Agricultural Systems 186 (2021) 102985

Contents lists available at ScienceDirect

Agricultural Systems

journal homepage: www.elsevier.com/locate/agsy

Crop production, water pollution, or climate change mitigation—Which drives socially optimal fertilization management most?

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ARTICLE INFO

Keywords:
Carbon sequestration
Climate change
Crop production
Soil carbon
Soil nitrogen
Water pollution

ABSTRACT

We introduce a multistep modeling approach for studying optimal management of fertilizer inputs in a situation where soil nitrogen and carbon dynamics and water and atmosphere externalities are considered. The three steps in the modeling process are: (1) generation of the data sets with a detailed simulation model; (2) estimation of the system models from the data; (3) application of the obtained dynamic economic optimization model considering inorganic and organic fertilizer inputs. We demonstrate the approach with a case study: barley production in southern Finland on coarse and clay soils. Our results show that there is a synergy between climate change mitigation and water protection goals, and a trade-off between pollution mitigation and crop production goals. If a field is a significant source of greenhouse gas (GHG) emissions and an insignificant source of water pollution, atmospheric externalities dominate the water externalities, even for a relatively low social cost of carbon (SCC). If a field is a significant source of water pollution, the SCC would have to be very high before atmospheric externalities dominate water externalities. In addition, an integrated nutrient management system appears better than a system in which only inorganic or organic fertilizer is used, although manure is not a solution to agriculture's GHG emissions problem. Moreover, GHG emissions and nitrogen and carbon leaching mitigation efforts should first be targeted at coarse soils rather than clay soils, because the marginal abatement costs are considerably lower for coarse soils.

1. Introduction

Sustained and environmentally friendly food production to feed the growing global population is one of the greatest challenges of our time (Tilman et al., 2001, 2002; Garnett et al., 2013). Reaching food security will become even more difficult due to the changing climate and the need to mitigate greenhouse gas (GHG) emissions in all sectors, including agriculture (Beddington et al., 2012; Poppy et al., 2014). Diminishing soil fertility and soil carbon (C) content are causing the declining productivity of soils (Bauer and Black, 1994; Amundson et al., 2015; Sanderman et al., 2017). Environmental security, on the other hand, is challenged by the negative externalities of crop cultivation, including nutrient and C losses to water ecosystems (Vitousek et al., 1997; Carpenter et al., 1998; Foley et al., 2005) and the C and nitrogen (N) losses to the atmosphere (Vermeulen et al., 2012; Campbell et al., 2017). This paper provides novel results regarding socially optimal fertilization management when both the water and atmospheric externalities are considered.

About 26% of global GHG emissions are caused by food production (Poore and Nemecek, 2018). Agriculture, however, also has the potential for climate change mitigation by restoring C to domesticated soils (Lal, 1999, 2001; Wang et al., 2017). Smith et al. (2008) estimated that technical mitigation potential from agriculture by 2030 is about 5500–6000 Mt. carbon dioxide equivalent (CO_2-eq) yr^{-1}, which corresponds to 15–17% of the global annual GHG emissions (Ritchie and Roser, 2017). However, there are doubts whether C sequestration in agricultural soils is politically and economically attainable (Amundson and Biardeau, 2018). Agriculture is also a huge source of water pollution: food production is responsible for about 78% of global eutrophication (Poore and Nemecek, 2018). The rates of applied fertilizers, and the ratio of the inorganic to organic fertilizer rates, are important factors in determining the annual N leaching from the field (Basso and Ritchie, 2005). When the crop productivity and the externalities are studied from the long-term perspective, considering both N and C is essential because of their interaction (Knops and Tilman, 2000; Karlsson, 2012; Liu et al., 2017). Nitrogen fertilizer and manure additions are necessary for

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https://doi.org/10.1016/j.agsy.2020.102985
Received 3 April 2020; Received in revised form 16 September 2020; Accepted 20 October 2020
Available online 2 November 2020

increasing plant available N, obtaining good yields, and returning high amounts of crop residues to the soil if leguminous crops are not an option in crop rotation (Christopher and Lal, 2007).

Crop production, C sequestration, and water and air externalities need to be considered simultaneously because decisions made to achieve a certain target affect the possibilities to reach other targets (cf. Aillery et al., 2005; Alexander et al., 2015; Sanderman et al., 2017; Amundson and Biardeau, 2018). In addition, it is important to study these aspects from a dynamic perspective, because in the long run both the damage (leaching and GHG emissions) and the benefits (crop yields) react to changes in the soil stocks.

To this end, we introduce a dynamic bioeconomic model for crop production that includes the soil N and C stocks, as well as the N and C losses to water ecosystems and to the atmosphere. A detailed simulation model is used for generating long-term field data to estimate the systems of models capturing the necessary elements for analysis. The obtained models are extended for the economic optimization of inorganic and organic fertilizer inputs in crop production. We address the following questions: (1) How do the soil N and C dynamics and the water and atmospheric externalities affect the optimal cultivation practices? (2) What kind of synergies and trade-offs are there among different agricultural goals? (3) What is the relative magnitude of the atmospheric and water damage? (4) How do the private marginal abatement costs depend on the increasing restrictions on atmospheric emissions and water pollution?

In the existing economics literature, the papers close in aim and scope to the one at hand are those focusing on optimal management of N and C in crop production in a dynamic setting. Several earlier papers have applied dynamic optimization in determining optimal N rates (Kennedy et al., 1973; Segarra et al., 1989; Jomini et al., 1991; Thomas, 2003; Lambert et al., 2007; Dhakal et al., 2019). Some papers extend the dynamic optimization to also include water body externalities (Yadav, 1997; Watkins et al., 1998; Nkonya and Featherstone, 2000; Huang et al., 2001; Martínez and Albiac, 2004; Hyytiäinen et al., 2011; Sela et al., 2017). There are also some papers focusing on the dynamic economic optimization of C sequestration in agriculture (Antle et al., 2001; Antle and Diagana, 2003; Graff-Zivin and Lipper, 2008; Nelson and Matzek, 2016; Berazneva et al., 2019). Despite the extensive literature, to the best of our knowledge there is no paper where both soil N and C are included as soil stocks in dynamic economic analysis. This gap in the literature is probably due to the lack of long-term field experiment data, where the changes in the soil N and C stocks would be documented. Furthermore, there are no economic papers including both water and atmospheric emissions in a dynamic analysis.[1]

We develop a dynamic modeling framework where annual fertilizer and manure decisions are optimized on the economic basis of profit maximization in a setting where different combinations of externalities are considered in addition to soil N and C interactions. The modeling process starts with an existing simulation model called the Coupled heat and mass transfer model for the soil-plant-atmosphere system (CoupModel), which is used to run long-term simulation experiments, where inorganic and organic N fertilizers are given in different amounts. We use the obtained data sets to estimate simpler system models, which are directly usable for dynamic optimization. The optimization is carried out for a case study: barley (*Hordeum vulgare* L.) cultivation on clay and coarse soils in southern Finland. The aim of the case study is to understand the relative importance of climate change mitigation, water protection, and high crop productivity as well as to evaluate the impacts of these factors on optimal fertilization decisions. We focus on barley production, because it is the largest grain crop in Finland by volume (Fig. 1).

[1] There are, however, papers where they are considered in a static setting, for example Gren and Ang (2019) and Lötjönen et al. (2020).

2. The model

2.1. The setup

We consider barley production on a representative field parcel of homogenous land characterized by a certain soil texture and a certain level of initial soil N and C stocks. The lower boundary for the layer of the ground entailing the stocks in the soil is at a depth of 1.5 m. The time horizon is infinite, and the time step is one year, $t \in \{0, 1, 2, \dots\}$. Here we describe the model components that are relevant for explaining the functioning of the barley production agroecosystem. All the endogenous variables (barley yield and the N and C losses to water ecosystems and to the atmosphere), as well as the state variables (soil N and C stocks), change simultaneously as a function of the exogenous variables, i.e., the annual input choices of inorganic N fertilizer and manure application (the control variables), and the weather variables. The existing literature gives prior information about the structure of the individual equations.

The primary output of the agroecosystem—the annual barley yield (kg ha^{-1} yr^{-1})—is described as an increasing function of the annual photosynthesis (kg ha^{-1} yr^{-1}), δ_t (Zelith, 1982; Wu et al., 2019), annual average temperature (°C yr^{-1}), $temp_t$, and annual precipitation sum (mm yr^{-1}), ppt_t (Schlenker and Roberts, 2009; Hakala et al., 2012; Mäkinen et al., 2018):

$$y_t = y(\delta_t; temp_t, ppt_t) \tag{1}$$

Annual photosynthesis, i.e., the primary production of the agroecosystem, is described as an increasing function of annual inorganic N fertilization (kg ha^{-1} yr^{-1}), N_t, solid manure (kg ha^{-1} yr^{-1}), φ_t, and soil N stock (kg ha^{-1} yr^{-1}), n_t (Sinclair and Hone, 1989; Wang et al., 2012; Jin et al., 2015; Song et al., 2019). Note that both inputs—inorganic N fertilizer and manure—increase the yield and as such they are imperfect substitutes for each other; both can be used to meet the plant's N requirements, but the yield response to these two inputs is not identical. Also, inorganic fertilizer and manure affect soil N and C processes and the loss processes in a different fashion.[2] Photosynthesis is also a function of temperature and precipitation (Hew et al., 1969):

$$\delta_t = \delta(N_t, \varphi_t, n_t; temp_t, ppt_t) \tag{2}$$

We also assume that the law of diminishing marginal returns is satisfied (Cannon, 1892; McNall, 1933).

The crop cultivation agroecosystem also produces several by-products. First, there are water body externalities. Annual N loss (kg ha^{-1} yr^{-1}), e_t^N, is an increasing convex function of the soil N stock, inorganic N fertilization, and solid manure. The annual N loss is also an increasing function of the total annual runoff sum (mm yr^{-1}) (drainage and surface runoff), $runoff_t$, and a decreasing function of the annual yield through the crop N uptake (the N that is annually removed from the system with the harvested yield). Also, increasing temperature may have a decreasing effect on N leaching because it increases evaporation:

$$e_t^N = e^N(N_t, \varphi_t, n_t, y_t; temp_t, runoff_t) \tag{3}$$

Annual C loss to water bodies (kg ha^{-1} yr^{-1}), e_t^c, is assumed to be an increasing convex function of manure, soil C stock, c_t, and the total annual runoff sum (mm yr^{-1}):

$$e_t^c = e^c(\varphi_t, c_t; runoff_t) \tag{4}$$

Second, there are the atmospheric externalities: GHG emissions from the field to the atmosphere. Carbon dioxide (CO_2) emissions are released from the field by soil respiration and plant respiration processes. Annual

[2] Many papers have indicated that manure additions to agricultural soils result in higher C sequestration compared with N fertilizer additions (Yang et al., 2004; Jarecki et al., 2005; Hati et al., 2006; Morari et al., 2006; Rudrappa et al., 2006; Su et al., 2006).

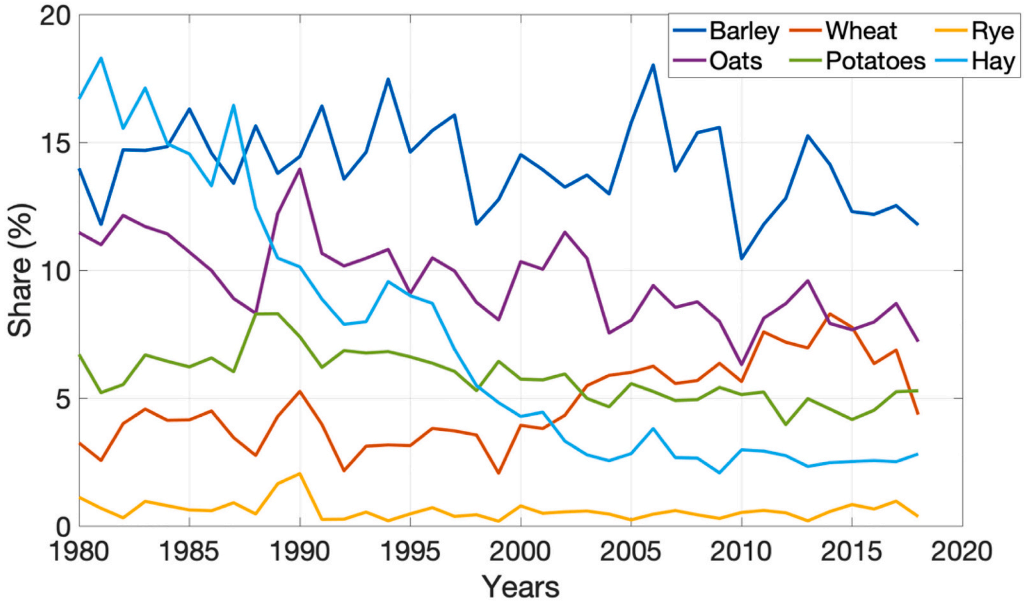

Fig. 1. Production shares of the main crops in Finland from 1980 to 2019 (Luke, Statistic database, 2019).

soil respiration (CO_2 kg ha^{-1} yr^{-1}), R_t^s, is an increasing function of the annual manure application, soil C stock, and temperature (Llouyd and Taylor, 1994; Schimel et al., 2001; Lai et al., 2012; Karhu et al., 2014; Wang et al., 2019). Also, it has been shown that there is a positive relationship between N fertilization and soil respiration (Wang et al., 2019), but N fertilization tends to flatten the otherwise declining trend of soil C stock (Ladha et al., 2011):

$$R_t^s = R^s(c_t, \varphi_t, N_t; temp_t, ppt_t) \tag{5}$$

Annual plant respiration (CO_2 kg ha^{-1} yr^{-1}), R_t^p, is an increasing function of annual yield and temperature (Reich et al., 1998; Atkin and Tjoelker, 2003; Atkin et al., 2005):

$$R_t^p = R^p(y_t; temp_t) \tag{6}$$

Agricultural soil also releases nitrous oxide (N_2O) emissions (N_2O kg ha^{-1} yr^{-1}). Annual N_2O emissions, ξ_t, are an increasing function of denitrification, denoted by η_t, inorganic N fertilizer and manure, soil N stock, and temperature and a decreasing function of the annual yield (Kirschbaum, 1995; Matson et al., 1998; Smith et al., 2003; Shcherbak et al., 2014):

$$\xi_t = \xi(\varphi_t, N_t, y_t, \eta_t; temp_t) \tag{7}$$

Denitrification, on the other hand, is an increasing function of the input decisions, soil N stock, and temperature (Stanford et al., 1975; Knowles, 1982)[3]:

$$\eta_t = \eta(N_t, \varphi_t, n_t; temp_t) \tag{8}$$

We also consider the C that is released back to the atmosphere as human (and animal) respiration when the yield is consumed, C_t^y. The main part of the C in the consumed yield is most likely used as energy. We assume that 40% of the yield is released back to the atmosphere as C. This corresponds to the amount of C in grains.[4]

Agroecosystems also fix GHG emissions from the atmosphere. Carbon sequestration in managed soils occurs when there is a net removal of atmospheric CO_2 (C inputs > C outputs). CO_2 is stored in plant biomass through photosynthesis. Hence, annual net GHG emissions (CO_2-eq kg ha^{-1} yr^{-1}), e_t^{GHG}, consist of annual soil respiration, plant respiration, the C in grains, N_2O emissions, and the C sequestered through photosynthesis, δ_t:

$$e_t^{GHG} = R_t^s + R_t^p + C_t^y + \epsilon\xi_t - \delta_t, \tag{9}$$

where ϵ is an emission factor that converts the annual N_2O emissions to CO_2-eqs. The value of ϵ is 298 (Forster et al., 2007).

