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Australian net (1950s–1990) soil organic carbon erosion: implications for CO_2 emission and land–atmosphere modelling

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Abstract. The debate remains unresolved about soil erosion substantially offsetting fossil fuel emissions and acting as an important source or sink of CO_2. There is little historical land use and management context to this debate, which is central to Australia's recent past of European settlement, agricultural expansion and agriculturally-induced soil erosion. We use "catchment" scale ($\sim 25\,km^2$) estimates of ^{137}Cs-derived net (1950s–1990) soil redistribution of all processes (wind, water and tillage) to calculate the net soil organic carbon (SOC) redistribution across Australia. We approximate the selective removal of SOC at net eroding locations and SOC enrichment of transported sediment and net depositional locations. We map net (1950s–1990) SOC redistribution across Australia and estimate erosion by all processes to be $\sim 4\,Tg\,SOC\,yr^{-1}$, which represents a loss of $\sim 2\,\%$ of the total carbon stock (0–10 cm) of Australia. Assuming this net SOC loss is mineralised, the flux ($\sim 15\,Tg\,CO_2$-equivalents yr^{-1}) represents an omitted 12 % of CO_2-equivalent emissions from all carbon pools in Australia. Although a small source of uncertainty in the Australian carbon budget, the mass flux interacts with energy and water fluxes, and its omission from land surface models likely creates more uncertainty than has been previously recognised.

1 Introduction

The estimated effect of soil redistribution on the carbon cycle ranges from an annual global net source of $4.4\,Pg\,CO_2\,yr^{-1}$ (Lal, 2003) to a global net sink of $7.3\,Pg\,CO_2\,yr^{-1}$ (Stallard, 1998). Uncertainty in these estimates is largely attributed to mineralisation rates in the soil organic carbon (SOC) pools. The mineralisation rates are expected to either increase due to the breakdown of soil structure during erosion (Lal and Pimentel, 2008) or reduce as a consequence of SOC burial during deposition (Stallard, 1998). Based on the argument that SOC is dynamically replaced in eroding regions (Harden et al., 1999), several researchers have used studies of water and tillage erosion at the field/hillslope scale to support the tenet that soil erosion is acting as a net biospheric carbon sink (cf. Van Oost et al., 2007; Dlugoss et al., 2012). Berhe and Kleber (2013) suggested that the mass balance of carbon inputs and outputs must be considered when inferring protection of soil organic matter against decomposition in dynamic landscapes (Berhe et al., 2007). While the debate about whether SOC erosion is a source or sink of CO_2 has raised awareness of the significance of soil erosion for carbon cycling (Doetterl et al., 2012B) and carbon accounting (Sanderman and Chappell, 2012), the impact of erosion on the carbon cycle is yet to be resolved, and it appears that changes in land use and management have been neglected (Chappell et al., 2012).

Clearing of land for agriculture (cultivation) and grazing is widely recognised as being responsible for accelerating soil erosion. Conservation agriculture (minimum/zero tillage) and soil conservation measures in general have been

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a successful response to soil erosion on agricultural land (cf. Montgomery, 2007). These changes to land use and management have created phases in the recent soil erosion history. For example, European settlement (from 1788) transformed the Australian environment with extensive clearing of native vegetation for agricultural production, primarily pastoralism and, to a lesser extent, cropping (McAlpine et al., 2009). Marx et al. (2014) associated agricultural expansion between 1880 and 1990, compounded by droughts and the dust bowl era, with increased soil erosion. Conservation agriculture implemented in the 1980s considerably reduced dust emission (Marx et al., 2014) and ^{137}Cs-derived net (1990–2010) soil erosion (in SE Australia; Chappell et al., 2012). Evidently, SOC redistribution is a function of its residence times in the landscape, which is dependent on the distribution and change in land use and management.

Here we focus on the later part of agricultural expansion in Australia (1950s–1990) and quantify SOC erosion across the continent. We account for all erosion processes (wind, water and tillage), specifically including wind erosion and dust emission, which has the potential to preferentially remove SOC rapidly from terrestrial ecosystems (Webb et al., 2012, 2013; Chappell et al., 2013). Our estimates at the landscape or "catchment" scale (e.g. $> 1\,\mathrm{km}^2$) use measurements from across Australia at the hillslope scale but do not rely on extrapolations based on modelled gross erosion, which typically exclude deposition processes and hence neglect the balance of C inputs and outputs (net SOC redistribution).

