Evaluation of Tolerable Erosion Rates and Time to Critical Topsoil Loss in Australia

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Cover photograph: Runoff after heavy rain leads to sheet and rill erosion, especially where vegetative cover is sparse. (Courtesy of Scott Wilkinson).

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Executive Summary

Soil is formed over such long time periods that it is considered to be a finite resource. In contrast, soil erosion in agricultural and grazing landscapes occurs at rates that may deplete the productive part of soil in a matter of decades. This report addresses the need to improve the monitoring and modelling of erosion processes around Australia as identified in a report (Leys et al. 2009) recently commissioned by the Australian Government Department of Agriculture, Fisheries and Forestry. This report’s goal is the identification of regions of Australia at risk of exceeding ‘tolerable’ rates of soil loss. The report provides the first quantitative evaluation of the amount of time land managers have to arrest soil erosion and establish a sustainable soil resource that balances erosion, deposition, and soil formation rates.

In an agricultural context, ‘tolerable’ soil erosion rate is the maximum level of soil erosion that will permit a high level of agricultural productivity to be sustained economically and indefinitely. It is common to use the soil production rate to set an ‘indefinitely sustainable’ or ‘tolerable’ erosion rate, i.e., where sustainable net soil loss from a specified region is defined as equal to the rate of soil production. In this report we used measured and modelled pre-European long-term denudation rates to estimate soil production rates. Assuming an average soil production rate of $1.5 \times 10^{-5}$ m yr$^{-1}$ and a bulk density of 1.3 t m$^{-3}$ uniformly over Australia gives a tolerable soil loss of 0.2 t ha$^{-1}$ yr$^{-1}$. In places where soil gain from dust deposition are important and equal to soil production, e.g., the western slopes of the Great Dividing Range, the tolerable soil loss becomes 0.4 t ha$^{-1}$ yr$^{-1}$.

An extension of the idea of tolerable soil erosion is the idea of the life span of soil, or “time to soil exhaustion” which can be quantified as the critical time, $T_c$, it takes to erode through a soil profile. We define it by:

$$T_c = \frac{S}{E - P}.$$

Evaluation of $T_c$ requires data for soil thickness ($S$), net soil erosion rate ($E$, given by the balance of erosion and deposition), and the soil production rate ($P$). Herein soil thickness is defined in terms of the soil A-horizon, since this is the part of the soil profile that is directly related to agricultural productivity. We evaluate inputs required for calculating $T_c$, namely maps of soil thickness, erosion rates, and soil production rates, then we estimate $T_c$, and identify catchments at risk of exceeding tolerable soil loss.

Two broad-scale methods for mapping soil erosion are used in this report. The first takes observed measures of soil erosion by water at the plot scale and generalises them using the Revised Universal Soil Loss Equation (RUSLE) approach so that the average rate of soil erosion by water on a hillslope can be predicted for all combinations of soil type, slope steepness, slope length, vegetation cover, and local rainfall conditions. A hillslope sediment delivery ratio of 10% is assumed for the transport of eroded hillslope soil into streams. The second uses a budget of the radionuclide caesium-137 ($^{137}$Cs) that labelled surface soils following the atmospheric nuclear weapons testing in the 1950s and 1960s. This method requires the use of geostatistics to interpolate between sparse measurements across the continent made during the $^{137}$Cs National Reconnaissance Survey in 1990.

The RUSLE modelling and the $^{137}$Cs-based technique give inconsistent results with around one order of magnitude difference observed in some regions. The two methods also identify different areas at risk of exceeding tolerable soil losses. RUSLE predictions suggest that northern Australia and coastal Queensland are most at risk. However, according to the $^{137}$Cs-based map WA and Victoria are most threatened. A map showing the composite of these results is shown below (Fig. i).

Estimates of the time to critical soil loss ($T_c$), defined here as the time for complete erosion of the soil A-horizon, are in the range 100-500 years for the most highly eroding areas. However, some studies have noted an exponential reduction in agricultural productivity with loss of topsoil. Thus loss of just a small fraction of the A-horizon (a few cm of soil) may in some regions lead to a significant decline (>25%) in soil agricultural productivity over time frames of less than 100 years.
Simple statistical comparisons of soil thickness datasets available through the Australian Soil Resource Information System suggest that these measurements have a 40% error (or 0.1 m in a 0.25-m thick soil). More research is required therefore to reduce the uncertainty in soil thickness and erosion rates. The discrepancies in predictions of erosion rates using the two methods (due to differences in exactly what is predicted and at what scale) can only be assessed with more work, including field verification.

Currently RUSLE modelling is the most credible method capable of providing national hillslope erosion assessment with a resolution fine enough for regional management. Important improvements in RUSLE erosion modelling can be obtained by improving land cover data using newly available high-resolution remote sensing imagery. However a shortcoming of the RUSLE method is that it does not account for rapid deposition of soil material moved over short distances and requires the introduction of a hillslope sediment delivery ratio (HSDR) to be comparable to sedimentation rates measured from surface water bodies. The HSDR is known to be spatially variable across landscapes but currently its pattern of variation has only been estimated for the Murray-Darling basin. Better estimates of HSDR pattern are required for the continent. Another shortcoming of the RUSLE method is that it only estimates water-borne erosion. An advantage of the $^{137}$Cs-based technique is that it accounts for net erosion by all surficial landscape processes (water, wind, and tillage erosion) directly. However there is a paucity of $^{137}$Cs-based erosion estimates for many important agricultural areas, including most of Queensland and NSW, northern WA and western SA. These areas require further sampling for $^{137}$Cs. There is a limited time frame for this work since the $^{137}$Cs technique will continue to lose sensitivity over the coming decades as $^{137}$Cs decays.

Fig. i. Map showing all areas identified as having a soil A-horizon life span of < 500 years. Up to 100,000 km$^2$ could be at risk. This map of time to critical A-horizon loss uses soil thickness data from the 2009 and 2001 Australian Soil Resource Information System. The erosion data are a composite of RUSLE predictions scaled by a 10% HSDR and $^{137}$Cs-based maps. A soil production rate of $1.5 \times 10^{-5}$ m yr$^{-1}$ has been assumed over the whole continent.
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1. Introduction

Soil is a central component of ecosystems; it is the basic resource for agriculture and it is involved in many other ecosystems services such as water filtration, waste decomposition, water and nutrient storage, land-air gaseous interchanges, and support for biodiversity. Soil loss can reduce agricultural productivity because it decreases the potential storage volume of water for crops, reduces the total nutrient supply, and threatens soil structure and tilth (Montgomery 2007; Verheijen et al. 2009; Li et al. 2009). While the effect of these losses on productivity and soil depth is highly variable and often difficult to assess (Larson et al., 1983; Hopkins et al., 2001; Warren, 2007) there is evidence that that crop productivity decreases exponentially with topsoil loss (Stocking, 2003). In Australia where much of the topsoil layer (A-horizon) is notoriously thin, crop productivity is considered more sensitive to erosion than Europe and North America (Biggelaar et al., 2003). The decline in soil organic matter (McGarity et al., 1998) and loss of sustainable vegetation cover (Karfs et al., 2009) are other observed effects of soil loss in specific regions of Australia.

Since pre-European time, rates of soil loss in Australia appear to have increased significantly from land clearing and land use change (Lu et al. 2001, 2003; Neil et al. 2002; McCulloch et al. 2003; Gale and Haworth 2005; Dosseto et al. 2008; Hughes et al. 2009). Soil formation rates in Australia are thought to be so slow that they can be considered negligible on the human time-scale (Edwards 1988; Pillans 1997), therefore it follows that the current soil cover is all that will be available for use for generations to come. Thus soil conservation and minimization of soil loss in intensively used landscapes is seen as an important land management endeavour to ensure the long-term sustainability of Australian landscapes.

Soil loss is the result of removal by gravity, water, ice or wind as transport agents. Indeed erosion is defined as: “(i) The wearing away of the land surface by rain or irrigation water, wind, ice, or other natural or anthropogenic agents that abrade, detach and remove geologic parent material or soil from one point on the earth’s surface and deposit it elsewhere, including such processes as gravitational creep and so-called tillage erosion; (ii) The detachment and movement of soil or rock by water, wind, ice, or gravity” (https://www.soils.org/publications/soils-glossary#). Soil detached by erosion becomes sediment. Soil loss is notoriously hard to measure: plot-based estimates of soil loss due to hillslope erosion are about an order magnitude higher than sediment yield gauged from catchments (Lu et al. 2006; Visser et al. 2007), presumably due to deposition of the eroded soil at various locations along the river network. Thus it is difficult to scale up from field-based measurements, and conversely it is equally difficult to scale down using whole-of-catchment sediment flux estimates. Thus most large-scale estimates of soil loss are estimated by modelling. Merritt et al. (2003) provide a detailed review of existing water-borne erosion and sediment transport models. There are notable differences in their representation of the various erosion processes (e.g. hillslope, gully, in-stream), routing and transport processes, and the deposition process. The types of inputs these models need are also varied but include: topographic attributes (e.g. slope), climatic inputs (rainfall or potential evapotranspiration), land use, soil characteristics, vegetation cover, and management actions.

The empirical Universal Soil Loss Equation (Wischmeier and Smith 1978) has been the most widely used model worldwide. It predicts sheet and rill erosion by water and is guided by many years of plot-based (~20 m length scale) experimental data (Nearing et al. 2000). In the USA it has been replaced by the more process-based Water Erosion Prediction Project (WEPP) (Nearing et al. 1989). In Europe, the new PESERA (Pan-European Soil Erosion Risk Assessment) model is a physically based and spatially distributed model combining the effect of topography, climate and soil into a single integrated forecast of runoff and soil erosion on a 1-km² grid (Kirkby et al. 2004). In England and Wales, PESERA is being validated by a caesium-137 (137Cs) survey (Des Walling, personal communication). In Australia, the SedNet model developed for the first National Land and Water Resources Audit (NLWRA) by Prosser

As a result of the NLWRA, there has been significant investment in controlling soil loss by water-borne erosion in some Australian Natural Resources Management regions. The effectiveness of the control measures needs to be assessed and their impact on erosion monitored. A recent report undertaken by the Australian Government Department of Agriculture, Fisheries and Forestry (Leys et al., 2009) has described the need to monitor soil resources over the long term in order to assess soil condition, prioritise government investments, and demonstrate the causal links between improved land management practices and soil erosion. The report identified priority short term projects required to provide base level erosion information against which long term changes can be measured, and included the need for an integrative assessment of current wind and water erosion rates across rural Australia. Leys et al. (2009) also propose the construction of a map of the “time to soil exhaustion” for the eroding hillslopes of Australia. It is the aim of this report to address these needs.

To estimate rates of hillslope topsoil depletion requires an accurate assessment of long-term average soil erosion under natural conditions. By default this erosion rate is often conservatively defined as an ‘acceptable’ or ‘tolerable’ level of erosion since this rate can represent that which is naturally sustainable. Where ‘tolerable soil erosion’ is exceeded in the landscape the concept of ‘soil life span’ becomes a concern. In addressing the designated priorities of Leys et al. (2009) our ultimate goal in this report is the identification of regions of Australia at risk of exceeding tolerable soil loss, and we start by assessing the adequacy of the basic inputs required for this determination. This initially includes an assessment of tolerable erosion rates for Australia, followed by the review and collation of available estimates of soil depth, soil formation rates and erosion rates for Australia. Erosion rate estimates include currently available model outputs, as well as a map generated by a new approach to erosion analysis using 137Cs-derived estimates. As part of our focus on the agricultural perspective of soil life span, the relationship between soil loss and agricultural productivity is considered.

1.1. Defining the concept of a ‘tolerable level of soil erosion’

Wischmeier and Smith (1978) developed the Universal Soil Loss Equation (USLE) for tackling sustainable agriculture in the USA. They defined a ‘tolerable’ soil erosion rate as “the maximum level of soil erosion that will permit a high level of crop productivity to be sustained economically and indefinitely”. Importantly, they note that the definition would be different if the objective was water quality rather than crop productivity. However historically, notions of ‘tolerable soil erosion’ have focused on the impact of erosion and soil loss on agricultural uses (Table 1), with the sustained high level of agricultural productivity over the long term being the main criterion, although the length of time involved is not clearly defined. Only recently has the impact of erosion on other ecosystem services been recognised (Verheijen et al. 2009).

Many of the definitions in Table 1 propose a direct relationship between tolerable soil erosion ($T$) and the rate of soil production. Li et al. (2009) propose multiple $T$-values: $T_1$ defines tolerable soil loss as equal to soil production rate; $T_2$ reflects the soil productivity; and $T_3$ reflects the impact on water and aquatic environments. This proposition of multiple $T$-values recognizes that $T$-values can vary depending on the endpoint or environmental objective.

---

1 Most soil eroded and transported by rainfall runoff is re-deposited quickly after removal, a short distance from its origin and the fraction of sediment delivered to streams is a power function of catchment area; the exponent used by Wischmeier and Smith (1978) was -0.2.
In the USA, T-values have been assigned for all soils by the U.S. Department of Agriculture; these are based on soil depth, soil formation rate, prior erosion, and productivity and range from 4.5 to 11.2 t ha\(^{-1}\) yr\(^{-1}\) (McCormack et al. 1982), equivalent to 0.4–0.9 mm yr\(^{-1}\) of erosion (assuming a soil bulk density of 1.2 t m\(^{-3}\)). Some researchers have expressed concern that T values themselves are set substantially higher than soil production rates because of political and economic considerations, while others contend that only highly erodible land is eroding faster than T values (Montgomery 2007). Nevertheless, soil conservation measures and incentives, in particular the adoption of no-till agriculture, helped reduce the total erosion from U.S. cropland from 3.4 billion tons in 1982 to 2.0 billion tons in 1997 (Montgomery 2007).

In Australia, there is little information on soil loss tolerance. An upper limit for T of 1 t ha\(^{-1}\) yr\(^{-1}\) has been proposed by Edwards and Zierholz (2000) based on soil production rates, although the authors suggest a value of less than 0.5 t ha\(^{-1}\) yr\(^{-1}\) is more likely. In this Report, we have followed the convention of Li et al. (2009) and others in Table 1 for our definition of tolerable soil erosion; i.e. T1 is equal to the rate of soil production, so that topsoil thickness is maintained and agricultural productivity is sustained indefinitely. With an agricultural productivity endpoint, the focus is on soil loss from hillslopes.

1.2. Time to critical soil loss

An extension of the idea of ‘tolerable soil erosion’ is the idea of ‘life span’ of soil. This is the number of years it will take, at current soil formation/erosion rates, for a soil to reach its finite point; i.e. the minimum soil depth required before it becomes economically unsustainable to maintain the current land use (Stocking and Pain 1983, cited by Verheijen et al. 2009). For commercial farming, the finite point has been defined at when yields fall to 75% below the maximum possible (Morgan 1987, cited by Verheijen et al. 2009). Given that ploughed fields erode substantially faster than rates of soil production and natural soil erosion, a limiting life span of an agricultural civilization can be estimated by the time needed for conventional agriculture to erode through the native stock of topsoil (Montgomery 2007). The simple notion of the “life span of agricultural soils predicts reasonably well the historical pattern of a 500- to several-thousand-year life span for major civilizations around the world, supporting the argument that it was not the axe that cleared forests but the plow that followed that undermined many ancient societies” (Montgomery 2007).

We define the critical time, Tc (years), as the time to erode through an entire soil profile. This can be estimated from:

\[
T_c = \frac{S}{(E - P)} \quad (\text{Eq. 1})
\]

where \(S\) is the initial soil thickness (m), \(E\) is the net soil erosion rate given by the difference between erosion and deposition (m yr\(^{-1}\)), and \(P\) is the soil production rate (m yr\(^{-1}\)) (modified from Montgomery 2007). For reasons related to agricultural productivity (discussed further in section 2.1 below) we define soil thickness as the soil A-horizon thickness. Using this formula, and generally assuming soil production rate as negligible, many have warned of impending critical soil loss or soil exhaustion, e.g. Montgomery (2007) in the quote above.

---

\(^2\) To convert erosion rate estimates from t ha\(^{-1}\) yr\(^{-1}\) to mm yr\(^{-1}\) or vice versa, soil bulk density data are required.
<table>
<thead>
<tr>
<th>Definition/interpretation</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>The maximum level of soil erosion that will permit a high level of crop productivity to be sustained economically and indefinitely</td>
<td>Wischmeier and Smith (1978)</td>
</tr>
<tr>
<td>The maximum level of soil erosion that will permit a relatively high level of soil organic matter to be sustained economically and indefinitely</td>
<td>Alexander (1988) after Wischmeier and Smith (1978)</td>
</tr>
<tr>
<td>Soil loss balanced by soil formation through weathering of rocks</td>
<td>Roose (1996)</td>
</tr>
<tr>
<td>Erosion that does not lead to any appreciable reduction in soil productivity</td>
<td>Roose (1996)</td>
</tr>
<tr>
<td>The maximum rate of soil erosion that permits an optimum level of crop productivity to be sustained economically and indefinitely</td>
<td>ISSS (1996) after Wischmeier and Smith (1978)</td>
</tr>
<tr>
<td>The average annual soil loss a given soil type may experience and still maintain its productivity over an extended period of time (permissible soil loss)</td>
<td>Kok et al. (1995)</td>
</tr>
<tr>
<td>The maximum permissible rate of erosion at which soil fertility can be maintained over 20-25 years</td>
<td>Morgan (2005)</td>
</tr>
<tr>
<td>(i) The maximum average annual soil loss that will allow continuous cropping and maintain soil productivity without requiring additional management inputs</td>
<td>SSSA (2001)</td>
</tr>
<tr>
<td>(ii) The maximum soil erosion loss that is offset by the theoretical maximum rate of soil development which will maintain an equilibrium between soil losses and gains</td>
<td>SSSA (2001)</td>
</tr>
<tr>
<td>Rate of soil erosion is not larger than the rate of soil production (acceptable rates of soil erosion)</td>
<td>Boardman and Poesen (2006)</td>
</tr>
<tr>
<td>Any mean annual cumulative (all erosion types combined) soil erosion rate at which a deterioration or loss of one or more soil functions (habitat, agricultural production, engineering, environmental regulation, archival information) does not occur. Soil formation rates are proposed as a basis for establishing tolerable soil erosion.</td>
<td>Verheijen et al. (2009)</td>
</tr>
<tr>
<td>Multiple T-values: T1 defines tolerable soil loss as equal to soil production rate; T2 reflects the soil productivity; and T3 reflects the impact on water and aquatic environments.</td>
<td>Li et al. (2009)</td>
</tr>
</tbody>
</table>
1.3. Identifying areas at risk of exceeding rates of tolerable soil erosion

Erosion studies mainly focus on uplands, often excluding lowland depositional areas where annual flooding replenishes mineral soils by alluvial deposition, or where hillslope sediments get deposited as colluvium\(^3\) that contributes to vertical soil accretion.