Last, we have model components that describe the evolution of the soil N and C stocks. The evolution of the soil N stock from one period to another is described using the N carryover equation (Kennedy et al., 1973):

$$n_{t+1} = n_t + \vartheta^n\left(n_t, c_t, N_t, \varphi_t, e_t^N, \xi_t, y_t; temp_t, ppt_t\right), \tag{10}$$

where ϑ^n is a soil N transition function. Soil N transition is an increasing function of annual N fertilizer input and a decreasing function of the soil N stock (Rasmussen and Rohde, 1988; Knops and Tilman, 2000). The transition function is also an increasing function of soil C stock (Islam et al., 2016) and a decreasing function of the annual yield via crop N uptake. Also, the N transition function is a decreasing function of annual N_2O emissions (via denitrification) and annual N leaching, as well as an increasing function of temperature (since the soil N mineralization rate is significantly affected by the temperature) (Stanford et al., 1973; Dessureault-Rompré et al., 2010; Guntiñas et al., 2012; Liu et al., 2017). The evolution of the soil C stock from one period to another is described using a C carryover equation:

$$c_{t+1} = c_t + \vartheta^c\left(c_t, n_t, \varphi_t, e_t^c, R_t^s, \delta_t; temp_t, ppt_t\right), \tag{11}$$

where ϑ^c is a soil C transition function. Soil C transition is an increasing function of current soil N stock (a positive interaction effect) (Blevins et al., 1983; Salinas-Garcia et al., 1997) and a decreasing function of the current (or the initial) soil C stock (Knops and Tilman, 2000; Wang et al., 2017; Zhao et al., 2018). The transition of soil C stock is also an increasing function of the manure application and the annual yield via the crop residual, and a decreasing function of annual C leaching and CO_2 emissions through soil respiration (Izaurralde et al., 2000; Zhao

[3] Precipitation affecting soil moisture could also be added to Eq. (8), but as the conditions are strongly time dependent, the effect of annual precipitation is difficult to incorporate.

[4] In reality, the C that is released back to the atmosphere would most likely be lower. However, the effect of this C component on the results is minor. If the component was lower, the optimal N and manure rates would be slightly higher when GHG emissions are considered in decision-making.

et al., 2018; Ghimire et al., 2019). In our simulations, crop residues were left in the field after harvest, and they were incorporated by ploughing.[5] Soil C mineralization and hence the C transition is also affected by the temperature: soil C decomposition is an increasing function of temperature (Knorr et al., 2005). However, it is unclear whether the temperature increases or decreases the transition (Davidson and Janssens, 2006; Koven et al., 2017; Tang et al., 2017; Wang et al., 2017). Therefore, we include a positive linear term and a negative quadratic term for temperature.

2.2. The social planner's problem

The social planner maximizes the net present value (NPV) of the revenues from the crop production minus the monetary value of the environmental damage of the N and C losses to water ecosystems and to the atmosphere:

$$\max_{N_t,\varphi_t,n_{t+1},c_{t+1}} \sum_{t=0}^{\infty} \beta^t \left[p^y y_t - p^N N_t - \varphi_t \left(\tau^M + p^M\right) - \left(\mu^N e_t^N + \mu^c e_t^c + \mu^{GHG} e_t^{GHG}\right) \right] \tag{12}$$

Subject to Eqs. (10) and (11), $N_t \geq 0$ and $\varphi_t \geq 0$,
with n_0 and c_0 given.

where y_t is the annual yield, N_t is the annual N fertilizer rate, φ_t is the annual solid manure rate, e_t^N is the annual N loss, e_t^c is the annual C loss, and e_t^{GHG} is the annual GHG emissions as defined by Eq. (9). Annual revenues and costs are discounted by a discount factor $\beta = (1 + \rho)^{-1}$, where $\rho \geq 0$ is the social planner's discount rate. Constant and exogenous market prices of the yield, N fertilizer, manure, and manure spreading cost (€ kg^{-1}) are given by p^y, p^N, p^M, and τ^M, respectively. Note that we consider here a situation where all the manure is bought and not produced on the farm. Fixed production costs are ignored because they do not influence the annual input choices. Annual N and C leaching to water ecosystems is valued with the constant external marginal damage costs of the N loss (€ N kg^{-1}), μ^N, and the C loss (€ C kg^{-1}), μ^c, respectively. These measure the additional damage caused by an additional unit of the given nutrient in the water ecosystem.[6] The annual GHG emissions are valued with the constant social cost of carbon (SCC) (€ C-eq kg^{-1}), denoted by μ^{GHG}. The SCC measures the change in the discounted value of economic welfare from an additional unit of CO_2-eq (or C-eq) emissions (Tol, 2011; Nordhaus, 2017; Pindyck, 2019).[7] Hence, we assume that a damage function is linear instead of being strictly convex. We make this typical assumption due to the data limitations; we have marginal damage cost estimates but no damage function for the emissions. The assumption of linear damage is reasonable because a single farm's contribution to climate change or eutrophication (e.g., in the Baltic Sea) is marginal. We also assume that the damage function is additively separable, implying that we ignore all the possible interactions between the damage due to the lack of data.[8] Initial soil N and C stocks, n_0 and c_0, are determined by the past nutrient management decisions, and therefore they are taken as given by the landowner.

2.3. Optimality conditions for the annual input decisions

Following Nguyen et al. (2016) we solved Eq. (12) with the method of Lagrange (A1). According to the obtained optimality conditions, the NPV over the planning horizon is maximized when in each period the inorganic and organic fertilizer application rates are at the level where the marginal value product of the fertilizer is equal to the opportunity cost of the marginal unit of fertilizer. The marginal value product of the fertilizer is the increase in the yield obtained by a marginal unit of fertilizer, valued with the market price of the yield. Opportunity costs consist of the price of the fertilizer, plus the water and atmospheric externalities of the unit of fertilizer, less the discounted next-period value of the carryover of the present fertilizer applications. The carryover of the fertilizer is valued with the shadow price of the soil stock in question. The optimal level of the soil N stock is where the marginal value product of the soil N stock is equal to the opportunity cost of the marginal unit of soil N stock. The opportunity costs consist of the present shadow price of the soil N stock and the water and atmosphere externalities of the soil stock, minus the N and C carryovers. Carryovers are discounted and valued with the next-period shadow prices of the stocks in question. Analogously, the optimal level of the soil C stock is where the discounted carryovers are equal to the present shadow price of the soil C stock plus the externalities of the soil C stock (A1).

The steady-state shadow prices of the soil N and C stocks are crucial for optimal long-term management if the system converges to the steady state in the long run. The formulas for the steady-state shadow prices of the soil N and C stocks show that if the loss processes are excessive, the planner becomes more impatient and depletes the soil stock at a higher rate compared with the case where the loss processes are moderate (A2). We may also conclude that the shadow prices of the soil stocks are increasing functions of the marginal value product of the soil N stock and decreasing functions of the water externalities. The effect of the atmospheric externalities depends on whether the soil is a sink or source of the GHG emissions.

3. Modeling process

The modeling process consisted of four separate steps (Fig. 2). In the first step, we used the CoupModel (Jansson, 2012) to carry out a large number of simulations mimicking fertilizer field experiments (A3).[9] The CoupModel was parameterized for barley production in southern Finland (Rankinen et al., 2007; Salo et al., 2016). The meteorological data (for the period 1980–2011) were from the Jokioinen Observatory of the Finnish Meteorological Institute (60.81° N, 23.50° E, altitude 104 m). In the second step, we transformed the output of these simulations to panel data sets describing barley production system responses to various combinations of inorganic and organic fertilizers on coarse and clay soils with various initial soil N and C stocks.[10] In this step, daily simulation output data were converted to annual data, because the time step of the analysis is one year. In the third step, these data sets were used as an input for estimating the system of models described using Eqs. (1)–(11).

[5] Ploughing depth was 30 cm, and thus crop residues were mixed into 0–30 cm layer. Ploughing was conducted on Julian day 300 i.e. 28th October on regular years and 29th October on leap years.

[6] The marginal damage cost of the C loss reflects the marginal decrease in peoples' recreational benefit when the water clarity decreases marginally. Dissolved organic matter decreases water clarity by absorbing light and turning it brown.

[7] The social cost of carbon (SCC) is a controversial concept, as it is prominently uncertain (Anthoff and Tol, 2013), and even considered to be misleading (Pezzey, 2018). In addition, a global SCC ignores the heterogeneous geography of climate damage and differences in country-level contributions to the global SCC (Ricke et al., 2018). Therefore, the range of possible estimates for the SCC is more relevant than a single estimate. Thus, a sensitivity analysis of the results with respect to the choice of the SCC is important.

[8] For example, there may be a negative correlation between decreased water clarity and eutrophication, because reduced water clarity reduces light penetration, which in turn reduces photosynthesis at lower depths and biological productivity of the lake ecosystem.

[9] The CoupModel is a process-based model developed to calculate water and heat fluxes and C and N cycles in the soil profile, which is divided into a number of layers (Jansson and Karlberg, 2004).

[10] The transformation was performed with Matlab (MathWorks Inc., 2019).

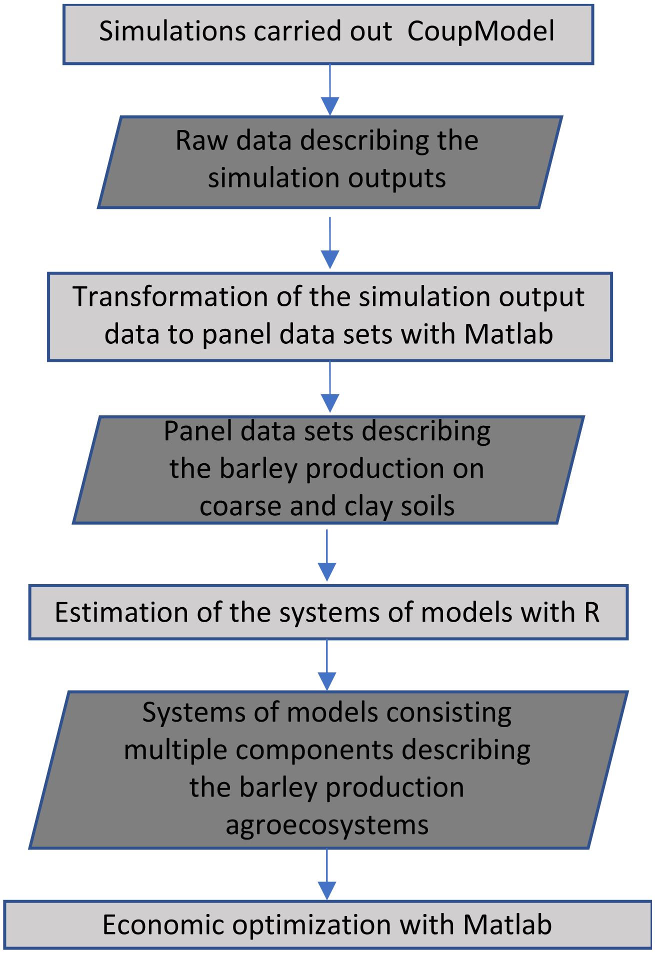

Fig. 2. Schematic diagram of the modeling process.

We applied three-stage least squares (3SLS) for estimating the simultaneous linear structural equation models (Zellner and Theil, 1962).[11] In the final step, these models were used for the economic optimization, i.e., to solve the problem described using Eq. (12). Numerical optimization was carried out with Matlab.

The economic parameters used in the study are shown in Table 1. The planning horizon was set to 150 years. Temperature and precipitation were treated as constant parameters in the optimization problem (they were fixed to their average level). It is clear that, given the effect of climate change, temperature and precipitation are not constants over the 150-year planning horizon. It is equally clear that prices are not constant over such a long planning horizon. A long-term horizon is applied to avoid the influence of the final period on the result, and to study whether the optimal path converges to a steady state or not. The influence of the far-future periods on the annual input decisions, however, is minor because the associated costs and benefits are discounted. The choice of a discount rate is pivotal in the long-term analysis (Weitzman, 1998, 2001), and therefore we studied the sensitivity of the results to this choice (see S1).

Table 1
Estimated price parameters applied in the numerical model.

Parameter	Estimated value	Source
Barley price	€ 0.136 kg^{-1}	www.lantmannenagro.fi
N fertilizer cost	€ 0.91 kg^{-1}	Luke (2015)
Manure spreading cost	€ 0–150 ha^{-1} (€ 75 ha^{-1} in the baseline scenario)	Decided by the authors. The spreading cost of manure depends on the distance between the barn and the field. The cost is unknown and is therefore treated as a sensitive parameter with a range of € 0–150 ha^{-1} (S3).
Manure price	€ 0 to 100 ha^{-1} (€ 50 ha^{-1} in the baseline scenario)	Decided by the authors. The price of the manure is unknown and it is treated as a sensitive parameter with a range of € 0 to 100 ha^{-1} (S3).
Marginal damage of N loss	€ 6.6 kg^{-1}	Gren and Folmer (2003)
Marginal damage of C loss*	€ 2.13 kg^{-1}	Miettinen et al. (2020), based on Ahola and Havumäki (2008) and Luhta (2017)
Social cost of carbon	€ 27.8 tCO_2^{-1} (31 tCO_2^{-1} in 2010 US dollars, for 2015)	Nordhaus (2017)
Discount rate	3%	Decided by the authors. The sensitivity of the results to the choice of the discount rate was studied (S1).

* Estimate for marginal damage of carbon to water bodies is not yet available. There are, however, some estimates for the damage of the sediment loss from peatlands in Finland. If we assume that the damage of sediment loss to the receiving water bodies is essentially similar from agricultural land and from peatlands, we may transfer these estimates to agriculture. Miettinen et al. (2020) model Finnish peatland forestry in northern Finland and present two marginal damage estimates for the sediment load damage based on their calculations and existing literature: € 4.1 kg^{-1} based on Ahola and Havumäki (2008) and € 0.16 kg^{-1} based on Luhta (2017). We use the average of these estimates.

4. Results of the case study

4.1. Model performance

The results of the 3SLS estimation for both soil textures are shown in A4. McElroy's R squared (McElroy, 1977) was 0.71 and 0.88 for models for clay and coarse textured soils, respectively, suggesting that the estimated models explain variation within the associated data sets with reasonable success. The adjusted r^2 of the individual model components ranged from 0.13 to 0.95 for the coarse soils models and from 0.28 to 0.8 for the clay soils models (Tables A1, A2). The model performance was best for the model describing N_2O emissions and worst for the model describing a N carryover, for both soil textures. Most of the correlation of the residuals of the systems of models were small, suggesting that the models describe the error structure of the data with reasonable accuracy (Tables A3, A4).

We also studied how the parameter estimates and the standard errors change if the two other methods for simultaneous equation estimation were used: two-stage least squares (2SLS) and seemingly unrelated regression (SUR). In addition, we estimated all the equations separately by ordinary least squares (OLS). Parameter estimates of the estimation methods were close to each other. In some equations, 3SLS and SUR estimates were similar but somewhat different from the 2SLS and OLS estimates. In the case of clay soils, 2SLS and OLS estimators give a different (and probably wrong) sign for the product term of C loss and total runoff in the C carryover equation. In addition, in the N loss equation, 2SLS and OLS estimators give a different (and probably wrong) sign for the quadratic term of the soil N. Standard errors for 3SLS estimates were smaller than for 2SLS estimates. However, in general, standard errors differ only a little across the different estimation

[11] This method was applied because it allows contemporaneous correlation across the individual equations within the system. The necessary condition for identification requires that each equation in the structural form of the system should exclude at least one exogenous variable that is present in the other equations. This requirement holds in this case, because various transformations of the exogenous variables were used in the equations. The estimation was carried out with statistical software R (R Core Team, 2018) with the systemfit package (Henningsen and Hamann, 2007).

methods. Surprisingly, for both coarse and clay soil models, most of the residual correlations were smallest with the 2SLS method and largest with the 3SLS method. Nevertheless, correlations were rather small even with the 3SLS method.

