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Trade-offs in soil fertility management on arable farms

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A B S T R A C T

Crop production and soil fertility management implies a multitude of decisions and activities on crop choice, rotation design and nutrient management. In practice, the choices to be made and the resulting outcomes are subject to a wide range of objectives and constraints. Objectives are economic as well as environmental, for instance sequestering carbon in agricultural soils or reducing nitrogen losses. Constraints originate from biophysical and institutional conditions that may restrict the possibilities for choosing crops or using specific cultivation and fertilization practices. To explore the consequences of management interventions to increase the supply of organic C to the soil on income and N losses, we developed the linear programming model NutMatch. The novelty of the model is the coherent description of mutual interdependencies amongst a broad range of sustainability indicators related to soil fertility management in arable cropping, enabling the quantification of synergies and trade-offs between objectives. NutMatch was applied to four different crop rotations subjected to four fertiliser strategies differing in the use of the organic fertilisers cattle slurry, pig slurry or compost, next to mineral fertiliser. Each combination of rotation and fertiliser strategy contributed differently to financial return, N emissions and organic matter inputs into the soil.

Our model calculations show that, at the rotational level, crop residues, cattle slurry and compost each substantially contributed to SOC accumulation (range 200-450 kg C ha^{-1} yr^{-1}), while contributions of pig slurry and cover crops were small (20-50 kg C ha^{-1} yr^{-1}). The use of compost and pig slurry resulted in increases of 0.61-0.73 and 3.15-3.38 kg N_2O-N per 100 kg extra SOC accumulated, respectively, with the other fertilizers taking an intermediate position. From a GHG emission perspective, the maximum acceptable increase is 0.75 kg N_2O-N per 100 kg extra SOC accumulated, which was only met by compost. Doubling the winter wheat area combined with the cultivation of cover crops to increase SOC accumulation resulted in a net GHG emission benefit, but was associated with a financial trade-off of 2.30-3.30 euro per kg SOC gained.

Our model calculations suggest that trade-offs between C inputs and emissions of greenhouse gases (notably N_2O) or other pollutants (NO_3, NH_3) can be substantial. Due to the many data from a large variety of sources incorporated in the model, the trade-offs are uncertain. Our model-based explorations provide insight in soil carbon sequestration options and their limitations vis-a-vis other objectives.

1. Introduction

The amount and quality of soil organic carbon (SOC) is often used as indicator of soil quality and productivity (Amundson et al., 2015; Powlson et al., 2011a). At the global scale, agricultural soils constitute a large C pool in the form of soil organic matter, and there is thus scope for large amounts of C to be lost or gained from soils as a consequence of farming practices (Smith, 2012). Management of arable land through repeated disturbance has turned many arable soils into C sources (Lal et al., 2007), contributing to climate change. Increased awareness of climate change and concerns about soil quality decline have led to increased emphasis on sequestering C in the soil: increasing SOC content is often seen as a desirable objective. Strategies to increase SOC content in crop rotations include cover crop cultivation (Poeplau & Don, 2015), nutrient and crop residue management (Lehtinen et al., 2014; Blair et al., 2006), application of manures and composts (Triberti et al., 2008) and no- or minimum-till farming (e.g. Powlson et al., 2014), with the latter a much debated option. While there are many advantages to increasing soil C stocks, there are a number of issues associated with soil C sequestration which make it a risky climate change mitigation option (Smith, 2012; Powlson et al., 2011b). These issues include the finiteness of the amount of C that can be stored in the soil, the reversibility of C sequestration, and a number of 'leakage' and pollution swapping issues. Despite these limitations, soil C sequestration may have a role in reducing the short term atmospheric CO_2

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concentration, thus buying time to develop longer term emission reduction solutions across all sectors of the economy (Smith, 2012).