The objective of this paper is to develop the first estimate of the impact of net soil redistribution by all processes on soil organic carbon (SOC) stocks across Australia. We use recent "catchment" scale ($\sim 25\,\mathrm{km}^2$) estimates of ^{137}Cs-derived net (1950s–1990) soil redistribution and SOC for Australia to calculate SOC net redistribution (carbon erosion). Our estimates of carbon erosion make explicit (a) the need to separately account for erosion and deposition and (b) the enrichment factor to account for the preferential removal by erosion of the fine, nutrient- and carbon-rich material from the soil. We classify total Australian net SOC redistribution by land use to demonstrate its impact for different current economic sectors. The significance for Australia is that there are no continental estimates of SOC redistribution. Consequently, it is expected that these estimates will reduce uncertainty and improve accuracy in carbon accounting with implications for greenhouse gas abatement and carbon sequestration storage and raise awareness of the agriculturally-induced impact of soil erosion on landscapes, agricultural systems and land–atmosphere interactions in land surface models.

2 Methods

2.1 Soil organic carbon redistribution model

Yan et al. (2005) provided a basis for further research on the estimation of eroded carbon. They suggested multiplying ^{137}Cs-derived wind erosion rates by the amount of carbon in the surface soil horizons or topsoil to estimate the annual average SOC loss to wind erosion in China. We modified the model of Yan et al. (2005; Eq. 1) by explicitly including an enrichment factor and separating the outcome of soil redistribution for net erosion (C_{eros}; t C ha^{-1} yr^{-1}):

$$C_{\mathrm{eros}} = E \times \mathrm{OC_e} \times P_\mathrm{e}, \qquad (1)$$

where E is ^{137}Cs-derived net soil redistribution (t soil ha^{-1} yr^{-1}), $\mathrm{OC_e}$ is the gravimetric ratio of organic carbon in the soil (gC / g soil) close to, or at, the source of erosion, and P is the enrichment factor (where, relative to the originating soil, $P > 1$ indicates an enrichment and $P < 1$ indicates a depletion) that accounts for the selective removal of SOC from the topsoil by wind erosion (Webb et al., 2012, 2013).

At locations where the outcome of all erosion events from all processes is net deposition, the modified model (Eq. 1) is inadequate. This is because material containing organic carbon deposited at a particular location has travelled from another (source) location where it likely preferentially removed organic carbon. During transport the coarser material will have been removed, leaving only the finest (nutrient and SOC-rich) fraction to reach its destination. Consequently, we require an additional model to handle the situation of SOC net deposition (C_{dep}; tC ha^{-1} yr^{-1}):

$$C_{\mathrm{dep}} = D \times \mathrm{OC_d} \times P_\mathrm{d}, \qquad (2)$$

where D is ^{137}Cs-derived net soil deposition (t soil ha^{-1} yr^{-1}). The implication of Eq. (2) is that, for the depositional locations, we need to know the SOC concentration ($\mathrm{OC_d}$) and the enrichment/depletion (P_d) of the material at its source, i.e. the source and sink must be linked. A justification for the values used in these terms is provided in Sect. 2.5.

2.2 ^{137}Cs-derived net (1950s–1990) soil redistribution

Soil erosion measurement and monitoring approaches, particularly in dryland environments (like Australia), require sufficiently long (ca. 15 years) and expensive campaigns to provide representative and reliable estimates of erosion rates (Roels, 1985). Even with such campaigns, the extrapolation of results from experimental plots and field studies to large areas is notoriously unreliable and unrealistic at the catchment scale and larger because of considerable spatial and temporal variation in sediment delivery (Roels and Jonker, 1983). The caesium-137 (^{137}Cs) technique overcomes most of the difficulties with long-term erosion monitoring programmes because it provides retrospective information on medium-term (ca. 40 years) net soil redistribution (Zapata, 2003) due to all processes including wind erosion and dust emission (Van Pelt, 2013). Although some limitations exist (Walling and Quine, 1991; Chappell, 1999; Parsons and Foster, 2011), the ^{137}Cs technique has been applied successfully in many countries at the field scale (Zapata, 2003) and used to investigate at the field scale whether accelerated erosion processes act as a source or a sink of atmospheric CO_2 (Quine and Van Oost, 2007).

Samples of ^{137}Cs have been combined with regionalised mapping techniques to make estimates over large areas and regions (de Roo, 1991; Chappell, 1998; Chappell and Warren, 2003) culminating recently in a map of Australian net soil redistribution for the continent (Chappell et al., 2011a, b). Statistically significant relationships between ^{137}Cs and SOC have been established for agricultural regions (e.g. Ritchie and McCarty, 2003; Ritchie et al., 2007; Wei et al., 2008), providing support for the movement of ^{137}Cs and SOC along the same physical pathways and through the same physical mechanisms (Martinez et al., 2009). These developments with the ^{137}Cs technique provide the opportunity to consider the net soil redistribution, of all erosion and deposition processes, at the catchment scale over large areas.

