ecosystems, for example, limestone karsts, as potentially relevant (Yan et al. 2011).

11. The accumulation of soil organic carbon is the result of the balance between inputs of carbon to the soil in organic matter from primary productivity and outputs from soil respiration (De Deyn et al. 2008). Abiotic factors, temperature and soil moisture are important in determining this balance, but many other factors also influence it, including soil biota diversity and composition (Nielsen et al. 2011).

12. The soil system is highly complex, and the dynamics of carbon storage and the permanence of carbon and soil organic matter (SOM) are best thought of as an ecosystem property (Schmidt et al. 2011). Therefore, different factors (biotic or abiotic) will control carbon stocks and the balance between inputs and outputs of carbon to and from the soils in different ecosystems. In this emerging understanding (Schmidt et al. 2011), organic matter inputs to soils consist of fresh plant litter (leaves, stems, roots and rhizosphere) and fire residues; inputs from roots and the rhizosphere are significant.

13. The permanence of organic matter and carbon is determined by multiple processes, including soil heterogeneity leading to physical isolation of organic matter, freezing and thawing, and microbial

processes. Furthermore, deep soil carbon is now thought to be more mobile and potentially just as active as carbon in upper layers of soil, and microbial products also make a larger contribution to deep soil carbon pools.

14. However, there are still significant gaps in knowledge. Schmidt et al. (2011) argue for three avenues of increased research:

(a) Applying a new generation of field experiments and analytical tools to study the processes driving SOM stabilization and destabilization;

(b) Developing a new generation of soil biogeochemistry models that represent the mechanisms driving soil response to global change; and

(c) Joining forces and connecting the disparate research communities that are studying, managing and predicting SOM cycling and terrestrial ecology.

15. Soil carbon cycling and soil fertility depend on both soil and above-ground biological diversity. Syntheses of recent research have reported a positive relationship between soil biodiversity and the ecosystem function of carbon cycling at low levels of soil biodiversity (Nielsen et al. 2011) and the importance of soil biodiversity in making the soil system resistant to perturbation (Fitter et al. 2005). Individual case-studies support this relationship, for example between carbon storage and fungal diversity in Mediterranean grassland ecosystems especially on slopes and at higher altitude (Persiani et al. 2008).

16. However, there is still significant uncertainty about the nature of the links between soil biodiversity and ecosystem carbon storage. This may have important implications, because soil diversity has been observed to decline sharply in polluted soils (Gans et al. 2005). A recent synthesis identified other threats to soil biodiversity (Turbe et al. 2010): (i) soil degradation, (ii) land-use management, (iii) climate change, (iv) chemical pollution, (v) genetically modified organisms (GMOs), and (vi) invasive species. Significant knowledge gaps remain with respect to the characterization, monitoring and understanding of the nature and mechanisms of diversity–function relationships in soils (Turbe et al. 2010).

17. Above-ground biodiversity can also affect soil carbon stocks. Studies, primarily in experimental grasslands, have found that increased biodiversity can lead to increased soil carbon storage and carbon stocks (Steinbeiss et al. 2008). Another study found that greater plant diversity increased the reliability of inputs to the soil, increasing the reliability of below-ground processes (Milcu et al. 2010). Agricultural biodiversity in terms of crop diversity has also been reported to enhance soil sequestration processes by maintaining continuous crop cover and reducing soil erosion (Hajjar et al. 2008).

Response of soil carbon pools to climate change

18. Significant research has focused on the response of soil carbon pools to climate change, although significant uncertainty remains (Eglin et al. 2011; Davidson & Janssens 2006; Schmidt et al. 2011; Trumbore & Czimczik 2008; Lützow & Kögel-Knabner 2009). On the one hand, increasing carbon dioxide and temperature are thought to increase primary productivity and therefore plant organic carbon inputs to soils (Eglin et al. 2011). However, this assertion has been challenged and qualified by significant uncertainty (Canadell et al. 2007) and the relationship between productivity and climate change is also offset by, for example, increased drought and changes in the water cycle (Granier et al. 2007). In addition, the relationship between increased production and increased soil carbon sequestration is also uncertain; a recent study in tropical forests estimated that increased litterfall would actually increase soil carbon release, a so-called priming effect (Sayer et al. 2011). On the other hand, increasing temperatures are thought to increase the rate of microbial decomposition and respiration by increasing the rate of

enzymatic reactions in the soil and these processes are thought to be more sensitive to temperature than increased productivity, particularly at lower temperatures. This would therefore increase release of carbon dioxide from soils under climate change (Kirschbaum 1995). These observations are supported by the higher proportion of soil carbon stocks in temperate, cooler climates compared to warmer tropical climates (Lal 2006). Recent meta-analysis supports the theory that temperature increases due to climate change will result in increased soil respiration and potentially, although not necessarily, to increased fluxes to the atmosphere (Bond-Lamberty & Thomson 2010).

19. Another recent synthesis came to the conclusion that it is not currently possible to predict with certainty the response of soil organic matter decomposition to temperature changes (Lützow & Kögel-Knabner 2009) because decomposition is limited by biological and biophysical restrictions, such as pH, water limitations, oxygen supply, accessibility of substrates, etc. The synthesis also identified significant knowledge gaps and suggested the following avenues for future study: (i) the temperature dependence of the mineralization (a process that makes substrates available) of more stable SOM pools; (ii) the role of soil biodiversity in carbon cycling; (iii) the importance of soil microbial biomass stocks, faunal composition, and priming effects; and (iv) the ability of microflora to adapt to decomposition rates. The prediction of net long-term carbon storage requires the consideration of feedback mechanisms and that external factors make this highly region- and ecosystem-dependent.

20. A recent meta-analysis determined that rising CO_2 concentrations will affect soil biodiversity (Blankinship et al. 2011). While initially, increased carbon dioxide increased soil biota abundance, this effect diminished over time while the negative effects of soil warming increased over time. The authors also determined that the positive effects of precipitation increased over time.

21. A final knowledge gap with respect to the response of soil organic carbon stocks to climate change is with respect to other interactions between climate change and biodiversity: for example, related to changes in precipitation regimes (Smith, Fang et al. 2008).

22. Although it is not possible to determine with certainty the relationship between soil carbon and climate change, at the global level, the balance of evidence would suggest that climate change will certainly affect soil carbon fluxes to the atmosphere and will likely increase them. At the ecosystem level, predictions can be made with more confidence. For example, there is now growing evidence for feedback between carbon stored in permafrost soils and climate change. Increasing Arctic temperatures leading to permafrost thawing and peat fires may release vast quantities of soil carbon (see below and Mack et al. 2011; Schuur et al. 2009) that will not be offset by a longer growing season (Schuur et al. 2009).

Soil carbon measurement

23. While above-ground biomass can be estimated using remote sensing (Goetz et al. 2009), the measurement of soil organic carbon stocks over large areas is much more onerous. Verifying changes in soil organic matter due to management is even more problematic. Measurement techniques for assessing soil organic matter (SOM), and by extension soil carbon, are relatively straightforward: established methods are available and individual samples are on the order of USD 20 (Cambridge Conservation Initiative 2011). The measurement of soil carbon requires the assessment of three variables: (i) soil carbon content; (ii) soil depth; and (iii) soil bulk density. Depth and bulk density together estimate soil mass per unit area, and soil carbon content determines what proportion of the mass is carbon.

24. Scaling from individual measurements to the landholder, project or landscape level is much more difficult. This is because SOM density can be highly heterogeneously distributed, as can soil horizons and bulk densities (soil density) (Cambridge Conservation Initiative 2011). To measure carbon storage over large areas requires high-resolution maps of soil type, depth and bulk density. While global soil databases exist and have been used to estimate soil carbon densities globally (e.g. IGBP-DIS 2000), it is recognized

that significant knowledge gaps remain in these databases: they are low-resolution and contain significant inaccuracies (Sanchez et al. 2009). A global soil mapping project is currently underway to provide a three-dimensional higher-resolution digital soil map of the globe (http://www.globalsoilmap.net/).

25. As a case-study of the challenges of measuring soil carbon at a project level, the Tropical Forest Group (TFG) / Save the Children project "Measuring Carbon in Ethiopian Rangelands" (Niles et al. 2010), funded by USAID, explored the possible opportunities for carbon finance for carbon sequestration in Ethiopian rangelands. At three sites – a pasture, woodland and forest – the project attempted to measure biomass and soil carbon in grazed and ungrazed areas. This allowed the team to compare the reported biomass carbon measurements with those estimated using global default values (IPCC Tier 1; IPCC, 2006). While they found agreement for the woodland site, default values derived from IPCC methodology (the *top-down* approach to carbon estimation) were substantially different from those the team measured (*the bottom-up* approach) in the pasture and forest sites. The team also attempted to measure soil carbon at each site. However, soils at the pasture site were rocky and varied in thickness from 0 to 15 cm, and the researchers were not able to estimate soil carbon using standard techniques as they were not able to derive estimates of bulk density and soil thickness. Lessons learned from this project include:

(a) Global IPCC (2006) default values are poorly predictive of Ethiopian carbon stocks in broad land-use classes; and

(b) Soil carbon measurement in these Ethiopian pasture lands is difficult due to rocky and variably thin soils and new techniques may need to be developed to obtain accurate estimates.