In a broad sense, hilly landscapes can be characterized as transport- or weathering-limited (Carson and Kirkby 1972). Assuming steady-state balance or equilibrium conditions, landscapes are transport-limited where rates of erosion are less than soil formation. Where erosion exceeds soil formation landscapes are weathering- or supply-limited. It follows therefore that soils should be thick in transport-limited landscapes and thin in weathering-limited situations. It also follows that thick soils should correspond to low erosion zones and thin soils to high erosion zones. Therefore, as a first approximation, we could assume that areas at risk of exceeding rates of tolerable soil erosion are those where thin soils occur and we could identify these simply by looking at a map of soil thickness. A map-to-map comparison of hillslope erosion and soil thickness could then be used to indicate areas at risk of exceeding rates of tolerable soil erosion, i.e. where erosion rates are high and soils are thin. Together with mapped estimates of soil erosion rates and estimates of soil production rates, a map of soil thickness can be used to calculate and map approximate time to topsoil loss and reduction in overall thickness using Eq. 1.

2. An assessment of the parameters required for calculating tolerable soil erosion and time to critical soil loss

2.1. Soil thickness (\(S\))

If tolerable soil erosion is to be quantified with agricultural productivity as the primary consideration or endpoint, then the implementation of Eq. 1 requires reliable information on A-horizon thickness. This is because the A-horizon is the uppermost zone of soil where most nutrients (including soil organic matter) are stored and cycled. It is also the most important soil zone in agricultural soils because crops and grasses are usually shallow-rooted. Globally, across all ecosystems, approximately 30%, 50%, and 75% of roots occur in the top 10 cm, 20 cm, and 40 cm, respectively (Jackson et al. 1996), and nutrients strongly cycled by plants, such as P and K, are more concentrated in the upper 20 cm of the topsoil (Jobbagy and Jackson 2001).

Nationwide coverage of A-horizon thickness exists only at 1-km and at 50-km resolution, generated during the National Land Water Resources Audit of Australia (NLWRA) as part of the 2001 Australian Soil Resource Information System (ASRIS) project. These were produced using “look-up tabled” (LUT) values linked to soil map units, many of which are from the digital Atlas of Soils in central Australia (Carlile et al. 2001). Bulk density estimates over the continent are also available at these resolutions and have been produced using the same approach, linking LUT values to soil map units. Bulk density is required to convert areal erosion rates in t ha\(^{-1}\) yr\(^{-1}\) to mass specific rates in m yr\(^{-1}\).

The 2001 ASRIS point database contains \(~100,000\) points with measurements of A-horizon and B-horizon thickness. Models of soil A-horizon thickness as a binary variable (thin, \(< 0.25\) m; thick \(\geq 0.25\) m) were produced with a relatively good level of accuracy at grid resolution of 250 m over agricultural catchments of Australia\(^4\) using this point database (Henderson et al. 2001). Since 2001 more detailed soil surveys with associated LUT have been incorporated into updates of ASRIS (referred to as 2009 ASRIS herein), the extent of this mapping is

\(^{3}\) Depositional areas can be identified using the Multi-resolution Valley Bottom Flat (MrVBF) surface of Gallant and Dowling (2003) to mask out these areas of deposition.

\(^{4}\) These were made available for public use at 1-km resolution through the NLWRA digital Atlas of Australian Natural Resources.
shown in Fig. 1 (refer to http://www.asris.csiro.au/methods.html for detailed explanation of information available currently in ASRIS).

**Fig. 1.** Current 2009 ASRIS data for A-horizon thickness. Unmapped areas are blank. Level 5 is mapping of land systems or land units with similar relief, modal slope, stream pattern, toposequences, local climate, lithology, soil association(s), vegetation type or sequence; its characteristic diameter is ~600 m. Level 4 shows land district soil mapping grouping geomorphologically related land systems and therefore is coarser-scaled than Level 5, with a characteristic diameter of ~2 km.

There are clearly large gaps in the most detailed coverages from the 2001 ASRIS and the current soil thickness maps over Australia (Fig. 2). However, except in New South Wales, most of the areas predicted by Lu et al. (2003) to have high annual hillslope erosion rates have some data, albeit at variable scales.

There are some discrepancies in the estimated soil thickness between data sources (Fig. 1 and 2, Table 2), which brings the quality and reliability of the data into question. In the 2009 ASRIS data (Fig. 1), there are mismatches across state boundaries, between Queensland and South Australia, South Australia and Victoria, and West Australia and the Northern Territory. Within Queensland, there are edge mismatches in estimated soil thickness across soil survey boundaries. The relative accuracy of the two datasets remains to be determined.

In order to assess the consistency of two data sets a comparison is made with using ~100,000 points where A-horizon thickness was recorded at the 1-km and 50-km resolution. This is done using summary statistics shown in Table 2. The medians for A-horizon thickness are close (within 0.05-0.10 m of each other) for all the datasets even though the range of the distributions can vary widely. The discrepancies between the datasets at specific points are more evident in pairwise plots (Fig. 3). The 2009 ASRIS data for the whole continent appear to have some anomalously high values (maximum thickness is 10.9 m), however the fit is good where soils were predicted to be thin by modelling (2nd and 3rd row, last column in Fig. 3).
Fig. 2. A-horizon thickness estimates (A) using the 2001 ASRIS soil survey data. (B) modelled using measurements at points and environmental variables as predictors as explained in Henderson et al. (2001); this covers only intensively used agricultural areas. Thin is < 0.25 m, thick is ≥ 0.25 m.
Table 2. Comparison of soil thickness data.

<table>
<thead>
<tr>
<th>Dataset</th>
<th>minimum (m)</th>
<th>1st quartile (m)</th>
<th>median (m)</th>
<th>mean (m)</th>
<th>3rd quartile (m)</th>
<th>maximum (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>2001 ASRIS point database</td>
<td>0.02</td>
<td>0.08</td>
<td>0.20</td>
<td>0.25</td>
<td>0.34</td>
<td>2.50</td>
</tr>
<tr>
<td>2001 ASRIS 1-km</td>
<td>0.08</td>
<td>0.18</td>
<td>0.25</td>
<td>0.27</td>
<td>0.34</td>
<td>1.10</td>
</tr>
<tr>
<td>2001 ASRIS 50-km</td>
<td>0.08</td>
<td>0.20</td>
<td>0.25</td>
<td>0.27</td>
<td>0.37</td>
<td>0.75</td>
</tr>
<tr>
<td>2009 ASRIS</td>
<td>0.10</td>
<td>0.20</td>
<td>0.30</td>
<td>0.37</td>
<td>0.48</td>
<td>10.9</td>
</tr>
</tbody>
</table>

Fig. 3. A matrix of scatter plots of A-horizon thickness data from various datasets, compared pairwise. Correlation coefficients between datasets are shown, in font size proportional to their value, in the lower left panel. Variable (in this case dataset) names for the axes are along the diagonal and refer to the corresponding row and column; axes are labelled on the corresponding row and column. The third row is the same 2009 ASRIS data as in the second row but with values >2 m omitted. The bottom row and last column are from the modelled 2001 ASRIS surface where thin=1 is < 0.25 m, and thick=2 is ≥0.25 m.
2.2. Soil production rates ($P$)

Soil is the surficial material product of the chemically and physically weathered underlying rock, called saprolite, that serves as a medium for plant growth. Factors affecting the formation and layering of soil on the Earth’s surface include weathering at the soil-saprolite interface, erosion, and bioturbation (e.g., Paton et al. 1995; Heimsath et al. 2000). Soil thickness represents the balance between soil production and removal by erosion. In theory, at equilibrium or when a steady-state balance exists in a landscape, the net rate of erosion is equal to the rate of soil formation (Carson and Kirkby 1972; Heimsath et al. 2000; Montgomery 2007; Dosseto et al. 2008). Soil production rates only equal net erosion rates if local soil thickness is roughly constant over time. Thus, over geological time scales, the existence of a soil mantle on convex hillslopes can be taken as an indication that the landscape is in steady state, i.e., soil production equals soil loss (Heimsath et al. 2009). Assuming steady state, long-term denudation rates (over thousands to millions of years) can be used to estimate soil production rates and vice versa.

To be in line with world-wide ranges quoted by Montgomery (2007), in Australia the long-term erosion rate should range between <0.1 to 10 μm yr$^{-1}$ on the gently sloping lowland landscapes of the ancient continental craton, and from 1 to 1000 μm yr$^{-1}$ on moderate gradient hillslopes of soil-mantled terrain. Rates observed in steep tectonically active alpine topography, >10$^4$ μm yr$^{-1}$ (>10 mm yr$^{-1}$, Montgomery 2007) would be exceptional in Australia.

2.2.1. Long-term denudation rates

Long-term denudation rates can be estimated using a variety of techniques, e.g.:

1) morpho-stratigraphic, e.g. plateau lowering and valley incision below valley-filling basalt flows (Gale 1992). Here the different stratigraphic layers need to be of known age.

2) concentrations of in situ produced cosmogenic $^{26}$Al ($t_{1/2}=0.705\times10^6$ yr) and $^{10}$Be ($t_{1/2}=1.5\times10^6$ yr) in quartz-bearing surface rocks that have been exposed to cosmic ray radiation (Belton et al. 2004). Cosmic ray production of nuclides is offset by radioactive decay of the nuclides in surfaces that are not eroding and by removal of the target material in eroding surfaces. At secular equilibrium, production balances loss via erosion and decay, such that saturation concentrations of cosmogenic nuclides are effectively determined by the magnitude of the erosion rate (Heimsath et al. 2009).

3) apatite fission track thermochronology, which relies on changes in the Earth’s temperature profile to estimate surface lowering over 10$^6$ to 10$^8$ Ma time scales (Kohn et al. 2002).

Gale (1992) synthesized 43 estimates of denudation rates around Australia and concluded that denudation rates in the Cenozoic (last 65 Myr) have been between less than 2 μm yr$^{-1}$ over central Australia and between 2 and 10 μm yr$^{-1}$ along portions of coastal Australia associated with river rejuvenation (Fig. 4). Although these estimates involve several assumptions, Gale (1992) believed that the potential errors consequent upon these assumptions are not sufficient to alter the calculated rates by a factor of more than two or three. In the Sydney Basin, on the basis of 15 previously published measurements obtained by a variety of techniques, Tomkins et al. (2007) estimated a long-term denudation rate of 21 ±7 μm yr$^{-1}$, within a factor of three of the estimate by Gale (1992).

Using a large dataset (~1,700 data points across Australia), Kohn et al. (2002) established that the continent-wide average over the most recent geological period (the Pleistocene, last ~15 Myr) ranged from 4 to 10 μm yr$^{-1}$. Their regional estimate for south-eastern Australia concurs with that of Tomkins et al. (2007) for the Sydney Basin.
Bierman and Caffee (2002) calculated erosion rates of 0.2 to 7 μm yr$^{-1}$ at 61 inselbergs across the N-S breadth of central Australia. Belton et al. (2004) estimated rates in the north-central Australian craton ranging from 0.3 to 4 μm yr$^{-1}$ depending on lithology. Both these estimates mainly encompassed arid and semi-arid regions of the continent, and the rates are generally lower (half to one third) compared to other estimates. Given the reported correlation between rainfall and long-term erosion (e.g. Bierman and Caffee (2002)) the arid environment may account for these lower erosion rates.

2.2.2. Soil production rate estimates

Early Australian studies reported soil production rates of 2-30 μm yr$^{-1}$ (Edwards and Zierholz 2000). Pillans (1997), using K/Ar dating and paleomagnetism on red soils developed on basalt in Queensland, estimates a soil formation rate of 0.3 μm yr$^{-1}$. However, this is likely to be an under-estimate because it does account for the neighbouring formation of authigenic clay-rich Vertosols from the weathering of the same basalt flow: the total volume of rock weathered to produce the red soil is probably greater than the volume of red soil observed.

Fig. 4. Long-term Cenozoic denudation rates over physiographic regions of Australia. Inland areas have ancient surfaces, low relief, and low denudation rates, whereas coastal areas have younger landscapes, higher relief and higher denudation rates (from Gale 1992, with permission). Note units of m Ma$^{-1}$ are equivalent to microns per yr.
In the last decade, the use of the decay rate of atmospheric fallout of cosmogenic nuclides such as $^{10}$Be and its movement down-profile through soil (e.g. Fifield et al. 2009), of in situ produced cosmogenic nuclides (e.g. Heimsath et al. 2000), and of U-series isotopes (e.g. Dosseto et al. 2008) as dating tools has enabled research on soil formation rates and soil production functions. Using $^{10}$Be Fifield et al. (2009) reported a soil formation rate of 10-27 $\mu$m yr$^{-1}$ in north-west Australia, a rate 4-10 times higher than at a site in southeastern Australia. Stone and Vasconcelos (2000) also noted a large variation in long-term erosion rates determined by cosmogenic isotopes, these variations being related to rainfall and lithology.

Using in situ produced cosmogenic nuclides and optically stimulated luminescence in quartz grains, Wilkinson and Humphreys (2005) estimate a soil production rate of 0.015 mm yr$^{-1}$ on sandstone in the Blue Mountains. Using in situ produced cosmogenic $^{10}$Be and $^{26}$Al, Heimsath et al. (2000) estimate a minimum rate of 0.007 mm yr$^{-1}$ at the regolith/soil interface of soil 100-cm thick and a maximum rate of 0.053 mm yr$^{-1}$ on bare granodiorite in NSW. These studies in NSW have been compared to more traditional estimates of weathering rates based on geochemical mass balances (e.g. Green et al. 2006; Yoo et al. 2007; Burke et al. 2009) and of soil thickness estimates based on geomorphological relationships (e.g. Wilkinson and Humphreys 2006).

2.2.3. The relationship between soil thickness and soil production

The extent of the Australian soil formation studies is limited but where they exist they have also been used to evaluate/validate the ASRIS soil thickness and bulk density estimates (Table 3). Generally the ASRIS data at 1-km and at 50-km resolution is within <40% of the reported soil thickness in the detailed field studies on soil formation.

Soil A-horizons are thickest (Fig. 2b) where the long-term denudation/soil production rates are thought to be highest, along the continental margins (Fig. 4). The pattern of soil A-horizon thickness is thus consistent with theoretical expectations: In hilly landscapes at steady state, soil thickness depends on the balance between soil production and net erosion. In other words, balances between soil production and transport determine whether soil exists on any given landscape, as well as how thick it might be. Soils should be thick in transport-limited landscapes where erosion is less than soil formation, and thin in weathering-limited situations, where erosion exceeds soil formation.

The relationship between thickness and soil production in southeastern NSW, where Heimsath et al. (2000) estimate a minimum rate of 7 $\mu$m yr$^{-1}$ at the regolith/soil interface of soil 100-cm deep and a maximum rate of 53 $\mu$m yr$^{-1}$ on bare granodiorite, shows an inverse relationship exists between soil formation rates and soil depth. Thus an exponential soil production function is indicated. However, in Arnhem Land, northern Australia, Heimsath et al. (2009) quantify a well-defined 'humped' soil-production function. Here soil production rates peak under a soil thickness of about 35 cm and no soil thicknesses between exposed bedrock and this thickness are observed, in contrast to exponential soil functions reported for the Bega valley in NSW. In the Blue Mountains, NSW, Wilkinson and Humphreys (2005, 2006) also argued for a humped soil production function. Thus both the exponential and the humped soil production functions are valid potential models of soil formation albeit perhaps under different climatic and tectonic conditions (Wilkinson and Humphreys 2005; Heimsath et al. 2009).

If soil production rates are maximal under a thick soil cover rather than at an exposed bedrock surface, as they are under an exponential soil production function model (Heimsath et al. 2000), geological denudation rates estimated from inselbergs, e.g. as by Bierman and Caffee (2002), would provide a lower limit for soil formation rates.

The soil production/weathering rates determined in these studies generally lies in the range 4-60 $\mu$m yr$^{-1}$. The mean of all soil production rate estimates is 17 ±5 $\mu$m yr$^{-1}$. If the long-term denudation rates of section 2.2.1 are included the mean becomes 15 ±4 $\mu$m yr$^{-1}$. 
Table 3. Estimates of soil formation rates and soil thickness data for those sites. Erosion rate estimates determined from Lu et al. (2003) are also shown.

