ANALYSIS
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Tillage exacerbates the vulnerability of cereal crops to drought

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Soils used for crop production cover 15.5 million km² and almost all have been tilled at some point in their history. However, it is unclear how the changes in soil depth and soil properties associated with tillage affect crop yields. Here we show that tillage on slopes thins soils and reduces wheat and maize yields. At the landscape scale, tillage erosion gradually reduces crop yields as the duration and intensity of tillage increase. Over the next 50–100 yr, the overall yields are likely to further decline as modern mechanized agriculture accelerates the process of tillage erosion compared with centuries of non-mechanized tillage. Arresting this downward trend will require more widespread adoption of no-tillage practices and avoidance of down-slope cultivation. The downward pressure on landscape-scale yields due to tillage erosion is expected to be amplified by climate-change-induced increases in dry spells during crop growth.

Tillage moves substantial amounts of soil down-slope, estimated to be approximately a fifth of that associated with water erosion and over twice the amount globally moved by wind erosion[1]. Soils used for crop production cover 15.5 million km² and almost all have been tilled at some point in their history, yet the role of tillage in reducing soil depth remains an under-recognized threat to plant production. We also know little about how the changes in soil depth and soil properties associated with tillage affect crop yields and threaten the delivery of the UN Sustainable Development Goals. On sloping land, tillage thins soils on slope convexities and causes soil accumulation in concavities (Fig. 1, left). Tillage translocation depends upon the speed and depths of tillage and the implements used, with inversion tillage tending to move more material than non-inversion cultivations[2]. Only no-tillage systems do not move substantial amounts of soil. Soil translocation by tillage is also affected by soil properties, soil status variables (for example, moisture, consolidation after preceding tillage) and slope gradient, with the greater movement occurring on steeper slopes and where there are changes in slope. In areas with a long history of cultivation, soil redistribution by tillage almost certainly started when the land was first cultivated for agriculture (Fig. 1, left); however, rates of movement from hand tools and ploughs pulled by animals[3] are much lower than those associated with mechanized agriculture and these rates have accelerated in recent decades as agriculture has intensified and machinery has increased in size and power[4].

As soils become thinner, and if the tillage depth is not reduced, material from the subsoil is mixed with the topsoil and over time the topsoil properties approach those of the subsoil (Fig. 2). This leads to a reduction in the quality of the A horizon, which contains most of the soil nutrients and biological activity and stores a substantial amount of the water needed for plant growth. In some case soil horizons with physical or chemical properties that are inhospitable for plants approach the surface. Therefore, soils on convexities where soil is lost are shallower, and hence mostly hold less water, are depleted in nutrients and carbon, and have poorer chemical and physical properties. The contrary is true for the concavities where the soil translocated from upslopes is accumulated. Here soil accumulates and is mixed with the existing A horizon, leading to deeper soils that are enriched in nutrients and carbon and able to store greater amounts of plant-available water. However, prolonged landscape erosion might also result in a degradation of topsoils at depositional sites because over time subsoil exposed to the surface at eroded sites will be redistributed to depositional sites[5] (Fig. 2).

Although the question of how erosion affects agricultural production has long been a research topic, there have been only isolated studies on the effects of tillage erosion[6]. This is surprising because tillage erosion affects all hilly agricultural landscapes, not just those prone to water and wind erosion. Thus, estimates of how tillage erosion affects agricultural yields at the landscape scale, especially under increasing mechanization, are lacking.

To illustrate the general problem of erosion and yield loss, we synthesize published information on the impacts of soil thinning on crop productivity. We then utilize soil redistribution and crop growth models to examine the effects of tillage over a landscape, where soils on the convexities lose and those in the concavities gain soil, to see whether the potential gains in crop production due to increased soil depth outweigh the losses due to thinning soils. Next, we examine the potential future impacts of likely changes in tillage equipment on soils and crop production to 2100. Finally, we discuss the wider implications of these findings for the sustainability of crop production in arable landscapes.

Results

Tillage erosion and crop productivity. Tillage results in reduced plant productivity in those parts of the landscape where the soil thins. Thinning soils have reduced water storage, and, where no fertilizers are applied, lower nutrient availability which leads to lower crop productivity. The negative relationship between soil loss and crop productivity determined at the plot scale by removing topsoil, so-called desurfacing experiments, is well documented and consolidated in several review papers (for example, ref. [7]). The effect of soil loss upon biomass production or yield is significantly more pronounced in the case of zero or low fertilizer inputs (Fig. 3) and therefore poses a significant problem in low-input, subsistence farming systems[8]. However, almost all these studies relate the change in crop yield to soil loss, but not to the change in soil depth, which has the potential to be a better predictor. The reduction in soil depth, and the associated ability of the soil to store and

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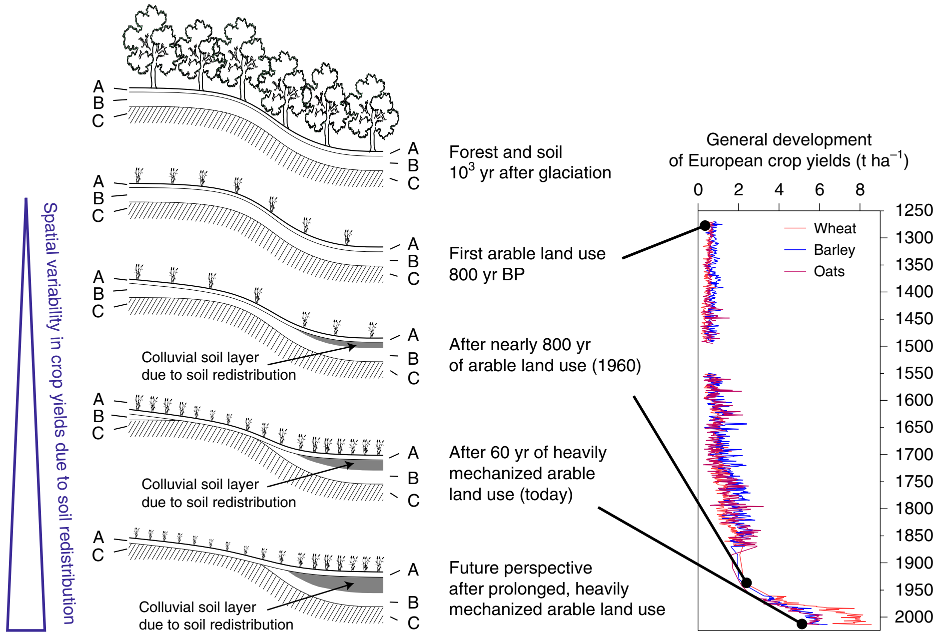