26. The measurement of SOC changes over large areas is even more difficult. At the field level and over a time series, this requires extensive soil sampling (Conant 2010). This is of particular concern for payment for ecological services (PES) or carbon finance schemes that aim to compensate landowners for management practices that increase soil organic matter and soil carbon. The cost associated with monitoring SOM changes has led to the development of practice-based assessment that couples limited sampling with SOM modelling with empirical scientific information about land-use changes (Smith, Martino et al. 2008; Conant 2010). However, a recent meta-analysis of the effect of land-use change on soil carbon stocks in tropical ecosystems urges caution in the extrapolation of empirical studies of SOC change to areas with different biophysical parameters (Powers et al. 2011).

27. At the regional level, the Global Environment Facility Soil Organic Carbon (GEFSOC) Project developed soil carbon inventories (in Brazil, India, Jordan, and Kenya) and further developed current soil carbon models (GEFSOC Project Team 2005). However, the results from the GEFSOC project are not immediately expandable into a global soil carbon database.

28. The Carbon Benefits Project (CBP),[3] funded by the Global Environment Facility (GEF), is a modelling system designed to be used with land management projects. The CBP system provides an online modular interface that allows project developers to employ modelling techniques to calculate the net carbon benefits of their projects. While the project is currently under development, it has the potential to increase the ease of soil carbon measurement at the project level.

29. A roadmap for the assessment of soil organic carbon change over large areas (Brown et al. 2010) describes some of the knowledge gaps and the prerequisites for such monitoring, in particular, the need to:

(a) Collect statistically rigorous ground-based measurements of SOC change at strategically selected sites;

---

[3] http://cbp-web1.nrel.colostate.edu/development/cbppim/.

(b) Analyse temporally targeted suborbital imaging spectroscopy to estimate SOC-controlling parameters; and

(c) Conduct spatially explicit biogeochemical modelling.

Deep soils – a key knowledge gap

30. The depth of soil measurement is also a key area of uncertainty. Studies of global soil carbon stocks range from consideration of 1 m of the soil profile (e.g. IPCC 2001, Powers et al. 2011) to 3 m (e.g. Jobbágy & Jackson 2000, Tarnocai et al. 2009), which reduces comparability and can drastically change estimates of total ecosystem carbon storage (see section III below). Many local and regional studies sample to less than 20 cm, and this has very important implications for soil C assessment and the response of soil C stocks to management and land-use changes (Harrison et al. 2011). The same study conducted a meta-analysis of land-use changes, for example, agricultural conversion to switchgrass and nitrogen fertilization of forests, and determined that the results and conclusions can change if soils are only sampled to a shallow depth. In light of recent advancement in the understanding of the active role deep soil organic matter can play (Schmidt et al. 2011), deeper soil layers should not be ignored when accounting for changes in carbon stocks through land management practices (Harrison et al. 2011).

Soil organic carbon and land-use change

31. Empirical studies of the effect of land-use change on soil carbon stocks are common but exhibit high variability and inconsistent methodology. However, using meta-analysis, several authors have drawn broad, general conclusions about which land-use changes affect soil carbon stocks. In the tropics, Powers et al. (2011) conducted a meta-analysis of 80 published studies and concluded that sampling was biased across precipitation regimes and soil types, and that these biases affected conclusions. However, in general they were able to conclude that, with significant qualifications:

(a) Conversion of forest to pasture increased soil carbon stocks in low-activity soils, but decreased soil carbon in high-activity soils, with a confounding effect of mean annual precipitation;

(b) Conversion of pasture to secondary forest increased carbon stocks; and

(c) Conversion of forest to cropland decreased carbon stocks, except in high-activity soils.

32. Another recent meta-analysis of soil organic carbon change in response to tropical land-use change reported similar results, with greater consideration of depth of soil carbon measurement (Don et al. 2011). The study reported SOC decreases with the following land-use transitions: primary forest to grassland, primary forest to cropland, primary forest to perennial cropland, primary to secondary forest, secondary forest to grassland, and grassland to cropland. The following land-use changes were reported as increasing carbon stocks: grassland to secondary forest, cropland to secondary forest, cropland to grassland, and cropland to fallow. Only a single transition, primary forest to secondary forest, had contradictory SOC changes depending on soil depth: in this transition, while upper layers of soil lost carbon, deeper layers were reported to have gained carbon. The greatest magnitude of SOC change involved transitions to and from cropland.

33. A further meta-analysis of land-use change on SOC stocks report similar, although less nuanced, results (Guo & Gifford 2002). The study concluded that SOC stocks decreased in the following conversions: pasture to plantation, native forest to plantation, native forest to crop, and pasture to crop. Land-use conversions that increased soil carbon stocks were: forest to pasture, crop to pasture, crop to plantation, and crop to secondary forest.

34. Other studies support the general conclusion that the clearing of forest land for cropland decreases soil carbon stocks, but that conversion to pasture does not (Murty et al. 2002). Lal (2008) asserts that conversion of natural ecosystems to agricultural ecosystems depletes soil carbon over a period of 20 to 50 years in temperate climes and 5 to 10 years in the tropics; he also reports that cultivated soils contain on average 50 to 70 per cent of the carbon content of undisturbed soils. Degraded ecosystems and those affected by desertification are widely reported to contain less soil carbon (Lal 2004; Olsson & Ard 2002; Lal 2009).

Management to increase soil carbon

35. Beyond practices to ensure that carbon stocks already stored in soils remain there (e.g. preventing conversion of ecosystems to cropland, combating land degradation and desertification, etc.), a range of management options also exists, primarily for croplands, degraded soils and rangelands. The feasibility of increasing the concentration of carbon in soils depends on the ecosystem, type of soil and condition (see also focal ecosystems, below). Generally speaking, this involves management practices that tip the balance of production and respiration: increasing net primary production (NPP) (for instance through irrigation, fertilizers, revegetation, etc.) or modifications that reduce carbon loss from soils (for instance re-wetting wetlands, etc.). Because degraded soils have depleted soil carbon stocks, they have some of the largest potential for enhancing carbon sequestration, which has been estimated at approximately 1 Gt C / yr in the global drylands (Lal 2009). The rate of carbon sequestration in soils depends on many factors but is generally faster in cooler soils and slower in warmer soils. Wetter soils also sequester more carbon as do clayey soils when compared to drier, sandier soils (Lal 2009).

36. While there is significant global potential for soil carbon sequestration, a recent critical review outlined three important caveats to the importance of soil carbon to mitigating climate change (Powlson et al. 2011):

(a) The quantity of carbon stored in soil is finite;

(b) The process is reversible; and

(c) Even if SOC is increased, there may be changes in the fluxes of other greenhouse gases, especially nitrous oxide (N_2O) and methane.

III. BIODIVERSITY AND ECOSYSTEMS IN THE GLOBAL CARBON CYCLE

Global carbon storage

37. The AHTEG report presents ecosystem carbon storage based on an IPCC (2001) assessment (Table 1). These estimates take into account only the first 1 m of soils and are thought to underestimate soil carbon content in some biomes. A recent recalculation (Eglin et al. 2011) of carbon storage values, including soil carbon stock estimates down to a depth of 3 m (largely from Jobbágy & Jackson 2000), revealed significantly higher estimates in nearly all biomes, including an approximately threefold increase in soil organic carbon stocks estimates for tropical forests (Table 2). UNEP-WCMC has produced a spatially explicit, top-down assessment of global carbon stocks (Kapos et al. 2008; Trumper et al. 2009) that integrates remotely-sensed land cover classifications (Global Land Cover 2000 or GLC2000) with IPCC Tier I default values for ecosystem carbon stocks (Ruesch & Gibbs 2008), combined with a spatially-explicit soil database (IGBP-DIS 2000) to better account for soil carbon stocks (Table 3). The analysis has recently been updated with improved spatial resolution of soil carbon stocks (Scharlemann et al. 2009), although carbon density estimates based on this product are not yet available.[4] Direct

---

[4] Please refer to http://www.carbon-biodiversity.net/GlobalScale/Map for updates.

comparisons between the IPCC (2001) and UNEP-WCMC assessments (Kapos et al. 2008; Trumper et al. 2009) are difficult because of the use of disparate biome classifications. Significant uncertainty arising from, for example, the use of default ecosystem carbon values, underlies all of these measurements and there is significant spatial heterogeneity in ecosystem carbon stocks within biomes.

Table 1: IPCC (2001) summary of global carbon stocks

	Vegetation (Gt C)	Soils (Gt C)	Total (Gt C)
Tropical forests	212	216	428
Temperate forests	59	100	159
Boreal forests	88	471	559
Tropical savannas	66	264	330
Temperate grasslands	9	295	304
Deserts and semi-desert	8	191	199
Tundra	6	121	127
Wetlands	15	225	240
Croplands	3	128	131

Table 2: Summary of global carbon stocks based on data from Eglin et al. (2011)[a]

	Vegetation (Gt C)	Soils (Gt C)	Total (Gt C)
Deserts and sclerophyllous shrubs	9	332	341
Crops	3.5	248	251.5
Tropical savannas	72.5	345	417.5
Temperate grasslands	16	172	188
Tundra	4	144	148
Tropical forests	276	692	968
Temperate forests	99	262	361
Boreal forests	72.5	150	222.5
Peatlands	15	400-500	415-515
Permafrost	-	1024	1024

[a] Eglin et al. (2011) provided mean values for soil C and ranges for vegetation C; the latter were then averaged to generate the estimates shown here for vegetation and total C respectively.