<table>
<thead>
<tr>
<th>Study location</th>
<th>Author(s) &amp; date</th>
<th>Estimate &amp; Method</th>
<th>Total soil depth (solum) observed</th>
<th>2001 ASRIS solum thickness (cm) &amp; Bulk Density (Mg m(^{-3}))</th>
<th>Soil formation rate determined m Myr(^{-1}) = m mkyr(^{-1}) = (\mu)m yr(^{-1})</th>
<th>Modelled pre-European erosion rate of (Lu et al. 2003) scaled by 10% HSDR ((\mu)m yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Davenport Range Central Australia</td>
<td>Belton et al., 2004</td>
<td>Long-term denudation rate: Fission track thermochronology and (^{26})Al and (^{10})Be</td>
<td>0.3 to 4 (\mu)m yr(^{-1})</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Australia-wide</td>
<td>Gale (1992)</td>
<td>Long-term denudation rate:</td>
<td>&lt;10 (\mu)m yr(^{-1})</td>
<td>20</td>
<td></td>
<td></td>
</tr>
<tr>
<td>N-S transect of Australia</td>
<td>Bierman and Caffee (2002)</td>
<td>Long-term denudation rate: (^{26})Al and (^{10})Be</td>
<td>0.2 to 7 (\mu)m yr(^{-1})</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sydney Basin</td>
<td>Tomkins et al. (2007)</td>
<td></td>
<td>15 to 28 (\mu)m yr(^{-1})</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nulla and Sturgeon province basalt flows, Qld 144.4667, -20.25</td>
<td>Pillans (1997)</td>
<td>soil depth chronofunction on red soils with K/Ar and (^{14})C dating,</td>
<td>60 cm</td>
<td>45 cm 1.38 Mg m(^{-3})</td>
<td>0.3 (\mu)m yr(^{-1})</td>
<td>20</td>
</tr>
<tr>
<td>Newnes plateau, Blue Mts, NSW, sandstone 150.2667, -33.3</td>
<td>Wilkinson (2005) and derived publications</td>
<td>geomorphology, stratigraphy, soil-vegetation relationships, accelerator mass spectrometry for (^{10})Be</td>
<td>mean soil depth under heath= 28 cm, under forest=66 cm</td>
<td>60 cm 1.3 Mg m(^{-3})</td>
<td>15 (\mu)m yr(^{-1})</td>
<td>1.4</td>
</tr>
<tr>
<td>Study location</td>
<td>Author(s)&amp; date</td>
<td>Estimate &amp; Method</td>
<td>Total soil depth (solum) observed</td>
<td>2001 ASRIS solum thickness (cm) &amp; Bulk Density (Mg m(^{-3}))</td>
<td>Soil formation rate determined (m) Myr(^{-1}) = (mm) kyr(^{-1}) = (\mu m) yr(^{-1})</td>
<td>Modelled pre-European erosion rate of (Lu et al. 2003) scaled by 10% HSDR ((\mu m) yr(^{-1}))</td>
</tr>
<tr>
<td>---------------</td>
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<td>---------------------------------</td>
<td>---------------------------------------------------------------</td>
<td>---------------------------------------------------------------</td>
<td>---------------------------------------------------------------</td>
</tr>
<tr>
<td>Bredbo R., NSW 149.3333, -35.9833</td>
<td>Heimsath et al. (2001)</td>
<td>geomorphology, stratigraphy, accelerator mass spectrometry for 10Be, 26Al</td>
<td>≥25-cm</td>
<td>70 cm, 1.5 Mg m(^{-3})</td>
<td>15 (\mu m) yr(^{-1})</td>
<td>230</td>
</tr>
<tr>
<td></td>
<td>Yoo et al. (2007)</td>
<td>geochemical mass balance, incorporating transport</td>
<td></td>
<td></td>
<td>20-30 (\mu m) yr(^{-1}) slope-averaged net weathering rate 14 (\mu m) yr(^{-1})</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Burke et al. (2009)</td>
<td>geochemical mass balance, Chemical index of alteration (CIA)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Dosseto et al. (2008)</td>
<td>U-series dating</td>
<td>100 cm</td>
<td>1.3 Mg m(^{-3})</td>
<td>min 4 ±1 (\mu m) yr(^{-1}) max 46 ± 14 (\mu m) yr(^{-1})</td>
<td>2.7</td>
</tr>
<tr>
<td></td>
<td>Green et al. (2006)</td>
<td>geochemical mass balance</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Heimsath et al. (2000)</td>
<td>(^{10})Be</td>
<td>100 cm</td>
<td></td>
<td>min 7 (\mu m) yr(^{-1}) max 53 (\mu m) yr(^{-1})</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Burke et al. (2009)</td>
<td>geochemistry, Chemical index of alteration (CIA)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burra Creek, NSW</td>
<td>Fifield et al. (2010)</td>
<td>(^{10})Be</td>
<td>75 cm</td>
<td>80 cm 1.5 Mg m(^{-3})</td>
<td>1-7 (\mu m) yr(^{-1})</td>
<td>5.9</td>
</tr>
<tr>
<td>Tin Camp Creek, NT 133.0667, -12.5</td>
<td>Heimsath et al. (2009)</td>
<td></td>
<td>≤35 cm</td>
<td>40 cm 1.2 Mg m(^{-3})</td>
<td>min. 5 (\mu m) yr(^{-1}), max 20 (\mu m) yr(^{-1})</td>
<td>13</td>
</tr>
<tr>
<td>Fingerpost Hill, WA 128.7802, -17.5609</td>
<td>Fifield et al. (2010)</td>
<td>(^{10})Be</td>
<td>150 cm</td>
<td>128 cm 1.2 Mg m(^{-3})</td>
<td>10 to 27 (\mu m) yr(^{-1})</td>
<td>47.3</td>
</tr>
</tbody>
</table>
2.2.4. Modelled pre-European settlement erosion rates

As an alternative to measuring long-term denudation rate and soil formation rates we now examine modelled estimates of pre-European settlement hillslope erosion rates of Lu et al. (2003) as a first order approximation (Fig. 5A). This approach assumes that landscapes were at steady-state before European settlement and that hillslope erosion by water is the dominant erosion process. The advantage of this approach is the high level of spatial detail given by the map, and the fact that the estimated erosion rates will be more relevant to contemporary landscape processes, rather than using estimates averaged over thousands or millions of years. The model used was the Revised USLE (RUSLE) for Australia (SOILLOSS by Rosewell 1993, cited by Lu et al. 2003). As the RUSLE model does not allow for re-deposition along hillslopes, this is only a rough approximation since net denudation equals the difference between erosion and deposition.

Eliminating outliers (i.e., pixels with values >130 t ha\(^{-1}\) yr\(^{-1}\)), the data distribution of this surface suggests that pre-European hillslope erosion ranged between 0 and 130 t ha\(^{-1}\) yr\(^{-1}\) or, assuming a soil bulk density of 1.3 t m\(^{-3}\), between 0 and 10 m mm yr\(^{-1}\). The median is 0.9 t ha\(^{-1}\) yr\(^{-1}\) or 0.07 mm yr\(^{-1}\), and the mean is 2.6 t ha\(^{-1}\) yr\(^{-1}\) or 0.2 mm yr\(^{-1}\). The spatial distribution shows that about two thirds of the continent has a long-term erosion rate of >0.02 mm yr\(^{-1}\) with discrete parts of northern Australia exceeding rates of 2 mm yr\(^{-1}\). The highest erosion rates under natural conditions are predicted to occur in northern Australia.

When compared to other long-term erosion estimates it is clear that the modelled pre-European hillslope erosion map (Fig. 5A) predicts erosion rates far greater than the estimates given in Table 3. A direct comparison of modelled estimates and soil production rates is shown in Table 3, with modelled estimates often too high by an order of magnitude. In addition, a simple calculation suggests that the pre-European hillslope erosion rates of Lu et al. (2003) are generally too large to be realistic. For example, at the median rate for the modelled map (0.1 mm yr\(^{-1}\)), it would only take three million years to lower the land surface by 300 m, resulting in the disappearance of a section equal to most of Australia as it looks today.

The most likely cause of these over-estimates is the failure of the model to account for deposition of mobilised (eroded) soil along hillslopes; i.e. the model predicts total or “gross” erosion rather than “net” erosion. This feature of the RUSLE model is well known, and is generally accounted for by using a hillslope sediment delivery ratio (HSDR) parameter that can be fixed or varied spatially to adjust the amount of sediment moving into streams (Prosser et al. 2001, 2003; Lu et al. 2006). The HSDR is defined as being <1, and is applied by multiplying HSDR by the estimate of gross hillslope erosion to obtain the proportion of eroded soil that reaches the stream network. Thus it reduces gross erosion to a value representing the quantity of soil leaving the hillslope. Values of HSDR between 0.05 and 0.70 (5% to 70%) are commonly employed, with the value being related to the catchment size (Lu et al., 2006; Prosser et al. 2001; Walling et al., 1983). An HSDR value of 5% is often assigned to large catchments encompassing many thousands of km\(^{2}\) (Prosser et al. 2001), however at smaller scales akin to typical hillslopes (0.1 to 10 km\(^{2}\)) HSDR values of 10% to 40% are more typical (Lu et al., 2006; Walling 1983; Verstraeten et al., 2009).

Fig. 5B shows the result of assuming a 10% HSDR for pre-European net hillslope erosion. This value of HSDR is commonly used to estimate sediment moving into surface waterways (Prosser et al. 2003; McKergow et al. 2005; Lu et al. 2006; Hancock et al. 2006) and provides a good broad-scale fit to the soil production rate estimates in Table 3. The application of 10% HSDR is equivalent to dividing the mapped estimates in Fig 5A by 10. Thus the median erosion rate becomes 0.007 mm yr\(^{-1}\); the mean is 0.02 mm yr\(^{-1}\), and ¾ of the continent has an erosion rate of <0.02 mm yr\(^{-1}\).

We acknowledge the application of a single HSDR value to the whole continent introduces additional uncertainty, and that variables such as slope, rainfall, and soil cover will play a role in determining sediment delivery. Although our use of a 10% HSDR may be in error by a
factor of 2 or 3 in some regions, it is likely that the spatial pattern of erosion captured by Fig. 5B is correct.

Fig. 5. Hillslope erosion rates (A) Pre-European “gross” hillslope erosion rates estimate from Lu et al. (2003). (B) using a Hillslope Sediment Delivery Ratio (HSDR) of 10%. Map A is equivalent to using a HSDR of 100%, and Map B is equivalent to dividing Map A by 10.
2.2.5. Æolian deposition: the role of wind

In Australia, wind-borne deposition is an important landscape process that can build up soil profiles. When dust is deposited onto a soil from a desert source area it may be more valuable for soil functions in its new location, in the way that Saharan dust boosts biomass production in Amazonian forests (Verheijen et al. 2009). Æolian deposits have been an important parent material for soils of south-eastern Australia (Fig. 6; Butler and Hutton 1956; McTainsh and Lynch 1996; Summerell et al. 2000; Chen et al. 2002; Hesse et al. 2003; Greene et al. 2009; Cattle et al. 2009). Hesse et al. (2003) estimate nearly constant rates of 0.20-0.50 t ha\(^{-1}\) yr\(^{-1}\) of dust accumulation on Central Tablelands of New South Wales for the last 50,000 years. Chen et al. (2002) give a minimum rate of 0.13 t ha\(^{-1}\) yr\(^{-1}\) for Æolian deposition in the late Pleistocene. McTainsh and Lynch (1996) report rates of present annual dust deposition on the order of 0.31-0.44 t ha\(^{-1}\) yr\(^{-1}\) in eastern Australia. De Deckker et al. (2008) report a single dust deposition event that blanketed Canberra with an estimated 0.06 t ha\(^{-1}\) of Æolian material in October 2002. Measurements of \(^{137}\)Cs made at CSIRO on this material and material from another major dust storm in 2009 indicate the dust is mainly surface soil (G. Hancock, unpublished data). Assuming a soil bulk density of 1.3 t m\(^{-3}\), these rates can be converted to give a range of 0.001 to 0.004 mm yr\(^{-1}\). These estimates suggest that Æolian deposition rates are of the same order of magnitude as soil formation rates from weathering of parent material discussed earlier. In fact, Greene et al. (2009) state that in many parts of southeastern Australia rates of soil formation and Æolian sediment accumulation are comparable. Thus total soil production rate estimates need to take Æolian deposition as well as weathering rates into account.

![Fig. 6. The approximate zone of parna-derived soil (grey) delineated by Butler and Hutton (1956) and the location of other study sites where Æolian deposits have been identified in soil (from Cattle et al. 2009, with permission).](image-url)
Central Australia is the major source of this aeolian material (McTainsh et al. 2005; Baddock et al. 2009; and see also the animation of dust storm dynamics from Sept. 2009 in on-line supplement). The dust trajectories observed from the remote sensing imagery suggests that wind-borne deposition could be occurring all along the western (windward) slopes of the Great Dividing Range along the entire east coast of Australia. In Queensland, an intriguing observation by Coventry and Williams (1983) that soil thickness and silt content in the upper 60 cm of soil were important in distinguishing between soil classes near Charters Towers may reflect an aeolian contribution to soil accretion: A bimodal particle size distribution dominated by a significant coarse silt peak, with a lesser peak in the clay/fine silt fraction, is considered by many workers to be diagnostic of a significant aeolian component (Greene et al. 2009). Thus soil thickness and aeolian deposition may be related in places.

This evidence suggests that wind-borne deposition or vertical accretion of soil may be offsetting water-borne hillslope erosion. Even if wind-borne deposits do not build-up soil thickness on slopes >6% (Greene et al. 2009) and they get eroded by water almost immediately, they could be offsetting topsoil loss from the A-horizon. In Australia where soils have undergone weathering over millions of years, replenishment of nutrients from dust deposition is likely to be a critical factor in maintaining plant productivity (e.g. Chadwick et al. 1999).

### 2.3. Estimation of modern or post-European settlement erosion rates ($E$)

Compared to long-term denudation rates contemporary erosion rates are thought to have increased under conventional agriculture by one to two orders of magnitude worldwide (Montgomery 2007). In Australia land cover/land use change is proposed as the cause of accelerated erosion rates in post-European time. However geological and modern erosion rates are not always different, particularly where land-use is unchanged. Fifield et al. (2009) conclude that in south-eastern NSW the long-term denudation rate is similar to the modern rate under native forest (0.0025 ±0.0012 mm yr$^{-1}$) whereas in northern WA, currently under grazing, it is now ten-fold higher (increasing from 0.020 ±0.007 mm yr$^{-1}$ to 0.2 mm yr$^{-1}$). In the Sydney Basin bioregion, an area thought to be at steady state, Tomkins et al. (2007) concluded that contemporary rates appear lower than long-term ones, 0.0055 ±4 mm yr$^{-1}$ compared to 0.0215 ±7 mm yr$^{-1}$. This pattern is probably due to the infrequent, episodic nature of driving events such as cyclones and wildfires. However Dosseto et al. (2008) conclude that in the Murray-Darling Basin rivers have modern sediment yields that are greater than what would be expected under steady state. Moreover they find that deviation from steady state is greatest in the Darling basin where fluvial sediment yields suggest that modern erosion rates are two orders of magnitude greater than prior to European settlement. Discrepancies in conclusions drawn by various studies may be due to differences in methods of erosion estimates and their assumptions.

Certain landscapes are more susceptible to erosion than others. Chappell and Brown (1993) report an erosion rate of 0.96 mm yr$^{-1}$ for sodic soils, some of the soils most vulnerable to water-borne erosion, including gullying (Boucher 1990; Rengasamy and Olsson 1991; Ford et al. 1993; Fitzpatrick et al. 1994), therefore this may serve as an upper limit for contemporary rates under undisturbed natural conditions. Sodic soils cover about 13% of Australia (Isbell et al. 1997).

Australia-wide, gully and streambank erosion likely contribute the largest amount of sediments to surface water bodies (Wallbrink et al. 1998; Wasson et al. 1998, 2002, 2010; Prosser et al. 2001; Hughes et al. 2009). In northern Queensland however, hillslope erosion may be important: it is estimated to range from <1 to >70 t ha$^{-1}$ per wet season (or assuming a bulk density of 1.3 t m$^{-3}$, up to 0.005 mm yr$^{-1}$) and to contribute ~60% of sediments delivered to streams (Bartley et al. 2004a; McKergow et al. 2005; Hateley 2007).
Estimates of contemporary rates from studies using geochemical tracers or ‘sediment source fingerprinting’, farm dam and lake sediments are tabulated in Tables 4 and 5. The eroded hillslope component of the total sediment flux at the outlet of the catchment is calculated by multiplying the topsoil fraction of the sediment, as determined using $^{137}$Cs and excess $^{210}$Pb topsoil tracers, by the sediment flux at the outlet. The hillslope denudation rates are then estimated on a ‘whole-of-catchment’ basis, by dividing the hillslope sediment flux at the catchment outlet by the catchment area. These large-scale catchment studies (Table 4) give hillslope denudation rates that compare well with the estimates for the Cenozoic, discussed earlier. However, these whole-of-catchment estimates of hillslope denudation are likely to be lower limits, with in-stream, reservoir, and floodplain deposition reducing the sediment flux seen at the outlet of the catchment. Thus losses of soil from hillslopes may well be higher than the catchment-wide estimates indicate.

Estimates of contemporary rates from regional modelling studies completed since the NLWRA are tabulated in Table 6. Often a combination of geochemical tracers, sedimentation history, and modelling are used to construct sediment budgets for catchments, so a number of studies are repeated in Tables 4-6.

2.3.1. Modelling estimates

The NLWRA and 2001 State of the Environment report used modelling to predict that most of northern Australia, the northern tablelands of New South Wales and the sheep-wheat belt of New South Wales are most at risk of soil loss by water-borne erosion (Lu et al. 2001; 2003) (Fig. 7). The models used were the Revised USLE (RUSLE) for Australia (SOILOSS by Rosewell 1993, cited by Lu et al. 2003) and SedNet. The outcomes of this work are supported by regional studies: e.g., the pattern of clay sediments in the Bonaparte Gulf (Gingele et al. 2001) tends to corroborate the smectitic clay soils, common in the Victoria River basin, as a major source of sediment and thus lends some support to the prediction by Lu et al. (2003) for this region. Increased sediment input to the Great Barrier Reef lagoon post-European settlement has been demonstrated, with the Fitzroy River basin identified as the single largest contributor (Neil et al. 2002; McCulloch et al. 2003; McKergow et al., 2005).

RUSLE/SOILOSS predictions depend on land cover, rainfall erosivity, soil erodibility, slope gradient and length. The standard conditions for RUSLE are small experimental plots: 22m unit plot, on a 9 percent slope, maintained in continuous fallow, tilled up and downhill periodically to control weeds and break crusts that form on the surface of the soil. The plots are ploughed, disked and cultivated as for cropping, except that no crop is grown on the plot (Renard et al. 1997). Importantly, RUSLE does not account for eroded soil (including wind-blown soil) that is re-deposited in adjacent plots. SOILOSS is an application of RUSLE that has been parametrized with a limited set of data around Australia. It is known to perform better in some terrains (sandstone-derived) than others (granite-derived) (Lu et al. 2003).

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5 It only lends partial support since other erosion processes (e.g. gully or streambank erosion) could be more important. Gully and bank erosion have been identified as the dominant erosive processes in the Daly River, Northern Territory (Wasson et al. 2010), Queensland Fitzroy River (Hughes et al. 2009), Mitchell River (Brooks et al. 2009), and Herbert River (Tims et al. 2010) basins.
Table 4. Catchment-wide derived estimates of hillslope soil loss using catchment sediment budgets and fallout radionuclide tracers ($^{137}$Cs, $^{210}$Pb$_{ex}$).

