

Invited Review

Impact of historical land use and soil management change on soil erosion and agricultural sustainability during the Anthropocene



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ARTICLE INFO

Article history:

Received 2 February 2016

Received in revised form 5 December 2016

Accepted 3 January 2017

Available online 6 January 2017

Keywords:

Agricultural sustainability
Soil erosion
Land use change
Soil management
Modelling
Anthropocene

ABSTRACT

Since prehistoric times, farmers have faced the challenge of balancing the demand for increasing food production from existing soil resources with conservation of these resources. Land use change from natural vegetation to agricultural land and intensification of agricultural soil management are closely linked to increased rates of soil erosion. This review analyzed and quantified the effects of changes in past land use and agricultural soil management on soil erosion. At a global scale, the period of the first significant land use change closely corresponds to a first wave of soil erosion. Equally important, however, are changes in past soil management. As shown by numerous case studies, changes in management under the same land use can convert sustainable agroecosystems into highly degraded systems. As soil erosion rates, soil profile truncation, agricultural yield, and biomass production are closely related, considering the interactions and feedback effects is important when modelling this system. This paper shows how modelling the dynamics of past soil erosion and agricultural sustainability raises similar challenges to those of quantifying future changes in climate or agricultural systems.

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1. Introduction

Soil erosion is a serious threat to global agricultural sustainability because soil resources are finite on a human time scale. Sustainable agriculture depends, among many other issues, on how efficiently it can use the natural resources, specifically soil and water. In a recent review, [Sposito \(2013\)](#) explained how man's use

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of soil and natural resources in their quest to face the food demand of the world population approaches the limits of three main constraints: land use, water use, and ability of agriculture to produce crops. Steffen et al. (2015) recognized that a “planetary boundary” limited the first constraint, the expansion of crop lands or the surface area of land used. Then, the second constraint, water use is limited: while rainfall or green water availability is relatively constant, additional use of blue water, or freshwater derived from surface or groundwater sources, was calculated by Hoekstra et al. (2012) to be limited to 20% of the global annual runoff. The final and third constraint, the ability of agriculture to increase or maintain crop yields, is continuously threatened and reduced by soil erosion and degradation. While better agricultural management, such as improved crop varieties or mineral fertilizer, has increased crop yields spectacularly over the past century, during the same period erosion rates have increased accordingly (Mazoyer and Roudart, 1997; Tilman, 1998). Such crop inputs are becoming increasingly scarce, and might be depleted in 50–100 years (Cordell et al., 2009). Access to these crop inputs is unequal and already difficult in less-developed countries, where the erosion of the soil resource potential is felt more directly. Soil erosion by water is therefore currently considered as one of the most significant soil degradation processes globally (Pimentel et al., 1995). However, to acquire a better understanding of the main current drivers and their effects, soil erosion needs to be interpreted and analyzed in its historical context. This review aims to give an overview of the effects of past land use and soil management changes on soil erosion and assess its effects on agricultural sustainability, focusing on the central role of soil resources.

As early as in Neolithic times (Meybeck and Vörösmarty, 2005), land use change has resulted in catastrophic episodes of soil erosion in many areas of the world (Bork and Lang, 2003). Over the last century however, intensification of land management, especially since the introduction of mechanized agriculture, has led to soil erosion levels and widespread soil degradation as yet unseen. Humans currently manage half of the earth's surface and their actions have led to the rapid evolution of basic soil characteristics, such as soil thickness, texture or nutrient content. Richter (2007) states that humans today are the planet's major soil forming factor, in a process that started with the dawn of agriculture. For several thousand years, our past land use and soil management have significantly affected global vegetation patterns (Kaplan et al., 2010) and possibly even atmospheric CH₄ and CO₂ concentrations (Ruddiman, 2003). However, the effects of these past farming activities on soil erosion dynamics and global soil resources have received much less research attention. While it is well known that soil erosion has led to a significant loss of fertile land, our quantitative understanding of the dynamics of past global soil erosion is still limited to a few case studies. It also remains extremely difficult to evaluate how global soil resources have changed over time due to erosion-induced changes in soil depth, soil texture, stoniness or a decrease in nutrient status. And, finally, the role human-induced soil erosion has played in agricultural yields and sustainability over the past millennia is largely unknown.

Part of this knowledge gap is due to the fact that feedbacks between erosion rates, soil properties, vegetation and, more specifically, agricultural crop yields, are complex and hard to quantify. The magnitude of erosion-induced changes will depend on soil profile characteristics. Soils are typically formed by different horizons, each with widely varying properties. Superficial soil horizons are generally more nutrient-rich and have a higher water holding capacity. Past –but also future– changes in crop yield and other ecosystem services can therefore be expected to be highly non-linear, with thresholds that are determined by the soil

horizons. Significant changes in soil functioning can thus occur much more suddenly than typically assumed ones based on assessments of whole profile depth. Taking a typical topsoil depth of 0.3 m, combined with typical soil erosion rates on cultivated land of 1 to 4 mm yr⁻¹, the time required to erode through the fertile topsoil is in the order of 75–300 years. Many areas around the world, such as the Mediterranean or Asia, have a long land use history that by far surpasses this order of magnitude. Therefore, it can be expected that many regions currently have a significantly lower soil resource capacity than before the first land use and are more vulnerable to future climate change.

The paper is organized as follows. In section 2 the dynamics of (pre)historic land use and soil management, and its drivers is reviewed. In section 3 an analysis of the effect of (pre)historic land use changes on soil erosion is provided, while in section 4 the effect of historic soil management dynamics on soil erosion is further examined. Finally, in section 5, key challenges are identified and a roadmap is established for modelling past soil erosion and its effects on agricultural sustainability. Also similarities between the modelling of past and future soil erosion processes are highlighted.

2. Historical trends and driving forces of land use, soil management and productivity

Land use changes are generally considered as one of the main factors of global change (Foley et al., 2005). In some areas of the world, especially in Europe, occupation of the terrain started already several millennia ago. Deforestation (Kaplan et al., 2009) and expansion of cropland (Ellis et al., 2013) already reached its highest levels several centuries ago. Although this process of conversion of natural vegetation had started in early hunter-gatherer societies, these changes had a limited impact and were geographically limited with respect to those of the last three centuries. The last phase of the Holocene witnessed global changes without precedent in the history of mankind, to a large extent derived from changes in soil use, both including land use or land cover changes and changes in soil management (DeFries et al., 2004). Therefore, and with the increasing recognition of the scientific community, the *Anthropocene* has been suggested as a new epoch of geologic time (Crutzen, 2002; Ellis, 2011; Steffen et al., 2011; Waters et al., 2016), reflecting the capacity of mankind to interfere with the physical and biological dynamics of ecosystems. There is some controversy about the start date of this new geological era, as to whether it began as from the Neolithic revolution (Ruddiman, 2003), the industrial revolution (Crutzen, 2002) or from World War II (Zalasiewicz et al., 2010). There is no doubt, as pointed out by Ellis (2015), that “the ecological patterns, processes, and dynamics of the present day, deep past, and foreseeable future are shaped by human societies”. This is also applicable to soils; many of their features are of anthropic origin (Certini and Scalenghe, 2011), as discussed in the introduction.

Case studies around the world have shown that there appears to be a parallel evolution pathway in land cover changes, with similar transition processes, as shown in Fig. 1. This figure shows how natural ecosystems start suffering perturbations since the very start of human presence in a particular area. During the first stages of colonization, when population densities are minimal, processes like deforestation and low-intensity agriculture take place (Boserup, 1965). As population density increases, periods of ever more intensive agriculture succeed each other. This intensification is fueled by the increase in urban demand and economic incentives. This common pattern of transition processes repeated itself in accelerated or slowed down form, or even modified, in different historic periods or study areas (DeFries et al., 2004).