Besides of CO_2, agricultural soils are also a source of nitrous oxide (N_2O) (Reay et al., 2012). Nitrous oxide is emitted largely during microbially governed transformation processes of soil-N, derived from crop residues and the application of inorganic and organic fertilizers. In developed, high-input agriculture, the N taken up by crops is typically no more than 60 per cent of that applied (Lassaletta et al., 2014; Janzen et al., 2003). The remainder is lost in various forms, with major environmental impacts such as high nitrate levels in drinking water aquifers and eutrophication of surface waters. Reducing N input is an important strategy in ameliorating the effect of arable crop production on N_2O emission and water quality (Hillier et al., 2009), but may have a penalty in terms of (economic) productivity.

Crop production and related soil management implies a multitude of decisions and activities on soil tillage, crop choice, rotation design, nutrient supply, water supply and crop protection. Within each of these management categories, many options are usually available to farmers, and the choices to be made and the resulting outcomes are subject to a wide range of economic and environmental objectives and constraints (Hengsdijk & van Ittersum, 2002; Groot et al., 2012). Finding ways to maintain farm profitability while reducing undesirable emissions or maintaining carbon stocks is complicated by interactions and feedbacks among agricultural practices. For example, the addition of organic materials to the soil, such as animal manures and composts, potentially increases SOC content, and increased yields resulting from fertiliser application can result in increased crop residue additions to the soil organic matter pool (Blair et al., 2006). However, large additions of mineral and organic fertilisers to the soil may enhance nitrogen losses to water and atmosphere or result in phosphorus saturation of agricultural soils. These and other examples illustrate the existence of conflicts or trade-offs between objectives of soil management (Powlson et al., 2011a). Given the complexity of interactions and conflicts, the selection of management options that result in a maximization of the net benefits from agriculture is no easy task.

Hengsdijk & van Ittersum (2003) presented an agro-ecological modelling approach, converting information on specific aims for agricultural systems into targeted identification and quantification of land use systems and their management options. In the approach, process based knowledge and empirical data regarding agronomic relationships are integrated and synthesised, using a variety of numerical tools, while taking into account available resources and prevailing land-related objectives (ten Berge et al., 2000). Typically, such 'engineered' land use systems are expressed in terms of inputs and outputs, including production, environmental and socio-economic characteristics. At relatively low costs and risks, agro-ecological modelling of land use systems enables the systematic exploration of land use options at farm and regional scales that are difficult to monitor otherwise. Such model-based land use systems hence provide a framework to disentangle the complex relationships between agricultural production, environment and economy and to explicate synergies and trade-offs between different goal variables, contributing to informed decision making with respect to future land use or research priorities. Currently, many descriptions and applications of such model studies exist (Janssen & van Ittersum, 2007), but to our knowledge no model is available that provides the required detail in nutrient management at farm level to reveal trade-offs resulting from soil fertility management. The purpose of this paper is to show how the NutMatch model can support multi-criteria decision making in nutrient and soil fertility management. To this end, the model is deployed for ex-ante assessments of choices in soil fertility management in arable farming in the Netherlands, illustrating long term consequences of these choices on farm income, nitrogen losses and the build-up of soil organic matter.

In the next section we present the linear programming (LP) model NutMatch. The novelty of this model is the coherent description of mutual interdependencies amongst a broad range of sustainability indicators related to crop production, soil fertility management, SOC content, N emissions and farm economics, enabling the quantification of synergies and trade-offs between objectives. NutMatch differs from most other modelling efforts related to soil fertility management in that it is a static optimization model that can be used for integrating several sustainability aspects within a whole farming system context. This can be contrasted with dynamic, process-oriented simulation models used for predicting nitrogen and soil fertility dynamics at the plot or higher scales in response to changed climate, management or land use (e.g. Ryals et al., 2015; Lugato et al., 2014; Viaud et al., 2010; Batlle-Aguilar et al., 2010), that lack the capacity to handle a range of objectives simultaneously.