<table>
<thead>
<tr>
<th>Study location (area covered)</th>
<th>Author(s) &amp; date</th>
<th>Catchment area (km$^2$)</th>
<th>Surface soil as % of total sediment load*</th>
<th>Catchment sediment load (kt yr$^{-1}$)</th>
<th>Hillslope erosion* (t ha$^{-1}$ yr$^{-1}$)</th>
<th>Hillslope erosion ** (mm yr$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lake Argyle, NT</td>
<td>Wasson et al. (2002)</td>
<td>46,000</td>
<td>10</td>
<td>24,000</td>
<td>0.52</td>
<td>0.035#</td>
</tr>
<tr>
<td>Brisbane River (below Wivenhoe)</td>
<td>Caitcheon et al. (2001); Wallbrink (2004); Douglas et al. (2003)</td>
<td>10,000</td>
<td>15</td>
<td>300</td>
<td>0.045</td>
<td></td>
</tr>
<tr>
<td>Western Port</td>
<td>Hancock et al. (2001); Wallbrink et al. (2003a)</td>
<td>3250</td>
<td>15</td>
<td>66-96</td>
<td>0.030-0.045</td>
<td>0.002-0.003</td>
</tr>
<tr>
<td>West Gippsland</td>
<td>Hancock and Pietsch (2006); Hancock et al. (2007)</td>
<td>11,800</td>
<td>11</td>
<td>118-150</td>
<td>0.012-0.015</td>
<td>0.001</td>
</tr>
<tr>
<td>East Gippsland</td>
<td>Hancock and Pietsch (2006); Hancock et al. (2007)</td>
<td>8,600</td>
<td>34</td>
<td>100</td>
<td>0.040</td>
<td>0.003</td>
</tr>
<tr>
<td>Mid-Murrumbidgee</td>
<td>Wallbrink et al. (1998)</td>
<td>13,500</td>
<td>15</td>
<td>580</td>
<td>0.07</td>
<td>0.005</td>
</tr>
<tr>
<td>Lake Burraborong</td>
<td>Caitcheon et al. (2007); Rustomji et al. (2008); Wilkinson et al. (2007); Blake et al. (2005)</td>
<td>9,000</td>
<td>67</td>
<td>290</td>
<td>0.23</td>
<td>0.015</td>
</tr>
<tr>
<td>Ovens River</td>
<td>DeRose et al. (2005)</td>
<td>7,200</td>
<td>17</td>
<td>80</td>
<td>0.02</td>
<td>0.001</td>
</tr>
<tr>
<td>Herbert River</td>
<td>Bartley et al. (2004b)</td>
<td>10,000</td>
<td>50</td>
<td>600</td>
<td>0.30</td>
<td>0.02</td>
</tr>
<tr>
<td>Logan-Albert</td>
<td>Hancock and Caitcheon (in prep.)</td>
<td>4,100</td>
<td>20</td>
<td>189</td>
<td>0.09</td>
<td>0.006</td>
</tr>
<tr>
<td>Chaffey Dam</td>
<td>Caitcheon et al. (1995)</td>
<td>420</td>
<td>50</td>
<td>3.7</td>
<td>0.05</td>
<td>0.003</td>
</tr>
</tbody>
</table>

*Estimates of hillslope losses have been made using estimates of the % hillslope contribution as determined from $^{137}$Cs and $^{210}$Pb$_{ex}$ surface soil tracers and annual load of sediment exported from the catchment at the outlet, i.e hillslope erosion rate = [annual export load x hillslope %]/catchment area. Load estimates are estimated from in-stream monitoring data and/or by SedNet modelling. These estimates of hillslope erosion are likely to represent a lower limit due to the fact that sediment is likely to be lost on route from the hillslope to the catchment outlet by in-stream deposition (mainly coarse sediment >63 μm) and by overbank deposition (coarse and fine sediment). Estimates of river sediment delivery rates are required to refine estimates.

**Assuming soil bulk density = 1.5 t m$^{-3}$

#Wasson et al. (2002) write <0.4 mm yr$^{-1}$, apparently in error
Table 5. Erosion rates estimated from farm dams, impoundments, reservoirs, and lakes as sediment traps.

<table>
<thead>
<tr>
<th>Study location/ Area covered</th>
<th>Author(s)&amp; date</th>
<th>Method</th>
<th>Scale (catchment area)</th>
<th>Mean hillslope erosion rate estimate ((t \text{ ha}^{-1} \text{ y}^{-1}))</th>
<th>Other comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>L. Barrine, Qld -17° 15´, 145° 38´</td>
<td>Walker et al. (2000)</td>
<td>Fallout radionuclides, charcoal, pollen, radiocarbon dating</td>
<td>1.0 km²</td>
<td>0.04</td>
<td>&quot;Undisturbed&quot; catchment sedimentation and erosion estimate for last ~1000 years.</td>
</tr>
<tr>
<td>Lake Wellington (West Gippsland) -38°05´, 147°19´</td>
<td>Hancock and Pietsch (2006)</td>
<td>Fallout radionuclides, optical dating in 4 sediment cores</td>
<td>11,800 km²</td>
<td>0.012-0.015</td>
<td>Erosion estimate pertains to last ~60 years.</td>
</tr>
<tr>
<td>L. Burrinjuck -34°59´, 148°38´</td>
<td>Wasson et al. (1987)</td>
<td>Fallout radionuclides, charcoal, pollen dating</td>
<td>13,000 km²</td>
<td>n.d.</td>
<td>Post 1925 sedimentation history only</td>
</tr>
<tr>
<td>L. Burragorang -33°59´, 150°25´ Warragamba dam, NSW -33°53´, 150°36´</td>
<td>Wasson et al. (1994), Rustomji (2006), Rustomji and Wilkinson (2007)</td>
<td>Fallout radionuclides, charcoal</td>
<td>9,000 km²</td>
<td>0.23</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Armstrong and Mackenzie (2002)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Chaffey Dam -31°21´, 151°08´</td>
<td>Caithcheon et al. (1995)</td>
<td>Reservoir sediment budget</td>
<td>420 km²</td>
<td>0.05</td>
<td>50% hillslope contribution assumed from (^{137})Cs measurements.</td>
</tr>
<tr>
<td>Kangaroo Creek Reservoir, Torrens R. catchment, SA -34°52´, 138°47´</td>
<td>Tibby et al. (2010)</td>
<td>Fallout radionuclides, charcoal dating</td>
<td>290 km²</td>
<td>n.d.</td>
<td>Post 1970 sedimentation history only</td>
</tr>
</tbody>
</table>
### Table 5 (continued).

<table>
<thead>
<tr>
<th>Study location/ Area covered</th>
<th>Author(s)&amp; date</th>
<th>Method</th>
<th>Scale (catchment area)</th>
<th>Mean hillslope erosion rate estimate (t ha(^{-1}) y(^{-1}))</th>
<th>Other comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jerrabombera Creek and Yass R. -35° 20´, 149° 10´</td>
<td>Neil and Mazari (1993)</td>
<td>14 farm dams</td>
<td>Hillslope scale (~0.1 to ~3 km(^2))</td>
<td>0.2 -0.6</td>
<td>Cultivated soils Pasture Note: a bulk density of 1.5 kg m(^{-3}) is assumed</td>
</tr>
<tr>
<td>Lake Illawarra, NSW -34° 31´, 150°50´</td>
<td>Chenhall et al. (1995)</td>
<td>Trace metals, ash, (^{137})Cs</td>
<td>270 km(^2) catchment area</td>
<td>2.5 to 10</td>
<td>Sedimentation more rapid on western side.</td>
</tr>
<tr>
<td>Western Sydney, NSW</td>
<td>Erskine et al. (2003)</td>
<td>yield in sediment cores (Neil and Galloway 1989)</td>
<td></td>
<td>3-7</td>
<td>Cultivated soils Grazed hillslopes</td>
</tr>
<tr>
<td>Cordeaux Reservoir, NSW -34°20´, 150°45´</td>
<td>Simms et al. (2008)</td>
<td>Two (^{137})Cs hillslope transects</td>
<td>Hillslope scale</td>
<td>0-0.4</td>
<td>Results obtained from two valley transects and pertain to the D&amp;M erosion model (Walling et al., 2002).</td>
</tr>
<tr>
<td>Little Llangothlin lagoon, NSW -30°05´, 151°47´</td>
<td>Gale and Haworth (2005)</td>
<td>(^{210})Pb chronology</td>
<td>3.2 km(^2) catchment area</td>
<td>2.7</td>
<td>This rate pertains to entire post-European period. The erosion rate is currently estimated to be 0.5 t ha(^{-1}) y(^{-1})</td>
</tr>
<tr>
<td>Tank Creek, near Broken Hill, NSW</td>
<td>Jones et al. (2000)</td>
<td>Dam sediment budget and (^{137})Cs profile of sediment cores</td>
<td>12.5 ha catchment area</td>
<td>0.71</td>
<td>0.78 t ha(^{-1}) y(^{-1}), pre-mid-1950s; 0.59 t ha(^{-1}) y(^{-1}), mid-1950s-present</td>
</tr>
<tr>
<td>Umberumberka, NSW</td>
<td>Wasson and Galloway (1986)</td>
<td></td>
<td>1915-1941 time period</td>
<td>5.2</td>
<td></td>
</tr>
</tbody>
</table>
Table 6. Hillslope erosion rates estimates from various models.

<table>
<thead>
<tr>
<th>Study Location</th>
<th>Author(s)&amp; date</th>
<th>Method</th>
<th>Samples / observations</th>
<th>Resolution</th>
<th>Mean rate t ha⁻¹ yr⁻¹</th>
<th>Error estimated</th>
<th>Other comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>MDB</td>
<td>DeRose et al. (2003, 2004)</td>
<td>SedNet*</td>
<td>9 arc sec or ~250 m</td>
<td>0.079</td>
<td>indirect</td>
<td>comparison with fallout radionuclide tracers</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Murrumbidgee R., NSW</td>
<td>Verstraeten et al. (2007); Wilkinson et al. (2004)</td>
<td>WATEM/SEDEM† SedNet</td>
<td>25-m</td>
<td>0.17</td>
<td>Yes</td>
<td>comparison with long-term mean load reported by Olive et al. (1996)</td>
<td>also see Olley and Scott (2002); Olley and Wasson (2003)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Goulburn-Broken R. catchments</td>
<td>DeRose et al. (2003)</td>
<td>SedNet</td>
<td>20 m</td>
<td>No</td>
<td></td>
<td>includes comparison w NLWRA</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Wilkinson et al. (2005, 2009)</td>
<td>SedNet 8 gauging water quality stations</td>
<td>20 m</td>
<td>0.046</td>
<td>indirect, 95% confidence interval reported for WQ data</td>
<td>modified rating curve method of Rustomji and Wilkinson (2008)</td>
<td>includes estimates of reservoir trapping efficiency</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ovens R. catchment</td>
<td>DeRose et al. (2005)</td>
<td>SedNet</td>
<td>20 m</td>
<td>0.025</td>
<td>indirect</td>
<td>comparison with fallout radionuclide and geochemical tracers</td>
<td>includes comparison w NLWRA and MDB</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lk Burragorang, Sydney Catchment</td>
<td>Rustomji (2006)</td>
<td>SedNet</td>
<td></td>
<td>0.04</td>
<td>indirect</td>
<td>comparison with fallout radionuclide and geochemical tracers, suspended sediment loads estimated at gauging stations</td>
<td>various parametizations of SedNet are explored. RUSLE performed poorly</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lk. Wingecarribee, Sydney Catchment</td>
<td>Olley and Deere (2003)</td>
<td>SedNet</td>
<td></td>
<td>0.1</td>
<td>indirect</td>
<td>comparison w radionuclide tracers of sediment sources</td>
<td></td>
</tr>
</tbody>
</table>
Table 6 (continued).

<table>
<thead>
<tr>
<th>Study Location</th>
<th>Author(s)&amp; date</th>
<th>Method</th>
<th>Samples / observations</th>
<th>Resolution</th>
<th>Mean rate t ha(^{-1}) yr(^{-1})</th>
<th>Error estimated</th>
<th>Other comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mt Lofty Ranges, SA</td>
<td>Wilkinson et al. (2005)</td>
<td>SedNet</td>
<td>10 m</td>
<td></td>
<td>indirect for Torrens catchment</td>
<td>No</td>
<td>comparison with fallout radionuclide and geochemical tracers</td>
</tr>
<tr>
<td>Daintree, Saltwater, Mossman and Mowbray catchments, north Qld</td>
<td>Bartley et al. (2004a)</td>
<td>SedNet</td>
<td>34 gauging stations</td>
<td>10 m</td>
<td>4.65</td>
<td>No</td>
<td>includes comparison w NLWRA and Herbert R. and Johnstone R. catchments</td>
</tr>
<tr>
<td>Mary R. catchment, Qld</td>
<td>DeRose et al. (2002)</td>
<td>SedNet</td>
<td>25 m</td>
<td></td>
<td>0.93</td>
<td>No</td>
<td>NLWRA estimated 5.4 t ha(^{-1}) yr(^{-1})</td>
</tr>
<tr>
<td>Herbert R. catchment</td>
<td>Bartley et al. (2004b)</td>
<td>SedNet</td>
<td></td>
<td></td>
<td>10% indirect</td>
<td></td>
<td>comparison with fallout radionuclide and geochemical tracers and water quality data</td>
</tr>
<tr>
<td>Herbert R. Floodplain</td>
<td>Visser et al. (2007)</td>
<td>field study</td>
<td></td>
<td></td>
<td>2.5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burdekin R. basin, Qld</td>
<td>Prosser et al. (2002)</td>
<td>SedNet</td>
<td>9 arc sec, ~250 m</td>
<td></td>
<td>9.5</td>
<td>No</td>
<td></td>
</tr>
<tr>
<td>Weany Creek sub-catchment of Burdekin R</td>
<td>Bartley et al. (2007a); Kinsey-Henderson et al. (2005b)</td>
<td>field study</td>
<td></td>
<td></td>
<td>0.21</td>
<td></td>
<td>Field-based flume study LISEM over-predicts sediment load by 47%</td>
</tr>
<tr>
<td>Bowen R. basin, sub-basin of Burdekin R.</td>
<td>Bartley et al. (2004c)</td>
<td>SedNet</td>
<td></td>
<td></td>
<td>1.4</td>
<td></td>
<td>72% from hillslope</td>
</tr>
</tbody>
</table>
Table 6 (continued).

<table>
<thead>
<tr>
<th>Region</th>
<th>Study Authors</th>
<th>Method</th>
<th>Results</th>
<th>Additional Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fitzroy R. basin, Qld</td>
<td>Dougall et al. (2006)</td>
<td>SedNet</td>
<td>3.2</td>
<td>part of Cogle et al. (2006)</td>
</tr>
<tr>
<td>Great Barrier Reef catchments</td>
<td>Cogle et al. (2006)</td>
<td>SedNet</td>
<td>2.8-75.3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Bartley et al. (2007b)</td>
<td>SedNet</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Western Port Bay</td>
<td>Hughes et al. (2003)</td>
<td>SedNet</td>
<td>1.8-0.1</td>
<td>Yes                                comparison with fallout radionuclide and geochemical tracers in companion report by Wallbrink et al. (2003)</td>
</tr>
<tr>
<td>Gippsland Lakes</td>
<td>Hancock et al. (2007)</td>
<td>SedNet</td>
<td>20-m</td>
<td>20 km²                              Yes Use of fallout tracers and lake sedimentation estimates to provide constraints comparison with fallout radionuclide and geochemical tracers</td>
</tr>
<tr>
<td>Daly-Douglas R., NT</td>
<td>Lynch and Hill (2007)</td>
<td>SedNet</td>
<td>not reported</td>
<td>No                                 preliminary results only</td>
</tr>
<tr>
<td>Tin Creek catchment, NT</td>
<td>Hancock et al. (2002)</td>
<td>SIBERIA§</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

* In SedNet, hillslope erosion from sheet and rill erosion processes is estimated using the Revised Universal Soil Loss Equation (RUSLE; Renard et al., 1997). The RUSLE calculates mean annual soil loss (Y, tonnes ha⁻¹ y⁻¹) as a product of six factors: rainfall erosivity (R), soil erodibility (K), hillslope length (L), hillslope gradient (S), ground cover (C) and landuse practice factor (P): Y = RKLSCP. The overall sediment budget in SedNet also includes estimates for gully and channel erosion (Prosser et al. 2001).

CSIRO Land and Water still hold all the results of the SedNet studies in digital format.

†WATEM/SEDEM is a European water-borne erosion model that also uses RUSLE to estimate mean annual soil loss (Verstraeten and Prosser 2008).

#Limburg Soil Erosion Model (LISEM) is a physically-based runoff and erosion model for research, planning and conservation purposes. It simulates the spatial effects of rainfall events on small watersheds. ([http://www.itc.nl/lisem](http://www.itc.nl/lisem))

§ SIBERIA is a physically-based model for simulating the geomorphic evolution of landscapes subject to fluvial and diffusive erosion and mass transport processes (Willgoose and Riley 1998). It uses a DEM to determine drainage areas and catchment geomorphology and dynamically adjusts the land surface as it changes through time in response to erosion and deposition. Dynamic models that employ DEMs have considerable advantages over traditional modelling approaches, such as the RUSLE, because they produce a better representation of slope and angle over the duration of the simulation. SIBERIA has also been used to model soil depth as a function of landscape evolution (Saco et al. 2006) and could be useful in the soil formation assessment of the current project.
Since the NLWRA and 2001 State of the Environment report, a project focusing on water-borne erosion in the Murray-Darling Basin, also using RUSLE/SOILLOSS and SedNet, found that the original estimates of total sediments and nutrients transported in rivers were overestimates, especially for gully and streambank erosion (Prosser et al. 2003). More detailed water-borne erosion modelling undertaken using high resolution (20-m) digital elevation models (DEM) and 25-m land use/land cover mapping in the Ovens catchment showed that hillslope erosion was half that estimated using 250-m data (De Rose et al. 2005). In the Mary River catchment, Queensland, De Rose et al. 2002 found that the average hillslope erosion estimated using RUSLE with a high resolution (25-m) DEM and detailed land use/land cover map was five times lower than the original NLWRA estimate of Lu et al. (2001). In the Fitzroy basin modelling gives a 30% range (uncertainty) in the predicted contribution of hillslope erosion (Wilkinson, 2008), and hillslope erosion from uncultivated land (i.e., grazed pasture/woodland) is a comparatively minor contributor of sediment to the river network (Hughes et al. 2009). Nevertheless, despite these differences a visual comparison of hillslope erosion maps for the catchments that drain into the Great Barrier Reef shows that the general spatial pattern using DEMs and land cover maps with higher resolution is essentially the same (i.e. comparing Fig. 7 above showing NLWRA with Fig. 1.8 in Cogle et al. 2006).

Further detailed studies in catchments around Australia have compared the results of NLWRA modelling with field measurements: Wasson et al. (2010) conclude from geochemical tracers that no discernible input of topsoil from cleared land adjacent to the river is evident in the Daly River basin whereas RUSLE predicted moderate hillslope erosion there (Fig. 7). A recent study in the Herbert catchment using $^{239}$Pu as a geochemical tracer (Tims et al. 2010) concluded that RUSLE modelling of Lu et al. (2003) had over-estimated contributions to Herbert River sediment from hillslope erosion by a factor of about 2 (~20% versus ~40% by RUSLE). Other research on erosion has compared sediment tracing methods with various modelling results and concluded that the RUSLE model over-estimates hillslope erosion in places as different as Arnhem Land, Northern Territory (Hancock et al. 2008) and the Hunter Valley, NSW (Martinez et al. 2009).

As noted earlier, the fact that the RUSLE model does not account for hillslope soil re-deposition is well known to SedNet modellers who use a hillslope sediment delivery ratio (HSDR) to adjust the amount of sediment moving into streams (Prosser et al. 2003; Lu et al. 2006). Similar to our pre-European erosion rate estimates (section 2.2.4), we have used a HSDR of 10% to provide a map of hillslope soil losses (Fig. 7B). As noted, while the application of HSDR may introduce errors equivalent to a factor of 2 or 3, the broad-scale spatial patterns are likely to be correct. The map in Fig. 7B obviously shows the same patterns as the gross erosion map (Fig. 7A) but with erosion reduced by a factor of 10, and identifies regions of rapid erosion (>0.2 mm yr$^{-1}$) in the north-west and coastal north-east and eastern Australia.
Fig. 7. Current average annual hillslope erosion rates (A) “gross” hillslope erosion rates predicted by Lu et al. (2003). (B) The same data with a HSDR of 10% applied (equivalent to reducing the rates shown in Fig. 7A by a factor of 10) is used to estimate net erosion rate.
2.3.2. Mapping net soil redistribution (mid 1950s to early 1990s) using the 137Cs technique.