The national reconnaissance survey of soil erosion in Australia was performed at the hillslope scale (Loughran et al., 2004). That measurement survey was used to make predictions of ^{137}Cs-derived net (1950s–1990) soil redistribution every 5 km across Australia (Chappell et al., 2011b). In contrast to gross erosion estimates typical of plots, traps and erosion models (e.g. Universal Soil Loss Equation), the approach used here estimates the net outcome of all erosion and deposition processes within the period 1950s–1990 at each pixel across Australia. We used these estimates to identify locations at the catchment scale which were either net (1950s–1990) eroding or net depositing (including stable) for use in the SOC redistribution model (Eqs. 1 and 2). The summation of these estimates for different land management types, regions and ultimately across the continent of Australia provides an estimate of the net outcome for the terrestrial ecosystem.

2.3 Soil organic carbon stocks

The Australian soil visible–near-infrared spectroscopic database (Viscarra Rossel and Webster, 2012) was used to predict the soil organic carbon (SOC) and bulk density (Bd; $\mathrm{g\,cm^{-3}}$) of 4000 surface soil samples (0–10 cm) to derive the soil organic carbon density (SOC_{den}) map. The soil samples originated from CSIRO's National Soil Archive, the National Geochemical Survey of Australia, and other regional- and field-scale surveys of the Australien states. Thus, SOC_{den} ($\mathrm{t\,ha^{-1}}$) was calculated by

$$\mathrm{SOC_{den}} = \mathrm{OC} \times B_\mathrm{d} \times d, \tag{3}$$

where d is the depth (cm) from where the samples were taken. The SOC_{den} values were mapped by ordinary kriging on an approximate 5 km grid to coincide with the other maps.

2.4 Carbon enrichment by size selective erosion

The carbon enrichment factor is a major source of uncertainty in estimating SOC redistribution because there is considerable spatial and temporal variability in SOC enrichment of eroded sediment (Schiettecatte et al., 2008; Wang et al., 2010; Nadeu et al., 2011; Webb et al., 2012). Owens and Walling (1998; p. 193) suggested that a good approximation to the enrichment ratio is based on a comparison of the particle size composition of the eroded material with that of the topsoil. Chappell et al. (2013) recently produced a map of SOC enrichment in dust for Australia by assuming that SOC enrichment is proportional to the enrichment of soil fines, estimated from a physically based model of particle size selectivity:

$$P = \text{eroded SOC/SOC in soil (dimensionless)}. \tag{4}$$

Chappell et al. (2013) had spatial information on SOC but little information on eroded SOC (D), and so approximated P using P' as

$$P' = D_\mathrm{f}/S_\mathrm{f} \text{ (dimensionless)}, \tag{5}$$

where D_f is mass $< 22\,\mu$m divided by the mass $\leq 52\,\mu$m and S_f is the equivalent ratio for the soil surface: mass $<$ 22 µm/mass $<$ 52 µm. The ratio P' estimates the proportion of fine material in transport. We consider it a reasonable first approximation to assume that the enrichment ratio for wind erosion (based on particle size) is also a good first approximation for enrichment by wind and water processes. In the absence of any other data, we use that Australian wind erosion enrichment ratio to estimate P in the SOC redistribution model (Eqs. 1 and 2).

2.5 Estimation of net (1950s–1990) soil organic carbon redistribution

The SOC redistribution model requires the depositional locations to be linked to their sources. Unfortunately, it is difficult to precisely determine the area from which the deposited material has originated particularly as this may change over time and may be associated with different scales of erosion (proximal and distal sources). However, the net soil depositional zones are not associated with floodplains and alluvial flats in Australia (Fig. 1). This finding suggests that they are associated with the accumulation of dust. Although it is difficult to determine the source areas precisely, they are likely from within the Lake Eyre and Lake Frome basins, areas which

Table 1. Calculation of soil organic carbon net (1950s–1990) redistribution and its proportion for land use classes across Australia.