4.2. Cases studied in the economic optimization

We study four cases in the economic calculations (Fig. 3). In case 1, the decision maker ignores both the water and atmospheric externalities, implying that $\mu^N = \mu^c = \mu^{GHG} = 0$ in Eq. (12). This case corresponds to the private optimum, and it is a reference case for the input and output reductions in other cases. In case 2, producers consider the water externalities but ignore the atmospheric externalities, implying that $\mu^{GHG} = 0$. In case 3, producers consider the atmospheric externalities but ignore the water externalities, implying that $\mu^N = \mu^c = 0$. In case 4, producers consider both the water and atmospheric externalities. This case corresponds to the social optimum, and it is a reference case in social welfare loss calculations. We calculate social welfare losses of the externalities as follows: we run the economic optimization model for case 4 and store the obtained optimal social NPV, denoted by NPV_4. Second, we run the economic optimization model for some other case, for example case 2, and store the obtained input vectors to obtain suboptimal input vectors corresponding to the producer's annual input choices when the atmospheric externalities of the production are ignored. Third, we simulate the economic optimization model for case 4 using the obtained suboptimal input vectors from case 2 and store the suboptimal NPV, denoted by NPV_2. When we run case 4 for the suboptimal input vector, the higher atmospheric emissions compared with those in the social optimum obtain their monetary value, and thus NPV_2 captures the cost of the externality that the producer ignored. Fourth, we obtain the social welfare loss of the atmospheric externality as the difference between NPV_2 and NPV_4. We have to take the difference, because GHG emissions are not zero in the social optimum either. We obtain the private cost in a similar fashion with the exception that the reference case is case 1 instead of case 4.

4.3. Differences between private and social optimums

Both inorganic N fertilizer and organic manure are used in private (case 1) and social (case 4) optimums on both soil textures (Table 2). The privately optimal manure rate is 70–80% higher than that in the social optimum, implying that manure greatly increases the loss processes. The socially optimal N fertilizer rate on clay soil is 2.6% higher than that in the private optimum, implying that N fertilizer has a negative net effect on the GHG emissions from clay soils through increased photosynthesis. The N fertilizer rate is higher in the social optimum also due to reduction of manure use, which means that less plant available N is coming from manure to the crop. However, on coarse soils the socially optimal N fertilizer rate is 56% lower than that in the private optimum, implying that the net effect of the N fertilizer on GHG emissions is positive: the denitrification effect dominates the photosynthesis effect. These results suggest that clay soils can retain more C compared with coarse soils.[12]

4.4. Effect of the water and atmospheric externalities on the cultivation practices

When only the water externalities are considered (case 2), the optimal N fertilizer application rate decreases by approximately 60% compared with the private optimum (case 1) on coarse soils due to a high amount of N leaching (Table 2). On clay soils, the impact of considering the water externalities on the optimal N fertilization rate is minor due to a lower amount of N leaching.[13] The effect of considering the water externalities is even greater on the optimal manure application rate, which decreases by 70–80% on both soil textures, suggesting that manure is a driving factor of both the N and C leaching on both soil textures.

When only atmospheric emissions are considered (case 3), the socially optimal N fertilizer rate is increased on clay soils, compared with case 1, because the net effect of N fertilizer on GHG emissions is negative on clay soils. However, on coarse soils the N fertilizer rate is lower in case 3 compared with case 1, because the net effect of N fertilizer on the GHG emissions is positive on coarse soils. The manure rate is lower on both soil textures in case 3 compared with case 1 due to the positive net effect of manure application on GHG emissions. Photosynthesis, soil respiration, and denitrification are all increasing functions of manure. GHG emissions from the field, on the other hand, are an increasing function of soil respiration and denitrification, and a decreasing function of photosynthesis. The result implies that the effect of the soil respiration and denitrification dominates the effect of the photosynthesis. Also, the manure rate decreases much more on coarse soils (−69%) than on clay soils (−24%), implying that the soil respiration and denitrification response is greater on coarse soils.

4.5. Synergies and trade-offs between different agricultural goals

There are synergies between water protection and climate change mitigation goals: in case 2, GHG emissions also decrease compared with the private optimum, and in case 3 both water losses are decreased (Table 2). In case 2, GHG emissions decrease because the input rates of both inorganic and organic fertilizers are lower than those in the private optimum. This results from the high N and C leaching responses, particularly to manure application. Surprisingly, on both soil textures GHG emissions are reduced even more in case 2 than in case 3.

In case 3, N and C leaching rates decrease because manure application decreases compared with case 1. On clay soils, however, the reduction in N loss is slight, because the inorganic N rate is increased to boost the photosynthesis. There is, however, a trade-off between water or atmospheric pollution reduction and crop production goals (Table 2). In case 2, yields are 13% and 24% lower compared with the private optimum, on clay and coarse soils, respectively. In case 3, yields are 1.5% and 11% lower compared with the private optimum, on clay and coarse soils, respectively.

4.6. Relative importance of water and atmospheric externalities

When only water externalities are considered and the atmospheric externalities are ignored, the social welfare loss is almost zero on both soil textures (Table 2). Instead, considering only atmospheric externalities and ignoring water externalities causes notable social welfare losses. However, the relative importance of the externalities depends on the applied damage costs of the water losses and the SCC. The threshold value for the SCC, where social costs of ignoring GHG emissions become greater than those of ignoring water externalities, is ∼€ 105 tCO_2^{-1} on coarse and ∼ € 240 tCO_2^{-1} on clay soils, when the damage costs of N and C losses to water bodies are at their baseline levels (Fig. 4). The marginal damage costs of N and C losses to water bodies measure people's valuation of the water pollution in some particular location. The damage cost

[12] This result is in agreement with some findings from the previous literature (Rasmussen and Rohde, 1988; Zhen et al., 2014). However, if the whole life cycle of the N fertilizer (including manufacturing of the fertilizers, which is a highly energy-intensive process) were considered, the net effect of the N fertilizer on the GHG emissions would most likely be positive (cf. Schlesinger, 1999).

[13] It is a general observation that the annual rate of N leaching from coarse soils is notably higher compared with that from clay soils (Bauer and Black, 1981; Campbell and Souster, 1982; Coote and Ramsey, 1983; Nichols, 1984), and it is caused by the clay soil's greater ability to hold nutrients compared with that of coarse soils (Foth, 1990; Hazelton and Murphy, 2007).

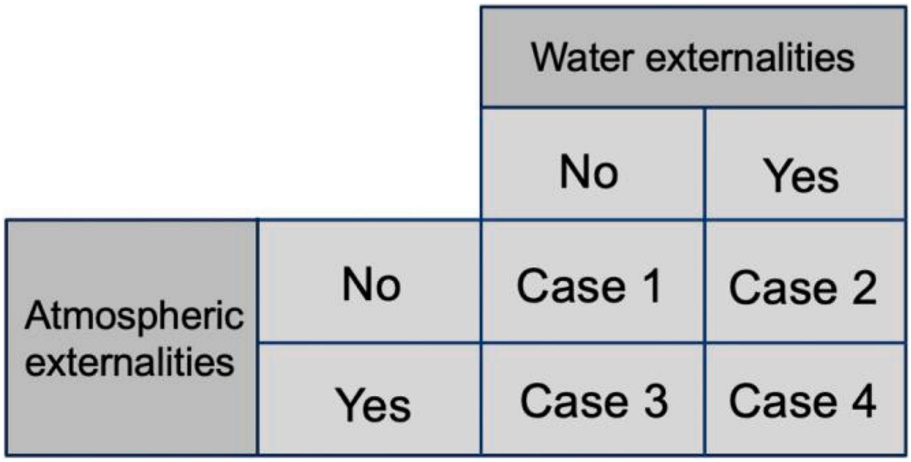

Fig. 3. Four cases according to which externalities are considered in the decision-making process: case 1 ignores both externalities (private optimum), case 2 considers water externalities but ignores atmospheric externalities, case 3 considers atmospheric externalities but ignores water externalities, and case 4 considers both externalities (social optimum).

Table 2
Mean (over the whole planning horizon: 150 years)* inorganic N rates, manure rates, soil N and C stocks, N and C losses to water bodies, net GHG emissions, and yields in different cases (case 1: no externalities considered; case 2: water externalities considered; case 3: atmospheric externalities considered; case 4: both externalities considered) on clay and coarse soils**.

Case/ Soil texture	Social welfare loss (€ ha^{-1})	Private cost (€ ha^{-1})	Mean N fertilizer rate (kg ha^{-1} yr^{-1})	Mean manure rate (kg ha^{-1} yr^{-1})	Mean soil N stock (kg ha^{-1} yr^{-1})	Mean soil C stock (kg ha^{-1} yr^{-1})	Mean N leaching (kg ha^{-1} yr^{-1})	Mean C leaching (kg ha^{-1} yr^{-1})	Mean net GHG emissions (CO_2-eq kg ha^{-1} yr^{-1})	Mean yield (kg ha^{-1} yr^{-1})
Clay soils										
Case 1	2062	0	54.1	8.56e+03	1.68e+04	1.76e+05	31.9	88.5	588	4110
Case 2	10	803	52.4	2.42e+03	1.43e+04	1.54e+05	20.5	80.5	364	3580
Case 3	1076	63	63.5	6.51e+03	1.60e+04	1.69e+05	28.4	85.3	397	4050
Case 4	0	716	55.5	2.60e+03	1.44e+04	1.55e+05	20.9	80.6	345	3630
Coarse soils										
Case 1	7220	0	112	6.51e+03	1.53e+04	1.65e+05	77.8	127	3600	4250
Case 2	71	1990	48.5	1.67e+03	1.37e+04	1.45e+05	45.7	120	915	3130
Case 3	1760	609	96.3	2.05e+03	1.36e+04	1.48e+05	58.7	118	1435	3780
Case 4	0	2270	48.9	1.30e+03	1.35e+04	1.43e+05	44.5	120	773	3050

* Time paths for the variables are shown in S2.
** Parameters are at their baseline level.

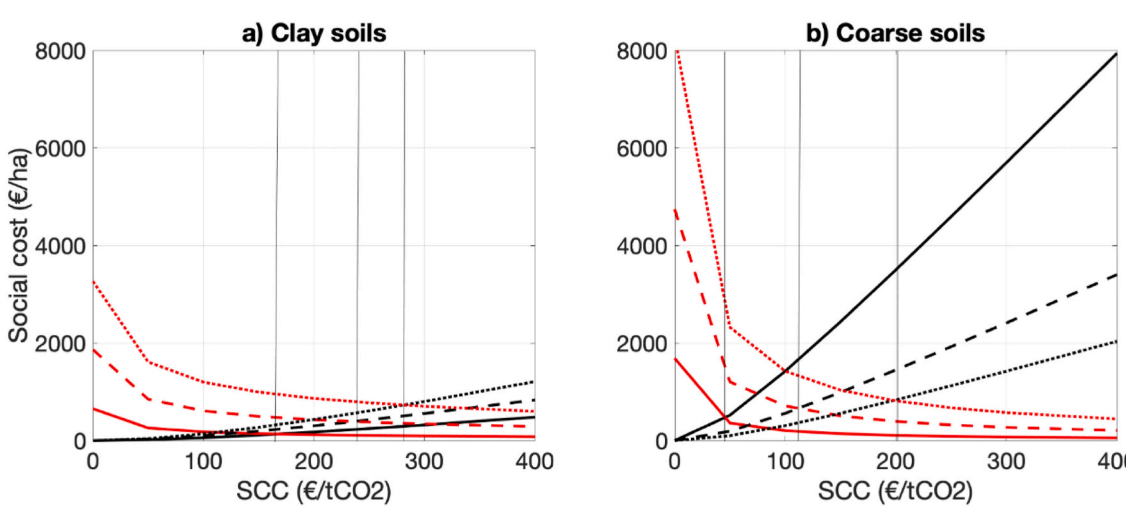

Fig. 4. Social cost in case 2 (black lines) and case 3 (red lines) as a function of the social cost of carbon (SCC) (note that the social welfare loss of cases 2 and 3 is calculated as the difference between the associated NPV and the NPV of case 4, i.e., the social optimum). Solid lines indicate the cases where the marginal damage costs of the water losses are 50% lower than the baseline estimates. Dashed lines are the baseline cases, and dotted lines are the cases where the damage costs are 50% above the baseline. Vertical straight lines show the SCC values where cases 2 and 3 intersect. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

is high for a pollution source that has a significant effect on the receiving water ecosystem and low for an insignificant source. If the damage costs of N and C losses to water bodies were 50% lower than the baseline, the threshold value for the SCC would be ~€ 38 tCO_2^{-1} and ~€ 130 tCO_2^{-1} on coarse and clay soils, respectively (Fig. 4). If the damage costs of N and C losses to water bodies were 50% higher than the baseline, the threshold value for the SCC would be ~€ 200 tCO_2^{-1} and ~ € 290 tCO_2^{-1} on coarse and clay soils, respectively (Fig. 4).

4.7. Marginal abatement cost curves for greenhouse gas emissions and N and C losses

We also study the abatement costs of the emissions to water and the atmosphere. The analysis is motivated by the suggestion by Pezzey (2018), according to which any SCC estimate will always be disputed. We obtain the total abatement costs for GHG emissions and N and C losses to water bodies by gradually increasing the marginal damage cost, calculating the resulting private NPVs, and subtracting them from the private NPV associated with the case where the marginal damage cost is zero. The abatement is calculated as the difference between cumulative emissions over the planning horizon for each marginal damage cost, and cumulative emissions for the case where the marginal damage cost is zero.

The marginal abatement cost curves (MACCs) are obtained by fitting curves to the obtained total costs and differentiating the fitted curves with respect to the level of abatement. The total costs and the fitted curves are shown in A5. The marginal abatement costs of the GHG emissions are almost zero on coarse soils until the abatement requirement is above 3000 kg CO_2-eq ha^{-1} (Fig. 5b), whereas they start to increase rapidly on clay soils when the abatement requirement is above 400 kg CO_2-eq ha^{-1} (Fig. 5a). Thus, climate change mitigation efforts should be first targeted at coarse soils. Also, the N and C loss mitigation efforts should first be targeted at coarse soils, because the N and C loss marginal abatement costs start to increase for lower abatement requirements on clay soils than on coarse soils (Fig. 5c, d, e, f). However, the difference in MACCs is minor for C losses.

In addition, when the producer uses both manure and fertilizer inputs simultaneously, the marginal abatement costs are notably lower compared with the situation where manure is not available. This means that simultaneous adjustment of the inputs enables better adaptation to the abatement requirements. The exception is the N loss on coarse soils where, interestingly, the marginal abatement costs are higher when both manure and N fertilizer inputs are used compared with the case where only N fertilizer input is used (Fig. 5d). On coarse soils, the NPV is higher when both inputs are used, but when the abatement requirements for the N losses increase, the NPV decreases at a higher rate when both inputs are used compared with the case where only N fertilizer is used, because in such a case the NPV is lower even without abatement. However, the total costs are almost the same in both cases. Fig. 5 also shows that the benefit of the simultaneous adjustment of the inorganic and organic fertilizer inputs in terms of reduced abatement costs is far greater on clay soils, suggesting that it would be expensive for the producer to reduce emissions from the field in an inorganic cropping system on clay soils.