Table 3: Trumper et al. (2009) summary of global carbon stocks

	Total (Gt C)
Tropical, subtropical forests	547.8
Tropical and subtropical grasslands, savannahs and shrublands	285.3
Deserts and dry shrublands	178
Temperate grasslands, savannas and shrublands	183.7
Temperate forest	314.9
Boreal forest	384.2
Tundra	155.4
Rocks and ice	1.47
Lakes	0.98

38. Absent or poorly captured by the global scale analyses above is the globally significant carbon stock present in the northern permafrost region, which have been recently estimated at 1672 Gt C,[5] nearly all in soils (Tarnocai et al. 2009). The same study estimated soil organic carbon stocks of 495.8 Gt C in the first meter, which is approximately three times the estimate of 121 Gt C for the same depth included in IPCC (2001). Peatlands also contain very high stocks of carbon in soils, with a global estimate of 550 Gt C (Parish et al. 2008). This carbon is not well reflected in the global analyses above.

Global carbon sequestration and mitigation potential

39. Globally, the earth's terrestrial ecosystems are estimated to have sequestered 2.6±0.7 Gt C / yr over the period from 1990 to 2000. There is significant year-to-year variability in this estimate. Due to wetter conditions in the tropics in 2008, the terrestrial carbon sink for that year was estimated at 4.7±1.2 Gt C / yr (Le Quéré et al. 2009). Estimates of terrestrial carbon sequestration include 2.1–3 Gt C / yr reported in Campbell et al. (2009).

40. Globally, degraded soils have been identified as having significant potential for enhanced soil carbon sequestration (Lal 2004). Lal (2004) estimated that enhanced management practices had the potential to sequester 0.4–1.2 Gt C / yr while simultaneously improving crop yields. It is important to note that there is a finite limit to sequestration potential, and that once a certain amount of carbon has been sequestered in soils, the soils are less able to function as carbon sinks. It is for this reason that degraded soils and ecosystems are thought to have the highest potential for carbon sequestration.

Relationship between biodiversity and the carbon cycle at the global level

41. A global mapping study concluded that the global distribution of organic carbon density is correlated to the spatial distribution of biological diversity, as measured by species richness (Strassburg et al. 2010). This suggests that links between activities to conserve biological diversity and organic carbon may exist at the global scale. Along these lines, there is active interest in developing spatially-explicit products to assist in conservation planning, such as the UNEP/CBD/LifeWeb Interactive Carbon Calculator.[6]

42. A global study of the relationship between biodiversity and carbon sequestration rates did not reveal any significant global correlation (Midgley et al. 2010), but correlations may exist at the local level. The relationship between biodiversity and carbon cycling is increasingly of interest but requires further study. This analysis also described several knowledge gaps and future research directions regarding the relationship between biodiversity and carbon cycling, such as:

(a) The potential future role of wildfire in compromising above-ground carbon storage and altering biodiversity;

(b) At high latitudes, decomposition represents a crucial threat for the long-term stabilization of the carbon cycle, while the expansion of woody vegetation (shrubs and trees) in these systems has a complex mix of adverse and positive implications for their biodiversity, carbon sequestration, and radiative balance; and

(c) A mechanistic understanding of how important plant species richness is for the global carbon cycle, especially through its role at local and regional scales, is lacking.

43. A case-study of spatial correlation between biodiversity and carbon stock in the United Kingdom concluded that although biodiversity and carbon sometimes coincide, this is not always the case

---

[5] To 3-m depth.

[6] http://www.cbd.int/lifeweb/carbon/.

(Anderson et al. 2009). In fact, the study's authors report that by examining subregions of the United Kingdom, one could draw radically different conclusions about the relationship between biodiversity and carbon stocks. This emphasizes the importance of multi-scale decision-making in determining environmental priorities. They suggest that regional-level assessments of biodiversity and carbon stocks should take precedence over global-level studies.

IV. BIOMES OF PARTICULAR IMPORTANCE: CARBON DENSITIES AND SEQUESTRATION RATES[7]

44. When soil carbon is considered in the assessment of organic carbon stocks and sequestration potential, several ecosystems that have until recently received limited attention in policymaking prove to be relevant for climate change mitigation, particularly at the local or national levels.

45. The aggregate biome carbon storage described in section III, above, is a function of density and areal extent. While the absolute size of terrestrial carbon pools is valuable for global policymaking, at the project level, ecosystem and landscape-level carbon densities and sequestration rates are most relevant. For comparative purposes, the mean carbon density of tropical forests has been estimated between 170–250 t C / ha in biomass (Malhi et al. 2006; Chave et al. 2008; Lewis et al. 2009) and 90–200 t C / ha in soil (Amundson 2001), for a combined density of 260–450 t C / ha. Particular forest types have been estimated to have much higher densities; for example, Australian temperate moist *Eucalyptus regnans* forest has been estimated to have a total carbon density of 1867 t C / ha (Keith et al. 2009).

46. It is important to note that the methods of measurement of soil carbon differ between authors and some studies report soil carbon to different depths (e.g., between 10–15 cm and 30 cm). This makes direct comparisons difficult and imprecise; depths of measurement are noted where possible.

47. The biomes addressed are structured after Epple (2012) and Trumper (2009).

Blue carbon: mangroves, tidal marshes and seagrass meadows

48. The carbon dynamics of coastal ecosystems have recently received attention from the international community (Herr & Pidgeon 2011; Nellemann et al. 2009). The ability of these ecosystems to store and sequester significant quantities of carbon is now recognized, although significant knowledge gaps remain.

49. In a recent study of sequestration rates in these ecosystems, several important knowledge gaps for coastal ecosystems were identified (Sifleet et al. 2011):

(a) How are sequestration rates affected by ecosystem loss, and what is the fate of existing sediment carbon stocks?

(b) How are sequestration rates and carbon stocks in sediments affected by climate change?

Mangroves

50. Mangroves are highly threatened coastal ecosystems, covering approximately 130–150 Mha (Mcleod et al. 2011), or a relatively restricted extent, primarily in tropical Asia. A review of mangrove carbon density, based on available information reported average soil carbon density[8] of 155 to

---

[7] Note that the figures for certain biomes in this section draw very heavily on the review by Epple (2012); other sections include diverse sources.

[8] First meter of soil only, which may represent a significant underestimate as mangrove sediments may be far deeper (Donato et al. 2011).

1150 t C / ha and biomass carbon density of 7 to 614 t C / ha (Sifleet et al. 2011).[9] A large proportion of carbon is stored in soils due to root-derived carbon stored as peat and protected from decomposition by anoxia; high storage densities may reflect past sea-level rise (McKee et al. 2007). Although there are few studies of the effect of land-use change/clearing on mangrove carbon stock, it is likely that mangrove clearing results in significant carbon efflux (e.g. Lovelock et al. 2011). It has also been reported that soil organic carbon in mangroves is particularly vulnerable in the case of land-use change (Matsui et al. 2010). Mangrove carbon sequestration rates are estimated at 0.03–6.54 t C / ha / yr, with most estimates below 1.9 t C / ha / yr (Sifleet et al. 2011).

51. Sifleet et al (2011) identified knowledge gaps with respect to the organic carbon stocks and sequestration rates in mangroves:

(a) Sampling is weak across different mangroves types (e.g. estuarine and coastal) and regions;

(b) Highly variable estimates of carbon density; and

(c) Limited observational studies on the effect of land-use change and clearing on mangrove soils carbon stocks.

52. Mangroves are highly biodiverse and provide multiple ecosystem services, including coastal protection. Conservation activities in mangroves are likely to produce strong co-benefits for biodiversity while conserving potential large carbon stocks. Additionally, although opportunity costs can be high for local stakeholders, mangrove conservation is likely to have a high benefit-cost ratio (TEEB 2009). A methodology[10] permitting the inclusion of projects seeking to restore degraded mangroves in the Clean Development Mechanism (CDM) carbon market has recently been approved. However, this methodology does not include the soil carbon pool.

Coastal marshes

53. Coastal tidal marshes are tidal ecosystems that range globally, but are most common in temperate areas; these store significant volumes of carbon in sediments due to anoxia (Sifleet et al. 2011). Global area has been estimated at 22–400 Mha (Mcleod et al. 2011). Current estimates of global loss rates of salt-marsh habitats are approximately 1–2 per cent per year with very high uncertainty (Sifleet et al. 2011).