Class	Description	Area* (ha $\times 10^8$)	Mean SOC (%)	Mean enrichment ratio	Mean net soil redistribution (t soil ha^{-1} yr^{-1})	Total net SOC redistribution (t C yr^{-1} $\times 10^6$)
1	Conservation and natural environments	2.465	0.70	1.97	−0.213	−0.717
2	Production from relatively natural environments	4.193	0.76	2.01	−0.220	−1.472
3	Production from dryland agriculture and plantations	0.511	1.64	1.38	−1.479	−1.710
4	Production from irrigated agriculture and plantations	0.026	1.71	1.65	−1.515	−0.115
5	Intensive uses	0.014	1.80	1.23	−1.272	−0.043
1 and 2	"Rangeland"	6.658	0.74	1.99	−0.217	−2.189
3 and 4	"Agriculture"	0.536	1.64	1.39	−1.480	−1.821
1–5	Australia	7.209	0.81	1.95	−0.313	−4.057

* Using an equal area projection the area of a pixel is approximately 4.53 km $\times$ 4.87 km $\approx$ 22.03 km^2, equivalent to 2203 ha.

are well established as a dust source region in Australia's rangelands (McTainsh, 1989). The Lake Eyre basin, in the arid continental interior of Australia, contains considerably smaller amounts of SOC than the coastal regions. Consequently, to implement Eq. 2, we assumed that $OC_d = 0.74\,\%$ and $P_d = 1.99$ (values from "Rangeland" in Table 1) and estimated SOC redistribution.

To place these maps into context, at each location across Australia we divided the SOC net redistribution by the SOC stock (0–10 cm) and multiplied by 100 to obtain a percentage. This process determined the proportion of SOC yr^{-1} removed by the net (1950s–1990) outcome of all erosion and deposition processes.

2.6 Australian land use

The Bureau of Rural Sciences provides a series of land use maps of Australia. The agricultural land uses are based on the Australian Bureau of Statistics' agricultural censuses and surveys for the years mapped. The spatial distribution of agricultural land uses was determined using Advanced Very High Resolution Radiometer (AVHRR) satellite imagery with ground control data (Knapp et al., 2006). These data were supplied at a 0.01° grid size with geographical coordinates (GDA94). The summary map provides an integer grid which represents an aggregation of the original attribute table which defines the agricultural commodity group, irrigation status and land use according to the Australian Land Use and Management Classification (ALUMC), version 5 (Table 1).

We followed Chappell et al. (2011b) and used land use data from 1992/1993, which are closest in time to the national ^{137}Cs reconnaissance survey. These data were resampled to an approximately 5 km grid for compatibility with the other data used here. We then calculated the SOC net redistribution for each land use zone and compared their magnitudes to sectoral contributions of the national carbon account.

3 Results

Our map of ^{137}Cs-derived net (1950s–1990) soil redistribution shows that nearly 5 times more soil was lost from the predominantly coastal, cultivated regions than from the mainly uncultivated rangeland interior of Australia (Fig. 2a) (Chappell et al., 2011). The cultivated regions of Australia generally have larger amounts of SOC than the rangelands (Fig. 2b). The pattern of SOC enrichment (Fig. 2c) is complicated by the highly variable soil types, textures and particle size distributions. Nevertheless, the SOC enrichment map shows that the greater part of Australia has associated enrichment values of 1–1.5. Large SOC enrichment values (up to 5) are found in the rangeland interior and in northern Australia at locations where net soil erosion is small. However, there is also a large SOC enrichment area in the west of Western Australia (WA) in the Gascoyne–Pilbara region. In contrast, SOC depletion ($P < 1$), where eroded SOC is smaller than SOC in the parent soil, occurs in patches throughout Australia and most notably in the sandy soils of the Wheat Belt region of WA. These enrichment values are consistent with the review of enrichment ratios provided recently by Webb et al. (2012). The SOC net redistribution map (Fig. 2d) is therefore a product of these previous maps in accordance with Eqs. (1) and (2). The SOC net deposition component equals the net soil deposition plus the deposition enrichment factor of ~ 0.015 ($OC_e = 0.0074$ multiplied by $P_e = 1.99$; values from "Rangeland" in Table 1).

Although Australian rangelands contain smaller amounts of SOC than cultivated regions, their large area has the