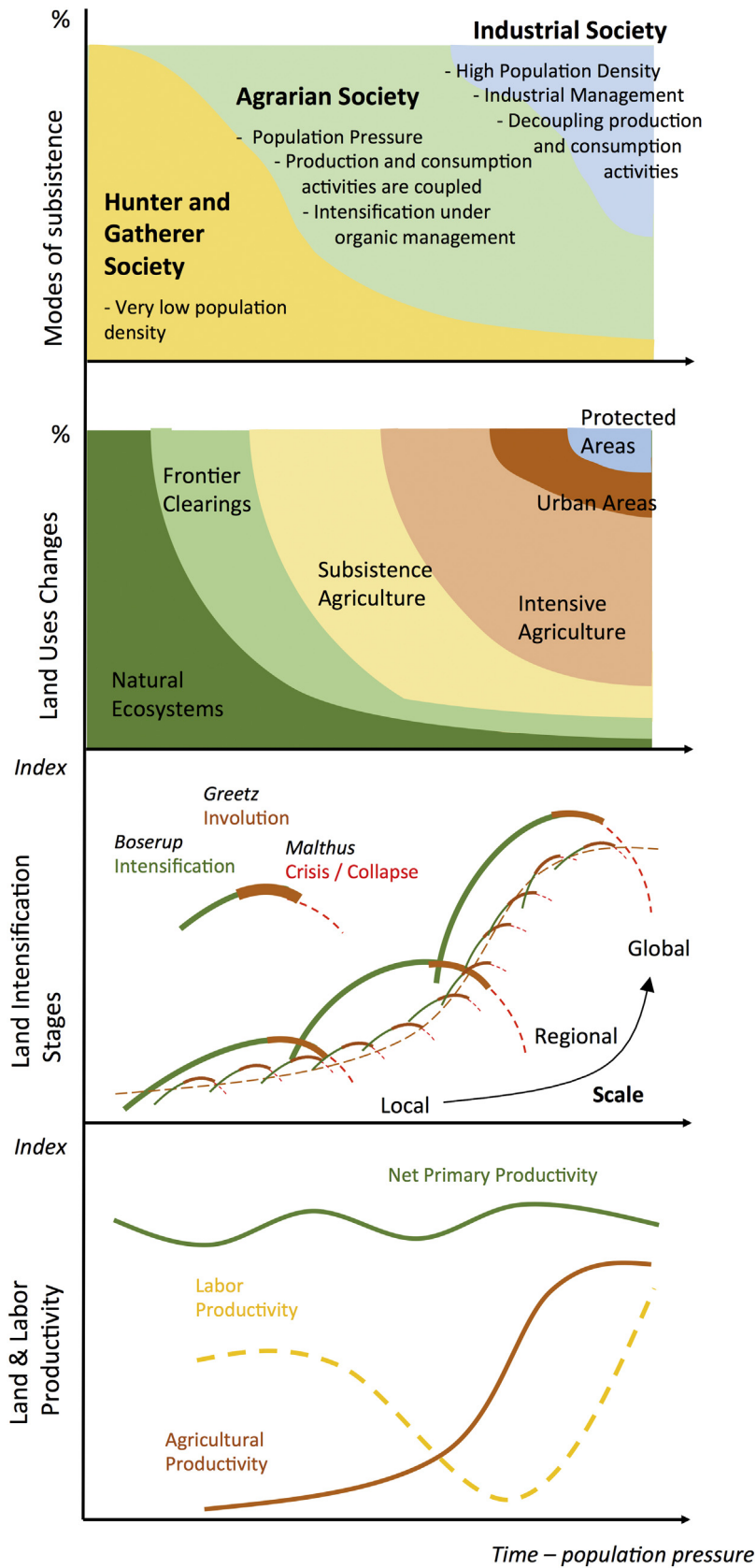


Fig. 1. Conceptual model showing the evolution over time of (a) models of subsistence; (b) land use changes; (c) land intensification stages and (d) land and labor productivity. (elaborated based on Boserup, 1965; Foley et al., 2005; Ellis et al., 2013; González de Molina and Toledo, 2014).

Kaplan et al. (2009) confirmed that agriculture was the main vector of deforestation processes in Europe from prehistoric times to industrialization. They also showed how from the start of industrialization in Europe, this process accelerated and culminated in the second half of the 20th century, after which the surface area occupied by forests started to increase again in many industrialized countries (Mather and Fairbairn, 2000; Mather and Needle, 1998). Parton et al. (2015) shows how also in areas of more recent colonization, the same patterns seem to have been followed.

This suggests that intensification of agriculture spares other areas by concentrating production on certain ones. Over the last years, the concept of “forest transition” was proposed by Mather (1992) and others (e.g. Grainger, 1995; Rudel et al., 2009), to describe this observed trend break of a constantly expanding cropland area. The concept refers to the shift from forest shrinking to forest expansion, observed at a national or regional scale around the world. This phenomenon has been documented in several European countries, North America and, more recently, in China, India, Vietnam, Costa Rica, Puerto Rico among others (Lambin et al., 2001; Lambin and Meyfroidt, 2010; Meyfroidt and Lambin, 2011). Rudel et al. (2009) and Borlaug (2007) have summarized the close links that exist between agricultural intensification and land sparing. The reasons for this transition can be found in the abandonment of agricultural activity and in the increase in environmental awareness, leading to the subsequent reforestation of old cropland or grazing areas (Mather and Needle, 1998). Scientific debate, however, has reached mixed conclusions. One reason is that in these studies, virtual or embodied land use being imported or exported at a national scale and sustaining a diet of high territorial requirements and of abandonment and conversion of large areas in the European Union, is not always being taken into account (Rudel et al., 2009; Meyfroidt et al., 2010; Kastner et al., 2012). Also, for example Rudel et al. (2009) conclude in their review that the link in the intensification–land-sparing hypothesis, between rising yields and declining cultivated areas, does not generally characterize agricultural sectors of between 1990 and 2005. This supports the idea that forest transition processes are happening at a regional or national scale, and fuels uncertainties whether or not agricultural intensification can really spare land.

Similarly to land use, intensification of soil management has also followed parallel trends globally, independently of where and when the case study occurred (Mustard et al., 2012). Foley et al. (2005) proposed a sequence of soil management or land use transitions that have intensity as their main driver. More recently and following the same criterion, a Land Intensification Theory was formulated by Ellis et al. (2013). Soil management intensification is defined here as an increase in agricultural operations and an increasing use of external inputs which, generally, results in an increase in agricultural production (Soto et al., 2016). The major body of existing literature tends to assume a Boserup-type maximum, in which the processes of intensification lead to an increase in productivity of the land but to a decrease in productivity of labor (Boserup, 1965) (Fig. 1d). Indeed, recent history shows a sustained growth in productivity, associated with more intensive management. However, two observations must be made on this generally accepted theory. Firstly, while it is clear that an increase in agricultural productivity has been documented (Bindraban et al., 2012; Currie et al., 2015), it is not at all evident whether net primary productivity also increased at the same rate (Krausmann et al., 2013; Smil, 2011). What is generally happening is that an increase in fruit or grain production is obtained, without a corresponding increase in total biomass production. Secondly, more and more studies seem to support that there was never a continuously growing productivity, but rather a complex succession that included retrogression and crisis phases (Ellis et al., 2013)

(Fig. 1c). The debate on collapse would be the most extreme example of this (e.g. Diamond, 2005). Finally, in relation to labor force productivity, currently it is known that it tended to decrease in preindustrial contexts, as was sustained by Boserup (1965), but in industrial societies a very significant growth occurred, associated with the process of industrialization of agriculture and of the agricultural sector in general (Fischer-Kowalski et al., 2014).

It has been debated in recent years whether it is possible to further intensify agricultural production without damaging natural resources. Agroecology defends that the only sustainable way to increase the productive land is by using agroecological methods (Nicholls et al., 2016), for example crop rotations (e.g. incorporating legumes) or increasing biodiversity with agroforestry techniques. However, outside the Agroecology branch, a new trend has emerged that defends the possibility of an “ecological or sustainable intensification”. This idea was introduced in the late nineties, linked to the need to increase productivity in degraded land in Africa (Pretty, 1997). There is no generally accepted definition of sustainable intensification, but the term is used as the enhancement of efficiency (Lang and Barling, 2012) or the “the use and optimization of biological regulation” (Doré et al., 2011) in order to improve soil productivity without degrading agroecosystems. It means an increase in production, while minimizing negative environmental impacts and avoiding the expansion of land used for cultivation (Godfray and Garnett, 2014). Because of this new further use of external inputs, some authors suggest sustainable intensification actually justifies new high-input models and the use of technologies, such as biotechnology (Loos et al., 2014).

With respect to the driving forces behind this intensification, the explanatory hypothesis based on the theories of Boserup (1965) received most attention. This hypothesis puts forward that the increase in land use and management intensity has been a consequence of the increase in population density and a decrease in the available land area. Many hypotheses are based on “population pressure” as a key variable causing land use changes (Currie et al., 2015). However, more recently, criticism of the hypothesis of Boserup and her followers has appeared. Lambin et al. (2001) oppose the idea that population and poverty trigger unsustainable intensification in smallholder agriculture. Other authors have limited the relation deforestation–population density to preindustrial societies (Kaplan et al., 2009) or, in general, the viability of the Boserupian theory in industrialized agricultures (Fischer-Kowalski et al., 2014). Ellis et al. (2013) propose to extend land-use intensification as the adaptive response of human populations not only to demographic pressures, but also to social, and economic ones. However, they warn that this process is non-linear, neither continuous, nor uniform, and can experience advances and regressions. This results, according to the social, demographic and economic conditions, in different soil uses that combine Boserupian with other Malthusian explanations. Among others, Kay and Kaplan (2015) sustain that intensification drivers are more complex than population, social or economic variables. They propose that there is a sequence of factors whose specific combination explains land use patterns. These authors describe, based on archaeological records and other sources, land use pattern characteristics in sub-Saharan Africa during the period 1000 BCE – 1500 CE, using a simple classification scheme with 17 categories that cover the range of human subsistence strategies from foraging to urbanized societies. These categories reflect specific combinations of diet, technology, culture, subsistence and urbanization that lead to distinctive patterns of land use intensity.

Finally, some authors state that land changes at all spatial levels are influenced by long-distance flows of raw material, energy, products, people, information and capital, creating a need for novel

theoretical and methodological approaches to the analysis of causal relationships in land system dynamics (Friis et al., 2015). They propose considering land systems as coupled human–environment systems. In this sense, land use systems could be better understood and analyzed when considering the result of a specific metabolic design and land use transitions as components of a more general socio-ecological transition (Fischer-Kowalski and Haberl, 2007; González de Molina and Toledo, 2014). Beyond the complexity of land use change across history cited above, the existence of land change regimes may be related to socio-metabolic changes. In this sense, preindustrial economies, characterized by their impossibility of establishing large trade networks, generated isolated settlements forced to integrate multiple land uses to meet subsistence (Fischer-Kowalski and Haberl, 2007). Industrialization allowed to decouple production and consumption activities worldwide (e.g. Erb et al., 2009) as well as to intensify agro-ecosystems (e.g. Soto et al., 2016), promoting the spread of monocultures and changes in landscape morphology (e.g. Infante-Amate, 2014). This perspective is also behind new proposals to understand the beforementioned Forest Transition, highlighting that forest increase may be explained by the growth of the fossil fuel economy (Erb et al., 2008; Gingrich et al., 2016). Carbon economy has allowed to abandon certain forest areas (Meyfroidt and Lambin, 2011) as well as to increase their carbon stocks (Gingrich et al., 2007; Kauppi et al., 2006), mainly due to the growing availability of fossil fuels that substituted traditional functions of forests (Sieferle, 2001). In short, a growing body of literature is putting the focus on explaining land use changes and their telecoupling effects from a biophysical perspective (Erb et al., 2008; Haberl, 2015).