Fig. 1 | Changes in soil properties and crop yields. Schematic illustration of the increase in spatial heterogeneity in soil properties (indicated as change in A, B and C soil horizons) due to (tillage) erosion following conversion from forests to arable affecting heterogeneity in crop yields. The yield effect is masked at the field scale due to the large increase in yields within the later part of the last century. Data illustrating changes in annual yields in Europe are taken from the United Kingdom[49].

supply water to plants, is more important in high-input agricultural systems than the loss of nutrients, which can, at least in the short term, be replaced by fertilization[9]. The loss of water storage capacity will be most important during prolonged dry spells or periods of drought during the cropping season. In the few experiments[10–12] which have mechanically removed soil and which have related soil depth to crop yield, there is an indication that the loss in yield is less pronounced than in those which have only measured soil depth reduction (Fig. 3). However, the relationship is uncertain: the experiments do not encompass situations where, because of soil thinning, only a little soil is left to be cultivated; nor do they include areas with pronounced drought during the growing period; moreover, the sample sizes are small, with crops limited to barley and maize. The relationship between soil depth and crop production is further complicated due to the properties of soil parent material, which may extend the rooting depth beyond the depth of the soil. For example, weakly consolidated and porous parent materials, such as loess, are penetrated by plant roots to access water. Although deposition of eroded material may cause soils to thicken, and thus, in periods of drought, have higher crop yields than comparable shallower soils, there are no standardized plot studies to illustrate this.

At the landscape scale, the response of crop productivity to landscape positions is complex. In Denmark crop yields were lower on slope convexities and higher on concavities and were related to changes in soil phosphorous content[6], whereas in England the crop response was more complex with locations associated with tillage erosion displaying nutrient depletion and low rates of crop production; however, there were also areas of low production associated with aggrading areas, and no consideration was given to changes in soil depth or to trade-offs between yields where soils are thinning and where they are thickening.

Tillage-induced soil redistribution and landscape-scale response of crop productivity. To understand the landscape-scale impacts of soil truncation and colluviation due to tillage on biomass production we coupled the well-established crop model AQUACROP[13] and the tillage erosion component of the model SPEROS-C[14]. We then applied the model for both wheat and maize in a test region of approximately 200 km^2 in the Uckermark, 100 km north of Berlin, Germany (Supplementary Fig. 3). This test region was chosen because: (1) it has been used for crop production over the last millennia, with intensive mechanization of agriculture and substantially enlarged field sizes since the 1960s under the German Democratic Republic; (2) it represents a typical ground moraine landscape, found in large areas of Europe and North America, dominated by a rolling topography and soils developed on glacial tills, a highly compacted and difficult-to-root parent material; (3) soil truncation due to tillage erosion is known to be widespread in the area[15,16]; and (4) results from earlier studies in the region dealing with different erosion processes[15] and erosion/biomass interactions can be utilized[16].

A series of biomass/soil-depth responses for the period 1964–2017 were produced using the AQUACROP model driven by soil properties generated by mixing an average non-eroded profile from the region (soil depths to C horizon, 1.4 m) following the loss or gain of soil at the surface and measured climate data.

There was a strong interaction between climate and the yield/soil-depth response. In wet years, when plant water is plentiful, there is a smaller difference in biomass production between shallow and deeper soils (Fig. 4, left) than in a dry year, when crops rely on water stored in the soil profile and the difference in biomass production between thinning and thickening soils is amplified (Fig. 4, left). Modelled reduction in yield started earlier with soil truncation in the case of winter wheat as compared to maize, but maize biomass immediately fell if soil thickness dropped below about 0.75 m. At depositional sites modelled winter wheat biomass profited from deeper soils, while this was not the case for maize. This general behaviour was also found when comparing remote-sensing-derived biomass proxy variables (Enhanced Vegetation Index, EVI) with patterns of modelled tillage erosion classes (Fig. 4, right).

The yield information resulting from the AQUACROP modelling of different soil profiles was extended across the landscape using SPEROS-C to model the spatially distributed tillage-induced

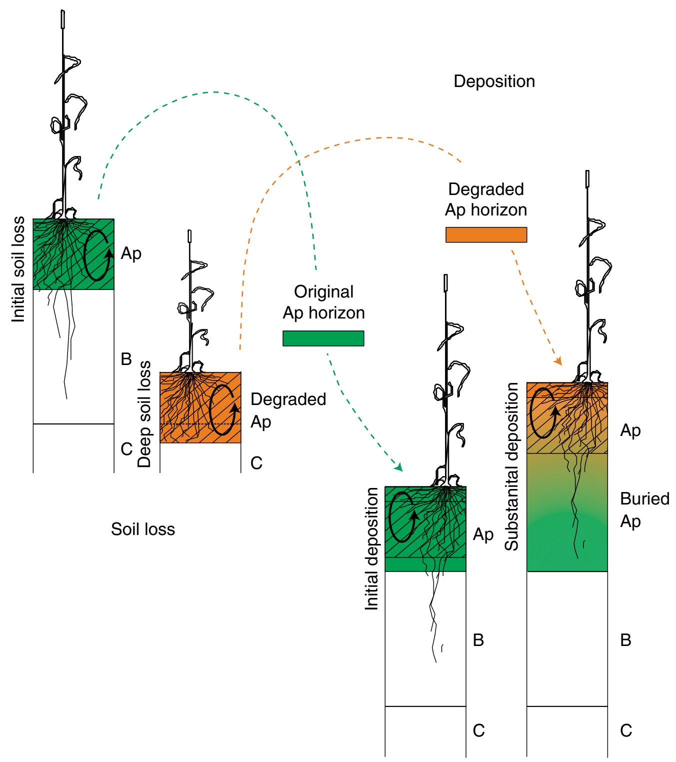

Fig. 2 | The effects of soil loss and deposition on topsoil properties. Schematic illustration of the effects of soil loss and deposition on topsoil properties following initial to long-term soil loss and deposition. Ap, plough horizon; B, mineral horizon; C, parent material of the systematic soil profiles.