Soil erosion is often insidious requiring the removal of considerable quantities before loss is noticeable. Soil erosion measurement and monitoring approaches, particularly in semi-arid environments where erosion is driven mainly by infrequent extreme events, require long and expensive campaigns to provide representative and reliable estimates. Even with the results of such long-lived studies the extrapolation of results from small experimental plots across large areas is notoriously unreliable. The caesium-137 (137Cs) technique for estimating soil redistribution overcomes these problems by providing retrospective information on medium-term (30–40 years) net erosion and deposition rates without the need for long-term monitoring programmes. It has been used successfully in many parts of the world to estimate net soil redistribution by the combined effect of wind, water and more recently by tillage erosion (Walling and Quine, 1993; Quine et al., 1996). Despite some limitations (Chappell, 1999) associated with dilution or unlabelled soil 137Cs, preferential removal of 137Cs-rich material by wind, the difficulty of establishing reference levels in wind eroded environments and some uncertainties in cultivated situations (Bremer et al., 1995), its use is well established. There are many examples of the application of the 137Cs technique to fields and small catchments in Australia (McCallan et al., 1980; Longmore et al., 1983; McFarlane et al., 1992; Loughran et al., 1989; 1990; 1993; Harper and Gilkes, 1994; Gillieson et al., 1996; Krause et al., 2003; Loughran and Balog, 2006; Hancock et al., 2008; Simms et al., 2008; Martinez et al., 2009). The 137Cs technique is commonly based on the establishment of the local amount of 137Cs deposited with rainfall from atmospheric weapons testing, mainly during the 1960s (Ritchie and McHenry, 1990). This local reference 137Cs inventory should come from a site that has been undisturbed by erosion or deposition. Net soil redistribution is based on the relationship between the amount of 137Cs at the reference site and the amounts at selected sample locations. The 137Cs loss or gain, relative to the reference inventory, is used to establish a relationship with soil erosion and deposition, respectively. A range of calibration approaches may be used (Walling and Quine, 1990; Sutherland and de Jong, 1990; Walling and He, 2001; Walling et al., 2002). In Australia empirical calibration relationships between 137Cs loss and gain (X) and soil erosion have been established from long-term experimental plots (Elliott et al., 1996). A calibration relationship is required to convert the percentage of 137Cs lost or gained (X) relative to the 137Cs inventory at the undisturbed reference location. This study follows that of Loughran et al. (2004) and uses the Australian Empirical Models (AEMs); one for calculating net soil loss (Y; kg ha\(^{-1}\) yr\(^{-1}\)) for sites which had never been cultivated (N=31; Equation 1) and the other for sites which had been used for cultivation (N=61; Equation 2):

\[
Y = 17.49 (1.0821)^X \quad (1)
\]

\[
Y = 296.1 (1.0539)^X \quad (2)
\]

Net soil accumulation was calculated using these equations in reverse mode for sites which had gained 137Cs. In this case, Y is net soil gain and X is the percentage 137Cs gain relative to the reference value. Both relationships were derived from long-term soil-loss measurements using runoff-erosion plots, or similar experiments in New South Wales, Queensland and Western Australia (Loughran and Elliott, 1996). Equations (1) and (2) are revisions of the original equations presented by Elliott et al. (1990), following correction of the measured sediment yields from plots of the Soil Conservation Service of New South Wales (Lang, 1992; Loughran and Elliott, 1996).

In Australia, 206 sites were sampled and analyzed for 137Cs during a reconnaissance study of soil re-distribution undertaken in the early 1990s as part of the National Soil Conservation Program (Loughran et al. 2004). Sampling was undertaken along single transects, down complete slopes, at paired sites in the same locality with typical land management practices within selected agricultural-economic regions (Fig. 8). At most of the identified sites a 137Cs reference was established in addition to 137Cs inventory from a hillslope. We have obtained the 137Cs data for these sites (Moore and Ciganovic, 1995; Elliott et al., 1996; Lorimer et al., 1996; Elliott et al., 1997; Richley et al., 1997; McFarlane et al., 2000; Elliott et al., 2002). Despite the paired approach of the national survey, 137Cs reference inventory data were not
available at all sites for a variety of reasons. In Tasmania (Richley et al., 1997) and Western Australia (McFarlane et al., 2000), some reference values appeared anomalously high or low for their districts and were not used in this analysis. Additional samples have been included based on studies performed since the National Reconnaissance Survey and corrected to the year 1990. There are two sites in New South Wales (Gary Hancock unpublished data; Blake et al., 2009), one site in the Australian Capital Territory (Wallbrink et al., 1994), one site in Victoria (Antonia Gamboa Rocha unpublished data), one site in Tasmania (Wallbrink et al., 1994), three sites in northern Queensland (Everett, 2009; Pfitzner et al., 2004) and one site in the Northern Territory (Hancock et al., 2008). Consequently, only 141 sites of the $^{137}$Cs reference inventory and 200 sites for $^{137}$Cs inventory values were used in the analysis performed here.

![Fig. 8. Location of sites sampled for $^{137}$Cs during the National Reconnaissance Survey of Australia by Loughran et al. (2004).](image)

The full potential has not been realised by most $^{137}$Cs studies for using point measurements to represent soil redistribution across scales of variation in the landscape particularly over areas larger than a hillslope (de Roo, 1991; Chappell, 1998). An example of this potential is Chappell and Warren (2003) who provided to date the largest map of $^{137}$Cs-derived net soil redistribution (approx. 20 km$^2$) within and between fields, farms and across a region of eastern UK using only 180 samples. To use $^{137}$Cs measurements at points to map net soil redistribution across vast expanses requires an approach (outlined below) which differs from those traditional soil $^{137}$Cs studies, by developing a reliable description of the variability in soil $^{137}$Cs over space (variogram). That variogram is used to provide the weight of influence for surrounding values when making estimates at unsampled locations (kriging).

Kriging is one of the most reliable two-dimensional spatial estimators (Laslett et al., 1987; Laslett and McBratney, 1990; Laslett, 1994) and produces more reliable estimates than other methods of interpolation (Webster and Oliver, 2001). Kriging provides estimation variances which are largely an expression of sample density. The estimates are vastly improved by the use of a secondary variable (co-kriging). Other geostatistical techniques are available to
tackle the deterministic and stochastic components e.g., regression-kriging. In all variants of kriging the map of local estimates may not be best as a whole. Estimation algorithms tend to smooth out local details of the spatial variation of the attribute. Such conditional bias is a serious shortcoming when trying to detect patterns of extremes such as zones of large and small rainfall. The smoothing tends not to be uniform as it depends on the local data configuration (Goovaerts, 1997). Thus, kriged maps may display artifact structures. Stochastic simulation for uncertainty modelling generates an ensemble of equally probable realisations of the property’s spatial distribution and enables differences amongst the realisations to be used as a measure of uncertainty. There are many (co)-simulation techniques to choose from and no one technique is best for all cases (Goovaerts, 1997). As with co-kriging, co-simulation improves the precision of the estimates in the presence of a (spatially) correlated secondary variable. Further details about stochastic simulation can be found in recent books on the subject (Goovaerts, 1997; Deutsch and Journel, 1998; Chiles and Delfiner, 1999).

Sequential indicator co-simulation is used here to provide many realisations or equally-probable $^{137}$Cs reference and net soil redistribution maps combined with rainfall and soil type data, as respective covariates. The realisations have the same resolution as the rainfall data, namely 5-km. The sequential indicator co-simulation approach also has the benefit of providing a map of the probability of exceeding a pre-defined threshold e.g., tolerable soil erosion which is useful for the allocation of remediation or conservation resources for policymakers. The aim is to provide a preliminary baseline across Australia of net (mid 1950s to early 1990s) soil redistribution and its uncertainty. The detailed methodology to achieve this aim is provided in two related papers (Chappell et al., in preparation$^6$). Since this work is not published, a brief outline of the datasets and methodology is provided in Appendix 1 and an assessment of the performance of the estimates is provided in Appendix 2.

2.3.2.1 Calibration of $^{137}$Cs difference to soil redistribution
The calibration of percentage $^{137}$Cs difference to net soil redistribution (t ha$^{-1}$ yr$^{-1}$) makes use of the Australian Empirical Models (AEMs) described above. However, these models also contribute a source of uncertainty. The model uncertainty was introduced using the bootstrap procedure (Appendix 2). The results of combining the spatial uncertainty and model uncertainty are shown in Figure 9. The map of net soil redistribution (Fig. 9a) shows areas that are stable, depositional or have very small erosion rates inland of south-eastern Australia, the northern parts of Australia and central Australia. The remainder of Australia is eroding. Areas that have large erosion rates include the main cultivated areas along the coastal regions of Western Australia, South Australia, Victoria, New South Wales and Queensland. The most eroded area is in the western most region of Western Australia (Pilbara /Gascoyne region) which has median erosion rates of more than 6 t ha$^{-1}$ yr$^{-1}$. The map of net soil redistribution interquartile range with model uncertainty (Fig. 9b) show similarly large magnitude areas of uncertainty in southern South Australia and the mid-west of New South Wales. The areas of greatest uncertainty include mid South Australia, mid New South Wales and south-west Queensland. This uncertainty surrounds the areas of stable or net deposition. The main cultivation regions of Australia have a similar uncertainty of ±2 t ha$^{-1}$ yr$^{-1}$. This means that we can be highly certain that these areas are net eroding. To generalise this point we calculated the probability of exceeding a threshold of 0.5 ha$^{-1}$ yr$^{-1}$ (suggested by Loughran et al., 2004). This rate is about twice the average soil production rate across the continent.

$^6$ Contact Adrian Chappell at Adrian.Chappell@csiro.au for further information.
Fig. 9. $^{137}\text{Cs}$-derived net (1950s-1990) soil redistribution rate (t ha$^{-1}$ yr$^{-1}$) over Australia. (A) The median soil redistribution rate. Negative values indicate soil loss and positive values indicate soil gain. (B) the interquartile range (t ha$^{-1}$ yr$^{-1}$)$^2$ with bootstrap uncertainty.
The probability map is shown in Fig. 10. On this basis, the areas showing the greatest probability of soil loss include the Pilbara / Gascoyne region, much of the agricultural regions of south-western Western Australia, a small portion of South Australia, the majority of Victoria and the cultivated areas (in 1992/93) of New South Wales and Queensland. Additional areas identified with a probability >0.5 and include the majority of Western Australia and the remainder of the cultivated areas (in 1992/93) of New South Wales and Queensland.

Fig. 10. Probability of exceeding a threshold of -0.5 t ha⁻¹ yr⁻¹ based on per-point realisations across Australia. Red means a high probability, blue means a low probability.

For Australia as a whole the net soil redistribution varies from -1.02 to +0.04 t ha⁻¹ yr⁻¹ (25th and 75th percentile) with a median of -0.19 t ha⁻¹ yr⁻¹. Negative values signify erosion (soil loss) and positive values signify deposition (soil gain). The net soil redistribution rates for uncultivated land in 1992/93 ranged between -0.8 to +0.04 t ha⁻¹ yr⁻¹ with a median of -0.16 t ha⁻¹ yr⁻¹. The net soil redistribution rates for cultivated land in 1992/93 ranged between -4.11 to +0.34 t ha⁻¹ yr⁻¹ with a median of -1.26 t ha⁻¹ yr⁻¹. At the median level, the results show that about eight times more soil was lost from cultivated areas than from uncultivated areas. A similar pattern was evident from analysis of the national ¹³⁷Cs reconnaissance samples (Loughran et al., 2004). An assessment of performance of the estimates against the measured data is provided in Appendix 2.

Overall, the pattern of ¹³⁷Cs-derived soil redistribution suggests that little net topsoil loss has occurred over large areas of Australia over the last 50 years, even where hillslope erosion is reportedly a dominant process, e.g. in the Queensland Wet Tropics (Hateley 2007). The areas of severe soil loss (>2 t ha⁻¹ yr⁻¹) in the Pilbara, West Australia, the New England Tablelands in NSW, and in Victoria (Fig. 9) coincide with areas where gully erosion is serious and landscape dissection is marked (Hughes et al. 2001). However, gullies did not form part of the sampling design and they tend to produce soil which is unlabelled in ¹³⁷Cs and complicate the use of the ¹³⁷Cs technique. Since soil type is used as a covariate in the ¹³⁷Cs-
derived estimates, the coincidence is likely caused by high soil erodibility in these regions. The results for the Pilbara are consistent with the RUSLE pattern and the 2001 SoE report which identified the Pilbara as one of the most degraded areas in Australia (http://www.environment.gov.au/soe/2001/publications/theme-reports/land/pubs/land.pdf). In Victoria large tracts of sodic soils are subject to gullying and tunnel erosion (Boucher 1990; Ford et al. 1993); also see the map at http://www.dpi.vic.gov.au/dpi/vro/vrosite.nsf/pages/lwm_land_deg_gully_dist. Typically, erosion of sodic soils is also characterized by a complete loss of the A-horizon (Chappell and Brown 1993).

2.3.3. Comparison between $^{137}$Cs-based estimates, modelling, and catchment sedimentation histories and limitations of erosion predictions by these methods

Comparison of the $^{137}$Cs-based soil redistribution map with hillslope erosion estimated from geochemical tracer and farm dam/reservoir sedimentation history studies and annual erosion predictions from the RUSLE/SOILLOSS model scaled by 10% HSDR are reported in Table 7. There is up to an order of magnitude difference between the $^{137}$Cs-based soil erosion estimates and the RUSLE/SOILLOSS model scaled by 10% HSDR, with the $^{137}$Cs-based estimates mostly being lower. Overall the RUSLE/SOILLOSS model shows a better match with tracer/farm-dam estimates. However, there are subtle differences in what is being measured or predicted between all these studies so the comparisons must be treated with caution. The continental scale of the $^{137}$Cs-derived net soil redistribution map makes it difficult to make a comparison with regional scales and detailed hillslope studies. Conversely, it is also difficult for studies conducted over small areas to make an assessment at the resolution (5 km) of this study.

Major limitations associated with the different methods are:

- The $^{137}$Cs-based map is based on relatively few measurements. There are little or no data for large areas of the continent, including the Wet Tropics in Queensland, western NSW, western Tasmania, WA, the Eyre Peninsula and L. Eyre regions of South Australia, where the uncertainty of the predictions is high (Fig. 9B).

- The $^{137}$Cs soil distribution analysis relies on soil type and, in the reference inventory map, rainfall as covariates. It does not include spatial variations in slope and vegetation cover. This may be why it does not capture known soil loss from regions of strong gradients in slope, seasonally high rainfall and changing vegetation cover, e.g. in the Kimberley (McFarlane et al. 2000).

- The spatial resolution of the $^{137}$Cs method is 5 km, thus soil losses on the hillslope scale (<5 km) may not be captured.

- Gross erosion estimates using RUSLE are based on results from ~20 m experimental plots, and while these provide information on hillslope soil mobility, they are clearly overestimates of hillslope soil losses. The introduction of a uniform 10% HSDR appears to provide better agreement with catchment data, however this application introduces an unknown additional level of uncertainty (perhaps a factor of 2) since HSDR is known to be spatially variable across landscapes.

- RUSLE estimates are very sensitive to the way vegetative cover is determined. For example erosion is over-estimated if dead grass cover is not taken into account.

- Geochemical tracer studies undertaken over large catchments are likely to have under-estimated hillslope losses by not including sediment deposition within the river network during overbank floods.
In summary, while the estimates of hillslope losses using RUSLE have an unknown uncertainty associated with them, the spatial patterns are likely to be correct given the high erosion rates observed in northern Australia (Prove et al. 1995; McFarlane et al. 2000; Hateley 2007; Fifield et al. 2009). The $^{137}$Cs-based estimates are only a preliminary assessment of hillslope soil loss, based on the sparse data collected during the National Reconnaissance Survey in 1990. While the results are encouraging, they do not show the expected erosion trends in high rainfall regions of coastal Queensland, coastal NSW, and the Kimberly. On the other hand, $^{137}$Cs-based predictions identify areas in southern Australia where the erosion risk is believed to be under-estimated by RUSLE (Hairsine et al. 2009). The method also has the capacity to identify regions where erosion is occurring as a result of factors not represented by RUSLE modelling, i.e. soil redistribution by wind.
Table 7. Comparison of catchment-wide derived estimates of hillslope soil loss. Note $^{137}$Cs-based estimates are negative when they indicate soil loss. Erosion estimates by other methods always indicate soil loss.