54. Carbon stocks estimated for biomass in salt marshes are 1.3–4.99 t C / ha for above-ground biomass and 0.9–13.9 t C / ha for below-ground biomass. Measured soil carbon stocks[11] have been reported to range between 47.45 to 1900 t C / ha, with most estimates falling between 245.5 t C / ha and 463.6 t C / ha. These stocks are very high and are probably an underestimate because they only address the first meter of soil, while some sediments are much deeper and are on average similar in size to those of moist tropical forest. Sequestration rates in salt marshes have been estimated at between 0.002 t C / ha to 17.13 t C / ha; with the majority of measurements below 2.2 t C / ha (Sifleet et al. 2011). Although no direct empirical studies are available on emissions following draining , studies of carbon stocks in drained salt marshes estimate emissions of 9.72 t C / ha / yr in the Southeastern United States and 23.2–45.5 t C / ha / yr in California (Sifleet et al. 2011).

---

[9] The figures presented here have been converted from those in the Sifleet (2011) reported in t CO_2 to t C by multiplying by (12/44).

[10] http://cdm.unfccc.int/methodologies/DB/CKSXP498IACIQHXZPEVRJXQKZ3G5WQ/view.html

[11] To the first meter of soil.

55. Because of their saline sediments, salt marshes are not a significant source of methane, a potent greenhouse gas, in contrasts to freshwater wetlands, which tend to release large quantities of methane (Sifleet et al. 2011).

56. Major information gaps identified concerning carbon and coastal marshes (Sifleet et al. 2011) include:

(a) There is very little information about the areal extent of salt marshes outside of North America;

(b) There is almost no information on sequestration rates outside of Europe and North America; and

(c) There is no direct empirical information about emissions from salt marsh clearing or draining.

Seagrass meadows

57. Seagrass meadows are coastal, underwater ecosystems that consist of mats of flowering plants. Seagrass meadows are widely considered some of the most productive ecosystems on the planet, and accumulate carbon primarily in sediments. The global area of seagrass beds is very poorly known: Silfleet et al. (2011) report global estimates from several studies at between 17–430 Mha, while Mcleod et al. (2011) estimate 170–600 Mha. Global estimates of seagrass loss are 11000 ha globally since 1980 and a total loss of 7.2 Mha in the last 100 years (Walcott et al. 2009).

58. A meta-analysis of seagrass carbon storage and sequestration reported seagrass carbon storage in above-ground biomass[12] of 0.002–6.2 t C / ha, with significant variability (Sifleet et al. 2011). Soil carbons stocks were reported as ranging between 218.2 t C / ha – 1800 t C / ha, although these were based on only seven measurements, did not include bulk density estimates and should be treated as highly preliminary. Carbon sequestration rates in seagrass meadows are more intensively studied and range between -21 t C / ha / yr to 23 t C / ha / yr, with most estimates below 1.9 t C / ha. In fact, many seagrass meadows were reported as having negative carbon sequestration rates (Sifleet et al. 2011).

59. There are significant knowledge gaps concerning seagrass meadows and carbon storage and sequestration (Sifleet et al. 2011) including:

(a) There is essentially no information on the effect of clearing/degradation on sequestration or storage of carbon;

(b) There is a lack of information on the areal extent of seagrasses; and

(c) There is very limited information on sediment carbon stocks; only from seven sites in Europe.

Tundra

60. Tundra ecosystems are characterized by cold temperatures and small, hardy vegetation. Tundra ecosystems have low above-ground biomass but a higher degree of below-ground biomass (De Deyn et al. 2008). Because of low temperatures that lead to slow decomposition rates and cryogenic mixing of soils, tundra ecosystems store vast quantities of carbon in sediments. Epple (2012) cites carbon storage densities between 128–440 t C / ha down to 1 m. However, tundra soils can be hundreds of meters deep.

---

[12] Above-ground in the sense of above sediments.

Total ecosystem storage down to 3 m has been reported as high as 1672 Gt C in permafrost regions (Tarnocai et al. 2009).

61. There is uncertainty and a significant knowledge gap as to the degree to which the soils and permafrost of the Arctic region are currently a carbon sink or source; studies based on bottom-inventory approaches are inconclusive (Hayes et al. 2011). There is significant concern that increased temperatures in the Arctic will thaw soils, increase decomposition and increase the frequency of fires. Together these are expected to release vast quantities of carbon (Schuur et al. 2009; Mack et al. 2011) in a positive feedback exacerbating climate change. A recent modelling approach to update IPCC modelling with the new understanding of soil organic pools and an improved model of permafrost dynamics reported that the permafrost region will probably become a net source of carbon by 2100, although the model displayed significant uncertainties (Koven et al. 2011).

Temperate grasslands and savannahs

62. Temperate grasslands are distributed globally and have been nearly wholly converted (Trumper et al. 2009). Temperate grasslands are characterized by high degrees of grazing. The literature review by Epple (2012) reported biomass carbon stocks between approximately 7–21 t C / ha in both above- and below-ground pools. Soil carbon estimates reported are 191 t C /ha for temperate grassland soils down to three meters (Jobbágy & Jackson 2000) and Epple (2012) reports values of 236 t C / ha for the top 1 m of soil. Finally, Amundson (2001) reports average values of 133 t C /ha for cool temperate steppe and 76 t C / ha for temperate thorn steppe. Trumper et al. (2009) reports average biomass values of 7 t C / ha and soil carbon values of 133 t C / ha.

63. The primary threat to soil carbon stocks in temperate grassland is overgrazing and degradation by livestock. Grazing land management may be an important technique to restore degraded soils and may have positive outcomes for biodiversity (Conant 2010). Sustainable grazing management practices mostly attempt to increase carbon inputs to the soil and have been estimated to be able to sequester 0.35 t C / ha / yr (Conant 2010).

64. Ducks Unlimited is currently implementing a project to develop carbon offsets for avoided conversion of grasslands in North America.[13] The project plans to develop a measurement, reporting and verification (MRV) methodology for the project to certify emission reductions. The project team has already identified lack of sampling information of greenhouse gases in their focal region of North American soils to calibrate models as an important gap to be filled.

Tropical grasslands and savannahs

65. Tropical grasslands and savannahs are broad ecosystem types comprising a continuum from treeless grasslands to open forest. The definition of tropical grasslands and savannahs for this note follows Epple (2012). Fire is an important characteristic of savannah biomes. Tropical grasslands and savannahs are primarily located in Africa, South America and Australia (Grace et al. 2006).

66. Total carbon stocks of global tropical grasslands based on the UNEP-WCMC top-down analysis have been estimated at 285 Gt C, with an average carbon density of 137 t C / ha (Trumper et al. 2009; Epple 2012). Grace (2006) provides literature-review based estimates of 9.4 t C/ ha for above-ground biomass carbon, 19 t C/ ha for below-ground biomass carbon, and 174 t C/ ha for soil organic carbon. Generally, above-ground biomass is a function of the density of trees in the ecosystem. A recent study of African savannahs reported total soil organic carbon stock of approximately 110 t C / ha, although it

---

[13] http://www.c-agg.org/docs/resources/CIG%20GHG%20DU.pdf.

reported very high spatial heterogeneity and little correlation between soil carbon stocks and vegetation. When soil carbon stocks are considered, these ecosystems can have total carbon densities that approach those of forests.

67. Ciais et al. (2011) and Epple (2012) reported strong knowledge gaps with respect to tropical savannahs and grassland ecosystems. They also reported that these ecosystems are not well incorporated into terrestrial ecosystem models, which makes their role in the global carbon cycle difficult to assess. As an example, they cite the fact that most models of fire activity for grasslands are parameterized by data derived from temperate grasslands.

68. Savannah ecosystems are highly threatened with habitat conversion, which has been estimated at around 1 per cent per year and may have constituted a large proportion of converted ecosystems (Grace et al. 2006). Land change to croplands or intensive grazing can release large quantities of carbon from biomass and soils but the long-term effect critically depends on soil type and subsequent management practices (Grace et al. 2006). In fact, because of high rates of loss and degradation, emission from savannah ecosystems may approach emissions from tropical deforestation (Grace et al. 2006).

69. Main carbon management strategies are grazing-land management (e.g. reduced stocking densities) and fire management. However, these areas frequently have nomadic high population densities; this means that implementation soil carbon sequestration schemes, through, for example, carbon credits for grazing land management schemes, will encounter issues of land tenure (Conant 2010; Epple 2012). Other potential options for soil carbon sequestration and the conservation of soil carbon stocks stem from fire management practices. While fire is a natural component of savannah ecosystems, it releases large quantities of carbon, although this is mostly a short-term loss and is re-accumulated upon regrowth (Ciais et al. 2011). Fire management may have negative or positive effects on biological diversity: high suppression regimes typically lead to rare, large fires and related effects on biodiversity. Alternatively, controlled dry-season fires may reduce impact on biodiversity while maintaining carbon sequestration benefit (Douglass et al. 2011). Douglass et al. (2011) emphasize the importance of management for carbon sequestration and biodiversity in Australia's savannahs.

70. A soil carbon sequestration project in Kenya hopes to increase soil carbon storage through control of grazing and fire,[14] and is currently developing a Verified Carbon Standard (VCS)[15] methodology to certify emissions reductions.

Deserts and dry shrublands

71. Deserts and dry shrublands cover a large surface area. Trumper et al. (2008) provide a global mean carbon density of desert and dry shrubland ecosystems of approximately 64 t C / ha (Epple 2012).