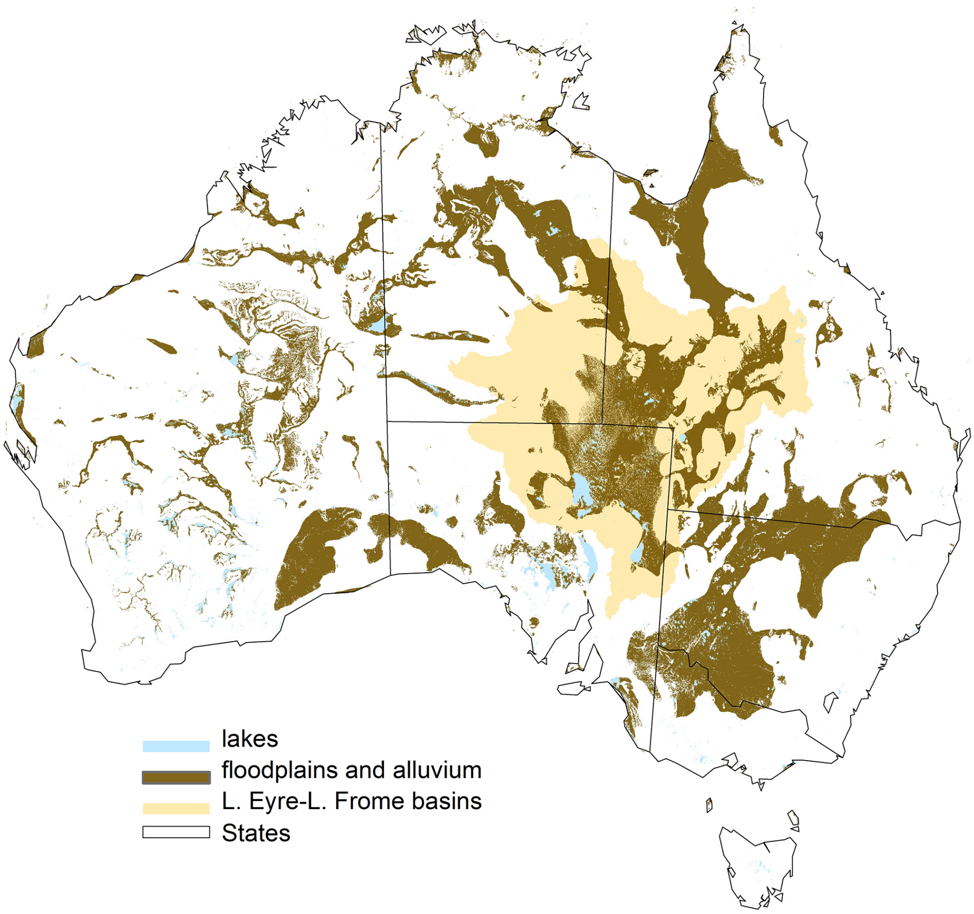

Figure 1. Map showing net deposition relative to floodplains and alluvial flats as mapped by Gallant and Dowling (2003).

potential to contribute considerably to SOC redistribution (Fig. 1e). Examining Australian SOC net redistribution on the basis of land use is instructive. Table 1 demonstrates that although rangeland regions (classes 1 and 2) contain only half as much topsoil SOC, their mean net soil redistribution is approximately 7 times smaller than that of the cultivated coastal regions (largely class 3). The cultivated regions contain the greatest amount of SOC and consequently produce areas with an "Agriculture" land use designation dominating the SOC net redistribution of Australia (Table 1).

The amounts of SOC net erosion appear substantial. Their proportions of SOC stock are all less than 1 % yr^{-1} because of the relatively large SOC stock (Fig. 2f). The spatial distribution of proportional loss matches that of net soil redistribution and SOC net redistribution. Cultivated regions, with the largest erosion and the largest SOC, have the greatest proportion of SOC stock removed. The total SOC net redistribution for Australia is on average -4.06×10^6 tC yr^{-1} (-4.06 Tg C yr^{-1}; Table 1) or approximately -1.63×10^8 tC (0.163 Pg C) for the period 1950s–1990. The Australian SOC stock (0–10 cm) amounts to 7.55×10^9 t C (7.55 Pg C), which suggests that, on average across Australia, approximately 2 % of SOC stock (0–10 cm) was removed from the land surface by soil erosion over this ca. 40-year period.

4 Discussion

4.1 Soil erosion and Australian land surface dynamics

An Australian survey of approximately 200 hillslope profiles showed net soil loss, which, aggregated across Australia, indicated that 60 % of sites had net soil losses greater than 1 t ha^{-1} yr^{-1} (Loughran et al., 2004). The regionalised net soil redistribution estimates of Chappell et al. (2011b; p. 17) used here provide more representative statistics than the original survey for soil redistribution across Australia. Only approximately 7 % of Australia had net soil losses of more than 1 t ha^{-1} yr^{-1}. Despite these findings, it may be argued by others that these regionalised estimates are unrepresentative of soil depositional areas associated with alluvial flats and floodplains. However, at least some of the original survey sites coincide with those types of geomorphological regions (Fig. 1).