4.8. Socially optimal paths for soil carbon and nitrogen

The optimized fertilizer management leads to increasing N and C stocks on clay soils in cases 1 and 3, suggesting that there is a synergy between C sequestration and crop production goals and C sequestration and climate change mitigation goals. Instead, the stocks are declining in the cases where water externalities are considered (cases 2 and 4) (Fig. 6a, c), suggesting that there is a trade-off between C sequestration and water protection goals. The explanation is the reduction in manure use and the fact that manure was the only C input in the study setup. When only atmospheric externalities are considered, it is optimal to increase C sequestration by applying both more inorganic and organic fertilizers. On coarse soils the soil stocks are declining in every case (Fig. 6b, d), suggesting that the optimal level of the soil C and N stock is below the initial level. The decline stems also from the applied prices (S3). Fig. 6 shows also that the optimal soil N and C paths are almost identical in cases 2 and 4, because the water externalities have such a strong influence on the input decisions. Also, the soil C/N ratio remains

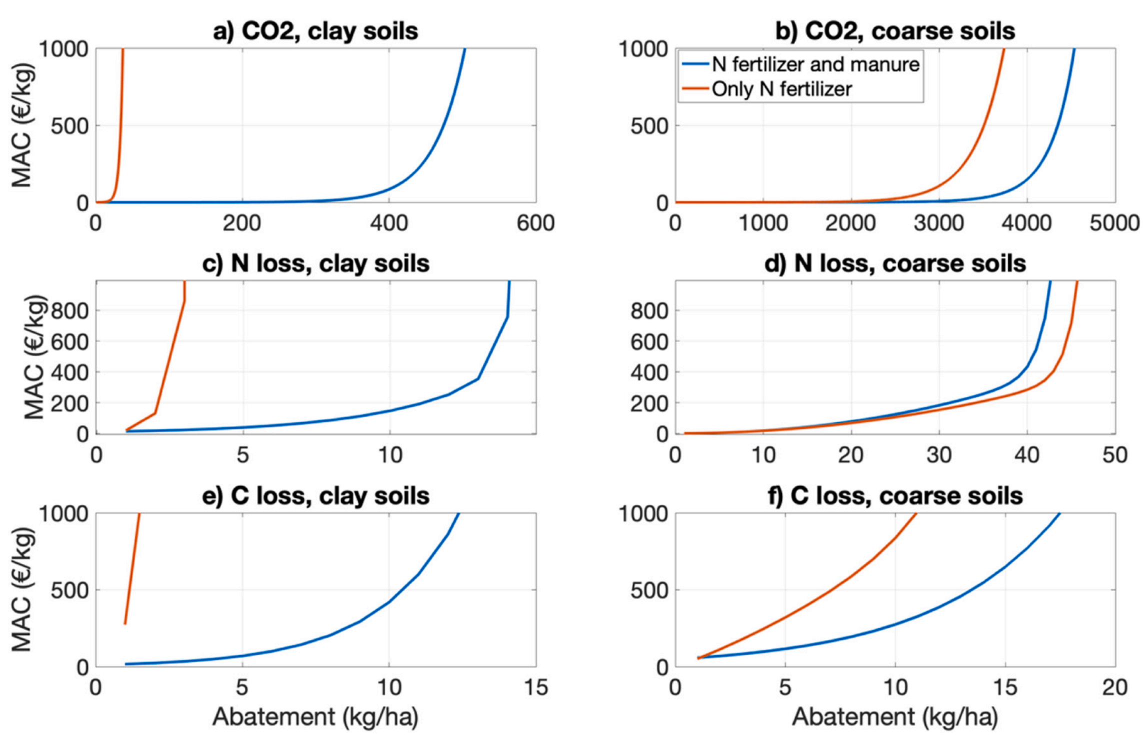

Fig. 5. Marginal abatement costs (MACs) of greenhouse gas emissions (a and b), N loss to water bodies (c and d), and C loss to water bodies (e and f) on clay and coarse soils for the cases where both inorganic and organic fertilizers are used and where only N fertilizer is used.

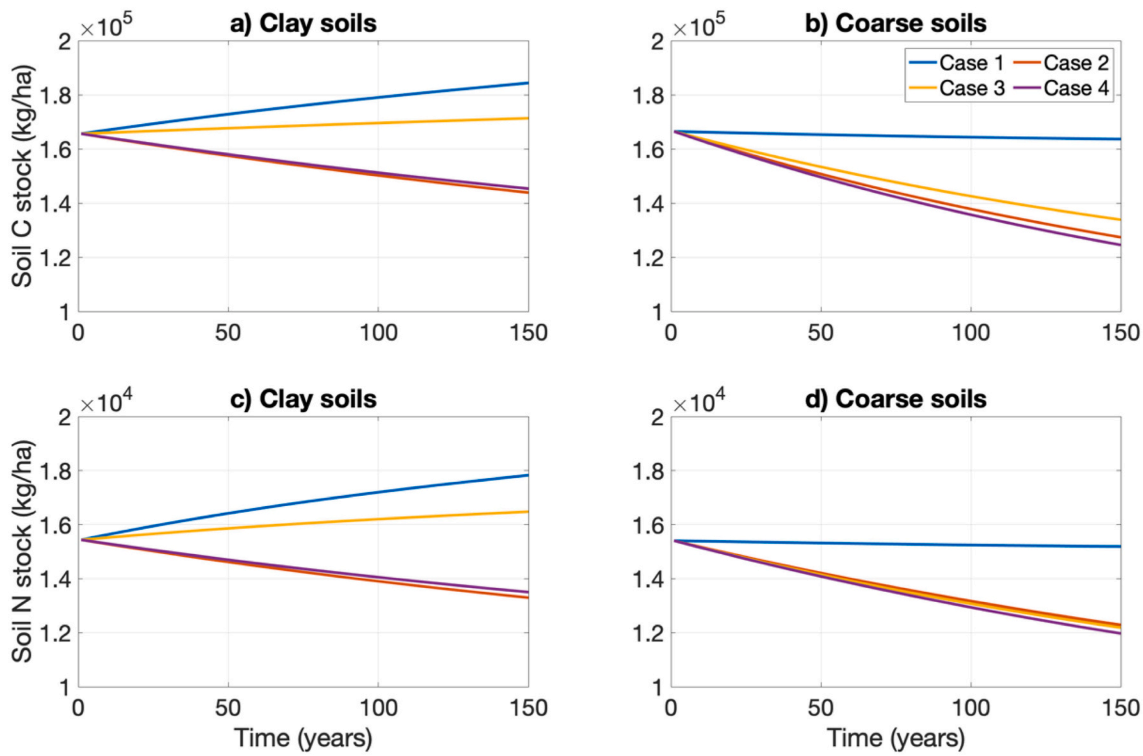

Fig. 6. Soil C and N stock development paths in cases 1–4.

almost the same throughout the planning horizon in the different cases (Fig. 6c, d). Last, Fig. 6 demonstrates the obtained theoretical result: the planner depletes the soil stock at a higher rate when the loss processes are high compared with the case where the loss processes are moderate.

Despite the declining trends shown in Fig. 6, soil N and C stocks are not completely depleted in any of the cases. Fig. 7 shows that if the initial soil N and C stocks are low, they are increased in all the cases by applying more of both fertilizers in the beginning of the planning horizon (S2). The convergence is slightly faster on coarse soils than on clay soils due to the naturally slower rate of change on clay soils, which results from the clay soil's greater ability to hold nutrients. The optimal soil C stock is higher on clay soils, which highlights the obtained theoretical result that the absolute value of the shadow prices of the soil stocks are decreasing functions of the water externalities. Hence, a rate of convergence toward the steady state is slower on clay soils, but the steady-state soil C stock is higher on clay soils than on coarse soils. The result supports the general observations from the previous literature, according to which clay content decreases the transition rate of soil C but increases the level of the steady-state soil C stock through its control on the accumulation and mineralization of C (Oades, 1988; Wang et al., 2017; Zhao et al., 2018). Clay soils can store more C than coarse soils due to their small particle size (Christensen, 2001).

5. Discussion

This study is the first attempt to include both N and C soil dynamics as well as the water and atmospheric externalities in the economic analysis to obtain insights about the sustainable fertilization management and abatement costs in crop production. We carried out an extensive modeling process in which we constructed large systems of

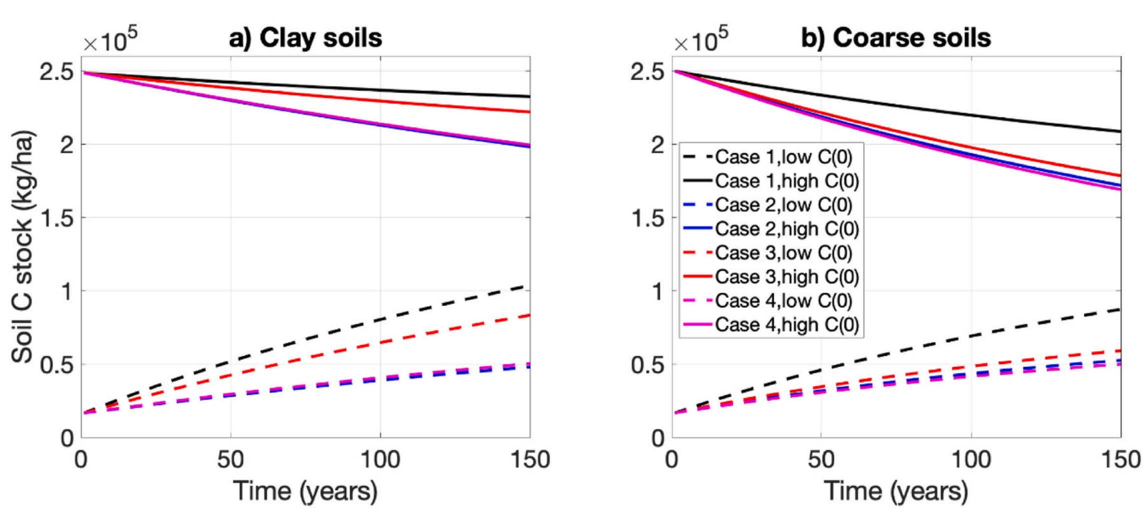

Fig. 7. Soil C development paths in cases 1–4 for high and low initial states.

models from simulated data. Complex and dynamic systems of models are necessary for describing agroecosystems, and they are increasingly used for solving problems in food production (Wallach et al., 2014). Complex interactions of the various biophysical components of agro-ecosystems are captured by detailed simulation models, which, however, are not directly usable for an economic optimization of the input use. To overcome this problem, we followed a tradition of simulating data with existing simulation models (Larson et al., 1996; Watkins et al., 1998; Huang et al., 2001; Martínez and Albiac, 2004). Our approach provides a framework for constructing models for an economic optimization of the inputs in crop production, by using a detailed simulation model as a starting point.

Our results suggest that there are synergies between water protection and climate change mitigation goals. Consequently, the social optimum was almost reached when only the water externalities were considered in decision-making. This result is in contrast to the previous literature regarding the regulation of an animal farm, stating that regulation of only water (air) emissions might inadvertently increase air (water) emissions (Aillery et al., 2005). The result can be explained by the high leaching responses to fertilizers. The sensitivity analysis showed that these results were surprisingly insensitive to the damage costs of water externalities (S4, S5). The relative importance of the atmospheric and water externalities, nevertheless, depends on the marginal damage costs. On fields where the GHG emissions are high, and which are not an important source of water pollution, atmospheric externalities are more important than water externalities, even for a relatively low SCC. If the fields are great sources of water pollution, the SCC would have to be very high before atmospheric externalities dominate water externalities. Moreover, if SCC increased in time with a 2% growth rate (cf. Nordhaus, 2017; Tol, 2011), the threshold value where the atmospheric externalities are more crucial than water externalities was reached in about 70 years on coarse soils and 110 years on clay soils (S1).

Our results also suggest that manure can be used to increase soil C stock (cf. Matson et al., 1997; Smith, 1997; Buyanovsky and Wagner, 1998). However, the net effect of the manure application on GHG emissions was positive, because manure was a driving factor of soil respiration (cf. Rochette and Gregorich, 1998; Lai et al., 2017; Yang et al., 2018), which is a primary source of CO_2 emissions from agricultural soils (Paustian et al., 2000; Schlesinger and Andrews, 2000; Lai et al., 2012). Thus, our results are in agreement with the conclusion of Schlesinger (1999) that manure is not a solution to agriculture's GHG emissions problem. It must be noted that we excluded the opportunity cost of the manure application from the analysis. The alternative use of manure may be even worse for the environment. Schlesinger (1999) claims, however, that considering the whole life cycle of the manure, including the production of the fodder and the associated deforestation, notably increases the social cost of the manure application compared with only partial life-cycle analysis.[14] Nevertheless, according to our results, manure was applied to some extent even in the cases where both the water and atmospheric externalities were considered, because manure application significantly increased crop yield (cf. Christopher and Lal, 2007; Salehi et al., 2017; Geng et al., 2019). Hence, our results suggest that an integrated nutrient management system is better than a system where only inorganic or organic fertilizer is used, in agreement with some previous studies (Basri et al., 2013; Omidire et al., 2015; Mahmood et al., 2017).

We found also that there were trade-offs between crop production and the mitigation goals. The trade-off between crop production and water protection goals is well known in the previous literature (Vitousek et al., 1997; Foley et al., 2005). Our result showing a trade-off between climate change mitigation and food production goals, however, is in contrast to the results in the previous literature (Bauer and Black, 1994; Lal and Bruce, 1999; Lal, 2004; Bot and Benites, 2005). This result could be different if there were some other ways to increase C stock than manure application, for example changing to no-till practices in crop production (Paustian et al., 1997; Lal et al., 1999; Follett, 2001). However, the role of no-till in climate change mitigation may be overstated, although it is beneficial for soil quality and climate change adaptation (Robertson et al., 2000; Baker et al., 2007; Powlson et al., 2014; VandenBygaart, 2016). When manure application was not possible, there indeed was a synergy between food production and climate change mitigation goals on clay soils. This was because the net effect of the N fertilizer on GHG emissions was negative (S7). At the same time, water pollution increased, implying that there was a trade-off between a water protection goal and a climate change mitigation goal. This result is in agreement with the previous literature (Aillery et al., 2005). Moreover, the net effect of N fertilizer on GHG emissions was positive on coarse soils, in agreement with the previous results (Robertson et al., 2000).

According to our results, GHG emissions and N and C leaching mitigation efforts should first be targeted at coarse soils, rather than clay soils, because the marginal abatement costs are considerably lower on coarse soils. In addition, integrated application of both inorganic and organic fertilizers is associated with lower marginal abatement costs of GHG emissions and C leaching to water bodies, independently of the coarse textures, compared with those associated with the application of only inorganic N fertilizer. The integrated fertilizer strategy leads also to lower marginal abatement costs of N leaching on clay soils, where annual N losses to water bodies are moderate. On coarse soils the application of only the inorganic fertilizer was associated with lower marginal abatement costs of N loss than integrated fertilization, because in such a case NPVs decrease at a higher rate for increasing abatement requirements. It must be noted that the marginal abatement costs could be considerably lower if more abatement measures were considered. For example, Moran et al. (2010) considered 97 abatement measures when estimating marginal abatement costs of the GHG emissions in UK.

