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Simulating the effect of tillage practices with the global ecosystem model LPJmL (version 5.0-tillage)

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Abstract. The effects of tillage on soil properties (e.g. soil carbon and nitrogen), crop productivity, and global greenhouse gas emissions have been discussed in the last decades. Global ecosystem models are limited in simulating tillage. Hence, they do not allow for analyzing the effects of tillage and cannot evaluate, for example, reduced-tillage or no-till as mitigation practices for climate change. In this paper, we describe the implementation of tillage related practices in the global ecosystem model LPJmL. The model is subsequently evaluated against reported differences between tillage and no-till management on several soil properties. To this end, simulation results are compared with published meta-analysis on tillage effects. In general, the model is able to reproduce observed tillage effects on global, as well as regional patterns of carbon and water fluxes. However, modelled N-fluxes deviate from the literature and need further study. The addition of the tillage module to LPJmL 5.0 opens opportunities to assess the impact of agricultural soil management practices under different scenarios with implications for agricultural productivity, carbon sequestration, greenhouse gas emissions and other environmental indicators.

1 Introduction

Agricultural fields are tilled for various purposes, including seedbed preparation, incorporation of residues and fertilizers, water management and weed control. Tillage affects a variety of biophysical processes that affect the environment, such as greenhouse gas emissions or soil carbon sequestration and can promote various forms of soil degradation (e.g. wind-, water- and tillage-erosion), leaching and runoff. Reduced-tillage or no-till is being promoted as a strategy to mitigate greenhouse gas (GHG) emissions in the agricultural sector (Six et al., 2004; Smith et al., 2008). There is an ongoing long-lasting debate about tillage and no-till effects on soil organic carbon (SOC) and GHG emissions (Schlüter et al., 2018). In general, reduced- or no-till tends to increase SOC storage through a reduced decomposition and thereby reducing GHG emissions (Chen et al., 2009; Willekens et al., 2014). However, several field studies have shown contradictory results (Grandy et al., 2006; van Kessel et al., 2013; Lugato et al., 2018; Powlson et al., 2014; Zhao et al., 2016). This is not surprising as tillage affects a complex set of biophysical factors. The effect of reduced-tillage or no-till impacts on SOC storage and GHG emissions varies depending on climate and soil conditions that influence plant and soil processes driving decomposition (Díaz-Zorita et al., 2002; Ogle et al., 2005).

In order to study the role of tillage for biogeochemical cycles, crop performance and mitigation practices, the effects of tillage on soil physical properties need to be represented in ecosystem models. Though tillage is

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already implemented in other ecosystem models in different levels of complexity (Lutz et al., under review; Maharjan et al., 2018), tillage practices in global ecosystem models are currently underrepresented.

This paper describes new routines as implemented into the Lund Potsdam Jena managed Land (LPJmL5) (von Bloh et al., 2018) that allows for studying different tillage practices. This enables us to quantify the effects of different tillage practices on biogeochemical cycles, crop performance and assessing questions related to agricultural mitigation practices.

2 Tillage effects on soil processes

Tillage affects different soil properties and soil processes, which result in a complex system with various feedbacks on soil water, temperature and carbon (C) and nitrogen (N) related processes (Fig. 1). Some processes are not taken into account in this initial implementation (e.g. soil compaction and water erosion) to limit model complexity, despite acknowledging that these processes can be important.

The effect of tillage has to be implemented and analyzed in conjunction with residue management as these management practices are often inter-related. The degree to which properties and processes are affected mainly depends on the tillage intensity. We here describe few selected processes (identified by numbered elements in Fig. 1), without distinguishing tillage intensities, even though these can be parametrized in LPJmL.

The presence of a residue layer on top of the soil column tends to increase water infiltration [1] by intercepting part of the rainfall, limiting soil crusting and reducing runoff (Ranaivoson et al., 2017). Moreover, it tends to lower soil evaporation [2] and to reduce the amplitude of soil temperature [3] (Enrique et al., 1999; Steinbach and Alvarez, 2006). Incorporating residues into the soil increases the soil organic matter (SOM) content of the tilled soil layer [4], while the bulk density of the tilled soil layer is decreased [5] (Green et al., 2003). An increase in SOM will positively affect the porosity and therefore the soil water holding capacity (WHC) [6] (Minasny and McBratney, 2018). The result of a decrease in bulk density affects the WHC through the porosity [7]. A change in WHC affects several water related processes. For instance, an increase in WHC reduces lateral runoff and leaching [8], whereas infiltration can be enhanced as the soil can store more soil moisture [9], which is beneficial for plant access to water [10]. The soil temperature is strongly related to soil moisture [11], through the heat capacity of the soil, i.e. a relatively wet soil heats up much slower than a relatively dry soil (Hillel, 2004). Changes in soil moisture and soil temperature influence several processes, including the rate of SOM mineralization [12]. The rate of mineralization affects the amount of CO_2 emitted from soils [13] and the inorganic N content of the soil. Inorganic N can then be taken up by plants [14], be lost as N_2O [15], or transformed in other forms of N (not shown). After the soil has been tilled, due to gravitational forces and precipitation, the soil over time consolidates, which means it slowly returns to its original density level before it was tilled.

[Fig. 1]

3 Implementation of tillage routines into LPJmL

3.1 LPJmL model description

The tillage implementation described in this paper was introduced into the dynamical global vegetation, hydrology and crop growth model LPJmL (version 5), which was recently extended by a terrestrial N cycle to

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also account for nutrient limitations (von Bloh et al., 2018). Previous comprehensive model descriptions and developments can be found in Schaphoff et al. (2018). The LPJmL model simulates the C, N and water cycles and explicit biophysical processes in plants (e.g. photosynthesis) and soil (e.g. mineralization of N and C). The water cycle explicitly considers evaporation, transpiration, soil infiltration and runoff. Soils in LPJmL are represented by five hydrologically active layers, each with a distinct layer thickness. The first soil layer, which is mostly affected by tillage, is 0.2 m deep. The following soil layers are 0.3, 0.5, 1.0 and 1.0 m thick, respectively, followed by a 10.0 m bedrock layer.

In LPJmL5, all organic matter pools are represented as C and N pools with variable C:N ratios (Appendix – Fig. 5a). The fraction of residues, which are harvested, can range between almost fully harvested (90%, Bondeau et al. 2007) or none, when all residues are left on the field. The C and N content in the residues that are not harvested (>10%) are transferred to the above-ground litter pool ($Litter_{ag}$). The C and N content in crop roots are transferred to the below-ground litter pool ($Litter_{bg}$). The litter pools are then subject to decomposition, after which the humified products are transferred to one of the SOM pools. The SOM pools consist of a fast pool with a turnover time of 30 years, and a slow pool with a 1000 year turnover time (Schaphoff et al., 2018). Carbon, water and N pools in vegetation and soils are updated daily as the result of computed processes (photosynthesis, autotrophic respiration, growth, transpiration, evaporation, infiltration, percolation, mineralization, nitrification, leaching and many more; for a full description see Bloh et al. (2018)). LPJmL5 has been evaluated extensively and demonstrated that the model performs credibly for reproducing C,- water and N fluxes in both agricultural and natural vegetation on various scales (Bloh et al., 2018; Schaphoff et al., 2018b).

3.2 Litter pools

In order to take care of residue management resulting to tillage, we have introduced an incorporated litter pool ($Litter_{inc}$) and a surface litter pool ($Litter_{surf}$). Crop residues not collected from the field are transferred to $Litter_{surf}$. A fraction of residues from $Litter_{surf}$ is then partially or fully transferred to the incorporated litter pool ($Litter_{inc}$), depending on the tillage practice;

$$Litter_{inc} = Litter_{inc} + Litter_{surf} \cdot TL, \tag{1}$$

and the $Litter_{surf}$ pool is reduced accordingly;

$$Litter_{surf} = Litter_{surf} \cdot (1 - TL), \tag{2}$$

where $litter_{inc}$ is the amount of incorporated surface litter C and N (treated separately but accounting for actual C:N ratios of the pools) in g m^{-2} after tillage. The parameter TL is the tillage efficiency, which determines the fraction of residues which are incorporated by tillage (0-1). To account for the vertical displacement of litter through bioturbation under natural vegetation and under no-till conditions, we assume that 0.2% of the $Litter_{surf}$ is transferred to $Litter_{inc}$ per day (equivalent to an annual bioturbation rate of 50%).

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$Litter_{inc}$ and $Litter_{surf}$ are subject to decomposition. The decomposition of $Litter_{inc}$ depends on soil moisture and temperature of the first soil layer, similar to $Litter_{ag}$ as described in Schaphoff et al. (2018). The decomposition of $Litter_{surf}$ is described below.

3.2.2 Decomposition

The decomposition of litter depends on the temperature and moisture of its surroundings. For the litter pools within the soil column ($Litter_{bg}$ and $Litter_{inc}$) decomposition depends on soil moisture and soil temperature of the upper soil layer, whereas the decomposition of the $Litter_{surf}$ depends on its own temperature and moisture, which are approximated by the model (Eq. (5), (12)). As the litter decomposes, a fixed fraction of the C is mineralized, i.e., emitted as CO_2 (70%), whereas the remaining humified C is transferred to the soil C pools following the usual litter and soil decomposition rules as described by von Bloh et al. (2018) and Schaphoff et al. (2018). The mineralized N (also 70%) of the decomposed litter is added to the ammonium pool of the first soil layer, where it is subjected to further transformation (von Bloh et al., 2018), whereas the humified organic N is allocated to the different organic soil N pools in the same shares as the humified C. The decomposition of litter $decom$ (in g C m^{-2} day^{-1}) is described by first-order kinetics (Eq. 3), following Sitch et al. (2003);

$$decom = Litter \cdot (1 - e^{-(k \cdot response)}), \tag{3}$$

where k is a decomposition rate in day^{-1} (specific for each "plant functional type") and $response$ the litter response function, which depends on the litter temperature (T_{litter} in °C) and litter moisture (S in mm);

$$response = T_{litter} \cdot (0.04021601 - 5.00505434 \cdot (S^3) + 4.26937932 \cdot (S^2) + 0.71890122 \cdot S). \tag{4}$$

T_{litter} is calculated as an average of soil temperature and air temperature. S depends on the water holding capacity of the litter layer (WHC_{litter}), the fraction of residue cover ($Cover_{surf}$) and the amount of water captured by the litter layer ($Infil_{surf}$).