2. Materials and Methods

2.1. Case study

The NutMatch model was applied to arable farming on sandy soils in the Netherlands. Here, arable farming is characterized by high intensity, expressed in the adoption of crop rotations with a large share of high-value crops (potatoes, vegetables) and the use of relatively high levels of external inputs such as pesticides and fertilisers. The use of organic and mineral fertilisers on arable farms is currently ceiled by legally binding maximum nitrogen and phosphorus application standards defined at the crop level (Schröder & Neeteson, 2008). Due to the ample supply of animal slurries in the Netherlands, suppliers pay arable farmers for using animal slurries in crop fertilisation. Therefore, arable farmers tend to import a large part of the maximum allowable phosphorus application (28.4 kg P or 65 kg P_2O_5 ha^{-1} yr^{-1} in 2014) as phosphorus in animal slurries.

Arable farmers are concerned that restrictions on the use of organic and mineral fertilizers will in the long term reduce soil fertility, jeopardizing quality production and economic profits (ten Berge et al., 2010, Reijneveld et al., 2009). While no general decline in soil fertility has been documented for the Netherlands as yet (Reijneveld et al., 2009), it is recognized that past management has resulted in high levels of soil fertility indicators such as SOC content, soil N supply and phosphorus status. Although Nitrates Directive regulations have resulted in reduced fertiliser inputs over time, nitrate leaching from agriculture still poses a serious problem, with nitrate concentrations in shallow groundwater under arable farming among the highest in the country. About seventy percent of arable farms on sandy soils have until now not been able to meet the EU target for shallow groundwater of 11.3 mg NO_3-N per litre (RIVM, 2012). Since 2000, average nitrate concentrations on arable farms in the sandy region (covering the southeast, east and northeast of the Netherlands, i.e. about half the agricultural area in the Netherlands), have varied from about 13.6 mg per litre to 19.2 mg NO_3-N per litre, with no clear trend.

2.2. Rotation and nutrient management variants

Based on the above regional context, arable cropping systems in NutMatch were described according to so-called design criteria (Hengsdijk & van Ittersum, 2003), each represented by a number of variants. Our design criteria were the composition of the rotation, nutrient sources used, and the level of N supply to individual crops relative to their full N demand at economically optimal N rate (Table 1). We defined four crop rotations differing in the relative areas of winter wheat, ware potato, sugar beet and silage maize, and differing in the use or not of a cover crop after winter wheat. The four rotations obviously have different nutrient requirements, financial returns and inputs of crop residues into the soil, affecting SOC and soil N dynamics. Of the crops considered, ware potato is the single most important crop in farm economic terms (see Supplementary Material). The crop with the largest crop residue input is winter wheat, with straw assumed to be incorporated into the soil. Cover crops after winter wheat bring

Table 1
Design criteria and their variants to characterise arable cropping systems on sandy soil in the Netherlands.

Design Criteria	Variants
Rotation	Four rotation types: 1. ROT1: 25% of each of the crops winter wheat, potato, maize and sugar beet; 2. ROT1+: as ROT1, with yellow mustard as cover crop after wheat; 3. ROT2: 50% winter wheat, 25% of each of the crops potato and sugar beet; 4. ROT2+: as ROT2, with yellow mustard as cover crop after wheat.
Nutrient source	Four variants: 1. Mineral fertilisers only; 2. Mineral fertilisers and cattle slurry; 3. Mineral fertilisers and pig slurry; 4. Mineral fertilisers and compost.
Relative N supply	Eight N rates per crop, ranging from economically optimal N rate to zero N rate.

additional organic inputs into the soil, but their cultivation comes with extra costs (Section 2.3).

Each crop could be grown at one of eight different levels of relative N supply, ranging from economically optimal N rate to zero rate. N supply is the sum of annual N inputs (expressed as fertiliser equivalents) and N mineralised annually from organic soil pools (see Section 2.4). Nitrogen application rate affects crop yield and N losses to groundwater (NO_3-N) and atmosphere (NH_3-N and N_2O-N).

Besides crop rotation and relative N supply, different nutrient sources were used (Table 1). In one variant, nutrient supply to crops was based on mineral fertilisers (NPK) only. Three other options combined mineral fertilisers with either cattle slurry, pig slurry or compost. The organic manures could be combined with mineral fertilisers in any ratio, depending on objectives and restrictions (e.g. on maximum N emissions) imposed in NutMatch.