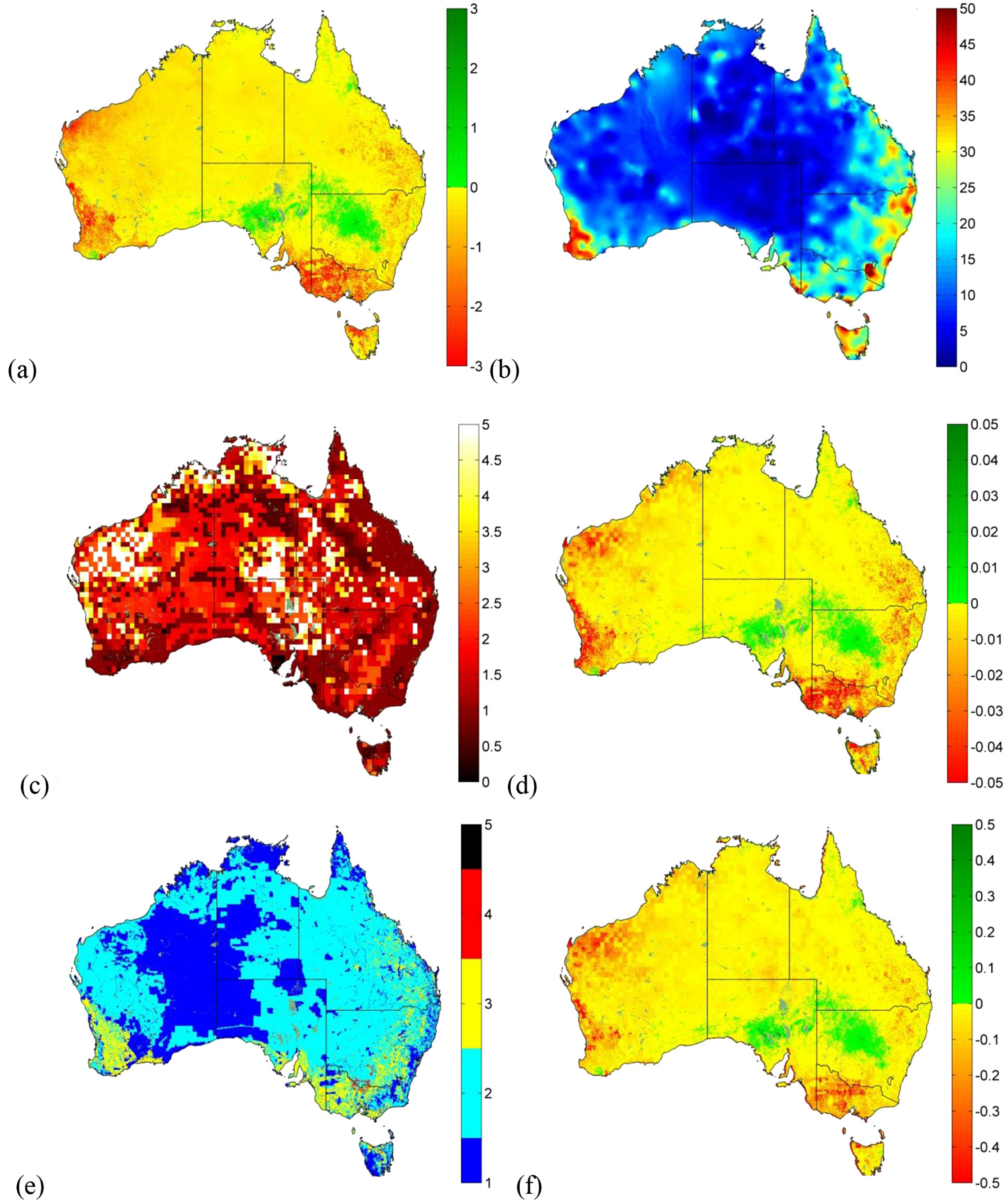

Figure 2. Maps for Australia showing **(a)** ^{137}Cs-derived median net (1950s–1990) soil redistribution (t soil ha^{-1} yr^{-1}). Positive values represent sites of net gain, while negative values are those of net loss. **(b)** Soil organic carbon (SOC) stocks (tC ha^{-1}; 0–10 cm). **(c)** Enrichment ratio at 50 km resolution. **(d)** SOC net redistribution (tC ha^{-1} yr^{-1}). **(e)** Land use (see Table 1). **(f)** SOC net redistribution as a proportion (% yr^{-1}) of SOC stocks.