3.3 Water fluxes

3.3.1 Litter infiltration

Precipitation and applied irrigation water in LPJmL5 is partitioned into interception, transpiration, soil evaporation, soil moisture and runoff (Jägermeyr et al., 2015). To account for the interception and evaporation of water by the surface cover, the water can now also be captured by $Litter_{surf}$ by infiltration ($Infil_{surf}$) and be lost through litter evaporation. Surplus water that cannot infiltrate into the $Litter_{surf}$ layer, i.e. more than WHC_{litter}, infiltrates into the first soil layer. Litter moisture (S) is calculated in the following way:

$$S_{(t+1)} = \min(WHC_{litter} - S_{(t)}, Infil_{surf} \cdot Cover_{surf}). \tag{5}$$

$Cover_{surf}$ is calculated by adapting the equation from Gregory (1982) that relates the amount of residues (dry matter) per m^2 to the fraction of soil covered by crop residue;

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$$Cover_{surf} = 1 - exp^{-A_m \cdot Litter_{surfOM}}, \quad (6)$$

where $Litter_{surfOM}$ is the total mass of dry matter residues in g m^{-2} and A_m is the area covered per mass of crop specific residue (m^2 g^{-1}). The total mass of residues is calculated in the following way:

$$Litter_{surfOM} = Litter_{surfC} \cdot CF_{SOM}, \quad (7)$$

where $Litter_{surfOM}$ is the total mass of residues in g SOM m^{-2}, $Litter_{surfC}$ is the amount of C stored in $Litter_{surf}$ in g C m^{-2}. To get the total amount of SOM in $Litter_{surfOM}$, we apply a factor of 2 (CF_{SOM}), based on the assumption that organic matter is 50% C, as in Pribyl (2010). We apply the average value of 0.004 for A_m from Gregory (1982) to all materials, neglecting variations in surface cover for different materials. WHC_{litter} (mm) is calculated by multiplying the WHC of a kg of litter (set to $2 \cdot 10^{-3}$ mm kg^{-1} SOM) with the litter mass ($Litter_{surfOM}$) following Enrique et al. (1999).

3.3.2 Litter and soil evaporation

Evaporation ($Evap_{Litter}$, in mm) from $Cover_{surf}$, is calculated in a similar manner as evaporation from the first soil layer where evaporation is a function of potential evapotranspiration (PET), evaporation available water (ω_{litter}) relative to WHC_{litter}, vegetated cover ($Cover_{veg}$) and radiation energy (Schaphoff et al., 2018). Here, also $Cover_{surf}$ is taken into account so that the fraction of soil uncovered is subject to soil evaporation as described in Schaphoff et al. (2018);

$$\omega_{litter} = S/WHC_{litter}, \quad (8)$$

$$Evap_{Litter} = PET \cdot \alpha \cdot \max(1 - Cover_{veg}, 0.05) \cdot \omega_{Litter}^2 \cdot Cover_{surf}, \quad (9)$$

where PET is calculated based on the theory of equilibrium evapotranspiration (Jarvis and McNaughton, 1986) and α the empirically derived Priestley-Taylor coefficient ($\alpha = 1.32$) (Priestley and Taylor, 1972).
The presence of $Cover_{surf}$ reduces the evaporation of a soil layer ($Evap_{Soil}$). $Evap_{Soil}$ (mm) occurs when there is not a full $Cover_{surf}$ ($Cover_{surf} < 1$). $Evap_{Soil}$ corresponds to the soil evaporation as described in Schaphoff et al. (2018), where $Evap_{Soil}$ depends on the available energy for vaporization of water and the available water in the upper 0.3 m of the soil (ω_{evap}). However, the fraction of $Cover_{surf}$ influences evaporation, i.e, a larger fraction of $Cover_{surf}$ results in a decrease in $Evap_{Soil}$. ω is calculated as the evaporation-available water relative to the water holding capacity in that layer (WHC_{evap});

$$\omega = \min\left(1, \frac{\omega_{evap}}{WHC_{evap}}\right), \quad (10)$$

where ω_{evap} is all the water above wilting point of the upper layer (0.2 m) and one third of the second layer (0.3 m) (Schaphoff et al., 2018);

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$$Evap_{Soil} = PET \cdot \alpha \cdot \max(1 - cover_{veg}, 0.05) \cdot \omega^2 \cdot (1 - Cover_{Surf}). \tag{11}$$

3.4 Heat flux

The temperature of the litter layer is calculated as the average of soil temperature of the previous day of the first layer ($T_{Soil,l=1}$ in°C) and air temperature (T_{air} in°C), in the following way:

$$T_{litter} = 0.5(T_{air} + T_{l=1}). \tag{12}$$

Equation (12) is an approximate solution for the heat exchange described in Schaphoff et al. (2013). In contrast to Schaphoff et al. (2013), the upper boundary condition (T_{upper} in °C) is no longer equal to T_{air}, but is now calculated by the $Cover_{surf}$ weighted average of T_{air} and T_{litter}:

$$T_{upper} = T_{air} \cdot (1 - Cover_{surf}) + T_{litter} \cdot Cover_{surf}. \tag{13}$$

3.5 Tillage effects on physical properties

3.5.1 Hydraulic properties

Previous versions of the LPJmL model are using static soil hydraulic parameters as inputs, which were calculated using the pedotransfer function (PTF) by Cosby et al. (1984). We now introduced a new approach using the PTF by Saxton and Rawls (2006), which was included in the model in order to dynamically simulate permanent wilting point (PWP), field capacity (FC), saturation (SAT) and saturated hydraulic conductivity (Ks). Owing to the effects of changes in SOM on hydraulic characteristics and on soil productivity, we included a PTF which also takes organic matter content of the soil into account. Though several methods exist to calculate feedbacks of SOM (Pachepsky and van Genuchten, 2011; Wösten et al., 1999) on hydraulic properties, we chose Saxton and Rawls (2006) since -to our knowledge- it was the only PTF where SOM feedbacks on those specific parameters were included. Other PTFs include texture only (Cosby et al., 1984; Rawls et al., 1982; Saxton et al., 1986) or calculate SOM effects on soil water parameters at continuous pressure levels (Van Genuchten, 1980; Vereecken et al., 2010).

Dynamic soil water properties are now calculated on a daily time step via the PTF. The model considers twelve soil textural classes for productive soils, all with a specific percentage of silt, sand (Sa in %v) and clay (Cl in %v) and a 13th class for unproductive land, which is referred to as "rock and ice". The textural classes were derived following the approach by Cosby et al. (1984), who used the midpoint values of each textural class from the USDA textural soil triangle to determine the average percentage of the soil separates sand, silt and clay. These percentages are then used in the PTF to calculate specific soil hydraulic properties for each textural class. PTF following Saxton and Rawls (2006):

$$PWP = 1.14 \cdot \lambda_{pwp} - 0.02, \tag{14}$$

$$FC = 1.238 \cdot (\lambda_{fc})^2 - 0.626 \cdot \lambda_{fc} - 0.015, \tag{15}$$

$$SAT = FC + 1.636 \cdot \lambda_{sat} - 0.097 \cdot Sa - 0.064, \tag{16}$$

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$$\lambda_{pwp,fc,sat} = \alpha \cdot Sa + \beta \cdot C + \gamma \cdot SOM + \delta \cdot Sa \cdot SOM + \varepsilon \cdot Cl \cdot SOM + \rho \cdot Sa \cdot Cl + \sigma, \quad (17)$$

$$BD = (1 - SAT) \cdot MD. \quad (18)$$

[Table 1]

SOM is the soil organic matter content in weight percent (%w), BD is the bulk density in kg m^{-3}, MD is the mineral density of 2700 kg m^{-3}. SOM is calculated using the slow and fast C pool as well as soil bulk density. This way, we ensure a feedback of organic material on soil water properties. SOM is calculated as following:

$$SOM = \frac{CF_{SOM} \cdot (SC_{fast} + SC_{slow})}{BD \cdot z} \cdot 100, \quad (19)$$

where SC_{fast} is the fast decaying C pool in kg m^{-2}, SC_{slow} is the slow decaying C pool in kg m^{-2}, BD is the bulk density in kg m^{-3} and z is the thickness of the specific soil layer in m. It was suggested by Saxton and Rawls (2006) that the PTF should not be used for high SOM values, so we only consider SOM of up to 5% when computing soil hydraulic properties. We treated soils with SOM content above this threshold as soils with 5% SOM content. Saturated hydraulic conductivity is also calculated using the PTF from Saxton and Rawls (2006) in the following way:

$$Ks = 1930 \cdot (SAT - FC)^{3-\phi}, \quad (20)$$

$$\phi = \frac{\ln(FC) - \ln(PWP)}{\ln(1500) - \ln(33)}, \quad (21)$$

where Ks is the saturated hydraulic conductivity in mm h^{-1} and ϕ is the slope of the logarithmic tension-moisture curve.

