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Wind erosion reduces soil organic carbon sequestration falsely indicating ineffective management practices

Adrian Chappell [a,*], Jeffrey A. Baldock [b]

[a] CSIRO Land and Water, G.P.O. Box 1666, Canberra ACT 2601, Australia
[b] CSIRO Agriculture, Urrbrae, SA 5064, Australia

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ABSTRACT

Improved management of agricultural land has the potential to reduce greenhouse gas emissions and to reduce atmospheric CO_2 via soil carbon sequestration. However, SOC stocks are reduced by soil erosion which is commonly omitted from calculations of crop production, C cycling, C sequestration and C accounting. We used fields from the wind eroded dryland cropping region of Western Australia to demonstrate the global implications for C sequestration and C accounting of omitting soil erosion. For the fields we previously estimated mean net (1950s–1990) soil erosion of 1.2 ± 1.0 t ha^{-1} y^{-1}. The mean net (1990–2013) soil erosion increased by nearly four times to 4.4 ± 2.1 t ha^{-1} y^{-1}. Conservation agriculture has evidently not reduced wind erosion in this region. The mean net (1990–2013) SOC erosion was up to 0.2 t C ha^{-1} y^{-1} across all sampled fields and similar to measured sequestration rates in the region (up to 0.5 t C ha^{-1} y^{-1}; 10 years) for many management practices recommended for building SOC stocks. The minimum detectable change (MDC; 10 years) of SOC without erosion was up to 0.2 t C ha^{-1} y^{-1} whilst the MDC of SOC with erosion was up to 0.4 t C ha^{-1} y^{-1}. These results illustrate the generally applicable outcome: (i) if SOC erosion is equal to (or greater than) the increase in SOC due to management practices, the change will not be detectable (or a loss will be evident); (ii) without including soil erosion in SOC sequestration calculations, the monitoring of SOC stocks will lead to, at best the inability to detect change and, at worst the false impression that management practices have failed to store SOC. Furthermore, continued omission of soil erosion in crop production, C accounting and C sequestration will most likely undermine confidence in policy designed to encourage adoption of C farming and the attendant benefits for soil stewardship and food security.

1. Introduction

The amount of carbon dioxide (CO_2) captured and converted to soil organic carbon (SOC) annually via terrestrial net primary productivity (NPP) or released as CO_2 by soil microbial activity is about an order of magnitude greater than the annual increase in atmospheric CO_2 (Houghton et al., 1992). Soil contributes substantially to the global carbon cycle and small changes in the SOC stock may result in large changes of atmospheric CO_2 particularly over tens to hundreds of years (Giorgi, 2006). Land use change e.g., by clearing native vegetation for agriculture and grazing, perturbs the terrestrial ecosystem. The soil adjusts to the new land surface conditions and changes its soil organic carbon by mineralisation in response to changed soil moisture and temperature. Similarly, changes in the structure of the vegetation changes the amount of momentum extracted by the surface roughness (height, spacing, density) making available surplus energy to impact the soil and cause soil erosion by wind and water. Soil erosion removes preferentially the fine nutrient- and C-rich fractions of the topsoil which influences the composition and structure of the soil surface changing the soil albedo and temperature of the soil, the soil moisture holding capacity, the fertility and ultimately the agricultural productivity of the soil. Thus, perturbations to the terrestrial ecosystem change the C cycle and dust cycle (Shao et al., 2011) and the development of new equilibria are likely for both C and soil redistribution.

Improved management of agricultural land has the potential to reduce greenhouse gas (GHG) emissions (Smith et al., 2008) and to reduce atmospheric CO_2 via soil carbon sequestration (Lal, 2004). Soil erosion has apparently declined as a likely consequence of the widespread adoption of conservation agriculture (Montgomery, 2007; Chappell et al., 2012). However, these small-scale (plot-based) measurements extrapolated to regional decreases hide considerable variability (Chappell and Viscarra Rossel, 2013) indicating that on some farms erosion has not

* Corresponding author.
E-mail address: adrian.chappell@csiro.au (A. Chappell).

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reduced. The associated management practices (e.g., no-tillage, residue retention and diversification of cropping systems) are believed to have considerable potential to sequester soil organic carbon (Lal, 2004, 2015). However, there is evidence that the relative gain in SOC stocks may be due to a reduction or cessation of SOC losses (e.g., wind erosion) rather than an actual increase in SOC stocks (Li et al., 2014). Given the inextricable relation between the loss and gain of SOC and the loss and gain of soil (containing SOC) it is surprising to find that models of agricultural production (e.g., APSIM), C cycling (e.g., RothC), C sequestration and C accounting (e.g., FullCAM) typically ignore the impact of soil erosion (Chappell et al., 2015b). Of particular importance to this paper is the work by Sanderman and Chappell (2013). They showed that modest unrecognised amounts of soil redistribution ($10–20\ t\ ha^{-1}\ y^{-1}$ from experimental sites around Australia) produced uncertainties in sequestration rates of similar magnitude to measured sequestration rates (ca. 0.1 to $0.3\ t\ C\ ha^{-1}\ y^{-1}$) for many management practices recommended for building SOC stocks (Sanderman et al., 2011).