Our results also showed that social welfare is not necessarily an increasing function of the rate of C sequestration on agricultural soils (cf. Amundson and Biardeau, 2018). According to our results there is a synergy between C sequestration and crop production goals on clay soils (cf. Matson et al., 1997; de Ridder et al., 2004; Roy et al., 2006; Thiaw et al., 2011). The rate of increase in C stocks in the private optimum on soils with low initial C stock was comparable to the results of some previous studies (e.g., Jenkinson et al., 1994; Powlson, 1994). When the initial soil C stock was low, it was increased by applying high inorganic and organic fertilizer rates, independently of the soil texture and the considered externalities (S2). This result supports the conclusion from the previous literature that one way to prioritize support for increased soil C sequestration is to identify those fields where soil C content is particularly low and where the links to food production gains are strongest (Dickie et al., 2014; Sanderman et al., 2017). For example, previous papers have suggested that the declining trend in soil C observed on agricultural soils in Finland could be slowed by applying organic manure (Akujärvi et al., 2014). In fact, insufficient organic C inputs is the primary cause leading to declining trends of soil C in Europe (Ciais et al., 2010). Moreover, there was clearly a trade-off between the C sequestration and water protection goals, independently of the soil texture.

There are multiple ways to extend the model used in this paper. First, phosphorus (P) could be included in the analysis, because P dynamics are an essential factor in optimizing long-term nutrient management. P was excluded from the analysis because the CoupModel does not include P. We also ignored the effects of climate change on agricultural productivity and the externalities (cf. Abler et al., 2002; Howden et al., 2007; Gornall et al., 2010). These effects could be important here, because the time horizon of the analysis is such long. However, the costs and benefits occurring in the distant future become eventually meaningless because those are discounted. Nevertheless, the effects of climate

[14] One possible way to reduce manure's C footprint is to use it for energy, for example for biofuel production.

change could be included, for example, by considering various climate change scenarios. However, because the model includes economic variables, also the socioeconomic scenarios would have to be considered in parallel with the climate change scenarios. Such a scenario analysis would be a natural extension of this study. Third, the possibility of crop rotations should be included in the analysis, because the N fertilizer requirement for optimal crop yield is often reduced in rotations compared with monoculture (Schmid et al., 1959; Heichel and Barnes, 1984; Franzluebbers et al., 1994). In addition, a model could be extended for other crops. Barley production does not require as much inputs as production of, for example, wheat. Therefore, the private costs, and the externalities, could be higher for wheat production systems than for barley production systems. Fourth, we could expand the analysis to include land-use change and alternative management practices/technologies. Fifth, the obtained models should be validated ideally with empirical data or alternatively by using other simulation models. Note, however, the results presented here are based on the calibrated and validated model (CoupModel). Last, the model could be used to study agri-environmental policy instruments, such as tax-subsidy schemes.

6. Conclusions

The social costs in terms of increased GHG emissions outweigh the gains of C sequestration on soils with moderate soil C stocks or high annual losses to water and the atmosphere. Thus, other technological solutions than spreading manure could be considered for increasing the climate change mitigation effect of cropland. Nevertheless, according to our results, an integrated nutrient management system is better than a system where only inorganic or organic fertilizer is used. Our results also suggest that water externalities are as important in determining the socially optimal cultivation practices as the atmospheric externalities, unless the field is a considerable source of GHG emissions and an insignificant source of water pollution. Also, there are trade-offs between climate change mitigation (as well as water protection) and crop production targets, and synergies between mitigation targets of water and atmospheric emissions. If manure was not an option, there was a synergy between climate change mitigation and crop production targets on clay soils. Moreover, our results suggest that GHG emissions and N and C leaching mitigation efforts should first be targeted at coarse soils, rather than clay soils, because of the considerably lower marginal abatement costs.

Declaration of Competing Interest

None.

Acknowledgements

This work was carried out as a part of the Oranki-project, which was financed by the Finnish Ministry of Agriculture and Forestry.

Appendix

A.1. Optimality conditions for the annual input decisions

The problem of a planner is the following:

$$\max_{N_t,\varphi_t,n_{t+1},c_{t+1}} \sum_{t=0}^{\infty} \beta^t \left[p^y y_t - p^N N_t - \varphi_t\left(\tau^M + p^M\right) - \left(\mu^N e_t^N + \mu^c e_t^c + \mu^{GHG} e_t^{GHG}\right) \right] \tag{A1}$$

Subject to
$n_{t+1} = n_t + \vartheta^n(n_t, c_t, N_t, \varphi_t, e_t^N, \xi_t, y_t; temp_t, ppt_t)$ and
$c_{t+1} = c_t + \vartheta^c(c_t, n_t, \varphi_t, e_t^c, R_t^s, \delta_t; temp_t, ppt_t)$,
$N_t \geq 0$, and $\varphi_t \geq 0$, with n_0 and c_0 given
The Lagrangian of the problem Eqn A1 is the following:

$$L = \sum\nolimits_{t=0}^{\infty} \left(\beta^t\, p^y y_t - \beta^t \left[p^N N_t + \varphi_t\left(\tau^M + p^M\right) \right] - \beta^t \mu^N e_t^N - \beta^t \mu^c e_t^c - \beta^t \mu^{\mathrm{GHG}} \left(R_t^s + R_t^p + C_t^y + \epsilon\xi_t - \delta_t\right) + \beta^{t+1} \lambda_{t+1}^n \left[n_t + \vartheta_t^n - n_{t+1}\right] + \beta^{t+1} \lambda_{t+1}^c \left[c_t + \vartheta_t^c - c_{t+1}\right] \right) \tag{A2}$$

where

$$y_t = y(\delta_t; temp_t, ppt_t)$$

$$\delta_t = \delta(N_t, \varphi_t, n_t; temp_t, ppt_t)$$

$$e_t^N = e^N(N_t, \varphi_t, n_t, y_t; temp_t, runoff_t)$$

$$e_t^c = e^c(\varphi_t, c_t; runoff_t)$$

$$R_t^s = R^s(c_t, \varphi_t, N_t; temp_t, ppt_t)$$

$$R_t^p = R^p(y_t; temp_t)$$

$$\xi_t = \xi(\varphi_t, N_t, y_t, \eta_t; temp_t)$$

$$\eta_t = \eta(N_t, \varphi_t, n_t; temp_t)$$

$$C_t^y = C(y_t)$$

$$n_{t+1} = n_t + \vartheta^n\left(n_t, c_t, N_t, \varphi_t, e_t^N, \xi_t, y_t; temp_t, ppt_t\right)$$

$$c_{t+1} = c_t + \vartheta^c\left(c_t, n_t, \varphi_t, e_t^c, R_t^s, \delta_t; temp_t, ppt_t\right)$$

In (A2), λ_t^n and λ_t^c are the Lagrange multipliers reflecting the shadow prices of the soil N and C stocks, respectively. The shadow price of a given stock measures a marginal change in social welfare, as the stock changes marginally. Because a yield response function is concave, first-order conditions (multiplied with β^{-t}) for the annual optimal decisions are obtained by differentiating Eqn A2. The condition for an optimal annual N fertilizer rate is determined as follows:

$$\boldsymbol{N_t}\ (\text{interior}): p^y y_{\delta,t}\delta_{N,t} + \beta\lambda^n_{t+1}\left(\vartheta^n_{N,t} + \vartheta^n_{e^N,t}\left(e^N_{N,t} + e^N_{y,t}y_{\delta,t}\delta_{N,t}\right) + \vartheta^n_{\xi,t}\left(\xi_{N,t} + \xi_{y,t}y_{\delta,t}\delta_{N,t} + \xi_{\eta,t}\eta_{N,t}\right) + \vartheta^n_{y,t}y_{\delta,t}\delta_{N,t}\right) + \beta\lambda^c_{t+1}\left(\vartheta^c_{R^s,t}R^s_{N,t} + \vartheta^c_{\delta,t}\delta_{N,t}\right)$$
$$= p^N + \mu^N\left(e^N_{N,t} + e^N_{y,t}y_{\delta,t}\delta_{N,t}\right) + \mu^{\text{GHG}}\left(R^s_{N,t} + R^p_{y,t}y_{\delta,t}\delta_{N,t} + C^y_{y,t}y_{\delta,t}\delta_{N,t} + \epsilon\left(\xi_{N,t} + \xi_{y,t}y_{\delta,t}\delta_{N,t} + \xi_{\eta,t}\eta_{N,t}\right) - \delta_{N,t}\right), \tag{A3}$$

where $p^y y_{\delta,t}\delta_{N,t}$ is the marginal value product (MVP) of N fertilizer. MVP measures the market value of a yield increase obtained by increasing input use marginally. The term p^N is the marginal cost of N fertilizer. The environmental marginal damages of N fertilizer consists of the effect of N fertilizer on N loss and on GHG emissions. The marginal effect of N fertilizer on N loss is strictly positive. Instead, the sign of the marginal effect of N fertilizer on GHG emissions depends on a relative magnitude of the terms capturing atmospheric outflows: soil and plant respiration, carbon in a consumed yield, and denitrification, and inflows: carbon sequestration by the photosynthesis (which also has an indirect decreasing effect on denitrification). If the decreasing effect of N fertilization on GHG emissions dominates the increasing effect, the optimal level of N fertilization is higher in the case where atmospheric externalities are considered, compared to the situation where those are ignored. The second term of Eq. (A3) captures the discounted marginal effect of N fertilizer on N carryover, and the third term captures the discounted marginal effect of N fertilizer on C carryover. The effect of N fertilizer on C transition is indirect as it occurs through an increased yield residue.

Condition for optimal manure rate is given by

$$\boldsymbol{\varphi_t}\ (\text{interior}): p^y y_{\delta,t}\delta_{\varphi,t} + \beta\lambda^n_{t+1}\left(\vartheta^n_{\varphi,t} + \vartheta^n_{e^N,t}\left(e^N_{\varphi,t} + e^N_{y,t}y_{\delta,t}\delta_{\varphi,t}\right) + \vartheta^n_{\xi,t}\left(\xi_{\varphi,t} + \xi_{y,t}y_{\delta,t}\delta_{\varphi,t} + \xi_{\eta,t}\eta_{\varphi,t}\right) + \vartheta^n_{y,t}y_{\delta,t}\delta_{\varphi,t}\right) + \beta\lambda^c_{t+1}\left(\vartheta^c_{\varphi,t} + \vartheta^c_{e^c,t}e^c_{\varphi,t} + \vartheta^c_{R^s,t}R^s_{\varphi,t} + \vartheta^c_{\delta,t}\delta_{\varphi,t}\right)$$
$$= \tau^M + p^M + \mu^N\left(e^N_{\varphi,t} + e^N_{y,t}y_{\delta,t}\delta_{\varphi,t}\right) + \mu^c e^c_{\varphi,t} + \mu^{\text{GHG}}\left(R^s_{\varphi,t} + R^p_{y,t}y_{\delta,t}\delta_{\varphi,t} + C^y_{y,t}y_{\delta,t}\delta_{\varphi,t} + \epsilon\left(\xi_{\varphi,t} + \xi_{y,t}y_{\delta,t}\delta_{\varphi,t} + \xi_{\eta,t}\eta_{\varphi,t}\right) - \delta_{\varphi,t}\right), \tag{A4}$$

where $p^y y_{\delta,t}\delta_{\varphi,t}$ is the marginal value product of manure, and $\tau^M + p^M$ is the marginal cost of manure (i.e. a spreading cost and a market price). Environmental damages of manure consists of three terms: C loss to waterbodies, N loss to waterbodies, and GHG emissions. Also the effect of a manure application on GHG emissions can be positive or negative, depending on the relative magnitude of the terms capturing atmospheric outflows and inflows. Compared to N fertilizer, manure rate has an increasing effect on both N and C losses to water ecosystems. The second termis the discounted marginal effect of manure on N carryover, and the third termis the corresponding effect on C carryover. Manure application has a direct positive effect on both soil N and C stocks.

Optimal condition for an annual soil N stock is

$$\boldsymbol{n_t}(\text{interior}): p^y y_{\delta,t}\delta_{n,t} + \beta\lambda^n_{t+1}\left(1 + \vartheta^n_{n,t} + \vartheta^n_{e^N,t}\left(e^N_{n,t} + e^N_{y,t}y_{\delta,t}\delta_{n,t}\right) + \vartheta^n_{\xi,t}\left(\xi_{y,t}y_{\delta,t}\delta_{n,t} + \xi_{\eta,t}\eta_{n,t}\right) + \vartheta^n_{y,t}y_{\delta,t}\delta_{n,t}\right) + \beta\lambda^c_{t+1}\left(\vartheta^c_{n,t} + \vartheta^c_{\delta,t}\delta_{n,t}\right)$$
$$= \lambda^n_t + \mu^N\left(e^N_{n,t} + e^N_{y,t}y_{\delta,t}\delta_{n,t}\right) + \mu^{\text{GHG}}\left(R^p_{y,t}y_{\delta,t}\delta_{n,t} + C^y_{y,t}y_{\delta,t}\delta_{n,t} + \epsilon\left(\xi_{y,t}y_{\delta,t}\delta_{n,t} + \xi_{\eta,t}\eta_{n,t}\right) - \delta_{n,t}\right), \tag{A5}$$

where $p^y y_{\delta,t}\delta_{n,t}$ is the marginal value product of a soil N stock, the second term is the discounted effect of a soil N stock on N carryover, and the third term is the discounted effect of a soil N stock on C carryover, and λ^n_t is the current shadow price of a soil N stock. Environmental marginal damages of a N stock consist of N loss to waterbodies and GHG emissions.

Optimal condition for annual soil C stock is the following:

$$\boldsymbol{c_t}(\text{interior}): \beta\lambda^c_{t+1}\left(1 + \vartheta^c_{c,t} + \vartheta^c_{e^c,t}e^c_{c,t} + \vartheta^c_{R^s,t}R^s_{c,t}\right) + \beta\lambda^n_{t+1}\vartheta^n_{c,t} = \lambda^c_t + \mu^c e^c_{c,t} + \mu^{\text{GHG}}R^s_{c,t}, \tag{A6}$$

where the first term is the discounted marginal effect of a soil C stock on C carryover, and the second term is the discounted marginal effect of a soil C stock on the N carryover. The first term on the right hand side of Eq. (A6) is the current shadow price of a soil C stock, the second term is the marginal damage of a soil C stock to water ecosystems and the third term is the marginal damage to the atmosphere.

A.2. Steady state

In a steady state optimal management remains the same from one year to another, and time indices can be dropped from the Eqs. (A3)–(A6). We can derive formulas for the steady-state shadow prices of soil N and C stocks from the Eqs. (A5) and (A6):

$$\lambda^n = \frac{(1+\rho)\left[(MVP_n - MD_n) - \phi_c^{-1}MD_c\left(\vartheta^c_n + \vartheta^c_y y_\delta\delta_n\right)\right]}{\phi_n - \phi_c^{-1}\vartheta^n_c\left(\vartheta^c_n + \vartheta^c_y y_\delta\delta_n\right)}, \text{ and} \tag{A7}$$

$$\lambda^c = \frac{(1+\rho)\left[\phi_n^{-1}(MVP_n - MD_n)\vartheta^n_c - MD_c\right]}{\phi_c - \phi_n^{-1}\vartheta^n_c\left(\vartheta^c_n + \vartheta^c_y y_\delta\delta_n\right)}, \tag{A8}$$

where MVP_n is the steady-state marginal value product of a soil N stock, MD_n is the steady-state marginal damage of a soil N stock, MD_c is the steady-state marginal damage of a soil C stock. The term $\phi_c = \rho - (\vartheta c + \vartheta_{e^c}{}^c e^c_c + \vartheta_{R^s}{}^c R^s_c)$ is the effective discount rate (i.e. the discount rate that considers the rate of change of a soil nutrient stock in addition to the time preference of a planner) for a soil C stock, and $\phi_n = \rho - (\vartheta^n_n + \vartheta_{e^N}{}^n e^N_n + \vartheta^n_\xi\xi_n + \vartheta^n_y y_\delta\delta_n)$ is the effective discount rate for a soil N stock. Note that $\phi_c > 0$ and $\phi_n > 0$, because $\vartheta^c_c < 0$, $\vartheta_{e^c}{}^c e^c_c < 0$, and $\vartheta_{R^s}{}^c R^s_c < 0$, as well as $\vartheta^n_n < 0$, $\vartheta_{e^N}{}^n e^N_n < 0$, $\vartheta^n_\xi\xi_n < 0$, and $\vartheta^n_y y_\delta\delta_n < 0$. The discount rate describes the impatience of a planner regarding the use of a resource. If the loss processes are significant, i.e. a carryover decreases fast as the losses increase, a planner becomes more impatient (i.e. an effective discount rate increases) and depletes a soil stock at a higher rate, compared to the case where loss processes are moderate.