3.5.2 Bulk density

Effects of tillage for the tillage layer (first topsoil layer of 0.2 m) are accounted for by adapting BD after tillage, which is then used to calculate a new SAT and FC. Ks is also newly calculated using SAT_{till} and FC_{till} in equation (23) and (24). A mixing efficiency (mE) depending on the intensity and type of tillage, which can be specified as a parameter and ranges between 0 and 1, determines the BD after tillage, following the APEX model approach (Williams et al., 2015). An mE of 0.90 represents a full inversion tillage practice, also known as conventional tillage (White et al., 2010). Using mE in combination with residue management after harvest, we are now able to simulate different tillage types and intensities, depending on the combination of settings. The BD change after tillage is following Williams et al. (2015):

$$f_{BDtill} = 1 - (1 - 0.667) \cdot mE. \quad (22)$$

Tillage density effects on saturation and field capacity follow Saxton and Rawls (2006):

$$SAT_{till} = 1 - (1 - SAT_0) \cdot f_{BDtill}, \quad (23)$$

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$$FC_{till} = FC_0 - 0.2 \cdot (SAT_0 - SAT_{till}), \quad (24)$$

where f_{BDtill} is the density effect on the top soil layer after tillage, SAT_{till} and FC_{till} are adjusted saturation and field capacity after tillage and SAT_0 is the saturation before tillage.

3.5.3 Reconsolidation of tillage effect

Depending on the structural composition of the soil and the amount of precipitation after the tillage event, with time the tilled soil layer reconsolidates to its state before tillage, also known as soil settling. This way the porosity and BD changes caused by tillage gradually decline, caused by a cycle of wetting and drying (Onstad et al., 1984). The reconsolidation of the soil is now accounted for using the approach by Williams et al. (2015) (Eqs. 25 to 27):

$$sz = 0.2 \cdot Infil_{soil} \cdot \frac{1+2 \cdot Sa/(Sa+e^{8.597-0.075 \cdot Sa})}{z_{till}^{0.06}}, \quad (25)$$

$$f = \frac{sz}{sz+e^{3.92-0.0226 \cdot sz}}, \quad (26)$$

$$f_{BDtill(t+1)} = f \cdot (1 - f_{BDtill}), \quad (27)$$

where sz is the scaling factor for the tillage layer, $Infil_{soil}$ is the infiltration rate into the layer in mm d^{-1} and z_{till} is the depth of the tilled layer in m. This allows for a faster settling of recently tilled soils with high precipitation and for soils with a high sand content. In contrast soils with a low sand content settle slower, especially in dry areas with low precipitation.

4 Model setup

4.1 Model input, initialization and spin-up

In order to bring vegetation patterns and SOM pools into a dynamic equilibrium stage, we make use of a 5000 years spin-up simulation, which recycles the first 30 years of climate input following the procedures of von Bloh et al. (2018). For simulations with land use inputs and to account for agricultural management, a second spin-up of 390 years is conducted, to account for historical land use change. The spatial resolution of all input data and model simulations is 0.5°. Land use data is based on crop-specific shares of MIRCA2000 (Portmann et al., 2010) and cropland and grassland time series since 1700 from HYDE3 (Klein Goldewijk et al., 2010) as described by Fader et al. (2010). We drive the model with daily mean temperature from the Climate Research Unit (CRU TS version 3.23, University of East Anglia Climate Research Unit, 2015; Harris et al., 2014), monthly precipitation data from the Global Precipitation Climatology Centre (GPCC Full Data Reanalysis version 7.0; Becker et al., 2013), (shortwave downward and net longwave downward) radiation data from the ERA-Interim data set (Dee et al., 2011). Static soil texture classes are taken from the Harmonized World Soil Database (Nachtergaele et al., 2009) and soil pH data from the WISE data set (Batjes, 2005). The NOAA/ESRL Mauna Loa station (Tans and Keeling, 2015) provides atmospheric CO_2 concentrations. Deposition of N was taken from the ACCMIP database (Lamarque et al., 2013).

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4.2 Simulation options and evaluation set-up

The new tillage management implementation allows for specifying tillage systems. We conducted contrasting simulations with or without application of tillage. The effect of tillage on current cropland was evaluated. The default setting for conventional tillage is: mE=0.9 and TL=0.95. In the tillage scenario, tillage is conducted twice a year, at sowing and after harvest. Soil water properties are updated on a daily basis, enabling the tillage effect to be effective from the subsequent day onwards until it wears off. Four different management settings (MS) for global simulations were used: 1) tillage performed and residue are left on the field (T_R), 2) tillage performed and residues are removed (T_NR), 3) no-till and residues retained on the field (NT_R), and 4) no-till and residues are removed (NT_NR) (Table 2). All of these 4 simulations were run from the year 1900 until 2009. Land use was introduced in 1700 and with a spin-up simulation of 390 years for T_R after the spin-up simulation with 5000 years with natural vegetation only. We used fertilizer data supplied by the Global Gridded Crop Model Intercomparison (GGCMI phase 1; Elliott et al., 2015). Fertilizers are applied at sowing and when the amount of fertilizer is larger than 5 g N m^{-2}, 50% is applied at sowing and 50% at a later stage in the growing season (depending on the phenological stage of the crop). From 1900 onwards the four new management options were introduced on current cropland. The outputs of these four different simulations were analyzed using the relative differences between each output variable using T_R as the default management;

$$RD = \frac{MS}{T\_R} - 1, \tag{28}$$

where RD is the relative difference between the management scenarios. The effects were analyzed using different time scales: the average after the first three years for short-term effects, the average after 9 to 11 years for mid-term effects and the average of year 19 to 21 for long-term effects. Depending on available reference data in the literature, the specific duration of the experiment was chosen. The results of the simulations are compared to literature values from selected meta-analyses. Meta-analyses were chosen in order to compare the globally modeled results to a set of combined results of individual studies from all around the world, rather than choosing individual site-specific studies. Results of individual site-specific experiments can differ substantially between sites, which hampers the interpretation at larger scales. We calculated the median and the 5th and 95th percentile (values within brackets) between MS in order to compare the model results to the meta-analyses, where averages and 95% confidence intervals (CI) are mostly reported. We chose medians rather than averages to reduce outlier effects. If region-specific values were reported in the meta-analyses, e.g. climate zones, we compared model results of these individual regions to the reported regional value ranges.

To analyze the effectiveness of individual processes (see Fig. 1) without too many blurring feedback processes, we conducted additional simulations of the four different MS on bare soil with uniform dry matter litter input of 75 g m^{-2}, 150 g m^{-2} and 300 g m^{-2} of uniform composition (C:N ratio of 20), no atmospheric N deposition and static fertilizer input (Elliott et al. 2015). This helps to isolate soil processes, as any feedbacks via vegetation performance is eliminated in this setting.

[Table 2]

5 Evaluation and discussion

5.1 Tillage effects on hydraulic properties

The calculated soil hydraulic properties of tillage for each of the soil classes prior to and after tillage is performed combined with 0% and 5% SOM in the tillage layer and a *mE* of 0.9 (table 3). In general, both tillage and a higher SOM content have an increasing effect on WHC, SAT, FC and Ks. Clay soils are an exception, since higher SOM content decreases their WHC, SAT and FC, and increases Ks. For the soil classes sand and loamy sand, the increasing effect on WHC, SAT and FC of increasing SOM content shows be the highest among all classes, while Ks decrease with increasing SOM content. The increasing effects of tillage on the hydraulic properties are generally weaker compared to an increase in SOM by 5% (maximum SOM content for computing soil hydraulic properties in the model). While tillage in sandy soils with a *mE* of 0.9 can increase WHC by 7%, an increase in 5% of SOM can increase WHC by 27%.

The PTF by Saxton and Rawls (2006) uses an empirical relationship between SOM, soil texture and hydraulic properties derived from the USDA soil database, implying that the PTF is likely to be more accurate within the US than outside. Nevertheless the PTF is used in a variety of global applications despite the limitations to validate it at that scale (Van Looy et al., 2017).

[Table 3]

5.2. Soil C stocks and fluxes

Model outputs for CO_2 emissions from cropland soils, as well as SOM and litter C stocks of the topsoil (0.3 m) were used to evaluate the effects of tillage and residues management on soil C stocks and fluxes. CO_2 emissions and SOM response after ten years duration of NT_R MS compared to T_R show a discrepancy, as both CO_2 emissions and SOM stocks increase (Fig. 2A and 2B). The reported numbers refer to the median value across all cropland grid cells globally. After a duration of ten years of applied MS, CO_2 emissions from NT_R compared to T_R are increased by +2.3% (5th, 95th percentile: -9.6%, +29.0%) (Fig. 2A), while at the same time topsoil and litter C is also increased by +5.7% (5th, 95th percentile: +1.7%, +14%) (Fig. 2B), i.e. the soil C stock has already increased enough to sustain higher CO_2 emissions. If we only look at the first three years after the change in MS, CO_2 emissions are substantially decreased by -12.2% (5th, 95th percentile: -18.3%, -2.8%) in a NT_R system compared to T_R (Fig. 2D). If we only analyze the tillage effect and do not take residues into account, topsoil and litter C decreases by -9.9% (5th, 95th percentile: -27.0%, -0.6%) in a T_NR system compared to a NT_NR system after ten years (Appendix – Fig. 4A), while CO_2 emissions are increased by +17.1% (5th, 95th percentile: 0.0%, +114.4%) (Appendix – Fig. 4B).