3. Effects on past land use changes on soil erosion

Land use and soil management intensification over time is due to a complex combination of driving factors, as was analyzed in the previous paragraph. The same is true for its effects. The most direct of these effects, and that is generally considered to be one of the most significant soil degradation processes, is soil erosion by water. While natural soil erosion rates are in equilibrium with the production of soil as a result of weathering, a shift from natural to

agricultural land use increases soil erosion by one to two orders of magnitude, resulting in rapid profile truncation (Montgomery, 2007). As shown in Fig. 1, the earliest human presence in a particular area leads to a perturbation of the natural ecosystem. It is, however, an open question as to when this human impact was high enough to cause significant soil erosion. Higher land use intensity can generally be associated with higher rates of soil erosion. Under lower land use intensity, much of the sediment can be trapped along the same hillslope by a strip of natural vegetation or a fallow plot. As agricultural expansion continues in a region, the landscape connectivity for sediment increases as field patches link together and form continuous pathways from hillslopes to streams, with the most suitable agricultural areas being occupied first. A first mention to this system was made by the Greek Theophrastus in 313 BCE, but the practice was much older (Semple, 1931). During intensification, farmers start cultivating also steeper areas, resulting in higher soil erosion rates. This all supports the idea that there is some threshold or limit of first significant land use, much like the concept of first significant land use proposed by Ellis et al. (2013).

At a global level, recent research has advanced significantly in the spatial and temporal reconstruction of past land use changes. Land use scenarios such as KK10 or HYDE have been applied successfully to show effects of human land use change on the global carbon cycle (Kaplan et al., 2010). However, at present, geomorphic information is limited to isolated case studies (Dotterweich, 2013). So far, no systematic attempt has been made to link past land use changes to soil erosion at a broader, regional or global scale and there is no information as to when a first significant erosion phase occurred. Therefore, this review has synthesized a global overview of soil erosion case studies from literature in Table 1 and compared this to the land use and land change scenario KK10 developed by Kaplan et al. (2010). In particular, the hypothesis tested was that the period of first significant land use, as proposed by Ellis et al. (2013), resulted in a first significant soil erosion signal. Only studies that performed an absolute dating of the erosion phases and that focused on hillslopes or small catchments were included where possible (i.e. where several studies were available for the same area, the study performed at the smallest spatial unit was included in

Table 1
Case studies on historical soil erosion, shown in Fig. 2.

Zone	Source	Location	Country	Period earliest erosion pulse ^a	
Northern and Central Europe	Vanwalleggem et al., 2006	Meerdaal Forest	Belgium	2	
	Rommens et al., 2006	Belgian loess belt	Belgium	2	
	Bertran, 2004	Quercy region	France	3	
	Lang, 2003	Rhine catchment	Germany	2	
	Dotterweich et al., 2003	Wolfsgraben	Germany	5	
	Bussmann et al., 2014	Osnabrueck	Germany	1	
	Dotterweich et al., 2012	Kasimiersz Dolny	Poland	2	
	Silva-Sánchez et al., 2014	NW Spain	Spain	2	
	Mediterranean	Fuchs, 2007a,b	Phlious Basin (NE Peloponnese)	Greece	1
		Engel et al., 2009	Lower Messenian Plain	Greece	2
Vanwalleggem (unpublished data)		Baena	Spain	3	
Dusar et al., 2012		Sagalassos	Turkey	1	
America	Stinchcomb et al., 2011	Delaware river valley (Eastern US)	USA	5	
	Beach et al., 2006	Mayan peninsula	Mexico	3	
	Beach et al., 2008	Mayan lowlands	Mexico	3	
Asia and Oceania	Gale and Haworth, 2005	East Coast	Australia	7	
	McWethy et al., 2009	New Zealand	New Zealand	5	
	He et al., 2006	Loess plateau	China	3	
	Rosen, 2008	Loess plateau	China	1	
	Bork and Ahrendt, 2006	Shaanxi	China	2	
	Li et al., 2010	Liangzhu and Qujialing sites	China	2	
	Mieth and Bork, 2005	Easter Island	Chile	5	

^a see legend Fig. 2. Period codes: 1 = > 8000 years; 2 = 5000–8000 years; 3 = 3000–5000 years; 4 = 2000–3000 years; 5 = 1000–2000 years; 6 = 500–1000 years; 7 = 250–500 years; 8 = 100–250 years).

Table 1). Grove and Rackham (2003) pinpointed the amount of uncertainty associated typically with palaeo-environmental studies and indicated that the chronology of alluvial sediments does not regularly correspond to the rise or decline of past land use. By selecting only those small-scale case studies, a direct link between land use and soil erosion can be expected, in other words the important complication of sediment reworking in the fluvial system of larger catchments can be avoided. The drawback is then that a single site could potentially not be representative for regional dynamics. For example, an erosion phase could be induced by climate extremes rather than by real changes in the land use. Identifying the correct cause of measured past erosion events is often complex and while most studies use an interdisciplinary approach to compare observed erosion pulses to climate or vegetation records and archaeological data, some level of uncertainty persists. However, by combining a high number of individual datasets at a global scale, it can be expected that this uncertainty would be reduced and that general patterns would emerge.

Fig. 2a shows the location of the documented historical erosion case studies listed in **Table 1** with respect to the global distribution of first significant land use. The global distribution of historical erosion case studies is clearly heavily skewed towards the European and North American context. While more and more studies are becoming available for China, much more work still needs to be done in other areas with a similar or even longer land use history, such as South America, Africa and parts of Asia.

Fig. 2b shows the results of comparing the timing of significant first land use changes and the timing of the first soil erosion pulses extracted from the geomorphological record. In general, there is a good correspondence between both, with most case studies scattered along the 1:1 line. In about 25% of the case studies, the erosion pulses and land use periods coincide, whereas in 50% of the cases they almost do (i.e. the difference in timing is limited to plus or minus one time period; area shaded in dark grey). Areas that plot below the 1:1 line correspond to areas where erosion was documented before major land use changes occurred. This is for example the case in the US, where [Stinchcomb et al. \(2011\)](#)

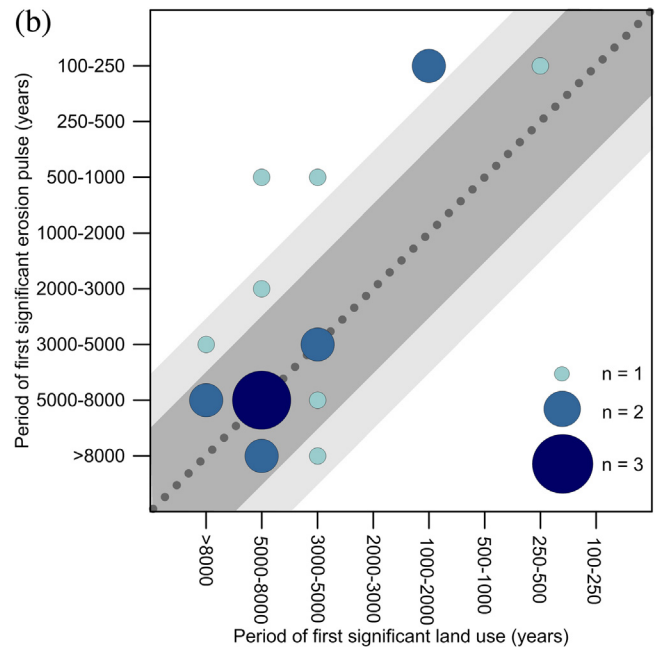


Fig. 2. (Continued)

observed soil erosion associated with agriculture from native Indian populations, while significant land use changes were only detected after the arrival of European farmers. This is actually a relatively common observation, and occurs in a third of all case studies. This observation points to one possible problem with this type of data, which might be the spatial representativeness of the geomorphic evidence, as early land use changes, in particular deforestation, were most probably much more limited in the landscape than current agricultural land use that has a much more generalized territorial impact ([Foucher et al., 2014](#)).