72. Biomass carbon stocks in deserts and shrubs are relatively low: Epple (2012) reports values from the literature of between 0.09 t C / ha – 3.3 t C / ha. Because aridity reduces the rate of microbial decomposition and soil respiration, soil carbon stocks in desert and dry shrublands can be relatively higher. Epple (2012) reports values in the literature between 14 t C / ha – 270 t C / ha, with lower stocks in tropical deserts and higher stocks in temperature and boreal deserts and shrublands. Vegetation distribution can have significant effect on the carbon storage in soils (White II et al. 2009).

73. While some recent studies have reported localized high measurements of carbon sequestration in desert ecosystems, a recent study indicates that there is still a significant knowledge gap as to the carbon

---

[14] http://www.jadorallc.com/mbirikani_project.html.

[15] VCS is a leading certifier of emissions reductions for voluntary carbon markets. http://www.v-c-s.org/methodologies/alm-adoption-sustainable-grassland-management-through-adjustment-fire-and-grazing.

balance of deserts ecosystems (Schlesinger et al. 2009). While widespread policies to restore degraded dry shrublands may increase soil carbon sequestration and might have positive economic and biodiversity outcomes, low carbon density means that project-based mitigation potential is likely limited (Epple 2012).

Agroecosystems: biodiversity and soil carbon sequestration in croplands and rangelands

74. Agricultural soils contain less soil carbon than natural ecosystems, and this loss of soil carbon from soils constitutes a major historical emission of carbon of around 42–78 Gt C or approximately 20 to 80 tons C / ha (Lal 2004). However, because of their relatively impoverished condition, there is significant mitigation potential in agricultural soils (Lal 2004). Total mitigation potential of the world's soils has been estimated at from between 0.4–0.6 Gt C / yr to a high of 0.6–1.2 Gt C / yr (Lal 2004).

75. Lal (2008) identified generic recommended management practices thought to increase soil carbon sequestration in agricultural landscapes:

(a) Development of a positive carbon balance through conversion of plow tillage to conservation tillage or no-till farming along with the use of crop residue mulch and cover cropping;

(b) Increasing in plant-available water resources in the root zone through enhancement of infiltration rate, water harvesting and recycling, supplemental irrigation, and minimizing losses due to soil evaporation;

(c) Creation of a positive nutrient budget through integrated nutrient management, manuring, and judicious use of chemical fertilizers;

(d) Adoption of complex cropping systems including agroforestry; and

(e) Choice of appropriate crops and pastoral species most suited for the specific soils and climatic conditions.

76. In another analysis, Smith, Fang et al. (2008) provide a list of management practices thought to increase soil carbon sequestration:

(a) Cropland management:
  - Agronomy;
  - Nutrient management;
  - Tillage/residue management;
  - Water management (irrigation, drainage);
  - Agroforestry;
  - Set-aside and land-use change;

(b) Grazing land management / pasture improvement:

  - Managing grazing intensity;
  - Increased productivity (e.g. fertilization);
  - Nutrient management;
  - Fire management;
  - Species introduction (e.g. legumes);

(c) Management of organic soils:

  - Avoided drainage of wetlands;

(d) Restoration of degraded soils:

  - Erosion control, organic amendments, nutrient amendments.

77. Smith, Martino et al. (2008) also provide tables of expected mitigation potential in terms of carbon sequestration organized by climate zone (cool-dry, cool-moist, warm-dry and warm-moist). The activities with the great mitigation potential are restoration of organic soils (wetlands) and degraded lands, across all climatic types. These restoration practices are likely to have positive biodiversity outcomes. At the project level, biodiversity impact certification schemes can be used to certify the effects of sequestration projects on biological diversity and promote best practices to achieve multiple benefits.[16]

78. In degraded semi-arid soils in particular, increasing fallow periods and the conversion of degraded croplands to rangelands are seen as options to increase soil carbon sequestration (Olsson & Ard 2002).

79. There have only been a handful of soil carbon sequestration projects attempting to access carbon finance. Nearly all projects focus on agricultural and rangeland management. A key constraining factor to the implementation of soil carbon sequestration projects is lack of appropriate and cost-effective methodologies to account for soil carbon changes due to land management. Indeed, a major problem identified from reviews of an early project in community rangeland management in Sudan was the lack of proper carbon accounting to verify reported sequestration.[17]

80. Measurement, reporting and verification (MRV) methodologies are essential for projects to certify emissions reductions due to project activities and consequently access carbon finance through the carbon compliance (e.g. UNFCCC/CDM, EU/ETS) or voluntary markets. MRV schemes ensure that project produce emissions reductions that are real, additional and permanent. Currently, soil carbon credits are not permitted in any international compliance market. They are, however, traded in the Canadian province of Alberta, where payment is made for conversion to no-till farming.[18] Agricultural carbon credits for grazing-land management and conservation agriculture (e.g. no-till farming) were traded on the voluntary Chicago Carbon Exchange[19] although that exchange is now closed. The Food and Agriculture Organization of the United Nations (FAO) has recently published a guide to smallholder soil carbon sequestration projects in agriculture (FAO 2011).

81. The Kenya Agricultural Carbon Project,[20] funded by the World Bank, is the first agricultural soil carbon sequestration project in Africa. The project is ongoing and seeks to promote sustainable agricultural practices (e.g. agroforestry, cover crops, crops rotation, mulching, etc.) to sequester an estimated 60,000 t CO_2 (~16 363 t C) annually. Important co-benefits include increasing soil fertility via increased soil organic matter and resilience to climate change. The project has developed a new VCS standard entitled “Adoption of Sustainable Agricultural Land Management”,[21] which was recently approved (December, 2011) after lengthy revisions. While this standard has the potential for use in other agricultural carbon sequestration projects, it has restrictive applicability conditions, among others, that: (i) the land must already be degraded, (ii) cultivated land in adjacent areas must be constant or increasing and (iii) the scientific work must prove that the RothC model of soil carbon dynamics is applicable to the region. These applicability conditions demonstrate the frequently high transaction costs of emerging soil carbon sequestration projects. Several NGOs have expressed concern with this project: for example, one group suggests that carbon credit generated by the project will amount to no more than about USD 24 per farmer over twenty years, although these figures are not peer-reviewed or validated.[22] The World Bank’s

---

[16] E.g., the Climate, Community and Biodiversity Project Design Standards (CCB Standards) - http://www.climate-standards.org/standards/index.html.

[17] Community Based Rangeland Rehabilitation, GEF Project # UNSO/SUD/93/G31.

[18] http://environment.alberta.ca/0923.html.

[19] https://www.theice.com/ccx.jhtml.

[20] http://web.worldbank.org/external/projects/main?pagePK=64283627&piPK=73230&theSitePK=40941&menuPK=228424&Projectid=P107798.

[21] http://www.v-c-s.org/sites/v-c-s.org/files/VM0017%20SALM%20Methodolgy%20v1.0.pdf.

[22] http://www.iatp.org/documents/elusive-promises-of-the-kenya-agricultural-carbon-project.

environmental impact assessment[23] estimates that the project activities will have beneficial outcomes for biodiversity: by promoting the use of native species in agroforestry and increasing agrobiodiversity with diverse plantings.

82. A survey of VCS methodologies under development[24] reveals five further methodologies primarily concerned with soil carbon at various stages of development.

83. In terms of activities to manage soil carbon, the implementation of sustainable agricultural land management techniques may or may not have beneficial impacts on biodiversity. A review of soil carbon management techniques and biodiversity (Cambridge Conservation Initiative 2011) concluded that these techniques can be beneficial to biodiversity, but only:

(a) If incentives for *land-sharing* approaches (e.g. retention of buffer strips, shade crops, and reduced stocking rates in natural grasslands) to support biodiversity do not displace production;

(b) If these management changes also occur in areas where underlying patterns of soil carbon coincide with biodiversity; or

(c) If the need for further land conversion is reduced by restoring carbon to degraded soils so as to increase productivity and generate biodiversity benefits via *land-sparing***.**

V. RELEVANCE TO THE AICHI BIODIVERSITY TARGETS

84. In Nagoya, in 2010, the Conference of the Parties to the Convention on Biological Diversity adopted the Strategic Plan for Biodiversity 2011-2020. The Strategic Plan includes five Strategic Goals and 20 Aichi Biodiversity Targets. As a global biodiversity strategy, the Strategic Plan creates a strong mandate for increased understanding and consideration of the relationship between biodiversity conservation and sustainable use and the conservation and restoration of organic carbon stocks. Indeed, the full and effective implementation of the Strategic Plan will have impacts on the global carbon cycle.

Target 5: By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.

85. The full and effective implementation of target 5 would make a significant contribution to efforts to mitigate global climate change. As outlined above, ecosystems store and sequester carbon dioxide. When these ecosystems are lost, degraded or converted to agriculture, their ability to sequester carbon can be compromised and their significant carbon stores can be released. Conversion of ecosystems and land-use change, primarily forest degradation has contributed more than 200 Gt C to atmosphere (SCBD 2009). Halting land-use change would reduce the current flux from land-use change of 1.5±0.7 Pg C / yr of carbon dioxide to the atmosphere (Le Quéré et al. 2009) and would secure the current terrestrial carbon sink of approximately 4.7±1.2 Gt C / yr. Recent research has reported that natural old-growth forests continue to accumulate carbon (Luyssaert et al. 2008), and that they are also indispensible in biodiversity conservation (Gibson et al. 2011).