<table>
<thead>
<tr>
<th>Study location</th>
<th>Author(s) &amp; date</th>
<th>Catchment area (km$^2$)</th>
<th>Hillslope contribution (%)</th>
<th>Load estimate (kt yr$^{-1}$)</th>
<th>Hillslope erosion estimate (t ha$^{-1}$ yr$^{-1}$)</th>
<th>median NRS $^{137}$Cs-based estimate * (zonal mean, t ha$^{-1}$ yr$^{-1}$)</th>
<th>RUSLE/SOILLOSS-based estimate (Lu et al. 2003) scales by 10% HSDR (zonal mean, t ha$^{-1}$ yr$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Brisbane River (below Wivenhoe)</td>
<td>Caitcheon et al. (2001); Wallbrink (2004); Douglas et al. (2003)</td>
<td>10,000</td>
<td>15</td>
<td>300</td>
<td>0.045</td>
<td>-0.29 for whole Brisbane R. catchment†</td>
<td>0.96 for whole Brisbane R. catchment</td>
</tr>
<tr>
<td>Western Port</td>
<td>Hancock et al. (2001); Wallbrink et al. (2003a)</td>
<td>3250</td>
<td>15</td>
<td>66-96</td>
<td>0.03-0.045</td>
<td>-1.4</td>
<td>0.02 for Bunyip R. catchment</td>
</tr>
<tr>
<td>West Gippsland</td>
<td>Hancock and Pietsch (2006); Hancock et al. (2007)</td>
<td>11,800</td>
<td>11</td>
<td>118-150</td>
<td>0.012-0.015</td>
<td>-0.94 to -0.23</td>
<td>0.13-0.01</td>
</tr>
<tr>
<td>East Gippsland</td>
<td>Hancock and Pietsch (2006); Hancock et al. (2007)</td>
<td>8,600</td>
<td>34</td>
<td>100</td>
<td>0.040</td>
<td>-0.67</td>
<td>0.07</td>
</tr>
<tr>
<td>Upper- Murrumbidgee</td>
<td>Verstraeten and Prosser (2008)</td>
<td>30,000</td>
<td></td>
<td></td>
<td>0.02-1.32</td>
<td>-0.02 to -0.71</td>
<td>0.07-1.4</td>
</tr>
<tr>
<td>Jerrabomberra Creek</td>
<td>Neil and Mazari (1993)</td>
<td>100?</td>
<td>100?</td>
<td>0.59</td>
<td>-0.44</td>
<td>0.2 for whole catchment</td>
<td>0.4</td>
</tr>
<tr>
<td>Mid-Murrumbidgee</td>
<td>Wallbrink et al. (1998)</td>
<td>13,500</td>
<td>15</td>
<td>580</td>
<td>0.07</td>
<td>-0.02 to -0.64</td>
<td>-0.06</td>
</tr>
<tr>
<td>Lake Burragorang</td>
<td>Caitcheon et al. (2007); Rustomji et al. (2008); Wilkinson et al. (2007); Blake et al. (2005)</td>
<td>9,000</td>
<td>67</td>
<td>290</td>
<td>0.23</td>
<td>-0.28</td>
<td>0.10</td>
</tr>
<tr>
<td>Ovens River</td>
<td>DeRose et al. (2005)</td>
<td>7,200</td>
<td>17</td>
<td>80</td>
<td>0.02</td>
<td>-1.07</td>
<td>0.17</td>
</tr>
<tr>
<td>Study location (area covered)</td>
<td>Author(s) &amp; date</td>
<td>Catchment area (km²)</td>
<td>Hillslope contribution (%)</td>
<td>Load estimate (kt yr⁻¹)</td>
<td>Hillslope erosion estimate (t ha⁻¹ yr⁻¹)</td>
<td>median NRS $^{137}$Cs-based estimate * (zonal mean, t ha⁻¹ yr⁻¹)</td>
<td>RUSLE/SOILLOSS-based estimate (Lu et al. 2003) scales by 10% HSDR (zonal mean, t ha⁻¹ yr⁻¹)</td>
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<td>-------------------------------------------------</td>
<td>-------------------------------------------------</td>
</tr>
<tr>
<td>Goulburn River</td>
<td>Wilkinson et al. (2009)</td>
<td>17 000</td>
<td>30-68</td>
<td>0.01-0.08</td>
<td>-1.22</td>
<td>0.11</td>
<td></td>
</tr>
<tr>
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<td>Wilkinson et al. (2009)</td>
<td>7 000</td>
<td>44-55</td>
<td>0.008-0.06</td>
<td>-1.45</td>
<td>0.05</td>
<td></td>
</tr>
<tr>
<td>Herbert River</td>
<td>Bartley et al. (2004b)</td>
<td>10,000</td>
<td>50</td>
<td>600</td>
<td>0.30</td>
<td>-0.030</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Hancock and Caitcheon (in prep.)</td>
<td>4,100</td>
<td>20</td>
<td>189</td>
<td>0.09</td>
<td>-0.19</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Wasson et al. (2002)</td>
<td>46,000</td>
<td>10</td>
<td>24,000</td>
<td>0.52</td>
<td>-0.36</td>
<td></td>
</tr>
<tr>
<td>L. Argyle</td>
<td>Walker et al. (2000)</td>
<td>1</td>
<td>100?</td>
<td>0.04</td>
<td>-0.32</td>
<td>1.2</td>
<td></td>
</tr>
<tr>
<td>Little Liangothlin lagoon, NSW</td>
<td>Gale and Haworth (2005)</td>
<td>3.23</td>
<td>100?</td>
<td>currently 0.5</td>
<td>-0.40</td>
<td>0.32</td>
<td></td>
</tr>
</tbody>
</table>
3. Setting tolerable erosion rates for Australia

Given that several definitions of tolerable soil erosion rates are based on soil production rates and a notional balance between the two (Table 1), we can use known soil production rates as a starting point for setting T-values for Australia. Li et al. (2009) call this the $T_1$-value (Table 1). Assuming that landscapes are in steady-state balance, we can also use known denudation rates since these are equivalent to soil production rates under this assumption. Therefore as a spatially undifferentiated first approximation, a rate of $1.5 \times 10^{-5}$ m yr$^{-1}$ (or 0.20 t ha$^{-1}$ yr$^{-1}$, assuming a bulk density of 1.3 t m$^{-3}$) is selected to represent $P$ and hence the sustainable erosion rate or $T_1$-value. This value corresponds to the mean value of long-term denudation and soil production estimates given in sections 2.2.1 and 2.2.2. It is also the average denudation rate over the last 65 Myr (Kohn et al. 2002). In places where soil gains from aeolian deposition are approximately equal to those from soil formation, e.g. the western slopes of the Great Dividing Range (Greene et al. 2009), the proposed rate of $1.5 \times 10^{-5}$ m yr$^{-1}$ may be doubled to $3.0 \times 10^{-5}$ m yr$^{-1}$ (0.4 t ha$^{-1}$ yr$^{-1}$). The $^{137}$Cs-based soil redistribution map suggests that areas of soil gain extend from SA to north Queensland (Fig. 9), thus the higher $T_1$-values could apply there too.

Fig. 11 shows areas exceeding $T_1$-values. Two maps are shown using different soil production terms; (A) $P = 15$ μm yr$^{-1}$, and (B) $P$ estimated from pre-European estimates of Lu et al. (2003) scaled by 10% HSDR (see Fig 5B). Both maps show regions of eastern Australia greatly exceeding $T_1$-value. However, an interesting and potentially important difference is the area exceeding $T_1$-value in the Kimberly and Pilbara regions is much reduced in map B. This difference reflects variations in the natural (pre-European) value of soil erosion, as shown in Fig. 5B, and indicates there may have been little change in net erosion as a result of European land use practices.

Setting tolerable erosion values that reflect reduction in productivity as a function of soil loss ($T_2$-values) is difficult at present given the lack of data on nutrient distribution with depth and the complex interactions between vegetation, microbiota, and soil organic matter. Crop yield decline follows a curvilinear, negative exponential in most eroding tropical and many temperate soils (Stocking 2003). Productivity diminishes rapidly in the early stages of erosion and different soils show different degrees of impact after varying amounts of time and prior erosion. To maintain yields within 75% of the maximum possible, topsoil loss would need to be <1 cm. Over a 200-year time frame, this suggests that an erosion rate of $5 \times 10^{-5}$ m yr$^{-1}$ or 0.85 t ha$^{-1}$ yr$^{-1}$ would be acceptable (assuming a bulk density of 1.3 t m$^{-3}$ and a soil production rate of 15 μm yr$^{-1}$ or 0.20 t ha$^{-1}$ yr$^{-1}$). This rate is three-times higher than the $T_1$-value (unadjusted for aeolian deposition) based on soil production rate given above. However, if a longer time frame for sustainable agriculture is required, say 1000 years, $T_2$-value becomes 0.32 t ha$^{-1}$ yr$^{-1}$, much closer to the value estimated for $T_1$-value.

Setting erosion values that consider the impact of sediment on aquatic environments ($T_3$-values) requires detailed information on hillslope sediment delivery rates. Currently we do not possess such information. The $^{137}$Cs National Reconnaissance Survey provided such information at only ~200 sites across Australia. Modelled estimates have been made for the Murray-Darling (Lu et al. 2006) and a small sub-catchment of the Burdekin basin (Kinsey-Henderson et al. 2005) but generally RUSLE estimates with a HSDR of 5-10% are able to account for the amount of sediment gauged in rivers. Applying a 10% HSDR uniformly over the continent gives a median of 0.09 t ha$^{-1}$ yr$^{-1}$ or 0.007 mm yr$^{-1}$ and a mean of 0.33 t ha$^{-1}$ yr$^{-1}$ or 0.025 mm yr$^{-1}$. This is close to the average of 0.1 t ha$^{-1}$ yr$^{-1}$ estimated for the Murray-Darling sub-catchments (Lu et al. 2006).

Clearly the selection of $T$-values depends on the endpoint desired. The previous estimate of 0.5-1.0 t ha$^{-1}$ yr$^{-1}$ as a tolerable erosion rate for Australia by Edwards and Zierholz (2000) is generally higher than our $T$-value estimates.
Fig. 11. Areas exceeding T1-values. Two maps are shown using different soil production terms: in (A) $P = 15 \, \mu m \, yr^{-1}$, and in (B) $P$ estimated from pre-European estimates of Lu et al. (2003) scaled by 10% HSDR (see Fig 5B).
4. Estimation of time to critical topsoil loss

The critical time, $T_c$, to erode through a soil profile (time to soil exhaustion) can be calculated using Eq. 1:

$$T_c = \frac{S}{(E - P)}$$

where $S$ is the initial thickness in m, $E$ is the net soil erosion rate in m yr$^{-1}$, and $P$ is the soil production rate in m yr$^{-1}$ (after Montgomery 2007). Here we use the complete A-horizon thickness to represent $S$.

4.1. Application of NLWRA data

Fig. 12 shows continental-scale maps estimating $T_c$ at 1-km resolution using NLWRA erosion data with a 10% HSDR applied to represent net erosion ($E$) (Fig 7B). Maps are shown whereby $T_c$ has been estimated using; (A) the 2001 ASRIS data for soil bulk density and A-horizon thickness; and (B) the 2009 ASRIS data. These different versions of the ASRIS soil maps help represent the spatial uncertainty associated with the soil thickness and bulk density data currently available. Again $P$ is assumed equal to $1.5 \times 10^{-5}$ m yr$^{-1}$ over the whole continent, the denudation rate over the Cenozoic according to Kohn et al. (2002).

The spatial pattern of areas at risk of excessive topsoil loss within decades does not change much regardless of the thickness surfaces, i.e. the patterns in Fig. 15A and 15B are essentially the same. Most the continent shows $T_c$ values that are either greater than 500 years or are indicating no net soil loss (blank areas). Coastal Queensland, the Wet Tropics, Mitchell Plains grasslands, New England Tablelands, and Victoria River basin in the NT are identified at highest risk, all having $T_c$ values less than 500 years. Note that the pattern and mineralogy of clay sediments in the Bonaparte Gulf (Gingele et al. 2001) corroborates the Victoria River basin’s smectitic clay soils as a major source of sediment and thus lends some support for the spatial prediction. Moreover, increased sediment delivery to the Great Barrier Reef since European settlement is documented (Neil et al. 2002; McCulloch et al. 2003). Hillslope erosion is a major erosive process in the Wet Tropics (Hateley 2007) with rates >70 t ha$^{-1}$ yr$^{-1}$ estimated (McKergow et al. 2005), some of the highest amount of average annual soil loss in Australia. Is the spatial pattern of $T_c$ correct? This needs to be investigated further with more on-the-ground measurements of erosion for ground-truthing of predictions.
Fig. 12. Estimates of time to critical topsoil loss (A) using A-horizon thickness from 2001 ASRIS in combination with 10% HSDR value applied to NLWRA gross erosion estimates. Areas in blank are those where soil production is $\leq 15 \, \mu m \, yr^{-1}$.
Fig. 12. (B) Estimates of time to critical topsoil loss using A-horizon thickness from 2009 ASRIS in combination with 10% HSDR value applied to NLWRA gross erosion estimates. Areas in blank are those where soil production is ≤15 μm yr⁻¹ and/or where there is no ASRIS 2009 data.
4.2. Application of $^{137}$Cs-derived erosion data

$T_c$ maps estimated from the $^{137}$Cs-derived erosion estimates are shown in Figs. 13 and 14. Maps generated using (A) median, (B) minimum and (C) maximum values of erosion per pixel are shown. As for NLWRA data, maps are generated using 2001 ASRIS soil thickness data and bulk density maps (Fig. 13) and 2009 ASRIS data (Fig. 14). The median soil loss estimate per pixel for the 100 simulations of $^{137}$Cs-derived net soil distribution map as the denominator in Eq. 1, assuming soil production rate is negligible. $T_c$ estimated thus is less than it would be if soil production rate was subtracted from the net erosion estimates.

Overall, $T_c$ predicted using the median $^{137}$Cs-derived net soil redistribution is far greater than $T_c$ predicted using the NLWRA estimates. Using the median of the $^{137}$Cs-derived net soil redistribution estimates to approximate net erosion losses, we would conclude that Australia’s topsoil can sustain us for many centuries to come except in a few localities in the Pilbara, the Wimmera, and central Queensland.

Clearly there are unresolved contradictions in the $T_c$ results, due to differences in predictions of erosion rates using different methods (differences in exactly what is predicted and at what scale) that require further investigation.

4.3. The effect of reduction in the A-horizon on agricultural productivity

As noted in section 3 above, agricultural productivity is reported to diminish exponentially with loss of organic matter and nutrients in the upper A-horizon (Stocking 2003). Given Australia is drought-prone, reduction in water storage capacity of soil may be the most significant impact on productivity. Thus loss of just a small fraction of the A-horizon (a few cm of soil) may in some regions lead to significant declines in productivity. However, it should also be noted that there are situations in which reduction in the soil A-horizon makes little or no impact on agricultural productivity (e.g., Bakker et al. 2004, Warren 2007). While estimates of the time to critical soil loss ($T_c$), defined here as the time for complete erosion of the soil A-horizon, are in the range 100-500 years for the most highly eroding areas, it is possible that productivity will be significantly affected in time frames of less than 100 years, especially in areas where the A-horizon is thin.
Fig. 13. Estimated time to A-horizon loss using 2001 ASRIS data and (A) median value per pixel for the 100 simulations of $^{137}$Cs-derived net soil redistribution as the denominator in Eq. 1. Areas of no soil loss or net gain are blank and affects mostly SA and NSW, and north Qld. (B) Same calculation using 2001 ASRIS data and minimum value of erosion per pixel for the 100 simulations of $^{137}$Cs-derived net soil redistribution. Most of the continent shows either no soil loss or net gain and is blank. (C) Same calculation using 2001 ASRIS data and maximum value of erosion per pixel for the 100 simulations of $^{137}$Cs-derived net soil redistribution.
Fig. 14. Estimated time to A-horizon loss using 2009 ASRIS data and (A) median value per pixel for the 100 simulations of $^{137}$Cs-derived net soil redistribution. Areas of no soil loss or net gain are blank—this affects mostly SA and Qld. (B) Same calculation using 2009 ASRIS data and maximum value per pixel for the 100 simulations of $^{137}$Cs-derived net soil redistribution. Most of the continent shows either no soil loss or net gain and is blank. (C) Same calculation using 2009 ASRIS data and minimum value per pixel for the 100 simulations of $^{137}$Cs-derived net soil redistribution.
5. Monitoring and evaluating erosion control measures

Beyond estimating what a ‘tolerable’ soil loss might be in a given region, control measures to reduce it need to be assessed for effectiveness and impact on erosion monitored. While the spatial modelling approaches identified above can be useful for identifying sources of sediments and estimating erosion rates quantitatively, they can also be used at a catchment scale to target investment to control erosion by changing land use/land cover and/or rehabilitating riparian zones. The models have been used to identify drivers or environmental conditions that promote erosion and changes in these variables have been simulated under various scenarios and the impact on erosion modelled. Several examples of such applications of spatial modelling studies have been identified (Table 8).

These approaches appear useful for ascertaining the appropriateness of control measures in catchments, where to invest, what to do, and how much to invest. For example, in the Mt. Lofty Ranges in SA, the SedNet model was used to determine that the management action with the best cost/benefit ratio is targeted riparian re-vegetation in hotspots of bank and gully erosion, with the installation of riparian filter strips around water supply storages at risk of sediment deposition as a second priority (Wilkinson et al. 2005). In South-East Queensland, Olley et al. (2006) showed that targeted re-vegetation along 20% of stream channel length could reduce sediment export to Moreton Bay by ~50%.

Whether such approaches could be used to monitor performance against five-year outcomes is unclear given the high spatiotemporal variability of surficial landscape processes, the delayed impact on erosion as measurable from gauging stations, and the uncertainty associated with such gauging measurements. For example, Bartley et al. (2007a) report ± 50% uncertainty associated with the measured suspended sediment load at the Weany Creek catchment outlet between 2003 and 2005.

Most of the examples in Table 8 use water quality as an endpoint for decision-making. Whether the likely topsoil loss leads to reduced productivity (as implied by the expression ‘soil exhaustion’) is another question that needs to be addressed by monitoring crop yields in areas at risk of exceeding tolerable rates of topsoil loss. More studies like those of Littleboy et al. (1992) and Freebairn et al. (2009), that link crop and erosion models, could help address the issue of hillslope erosion’s impact on crop productivity. We have identified areas at risk, where the hillslope erosion currently appears to exceed the pre-European rate (Fig. 12-14), and where such studies should be pursued if an agricultural endpoint is used to define tolerable soil erosion.