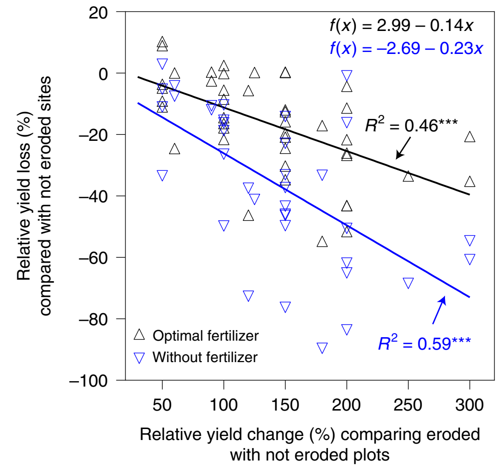

Fig. 3 | Effects of topsoil removal on yields. Effects of removing topsoil on wheat yields under optimal fertilization and no fertilization[9,17,33–40].

soil thinning and thickening for about the last 1,000 yr. To achieve this, low values of tillage intensity for the first 940 yr and much higher values for the last approximately 60 yr were assumed, and the latter were calibrated against soil redistribution patterns derived from radionuclide data of a small sub-catchment[15] (Supplementary Fig. 3). Soil redistribution due to tillage lowered overall simulated biomass production in a 'normal-to-dry' year (Fig. 4) from 215,000 t to 202,000 t for wheat, and from 276,000 t to 269,000 t for maize. In a wetter year yield reductions were lower with 317,000 t of wheat and 415,000 t of maize reduced to 308,000 t or 411,000 t, respectively (Fig. 5).

Future production. Agricultural production has been possible in the Uckermark for at least 1,000 years. Our modelling results suggest that by continuing to till the Uckermark soils, mean yields on the landscape scale will continue to decline and that this decline increases with tillage intensity and reduced water availability (Fig. 5). In 50 yr, we expect reductions in normal-to-dry years winter wheat biomass of between 6.6% and 7.1%, depending on the intensity of tillage (see Fig. 5 for scenarios). These differences increase to between 8% and 10% at 100 yr. Maize biomass was less affected in normal-to-dry years, with reductions of between 3.1% and 4.0% for the 50 yr time horizon and 3.9% to 5.9% at 100 yr in the future. In wet years reductions were smaller at between 3.3% and 4.4% for wheat and between 1.1% and 1.9% for maize for the 50 yr scenario, whereas for the 100 yr scenarios, reductions are between 4.3% and 5.9% for wheat and between 1.9% and 3.2% for maize.

Discussion

The reviewed desurfacing experiments illustrated, as expected, the negative effect of soil loss on crop yields (Fig. 3). Experiments that add topsoil to plots to test potential positive yield effects[17] are much rarer. Therefore, even if these plot experiments give a first indication, it is difficult to use them to understand the effect of soil redistribution on crop yields at the landscape scale. To overcome this, studies have determined soil loss and deposition using radionuclide erosion tracers, such as ^{137}Cs, associated with atomic weapons testing, and compared them with yield data[6,18]. However, such tracer-based approaches miss the long-term effect associated with centuries to millennia of soil loss and gain as they only focus on the last approximately 70 yr to explain spatial distributed yield effects. They are also, mostly, limited to small test sites as the effort required for soil sampling and analysis is substantial. Our modelling approach allows a much longer perspective, focusing on soil loss or gain since start of cultivation in the test region roughly 1,000 yr ago. Obviously, modelling land management over such a long time span is challenging and model parameterization requires a number of assumptions, such as the historical tillage intensity (see Supplementary Information for a discussion of uncertainty). However, as we were interested in relative changes in spatial variability of crop biomass production following soil redistribution, which we know to be important for the test area[16], we are confident that our parsimonious model system is robust enough to illustrate the general problem associated with tillage erosion in regions only slightly affected by other erosion processes[15].

Our modelled landscape-scale yield losses suggest that deeper soils in depositional environments at least partly compensate for yield losses in erosional settings and that agricultural production is likely to continue in the Uckermark. Tillage erosion reduced landscape-scale yield potential, but yields did not collapse. As in other empirical studies focusing on erosion since the 1960s[6], the significant differences in soil properties in eroded or depositional environments resulted in differences in crop yield, with the lowest yields on the hillslopes and the highest in the valley bottoms. The lack of a major decline on yields may be because the area with more than 0.3 m soil loss, resulting in a substantial yield loss (Fig. 4, left), is relatively small in our test region. Our findings are similar to results from a smaller catchment in Turkey[19]. In that study soil truncation was modelled along with its potential yield effects on winter barley for a mountainous catchment in the Mediterranean region over a period of 4,000 yr, with catchment-scale crop yields estimated to drop by 22% from 2.80 $t\,ha^{-1}\,yr^{-1}$ before widespread deforestation to 2.19 $t\,ha^{-1}\,yr^{-1}$ at present, whereas deeper soils in the valley bottoms at least partly compensated for substantial yield losses on the hillslopes. An additional factor that may explain the larger differences in the Turkish study[19] is that the work focuses