Abdalla et al. (2016) reviewed the effect of tillage, no-till and residues management and they found that if residues are returned, tillage has a decreasing effect on topsoil SOM content by 5.0% (95th CI: -1.0%, +9.2%) and an increasing effect on CO_2 emissions +23% (95th CI: -35.0%, -13.8%) (Table 4). These findings of Abdalla et al. are in contradiction to our findings for CO_2 emissions after a ten year period, nevertheless if we only take the first three years duration of MS into account, CO_2 emissions are decreased as suggested by the literature.

This supports the findings from Abdalla et al. (2016) and highlights the importance of accounting for the duration of the experiment after which the different MS are compared. Abdalla et al. (2016) also reported a decrease in SOM (-12%) and an increase in CO_2 emissions (+18%) of a T_NR system compared to a NT_NR system. T_NR was reported to decrease SOM content, while at the same time CO_2 emissions are increased, due to a higher soil temperature in a tilled soil and an increased decomposition. The updated LPJmL reproduced these patterns.

A strong CO_2 response can be found in areas where SOM increases the most (e.g., northern Mexico and western Australia). This is also true for yields, here shown for maize yields after ten years of NT_R MS (Fig. 2C), which are mostly increasing in areas with strong SOM increase (e.g., Argentina, mid-west USA, northeaster China and south-western Russia). These areas all have a warm temperate dry climate according to the IPCC climate zone classification (Carré et al., 2010). This positive feedback could be driven by a positive water-savings effect from NT_R, where water which is saved due to NT_R leads to a higher productivity. NT_R for example reduces evaporation substantially compared to T_R and has other positive water-saving feedbacks, which are further discussed in chapter 5.3. In areas with higher productivity, we also have a higher residues input, since litter fall is a function of plant productivity (see Eq. (6)). If productivity feedbacks are disabled, using the simulation from a bare soil experiment, there is no difference in CO_2 emissions between NT_R and T_R (Appendix – Fig. 6).

Our simulations of NT_R and T_R show that NT_R has a positive effect on SOM (topsoil and litter) and this effect increases over time. Our model is generally reliable to reproduce SOM increase under NT_R for a duration of ten years and increasing CO_2 emissions under T_R for a duration of three years. Differences to literature estimates occur after ten years under NT_R with regard to CO_2 emissions because productivity feedbacks under NT_R are taken into account in our model.

Ogle et al. (2005) conducted a meta-analysis and reported SOM changes from NT_R for different climatic zones. They found a +23%, +17%, + 16% and +10% mean increase in SOM after converting from a conventional tillage to a no-till system for more than 20 years for tropical moist, tropical dry, temperate moist and temperate dry climates, respectively. Ogle et al. (2005) analyzed the data based on linear mixed-effect models, which do not account for interactions between effects. This could explain why we were not able to reproduce these high numbers in SOM increase, since our model results range between a 5.1% to 11.9% increase in SOM after 20 years from tropical moist to temperate dry climates, respectively. LPJmL was also not able to reproduce the gradient found by Ogle et al. (2005). There is high discrepancy in the literature in regard to no-till effects on SOM, since the high increase found by Ogle et al. (2005) is not supported by the findings of Abdalla et al. (2016). Ranaivoson et al. (2017) found that crop residues left on the field increases SOM, which is in agreement with our simulation results.

[Fig. 2]

5.3 Water fluxes

Water fluxes are highly affected by tillage and residue management (Fig. 1). Residues, which are left on the soil surface, create a barrier that reduces evaporation from the soil. In addition, a residue cover effectively protects the soil surface from structural degradation through the impact of rain drops, thereby increasing rainfall infiltration. Generally, residues, which are incorporated through tillage, loose the function to protect the soil.

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Both, the reduction of soil evaporation and the increase of rainfall infiltration contribute to increased soil moisture and hence plant water availability. Because we could not find suitable approaches to account for the processes leading to increased rainfall infiltration, our implementation only captures the reduction of soil evaporation. However, despite the significant increase in rainfall infiltration and corresponding reduction in surface runoff found in a number of field studies (Ranaivoson et al., 2017), the contribution to plant water availability is likely to be much smaller as a substantial portion of it will be lost through subsurface runoff (lateral runoff and seepage). In cases where the reduction of soil evaporation alone is larger than the increased plant transpiration, the resulting increase in soil moisture may even lead to an overall increase in total runoff (sum of all surface and subsurface runoff components) (Fig. 3A).

Steiner (1989) conducted field and laboratory trials and reported functions for wheat and sorghum to estimate changes in evaporation based on the residue amount. These functions were used to evaluate the evaporative reduction from a layer of residues using the bare soil simulations. We find that an application of 75 g C m^{-2} yr^{-1} of residues reduces evaporation by -18.2% (5th, 95th percentile: -34.0%, -2.1%) (Appendix – Fig. 6B), 150 g C m^{-2} yr^{-1} by -40.3% (5th, 95th percentile: -55.6%, -9.0%) (Appendix – Fig. 6C) and 300g C m^{-2} yr^{-1} by -62.2% (5th, 95th percentile: -73.4%, -34.4%) (Appendix – Fig. 6D). Using the functions provided by Steiner (1989), residue amounts can be translated into a reduction of evaporation by -36.3% for wheat and -16.5% for sorghum for the low application rates, by -50.2% for wheat and -30.7% for sorghum for the medium application rates and by -64.0% for wheat and by -44.9% for sorghum for the high application rates, respectively (Table 4). These values for evaporation reduction from prescribed residue loads are well reproduced by the model. Overall, soil evaporation in the first 3 years of MS duration in the NT_R scenario is reduced by -28.4% (5th, 95th percentile: -49.0%, -11.3%) compared to the T_R (Fig. 3B).

[Fig. 3]

5.4 N_2O fluxes

Overall, switching from tillage to no-till management with additional residue input (NT_R vs. T_R) increases N_2O emissions by +7.5% (5th, 95th percentile: -6.7%, +68.9%) (Appendix – Fig. 7A). The strongest increase is found in the warm temperate zone where the average increase is 11.3% (5th, 95th percentile: +0.7%, +75.7%) (Appendix – Fig. 7B). The lowest increase is found in the tropical zone +2.9% (5th, 95th percentile: -8.5%, +43.3%) (Appendix – Fig. 7C).

The increase in N_2O emissions after switching to no-till is in agreement with several literature studies (Linn and Doran, 1984; Mei et al., 2018; van Kessel et al., 2013; Zhao et al., 2016) (Table 4). Mei et al. (2018) reports an overall increase of +17.3% (95th CI: +4.6%, +31.1%), which is higher than our values, but both ranges mostly overlap. However, although the overall effect is in agreement with Mei et al. (2018), the spatial patterns over the different climatic regimes are in less agreement. We strongly underestimate the increase in N_2O emissions in the tropical zone compared to Mei et al. (2018), who reported an increase of +74.1% (95th CI: +34.8%, +119.9%). Moreover, the N_2O emissions in arid regions after switching to no-till are underestimated (Appendix – Fig. 8B), but still within the range, compared to van Kessel et al. (2013), who reported an increase of +35.0% (95th CI: +7.5%,+69%). In the cold temperate (Appendix – Fig. 7D) and humid zones (Appendix – Fig. 8A) we slightly overestimate on average, and the 95th percentile of our ranges is relatively high compared to Mei et al. (2018) (average: -1.7% and 95th CI: -10.5%, +8.4%) and van Kessel et al. (2013) (average: -1.5% and 95th CI: -11.6%,

+11.1%). This is also the case for the warm temperate zone, though the median and average increase is in agreement with Mei et al. (2018), who report an increase of +17% (95th CI: +6.5%, +29.9%) (Table 4).

The increase in N_2O emissions under NT_R can be explained by two mechanisms. Firstly, under no-till with residues, more water can infiltrate into the soil and less water is lost through evaporation. This can cause anaerobic conditions, which trigger N_2O emissions from denitrification. Secondly, no-till tends to increase bulk density and moisture content, which results additionally in a larger water-filled pore space (Fig. 1) which can increase the denitrification rate, and therefore N_2O emissions (van Kessel et al., 2013; Linn and Doran, 1984).

However, the impact of no-till on N_2O emissions has been variable with both increases and decreases in emissions reported (van Kessel et al., 2013). This variation in response is not surprising, as tillage affects several biophysical factors that influence N_2O emissions (Fig. 1) in possibly contrasting manners (van Kessel et al., 2013; Snyder et al., 2009). For instance, no-till can lower soil temperature, which can reduce N_2O emissions (Six et al., 2004). Moreover, under T_R, more C (from residues) is incorporated into the soil, which leads to more substrate for N_2O emissions. Reduced N_2O emissions under no-till compared to the tillage MS can also be observed in the model results, for instance in North-East India, South-East Asia and areas in Brazil (Appendix – Fig. 7A).

Various studies where field experiments are conducted report high uncertainties associated with the estimation of N_2O emissions, due to significant spatial and temporal variability, which hampers the evaluation of the model results (Chatskikh et al., 2008; Mangalassery et al., 2015). Moreover, the relevant processes behind N_2O emissions are still not fully understood (Lugato et al., 2018).

The deviations from the model results compared to the meta-analyses especially for specific climatic regimes (i.e. tropical- and cool temperate) cannot be explained other than N_2O emissions are sensitive to subtle changes in soil moisture, forms of reactive N and timing, which renders all comparisons to patchy data difficult. Additional model evaluation is needed by e.g., conducting sensitivity analysis of specific inputs (e.g., soil type-, N-fertilizer) in different climate regimes for testing the model behavior.

[Table 4]

5.5 General discussion

The implementation of tillage into the global ecosystem model LPJmL opens opportunities to assess the effects of tillage and no-till practices on agricultural productivity and its environmental impacts, such as nutrient cycles, water consumption, GHG emissions and C sequestration. The implementation involved 1) the introduction of a surface litter pool, 2) dynamic accounting for SOM in computing hydraulic properties, and 3) tillage effects on physical properties.