Note also that the sign of the denominator in Eq. (A7) depends on the relative magnitude of the terms ϕ_n and $\phi_c^{-1}\vartheta_c^n(\vartheta_n^c + \vartheta_y^c y_\delta \delta_n)$. This is because the product term $\vartheta_c^n(\vartheta_n^c + \vartheta_y^c y_\delta \delta_n)$ is positive. If the denominator is positive, the sign of λ^n depends on the marginal value product of a soil N stock (MVP_n) and the marginal damages of soil N and C stocks (MD_n and MD_c, respectively). Correspondingly, if the denominator in (A((A8)Eqn A8 is positive, the sign of λ^c depends on the same marginal values and damages as the λ^n (i.e. MVP_n, MD_n and MD_c). It is clear that the absolute value of the shadow prices of both soil N and C stocks are increasing functions of the marginal value product of soil N stock and decreasing functions of the water externalities. The effect of the atmospheric externalities depends on whether a soil is a sink or source of GHG emissions. In addition, in the case of private optimum, both externalities are ignored, and the shadow prices reduce to the following forms:

$$\lambda^n = \frac{(1+\rho)MVP_n}{\phi_n - \phi_c^{-1}\vartheta_c^n\left(\vartheta_n^c + \vartheta_y^c y_\delta \delta_n\right)}, \text{ and} \tag{A9}$$

$$\lambda^c = \frac{(1+\rho)\phi_n^{-1}MVP_n\vartheta_c^n}{\phi_c - \phi_n^{-1}\vartheta_c^n\left(\vartheta_n^c + \vartheta_y^c y_\delta \delta_n\right)} \tag{A10}$$

In this case both λ^n and λ^c are positive if the denominators in Eqn A9 and in Eqn A10 are positive.

A.3. Description of the simulations

We run simulations for inorganic nitrogen (N) rates ranging from 0 to 200 kg ha^{-1} yr^{-1} with 25 kg ha^{-1} steps, and for the following solid manure applications: 0, 6138, 18,600, 24,738, 30,876, and 37,200 kg ha^{-1} yr^{-1}. Thus, there are 63 combinations of inorganic and organic fertilizer rates. We repeated the simulations for 9 initial states: initial total amount of N contained in humus in the whole soil profile was changed from −50% to 50% of the baseline level with 25% step (while holding initial organic C/N ratio fixed). Furthermore, initial organic C/N ratios were changed using −50%, −25%, 0%, +50%, and + 100% of the baseline (while holding initial organic N content fixed). The baseline N and C stocks were 1.54e+4 kg ha^{-1} and 1.65e+5 kg ha^{-1} (the baseline C/N ratio was 10.7). We did all the simulations for coarse and clay soils (on average 27% and 60% of the agricultural soils in Finland are clay and coarse soils, respectively (Ylivainio et al. 2015)). The total number of the simulations was 1134.

A.4. Estimated systems of models

Table A1
Estimated system of models for coarse soils (standard errors are in the brackets) (Note that manure was coded as a number between 0 and 2) using four methods: 3SLS, 2SLS, SUR, and OLS.

Coarse soils	3SLS	2SLS	SUR	OLS
Parameter	Estimate	Estimate	Estimate	Estimate
Yield response function (Eq. (1): $y_t = y(\delta_t; temp_t, ppt_t)$)				
Photosynthesis	1.43 (1.14e-02)	1.46 (1.24e-02)	1.43 (1.14e-02)	1.46 (1.24e-02)
Photosynthesis^2	−8.94E-05 (1.28e-06)	−8.88E-05 (1.39e-06)	−8.82E-05 (1.28e-06)	−8.88E-05 (1.39e-06)
Temperature	8.22E+01 (4.98)	8.53E+01 (5.46)	8.19E+01 (4.98)	8.53E+01 (5.46)
Precipitation	−1.21 (3.94e-02)	−1.51 (4.21e-02)	−1.20 (3.94e-02)	−1.51 (4.32e-02)
Adjusted R-Squared	0.61	0.62	0.61	0.96
N carryover equation (Eq. (10): $n_{t+1} = n_t + \vartheta^n(n_t, c_t, N_t, \varphi_t, e_t^N, \xi_t, y_t; temp_t, ppt_t)$)				
Intercept	−2.03E+02 (2.47e+01)	−2.22E+02 (2.47e+01)	−1.783E+02 (2.42e+01)	−2.26E+02 (2.43e+01)
Soil N	−5.44E-03 (7.00e-04)	−4.80E-03 (7.02e-04)	−5.034E-03 (6.70e-04)	−4.65E-03 (6.92e-04)
Soil C	2.08E-04 (3.85e-05)	2.25E-04 (3.86e-05)	1.93E-04 (3.82e-05)	2.18E-04 (3.84e-05)
N fertilizer	5.78E-01 (3.99e-02)	5.31E-01 (4.00e-02)	5.99E-01 (3.95e-02)	5.48E-01 (3.96e-02)
Manure	1.79E+02 (4.11)	1.76E+02 (4.13)	1.83E+02 (4.02)	1.78E+02 (4.04)
log(N loss +1)	−6.35 (3.90)	−9.55 (3.92)	−1.30E+01 (3.57)	−1.15E+01 (3.58)
N yield-uptake	−8.99E-01 (4.07e-02)	−8.39E-01 (4.97e-02)	−9.22E-01 (4.01e-02)	−8.62E-01 (4.03e-02)
Temperature	2.74E+01 (9.12)	3.78E+01 (9.14)	2.11E+01 (9.08)	3.57E+01 (9.11)
Temperature^2	−2.09 (1.00)	−3.44 (1.01)	−1.61 (9.97e-01)	−3.23 (1.00)
Deposition	3.65E+01 (3.17)	3.75E+01 (3.18)	4.02E+01 (3.00)	4.08E+01 (3.01)
Adjusted R-Squared	0.13	0.13	0.13	0.13
C carryover equation (Eq. (11): $c_{t+1} = c_t + \vartheta^c(c_t, n_t, \varphi_t, e_t^c, R_t^s, \delta_t; temp_t, ppt_t)$)				
Soil C	−3.56E-03 (7.41e-05)	−3.53E-03 (7.85e-05)	−3.93E-03 (7.06e-05)	−3.86E-03 (7.54e-05)
Soil N	1.87E-03 (9.55e-04)	4.82E-03 (1.02e-03)	5.15E-04 (9.49e-04)	3.36E-04 (1.02e-03)
Yield	9.96E-02 (2.60e-03)	7.13E-02 (2.69e-03)	9.18E-02 (2.54e-03)	6.92E-02 (2.64e-03)
Manure	1.51E+03 (7.18)	1.51E+03 (7.53)	1.50E+03 (6.82)	1.49E+03 (7.21)
log(C loss)	−1.33E+02 (9.37)	−1.81E+02 (9.73)	−7.54E+02 (8.40)	−1.24E+02 (8.81)
log(soil respiration)	−2.48E+02 (5.39)	−2.32E+02 (5.60)	−2.13E+02 (4.87)	−2.16E+02 (5.12)
Temperature	6.28E+02 (1.38e+01)	6.51E+02 (1.43e+01)	5.17E+02 (1.33e+01)	5.71E+02 (1.38e+01)
Temperature^2	−7.42E+01 (1.52)	−7.69E+01 (1.57)	−6.21E+01 (1.47)	−6.80E+01 (1.53)
Precipitation	1.37 (3.26e-02)	1.51 (3.35e-02)	1.10 (3.14e-02)	1.33 (3.24e-02)
Adjusted R-Squared	0.83	0.83	0.83	0.89
Photosynthesis function (Eq. (2): $\delta_t = \delta(N_t, \varphi_t, n_t; temp_t, ppt_t)$)				
Intercept	−7.26E+03 (1.15e+02)	−5.83E+03 (1.18e+02)	−5.53E+03 (1.10e+02)	−4.96E+03 (1.13e+02)

(continued on next page)

Table A1 (*continued*)

Coarse soils	3SLS	2SLS	SUR	OLS
Parameter	Estimate	Estimate	Estimate	Estimate
N fertilizer^0.5	2.68E+02 (2.68)	2.60E+02 (2.77)	2.24E+02 (2.47)	2.23E+02 (2.55)
Manure^0.5	2.30E+03 (2.74e+01)	2.20E+03 (2.83e+01)	1.74E+03 (2.40e+01)	1.72E+03 (2.48e+01)
Soil N	1.85E-02 (1.80e-03)	1.85E-02 (1.83e-03)	2.27E-02 (1.77e-03)	2.38E-02 (1.81e-03)
Temperature	2.53E+02 (3.19e+01)	4.31E+02 (3.30e+01)	2.74E+02 (3.15e+01)	4.72E+02 (3.25e+01)
Temperature^2	−2.64E+01 (3.49)	−4.69E+01 (3.61)	−3.01E+01 (3.45)	−5.20E+01 (3.56)
Total runoff	−3.09 (6.91e-02)	−2.56 (7.13e-02)	−2.95 (6.70e-02)	−2.56 (6.90e-02)
N fertilizer*manure	−4.13 (1.40e-01)	−3.22 (1.47e-01)	−1.58 (1.28e-01)	−1.01 (1.34e-01)
Deposition^0.5	3.86E+03 (5.48e+01)	2.96E+03 (5.64e+01)	3.29E+03 (5.09e+01)	2.73E+03 (5.23e+01)
Adjusted R-Squared	0.66	0.68	0.68	0.68
N loss function (Eq. (3): $e_t^N = e^N(N_t, \varphi_t, n_t, y_t; temp_t, runoff_t)$)				
Total runoff	2.51E-01 (2.06e-03)	2.34E-01 (2.25e-03)	2.51E-01 (2.06e-03)	2.34E-01 (2.25e-03)
Soil N	2.01E-03 (6.82e-05)	2.18E-03 (7.36e-05)	2.04E-03 (6.79e-05)	2.18E-03 (7.36e-05)
Manure	5.99E+01 (5.26e-01)	5.71E+01 (5.49e-01)	5.93E+01 (5.23e-01)	5.71E+01 (5.49e-01)
N fertilizer	3.59E-01 (5.22e-03)	3.52E-01 (5.54e-03)	3.62E-01 (5.20e-03)	3.52E-01 (5.54e-03)
Yield^2	−1.15E-06 (3.22e-08)	−1.03E-06 (3.47e-08)	−1.15E-06 (3.21e-08)	−1.03E-06 (3.47e-08)
Temperature	−1.85E+01 (2.10e-01)	−1.76E+01 (2.27e-01)	−1.85E+01 (2.09e-01)	−1.76E+01 (2.27e-01)
Adjusted R-Squared	0.61	0.62	0.61	0.89
C loss function (Eq. (4): $e_t^c = e^c(\varphi_t, c_t; runoff_t)$)				
Soil C	6.34E-04 (3.13e-06)	6.22E-04 (3.25e-06)	6.37E-04 (3.13e-06)	6.21E-04 (3.25e-06)
Temperature	−8.50 (1.56e-01)	−9.72 (1.59e-01)	−8.49 (1.56e-01)	−9.75 (1.59e-01)
Manure	1.31E+01 (3.70e-01)	1.13E+01 (3.76e-01)	1.28E+01 (3.69e-01)	1.12E+01 (3.76e-01)
Total runoff	2.49E-01 (1.53e-03)	2.47E-01 (1.56e-03)	2.49E-01 (1.53e-03)	2.47E-01 (1.56e-03)
Yield	−8.43E-03 (1.59e-04)	−8.93E-03 (1.63e-04)	−8.51E-03 (1.58e-04)	−5.86E-03 (1.62e-04)
Adjusted R-Squared	0.81	0.81	0.81	0.96
N2O emissions (Eq. (7): $\xi_t = \xi(\varphi_t, N_t, y_t, \eta_t; temp_t)$)				
Denitrification	9.31E-01 (5.52e-03)	9.61E-01 (6.58e-03)	9.25E-01 (4.87e-03)	9.41E-01 (5.80e-03)
Temperature	1.29 (2.77e-02)	2.25 (3.17e-02)	1.29 (2.75e-02)	2.23 (3.15e-02)
Temperature*precipitation	−2.25E-03 (3.31e-05)	−3.38E-03 (3.69e-05)	−2.26E-03 (3.28e-05)	−3.37E-03 (3.67e-05)
Manure	8.87E-01 (4.41e-02)	8.31E-01 (4.84e-02)	8.24E-01 (4.34e-02)	8.89E-01 (4.75e-02)
Yield	−1.97E-04 (1.94e-05)	−1.05E-04 (2.10e-05)	−1.81E-04 (1.92e-05)	−9.33E-05 (2.07e-05)
N fertilizer	1.08E-02 (4.89e-04)	9.14E-03 (5.42e-04)	1.07E-02 (4.77e-04)	9.85E-03 (5.30e-04)
Denitrification^2	−1.05E-03 (7.37e-05)	−1.52E-03 (8.82e-05)	−9.22E-04 (6.35e-05)	−1.23E-03 (7.59e-05)
Soil N	9.45E-05 (4.50e-06)	2.02E-05 (4.97e-06)	9.71E-05 (4.47e-06)	2.35E-05 (4.94e-06)
Adjusted R-Squared	0.95	0.96	0.95	0.98
Soil respiration (Eq. (5): $R_t^s = R^s(c_t, \varphi_t, N_t; temp_t, ppt_t)$)				
Intercept	−9.77E+02 (5.06e+01)	−1.52E+03 (6.46e+01)	−9.51E+02 (5.03e+01)	−1.53E+03 (6.44e+01)
Soil C	2.77E-03 (1.42e-04)	4.16E-03 (1.81e-04)	2.91E-03 (1.41e-04)	4.14E-03 (1.81e-04)
Manure	6.94E+03 (1.95e+01)	6.87E+03 (2.09e+01)	6.93E+03 (1.92e+01)	6.86E+03 (2.08e+01)
Temperature*precipitation	5.92E-01 (9.80e-03)	7.11E-01 (1.20e-03)	5.90E-01 (9.77e-03)	7.10E-01 (1.20e-03)
N fertilizer	6.45 (2.08e-01)	6.52 (2.24e-01)	6.53 (2.06e-01)	6.46 (2.23e-01)
Yield	−1.89E-01 (1.02e-02)	−1.98E-01 (1.22e-02)	−1.98E-01 (1.01e-02)	−1.93E-01 (1.20e-02)
Adjusted R-Squared	0.9	0.9	0.9	0.9
Denitrification (Eq. (8): $\eta_t = \eta(N_t, \varphi_t, n_t; temp_t)$)				
Soil N	2.00E-04 (7.27e-06)	2.18E-04 (7.80e-06)	2.29E-04 (7.02e-06)	2.42E-04 (7.52e-06)
Manure^2	2.61 (7.11e-02)	2.48 (8.12e-02)	2.17 (6.45e-02)	2.09 (7.44e-02)
N fertilizer^2	2.75E-04 (6.32e-06)	2.81E-04 (7.32e-06)	2.53E-04 (6.16e-06)	2.59E-04 (7.08e-06)
Manure*N fertilizer	5.18E-02 (1.07e-03)	5.01E-02 (1.30e-03)	5.67E-02 (1.02e-03)	5.51E-02 (1.23e-03)
Adjusted R-Squared	0.63	0.63	0.63	0.84
N yield uptake (in the analytical model this function is captured by the yield in Eq. (10))				
Photosynthesis^2	1.64E-06 (3.62e-08)	1.86E-06 (4.08e-08)	1.66E-06 (3.62e-08)	1.87E-06 (4.08e-08)
N fertilizer	5.20E-01 (6.54e-03)	4.80E-01 (7.29e-03)	5.17E-01 (6.53e-03)	4.80E-01 (7.29e-03)
Manure	3.96E+01 (5.92e-01)	3.73E+01 (6.54e-01)	3.94E+01 (5.92e-01)	3.73E+01 (6.54e-01)
Adjusted R-Squared	0.5	0.5	0.5	0.88
Plant respiration (Eq. (6): $R_t^p = R^p(y_t; temp_t)$)				
Yield	6.62E-03 (6.74e-05)	6.64E-03 (6.92e-05)	6.26E-03 (6.64e-05)	6.41E-03 (6.83e-05)
Temperature	1.48E+01 (3.64e-01)	1.51E+01 (3.88e-01)	1.25E+01 (3.49e-01)	1.28E+01 (3.74e-01)
Temperature^2	−1.51 (3.91e-02)	−1.61 (4.17e-02)	−1.28 (3.77e-02)	−1.38 (4.04e-02)
Precipitation	−2.31E-02 (1.52e-03)	−1.83E-02 (1.62e-03)	−1.23E-02 (1.45e-03)	−7.93E-03 (1.56e-03)
Total runoff	−1.92E-01 (2.04e-03)	−2.16E-01 (2.17e-03)	−1.73E-01 (1.94e-03)	−2.01E-01 (2.08e-03)
Precipitation*total runoff	2.11E-04 (2.72e-06)	2.33E-04 (2.89e-06)	1.83E-04 (2.54e-06)	2.09E-04 (2.73e-06)
Adjusted R-Squared	0.58	0.58	0.58	0.91