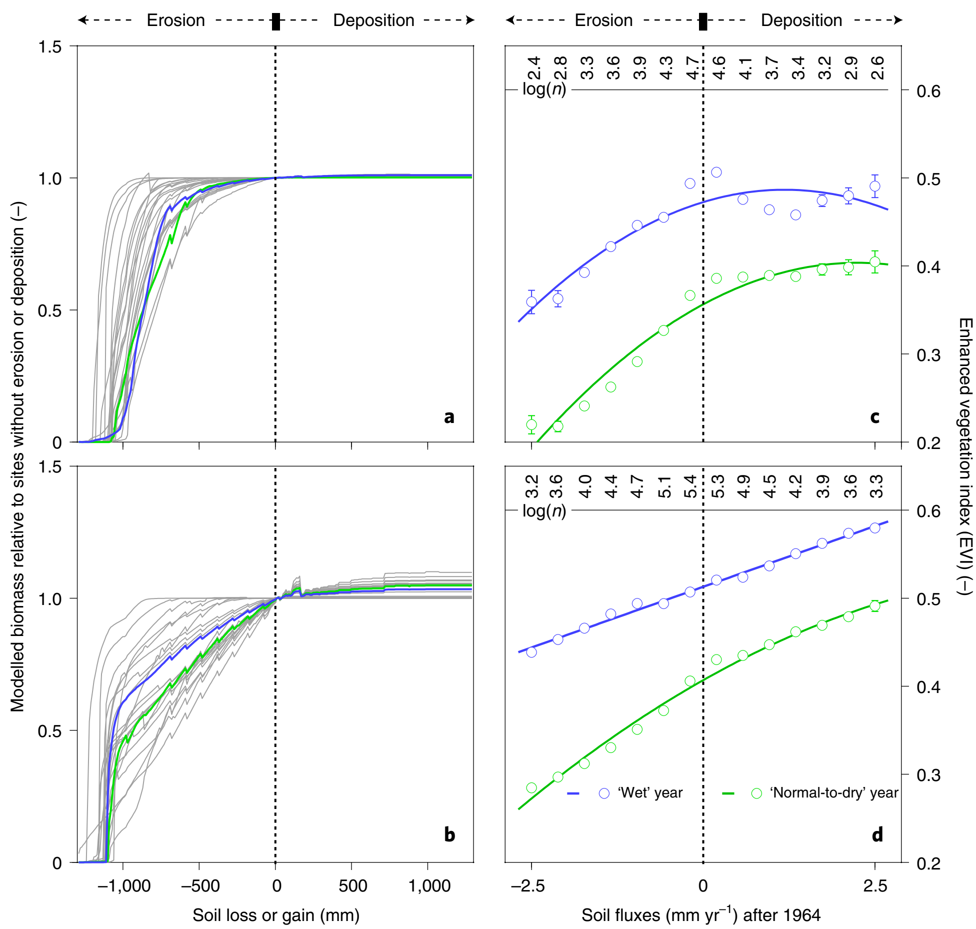

Fig. 4 | Modelled biomass production and EVI. a,**b**, AQUACROP-modelled biomass production of maize (**a**) and winter wheat (**b**) depending on soil loss and gain. Grey lines result from the different climate between 1992 and 2017, while the blue and green lines represent a 'wet' and 'normal to dry' year used for further analysis. Data are given as relative biomass using biomass from profiles with no soil erosion or deposition as 1. **c**,**d**, Mean EVI as a proxy variable for biomass in different classes of modelled tillage-erosion-induced soil fluxes for maize (**c**) and winter wheat (**d**). The EVI was derived from RapidEye satellite images from 2010 and 2015 (bands in the visible–near infrared spectrum; resolution, 5 × 5 m^2). The tillage erosion modelling and the EVI analysis are described in detail in ref. [16]. Note that the modelled soil fluxes are aggregated in classes and are only indicators of soil truncation and/or colluvial deposition because the duration of tillage erosion was not analysed. *n*, number of pixels analysed per soil erosion class; error bars indicate 95% confidence intervals of EVI within each class.

on areas of substantial water erosion[18] representing environments with steep slopes and heavy rainfall events, where water erosion is well-recognized to be an important soil threat. In contrast, in the Uckermark water erosion plays only a minor role[15].

A more substantial overall relative yield effect was modelled for winter wheat versus maize (Fig. 5). Comparing the remote-sensing-based biomass proxy with the modelled biomass based on soil loss and gain (Fig. 4) gives an indication why the modelled maize yield effect following landscape-scale soil redistribution is somewhat underestimated. While modelled winter wheat yields immediately react to soil truncation (Fig. 4a), this is not the case for maize, which does not react to soil thinning of less than approximately 0.3 m (Fig. 4a). The modelled response reflects the parameterization of the crop model, which produces a higher water-use efficiency of maize over winter wheat, meaning that the maize produces more biomass per litre of water than the wheat and therefore is less prone, in the model realization, to a reduction in water availability due to soil thinning. This is in agreement with results from the Berlin area[20], but not all studies concur. Europe-wide modelling suggests that in Europe maize is more substantially affected by droughts than wheat[21] which is supported by findings[16] based on remote sensing of our study area (see also Fig. 4, right) that slightly larger landscape-scale yield effects can be expected in the case of maize. Therefore, it is likely that our modelled yield effects are conservative for maize.

Based on the reference soil profiles from the Uckermark, we can identify a threshold of approximately 0.3 m of soil loss beyond which affected soils contribute little biomass (Fig. 4). In our future scenarios, the area of soil in this class (soil thinning, >0.3 m) increases up to 100%. The increase in the area of soils that are non-productive highlights the need for urgent action to reduce soil thinning due to tillage.