In general, a global model implementation on tillage practices is difficult to evaluate, as effects are reported often to be quite variable, depending on soil conditions. We find that the model results for NT_R compared to T_R are in agreement with literature for C stocks and fluxes, water fluxes and to a lesser extent N_2O emissions when compared to reported impact ranges in meta-analyses. Effects can also change over time so that a comparison needs to also consider the timing, history and duration of management changes. For C, e.g., we see that NT_R has a positive effect on SOM and reduces CO_2 emissions the first years after adapting to NT_R, but increases CO_2 emissions in the mid- and long-term owing to a larger accumulation of SOM.

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In this study, model results were evaluated with data ranges as compiled by meta-analyses, which implies several limitations. Due to the limited amount of available meta-analyses, not all fluxes and stocks could be evaluated within the different management scenarios. Especially for testing residue-only effects, it would have been good to have additional studies to analyze the effects of $Cover_{Surf}$, which has a strong influence on water-fluxes (e.g., evaporation) and thus affects various other relevant fluxes that are sensitive to soil moisture as well. Also, the sample size was sometimes low, which may result in biases if not all conditions (e.g., climate and soil combinations) were tested, and it remains unclear how these can be best compared to a full sampling of the global cropland as in the modeling results. Nevertheless, the meta-analyses gave the best overview of the overall effects of tillage practices that have been reported for various individual experiments.

When applying the model, it is important to be aware that not all processes related to tillage and no-till are taken into account. For instance, NT_R can improve soil structure (e.g., aggregates) due to increased faunal activity (Martins et al., 2009), which can result in a decrease in BD. Although tillage has several advantages for famers (e.g. residue incorporation and topsoil loosening), it can have several disadvantages as well. For instance, tillage can result in compaction of the subsoil, which result in an increase in BD (Podder et al., 2012). Moreover, the absence of a residue layer can drive soil crusting which affects the infiltration of soil water. However, Strudley et al. (2008) observed mixed effects of tillage and no-till on hydraulic properties (such as BD). Nevertheless, they motivate more fruitful investigations into agricultural management practices and their interacting influences on soil hydraulic properties.

One of the primary reasons for tillage, weed control, is not accounted for in LPJmL or most other ecosystem models. As such, different tillage and residue management strategies can only be assessed with respect to their biogeochemical effects, but only partly with respect to their effects on productivity and not with respect to some environmental effects (e.g. pesticide use).

6 Conclusion

We described the implementation of tillage related practices in the global ecosystem model LPJmL 5.0-tillage. The extended model was tested under different management scenarios and evaluated by comparing to reported impact ranges from meta-analyses on C, water and N dynamics as well as on crop yields.

We were able to broadly reproduce reported tillage effects on global stocks and fluxes, as well as regional patterns of these changes, with LPJmL 5.0-tillage but deviations in N-fluxes need to be further examined. Not all effects of tillage, including one of its primary reasons, weed control, could be accounted for in this implementation. Nonetheless, the implementation of more detailed tillage-related mechanics into LPJmL improves our ability to represent different agricultural systems and to understand management options for climate change adaptation, agricultural mitigation of GHG emissions and sustainable intensification.

Code and data availability. The source code and data is available upon request from the main author for the review process and for selected collaborative projects. The source code will be generally available after final publication of this paper and a DOI for access will be provided.

Author contributions. F.L and T.H. both share the lead authorship for this manuscript. They had an equal input in designing and conducting the model implementation, model runs, analysis and writing of the manuscript. S.R.

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contributed to simulation analysis and manuscript preparation/evaluation. J.H. contributed to the code implementation, evaluation and analysis and edited the paper. S.S. contributed to the code implementation and evaluation and edited the paper. W.v.B. contributed to the code implementation and evaluation and edited the paper. J.S. contributed to the study design and edited the paper. C.M. contributed to the study design, supervised implementation, simulations and analyses and edited the paper.

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Table 1: Corresponding coefficients for the function λ_{pwp}, λ_{fc} and λ_{sat}.

	α	β	γ	δ	ε	ρ	σ
λ_{pwp}	-0.024	0.0487	0.006	0.005	-0.013	0.068	0.031
λ_{fc}	-0.251	0.195	0.011	0.006	-0.027	0.452	0.299
λ_{sat}	0.278	0.034	0.022	-0.018	-0.027	-0.584	0.078

Table 2: LPJmL simulation settings for the evaluation.

Scenario	*Simulation abbreviation*	*Retained residue fraction on field*	*Tillage efficiency (TL_{Frac})*	*Mixing efficiency of tillage (mE)*
Tillage + residues	T-R	1.0	0.95	0.90
Tillage + no residues	T-NR	0.1	0.95	0.90
No tillage + residues	NT-R	1.0	0	0
No tillage + no residues	NT-NR	0.1	0	0

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Table 3: Percentage values for each soil textural class of silt, sand and clay content used in LPJmL and correspondent hydraulic parameters before and after tillage with 0% and 5% SOM using the Saxton and Rawls (2006) pedotransfer function.

				pre tillage								**after tillage**							
				0% SOM				**5% SOM**				**0% SOM**				**5% SOM**			
Soil class	Silt (%)	Sand (%)	Clay (%)	WHC	SAT	FC	Ks	WHC	SAT	FC	Ks	WHC	SAT	FC	Ks	WHC	SAT	FC	Ks
Sand	5	92	3	0.30	0.68	0.31	244.11	0.38	0.85	0.45	202.26	0.32	0.78	0.33	365.15	0.39	0.90	0.46	249.47
Loamy sand	12	82	6	0.31	0.66	0.34	124.47	0.39	0.83	0.47	142.43	0.33	0.76	0.36	219.26	0.40	0.88	0.48	188.55
Sandy loam	32	58	10	0.31	0.59	0.37	43.16	0.38	0.77	0.48	80.17	0.34	0.72	0.40	110.34	0.39	0.84	0.49	125.69
Loam	39	43	18	0.31	0.57	0.42	13.54	0.36	0.73	0.50	38.35	0.34	0.70	0.45	53.18	0.38	0.81	0.52	74.40
Silty loam	70	17	13	0.29	0.48	0.37	6.12	0.33	0.68	0.45	42.91	0.32	0.64	0.40	45.16	0.35	0.78	0.47	90.51
Sandy clay loam	15	58	27	0.35	0.65	0.52	9.00	0.40	0.76	0.59	16.31	0.37	0.76	0.54	33.07	0.41	0.83	0.60	36.05
Clay loam	34	32	34	0.31	0.61	0.51	2.76	0.33	0.70	0.55	10.38	0.33	0.72	0.54	19.51	0.35	0.79	0.57	30.84
Silty clay loam	56	10	34	0.25	0.54	0.46	2.05	0.26	0.66	0.48	15.13	0.28	0.68	0.48	20.84	0.28	0.76	0.50	44.88
Sandy clay	6	52	42	0.39	0.70	0.64	1.00	0.40	0.74	0.67	1.51	0.40	0.79	0.65	8.02	0.41	0.82	0.68	8.03
Silty clay	47	6	47	0.44	0.75	0.72	0.19	0.44	0.76	0.74	0.06	0.46	0.83	0.73	2.95	0.46	0.83	0.75	1.81
Clay	20	22	58	0.29	0.68	0.63	0.45	0.26	0.67	0.60	1.07	0.31	0.78	0.65	5.79	0.28	0.77	0.62	9.08
Rock	0	99	1	0.00	0.01	0.01	0.10	0.00	0.01	0.01	0.10	0.00	0.01	0.01	0.10	0.00	0.01	0.01	0.10

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Table 4: Comparison of simulated model output and literature values from meta-analysis.

Variable/Scenario	Soil depth (m)	# paired treatments	Literature mean in % (95% confidence interval)	Time horizon (years)	Model (median in %)	Model (5% and 95% percentile)	Reference
notill residue - till residue							
SOM (0.3m)	0-0.3	101	+5.0 (-1.0, 9.2)*‡	**	+5.7	+1.7, +14.0	Abdalla et al., 2016
CO2		113	-23.0 (-35.0, -13.8)*	**	+2.3	-9.6, +29.0	Abdalla et al., 2016
N2O		100	+36.1 (+25.0, +47.8)*	**	+7.5	-6.7, +68.9	Mei et al., 2018
N2O (tropical)		123	+74.1 (+34.8, +119.9)†‡	**	+2.9	-8.5, +43.3	Mei et al., 2018
N2O (warm temperate)		62	+17.0 (+6.5, +29.9)†‡	**	+11.3	+0.7, +75.7	Mei et al., 2018
N2O (cool temperate)		27	-1.7 (-10.5,+8.4)†‡	**	+8.8	-3.1, +170.5	Mei et al., 2018
N2O (arid)		56	+35 (+7.5, +69.0)*	**	+8.2	-6.8, +69.9	Kessel et al., 2013
N2O (humid)		183	-1.5 (-11.6, +11.1)*	**	+5.9	-6.5, +65.6	Kessel et al., 2013
Yield (wheat) B		47	-2.6 (-8.2, +3.8)*	10§	+4.3	-9.4, +58.7	Pittelkow at al. 2015b
Yield (maize) B		64	-7.6 (-10.1, -4.3)*	10§	+3.4	-23.1, +62.9	Pittelkow at al. 2015b
Yield (pulses) B		12	-2.4 (-9.0, +4.9)*	10§	+10.2	0.0, +215.7	Pittelkow at al. 2015b
Yield (rapeseed) B		10	+0.7 (-2.8, +4.1)*	10§	+2.8	-27.3, +50.6	Pittelkow at al. 2015b
notill residue - notill noresidue							
Evaporation		3	-16.5[B] , -36.3[BB]	**	-18.2	-34.0, -2.1	Steiner 1989
Evaporation		3	-30.7[D], -50.2[DD]	**	-40.3	-55.6, -9.0	Steiner 1989
Evaporation		3	-44.9[E], -64.0[EE]	**	-62.2	-73.4, -34.4	Steiner 1989
till nores-no till-nores							
SOM (0.3m)	0-0.3	46	-12.0 (-15.3, -5.1)*	**	-15.1	-41.2, -0.4	Abdalla et al.,

						2016
CO_2	46	+18.0 (+9.4, +27.3)*	**	+17.1	+0.0, +114.4	Abdalla et al., 2016
Yield (wheat) B	8	+2.7 (-6.3, +12.7)*	10§	-0.6	-8.4, +20.9	Pittelkow at al. 2015b
Yield (maize) B	12	-25.4 (-14.7,-34.1)*	10§	-0.5	-13.4, +5.7	Pittelkow at al. 2015b
till nores-till res						
N_2O	105	+1.3 (-5.4, +8.2)*‡	**	-8.4	-19.5, +4.0	Mei et al., 2018

*estimated from graph

**Time horizon of the study is unclear in the meta-analysis. The average over the first three years of model results is taken.