On the other hand, areas plotting above the curve correspond to areas where erosion was not measured until well after significant

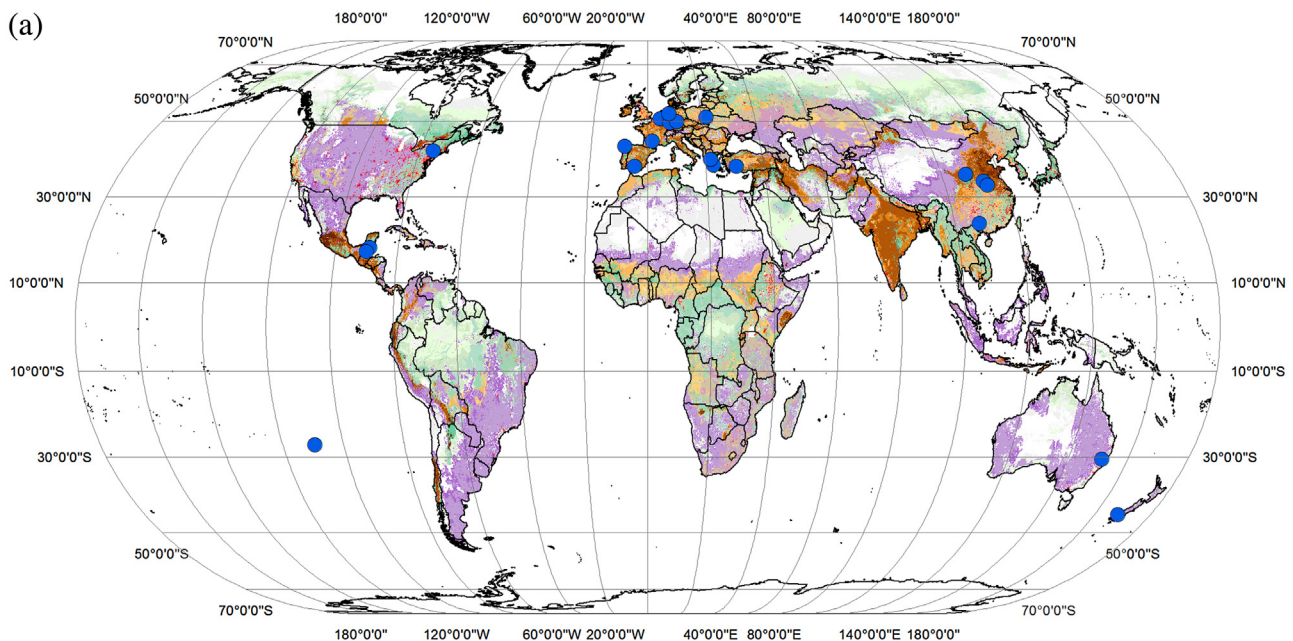


Fig. 2. (a) Map of period of first significant land use change. The location of case studies of (pre)historical soil erosion is shown in blue. (b) Relationship between first significant land use change and first significant erosion pulse. Colors codes indicate number of correspondences. Dark and light shaded areas indicate a time difference of respectively 1 and 2 periods around the dotted 1:1 line.

land use changes set in. This is a more rare observation and only occurred in 10% of the cases.

In conclusion, in many case studies, important and widespread erosion peaks were found with the arrival of the first farmers all over the world. This good match between soil erosion and land use phases has important implications as it shows the potential of using global land use scenarios, such as KK10, for modelling past soil erosion rates in areas where little or no geomorphic evidence is available, or for extrapolating local case studies to a regional scale. These results also corroborate findings of other studies, like that of [Dotterweich \(2013\)](#), who presents a global synopsis of different case studies about the geomorphic evidence of soil erosion in humid and subhumid areas since the beginning of agriculture. He provides a comprehensive collection on the development of soil erosion research and soil conservation, giving examples from ancient Greek and Roman Times and from central Europe, southern Africa, North America, the Chinese Loess Plateau, Australia, New Zealand, and Easter Island. Soil erosion has been a significant factor in a complex causality spiral leading to socioeconomic instability and land use changes ([Dotterweich and Dreibrodt, 2011](#)), and could possibly be linked to the decline of numerous civilizations around the world. [Dotterweich \(2013\)](#) concludes that globally, agricultural soil erosion occurred in three main waves ([McNeill and Winiwarter, 2004](#)). The first wave is generally assumed to have started around the second millennium BC and over the next 3000 years the soil erosion rates increased because farmers in Eurasia, Africa, and the Americas gradually converted a moderate proportion of the world's forests into farmland or pasture. The second wave took place during the sixteenth to nineteenth centuries with the introduction of stronger and sharper plowshares, which helped to break the sod of the South American pampas, Eurasian steppe, and the North American prairies. And the third wave began with rapid population growth after the mid-twentieth century when people started to clear rainforests and steep areas to exploit wood for timber or fuel or to expand agricultural land.

4. Effect of past agricultural soil management changes on soil erosion

The majority of the studies discussed so far reveal a close relationship between soil erosion and land use changes, especially deforestation and the advance of agricultural land. However, it is much more difficult to analyze the effect of past soil management changes on soil erosion, although soil management changes are potentially as important as land use changes. Current knowledge shows us that under the same land use, different agricultural practices can change erosion rates by an order of magnitude, as concluded by [Montgomery \(2007\)](#) based on a comparison of traditional and conservation agriculture. As sustained by [Dotterweich \(2013\)](#), local and regional variations in natural situations, cultural traditions, and socioeconomic conditions played a major role in the dynamics and rates of soil erosion in a long-term perspective. As intensification continues and the available land is taken up, farmers need to intensify the existing land and resort to changing soil management. Even under forest or grassland, more intense use leads to activities such as overstocking, charcoal burning or iron smelting, all significantly altering the hydrological and sedimentological response. Yet surprisingly, almost no studies exist at a temporal resolution that would allow the resolution of the effects of changing historical soil management or of the implementation of conservation structures such as terraces. Nevertheless, according to the literature, such practices were well known and have been used almost since the introduction of agriculture. For example [Arnáez et al. \(2015\)](#) state that terracing dates back to over five thousand years ago in south-east Asia, after which it spread to the Mediterranean. Terraces are also found in

ancient American civilizations. This knowledge gap, in which almost no information is available on the different past agricultural practices or forms of soil management with respect to soil erosion can be addressed by field observations at a higher temporal resolution, by modelling, or by a combination of both.

An example of the first type of study, field observations at a high temporal resolution, is given by [Foucher et al. \(2014\)](#), who studied in detail sediment deposits in a 24 km² cultivated lowland catchment in France. A reconstruction of sedimentation rates by fallout radionuclides, allowed them to distinguish seven different periods of large variations of sedimentation rates over the last 60 years and link these to management changes. They observed how a major land consolidation scheme between 1945 and 1960 led to a dramatic increase in the sedimentary production, from 40 t yr⁻¹ before 1950 CE to about 13,000 t yr⁻¹ in the 1950s and 1960s. Since then, erosion and transfers decreased regularly due to management changes that decreased the connectivity in the catchment, although current sediment production was still 60-fold higher than before 1950. Similar observations were made in other countries, albeit with less temporal detail. [van der Post et al. \(1997\)](#) identified an increase in modern sedimentation rates in the English Lake District. They interpreted this as a direct consequence of increased sheep stocking rates in the nineties. In a review of lake sediment rates across the Midwestern United States since the European settlements in the 1800s, [Heathcote et al. \(2013\)](#) also observed an increasing pattern. They attributed this to an increase in agricultural intensification, rather than in agricultural land area, and to a failure of traditional soil conservation programs. The largest increases in sediment deposition occurred after 1950, concurrent with agricultural intensification, while total agricultural land area remained stable already since the early 1900s. Over a much longer time scale, [Currás et al. \(2012\)](#) presented a high-resolution study of the sediments in a Mediterranean dry lake in Central Spain. Their palaeolimnological reconstruction extends from the 9th century BC till the 8th century AD. While they attributed most changes in their sedimentation record to land use changes, some inferences about management could be made during particular periods. For example, they observed the disappearance of coprophilous fungal spores and low abundance of apophytes in the Visigoth period, a period of lower population pressure, which they attributed to a reduction in grazing pressure. Of course, the longer the temporal scale, the more difficult it becomes to establish a direct link between management and observed sedimentation rates. However, as these case studies show, high-resolution palaeolimnological data has an important potential for inferring not only climatic or land use forcings, but also for detecting such changes in past management. Other field methods for measuring historical soil erosion such as studies of soil profile truncation only allow to measure total average erosion and need to be combined with modelling in order to reconstruct temporal patterns. The same is true for direct field observations from long-term field experiments. These offer unique and valuable information on management effects on soil erosion, but are limited to several decades at best ([Karlen et al., 2014](#)). One well-known disadvantage of palaeolimnological data is the issue of equifinality, as different causes can result in a similar effect on lake fauna or level. Therefore, as shown by [Currás et al. \(2012\)](#) or [Foucher et al. \(2014\)](#), it is important to combine multiple proxies (pollen, non-pollen, palynomorphs, macrocharcoal, ostracods, diatoms, other biotic remains and sedimentology) and to have access to detailed historical or archaeological records to accurately interpret the causality of observed variations in sedimentation rates.

A combination of modelling with field observations is a second powerful tool, used in another case study by [Vanwalleggem et al. \(2011\)](#) in Montefrío, Southern Spain, and extending over almost three centuries. By combining empirical field data and erosion

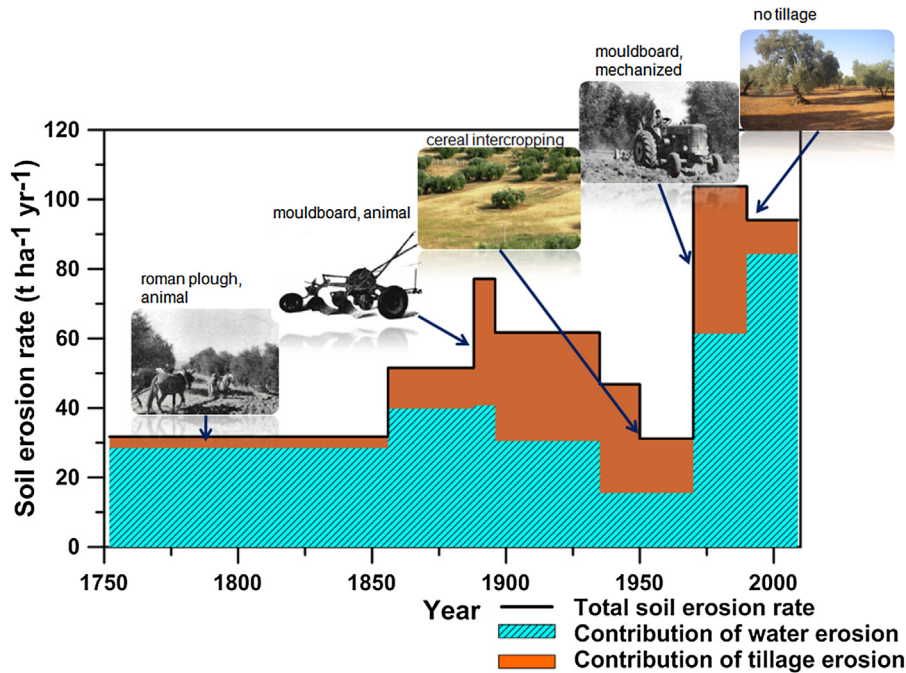


Fig. 3. Water and tillage erosion between 1756 and 2010 under olive in Montefrío, S Spain, with identification of the main agricultural management changes (adapted from Vanwalleghem et al., 2011).