86. With respect to soil carbon, reducing the rate of loss of highly threatened ecosystems including peatlands, mangroves and tidal wetlands and seagrasses will conserve significant carbon stocks that might otherwise be released as a result of degradation or conversion. In addition, if Parties are able to produce

---

[23] http://www-wds.worldbank.org/external/default/main?pagePK=64193027&piPK=64187937&theSitePK=523679&menuPK=64187510&searchMenuPK=51351213&theSitePK=40941&entityID=000334955_20100205054330&searchMenuPK=51351213&theSitePK=40941.

[24] Available at http://www.v-c-s.org/methodologies/in-development.

high-resolution maps of biomass and soil carbon densities, these might be used to prioritize activities towards achieving target 5 for areas of high carbon density. This is particularly important in light of the information presented above: soil carbon stocks tend to be variable and heterogeneously distributed even among biomes.

87. To fully account for carbon mitigation benefits of the implementation of target 5, much better and higher-resolution spatial information on the distribution of global carbon stocks is required, including a comprehensive global treatment of soil carbon stocks. National-level accounting of carbon stock, including soils, will inform this process.

88. Ecosystem-based mitigation activities, in particular financial incentives to avoid conversion (e.g. REDD+, soil sequestration carbon payments), may be an important tool to reach target 5, although issues of biodiversity safeguards and carbon leakages need to be comprehensively addressed.

Target 7: By 2020, areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.

89. While carbon densities and sequestration potential per unit area are relatively low in agricultural systems, the large extent of croplands creates great scope for action. As outlined above, a host of management activities are known to increase soil carbon sequestration. Implemention either at the project level, potentially using MRV protocols to access carbon finance, or as national-level mitigation policies, will likely have impacts on biodiversity. As pointed out earlier, the nature of interventions, either “land-sharing” or “land-sparing” approaches, can be beneficial to biodiversity, but they can also have negative impacts.

Target 9: By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.

90. Invasive alien species cause great changes to ecosystem function, including large changes in carbon cycling dynamics in all pools (Liao et al. 2008). Generally, invasive plants increase productivity and respiration, which potentially increases carbon storage, although few studies account for all carbon pools and converse results have been reported (Koteen et al. 2011). A study of invasive grasses in California’s grasslands reported a reduction in ecosystem carbon storage, particularly in soils (Koteen et al. 2011). This invasion, which replaced drought-tolerant perennial grasses with invasive annuals, reduced carbon storage by 40 t C / ha in the first 50 cm of soil. There are significant gaps in knowledge with respect to the long-term effect of invasion on carbon cycling, especially under climate change, and what the consequences of the implementation of target 9 on carbon cycling might be.

Target 11: By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes.

91. It is already established that protected areas, in general, contain more carbon per unit area than unprotected areas: they contain 15.2 per cent of the global carbon stock (312 Gt C) while protecting only 12.2 per cent of the earth’s surface (Campbell et al. 2008). Full and effective implementation of target 11 would safeguard stocks of carbon from degradation. The exact quantity of carbon secured if target 11 is implemented would require detailed spatial analysis, but estimates should be feasible given global carbon

storage data sets and several scenarios of coverage choices by protected area planners (e.g. representativeness of ecoregions, etc.)

92. The consideration of soil carbon together with biomass carbon in a spatially explicit framework will allow Parties to make conservation decisions to maximize carbon sequestration and biodiversity conservation. In a climate where conservation resources are limited, it is important to emphasize that although all ecosystems store carbon, certain ecosystems store particularly high carbon densities, primarily in the soils, for example: peatlands, mangroves, tidal marshes and other wetlands. Decision support tools like the UNEP/CBD/LifeWeb carbon calculator are useful planning tools but they do not replace fine-scale national or regional data for high carbon density ecosystems.

93. Given the very high carbon densities of these ecosystems, high levels of threat and associated biodiversity and ecosystem services (Sifleet et al. 2011), full and effective coverage of 10 per cent of marine and coastal areas – and in particular mangroves, seagrass beds and tidal marshes – may have strong benefits for climate change mitigation and biodiversity conservation, despite significant knowledge gaps. Indeed, one study estimated that the global importance of carbon sinks in these ecosystems might rival those of terrestrial forests; although because of significant uncertainly in rates and areal extents, more study is required (Mcleod et al. 2011).

Target 15: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification.

94. Restoration of degraded ecosystems can sequester carbon and increase the contribution of biodiversity to organic carbon stocks. Much attention has focused on forest restoration, for example the Bonn Challenge.[25] Degraded non-forest ecosystems have significant potential for restoration to promote both carbon sequestration and biodiversity conservation. Indeed, a recent research study has shown that biodiversity restoration in grasslands can have important soil carbon sequestration benefits, especially if leguminous grasses are sown (De Deyn et al. 2011).

95. As recent meta-analyses of the effect of land-use changes on SOC stocks demonstrate (above), there is significant potential to increase SOC stocks by promoting the following land-use transitions:

- Croplands to grasslands;
- Croplands to fallow;
- Croplands to secondary forest or pasture.

96. If these land-use transitions are accomplished on degraded lands, the effect would be expected to be greater. Additionally, the restoration of degraded croplands lands has been widely identified as being beneficial to biodiversity, with the caveats identified above concerning land-sharing or land-sparing (Cambridge Conservation Initiative 2011). While forest-to-pasture transitions may increase SOC stocks, they will probably not increase whole ecosystem carbon storage and would likely have negative impacts on biodiversity. Finally, afforestration of croplands and degraded croplands should be undertaken with consideration of safeguards with regard to water usage, invasive species and biodiversity conservation.[26]

97. Degraded soils contain on average less carbon than healthy soils. As described above, dryland ecosystems (grasslands, savannahs and deserts) are vulnerable to desertification. While the potential of soil carbon sequestration to restore soil carbon stocks is low per unit area, their large extent means that the overall sequestration potential of drylands is very high.

---

[25] http://ideastransformlandscapes.org/.

[26] See CBD Technical Series No. 51: REDD+ and Biodiversity.

REFERENCES

Amundson, R. (2001). The Carbon Budget in Soils. *Annual Review of Earth and Planetary Sciences*, 29, pp. 535-562. Available at: http://www.annualreviews.org/doi/pdf/10.1146/annurev.earth.29.1.535.

Anderson, B.J. et al. (2009). Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology*, 46(4), pp. 888-896. Available at: http://doi.wiley.com/10.1111/j.1365-2664.2009.01666.x [Accessed July 15, 2011].

Blankinship, J.C., Niklaus, P.A. & Hungate, B.A. (2011). A meta-analysis of responses of soil biota to global change. *Oecologia*, 165(3), pp. 553-65. Available at: http://www.ncbi.nlm.nih.gov/pubmed/21274573 [Accessed July 8, 2011].

Bond-Lamberty, B. & Thomson, A. (2010). Temperature-associated increases in the global soil respiration record. *Nature*, 464(7288), pp. 579-82. Available at: http://www.ncbi.nlm.nih.gov/pubmed/20336143 [Accessed June 15, 2011].

Brown, D.J. et al. (2010). Soil Organic Carbon Change Monitored Over Large Areas. *Eos*, 91(47), pp. 441-456.

Cambridge Conservation Initiative (2011). *Can Payments for Soil Carbon Management Contribute to Biodiversity Conservation in production landscapes?*

Campbell, A. et al. (2008). Carbon Storage in Protected Areas – Technical Report, UNEP-WCMC

Campbell, A. et al. (2009). Review of the Literature on the Links between Biodiversity and Climate Change: Impacts, Adaptation and Mitigation. Secretariat of the Convention on Biological Diversity, Montreal. Technical Series No. 42, 124 pages. Available at: http://www.cbd.int/doc/publications/cbd-ts-42-en.pdf.

Canadell, J. et al. (2007). Factoring out natural and indirect human effects on terrestrial carbon sources and sinks. *Environmental Science & Policy*, 10(4), pp. 370-384. Available at: http://linkinghub.elsevier.com/retrieve/pii/S1462901107000238 [Accessed June 16, 2011].

Chave J. et al. (2008). Above-ground biomass and productivity in a rain forest of eastern South America. *Journal of Tropical Ecology* 24, pp. 355–366. doi:10.1017/S0266467408005075

Ciais, P et al. (2011). The carbon balance of Africa: synthesis of recent research studies. *Philosophical transactions. Series A, Mathematical, physical, and engineering sciences*, 369(1943), pp. 2038-57. Available at: http://www.ncbi.nlm.nih.gov/pubmed/21502175 [Accessed June 23, 2011].

Conant, R.T. (2010). Challenges and opportunities for carbon sequestration in grassland systems. *Integrated Crop Management*, 9, pp. 1-55.