We find three misconceptions which are commonly used, at least in Australia, to justify and perpetuate the exclusion of soil erosion from crop production modelling, C cycling, C sequestration and C accounting. The first misconception is that soil erosion merely redistributes SOC within a field. Wind erosion and dust emission readily remove considerable quantities of soil and SOC from Australian fields in agricultural and rangeland regions (Chappell et al., 2012; Webb et al., 2013). The material is transported rapidly across vast distances and either deposited in other regions or removed from the terrestrial ecosystem to the ocean. It is not uncommon for water erosion to remove soil on sloping land (Loughran et al. 2004) and for the material to be deposited on adjacent fields or transported greater distances by streams and rivers and subsequently deposited on floodplains or in the ocean. In the case of either wind or water erosion, large amounts of soil erosion are not needed to make a significant difference to the SOC change over time because eroded material is typically considerably enriched (relative to the soil) with SOC (Webb et al., 2012). The second misconception (particularly prevalent in Australia) is that soil erosion is no longer a priority. However, each year cropland is lost due to soil degradation, predominantly by soil erosion (Lal, 2004) significantly reducing the cropland available for food production. The rate of soil loss is or has been until recently, considerably faster than the rate of soil renewal (Montgomery, 2007) imperilling future human food security and the environment (Koch et al., 2015). Consequently, soil erosion is one of the most serious environmental and public health problems facing human society today (Amundson et al., 2015). Sustainable intensification of agricultural production must reduce soil loss and maintain and enhance the soil resource. The third misconception is that there is no reliable and/or cost-effective method to estimate medium-term (ca 30–40 years) soil erosion. In contrast, a straightforward method has been demonstrated using the well-established ^{137}Cs technique to estimate the average net (time-integrated) soil redistribution (erosion and deposition) due to all processes of wind, water and tillage at a given location (Kachanoski and de Jong, 1984; Chappell et al., 2012). Recent work has provided a cost-effective approach to establishing the spatial mean (e.g., over a field) of ^{137}Cs-derived net soil redistribution and to detect change in the spatial mean over time (Li et al., 2015; Chappell et al., 2015a).

Soil monitoring networks are being considered in many countries (Morvan et al., 2008) to better understand carbon balances in terrestrial ecosystems and to verify the effects of land use or management practices on SOC change (Goidts et al., 2009). Consequently, it is essential to assess whether changes in SOC are detectable amongst the uncertainties caused by spatial heterogeneity, temporal variation, sampling approaches and analytical errors (Saby et al., 2008). These sources of uncertainty must be minimised to ensure that SOC sequestration is attributable to a net depletion of atmospheric CO_2 (Olson, 2013). Olson (2013) suggested that SOC sequestration should include only C acquired directly from the atmosphere and from a pre-defined monitored area. Natural or human-induced erosion and deposition of soil and the associated C it contains should be quantified and subtracted otherwise changes to SOC stocks may be attributed falsely to land use management practices in one of two ways:

1. Net increase in SOC stock may be due (at least partially) to soil deposition (e.g., accumulation of SOC enriched-dust) and performance of the management practice will be over-estimated.
2. Net decrease in SOC stock may be due (at least partially) to soil erosion and will reduce SOC storage and cause management practices to appear falsely ineffective i.e., performance of the management practice will be under-estimated.

The aim here is to demonstrate the impact that soil organic carbon erosion can have on C sequestration and the arising implications for C accounting. We describe and use a cost-effective sampling framework which establishes uncertainty without prior information about the spatial distribution of SOC. We show how a detectable change in SOC is calculated with and without soil erosion. We apply this sampling and detectable change framework to six fields in the typically wind eroded dryland cropping region of south-west Western Australia. We measure ^{137}Cs to estimate soil erosion and SOC and demonstrate for these fields the impact for C sequestration and C accounting of omitting SOC erosion. We believe the arising implications are generally applicable and demonstrate that SOC erosion should be included in the frameworks being considered by governments to mitigate against increases in CO_2.

2. Sampling theory and statistics

2.1. Sampling design

Simple random sampling may be adopted across a field but the sampling variance is usually larger than with most other types of design (for the same cost) because spatial coverage may be poor (Brus and Noij, 2008; de Gruijter et al., 2006). With stratified simple random sampling the field may be divided into (equal area) strata and simple random sampling applied within each stratum. The strata may be defined using ancillary variables and an arising classification. However, a regular grid may be used to stratify the field prior to sampling and in this case the strata have equal area and therefore equal volumes of soil may be collected at the sample location (Fig. 1). In this case, the field is stratified assuming no prior information about spatial variability within the field. The field is divided into four (or more) approximately equal areas or strata. Within each strata three locations to obtain soil samples are chosen using random numbers. The locations within each strata reduce the likelihood that they would create a clustered sample across the field. Consequently, these locations within each strata are assumed to represent a cost-effective approach to obtaining an unbiased sample of the spatial variability in soil properties across the field.

Many soil samples may be obtained from across the field but SOC and ^{137}Cs measurement costs of each sample make it expensive to ensure that they represent the field or area of interest adequately. A straightforward solution is to create a composite (or