† includes conservation till

†† at least 30% on soil

‡ Residue management for conventional till unsure

§ Time horizon not explicitely mentioned by author

[B] 75g/m2 dry matter sorghum, [BB] 75g/m2 dry matter wheat

[D] 150g/m2 dry matter sorghum, [DD] 150g/m2 dry matter wheat

[E] 300g/m2 dry matter sorghum, [EE] 300g/m2 dry matter wheat

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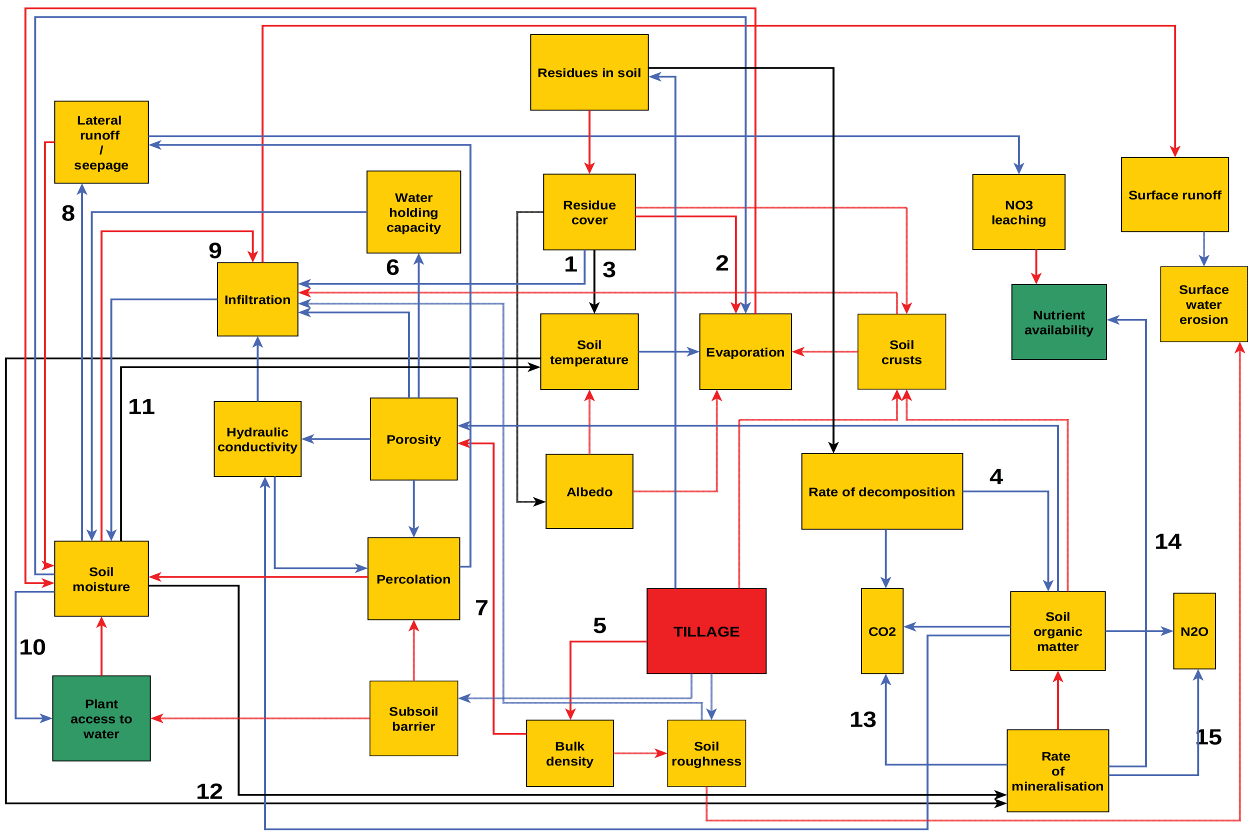

Fig. 1: Flow chart diagram of feedback processes caused by tillage, which are considered (dashed lines) and not considered (dashed lines) in LPJmL. Blue lines highlight positive feedbacks, red negative and black are ambiguous feedbacks.

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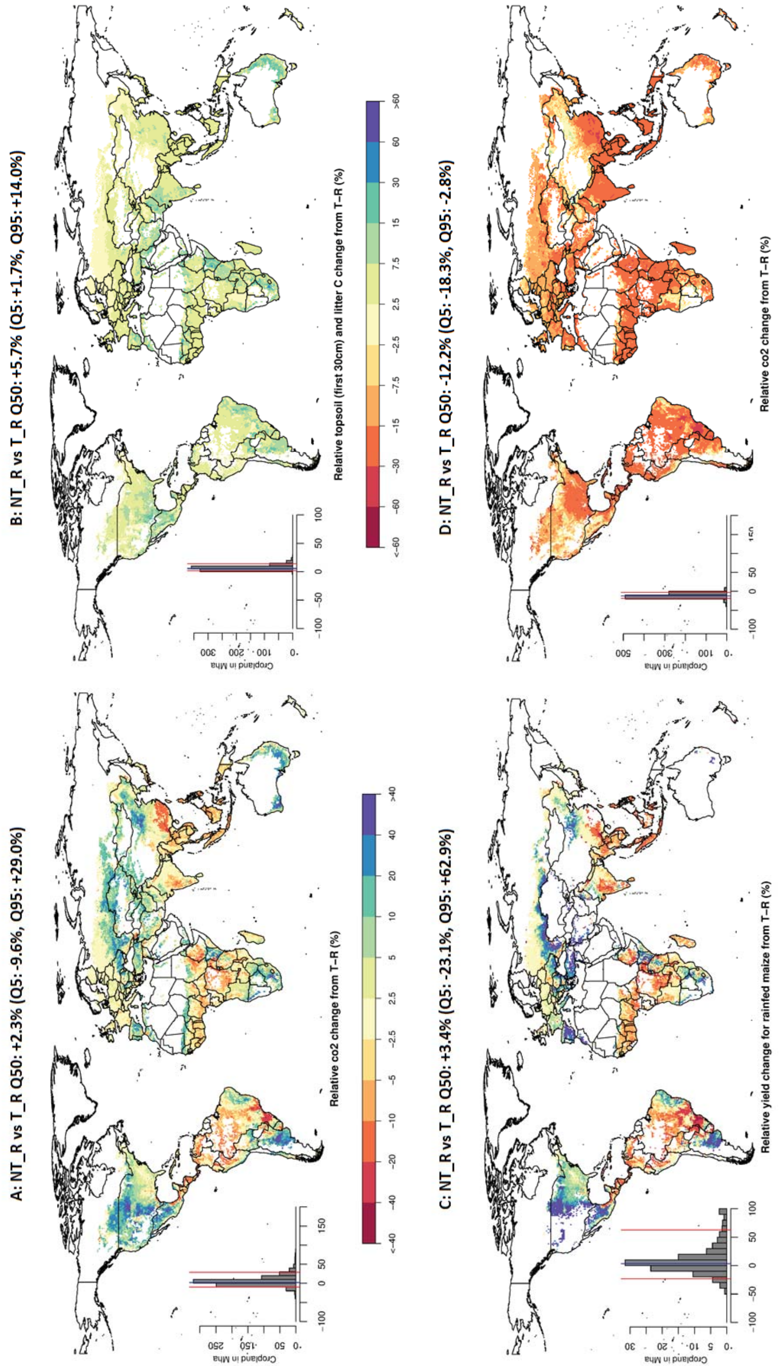

Fig. 2: Relative C dynamics comparing NT_R vs. T_R – Relative CO_2 change after ten years (A), relative topsoil and litter C change after ten years (B), relative yield change for rain-fed maize after ten years (C), relative CO_2 change after three years (D).