Table A2
Estimated system of models for clay soils (standard errors are in the brackets) (Note that manure was coded as a number between 0 and 2), using four methods: 3SLS, 2SLS, SUR, and OLS.

Clay soils	3SLS	2SLS	SUR	OLS
Parameter	Estimate	Estimate	Estimate	Estimate
Yield response function (Eq. (1): $y_t = y(\delta_t; temp_t, ppt_t)$) (note that this model specification differs somewhat from one used in the analytical derivations, which, however, does not affect the results)				
Photosynthesis	7.78e-01 (3.88e-03)	7.74e-01 (4.62e-03)	7.78e-01 (3.88e-03)	7.74e-01 (4.62e-03)
Soil N	3.43e-02 (1.37e-03)	2.58e-02 (1.60e-03)	3.35e-02 (1.37e-03)	2.58e-02 (1.60e-03)
Soil N*manure	−2.75e-02 (7.71e-04)	−2.47e-02 (8.17e-04)	−2.67e-02 (7.68e-04)	−2.47e-02 (8.17e-04)
N fertilizer*soil N	−1.61e-04 (6.94e-06)	−1.24e-04 (8.27e-06)	−1.61e-04 (6.93e-06)	−1.24e-04 (8.27e-06)
Adjusted R-Squared	0.49	0.49	0.49	0.94
N carryover equation (Eq. (10): $n_{t+1} = n_t + \vartheta^n(n_t, c_t, N_t, \varphi_t, e_t^N, \xi_t, y_t; temp_t, ppt_t)$)				
Intercept	−9.89e+01 (7.48)	−1.50e+02 (7.69)	−8.33e+01 (6.89)	−1.38e+02 (7.11)
Soil N	−5.19e-03 (2.69e-04)	−4.54e-03 (2.74e-04)	−5.25e-03 (2.68e-04)	−4.68e-03 (2.73e-04)
Soil C	2.67e-04 (1.62e-05)	2.10e-04 (1.64e-05)	2.79e-04 (1.60e-05)	2.24e-04 (1.63e-05)
N fertilizer^0.5	4.21e+00 (2.37e-01)	5.77e+00 (2.45e-01)	3.72e+00 (2.31e-01)	5.30e+00 (2.39e-01)
Manure	1.23e+02 (1.82e+00)	1.38e+02 (1.87e+00)	1.18e+02 (1.78e+00)	1.34e+02 (1.82e+00)
N loss^2	−7.64e-04 (1.27e-04)	−9.88e-04 (1.31e-04)	−6.24e-04 (1.15e-04)	−9.00e-04 (1.20e-04)
N yield-uptake^2	−6.83e-04 (3.34e-05)	−9.97e-04 (3.47e-05)	−4.82e-04 (3.03e-05)	−8.51e-04 (3.16e-05)
N2O	−2.82e+00 (5.73e-01)	−2.49e+00 (5.97e-01)	−4.19e+00 (5.59e-01)	−3.42e+00 (5.83e-01)
Deposition	1.76e+01 (1.40e+00)	2.59e+01 (1.44e+00)	1.46e+01 (1.25e+00)	2.38e+01 (1.30e+00)
Adjusted R-Squared	0.28	0.28	0.27	0.28
C carryover equation (Eq. (11): $c_{t+1} = c_t + \vartheta^c(c_t, n_t, \varphi_t, e_t^c, R_t^s, \delta_t; temp_t, ppt_t)$)				
Soil C	−3.76e-03 (7.41e-05)	−4.30e-03 (7.41e-05)	−3.95e-03 (8.76e-05)	−4.43e-03 (1.12e-04)
Soil N	1.97e-02 (1.23e-03)	7.21e-03 (1.23e-03)	1.92e-02 (1.23e-03)	6.96e-03 (1.53e-03)
Manure	9.61e+02 (1.00e+01)	8.58e+02 (1.00e+01)	9.51e+02 (9.97)	8.48e+02 (1.12e+01)
Photosynthesis	3.41e-02 (3.11e-03)	4.43e-02 (3.11e-03)	3.89e-02 (3.11e-03)	4.66e-02 (3.83e-03)
log(soil respiration)	−1.59e+02 (6.23e+00)	−1.27e+01 (6.23e+00)	−1.48e+02 (6.20e+00)	−4.26e+00 (8.04e+00)
Temperature	2.93e+02 (1.50e+01)	1.12e+02 (1.50e+01)	2.90e+02 (1.50e+01)	1.13e+02 (1.97e+01)
Temperature^2	−4.52e+01 (1.64e+00)	−2.63e+01 (1.64e+00)	−4.48e+01 (1.65e+00)	−2.64e+01 (2.16e+00)
C loss*total runoff	−3.80e-03 (2.72e-04)	3.81e-04 (3.55e-04)	−2.64e-03 (2.65e-04)	1.09e-03 (3.47e-04)
Precipitation	1.28e+00 (5.16e-02)	3.31e-01 (6.72e-02)	1.12e+00 (5.09e-02)	2.25e+00 (6.61e-02)
Adjusted R-Squared	0.52	0.54	0.53	0.66
Photosynthesis function (Eq. (2): $\delta_t = \delta(N_t, \varphi_t, n_t; temp_t, ppt_t)$)				
N fertilizer^0.5	2.08e+02 (2.91e+00)	2.08e+02 (2.97e+00)	1.97e+02 (2.84e+00)	1.98e+02 (2.90e+00)
Manure^0.5	2.60e+03 (2.81e+01)	2.54e+03 (2.87e+01)	2.54e+03 (2.79e+01)	2.48e+03 (2.84e+01)
Soil N	1.83e-02 (1.95e-03)	2.28e-02 (1.99e-03)	1.90e-02 (1.95e-03)	2.34e-02 (1.99e-03)
Temperature	1.22e+03 (2.26e+01)	1.11e+03 (2.31e+01)	1.27e+03 (2.25e+01)	1.15e+03 (2.29e+01)
Temperature^2	−1.30e+02 (2.70e+00)	−1.14e+02 (2.76e+00)	−1.34e+02 (2.70e+00)	−1.18e+02 (2.74e+00)
Total runoff	−3.55e+00 (7.18e-02)	−3.28e+00 (7.26e-02)	−3.56e+00 (7.18e-02)	−3.30e+00 (7.26e-02)
N fertilizer*manure	−4.76e+00 (1.55e-01)	−4.39e+00 (1.58e-01)	−4.33e+00 (1.52e-01)	−3.99e+00 (1.56e-01)
Adjusted R-Squared	0.56	0.56	0.56	0.96
N loss function (Eq. (3): $e_t^N = e^N(N_t, \varphi_t, n_t, y_t; temp_t, runoff_t)$)				
Total runoff	1.29e-01 (1.44e-03)	1.28e-01 (1.44e-03)	1.29e-01 (1.44e-03)	1.28e-01 (1.55e-03)
Soil N^2	1.85e-08 (2.12e-09)	−6.42e-09 (2.28e-09)	1.85e-08 (2.12e-09)	−6.42e-09 (2.28e-09)
Manure	3.14e+01 (9.98e-01)	1.37e-04 (1.08)	3.13e+01 (9.99e-01)	1.37e+01 (1.08)
N fertilizer^2	7.49e-04 (1.45e-05)	7.35e+01 (1.51e-05)	7.47e-04 (1.45e-05)	7.35e-04 (1.52e-05)
Yield	−3.66e-03 (1.36e-04)	−3.27e-03 (1.43e-04)	−3.66e-03 (1.36e-04)	−3.27e-03 (1.43e-04)
Temperature	−5.35e+00 (1.49e-01)	−4.11e+00 (1.60e-01)	−5.34e+00 (1.49e-01)	−4.11e+00 (1.60e-01)
Manure*Soil N	2.16e-04 (6.14e-05)	1.28e-03 (6.69e-05)	2.16e-04 (6.14e-05)	1.28e-03 (6.69e-05)
Adjusted R-Squared	0.56	0.57	0.56	0.86
C loss function (Eq. (4): $e_t^c = e^c(\varphi_t, c_t; runoff_t)$)				
Soil C	4.16e-04 (2.05e-06)	3.94e-04 (2.16e-06)	4.15e-04 (2.04e-06)	3.94e-04 (2.16e-06)
Temperature	−7.29 (1.16e-01)	−7.89 (1.24e-01)	−7.32 (1.16e-01)	−7.89 (1.24e-01)
Manure	4.35 (2.41e-01)	2.08 (2.44e-01)	4.39 (2.41e-01)	2.08 (2.44e-01)
Total runoff	1.79e-01 (1.12e-03)	1.78e-01 (1.19e-03)	1.79e-01 (1.12e-03)	1.78e-01 (1.19e-03)
Yield	−4.70e-03 (9.83e-05)	−2.44e-03 (1.04e-04)	−4.69e-03 (9.83e-05)	−2.44e-03 (1.04e-04)
Adjusted R-Squared	0.79	0.8	0.79	0.79
N2O emissions (Eq. (7): $\xi_t = \xi(\varphi_t, N_t, y_t, \eta_t; temp_t)$)				
Denitrification	1.01e-01 (6.37e-04)	9.80e-02 (6.37e-04)	1.01e-01 (6.37e-04)	9.80e-02 (6.75e-04)
Manure	1.69e-01 (1.74e-02)	1.77e-01 (1.74e-02)	1.69e-01 (1.74e-02)	1.77e-01 (1.80e-02)
Yield	−6.76e-05 (5.09e-06)	−4.19e-05 (5.09e-06)	−6.76e-05 (5.09e-06)	−4.19e-05 (5.23e-06)
N fertilizer	1.27e-03 (1.55e-04)	1.27e-03 (1.55e-04)	1.22e-03 (1.55e-04)	1.27e-03 (1.61e-04)
Soil N^2	1.08e-10 (5.42e-11)	−8.96e-11 (5.48e-11)	9.98e-11 (5.42e-11)	−8.96e-10 (5.48e-11)
Adjusted R-Squared	0.8	0.8	0.8	0.9
Soil respiration (Eq. (5): $R_t^s = R^s(c_t, \varphi_t, N_t; temp_t, ppt_t)$)				
Soil C	6.83e-04 (2.97e-04)	1.16e-03 (3.21e-04)	1.56e-04 (2.95e-04)	4.62e-04 (3.17e-04)
Manure^2	1.67e+03 (2.28e+01)	1.56e+03 (2.49e+01)	1.58e+03 (2.21e+01)	1.47e+03 (2.4e+01)
Temperature*precipitation	7.37e-01 (1.14e-02)	6.97e-01 (1.21e-02)	7.52e-01 (1.13e-02)	7.155e-01 (1.20e-02)
N fertilizer	4.97e+00 (2.30e-01)	5.60e+00 (2.39e-01)	5.13e+00 (2.30e-01)	5.74e+00 (2.39e-01)

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Table A2 (*continued*)

Clay soils	3SLS	2SLS	SUR	OLS
Parameter	Estimate	Estimate	Estimate	Estimate
Soil C*manure	6.28e-03 (2.65e-04)	7.32e-03 (2.92e-04)	7.10e-03 (2.59e-04)	8.27e-03 (2.85e-04)
Adjusted R-Squared	0.66	0.66	0.66	0.92
Denitrification (Eq. (8): $\eta_t = z(N_t, \varphi_t, n_t; temp_t)$)				
Soil N	1.53e-04 (9.32e-06)	1.93e-04 (9.88e-06)	1.67e-04 (9.27e-06)	2.05e-04 (9.81e-06)
Manureˆ2	5.54e+00 (9.26e-02)	5.24e+00 (1.00e-01)	5.34e+00 (9.09e-02)	5.05e+00 (9.84e-02)
N fertilizerˆ2	3.65e-04 (8.90e-06)	3.32e-04 (9.72e-06)	3.56e-04 (8.85e-06)	3.21e-04 (9.66e-06)
Manure*N fertilizer	7.52e-02 (1.52e-03)	8.02e-02 (1.69e-03)	7.72e-02 (1.51e-03)	8.26e-02 (1.68e-03)
Adjusted R-Squared	0.7	0.7	0.7	0.87
N yield uptake (in the analytical model this function is captured by the yield in Eq. (10))				
Yield	2.41e-02 (2.40e-04)	1.81e-02 (2.72e-04)	2.44e-02 (2.39e-04)	1.83e-02 (2.71e-04)
N fertilizerˆ2	2.61e-03 (4.06e-05)	3.17e-03 (4.73e-05)	2.59e-03 (4.06e-05)	3.17e-03 (4.73e-05)
Manureˆ2	3.70e+01 (4.45e-01)	4.43e+01 (4.69e-01)	3.61e+01 (4.40e-01)	4.37e+01 (4.66e-01)
Adjusted R-Squared	0.36	0.37	0.35	0.86
Plant respiration (Eq. (6): $R_t^p = R^p(y_t; temp_t)$)				
Yield	4.45e-03 (5.64e-05)	4.34e-03 (5.86e-05)	4.36e-03 (5.62e-05)	4.29e-03 (5.84e-05)
Temperature	6.29e+00 (3.40e-01)	5.96e+00 (3.55e-01)	5.36e+00 (3.31e-01)	4.97e+00 (3.46e-01)
Temperatureˆ2	−6.62e-01 (3.65e-02)	−6.90e-01 (3.81e-02)	−5.72e-01 (3.57e-02)	−5.90e-01 (3.73e-02)
Precipitation	2.16e-02 (1.39e-03)	3.20e-02 (1.45e-03)	2.57e-02 (1.35e-03)	3.58e-02 (1.42e-03)
Total runoff	−1.16e-01 (1.98e-03)	−1.38e-01 (2.06e-03)	−1.09e-01 (1.92e-03)	−1.32e-01 (2.00e-03)
Precipitation*total runoff	9.52e-05 (2.38e-06)	1.08e-04 (2.48e-06)	8.52e-05 (2.27e-06)	9.88e-05 (2.38e-06)
Manure	1.58e+00 (1.21e-01)	1.79e+00 (1.22e-01)	1.54e+00 (1.21e-01)	1.77e+00 (1.22e-01)
Adjusted R-Squared	0.49	0.5	0.49	0.92

Table A3
The correlations of the residuals for the coarse soil models.