In our scenarios we addressed different potential trajectories of future tillage practice (Fig. 5), but we did not use future climate scenarios, which indicate longer dry spells or phases of droughts during the growing period in the region[22]. However, our analysis comparing a normal-to-dry year with a wet year clearly indicates that the downward landscape-scale effect of tillage erosion on crop yields in the region is more pronounced in case of drier conditions during the growing season (Figs. 4 and 5). Hence, there is clear evidence that projected future climate conditions will amplify the downward landscape-scale yield effect. Moreover, we ignored the potential effects of soil quality loss due to deposition of depleted topsoil material coming from strongly eroded sites (Fig. 2). This will

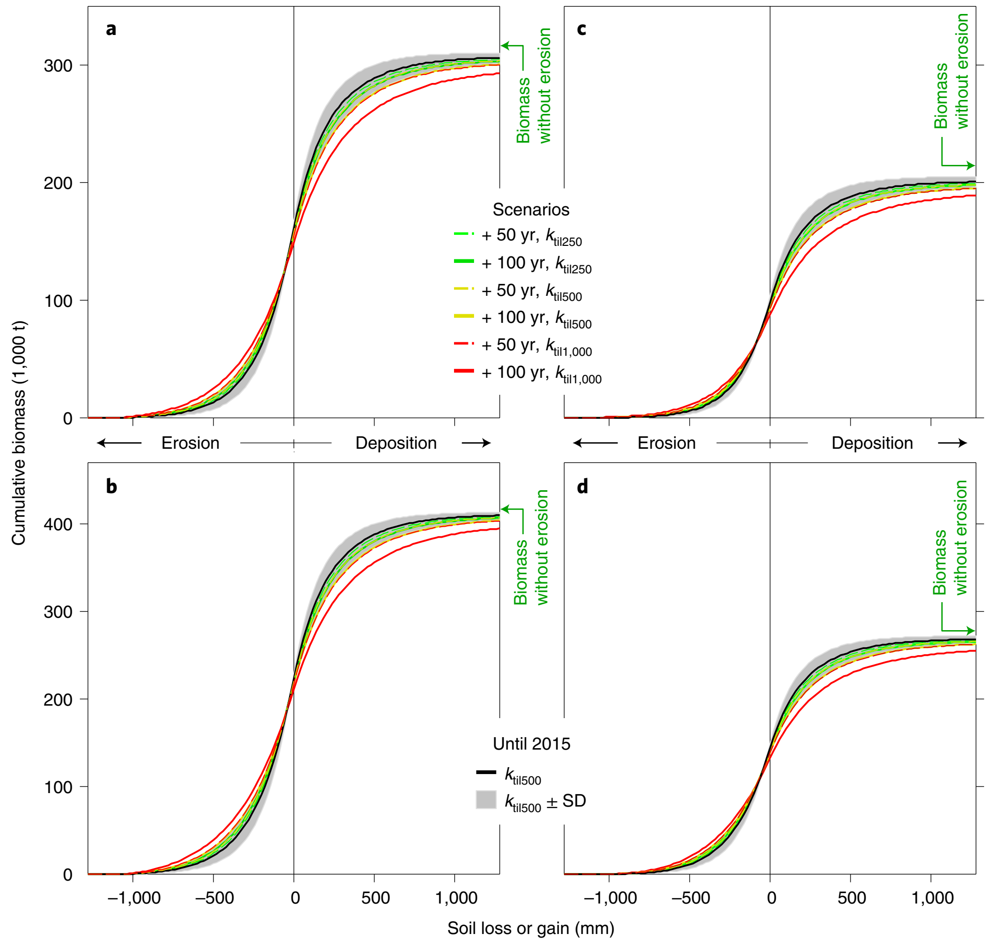

Fig. 5 | Changes in modelled cumulative landscape-scale biomass production. a–d, Changes in modelled cumulative landscape-scale biomass production for different tillage scenarios for 'wet' (**a**,**b**) and 'normal-to-dry' years (**c**,**d**) for winter wheat (**a**,**c**) and silage maize (**b**,**d**). Definitions of years are given in the Supplementary Information. The bold black lines indicate the actual mean with grey shaded areas giving the uncertainty of the results (±1 s.d. of modelled mean soil loss or gain). The coloured lines indicate six scenarios for 50 and 100 yr of additional tillage, with reduced tillage (green), tillage equal to the mean of the last 50 yr (yellow), and increased tillage representing the use of heavier and faster machines (red). The tillage intensity of the different scenarios is given as the tillage coefficient k_{til} of 250, 500, and 1,000 used in SPEROS-C. The green arrows on the right side of the panels indicate modelled cumulative biomass without soil loss or deposition. Note: increasing soil loss leads to decreasing biomass production but the cumulative biomass production of erosional sites (as given in the figure) increases as the area affected by soil loss is increasing.

also strengthen the negative yield effect which might foster future adaptions of management towards irrigation in this already dry region of northern Germany.

The increasing mechanization of agriculture, with significant innovations in agricultural machinery during the 1950s and 1960s increasing the size, weight and speed of tractors and cultivators, has played a significant role in accelerating tillage erosion. For example, the front and rear axle loads of tractors in Illinois (United States) have increased since 1960 by a factor of four and two, respectively[4]. It is hard to find measurements of pre-mechanization tillage speeds, but contemporary measurements of horse-drawn ploughs in Ethiopia place speeds at between 3.2 and 4.7 km h^{-1} (ref. [23]), or between 1.0 and 1.8 km h^{-1} in the case of ox-drawn mouldboard ploughing in Cuba[3]. By 1925, with the introduction of the tractor, a plough would typically be pulled through the soil at 4–6.4 km h^{-1} (ref. [24]) and ploughing speeds are now in the range of 6.4–11.3 km h^{-1} (ref. [25]), or even higher in case of non-inversion tillage (personal communication with farmers from the test site). Such an increase in tillage speeds accelerates tillage erosion[2] and hastens the thinning and thickening of soils in sloping landscapes. There is some indication that the rate of tillage translocation in mechanized agriculture may be slowing. In recent decades there have been substantial shifts in the types of machinery used to cultivate soils. At one time, mechanized cereal production was dominated by the mouldboard plough as a primary cultivation tool. Although, still widely used, there has been a shift towards non-inversion tillage and no-tillage systems. However, non-inversion systems utilizing chisel ploughs in combination with powerful tractors have been demonstrated to move as much, if not more, soil than plough-based systems[26] due to high tillage speeds. No-tillage systems, where seeds are sown directly into undisturbed soil, translocate an order of magnitude less soil than conventional tillage systems[27] and therefore represent the best option for reducing the translocation of soil in agricultural landscapes.