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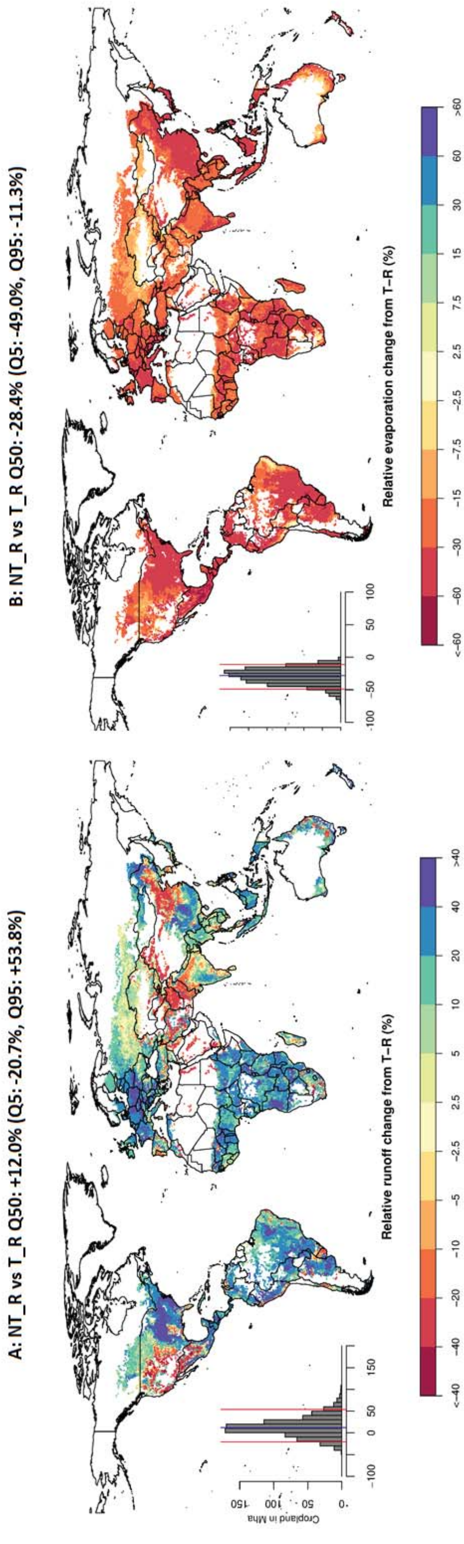

Fig. 3: Relative changes in runoff (A) and evaporation (B) comparing NT_R vs. T_R for the average of the first three years after implementation.

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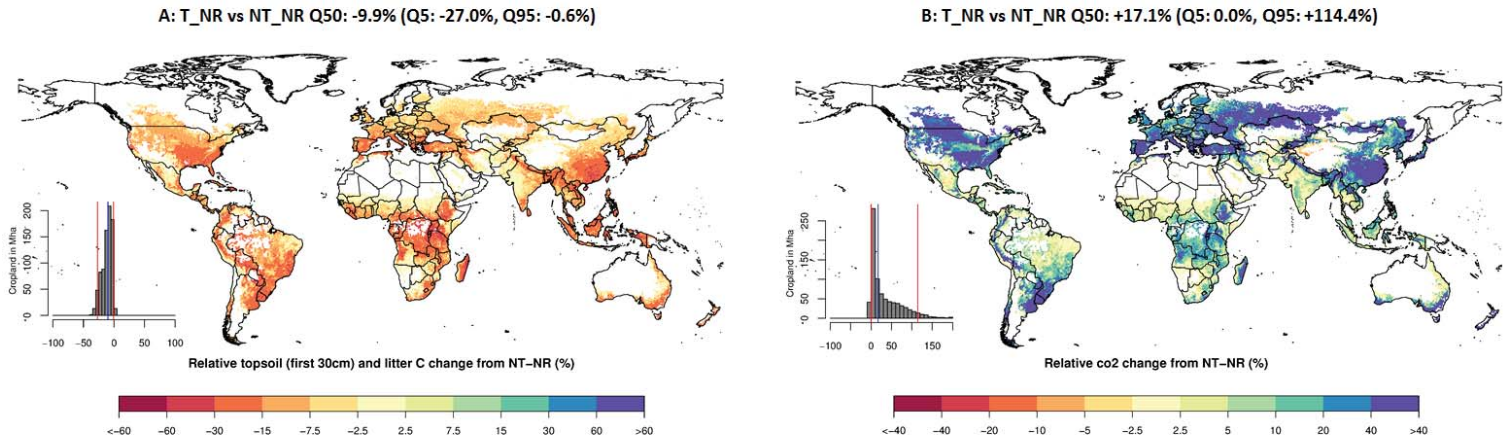

Appendix – Fig. 4: Relative topsoil and litter carbon change for T_NR vs. NT_NR after ten years of experiment duration (A), Relative CO_2 change for T_NR vs. NT_NR after ten years of experiment duration (B).

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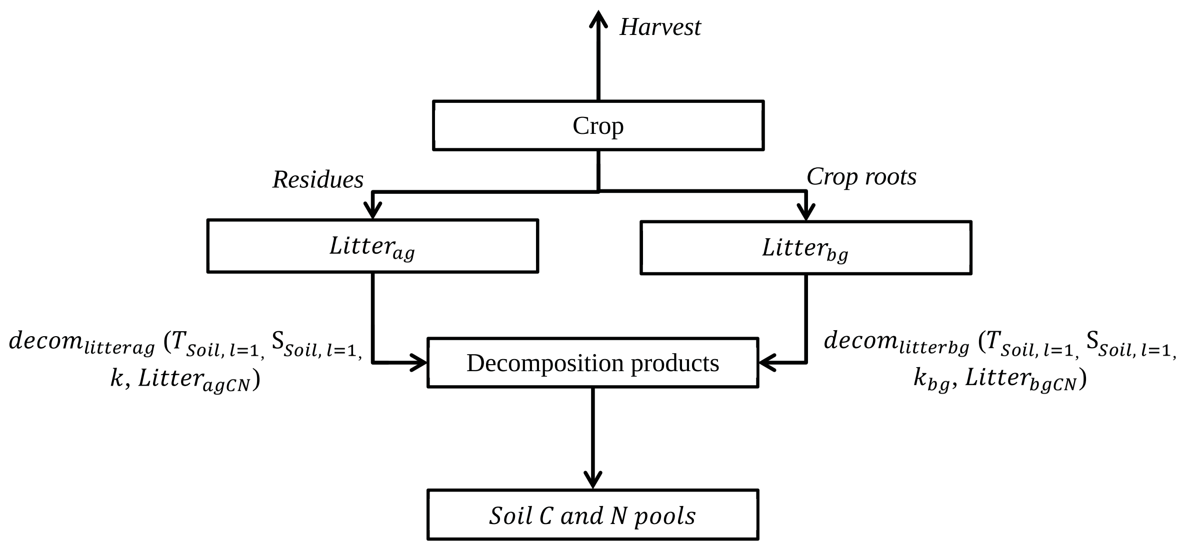
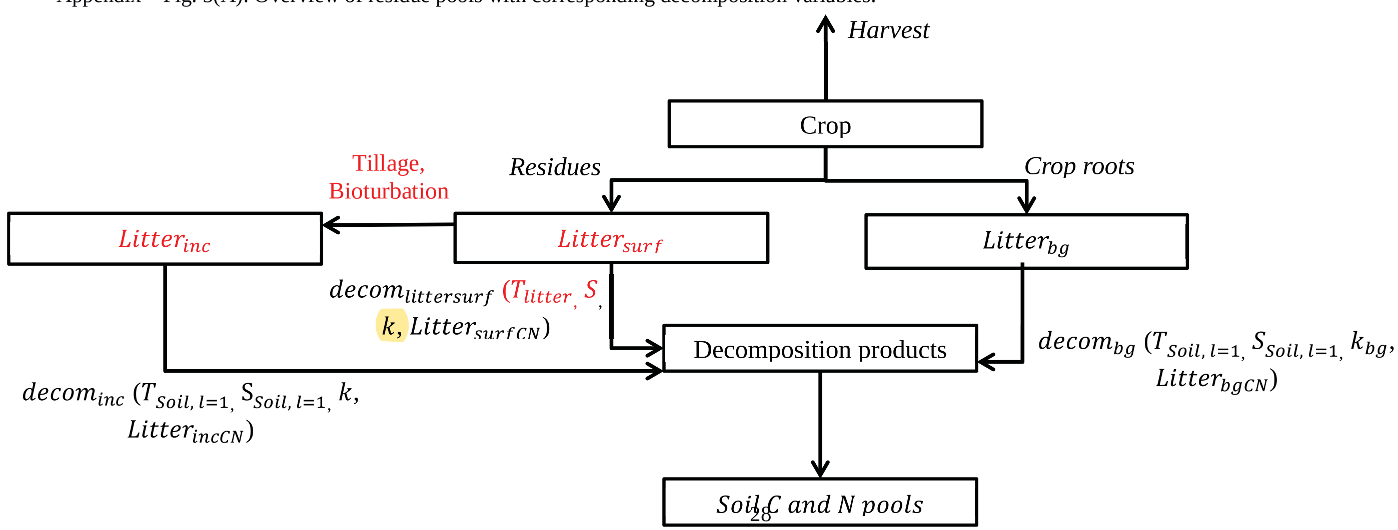


Appendix – Fig. 5(A): Overview of residue pools with corresponding decomposition variables.



Appendix – Fig. 5(B): Overview of residue pools and the new pool for incorporated residues with corresponding decomposition variables.