model analysis, they were able to link the soil erosion rates under olive cultivation, one of the oldest and most characteristic crops of the Mediterranean, to different management periods. Thanks to an interdisciplinary collaboration, input parameter sets for the water erosion model RUSLE and a tillage erosion model, could be derived for 8 different period between 1756 and 2010, based on historic and archival data from that site. Cumulative modelled erosion rates were then compared to observed total soil truncation to validate the model. Fig. 3 shows how soil erosion rates increased about three fold between the start and the end of the studied period, despite being under the same land use. In 1750, olive farms were extensively managed, producing 200 kg ha^{-1} and their management caused relatively low soil loss rates, albeit far from being sustainable. However, actual crop management has intensified through chemical treatments, agricultural machinery, and removal of native plants and grasses. Now, olive yields have increased tenfold ($2,5 \text{ t ha}^{-1}$), but soil losses have also increased to $90\text{--}100 \text{ t ha}^{-1} \text{ year}^{-1}$. Over the last few years, soil erosion rates have been slightly reduced due to a shift towards no tillage, which increased water erosion rates but lowered tillage erosion rates considerably. After identifying the main periods of soil erosion, Mabit et al. (2012) further validated the erosion model with radionuclide tracers and Infante-Amate et al. (2013) explored the social and institutional context to explain the factors that changed management of olive groves and influenced soil losses. In sum, in this case study, two main conclusions were reached: (1) although the land use has been constant over the last 250 years (olive), soil management has caused large variations in the erosion rates and sustainability of the agro-ecosystem and (2) high erosion rates are not related to socio-demographic or economic drivers but rather the result of poor soil management, which is the product of specific socio-metabolic drivers.

These case studies stress the importance of accurately incorporating not only land use but also soil management into soil erosion models. Both examples show how the same land use results in significantly differing soil erosion rates due to management effects. The next section discusses key challenges to perform this analysis on a historical scale, and shows the similarity between

reconstructing past processes with the modelling of future soil erosion rates.

5. From the past into the distant future: key challenges for modelling the effect of past and future soil erosion on agricultural sustainability

5.1. Modelling past and future soil erosion

Analysis with simulation models is an effective way to evaluate the impact of past or future changes in land use, management or climate on soil erosion in agricultural systems. Although the effects of past changes in any of these driving factors on soil erosion rates can be assessed in case studies, based on detailed field observations, the extrapolation of such cases to regional or global scales is limited to model simulations. With respect to future predictions, simulation models are actually the only available tool. This is then one of the major uses of models, as tools for decision-making in engineering and environmental planning (Engel et al., 1993). Many different erosion models exist, from simple, empirical ones such as the Universal Soil Loss Equation (USLE) to complex, physically-based models (e.g. WEPP). The USLE (Wishmeier and Smith, 1978), and its revised version RUSLE (Dabney et al., 2012) is undoubtedly the model most widely used for soil erosion modelling. There are several reasons for this, among which are its relative simplicity (compared to physically-based models), the fact that comparisons with other models showed that the average error and model efficiency in predicting soil loss are similar (e.g. Morgan and Nearing, 2002), and the possibility of introducing directly, or through external calibration of key parameters, the effect of land use, management or climate on key drivers of soil erosion such as rainfall erosivity or soil ground cover. The various erosion models have all been extensively calibrated and validated under present-day conditions.

When modelling soil erosion over longer periods, from decades to centuries or even millennia, some problems arise. Long-term modelling of soil erosion and its consequences, soil truncation and losses in yield or biomass production, are subject to some key

limitations that are similar for both, backward modelling over (pre) historical time scales, and forward modelling into the future. These key limitations are related to (i) feedback effects that are generally not considered and (ii) the uncertainty related to the determination of the input factors.

The first key limitation of the feedback mechanisms involved is represented schematically in Fig. 4. This figure gives an overview of the main and secondary input factors for calculating long-term trends in soil erosion, soil resources and yield. The quantification of the relationship between soil erosion and yield is the subject of the next paragraph. Irrespective of the erosion model used, their key primary input parameters being similar. Topography and soil are typically considered static, while climate, land use and soil management change over time in most simulation studies. In a broader approach, socio-economics should also be considered at this level. Most erosion studies do not include socio-economics explicitly, although its influence on soil erosion is well known. Clear examples are the effect of agricultural policies on soil conservation measures or the pricing of agricultural commodities on land use changes (Boardman et al., 2003). An exception to this is the study by Ye and Van Ranst (2009). They presented a scenario analysis for evaluating the impact of soil erosion on future food security in China and took into account socio-economic variables, such as population growth and urbanization rate, at the highest level.

The secondary or derived input factors are calculated from the primary variables, and are the specific inputs required by any erosion model. For the example of RUSLE, the long-time average

annual soil loss, carried by runoff from specific field slopes in specific crop and management systems, is computed as the linear product of six factors: a rainfall erosivity factor; a soil erodibility factor; a slope length factor; a slope gradient factor; a cover-management factor; and finally a support practice factor. There are extensive definitions of the meaning and procedures to determine these factors (e.g. Renard et al., 1997), but here we will focus on interactions that could be important for modelling over long time scales.

The first of the interactions commonly missing in erosion models is the dynamic feedback of topography on erosion and deposition processes (Dabney et al., 2013) (Fig. 4, far left). This is a well-known problem in geomorphological studies as the initial topography is unknown (Hancock et al., 2016). Over long time scales, of centuries to millennia, slope gradient might change considerably. As erosion tends to smooth out topography, keeping it static can lead to a considerable underestimation of past erosion processes or an overestimation of future soil erosion. Some recent long-term models such as WaTEM-LT (Peeters et al., 2006), LAPSUS (Debolini et al., 2015) and LandSoil (Ciampalini et al., 2012) are erosion and sediment transport models in which erosion and landforms are coupled and coevolve. However, in turn, they present the drawback of the representation of erosion being simplified and are more comparable to Landscape Evolution Models (LEMs). Literature support for their used erosion parameters is more limited and harder to determine from current agricultural analogs. On the other hand, in some cases, in which past agricultural land uses were significantly different from their

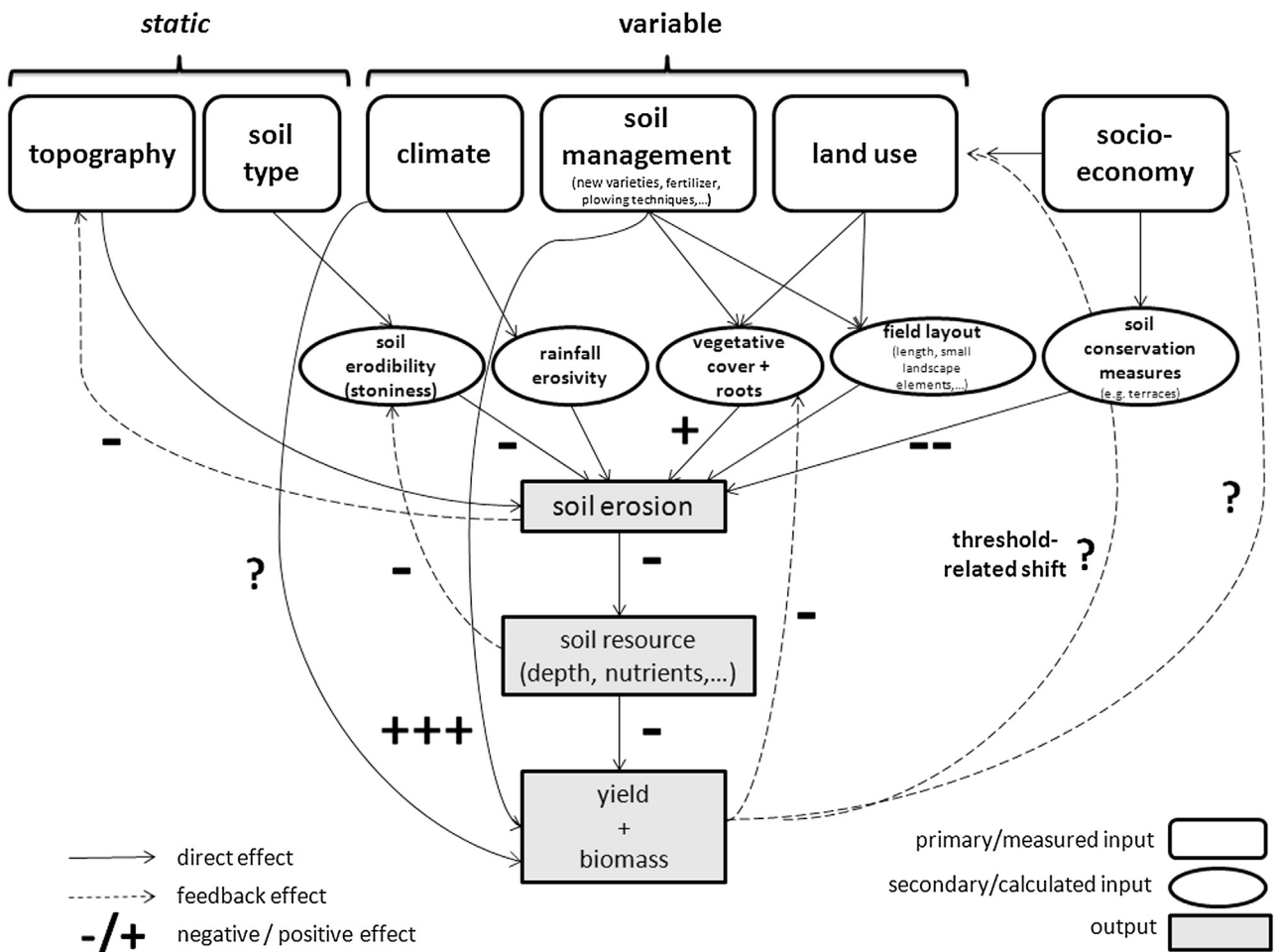


Fig. 4. Interactions and importance of difference factors for the prediction of long-term trends in soil erosion, soil resources and yields.