Perhaps most important for designing effective policy- and decision-making is the correct identification of the major erosion process(es) and sources of sediment. Control measures for gullying would be different than for hillslope erosion. For example, a major catchment revegetation program in the Ord, designed to slow sedimentation into Lake Argyle, appears to have had little effect because it did not target gullies, the major source of sediment (Wasson et al. 2002).
<table>
<thead>
<tr>
<th>Study</th>
<th>Author(s)&amp; date</th>
<th>Method</th>
<th>Resolution</th>
<th>Meas. Units</th>
<th>Cost:benefit estimated</th>
<th>Error estimated</th>
<th>Other comments</th>
</tr>
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<tbody>
<tr>
<td>Location/ Area covered</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Emerald, Dalby and Gunnedah, Qld</td>
<td>Littleboy et al. (1992)</td>
<td>PERFECT</td>
<td>cm yr⁻¹</td>
<td>No</td>
<td>Yes</td>
<td>range in variation of simulated results; productivity half-life concept; cites several earlier studies</td>
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</tr>
<tr>
<td></td>
<td>Freebairn et al. (2009)</td>
<td>APSIM</td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>state of NSW</td>
<td>Smyth and Young (1998)</td>
<td>USLE</td>
<td>1-10km grids as inputs</td>
<td>t ha⁻¹ yr⁻¹</td>
<td>Yes</td>
<td>for modelled input surfaces only</td>
<td>refers to studies of Aveyard (1983, 1988) on erosion’s impact on productivity</td>
</tr>
<tr>
<td>L. Wingecarribee catchment, NSW</td>
<td>Olley and Deere (2003)</td>
<td>SedNet</td>
<td>25-m DEM, 5-km land cover</td>
<td>t ha⁻¹ yr⁻¹, kt y⁻¹</td>
<td>No</td>
<td>indirect</td>
<td>comparison w radionuclide tracers of sediment sources</td>
</tr>
<tr>
<td>Weany Creek sub-catchment of Burdekin R. basin</td>
<td>Kinsey-Henderson et al. (2005a)</td>
<td>SedNet, grazing scenarios</td>
<td>5-m DEM accurate to better than 50 cm in vertical resolution; C-factor from 2-m pixel re-sampled photo mosaic</td>
<td></td>
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<tr>
<td>Moreton Bay, Qld</td>
<td>Olley et al.</td>
<td>SedNet, ANNEX</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Catchment Description</td>
<td>Study Refs</td>
<td>Methodology</td>
<td>Units</td>
<td>Cost Use</td>
<td>Landscape Parameter</td>
<td>Notes</td>
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</tr>
<tr>
<td>upper Murrumbidgee R. catchment</td>
<td>Wilkinson et al. (2004)</td>
<td>SedNet and RARC*, scenario analysis</td>
<td>No</td>
<td>x</td>
<td>against gauging station</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Goulburn-Broken catchments</td>
<td>Wilkinson et al. (2005a)</td>
<td>SedNet and RARC, scenario analysis</td>
<td>20 m</td>
<td>t ha⁻¹ yr⁻¹, kt yr⁻¹</td>
<td>No</td>
<td>x</td>
<td>against gauging stations</td>
</tr>
<tr>
<td>south Para, Torrens and upper Onkaparinga catchments, Mount Lofty Ranges, SA</td>
<td>Wilkinson et al. (2005b)</td>
<td>SedNet and RARC, scenario analysis</td>
<td>10-m DEM, 20-m C-factor</td>
<td>sediment t ha⁻¹ yr⁻¹, kt yr⁻¹ at outlet</td>
<td>Yes</td>
<td>x</td>
<td>fixed at 50%</td>
</tr>
<tr>
<td>Douglas Shire, Qld</td>
<td>Bohnet et al. (2007)</td>
<td>SedNet, scenario analysis</td>
<td></td>
<td></td>
<td></td>
<td>x</td>
<td></td>
</tr>
<tr>
<td>Nambucca R. catchment, NSW</td>
<td>Higgins et al. (2008)</td>
<td>mathematical programming for spatial optimization</td>
<td>400-m</td>
<td>cost is used as a constraint</td>
<td>x</td>
<td>multi-objective optimization for biodiversity, erosion control (using travel time approximation) and carbon sequestration</td>
<td></td>
</tr>
<tr>
<td>MDB, focused on Goulburn, Namoi, Murrumbidgee, and Balonne catchments</td>
<td>Lu et al. (2004)</td>
<td>SedNet, scenario analysis, genetic algorithm</td>
<td>9 arc sec or ~250 m</td>
<td>t km⁻² yr⁻¹, kt yr⁻¹</td>
<td>Yes, Cost versus sediment reduction curves</td>
<td>x</td>
<td></td>
</tr>
</tbody>
</table>

*Rapid Appraisal of Riparian Condition (RARC)
6. Conclusions

There are still large uncertainties associated with the various surfaces required to calculate time to critical soil loss. While there are some inconsistencies between different soil thickness maps these are small compared to the discrepancies and potential errors in the different erosion maps and in the soil production rate estimates. Overall statistical comparisons of soil thickness datasets suggest that the measurements have a 40% error whereas for some regions of the country there is an order-of-magnitude difference between erosion rate estimates. More research is required therefore to measure and map soil thickness and topsoil erosion rates, to reduce the uncertainty in erosion rate estimates.

In many parts of Australia soil production rates are not negligible compared to contemporary erosion rates and should not be ignored when estimating time to critical soil loss. The proposition that soil production rates be used set tolerable erosion rates may therefore be a good one. However more research is required to establish spatially explicit soil production rates. As a spatially undifferentiated first approximation the average denudation rate over the Cenozoic can be used as a sustainable erosion rate. This rate, $1.5 \times 10^{-5}$ m yr$^{-1}$ (or 0.2 t ha$^{-1}$ yr$^{-1}$, assuming a bulk density of 1.3 t m$^{-3}$), assumes steady state landscape balance; i.e. soil denudation equals soil production. In places where soil gains from aeolian deposition are important, e.g. the western slopes of the Great Dividing Range, this rate may be doubled to $3.0 \times 10^{-5}$ m yr$^{-1}$ (0.4 t ha$^{-1}$ yr$^{-1}$).

Estimates of the time to critical soil loss ($T_c$), defined here as the time for complete erosion of the soil A-horizon, are in the range 100-500 years for the most highly eroding areas. Whereas our topsoil resources may only sustain us for a few centuries more if the NLWRA hillslope net erosion estimates scaled by 10% HSDR is used, using the $^{137}$Cs-derived net soil redistribution surface to estimate topsoil losses indicates that hillslope erosion is less worrisome over most of Australia.

Some studies have noted an exponential reduction in agricultural productivity with loss of topsoil, an important issue if soil productivity is considered as the major criterion for agricultural sustainability. If the NLWRA erosion estimates and $T_c$ values are accepted, loss of just a small fraction of the A-horizon (a few cm of soil) may in some regions lead to significant declines in soil productivity over time frames of less than 100 years.

Contradictory conclusions can be drawn using different approaches to mapping erosion when estimating time to critical soil loss and areas at risk. Using maps of erosion modelled with RUSLE/SOILLOSS or $^{137}$Cs inventories, different areas are identified as being at risk of exceeding tolerable soil losses. The RUSLE/SOILLOSS model currently has wide acceptance both in Australia and internationally. The $^{137}$Cs-based map provides a preliminary estimate across Australia of net soil redistribution (from mid-1950s to early 1990s) and, because it relies on relatively few data points and only rainfall and soil-type as co-variates, has large areas of relatively high uncertainty. While the $^{137}$Cs net soil redistribution pattern does not show the expected erosion trends in some regions, it appears to have identified distinct regions of high erosion (and net gain, possibly by wind-borne deposition), that were missed by RUSLE/SOILLOSS. If we rely on the predictions from RUSLE/SOILLOSS, northern Australia and coastal Queensland and NSW are most at risk of exceeding tolerable soil losses, but according to the $^{137}$Cs-based map, south-west WA, western Victoria, and western Tasmania are most threatened. A composite map seems the best way at present to estimate potential areas at risk (Fig. 15).
7. Recommendations for further work

1. **Integrated studies of soil formation and erosion: geomorphometric, geochemical, and modelling approaches in key catchments across Australia**

We need to improve our understanding of the processes that drive landscape evolution. Fieldwork to collect new measurements of soil depths and erosion rates across Australia is required as there are major discrepancies between results obtained by modelling and by measurement, especially for erosion rates. Priority areas would be those where agriculture is important and there are large discrepancies in the estimated time to critical soil loss maps using different denominators in Eq. 1, i.e. areas where time to critical soil loss is <100 years in Fig. 12 to Fig. 14. The Wimmera-Avon River basins in Victoria; Namoi R. and Hunter River basins in New South Wales; upper Mitchell R., Normanby R., Endeavour R., lower Burdekin R., Don R., Fitzroy R., upper Condamine River in Queensland; Ord R., Keep R. and Victoria River, Northern Territory; and Port Hedland Coast in WA would be good candidates for future work (Fig. 15). Current sediment transport studies in the Mitchell River and Daly River basins should be broadened to include soil thickness measurements and soil production rate estimates. Measurement of total elemental chemistry on water samples would also provide an additional means of estimating denudation rate by geochemical mass balance.

By integrating point-specific and hillslope transect studies of soil thickness, weathering rates, and soil formation with cosmogenic nuclide and other geochemical tracers studies of sediment provenance and transport across catchments we can gain a more complete understanding of the geomorphological processes active across Australia. Developing an improved, empirically based understanding of soil development and sediment production on slopes in Australian landscapes will in turn allow us to develop more realistic process-based soil formation, soil thickness, and hillslope sediment production models. Quantitative formulation of what controls soil thickness and sediment transport relationships remain two of the most significant gaps in our understanding of Earth surface processes.

Integrated studies of soil formation and sediment transport on the western slopes of NSW should also quantify the influx of wind-borne deposition on vertical accretion of soil that could be offsetting water-borne hillslope erosion (and replenishing nutrients). The soil redistribution map based on the $^{137}$Cs measurements from the National Reconnaissance Survey shows that even in very recent time (30-40 years prior to 1990) soil accretion, most likely due to wind-borne deposition, is important on the western slopes of the Great Dividing Range (Fig. 9A). There are a number of geochemical sediment tracing and finger-printing techniques that are used to identify the provenance and track the transport of water-borne sediments (e.g. Hancock and Pietsch 2008) and of wind-borne deposits (Gatehouse et al. 2001; De Deckker et al. 2008; Greene et al. 2009). These biogeochemical tracers could be used also to investigate the importance of aeolian material in soils and distinguishing between sources of soil parent material.
Fig. 15. Map showing river basins and areas identified as having a soil A-horizon life span of < 500 years. Up to 100,000 km² could be at risk. This map of time to critical A-horizon loss uses soil thickness data from the 2009 and 2001 Australian Soil Resource Information System. The erosion data are a composite of RUSLE predictions scaled by a 10% HSDR and ¹³⁷Cs-based maps. A soil production rate of 1.5 x 10⁻⁶ m yr⁻¹ has been assumed over the whole continent.
2. Improved erosion modelling and validation by field ground-truthing

Discrepancies between predictions of erosion rates using different methods are up to an order of magnitude. We need to determine the most accurate method of estimating erosion rate. SedNet is now a well-established modelling approach for hillslope erosion in Australia and we should determine whether using higher resolution data inputs gives results that better match with sediment tracing and gauging data across the continent. It is recommended that:

a) SedNet is run with high resolution DEM (a 30-m DEM from corrected SRTM will soon be available) and land cover data for all coastal catchments in eastern and southern Australia, in particular for the Hunter, Wimmera-Avon, Ord, Keep, upper Mitchell, lower Burdekin and other catchments where hillslope erosion under current land use is potentially much greater than what it was in pre-European time. This would have the added benefit of also estimating bank and gully erosion which are generally more of a problem for water quality than hillslope erosion.

b) Newer, higher resolution input data layers would produce more accurate predictions using RUSLE/SOILLOSS: The one arc sec (~30-m) SRTM DEM is a large improvement on contour-derived DEMs for the slope factor (S-factor) in RUSLE (Kinsey-Henderson, 2007). Thus it appears that high resolution DEMs produce predictions that better match field-based measurements than the relatively coarse 250-m DEM used in the NLWRA. The cover-factor spatial time series could be improved by using MODIS, LandSAT and/or AussieGrass data and by accounting for dead grass, which was neglected during the NLWRA. Soil erodibility (K-factor) also could be updated with 2009 ASRIS data.

c) Develop a spatially explicit hillslope delivery ratio (HSDR) for the whole continent. Currently one exists only for the Murray-Darling basin and a small portion of the Burdekin River basin. Acquiring a spatially variable HSDR is essential to minimise uncertainty in the setting of T3-values, i.e. tolerable soil losses with aquatic habitat quality as the endpoint.

d) Compare the new modelling results with those from a new $^{137}\text{Cs}$ survey (see Recommendation 3 below), SIBERIA (e.g. Hancock et al. 2008) and other spatially explicit erosion models.

e) Discrepancies exist in conclusions that can be drawn using different approaches to mapping erosion when estimating time to critical soil loss. Using maps of erosion modelled with RUSLE or $^{137}\text{Cs}$ inventories, different areas are identified as being at risk of exceeding tolerable soil losses. Areas at risk need to be ascertained more definitively.

f) An alternative, independent method of assessment of spatial erosion patterns would be to link airborne gamma radiometrics from Geoscience Australia with point measurements on sediments to produce better maps of sedimentation patterns. Currently used sediment tracing technology is expensive and provides point data for a few locations rather than the spatially distributed information that is really needed. Combining gamma radiometric coverage with sediment tracing data where it has been collected already would allow verification of regional-scale process models and to calibrate regional-scale sediment transport models. Wilford et al. (1997) and Pickup and Marks (2000, 2001) have had some success in identifying pedogenetic and transport-related patterns in a landscape dominated by geological differences and weathering effects that are sufficiently encouraging to apply the technique in other areas now that gamma radiometric coverage across Australia is almost complete.

g) Fieldwork to collect new measurements and validate erosion rate predictions obtained by modelling is essential.

3. Further $^{137}\text{Cs}$ survey over Australia

A new $^{137}\text{Cs}$ inventory survey is recommended, linking terrestrial and aquatic environments to verify how much surface soil is moving and delivered into streams. There is a limited time frame
for this work since the $^{137}\text{Cs}$ technique will continue to lose sensitivity over the coming decades as $^{137}\text{Cs}$ decays. New samples, with $^{137}\text{Cs}$ inventories re-calculated to 1990, can be combined with those collected during the National Reconnaissance Survey by Loughran et al. (2004). More $^{137}\text{Cs}$ measurements should improve the variogram (Fig. 4 in Appendix 1) and reduce the uncertainty of the $^{137}\text{Cs}$ inventories and predicted net soil redistribution surface (Fig. 9 above). Northern Australia and Queensland in particular need to be sampled more to rationalise the difference in hillslope erosion rates predicted by methods based on the $^{137}\text{Cs}$ National Reconnaissance Survey of Loughran et al. (2004) and RUSLE/SOILLOSS used by Lu et al. (2003). More empirical models for northern Australia should also be developed to improve the accuracy of the conversion of $^{137}\text{Cs}$ loss-gain (Bq m$^{-2}$) to soil loss-gain (t ha$^{-1}$ yr$^{-1}$).

4. The effect of surface soil erosion on organic carbon and nutrient cycling

Decomposed organic matter attached to soil particles is moved with the soil as it is eroded, transported and deposited. Thus erosion can result in significant changes in the soil organic carbon (SOC) cycle, with depletion of SOC in the eroded profile, oxidation of SOC during transport and changes in soil-atmosphere C exchange upon deposition and burial. All these processes can alter the soil-atmosphere CO$_2$ flux. Studies have shown that soil erosion may represent a significant C source to the atmosphere (e.g. Lal et al. 2003), however more recent work indicates a small net sink (Van Oost et al. 2007). The latter authors made global estimates of SOC fluxes from tilled (cultivated) and uncultivated soils using small-scale $^{137}\text{Cs}$ and SOC budgets for 10 different regions, then extrapolating to a global scale using mechanistic models with controlling factors of land use, topography, climate and soils. In many regions erosion from cultivated soils dominates on a continental scale (e.g. Europe, North America and West Asia), but in other areas, such as Australia, erosion from uncultivated (grazed) hillslopes dominates. Van Oost et al. (2007) estimate that on a continental scale, erosion from grazed hillslopes in Australia exceeds erosion from cultivated soils by a factor of 20, contrary to our findings using $^{137}\text{Cs}$–based soil redistribution.

Improved estimates SOC fluxes for Australia should be obtained using the best available erosion map together with improved measurements of $^{137}\text{Cs}$ and SOC for uncultivated soils. In cultivated soils a strong relationship is expected between SOC and $^{137}\text{Cs}$ due to the mixed upper plough layer of these soils (~20 cm depth), simplifying soil redistribution and lateral SOC flux estimates. But in uncultivated soils, especially where the soil A-horizon is thick (>20 cm), $^{137}\text{Cs}$ and SOC can show very different soil depth profiles. Thus soil redistribution and deposition rates based on $^{137}\text{Cs}$ patterns may not necessarily reflect SOC. This is especially true where rilling and subsoil erosion is significant, and where soils have a well-developed A-horizon. In these situations lateral SOC fluxes would be underestimated using $^{137}\text{Cs}$ estimates of erosion.

The relationship between $^{137}\text{Cs}$ and SOC should therefore be established for uncultivated Australian soils for various soil types and regions. Such a relationship will allow better estimates of atmospheric-soil SOC budgets.
8. References


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9. Appendix 1

*Data and methods for estimating $^{137}$Cs-derived net (ca. 40 year) soil redistribution at unsampled location*

1. Data

1.1 Archived Australian rainfall data

An archive of Australian annual monthly rainfall data was constructed from ground-based observational data (Jeffrey et al. 2001). In that study the observations were used to make estimates on a regular 0.05° (approx. 5 km) grid using ordinary kriging. The per-point monthly data for each year between 1954 and 1990 were summed before the mean annual rainfall was calculated for each point (Fig. 1). This layer was used to improve the estimation of the $^{137}$Cs reference inventory across Australia.

![Mean annual rainfall (mm) between 1954 and 1990 for Australia using 0.05° (approx. 5 km) grid.](image)

1.2 Land-use data

The Bureau of Rural Sciences provides a series of land use maps of Australia. We used here the 1992/93 data which is closest in time to the national $^{137}$Cs reconnaissance survey. The land use data includes non-agricultural uses based on existing digital maps covering four themes: protected areas, topographic features, tenure and forest. 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 (Bureau of Rural Sciences, 2006). The data were supplied at a 0.01° grid size with geographical coordinates (GDA94). The data were re-sampled to a 0.05° degree grid for...
compatibility with the rainfall data and attribute values were associated with every grid point (Fig. 2a).

Fig. 2. Land-use for Australia (a) from the Bureau of Rural Sciences likelihood maps. The first 13 classes shown are used in subsequent analyses and represent the main attributes of the Australian Land use and Management Classification version 5 (see Table 1). Reclassified (b) Australian land use (1992/93) used to determine which model to use in the $^{137}$Cs calibration to soil redistribution (1=never cultivated; 2=cultivated).
The summary map was used here and it shows the non-agricultural land uses and a likely arrangement of the agricultural land uses based on probability maps. The summary map provides an integer grid with a value attribute table with attributes defining the agricultural commodity group, irrigation status and land use according to the Australian Land Use and Management Classification (ALUMC), Version 5 (Table 1).

Table 1. Values and meaning of the Australian Land Use and Management Classification (ALUMC), Version 5.