3SLS											
	eq1	eq2	eq3	eq4	eq5	eq6	eq7	eq8	eq9	eq10	eq11
eq1	1.000	−0.066	−0.224	0.077	0.022	0.244	0.116	−0.066	−0.019	−0.418	−0.073
eq2	−0.066	1.000	0.097	−0.015	−0.026	−0.030	0.001	0.017	0.023	0.024	0.028
eq3	−0.224	0.097	1.000	0.072	−0.101	−0.075	−0.142	0.045	−0.040	0.202	0.271
eq4	0.077	−0.015	0.072	1.000	0.032	0.026	0.338	−0.208	−0.087	−0.121	0.062
eq5	0.022	−0.026	−0.101	0.032	1.000	0.159	0.168	−0.083	0.327	0.034	−0.274
eq6	0.244	−0.030	−0.075	0.026	0.159	1.000	0.072	0.085	0.153	−0.115	0.038
eq7	0.116	0.001	−0.142	0.338	0.168	0.072	1.000	−0.572	−0.106	−0.285	−0.237
eq8	−0.066	0.017	0.045	−0.208	−0.083	0.085	−0.572	1.000	0.433	0.360	0.185
eq9	−0.019	0.023	−0.040	−0.087	0.327	0.153	−0.106	0.433	1.000	0.185	−0.124
eq10	−0.418	0.024	0.202	−0.121	0.034	−0.115	−0.285	0.360	0.185	1.000	−0.006
eq11	−0.073	0.028	0.271	0.062	−0.274	0.038	−0.237	0.185	−0.124	−0.006	1.000
2SLS											
	eq1	eq2	eq3	eq4	eq5	eq6	eq7	eq8	eq9	eq10	eq11
eq1	1.000	−0.058	−0.165	0.008	−0.001	0.169	0.058	−0.042	−0.026	−0.396	−0.058
eq2	−0.058	1.000	0.092	−0.015	−0.017	−0.025	0.010	0.005	0.017	0.012	0.028
eq3	−0.165	0.092	1.000	0.119	−0.089	−0.039	−0.118	0.024	−0.039	0.158	0.276
eq4	0.008	−0.015	0.119	1.000	0.010	−0.015	0.243	−0.096	−0.043	−0.106	0.088
eq5	−0.001	−0.017	−0.089	0.010	1.000	0.152	0.155	−0.062	0.321	0.058	−0.247
eq6	0.169	−0.025	−0.039	−0.015	0.152	1.000	0.110	0.040	0.147	−0.075	0.039
eq7	0.058	0.010	−0.118	0.243	0.155	0.110	1.000	−0.456	−0.070	−0.211	−0.189
eq8	−0.042	0.005	0.024	−0.096	−0.062	0.040	−0.456	1.000	0.431	0.335	0.099
eq9	−0.026	0.017	−0.039	−0.043	0.321	0.147	−0.070	0.431	1.000	0.176	−0.145
eq10	−0.396	0.012	0.158	−0.106	0.058	−0.075	−0.211	0.335	0.176	1.000	−0.032
eq11	−0.058	0.028	0.276	0.088	−0.247	0.039	−0.189	0.099	−0.145	−0.032	1.000
SUR											
	eq1	eq2	eq3	eq4	eq5	eq6	eq7	eq8	eq9	eq10	eq11
eq1	1.000	−0.066	−0.223	0.057	0.023	0.246	0.113	−0.063	−0.018	−0.417	−0.062
eq2	−0.066	1.000	0.098	−0.013	−0.017	−0.026	0.006	0.010	0.024	0.025	0.024
eq3	−0.223	0.098	1.000	0.103	−0.121	−0.066	−0.151	0.066	−0.034	0.200	0.280
eq4	0.057	−0.013	0.103	1.000	0.035	-0.074	0.290	−0.136	−0.046	−0.099	0.083
eq5	-0.023	−0.017	−0.121	0.035	1.000	0.160	0.166	−0.082	0.326	0.033	−0.283
eq6	0.246	−0.026	−0.066	0.074	0.160	1.000	0.073	0.087	0.156	−0.116	0.064

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Table A3 (*continued*)

SUR	eq1	eq2	eq3	eq4	eq5	eq6	eq7	eq8	eq9	eq10	eq11
eq7	0.113	0.006	−0.151	0.290	0.166	0.073	1.000	−0.573	−0.114	−0.288	−0.241
eq8	−0.063	0.010	0.066	−0.136	−0.082	0.087	−0.573	1.000	0.433	0.357	0.213
eq9	−0.018	0.024	−0.034	−0.046	0.326	0.156	−0.114	0.433	1.000	0.183	−0.102
eq10	−0.417	0.025	0.200	−0.099	0.033	−0.116	−0.288	0.357	0.183	1.000	−0.004
eq11	−0.062	0.024	0.280	0.083	−0.283	0.064	−0.241	0.213	−0.102	−0.004	1.000

Table A4
The correlations of the residuals for the clay soil models.

3SLS	eq1	eq2	eq3	eq4	eq5	eq6	eq7	eq8	eq9	eq10	eq11
eq1	1.000	0.146	0.457	−0.040	0.070	0.349	0.095	−0.217	−0.247	−0.557	−0.210
eq2	0.146	1.000	0.297	0.007	−0.016	0.043	−0.025	0.087	−0.031	−0.151	0.015
eq3	0.457	0.297	1.000	−0.074	−0.103	0.220	−0.123	0.133	−0.256	−0.521	−0.011
eq4	−0.040	0.007	−0.074	1.000	0.066	0.233	0.020	−0.056	0.103	0.062	0.172
eq5	0.070	−0.016	−0.103	0.066	1.000	0.251	0.216	−0.205	0.167	0.148	−0.121
eq6	0.349	0.043	0.220	0.233	0.251	1.000	−0.017	−0.120	0.073	−0.041	0.001
eq7	0.095	−0.025	−0.123	0.020	0.216	−0.017	1.000	−0.113	−0.119	−0.096	−0.081
eq8	−0.217	0.087	0.133	−0.056	−0.205	−0.120	−0.113	1.000	0.214	0.155	0.239
eq9	−0.247	−0.031	−0.256	0.103	0.167	0.073	−0.119	0.214	1.000	0.342	0.157
eq10	−0.557	−0.151	−0.521	0.062	0.148	−0.041	−0.096	0.155	0.342	1.000	0.088
eq11	−0.210	0.015	−0.011	0.172	−0.121	0.001	−0.081	0.239	0.157	0.088	1.000

2SLS	eq1	eq2	eq3	eq4	eq5	eq6	eq7	eq8	eq9	eq10	eq11
eq1	1.000	0.142	0.459	−0.044	0.043	0.242	0.061	−0.216	−0.250	−0.491	−0.167
eq2	0.142	1.000	0.262	0.021	0.006	0.025	−0.017	0.070	−0.008	−0.096	0.009
eq3	0.459	0.262	1.000	−0.051	−0.085	0.153	−0.152	0.088	−0.241	−0.474	−0.022
eq4	−0.044	0.021	−0.051	1.000	0.054	0.123	0.007	−0.028	0.109	0.097	0.145
eq5	0.043	0.006	−0.085	0.054	1.000	0.210	0.219	−0.160	0.144	0.151	−0.100
eq6	0.242	0.025	0.153	0.123	0.210	1.000	−0.028	−0.121	0.101	0.032	0.028
eq7	0.061	−0.017	−0.152	0.007	0.219	−0.028	1.000	−0.108	−0.082	−0.079	−0.049
eq8	−0.216	0.070	0.088	−0.028	−0.160	−0.121	−0.108	1.000	0.202	0.169	0.176
eq9	−0.250	−0.008	−0.241	0.109	0.144	0.101	−0.082	0.202	1.000	0.323	0.131
eq10	−0.491	−0.096	−0.474	0.097	0.151	0.032	−0.079	0.169	0.323	1.000	0.050
eq11	−0.167	0.009	−0.022	0.145	−0.100	0.028	−0.049	0.176	0.131	0.050	1.000

SUR	eq1	eq2	eq3	eq4	eq5	eq6	eq7	eq8	eq9	eq10	eq11
eq1	1.000	0.150	0.457	−0.040	0.069	0.348	0.095	−0.218	−0.248	−0.561	−0.207
eq2	0.150	1.000	0.315	−0.006	−0.028	0.037	−0.018	0.091	−0.040	−0.187	0.013
eq3	0.457	0.315	1.000	−0.074	−0.107	0.202	−0.123	0.131	−0.257	−0.525	−0.010
eq4	−0.040	−0.006	−0.074	1.000	0.064	0.232	0.016	−0.048	0.104	0.059	0.184
eq5	0.069	−0.028	−0.107	0.064	1.000	0.251	0.216	−0.204	0.167	0.147	−0.117
eq6	0.348	0.037	0.202	0.232	0.251	1.000	−0.017	−0.126	0.074	−0.041	0.015
eq7	0.095	−0.018	−0.123	0.016	0.216	−0.017	1.000	−0.111	−0.121	−0.096	−0.086
eq8	−0.218	0.091	0.131	−0.048	−0.204	−0.126	−0.111	1.000	0.209	0.151	0.237
eq9	−0.248	−0.040	−0.257	0.104	0.167	0.074	−0.121	0.209	1.000	0.341	0.163
eq10	−0.561	−0.187	−0.525	0.059	0.147	−0.041	−0.096	0.151	0.341	1.000	0.096
eq11	−0.207	0.013	−0.010	0.184	−0.117	0.015	−0.086	0.237	0.163	0.096	1.000

A.5. Total cost curves

We obtain total abatement costs for GHG emissions, and N and C losses to waterbodies by increasing gradually the respective marginal damage costs, and calculating the resulting private NPVs, and subtracting those from the private NPV associated with the case where marginal damage costs are zero. Marginal abatement cost curves (MACCs) are obtained by fitting curves to the obtained total costs and differentiating the fitted curves with respect to the level of abatement. Fig. A1 shows the total costs and the fitted curves. Table A5 shows the equations of the fitted total cost curves.

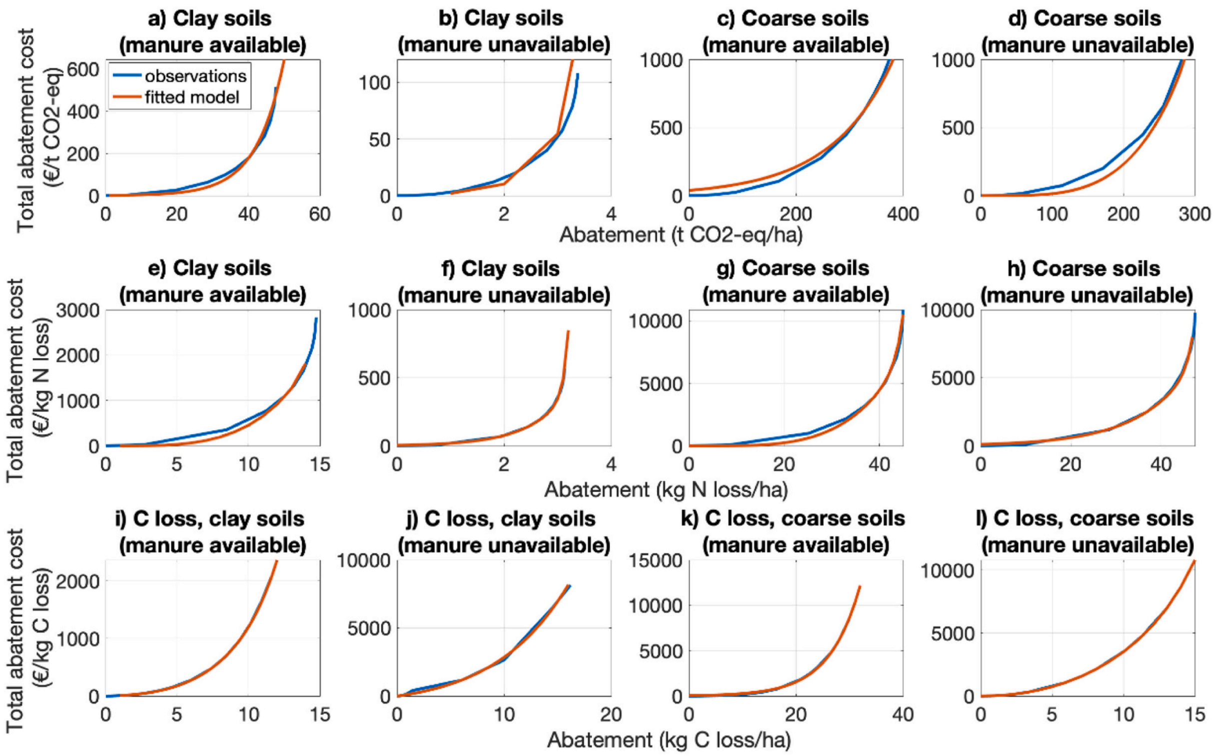

Fig. A1. Total abatement costs of GHG emissions, N loss to waterbodies, and C loss to waterbodies on clay and coarse soils for the cases where both inorganic and organic fertilizers are used, and where only N fertilizer is used.

Table A5
Fitted total cost curves (abat stands for an abatement).

Emissions	Soil texture			
	Clay soils		Coarse soils	
	Manure available	Manure unavailable	Manure available	Manure unavailable
GHG emissions	Cost = 1.03*exp.(0.1288*abat)	Cost = 0.3609*exp.(1.672*abat)	Cost = 4.104e-13*exp.(0.0796*abat) + 38.66*exp.(0.008509*abat)	Cost = 2.033e-11*exp.(0.09816*abat) + 1.237e-07*abat^4.03
N loss	Cost = 8.12e-11*exp.(2.028*abat) + 0.06911*abat^3.818	Cost = 1.987e-14*exp.(b*abat) + c*exp.(d*abat)	Cost = 7.151e-11*exp.(0.7001*abat) + 0.0009879*abat^4.146	Cost = 4.084e-13*exp.(0.7701*abat) + 112.6*exp.(0.08406*abat)
C loss	Cost = 6.237*exp.(0.4476*x) + 2.202*x^2.472	Cost = −2.165e+06*exp.(0.09704*abat) + 2.165e+06*exp.(0.09709*abat)	Cost = 49.92*exp.(0.1718 *abat)	Cost = 6.664*exp.(0.4144*abat) + 20.79*abatˆ2.173

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Appendix B. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.agsy.2020.102985.

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