present-day counterparts, or where no historical information exists, our limited knowledge of the system might justify the use of simplified LEMs. Peeters et al. (2006) and Temme et al. (2011) researched model soil erosion in the Belgian loess belt over the last 2500 years with different models, but using a constant set of input parameters through time. While their models successfully reproduced the spatial patterns of erosion and deposition observed, their results offer no insight into the temporal dynamics due to the different land uses or agricultural practices over this long time period that included important agricultural innovations such as the moldboard plow or mechanization. Ciampalini et al. (2012) apply their model LandSoil to a Mediterranean catchment. In their study, they take into account field layout and detailed management effects on soil surface characteristics. However, this type of detailed approach is highly data-demanding and difficult to parameterize over long time scales. Their study is limited to a medium term time scale of 10–100 years.

The second feedback loop that is often not considered in erosion models is due to dynamic soil properties, which include those impacting crop productivity but especially those that change the soil's erodibility. Over longer timescales, sediment supply and availability can change in response to ongoing erosion. Soil erosion generally pushes soil profiles into becoming coarser, to accumulate a protective surface stone cover or armor (Willgoose and Sharmeen, 2006), or –in extreme cases– even to be completely truncated and deprived of any erodible material. Ample evidence from landscapes with a long land use history illustrates this process. Anselmetti et al. (2007) studied lake sediments caused by ancient Maya land use in Northern Guatemala. After an initial phase of important soil loss, they observed a steep decline in sediment accumulation that contrasted with the continuously growing population densities. This decoupling between the land use and climate could only be explained by accounting for the changing soils in the catchment.

In addition to these pedogenetic and topographic feedback mechanisms, people also respond to soil erosion by adopting conservation practices such as for example terraces, although there might be a significant time lag between the cause and reaction. For

example, Beach et al. (2002) documented severe Maya soil erosion starting in the Preclassic period (1500 BCE – CE250), while the first terraces did not appear until the Early Classic period (250 CE–600). This time lag can possibly be explained by the fact that yield reduction due to soil erosion and truncation is often a slow process, which will be discussed in detail in the next paragraph. Such conservation measures are specifically designed to drastically reduce soil erosion rates, and will have a much greater effect compared to the previous feedback mechanisms, as indicated in Fig. 4 by the double minus symbols. In any case, even without conservation measures, continued erosion of the soil's natural resource results in a lower yield and biomass production. This will have a direct effect on the amount of soil cover and roots that provide resistance to erosion, an effect that is rarely accounted for. In extreme cases, if conservation measures do not reduce soil erosion sufficiently or, specifically, a (large) reduction in yields will result, leading eventually to a land use change, which can be sudden in some cases. In Germany, Bork (1998) describes how widespread gully erosion after extreme rainfalls in the 15th century led to widespread abandonment of the eroded farmlands. These areas that have since been conserved as forests still preserve these intact medieval gully networks. The choice of pathways between implantation of conservation structures and/or land use changes is complex and has important interrelations with socio-economic decision making processes, as shown by the feedback arrows indicated in Fig. 4.

The second key limitation related to long-term erosion modelling is the uncertainty limited to the parameterization of the input factors. Again, many of the challenges for determining the correct variation of climate, land use and management over time are similar between modelling soil erosion over historic time scales and modelling future soil erosion. Fig. 5 summarizes the different sources of input for generating information on the three key variable input variables and shows a conceptual representation of the error associated with these values, both for future as for past predictions. The uncertainty is expected to increase over time, both for predictions of the past and the present. Uncertainty is smallest, although greater than zero, for present-day conditions when

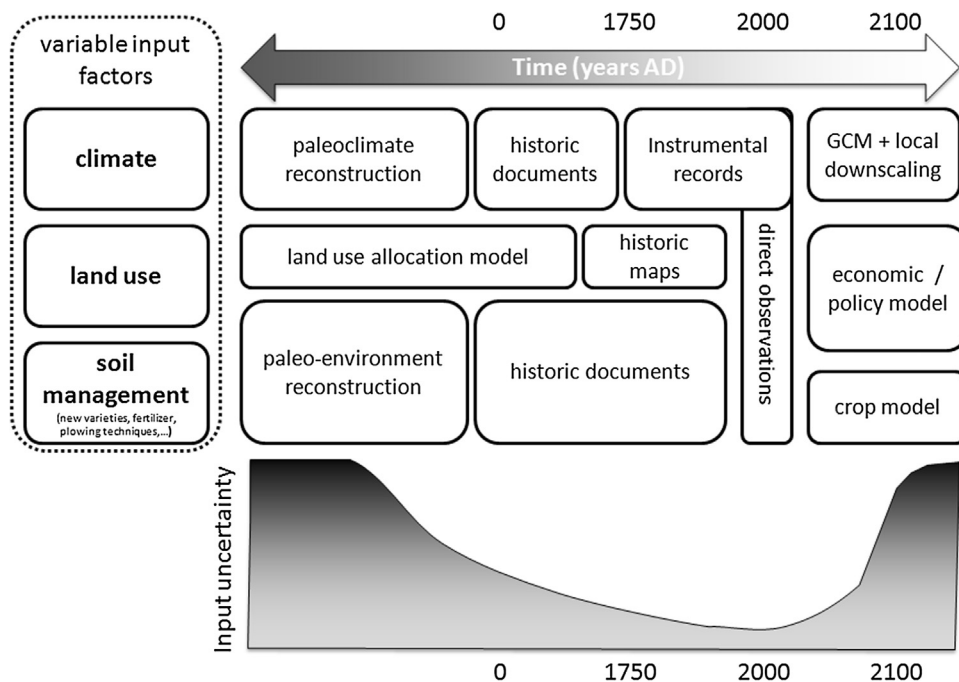


Fig. 5. Sources for input factors of long-term soil erosion models and their uncertainty as a function of the time scale considered.

simulating from direct observations. With respect to climate, as long as instrumental records are available, roughly over the past 150–250 years, uncertainty is low. For their historical erosion study, Vanwallegghem et al. (2011) used instrumental rainfall record observations from the San Fernando observatory, established in 1751 and keeping one of the longest instrumental records in Europe (Gallego et al., 2007). To extend such series further however, one must rely on proxy data that can range from historical documents containing weather descriptions to tree rings or ice cores (Jones, 2004). Documentary sources, although often questioned because of their subjective nature, have the advantage that apart from temperature or precipitation, they often yield information on additional variables, such as the start of the growing season or extreme events that are significant for modelling agricultural systems. Future soil erosion modelling has focused mainly on climate change effects on rainfall erosivity, derived from downscaled GCM predictions. Most erosion models however, such as RUSLE, do not link crop growth to climate inputs, and the user has to describe the growth and specify vegetation yield. It contains algorithms that allow estimates of the timing and amount of residue creation by perennial vegetation to dynamically adjust to alternative harvest management schemes although these routines are not linked to climate. It also includes similar algorithms for the definition of the surface roughness based on tillage operations and its decay from time and rainfall. As a result, RUSLE relies on an external estimation of the impact of climate and management conditions of these variables for a rigorous determination of the soil cover factor (Dabney et al., 2013), as opposed to more process-based models used in the context of climate change such as WEPP (e.g. O'Neal et al., 2005). However, in many of the climate change studies, RUSLE has been used assuming soil cover values derived from current conditions without a deep exploration of the effect of climate change on variables that might affect vegetation cover (e.g. Mello et al., 2015). This is obviously a simplification as changing temperature and soil moisture conditions will affect planting dates and may shift the growing season, leading to changes in the seasonal variation of vegetation cover. Mullan et al. (2012), use a scenario approach to include the uncertainties related to the indirect effects of future land use change impacts. When considering only downscaled climate change predictions, they predict an erosion decrease in their N Ireland study area, whereas they obtain large increases when including land use changes. This drastic change illustrates the high uncertainty of erosion model predictions under climate change, both for prediction of the future and the past, and shows that a proper determination of the effect of rainfall changes on rainfall erosivity and soil cover, and especially their interactions, is key to obtaining a robust and reliable analysis.

Another level of uncertainty is added by socio-economic drivers that control land use and soil management intensity. The latter are expressed not only by the total farmed area, but also the location, field size and crop type. Field size is an important, yet often overlooked factor that controls the length over which runoff water can collect and concentrate (Fig. 4). Very few data are available on the historical evolution of agricultural field size, but it is clear that current commercial agriculture supports much larger field sizes compared to those of subsistence agriculture. In addition, increase in field size is accompanied by the tendency for large blocks of the same crop, giving rise to bare ground at the same time of the year and, therefore, the possibility of runoff flows over long distances. White and Roy (2015) measured field size changes over the last 25 years from Landsat imagery. Over historical time scales, maps, documents and archaeological evidence could provide the necessary information, as shown in Fig. 5. However, field location is often not known if such historical maps are absent. Although, locally, medieval maps are available for landscape reconstruction, most

regions do not have accurate maps spanning more than a couple of centuries (Jongepier et al., 2016). De Brue and Verstraeten (2014) showed, with a soil erosion and sediment distribution model, that different spatial allocations of land uses lead to a significantly different estimation of sediment fluxes in the Belgian loess belt.