<table>
<thead>
<tr>
<th>Values</th>
<th>Attribute</th>
<th>Meaning</th>
</tr>
</thead>
<tbody>
<tr>
<td>110-117</td>
<td>1</td>
<td>Nature conservation</td>
</tr>
<tr>
<td>120-125</td>
<td>2</td>
<td>Other protected areas including indigenous uses</td>
</tr>
<tr>
<td>130-134</td>
<td>3</td>
<td>Minimal use</td>
</tr>
<tr>
<td>200, 210-210</td>
<td>4</td>
<td>Grazing natural vegetation</td>
</tr>
<tr>
<td>220-222</td>
<td>5</td>
<td>Production Forestry</td>
</tr>
<tr>
<td>310-314, 410-414</td>
<td>6</td>
<td>Plantation Forestry</td>
</tr>
<tr>
<td>300, 320-325</td>
<td>7</td>
<td>Grazing modified pastures</td>
</tr>
<tr>
<td>330-338</td>
<td>8</td>
<td>Dryland cropping</td>
</tr>
<tr>
<td>340-354</td>
<td>9</td>
<td>Dryland horticulture</td>
</tr>
<tr>
<td>400, 420-438</td>
<td>10</td>
<td>Irrigated pastures and cropping</td>
</tr>
<tr>
<td>440-454</td>
<td>11</td>
<td>Irrigated horticulture</td>
</tr>
<tr>
<td>500, 510-526</td>
<td>12</td>
<td>Intensive animal and plant production</td>
</tr>
<tr>
<td>542</td>
<td>13</td>
<td>Rural residential</td>
</tr>
<tr>
<td>530-541, 550-575</td>
<td>14</td>
<td>Urban intensive uses</td>
</tr>
<tr>
<td>580-595</td>
<td>15</td>
<td>Mining and Waste</td>
</tr>
<tr>
<td>600, 610-663</td>
<td>16</td>
<td>Water</td>
</tr>
</tbody>
</table>

For use with the $^{137}$Cs-derived net soil redistribution models, the classification was aggregated to land which had never been cultivated (classes ≤ 220) and that which had been used for cultivation (>220 classes). In common with Loughran et al. (2004) regardless whether land was cultivated continuously or in rotation the Australian Empirical Model (AEM) for cultivated land was applied. This is expected to over-estimate the net soil redistribution at locations cultivated in rotation. However, information necessary to resolve this issue was not available for the modelling. Intensive land uses (classes > 530) and Water (classes > 600) were excluded from the analysis. Fig. 2b shows the map which was used to determine which model to use in the later $^{137}$Cs conversion to soil redistribution.
1.3 Australian Soil Classification

Much of Australia is formed from old, deeply weathered soils. These old surfaces are being dissected and new soils have developed in the weathered material and on unweathered parent rocks exposed by erosion (Isbell et al., 1997). Past and present climates have interacted with the soil to produce environments in which the modern soil pattern has developed (Beckmann, 1983). This rejuvenation of Australian landscapes in terms of soil development makes for a complex genetic history. Nevertheless, the Australian Soil Classification provides a hierarchical, multi-categorical general purpose scheme with classes defined on the basis of diagnostic horizons or materials and their arrangement in vertical sequence as seen in exposed profiles (Isbell, 1996).

The 13 Soil Orders (excluding Anthroposols) are used here to provide data which relate to contemporary medium-term soil erosion and deposition across the continent. The rationale for its use is that it provided information on the relative erodibility of the soils. The categorical nature of these data makes that relationship more complicated to establish than with continuous rainfall data. However, the reduction of the data to indicators and the Bayesian approach ensures that different types of data may be combined to improve the estimation procedure. In this analysis Soil Orders were obtained from the Atlas of Australian Soils, where the dominant soil type for each mapping unit had been interpreted from soil-landscape descriptions and Northcote Principal Profile Forms (Isbell et al, 1997). This dataset was converted to a 5 km x 5 km point raster with an origin that aligned with other datasets in this study (Fig. 3). The conversion process selected the mapping unit that fell in the centre point of the pixel, although a majority filter may have provided a more appropriate point value.

Fig. 3. Distribution of the orders of the Australian Soil Classification

3. Methods

The methodology is reduced to the following key stages:

1. model the median indicator variogram of fallout and soil redistributed $^{137}$Cs
2. compute multiple realisations (maps) of the reference $^{137}\text{Cs}$ inventory and the $^{137}\text{Cs}$ inventory associated with soil redistribution using rainfall and soil type data, respectively;

3. calculate the $^{137}\text{Cs}$ loss / gain relative to the $^{137}\text{Cs}$ reference inventory and its uncertainty by sampling, without replacement, the realisations

4. estimate the net soil redistribution rates using the Australian Empirical Models between $^{137}\text{Cs}$ and soil redistribution and its spatial uncertainty by performing a bootstrap (re-sampling with replacement) of the regression models;

5. calculate net soil redistribution for Australia as a whole and separated between land-use in 1992/93 and produce a map of the probability of exceeding a ‘tolerable soil erosion’ rate.

3.1 Modelling the spatial variation of fallout and soil $^{137}\text{Cs}$

There is a basis for the $^{137}\text{Cs}$ distribution to be anisotropic, particularly over small regions and in directions of preferential soil movement (e.g., wind erosion). However, there were insufficient data to model reliably the anisotropic spatial variation (Webster and Oliver, 1992). Furthermore, it could be argued that the spatial variation in $^{137}\text{Cs}$ is closely related to some systematic change across the continent. There was no evidence for this type of underlying trend in the variograms and so it was assumed not to exist. Consequently, omni-directional sample indicator variograms represented the average variation in all directions and were calculated for the $^{137}\text{Cs}$ reference inventory and the $^{137}\text{Cs}$ inventory data (Fig. 4). Although sample variograms from indicator data are usually well-behaved and do not suffer from the adverse effects of erratic outlier values (Isaaks and Srivastava, 1989), they are not well-defined at the margins or extremes of a distribution because they depend on the spatial distribution of only a few pairs of indicator data (Goovaerts, 1997). Nevertheless, the magnitude and spatial connectivity of extremes in soil erosion patterns are particularly important to spatial simulations and so too are the number of thresholds used in the simulation. The data are equally distributed at the median and the variogram at thresholds other than the median may be inferred using the mosaic model (Journel, 1984) defined above. The need for calculation and model-fitting of variograms for multiple thresholds at the margins of the distribution meant that the mosaic model offered a valuable compromise in providing reliable variograms at the expense of restricted spatial structural information at each threshold. Furthermore, the median approximation also reduced the two major causes of order relations problems (Deutsch and Journel, 1998): the occurrence of negative kriging weights and the lack of data in some classes of threshold values. A comprehensive explanation of order relation deviations and methods for correcting them is beyond the scope of this paper and is available elsewhere (e.g., Goovaerts, 1997 p. 319).

The $^{137}\text{Cs}$ reference inventory and $^{137}\text{Cs}$ inventory values cumulative distribution functions were calculated for all locations. The $K=7$ quantiles (0.05, 0.1, 0.25, 0.5, 0.75, 0.9 and 0.95) were established and also used in the simulations. The percentiles were used to transform the $^{137}\text{Cs}$ values into indicator variables. The kriging estimates and therefore the stochastic indicator simulations are largely controlled by the first few lags (Webster and Oliver, 2001). Since the modelling of direct indicator sample variograms of these first few lags is problematic because of the erratic nature of few sparse data it is more consistent and reliable to use the mosaic model to infer the models. No constraint intervals were used in the analysis. The median indicator omni-directional variogram of $^{137}\text{Cs}$ reference inventory and $^{137}\text{Cs}$ inventory was computed to under half (2500 km) the maximum separation distance between soil $^{137}\text{Cs}$ sample locations. These variograms were fitted, using weighted least squares, with several models authorised for kriging (linear, spherical, exponential, power, and circular). The models that fitted best, in the least-squares sense, were selected using the square root of the mean squared difference between the model and the observations (RMSE) and the Akaike Information Criterion (AIC) was used to judge the level of complexity in the nesting of models. The model judged to fit the sample variograms best, in the least squares sense, was the spherical model and its is described below in its isotropic form (Eq. 1) where the lag $h$
becomes the scalar $h=|h|$ (Webster and Oliver, 1990). The quantity $c$ is the sill variance and $a$ is the range of the bounded models:

Spherical:  
\[ \gamma(h) = c \left\{ 1.5 \left( \frac{h}{a} \right) - 0.5 \left( \frac{h}{a} \right)^3 \right\} \text{ for } h \leq a \]  
\[ \gamma(h) = c \quad \text{for } h > a \]  

(Eq. 1)

The parameters of these models describe the structure of spatial variation (Chappell and Oliver, 1997; Chappell and Agnew, 2008) and were used in the co-simulations.

3.2 Sequential indicator co-simulations of $^{137}$Cs (spatial uncertainty)

Instead of a map of local best estimates, stochastic simulation generates a map or a realization of $z$ values \( \{z^{(l)}(u), u \in A\} \) where $l$ denotes the $l$th realization which reproduces statistics pertinent to the problem e.g., data values are honoured at their locations \( z^{(l)}(u) = z(u) \) \( \forall u = u_i, i = 1, \ldots, n \) when the realization is said to be conditional, the histogram of simulated values may reproduce the sample histogram and the set of indicator variograms may be reproduced by variograms of the simulated values (Deutsch and Journel, 1998). Unlike sequential Gaussian simulation the indicator approach is not dependent on an underlying distribution and allows one to account for class-specific patterns of spatial continuity using different indicator variograms (Goovaerts, 1997). The sequential indicator simulation (SIS) approach of a single continuous attribute $z$ at $N$ grid nodes $u_j$ conditional on the $z$-data, proceeds by discretising the range of variation of $z$ into $K+1$ classes (Eq. 10). A random path is defined that visits each node of the grid only once. At each node (Goovaerts, 1997):

1. Determine the $u'$ of $K$ conditional cumulative distribution function (ccdf) values \( F(u'; z_i[(n)]) \) using median indicator kriging. The conditioning information consists of indicator transforms of neighbouring original $z$-data and previously simulated $z$-values;

2. Correct for any order relation deviations and build a complete ccdf model \( F(u'; z_i[(n)]) \), \( \forall z \) using the median approximation indicator kriging estimation algorithm;

3. Draw a simulated value $z^{(l)}(u')$ from the ccdf;

4. Add the simulated value to the conditioning data set;

5. Proceed to the next node along the random path and repeat steps 1 to 4.

This process is repeated with different random paths for each realisation \( \{z^{(l)}(u'), j=1, \ldots, N\} \), $l \neq l$. Sequential indicator co-simulation was used here to generate 100 realisations separately of the $^{137}$Cs reference inventory data using the long-term mean annual rainfall data and $^{137}$Cs inventory data using the Australian Soil Classification data. This co-simulation approach ensures that the values of the primary data are honoured and that approximately reproduced were the sample cumulative distribution function and the variogram models for the five thresholds using the median indicator approximation (Deutsch and Journel, 1998). The simulation algorithm’s realisations should approximately represent the statistics modeled from the data and theoretically the traditional variogram may be reproduced using the median indicator approximation (Deutsch and Journel, 1998). However, the model statistics are inferred from observed $^{137}$Cs sample statistics that are uncertain because of limited sample size and measurement error. In this case, the exact reproduction of the model statistics by each simulated realization is not desirable. The Australian continent was discretised into a grid with 572721 nodes (841 x 681) using a grid-spacing of 5 km (in both x- and y-directions) which was approximately 0.05° in latitude and longitude. The estimates were made to coincide with the nodes of the rainfall grid. The 100 realisations were post-processed by calculating the per-point average and variance to summarise the uncertainty information.
Fig. 4. Omnidirectional variograms of (a) $^{137}$Cs reference inventory, (b) $^{137}$Cs inventory and the respective median indicators (c & d).
3.3 Compounding uncertainty between 137Cs maps and the soil redistribution calibration model

The realisations for the 137Cs reference inventory and 137Cs inventory provide an assessment of uncertainty in the estimates over space. The realisations are equally likely and consequently each of these sources of spatial uncertainty were combined to form the percentage 137Cs difference relative to the reference inventory by random selection of a realisation without replacement. That spatial uncertainty needed to be considered when using the calibration models that relate percentage 137Cs difference (X) to soil redistribution (Y) in equations 1 and 2. A bootstrap procedure (Efron and Tibshirani, 1986) was selected to provide the uncertainty in the calibration model due to variation in the underlying data (Elliott et al., 1990). To undertake the bootstrap procedure the original data was required. Unfortunately, the data could not be obtained and instead they were digitised from Elliott et al., (1990; figure 1 and figure 2). The digitised data were corrected following Lang (1992) and treated as original data for the purposes of this bootstrap procedure. With replacement, 100 random samples of the data were selected and a log-linear regression equation was calculated. Each bootstrap outcome produced values for each of two regression parameters. In this procedure a particular datum from the original set could appear multiple times in a given bootstrap sample. This means that more or less of the original data are used and therefore different configurations of the data (in property space) were included in the regression analysis. A formal description of the bootstrap follows:

(i) the calibration data set \( Z_N \) was randomly sampled with replacement so that \( Z_N = (z_1, z_2, \ldots, z_n) \) where \( z_n = (x_n, y_n) \) and the size of the data is denoted by \( N \).

(ii) This was done \( b \) times, \( b = 1, 2, \ldots, B \), producing \( B \) bootstrap data sets, where each of these bootstrap samples (or realisations of the original population), \( Z^b_N \), is the same size as the original. In this case \( N = 100 \) and \( B = 30 \).
1. Sequential indicator (median approximation) simulation and co-simulation of 137Cs reference values

The per-point median (a) and interquartile range (b) of the sequential indicator co-simulations for 137Cs reference values using mean annual rainfall as the secondary variable is shown in figure 1. The per-point mean map (Fig. 1a) shows no patches, broad areas of similarly high values along the south-eastern and south-western coasts and a broad band from the Australian Bight to the north-eastern Peninsula of 137Cs reference inventory values of <50 mBq cm⁻². The per-point interquartile range simulation map using rainfall as a secondary variable (Fig. 1b) shows a much less erratic pattern and much darker tones across Australia indicating much smaller values than the previous map. There appear to be some artefacts in the co-simulation maps which coincide with the thresholds used to discretise the rainfall cdf and manifest as rapid changes around particular isohyets. They were deemed to have an insignificant impact on the results and were ignored.
Fig. 1  Per-point median (mBq cm\(^{-2}\)) (a) and interquartile range (mBq cm\(^{-2}\)) (b) for the \(^{137}\)Cs reference inventory realisations of sequential indicator (median approximation) co-simulation using mean annual rainfall as a secondary variable. The black dots represent the location of the \(^{137}\)Cs reference inventory measurements.

The performance of the co-simulations is shown in the q-q plot (Fig. 2). The q-q plot displays the quantiles of a distribution’s data against the same quantiles of another distribution. The quantiles of a distribution are the percentiles when the distribution is accumulated e.g., the 10%, 25%, 50% of a cumulative frequency plot of the measured \(^{137}\)Cs data would return measured values of that distribution. Similarly, if the distribution of one realisation were accumulated and those same percentiles were selected it would return the values of the predicted \(^{137}\)Cs. These returned values for the measured and predicted \(^{137}\)Cs are plotted in the q-qplot for the same percentiles. If many percentiles are selected to represent the entire distribution and they plot on an approximately straight line then the predictions would be considered to consistently match the measured values. Furthermore, if that approximately straight line follows the 1:1 line between the axes then the predictions can be considered to be consistent and without bias (i.e., no larger or smaller values relative to the measured values).

The quantiles of the measured \(^{137}\)Cs reference values are plotted against estimates from the 100 realisations. Each line of symbols (not differentiated) represents a realisation. There is good correspondence between these quantiles since they plot approximately along a straight line slightly below the 1:1 line. This indicates that the simulations have faithfully reproduced the characteristics of the sample distribution. However, the realisations appear to underestimate the \(^{137}\)Cs reference inventory values relative to the measurements. This bias may cause a tendency to reduce \(^{137}\)Cs loss and enhance \(^{137}\)Cs gain in the overall process, although the extent to which this will occur is dependent on all the other stages of the
methodology. The straight lines represent the quantiles of the standard Normal distribution. These indicate that the distributions of the realisations are approximately Normal.

Fig. 2  A q-q plot of the $^{137}$Cs reference inventory realisations and the sample values.

2. Sequential indicator (median approximation) co-simulation of $^{137}$Cs inventory

The simulations of $^{137}$Cs inventory were performed using the Australia Soil Classification. The per-point median of the realisations (Fig. 3a) shows a similar pattern to that of the $^{137}$Cs reference inventory. However, the majority of Australia is covered in a much darker tone than that of the $^{137}$Cs reference inventory indicating smaller values. Large values of $^{137}$Cs inventory are found along the south-eastern and south-western coastal regions and those large values extend inland in south-eastern Australia. The per-point variance of the realisations (Fig. 3b) is speckled indicating small scale variation, due to the slightly larger nugget variance in the $^{137}$Cs inventory variogram in comparison to that of the $^{137}$Cs reference inventory variogram. The light tones indicate small variance. Areas of large per-point variance are generally associated with the large $^{137}$Cs values. The reason for this coincidence between large per-point median and variance values of $^{137}$Cs is evident from Fig. 4. Uncertainty in the q-q plot (Fig. 4) increases with $^{137}$Cs values which are greater than around 75 mBq cm$^{-2}$. The performance of the simulations using the soil classification is shown in the q-q plot (Fig. 4). There is good correspondence between the simulations and the 1:1 line which indicates that there is little bias in the estimates. This indicates that the simulations have faithfully reproduced the characteristics of the $^{137}$Cs inventory sample distribution.
Fig. 3. Per-point median (mBq cm$^{-2}$; a) and interquartile range (mBq cm$^{-2}$) (b) for the $^{137}$Cs inventory realisations of sequential indicator (median approximation) co-simulations using mean annual rainfall as a secondary variable. The dots represent the location of the $^{137}$Cs inventory measurements.
3. Difference between $^{137}$Cs reference inventory and $^{137}$Cs inventory values

The percentage loss or gain of $^{137}$Cs includes the uncertainty in the $^{137}$Cs reference inventory and $^{137}$Cs inventory realisations. The cumulative distribution function of the difference values is shown in Fig. 5. These results show that between 60% and 80% of Australia has lost $^{137}$Cs, between 20% and 40% of Australia has gained $^{137}$Cs and that around 20% of Australia has neither lost nor gained $^{137}$Cs. The uncertainty in these estimates is derived from the differences in the $^{137}$Cs reference inventory realisations and how they have been combined randomly with the $^{137}$Cs inventory realisations.
Fig. 5. Cumulative distribution function of the percentage difference between $^{137}\text{Cs}$ inventory and $^{137}\text{Cs}$ reference inventory values relative to the latter.

The map of per-point median difference (%) shows the areas across Australia which have lost $^{137}\text{Cs}$, relative to the reference inventory (Fig. 6a). The main areas are concentrated in southeastern Australia, the coastal region of Western Australia and part of the Northern Territory and Queensland. Most $^{137}\text{Cs}$ has been gained in South Australia and western New South Wales. The per-point variance map (Fig. 6b) shows that the uncertainty in these estimates is an order of magnitude larger than those associated with either the $^{137}\text{Cs}$ reference inventory or $^{137}\text{Cs}$ inventory values. The uncertainty appears greatest in areas where the percentage $^{137}\text{Cs}$ gain is greatest.
Fig. 6. Per-point median (a) and interquartile range of the difference (%) between $^{137}$Cs reference inventory and $^{137}$Cs inventory realisations of sequential indicator (median approximation) co-simulations.
The performance of the estimates (with model uncertainty) is shown in the q-q plot (Fig. 7) of the soil erosion realisations compared with the estimates made at the sample locations. Overall there is good correspondence between the simulations and the soil redistribution rate based on the samples alone. There is a small bias in the realisations relative to the sample estimates since the former fall above the 1:1 line. The most striking feature of this q-q plot is the increased range of the realisations relative to the sample data. This large uncertainty is due to the combination of spatial uncertainty and model uncertainty in the bootstrap procedure.

Fig. 7. A q-q plot of measured and estimated $^{137}$Cs-derived net soil redistribution (t ha$^{-1}$ yr$^{-1}$).
Additional References