As our results suggest that tillage erosion rates accelerated during phases of intensified agricultural mechanization, it is clear that the most pronounced future changes in tillage erosion can be expected in those regions where agricultural mechanization is still minimal. The region with the largest mechanization gap, and hence largest potential for accelerating tillage erosion, is Africa. Tractor use in sub-Saharan Africa was 1.3 tractors per 1,000 ha in 2002 (the last date for which the Food and Agriculture Organization[28] holds data on farm machinery for much of the world) compared with 9.1 tractors per 1,000 ha in South Asia and 10.4 tractors per 1,000 ha in

Latin America for the same year[29]. With increasing areas of land coming into cultivation[28] and gross national per-capita income rising across sub-Saharan Africa (The World Bank, 2020), it can be assumed that agricultural mechanization may also increase in the region[30], presenting a risk to the soils of the continent and highlighting the need to develop no-tillage systems adapted to the regional socioeconomic and environmental conditions.

Tackling the impact of tillage erosion is problematic. In cases, such as the Uckermark, where redistribution may have already gone too far, the only option is to relocate soil from the base of the slope to the top of slope in an attempt to rebuild soils. Anecdotally, farmers have been doing this for generations (D. Lobb, personal communication) and it has been demonstrated experimentally that adding 10 cm of topsoil to severely thinned hillcrests in Manitoba, Canada led to significantly greater yields than on those sites with no soil additions[31]. Clearly, the best option is to prevent tillage erosion altogether and the adoption of practices that have virtually eliminated soil redistribution has taken place at scale. Large tracts of arable land in South America are managed using no-tillage systems that minimize soil disturbance[32] and are one of the central platforms of conservation agriculture. In addition, soil conservation practices that seek to reduce the effective slope of the land, for example, terracing or the use of contour grass strips, force farmers to cultivate parallel to the contour, reducing tillage erosion rates.

Further work is needed to adapt the principles of no-tillage and/or at least to take tillage erosion into account. This could include using precision agriculture to manage tillage speeds and depths in sloping agricultural land to arrest the redistribution of soils in agricultural landscapes. In addition, we need to understand better how soil redistribution impacts on soil carbon, nitrogen and phosphorus cycles and water availability in contrasting arable landscapes and how these changes impact on biomass production. This understanding should lead to the development of models which will allow us to assess the future sustainability of agricultural production in response to climate, land-use and technological change.

Methods

Measured soil loss and yields from plots. To identify relevant data on soil loss or soil truncation on biomass production and yields, we conducted a systematic search of the experimental erosion literature. We focused on soil surface removal plot experiments (often called desurfacing experiments), where yields on plots, typically of the order of a few tens of square metres, without artificial removal of surface soil are compared to treated plots. We used predefined search terms (desurfacing, soil removal, erosion and yield) in the ISI Web of Knowledge and Scopus databases. From the search results we identified two sets of plot studies: (1) studies containing plot data on the extent of desurfacing and its effects on wheat yields, which also contain fertilized and not fertilized areas, allowing a comparison of the effect of surface lowering on yields in high-input versus low-input agricultural systems (Fig. 3)[9,17,33–40]; (2) studies where the relative or absolute reduction of soil thickness was given—here only three studies[10–12] analysing maize and barley yields were identified.

Modelled soil redistribution and biomass for landscapes. We used well-established modelling tools to illustrate the impact of long-term tillage on crop production in the Quillow catchment representing the Uckermark region (Supplementary Fig. 3), an area of predominantly arable farming in northern Germany, following a three-step approach: (1) area-specific non-eroded soil profiles modified to represent soil truncation or colluvial deposition were used to model biomass production under different soil erosion conditions with AQUACROP[13]; (2) long-term landscape-scale soil redistribution due to tillage was modelled based on the spatially distributed model SPEROS-C[14,41]; (3) profile-based modelled biomass and modelled spatially distributed soil thinning or thickening due to tillage erosion were then combined to determine overall impacts of soil redistribution by tillage on biomass production on a landscape scale.

Test catchment. The Quillow test catchment (196 km^2) is located about 100 km north of Berlin (Supplementary Fig. 3). It represents a typical ground moraine landscape, formed after the retreat of the Weichselian glaciers (~15,000 yr BP), typically found in large areas of northeastern Germany. The hilly area is characterized by small hummocks and a large number of kettle holes draining via groundwater. The mean slope of the catchment is about 7%. Land use is dominated by arable land (~70%). Due to its fertile soils large parts of the catchment have been used for agricultural production for over 1,000 years and some areas since Neolithic times[42,43]. Beginning in the 1960s, agriculture was intensively mechanized, and field sizes were substantially enlarged (current mean field size, 22 ha). Crops typically planted are winter wheat (*Triticum aestivum* L.), winter barley (*Hordeum vulgare* L.), rapeseed (*Brassica napus* L.) and maize (*Zea mays* L.). The catchment is characterized by a subcontinental climate with an average annual air temperature of 8.6 °C and a mean annual precipitation of about 500 mm (30 yr average, 1981–2010). The average precipitation during the growing season (April–September) is approximately 350 mm (1989–2017) (meteorological data from the Dedelow Experimental Field Station of the Leibniz Centre for Agricultural Landscape Research (ZALF) (53° 36′ N, 13°80′ E; Supplementary Fig. 4)). The primary soils developed from glacial till are Luvisols[44], but typical sequences of erosion-affected soils can be found due to the long history of arable land use and the hilly terrain.

Daily soil temperature, global radiation, relative humidity, air temperature, potential evaporation and wind speed between 1992 and 2017 used for modelling were taken from climatic records from the Dedelow Experimental Field Station. Due to the 25 yr record a wide range of annual and seasonal climatic variation (especially precipitation; Supplementary Fig. 4) was used for the modelling. Data for the spatial distribution of the EVI, a proxy variable for biomass production, of winter wheat were derived from RapidEye data in 2010 and 2015 taken from Öttl et al.[16]. Note that 2010 and 2015 represent a 'wet' and a 'normal-to-dry' year regarding precipitation during the growing season (Supplementary Fig. 4).

Topography for use in SPEROS-C is represented by a 5 m × 5 m Lidar-based digital elevation model (Landesamt für Umwelt & Landesvermessung und Geobasisinformation Brandenburg, 2012). Field boundaries are also important for modelling tillage erosion and are taken from an earlier study[16] and represent the situation in 2010.