With respect to soil management, this can be reconstructed based on historical documents or by simulation of land use intensity from food supply and demand. The primary source of historical soil management data stems from cadastre registers that contain inventories of different soil management types with the original purpose of establishing tax quota. Another important source for reconstructing historical soil management are private accounting documents, such as farm lease agreements wherein the owner established specific agricultural management practices, and, to a lesser degree, other sources such as notary registers (Garcia-Ruiz et al., 2012). Soil management has a direct effect on soil erosion through the frequency of plowing, fertilization, and other farming operations and results in significantly different amount of ground cover, crop residue left on the fields, soil roughness or moisture. All these factors combined are important for accurately estimating the cover factor in RUSLE, or in other erosion models. Similar to land cover change studies, several groups have studied the evolution of historical soil management. Vanwallegghem et al. (2011) derive farming operations directly from historical documents, mostly invoices and rent agreements, for their local case study and use it to calculate soil erosion. Garcia-Ruiz et al. (2012) also reconstruct historical cropland intensification at a parish scale in order to reconstruct nutrient balances of nitrogen, phosphorus and potassium. At regional scales, McGrath et al. (2015) present an interesting approach for reconstructing historical forest management from 1600 to 2010. They simulate management from comparing simulated demand and supply, based o.a. on population estimates and wood use. Jepsen et al. (2015) present an expert-based land management reconstruction for the past two centuries in Europe, although their regional assessment misses the level of detail required for modelling soil erosion. As far as we know, no such detailed regional study has been conducted for agricultural soil management.

5.2. Modelling soil truncation effects on yield and agricultural sustainability

From the previous paragraphs, it has become evident that past conversion of natural land into croplands, and subsequent agricultural soil management intensification has led to a significant increase in erosion rates globally. Soil erosion indisputably affects the ability of soils to produce foods. However, the extent to which a soil's crop yield potential has been affected by historical soil erosion is largely unknown at present. In some areas used for agriculture for millennia, such as the Mediterranean or parts of the Ethiopian highlands, soil erosion might have completely destroyed the productivity of cropland in hilly or mountainous areas (Govers et al., 2014). In order to assess the long-term effect of soil erosion on yields and agricultural sustainability, the first step is to accurately model historical erosion rates. The next step is then to assess how the resulting soil truncation has affected crop yields and agricultural sustainability. As discussed in the previous paragraph, erosion models allow us to quantify these increased erosion rates, albeit with a certain degree of uncertainty, that depends on our ability to estimate the erosion model's input factors over past or future periods. However, although the impact of soil erosion on crop yield is undisputable (Ye and Van Ranst, 2009), the exact relationship between soil erosion and productivity is difficult to quantify. Erosion-induced effects of soil erosion on crop yield are hard to generalize (Lal and Moldenhauer, 1987) because they vary among soils, management systems, crops and

climate. Soil profile truncation will affect plant growth mainly through: (1) plant-available water; (2) rooting depth; and (3) vertical distribution of nutrients (Lal, 1998). Soil type and horizon distribution is therefore the first factor leading to widely different crop yield responses to soil erosion. Deep, fertile soils, for example loess-derived Luvisols or Fluvisols that get regularly replenished by floodplain deposits, can sustain high soil erosion rates for several centuries without any effect on crop yield. The same erosion rates in soils that are shallow, or underlain by a hard, impeding layer, or that concentrate plant-available nutrients in the topsoil, could lead to dramatic yield losses over a few decades. Examples of such soils are Leptosols, Durisols and most tropical soils. A classification of global soil types into deep, fertile soils (type A) and soils that are vulnerable to soil erosion (type B) is shown in Table 2. A third soil type (type C) represents soils that do not fall into either class, either because that soil type has a marginal agricultural suitability or because it cannot easily be classified into A or B. From Table 2 it can be seen that both A and B type soils are about equally important on a global level, with the former occupying 40% of the global surface area versus 34% for the latter. This implies that the removal of topsoil by soil erosion is potentially an important problem for most farmers, both at present and in the future, especially since the most vulnerable farmers have less access to additional inputs to compensate for erosion-induced yield losses.

Soil management or technology inputs is the second main factor that affects the erosion-crop yield relationship. In fact, it can

Table 2

Global distribution of soil types and their classification into insensitive (type A) and sensitive (type B) to erosion. Type C soils cannot be classified as either or are not suitable for agriculture.

	Type ^a	Surface (10 ⁶ km ²) ^b
Acrisols	A	10,0
Albeluvisol	B	3,2
Alisols	A	1,0
Andosols	A	1,1
Anthrosols	C	0,0
Arenosols	B	13,0
Calcisols	B	10,0
Cambisols	A	15,0
Chernozems	A	2,3
Cryosols	C	18,0
Durisols	B	n.a.
Ferralsols	A	7,5
Fluvisols	A	3,5
Gleysols	C	7,2
Gypsisols	B	1,0
Histosols	C	3,5
Kastanozems	A	4,7
Leptosols	B	16,6
Lixisols	B	4,7
Luvisols	A	5,5
Nitisols	A	2,0
Phaeozems	A	1,9
Planosols	B	1,3
Plinthosols	B	0,6
Podzols	C	4,9
Regosols	A	2,6
Solonchaks	C	2,6
Solonetz	B	1,4
Stagnosols	C	1,8
Technosols	A	n.a.
Umbrisols	C	1,0
Vertisols	A	3,4
Total A (%)		40
Total B (%)		34
Total C (%)		26

^a type A = deep, fertile soils; type B = shallow soils or soil with a layer impeding plant growth at shallow depth; type C = soils that are not suitable for agriculture or do not follow the previous classification.

^b Source: IUSS Working Group WRB (2007).

be singled out as the reason why historically higher crop yields have been observed in many agricultural systems, despite a severe erosion phase. This trend is shown conceptually in Fig. 6. Compared to the reference situation without erosion and technology inputs, erosion will yield to lower yields. However, the addition of technology inputs will increase yield levels significantly for both cases. The increase in most agricultural systems has been so significant that erosion effects were hardly felt, because comparison is made to the reference situation. Lal (1998) cites several examples of sites in Whitman County, Washington, that, despite dramatic increases in soil erosion rates, has seen a doubling of measured grain yield since the 1930s. Vanwalleghem et al.'s study (2011) of historical olive yield in SE Spain gave even more dramatic results. Despite a cumulative soil loss of almost half a meter over the last 250 years, they documented a continuously increasing production of olive fruit and overall biomass during the same period, which was the sole result of improved agronomic management. On the other hand, Bakker et al. (2004) concluded from a systematic analysis of plot data that soil erosion-induced losses were, on average, ca. 4% of the total crop yield per 10 cm of lost soil for crops under intensive agriculture. At the regional scale, predicted crop yield reductions by soil erosion were simulated for the US by den Biggelaar et al. (2001) and for Europe by Bakker et al. (2007). den Biggelaar et al. (2003) made a global review of erosion-yield relationships and found that relative erosion-induced yield losses were two to six times smaller in North America and Europe than in Africa, Asia, Australia and Latin America. They attributed this to the lower absolute yields in the latter countries and lower inputs to replenish the lost nutrients.

In spite of these uncertainties and site-to-site differences, it is important to quantify the effect of soil erosion on crop yields and agronomic sustainability over long time periods as this leads to essential insights on how soil erosion might have influenced the dynamics of agricultural societies and their ability to sustain population growth. It is clear from the first paragraphs that the natural potential of many soils around the world has decreased significantly due to historical soil erosion, and will continue to do so over the next decades. A recent review by Bindraban et al. (2012) clearly concluded that continuing soil degradation remains a serious threat to future food security. This calls for a modelling framework, that can again be applied both backward in time to reproduce historical evolution and forward to quantify the effect of future climate change or of socio-economic changes. In general, two broad modelling approaches can be distinguished. The first would be to assess the impact of soil erosion on crop yields at regional to subcontinental scales in a more qualitative way, while the second would be a more detailed, quantitative assessment which is necessarily limited to the local scale. Although Bindraban et al. (2012) distinguish a third, intermediate category of quantitative assessment at the country scale, the authors of the present work are not aware of such studies existing and argue that the amount of uncertainty related to erosion-yield predictions is too great to produce meaningful assessments at such spatial scales and over long time periods. Some studies published recently provide erosion-yield predictions of the first type (qualitative and large spatial scales) for the next few decades. At present, there are no studies estimating the effect of past soil erosion on the historical evolution of potential yield degradation. Ye and Van Ranst (2009) use an advanced erosion-yield model, with several degradation and population scenarios, that predicts up to a 30% productivity loss by 2050 for China. However, the large national scale obliged the authors to use qualitative soil degradation information, rather than quantitative soil erosion rates and truncation data. Also, as admitted by the authors, important factors such as changes in land use, water availability and climate change could not yet be taken

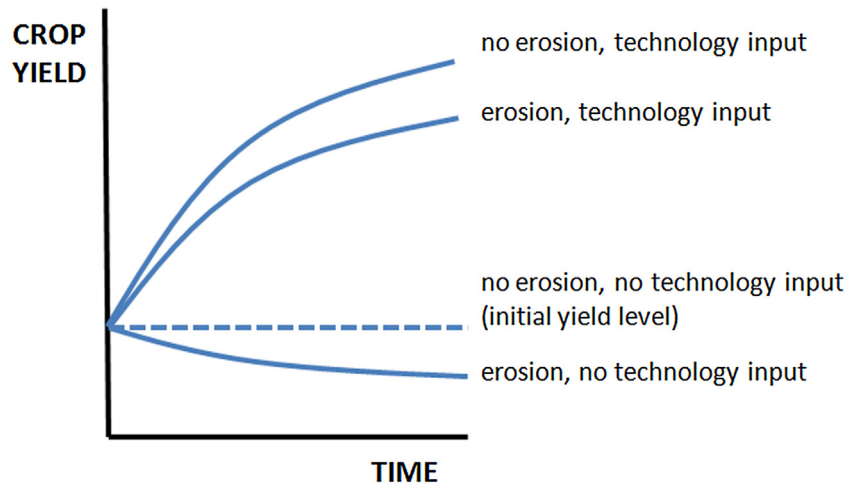


Fig. 6. Relationship between crop yield, erosion and technology inputs.

into account. Such limitations can be overcome at the local scale, as long as an accurate erosion–yield response curve can be established. A good example is given in Gómez et al. (Gómez et al., 2014b) for Mediterranean olive orchards that are heavily dependent on water availability during the growing season. Fig. 7 illustrates this point showing the cumulative impact of soil losses and soil truncation on the water balance and resulting yield under olive groves in Andalusia. Two contrasting situations are shown, taken from Gómez et al. (2014b).