Davidson, E.A. & Janssens, I.A. (2006). Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, 440(7081), pp. 165-173. Available at: http://www.nature.com/nature/journal/v440/n7081/full/nature04514.html [Accessed June 11, 2011].

De Deyn, G.B., Cornelissen, J.H.C. & Bardgett, Richard D (2008). Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecology letters*, 11(5), pp. 516-31. Available at: http://www.ncbi.nlm.nih.gov/pubmed/18279352 [Accessed June 10, 2011].

De Deyn, G.B. et al. (2011). Additional carbon sequestration benefits of grassland diversity restoration. *Journal of Applied Ecology*, 48(3), pp. 600-608. Available at: http://doi.wiley.com/10.1111/j.1365-2664.2010.01925.x [Accessed July 26, 2011].

Don, A., Schumacher, J. & Freibauer, A. (2011). Impact of tropical land-use change on soil organic carbon stocks - a meta-analysis. *Global*

Change Biology, 17(4), pp. 1658-1670. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2010.02336.x [Accessed July 7, 2011].

Donato, D.C. et al. (2011). Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience*, 4(5), pp. 293-297. Available at: http://www.nature.com/doifinder/10.1038/ngeo1123 [Accessed July 16, 2011].

Douglass, L.L. et al. (2011). The effect of carbon credits on savanna land management and priorities for biodiversity conservation. *PloS one*, 6(9), p.e23843. Available at: http://www.ncbi.nlm.nih.gov/pubmed/21935363 [Accessed September 23, 2011].

Eglin, T. et al. (2011). Overview on Response of Global Soil Carbon Pools to Climate and Land-Use Changes. In *Sustaining Soil Productivity in Response to Global Climate Change*. pp. 183-199.

Epple, C. (2012). The climate relevance of ecosystems beyond forests and peatlands: A review of current knowledge and recommendations for action. – BfN-Skripten 312. Bonn (German Federal Agency for Nature Conservation), 52 p. Available from http://www.unep-wcmc.org/medialibrary/2012/08/02/9607aa21/Epple2012Ecosystems.pdf.

FAO (2011). *Climate Change Mitigation Finance for Smallholder Agriculture A guide book to harvesting soil carbon sequestration benefits*,

Fitter, A. et al. (2005). Biodiversity and ecosystem function in soil. *Functional Ecology*, pp. 369-377.

Gans, J., Wolinsky, M. & Dunbar, J. (2005). Computational improvements reveal great bacterial diversity and high metal toxicity in soil. *Science*, 309(5739), pp. 1387-90. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16123304 [Accessed July 6, 2011].

GEFSOC Project Team (2005). *Assessmen off soil organic carbon stocks and change a national scale* K. P. and S. W. Eleanor Milne, Mark Easter, Carlos Eduardo Cerri, ed.,

Gibson, L. et al. (2011). Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature*, pp. 1-6. Available at: http://www.nature.com/doifinder/10.1038/nature10425 [Accessed September 14, 2011].

Goetz, S.J. et al. (2009). Mapping and monitoring carbon stocks with satellite observations: a comparison of methods. *Carbon balance and management*, 4, p.2. Available at: http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=2667409&tool=pmcentrez&rendertype=abstract [Accessed July 11, 2011].

Grace, J. et al. (2006). Productivity and carbon fluxes of tropical savannas. *Journal of Biogeography*, 33(3), pp. 387-400. Available at: http://doi.wiley.com/10.1111/j.1365-2699.2005.01448.x [Accessed June 21, 2011].

Granier, A. et al. (2007). Evidence for soil water control on carbon and water dynamics in European forests during the extremely dry year: 2003. *Agricultural and Forest Meteorology*, 143(1-2), pp. 123-145. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0168192306003911 [Accessed July 20, 2011].

Guo, L.B. & Gifford, R.M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*, 8(4), pp. 345-360. Available at: http://doi.wiley.com/10.1046/j.1354-1013.2002.00486.x.

Hajjar, R., Jarvis, D.I. & Gemmill-Herren, B. (2008). The utility of crop genetic diversity in maintaining ecosystem services. *Agriculture, Ecosystems & Environment*, 123(4), pp. 261-270. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0167880907002010 [Accessed August 18, 2011].

Harrison, R.B., Footen, P.W. & Strahm, B.D. (2011). Deep Soil Horizons : Contribution and Importance to Soil Carbon Pools and in Assessing Whole-Ecosystem Response to Management and Global Change. *Forest Science*, 57, pp. 67-76.

Hayes, D.J. et al. (2011). Is the northern high-latitude land-based CO_2 sink weakening? *Global Biogeochemical Cycles*, 25(3), pp. 1-14. Available at: http://www.agu.org/pubs/crossref/2011/2010GB003813.shtml [Accessed September 23, 2011].

Herr, D. & Pidgeon, E. (2011). *Blue Carbon Policy Framework: Based on the first Workshop of the International Blue Carbon Policy Working Group*, Gland, Switzerland and Arlington, USA: IUCN, CI.

IGBP-DIS, 2000. *Global Soil Data Products CD-ROM*, Oak Ridge, Tennessee. Available at: http://www.daac.ornl.gov/SOILS/guides/igbp-surfaces.html.

IPCC (2000). *Land Use, Land Use Change, and Forestry. Summary for Policymakers.* Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

IPCC (2001). *Climate change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change.* [Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell, and C.A. Johnson (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 881 pp.

Jobbágy, E.G. & Jackson, R.B., 2000. The Vertical Distribution of Soil Organic Carbon and its Relation to Climate and Vegetation. *Ecological Applications*, 10, pp. 423- 436.

Kapos, V. et al. (2008). *Carbon and biodiversity: a demonstration atlas*, Cambridge, UK: UNEP-WCMC.

Keith, H., Mackey, B.G. & Lindenmayer, D.B. (2009). Re-evaluation of forest biomass carbon stocks and lessons from the world’s most carbon-dense forests. *Proceedings of the National Academy of Sciences of the United States of America*, 106(28), pp. 11635-40. Available at: http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=2701447&tool=pmcentrez&rendertype=abstract [Accessed August 9, 2011].

Kirschbaum, M.U.F. (1995). The Temperature Dependence of Soil Organic Matter Decomposition, and the Effect of Global Warming on Soil Organic C Storage. *Journal of Soil Science*, 27(6), pp. 753-760.

Koteen, L.E., Baldocchi, D.D. & Harte, J. (2011). Invasion of non-native grasses causes a drop in soil carbon storage in California grasslands. *Environmental Research Letters*, 6(4), p.044001. Available at: http://stacks.iop.org/1748-9326/6/i=4/a=044001?key=crossref.e95e1c3c3ab2b1bf8f16344fed718099 [Accessed October 15, 2011].

Koven, C.D. et al. (2011). Permafrost carbon-climate feedbacks accelerate global warming. *Proceedings of the National Academy of Sciences of the United States of America*, (15). Available at: http://www.ncbi.nlm.nih.gov/pubmed/21852573 [Accessed August 24, 2011].

Lal, R. (2004). Soil carbon sequestration impacts on global climate change and food security. *Science*, 304(5677), pp. 1623-7. Available at: http://www.ncbi.nlm.nih.gov/pubmed/15192216 [Accessed June 12, 2011].

Lal, R. (2006). Carbon Management in Agricultural Soils. *Mitigation and Adaptation Strategies for Global Change*, 12(2), pp. 303-322. Available at: http://www.springerlink.com/index/10.1007/s11027-006-9036-7 [Accessed August 11, 2011].

Lal, R. (2008). Promise and limitations of soils to minimize climate change. *Journal of Soil and Water Conservation*, 63(4), p.113A-118A. Available at: http://www.jswconline.org/cgi/doi/10.2489/63.4.113A.

Lal, R. (2009). Sequestering carbon in soils of arid ecosystems. *Land Degradation & Development*, 454, pp. 441-454.

Le Quéré, C. et al. (2009). Trends in the sources and sinks of carbon dioxide. *Nature Geoscience*, 2(12), pp. 831-836. Available at: http://www.nature.com/doifinder/10.1038/ngeo689 [Accessed June 10, 2011].

Lewis, S. L. et al. (2009). Increasing carbon storage in intact African tropical forests. *Nature* 457, pp. 1003-1006.

Liao, C. et al. (2008). Altered ecosystem carbon and nitrogen cycles by plant invasion: a meta-analysis. *The New phytologist*, 177(3), pp. 706-14. Available at: http://www.ncbi.nlm.nih.gov/pubmed/18042198 [Accessed June 17, 2011].

Lovelock, C.E., Ruess, R.W. & Feller, I.C. (2011). CO_2 efflux from cleared mangrove peat. *PloS one*, 6(6), p.e21279. Available at: http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=3126811&tool=pmcentrez&rendertype=abstract [Accessed September 29, 2011].

Lützow, M. & Kögel-Knabner, I. (2009). Temperature sensitivity of soil organic matter decomposition—what do we know? *Biology and Fertility of Soils*, 46(1), pp. 1-15. Available at: http://www.springerlink.com/index/10.1007/s00374-009-0413-8 [Accessed June 24, 2011].