Crop modelling. We assumed that crops were not nutrient limited because farmers used inorganic fertilizers to manage crop nutrition. Therefore, we focused on the impact of soil redistribution on water availability. To simulate crop response to change in water availability in different soils the daily timestep FAO–AQUACROP model[13] was used. The Food and Agriculture Organization offers a menu-driven version of the model and versions suitable for executing without a graphical interface. For this work we used a version of the model designed for use in a geographic information system (GIS) environment (AQUACROP GIS file builder and AQUACROP plugin v.4.0) which allowed multiple runs to be made quickly with different parameter sets. The model is described in detail in Steduto et al.[13], but in brief it calculates transpiration which is translated into biomass. This is adapted to local conditions using the biomass productivity parameter which is normalized for evaporative demand and air CO_2 concentration. The crops' response to water is simulated based on water stress and its effect on canopy expansion, stomatal control of transpiration, canopy senescence and the harvest index. Soil water availability is determined for up five soil layers for which the user specifies characteristics. The model calculates a water balance for each time step based on infiltration, drainage, runoff, root uptake in different layers, deep percolation, evaporation, transpiration and capillary rise.

Soil data. The soil component of AQUACROP was parameterized based on three representative soil profiles unaffected by soil loss and deposition drawn from the database of ZALF, Müncheberg, Germany (Supplementary Table 2). These profiles (Calcic Luvisols; WRB, 2015)[44] represent about 20% of arable land, while in general the Uckermark soils have been greatly modified by soil erosion over the past centuries[42,45]. The profiles were then combined into one standard profile as follows. Almost all of the pedogenic horizons were the same for the three soils and only differed in their depth and slight textural variation. Therefore, the mean values of depth and sand and clay content were calculated for each horizon. A different approach was required for one of the profiles where two of its horizons did not exist in the other two soils (Bt3 and Bt4). In this case the data for the Bt3 and Bt4 horizons were combined with the Bt2 horizon for that soil to give a new average condition. The mean values for horizon depth and the hydraulic parameters for each horizon used in AQUACROP are given in Supplementary Table 3.

The response of crop yield to tillage erosion was simulated based on modifying the standard profiles (Supplementary Tables 2 and 3) as follows. In the case of deposition, the profile was grown by changing the Ap horizon depth. The properties of the new material were assumed to be the same as that of the existing Ap horizon. In the case of a profile losing soil, the soil was lost from the surface soil and the depth of each horizon, except that at the base of the profile, which extended into the horizon below by an amount corresponding to the eroded depth. The properties of each of the horizons were then recalculated by mixing in the relevant proportion of the horizon below. For example, the loss of 10 mm from the surface of a 100-mm-thick Ap horizon would mean that 10 mm of material from the B horizon was incorporated into the Ap and the properties of the Ap would reflect a mixture of 90% of the original Ap and 10% of the B horizon. New soil profiles representing 1 cm steps of soil thinning and thickening were created for a maximum soil loss and gain of 130 cm, which in case of soil loss would represent a total loss of the A and B horizon. Based on the changes in texture, soil organic carbon and thickness of the new profiles the hydraulic properties

(Supplementary Table 3) of the new profiles are derived within AQUACROP using standard pedotransfer functions[46]. However, it is important to note that moisture content at field capacity and wilting point in the eroded profiles will be somewhat underestimated because the dynamic replacement of soil organic carbon is ignored at the newly created eroded profiles. This will affect plant-available water but not nutrient supply because we assume no nutrient limitation of any of the profiles. Overall, 260 new soil profiles representing different stages of soil thinning and thickening were created and used for modelling biomass production for the different climatic conditions. These new profiles and their crop yields were translated to the entire test catchment using the modelled spatially distributed soil thinning or thickening based on SPEROS-C (see below).

Crop parameters were drawn from standard AQUACROP files for maize and winter wheat and modified for northern European conditions to reflect planting timings and plant growth curves. To check the performance of modelled crop biomass and yield for maize and winter wheat in the Uckermark, biomass was modelled for the average undisturbed soil profile for the years 1992–2017 and compared with the mean biomass for the region as given in agricultural statistics (see Supplementary Information for more detail). There was no significant difference in wheat yield between the simulated and measured data over the study period; however, simulated maize biomass was an average of 18% lower year-on-year. We concluded that that the modelling approach was reasonable as an approach focusing on relative differences in biomass production at different landscape positions.

Soil redistribution by tillage modelling. Soil redistribution by tillage was modelled using a diffusion-type equation developed by Govers et al.[47] as implemented in the spatially distributed model SPEROS-C. Tillage erosion is modelled using equations (1) and (2) in a spatial context.

$$Q_{\mathrm{til}} = -k_{\mathrm{til}} \times s = -k_{\mathrm{til}} \times \frac{\partial h}{\partial x} \tag{1}$$

where Q_{til} is the net flux due to tillage, k_{til} is the tillage transport coefficient (kg m^{-1} yr^{-1}), s is the tangent of the local slope gradient (-), h is the height at a given point of the hillslope (m) and x is the distance in the horizontal direction (m). The local tillage-induced erosion or deposition rate E_{til} (kg m^{-2} yr^{-1}) is been calculated as

$$E_{\mathrm{til}} = -\frac{\partial Q_{\mathrm{til}}}{\partial x} = k_{\mathrm{til}} \times \frac{\partial^2 h}{\partial x^2} \tag{2}$$

The tillage transport coefficient k_{til} depends on the tillage implement, tillage speed, tillage depths, bulk density, texture and soil moisture at the time of tillage[2].

The model calculates sediment redistribution within each field with a raster resolution of 5 m × 5 m, which results in data for roughly 5 million raster cells within the Quillow catchment.