Olive is a mostly rainfed crop, especially in steep areas under arid and semi-arid conditions. It is under these conditions that erosion is highest, and also that the role of soil as a water reservoir is vital, and plays a major role in determining the potential productivity of an olive orchard (assuming that other factors such as nutrient availability or pests and diseases are not limiting). In this simulation, two cases are compared. The first, Obejo, is typical for mountain areas and is characterized by shallow, coarse-textured soils. The second, Cordoba, represents orchards cultivated in the

rolling landscape of the Guadalquivir river valley on deeper soils with finer texture, fewer stones and better water-holding capacity. The two types of agricultural system correspond to, respectively, types B and A soils distinguished above. The first type can be expected to be more sensitive to changes under soil erosion pressure. The results shown in Fig. 7 clearly corroborate this: in the orchards represented by the Cordoba scenario, the deep soils result in a low yield sensitivity to the decrease in available soil depth, unless these values reach an inflexion point of close to 60 cm, where they effectively convert from a type A into type B soil. The simulations for this typology of olive orchard are, however, extremely sensitive to a reduction in average rainfall values, especially of below 400 mm, due to their larger crown canopies and greater water use. A contrasting situation occurs for the Obejo case, representative of extensive Mediterranean areas with shallow, stony soils. Here, there is a significant decrease in potential yield productivity due to the low water holding capacity of the soil, because these soils are coarser and rainfall infiltrates rapidly below

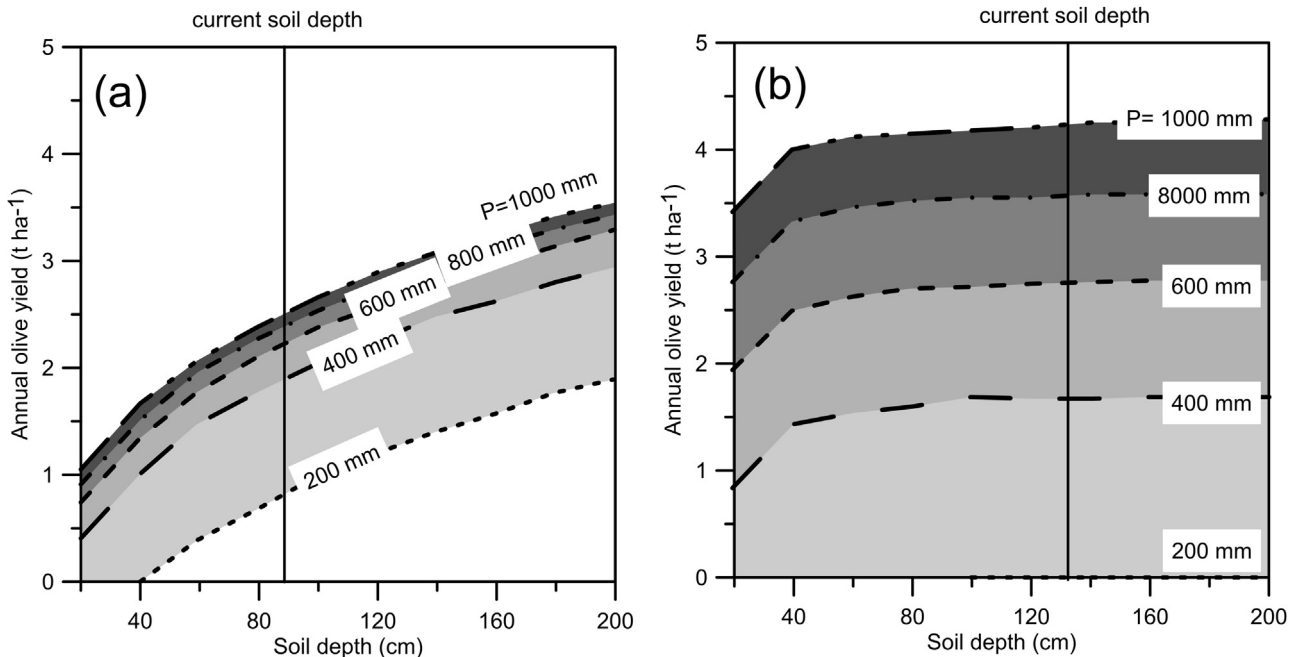


Fig. 7. Relationship for given annual rainfall (P) between potential yield and soil depth for the Obejo (a); and Cordoba (b) case study. Current soil depth at each site is indicated. Adapted from Gómez et al. (2014a,b).

the olive root zone into the relatively permeable parent material. Here however, the detailed simulations revealed that, surprisingly, climate change sensitivity is lesser, due to the smaller crown canopy and reduced tree density. Only below very low rainfall values, of around 200 mm, was a yield response observed. This truncation-yield model can then be connected to observed or modelled decreases of soil losses in olive orchards in the region, that range from 10 to 100 t ha⁻¹ yr⁻¹, with cumulative losses of up to 45 cm of soil in the last 250 years (Vanwallegem et al., 2011). Although water-holding capacity of degraded soils can be restored with management systems that increase soil structure and soil organic carbon, such actions are complicated. In the case of the soils mentioned in this example, this increase can be in the range of 7 to 18% after incorporation of soil amendments (García-Ruiz et al., 2012). This increase, however, will be concentrated near the soil surface so if soil erosion is not stopped first its positive impact will be rapidly lost again. These simulations at the local scale show the importance of quantitative modelling, as opposed to more qualitative estimates, since complex interactions, for example between rainfall and crop characteristics (in this case study, crown size and planting density) can alter the effects of erosion on crop yields.

6. Conclusions

Our analysis demonstrates the strong links between historical land use and soil management changes, soil erosion and agricultural sustainability. Common trends have been identified in the historical evolution of land use and soil management intensification worldwide. The driving forces behind this intensification are a complex mixture of population, economic and social variables, but recent research indicates that even more complex issues might underlie observed changes. Its effects on global soil resources are equally complex.

However, some general patterns emerge from the review of existing studies on historical soil erosion and their comparison to global land use dynamics. First phases of significant soil erosion are shown to correspond well with first significant land use changes at a global scale. At the same time, our novel dataset shows important gaps in areas such as Africa or Asia, where more research on historical soil erosion is needed. Historical soil management is equally important and also very closely linked to soil erosion rates, but very few studies exist on the topic. Only recently, have we started to see the appearance of case studies that analyze the relationship soil management-erosion at a historical scale (e.g. Vanwallegem et al., 2011), but much more empirical work is needed with a sufficiently high spatial and temporal resolution in order to resolve this coupling.

More work also needs to be done on the prediction of the impact of past soil erosion on soil resources and agricultural crop yields. From the existing body of work, it is apparent that a modelling approach is needed that links crop and erosion models and that incorporates the interactions between the different processes involved. One of the pathways in which these studies could be pursued can be through the use of relatively simple, well-understood models that could be calibrated through available information. Not enough historical information is generally available to allow the use of more complex erosion or crop models. Two main limitations were identified in this review, that were similar between backward modelling of past erosion processes and forward modelling of future erosion dynamics. The first area of limitations is related to feedbacks that are not represented in current models. Such feedbacks include dynamic topography and evolving soil properties, but also more complex responses related to human behavior and socio-economics. Those responses might result in the adoption of conservation practices

or land use changes. A second main area of limitation is the uncertainty related to constraining the model input factors. Reconstruction of past climate and land use is common among many branches of natural sciences and agricultural management is perhaps the most challenging to tackle. Historical or archaeological records might provide answers to where and when those certain changes in agricultural practices occurred, but for broad regional evaluations that information will have to be completed by multi-disciplinary modelling.

Finally, the last hurdle is to link the historical evolution of soil erosion to agricultural crop yields. Both local soil resources such as technology inputs or agricultural soil management are key to understanding this coupling.

Acknowledgements

This research was funded under the research projects “AGL2012-40128-C03-01”, “AGL2012-40128-C03-02” and “AGL2012-40128-C03-03”. The first author is grateful for the funding by the Ministry of Economy and Competitiveness under the Ramon y Cajal Fellowship programme. We thank Dr. Jed Kaplan for sharing the KK10 land use scenario data shown in Fig. 2 and for many stimulating discussions on this topic, especially in the framework of COST Action ES0805 The Terrestrial Biosphere in the Earth System.

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