It is reasonable to conceive of soil and SOC moving over time through a series of landscape stores before reaching a river network. In this conception, SOC residence times are unspecified, and yet it is widely accepted that old, weathered, low-relief landscapes (like Australia; Wasson et al., 1996; Lu et al., 2003) have small sediment delivery ratios and therefore small SOC net deposition (Roehl, 1963; Walling, 1983). In contrast to this general conception, our results show for the period 1950s–1990 that there was a SOC net loss for the majority of Australian hillslopes at the catchment scale (Chappell et al., 2011b). We contend that the general conception does not account for land surface dynamics influenced strongly by changes in land use and management, which are captured in our data. European settlement (from 1788) transformed Australia's environment with extensive clearing of native vegetation for agricultural production, primarily pastoralism and, to a lesser extent, cropping (McAlpine et al., 2009). Marx et al. (2014) used cores from a mire in the Snowy Mountains of Australia to reconstruct the past environment and used dust deposition as a proxy for soil erosion to show a rapid increase after 1879 associated with agricultural expansion 1880–1989 and the onset of agriculturally-induced wind erosion from the Murray–Darling Basin compounded by droughts (Federation, 1895–1903; 1911–1915, 1970s and 1980s) and the dust bowl era of the late 1930s and early 1940s.

Conservation agriculture has had a significant impact on soil erosion around the world (Montgomery, 2007). In Australia these practices and broader soil conservation measures were implemented in the 1980s and since then appear to have considerably reduced dust emission (Marx et al., 2014) and net (1990–2010) soil erosion (in SE Australia) despite considerable spatial variation remaining (Chappell et al., 2012). Notwithstanding this broad assessment, conservation agriculture (minimum/zero tillage) may not necessarily reduce soil erosion in some regions. This latter phase (1990–present) of agricultural stabilisation may well conform to the general conception of small sediment delivery ratios and slow reworking of sediments. However, it will likely take some time for the ecosystem to adjust to this new (dynamic) equilibrium and hence for SOC to develop the expected net SOC sink. SOC redistribution is a function of its residence times in the landscape, which must be contextualised for specific periods, land use change and management policies, etc. We believe our results provide a reasonable first approximation of the catchment-scale SOC net redistribution for Australia during the 1950s–1990.

4.2 Net soil redistribution (erosion and deposition) by all processes

To provide accurate and precise estimates of SOC net redistribution, it is essential to account for all erosion and deposition processes. The use of ^{137}Cs is evidently valuable in this respect. However, samples of ^{137}Cs must be obtained to represent the underlying population of soil redistribution processes and the scale at which they impact the carbon budget. Although it is logistically straightforward to conduct experiments at the field scale, estimates of SOC net redistribution are required at the catchment scale and larger when making regional/continental-scale assessments. It does not necessarily follow that investigations of SOC redistribution at the field scale (perhaps dominated by water erosion) are representative of the outcomes of the processes at the catchment scale (Doetterl et al., 2012a) by the combined effect of wind, water and tillage erosion. Our regionalised approach used here removes bias due to sampling (some fields and not others) and ensures that estimates at the catchment scale represent the small-scale variation.

Lal (2003; Table 11) estimated gross SOC erosion for Oceania at 20–40 Tg SOC yr^{-1}. Van Oost et al. (2007) estimated the gross SOC erosion by water and tillage for Oceania cropland and pastureland at 5.1 and 19.8 Tg SOC yr^{-1}, respectively. Doetterl et al. (2012) reduced their previous collaborative estimate of the gross SOC erosion by water and tillage for Oceania cropland and pastureland at 4.9 and 10.5 Tg SOC yr^{-1}, respectively. Dymond (2010) estimated that gross SOC flux for New Zealand was a sink of 3.1 Tg SOC yr^{-1} due primarily to soils regenerating from SOC erosion to the sea floor, where SOC was assumed permanently buried.

It is difficult to reconcile the differences between gross erosion and net (^{137}Cs-derived) erosion estimates (cf. Chappell et al., 2011b; p. 20). However, we expect our results to be considerably smaller than gross SOC erosion estimates because they include deposition within the landscape. It is therefore encouraging that our results are considerably smaller than the gross SOC estimates of Lal (2003). It is also encouraging that our net SOC erosion results for predominantly cropland (-1.82 Tg SOC yr^{-1}) and rangeland (-2.19 Tg SOC yr^{-1}) are smaller than the gross SOC erosion results of Van Oost et al. (2007). Their results for (and the subsequent reduction by Doetterl et al., 2012b) show an order of magnitude larger SOC loss from pasture regions in Oceania, which contrasts markedly with our results for Australia (1950s–1990). However, their global model estimates for New Zealand and Papua New Guinea explain about 86 % of the gross SOC erosion on pasturelands for Oceania, with Australia at -2.86 Tg C yr^{-1} (K. Van Oost, personal communication, 2014). This partition reveals consistency in our net SOC erosion estimates for Australian rangeland being smaller than that gross SOC erosion estimate.