While this tillage erosion approach generally leads to reasonable results over several decades as validated against tracer data[15], the modelling over several centuries (necessary to address long-term soil loss and gain) in our test site is associated with large uncertainties. These are mostly associated with missing or only weak parameter values, especially for land use and land management over such a long time span. Moreover, the change in topography over time is difficult to address. Although it is documented that arable land use started roughly 1,000 yr BP[43] in the test region, we do not have detailed land-use change or field layout data, or detailed data concerning land management (crop rotations, tillage intensity, and so on) for that long period. Consequently, the uncertainties associated with these unknown details were addressed by taking a simplified approach to parameterization. Topography and field layout were not reconstructed. To avoid artificial results due to a stable digital elevation model, all modelled tillage erosion results were smoothed using a moving average in 3 × 3 raster kernels. Potential erosion patterns associated with the field borders of smaller fields were ignored, as the soil loss and gain patterns along former field boarders would be partly erased due to the use of heavy machinery from the 1960s onwards.

To robustly estimate the cumulative tillage erosion, which can be expressed as a cumulative k_{til} (see yearly k_{til} in equations (1) and (2)), since cultivation started, we used two independent approaches.

(1) Based on Kappler et al.[43], we simply assume that tillage in the entire area used for agriculture today started 1,000 yr ago. To account for differences in tillage intensity during this long period the k_{til} value for different time periods was estimated based on (i) earlier erosion tracer ($^{239+240}$Pu) measurements[15] representing roughly the time from the 1960s to 2015, which were performed within the test site (see super test site in Supplementary Fig. 3); or (ii) literature values for different kinds of tillage techniques ranging from horse-drawn to mechanized systems[2] and the assumption that before 1850 no mechanized tillage occurred. This resulted in a cumulative k_{til} value of 186,125 kg m^{-1} (Supplementary Table 1). Assuming that SPEROS-C in general is able to reproduce tillage erosion patters (see ref. [15]) in the region, this should result in a reasonable soil truncation and colluviation, which, however is associated with relative large uncertainties due to the rough estimates of the model input parameters.

(2) To achieve more confidence in these results we used a second independent approach using the current remote-sensing-based spatial distribution of biomass production of winter wheat in the test area, in combination with modelled soil loss/deposition-affected biomass. Based on Öttl et al.[16], the raster cell-specific, EVI-derived, mean biomass of winter wheat for 2010 and 2015 (approximately 5 × 10^6 raster cells) was determined, while using an EVI–biomass relation presented by Jin et al.[48]. The spatially distributed mean biomass was converted into spatially distributed soil loss and soil gain using the results of the winter wheat modelling for these specific years with AQUACROP, assuming different soil truncation and colluviation (Fig. 4). Under the assumption that tillage erosion is one of the dominant processes in the variability in soil properties in the region[15,19], the resulting soil loss and gain map was used to test and calibrate the outputs of SPEROS-C. This was done for eight different erosion and deposition classes by increasing the cumulative k_{til} in a stepwise fashion so that the number of raster cells per soil erosion class in SPEROS-C aligned with the class derived from the biomass/soil erosion map. The calibration was done for each of the soil erosion classes separately so that a mean optimal cumulative k_{til} and its standard deviation could be calculated. Keeping the simplified time periods with different k_{til} values as used in the first approach (Supplementary Table 1) with a constant k_{til} before 1850 (150 kg m^{-1} yr^{-1}), the mean time since tillage was introduced could be calculated to additionally test plausibility. Based on the calibration, it could be estimated that tillage started in the region ~1,073 ± 299 yr BP (based on modelling from 2015 backwards). This compares well with the results of the first approach and our original assumption of 1,000 yr BP as the start date for tillage in the region and is in line with the geoarchaeological findings of Kappler et al.[43], indicating that substantial agriculturally induced soil erosion in the region did not occur before the beginning of the last millennium.

Given the challenges of modelling soil redistribution by tillage over 1,000 yr, the similar results of both approaches described give confidence in the robustness of the cumulative tillage erosion modelling. However, to account for at least some uncertainty in the cumulative tillage erosion, all modelled erosion results are always based on the mean calibrated cumulative k_{til} and its standard deviation (Supplementary Table 3). This results in a range of soil truncation and colluvial accumulation as given in Supplementary Fig. 2.

Combined biomass and tillage erosion modelling. Results from tillage erosion and biomass modelling were combined in a GIS. Therefore, the soil loss or gain from the tillage erosion model was classified into 130 thinning and 130 thickening soil profiles as used to determine crop growth based on the modified standard soil profiles with AQUACROP. This resulted in a spatially distributed profile information in a 5 m × 5 m raster for the entire test region. For each of the 260 soil profiles the biomass was modelled for winter wheat and maize with AQUACROP for the different years. The results of the biomass modelling were then spatially distributed according to the profiles modelled with SPEROS-C.

This coupling allowed the determination of the effects of tillage erosion on biomass production at a landscape scale, while considering the effect of different seasonal weather conditions and crops. To determine the net effect on biomass production on all fields the results of the combined models were compared with the results from an AQUACROP modelling on the undisturbed standard soil profiles.

This combined modelling also allowed the assessment of future effects of soil loss and gain on biomass production for different tillage scenarios for the next 50 and 100 yr. These tillage scenarios use k_{til} values from other studies[2] to address future reduced tillage (k_{til} = 250 kg m^{-1} yr^{-1}), intensified tillage (k_{til} = 1,000 kg m^{-1} yr^{-1}) or a business-as-usual approach with a k_{til} of 500 kg m^{-1} yr^{-1}.

It is important to note that our scenarios are based on 'wet' and 'normal-to-dry' years for the relatively dry region. Hence, the scenarios are somewhat conservative regarding the effect of soil redistribution by tillage on yields because the effects we simulated would be amplified in the case of potentially reduced or more variable rainfall during the growing season.

Data availability

Source data are provided with this paper.

Code availability

Aquacrop for GIS is available as an executable file from https://www.fao.org/aquacrop/software/aquacrop-gis/en/. SPEROS-C is available on request from P.F.

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Author contributions

J.N.Q. and P.F. contributed equally to the design, literature review, modelling and manuscript preparation. L.K.Ö. supported the modelling and evaluated the remote sensing data for the test site.

Competing interests

The authors declare no competing interests

Additional information

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