Luyssaert, S. et al. (2007). CO_2 balance of boreal, temperate, and tropical forests derived from a global database. *Global Change Biology*, 13(12), pp. 2509-2537. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2007.01439.x [Accessed June 10, 2011].

Luyssaert, Sebastiaan et al. (2008). Old-growth forests as global carbon sinks. *Nature*, 455(7210), pp. 213-5. Available at: http://www.ncbi.nlm.nih.gov/pubmed/18784722 [Accessed July 26, 2011].

Mack, M.C. et al. (2011). Carbon loss from an unprecedented Arctic tundra wildfire. *Nature*, 475(7357), pp. 489-492. Available at: http://www.nature.com/doifinder/10.1038/nature10283 [Accessed July 27, 2011].

Malhi Y., Wood, D., Baker, T.R., et al., (2006). The regional variation of aboveground live biomass in old-growth Amazonian forests, *Global Change Biology*, 12, pp. 1107–1138, doi: 10.1111/j.1365-2486.2006.01120.x.

Matsui, N. et al. (2010). Mangrove rehabilitation dynamics and soil organic carbon changes as a result of full hydraulic restoration and re-grading of a previously intensively managed shrimp pond. *Wetlands Ecology and Management*, 18(2), pp. 233-242. Available at: http://www.springerlink.com/index/10.1007/s11273-009-9162-6 [Accessed July 29, 2011].

McKee, K.L., Cahoon, D.R. & Feller, I.C. (2007). Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecology and Biogeography*, 16(5), pp. 545-556. Available at: http://doi.wiley.com/10.1111/j.1466-8238.2007.00317.x [Accessed August 12, 2011].

Mcleod, E. et al. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. *Frontiers in Ecology and the Environment* 9(10), pp. 552-560. Available at: http://www.esajournals.org/doi/abs/10.1890/110004 [Accessed October 7, 2011].

Midgley, G.F. et al. (2010). Terrestrial carbon stocks and biodiversity: key knowledge gaps and some policy implications. *Current Opinion in Environmental Sustainability*, 2(4), pp. 264-270. Available at: http://linkinghub.elsevier.com/retrieve/pii/S1877343510000461 [Accessed August 23, 2011].

Milcu, A. et al. (2010). Plant diversity enhances the reliability of belowground processes. *Soil Biology and Biochemistry*, 42(12), pp. 2102-2110. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0038071710002932 [Accessed August 31, 2011].

Murty, D. et al. (2002). Does conversion of forest to agricultural land change soil carbon and nitrogen? a review of the literature. *Global Change Biology*, 8(2), pp. 105-123. Available at: http://doi.wiley.com/10.1046/j.1354-1013.2001.00459.x.

Nellemann, C. et al. (2009). *Blue carbon: the role of Health Oceans in Binding Carbon*, United Nations Environment Programme, GRID-Arendal. Available at: www.grida.no.

Nielsen, U.N. et al. (2011). Soil biodiversity and carbon cycling: a review and synthesis of studies examining diversity-function relationships. *European Journal of Soil Science*, 62(1), pp. 105-116. Available at: http://doi.wiley.com/10.1111/j.1365-2389.2010.01314.x [Accessed June 22, 2011].

Niles, J.O. et al. (2010). *Carbon Finance in Ethiopian Rangelands : Opportunities for Save the Children / US Leadership*.

Olsson, L. & Ard, J. (2002). Soil Carbon Sequestration in Degraded Semiarid Agro-ecosystems: Perils and Potentials. *Ambio*, 31(6), pp. 471-477.

Parish, F. et al. (2008). *Assessment on Peatlands, Biodiversity and Climate Change: Main Report* F. Parish et al., eds., Global Environment Centre, Kuala Lumpur and Wetlands International, Wageningen.

Persiani, A. et al. (2008). Mediterranean grassland soil fungi: Patterns of biodiversity, functional redundancy and soil carbon storage. *Plant Biosystems - An International Journal Dealing with all Aspects of Plant Biology*, 142(1), pp. 111-119. Available at: http://www.tandfonline.com/doi/abs/10.1080/11263500701872713 [Accessed October 19, 2011].

Powers, J.S. et al. (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(15), pp. 6318-6322.

Powlson, D.S., Whitmore, A.P. & Goulding, K.W.T. (2011). Soil carbon sequestration to mitigate climate change: a critical re-examination to identify the true and the false. *European Journal of Soil Science*, 62(1), pp. 42-55. Available at: http://doi.wiley.com/10.1111/j.1365-2389.2010.01342.x [Accessed July 18, 2011].

Ruesch, A. & Gibbs, H.K. (2008). *New IPCC Tier-1 Global Biomass Carbon Map For the Year 2000*, Oak Ridge, Tennessee. Available at: http://cdiac.ornl.gov.

Sanchez, P.A. et al. (2009). Digital Soil Map of the World. *Science*, 325, pp. 6-7.

Sayer, E.J. et al. (2011). Soil carbon release enhanced by increased tropical forest litterfall. *Nature Climate Change*, 1(9), pp. 1-4. Available at: http://www.nature.com/doifinder/10.1038/nclimate1190 [Accessed August 23, 2011].

Scharlemann, J.P.W. et al. (2009). *Updated global carbon map*,

Schlesinger, W.H., Belnap, J. & Marion, G. (2009). On carbon sequestration in desert ecosystems. *Global Change Biology*, 15(6), pp. 1488-1490. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2008.01763.x [Accessed December 13, 2011].

Schmidt, M.W.I. et al. (2011). Persistence of soil organic matter as an ecosystem property. *Nature*, 478(7367), pp. 49-56. Available at: http://www.nature.com/doifinder/10.1038/nature10386 [Accessed October 5, 2011].

Schuur, E.A. et al. (2009). The effect of permafrost thaw on old carbon release and net carbon exchange from tundra. *Nature*, 459(7246), pp. 556-559. Available at: http://www.ncbi.nlm.nih.gov/pubmed/19478781 [Accessed July 21, 2011].

SCBD (2009). Connecting Biodiversity and Climate Change Mitigation and Adaptation: Report of the Second Ad Hoc Technical Expert Group on Biodiversity and Climate Change. Secretariat of the Convention on Biological Diversity, Montreal. Technical Series No. 41, 126 pages. Available at: http://www.cbd.int/doc/publications/cbd-ts-41-en.pdf.

SCBD (2011). REDD+ and Biodiversity. Secretariat of the Convention on Biological Diversity, Montreal. Technical Series No. 59, 65 pages. Available at http://www.cbd.int/doc/publications/cbd-ts-59-en.pdf.

Sifleet, S., Pendleton, L. & Murray, B.C. (2011). *State of the Science on Coastal Blue Carbon A Summary for Policy Makers*, Available at: http://nicholasinstitute.duke.edu/economics/naturalresources/state-of-science-coastal-blue-carbon.

Smith, P., Fang, C. et al. (2008). Impact of Global Warming on Soil Organic Carbon. *Advances in Agronomy*, 97, pp. 1-43.

Smith, P., Martino, D. et al. (2008). Greenhouse gas mitigation in agriculture. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, 363(1492), pp. 789-813. Available at: http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=2610110&tool=pmcentrez&rendertype=abstract [Accessed June 14, 2011].

Steinbeiss, S. et al. (2008). Plant diversity positively affects short-term soil carbon storage in experimental grasslands. *Global Change Biology*, 14(12), pp. 2937-2949. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2008.01697.x [Accessed August 31, 2011].

Strassburg, B.B.N. et al. (2010). Global congruence of carbon storage and biodiversity in terrestrial ecosystems. *Conservation Letters*, 3(2), pp. 98-105. Available at: http://doi.wiley.com/10.1111/j.1755-263X.2009.00092.x [Accessed July 20, 2011].

Tarnocai, C. et al. (2009). Soil organic carbon pools in the northern circumpolar permafrost region. *Global Biogeochemical Cycles*, 23(2), pp. 1-11. Available at: http://www.agu.org/pubs/crossref/2009/2008GB003327.shtml [Accessed July 25, 2011].

Trumbore, S.E. & Czimczik, C.I. (2008). An Uncertain Future for Soil Carbon. *Science*, 321, pp. 1455-1456.

Trumper, K. et al. (2009). *The Natural Fix? The role of ecosystems in climate mitigation*, Cambridge, UK: United Nations Environment Programme, UNEP- WCMC. Available at: http://www.grida.no/publications/rr/natural-fix/.

Turbe, A. et al. (2010). Soil biodiversity: functions, threats and tools for policy makers. *Reproduction*, (February).

Walcott, J., Bruce, S. & Sims, J. (2009). *Soil carbon for carbon sequestration and trading : a review of issues for agriculture and forestry*, Canberra.

White II, D.A. et al. (2009). Vegetation controls on soil organic carbon dynamics in an arid, hyperthermic ecosystem. *Geoderma*, 150(1-2), pp. 214-223. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0016706109000639 [Accessed July 28, 2011].

Yan, J. et al. (2011). Carbon uptake by karsts in the Houzhai Basin, southwest China. *Journal of Geophysical Research*, 116, p.G04012. Available at: http://www.agu.org/pubs/crossref/2011/2011JG001686.shtml [Accessed December 29, 2011].