For the Australian terrestrial carbon budget, Haverd et al. (2013) estimated the gross loss of carbon due to riverine and dust transport processes to be approximately 2.3 and 1 Tg SOC yr^{-1}, respectively (with 100 % uncertainty). Our physically based model estimates of gross SOC dust emission (2000–2011) was 1.6 Tg SOC yr^{-1} (Chappell et al., 2013). The net SOC dust flux is likely to be smaller for this period, which coincides with agricultural stabilisation (Marx et al., 2014). However, during the previous period of agricultural expansion and agriculturally-induced soil erosion, it was likely much larger (Marx et al., 2014), and could be approximated by our estimate of gross SOC dust emission. Subtracting that rate (1.6 Tg SOC yr^{-1}) from our estimate of total SOC net redistribution for Australia (-4.06 Tg C yr^{-1}) suggests that net SOC erosion by water was (1950s–1990) about 2.5 Tg SOC yr^{-1}.

5 Conclusions and implications for SOC cycling and SOC accounting

We find that net (1950s–1990) SOC redistribution for Australia is -4.06 Tg SOC yr^{-1}. Assuming that this material has been mineralised during transport by wind, water and tillage the net redistribution for Australia amounts to a loss of 14.87 Tg CO_2-equivalents yr^{-1} (using an elemental to molecular mass conversion factor of 44/12). We acknowledge that this assumption neglects the fate of SOC in the atmosphere, water courses and ultimately the oceans; its impact on gross primary productivity of forests; or its direct effect on radiative forcing. Biochemical reactions of SOC dust in the atmosphere and oceans may also counter the effects of SOC mineralisation that result in CO_2 production (Chappell et al., 2013). Even if the greater part of the SOC is not mineralised, a significant proportion is likely deposited in the marine environment and therefore not an atmospheric gain, but a terrestrial loss nonetheless. More work is required to elucidate the types and significance of these processes to determine the fate and in particular the impact of SOC dust.

It has been argued that the disturbance of SOC by erosion may accelerate its mineralisation and its conversion to CO_2, and support a hypothesis that SOC erosion is a source of CO_2 (Lal and Pimentel, 2008). Conversely, Harden et al. (1999) argued that erosion, transport and deposition of soil should act as a net carbon sink due to the dynamic replacement of SOC in eroding regions. Several studies have supported this latter hypothesis to suggest that soil erosion is acting as a biospheric net sink of CO_2 (cf. Van Oost et al., 2007). Following the logic of these hypotheses, it is difficult to avoid a conclusion here that during the period of agricultural expansion agriculturally-induced erosion net (1950s–1990) SOC erosion is a source of CO_2 for Australia.

Evidently, the losses of SOC due to soil erosion are of little consequence to the Australian carbon budget (Haverd et al., 2013). However, soil erosion and particularly the dynamics associated with the historical phases of agricultural expansion and stabilisation (Montgomery, 2007; Marx et al., 2014) are omitted from the land surface model used for the Australian carbon budget (CABLE / BIOS2). The implications are that substantial loss over time of organic-rich topsoil and changes to the soil albedo, soil temperature, moisture holding capacity and hydraulic properties have been omitted. That the CABLE / BIOS2 model has been shown to perform adequately without these fundamental dynamics in land–atmosphere interactions suggests to us that the tuning of the model is likely hiding the erosion impact. We believe that including a soil erosion component in this and other land surface models will likely provide a straightforward mechanism to demonstrate the impact of land use and management dynamics on land–atmosphere interactions.

Australian national carbon accounting provides the CO_2-equivalent emissions for national land use change, which represents total emissions from all carbon pools (below- and above-ground biomass, soil carbon and litter). Between 1988 and 1990 these emissions were 115–126 Tg CO_2-equivalents (Australian Greenhouse Office, 2005). Our results are approximately 12 % of those CO_2-equivalent emissions from all carbon pools in Australia. However, soil erosion is not explicitly included in Australian national SOC accounting, which renders estimates of CO_2 flux from soils highly uncertain. The inclusion of an erosion component may substantially reduce that uncertainty (Chappell et al., 2012, 2013; Sanderman and Chappell, 2012) and improve the accuracy for the reporting of GHG emissions.

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