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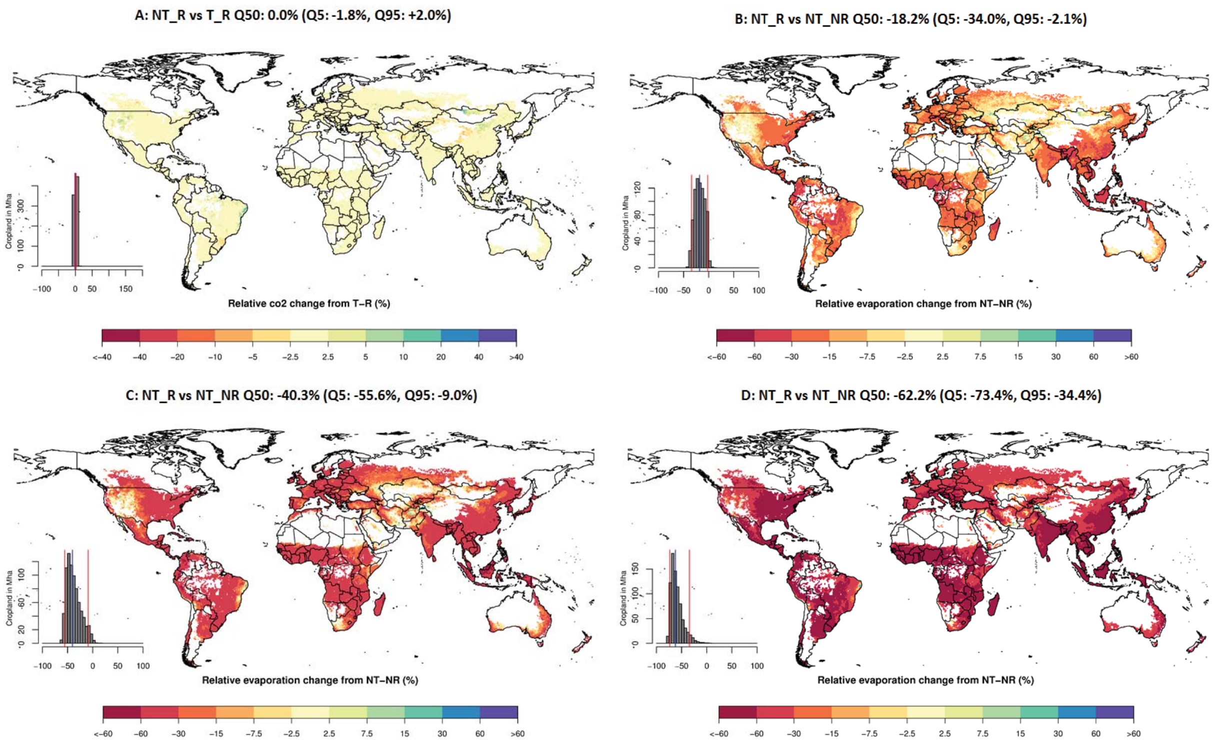

Appendix – Fig. 6: Relative CO_2 emission change for NT_R vs. T_R from bare soil experiment for the first three years with C m^{-2} yr^{-1} fixed residue amount input (A), relative soil evaporation change for NT-R vs. NT-NR from the bare soil experiment for the first three years with 75g C m^{-2} yr^{-1} fixed residue amount input (B), relative soil evaporation change for NT_R vs. NT_NR from bare soil experiment for the first three years with 150g C m^{-2} yr^{-1} fixed residue amount input (C), Relative soil evaporation change for NT_R vs. NT_NR from bare soil experiment for the first three years with 300g C m^{-2} yr^{-1}fixed residue amount input (D).

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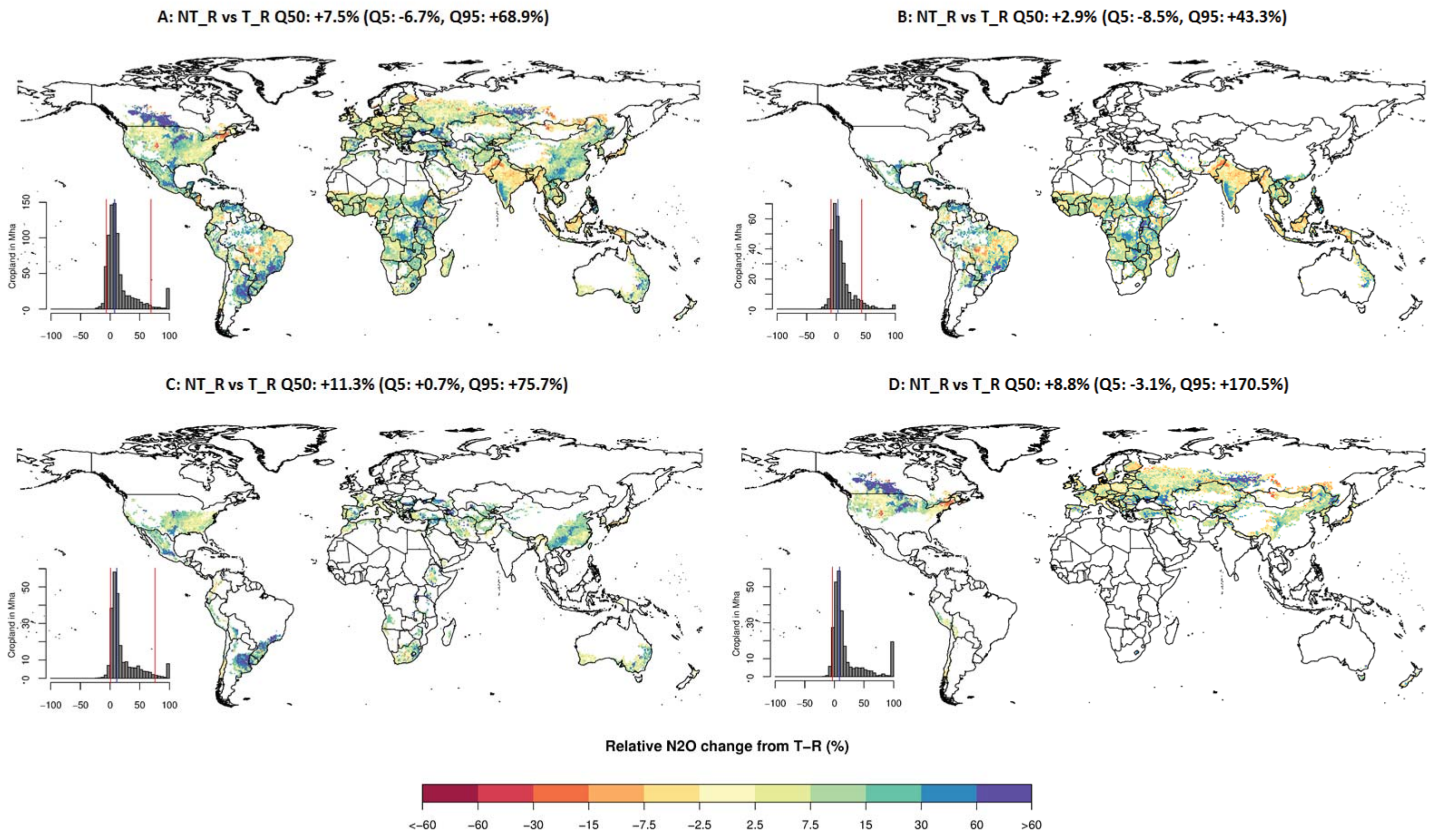

Appendix – Fig. 7: Relative changes in N_2O emissions compared to T_R (A), Relative changes in N_2O emissions compared to T_R in tropical regions (B), Relative changes in N_2O emissions compared to T_R in the warm temperate regions (C), Relative changes in N_2O emissions compared to T_R in the cold temperate regions (D).

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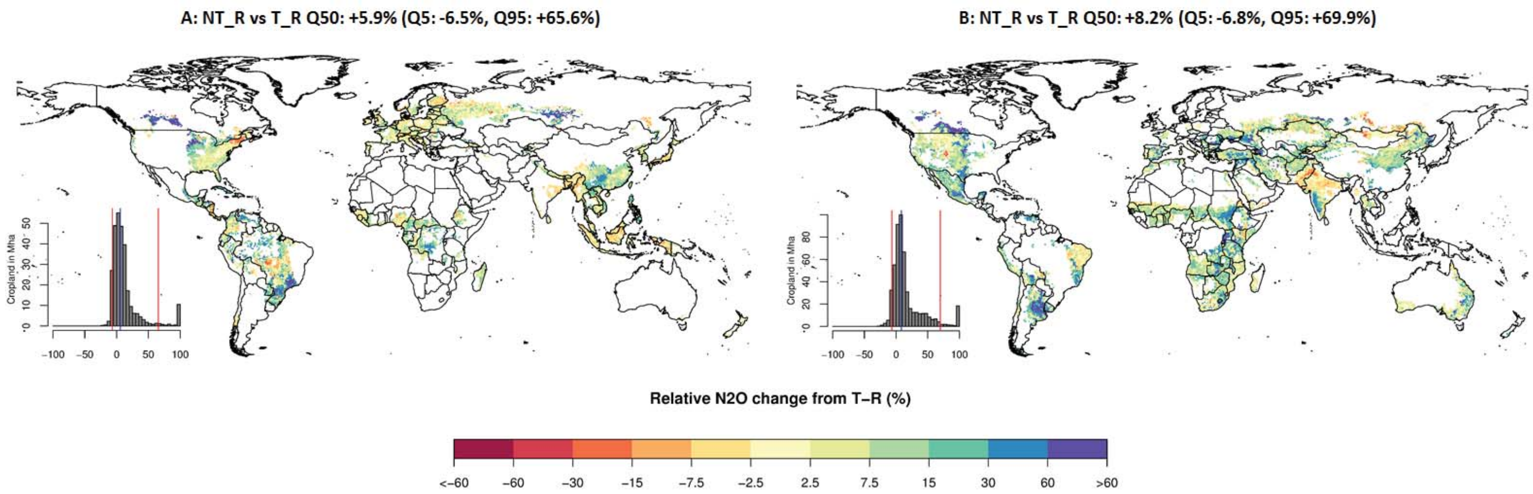

Appendix – Fig. 8: Relative changes in N_2O emissions compared to T_R in the humid regions (A), Relative changes in N_2O emissions compared to T_R in the arid regions (B).