2.3. Modelling framework

NutMatch optimises soil fertility management in arable crop rotations. The model is formulated as an optimisation matrix, consisting of rows and columns. The rows in this matrix are linear mathematical equations representing objective functions and restrictions with respect to crop production, supply of nutrients to crops from fertilisers and the soil, nutrient balances, build-up of soil carbon and N emissions. The columns are the decision variables in these equations, representing cropping activities (unit: ha) with different variants of fertiliser use (mineral and organic) and different levels of N supply (Table 1), mineral and organic fertiliser activities (Mg yr^{-1}) and a number of related decision variables required to fully formulate the LP problem to be optimised (see Supplementary Material for a list of all decision variables). An activity is a coherent set of operations with corresponding inputs and outputs, resulting in, e.g., the delivery of marketable crop products, maintenance of soil fertility and N emissions. Each possible model outcome is a ‘farm configuration’ that represents one of the four defined rotations, fertilised according to one of the four nutrient source variants, with each crop grown at its own level of relative N supply. Each activity is characterised by a set of pre-defined coefficients that express the activity's claim on available resources (e.g. land, inputs required) and its contributions to defined objective variables (e.g. income, build-up of soil carbon) and other desired or undesired outputs (e.g. crop yields, crop residue returns, N emissions). The sum of activities' claims is subject to a series of constraints, which represent the restrictions imposed (e.g. on N emissions) and the minimum or maximum amount of a certain resource that can be used (e.g. maximum 170 kg N ha^{-1} yr^{-1} in animal manures, as stipulated by the Nitrates Directive). Coefficients were calculated based on various literature and secondary data sources (see below and Supplementary Material). They should incorporate all relevant non-linearities that are so common in agriculture. For example, the non-linear response of crop yield and N leaching to N supply is embedded in the respective coefficient values for each of the eight defined levels of N supply. All coefficients are quantified at the activity level and are defined prior to NutMatch optimisations. Hence, they are presumed independent of resulting farm configurations. NutMatch combines all activities and constraints with underlying coefficient tables in one optimisation matrix, which can then be optimised for one objective function. Objective functions used in this paper are:

(1) financial return (maximise), calculated as revenues from sold crop products minus fertilisation costs and other variable costs attributed to crop cultivation,
(2) nitrate leaching to groundwater (minimise),
(3) ammonia emission (minimise),
(4) nitrous oxide emission (minimise),
(5) build-up of soil carbon (maximise).

In our optimizations with NutMatch, focus was not on maximising or minimising single objectives in isolation. Instead, we considered two or three objectives simultaneously, with one objective being maximised or minimised, with the other one(s) serving as restriction. E.g. “maximise financial return, while nitrate leaching to groundwater may not exceed 80 kg N”. Separate optimizations were done for each of the 16 variants of four rotations and four nutrient source variants (Table 1), which were hence imposed onto the model before optimization. Other than the rotation and nutrient source variants, the eight relative N supply variants were not imposed onto the model. Instead, N supply to single crops was an outcome of model optimizations, with the selected level per crop and the share of N from organic fertilisers in the total N supply depending on restrictions imposed in the model. Further details on the set up of our optimizations are given in Section 2.6. The definition of objective functions and their quantification are further detailed in following sections.

NutMatch incorporates costs and revenues associated with cultivation of crops and selling of crop products. To exclude economy of scale effects, labour costs and fixed costs of land, buildings and machinery were not considered. Costs attributed to crop cultivation included costs for seeds, fertilisers, crop protection products and fuel. With the exception of fertiliser costs, these costs were assumed to be independent of soil fertility management, and hence were assigned to crops as fixed cost terms. Costs and 5-year average farm product prices were based on default values for the sandy regions in the Netherlands (KWIN-AGV, 2012). Revenues were calculated as the product of crop yields as defined by relative N supply and product price. Costs of fertilisers refer to the purchase of mineral fertilisers (calcium ammonium nitrate, potassium chloride and triple superphosphate) and organic fertilisers (cattle slurry, pig slurry or compost). Depending on regional and seasonal supply and demand, prices of organic fertilisers vary widely across Europe. We fixed manure prices on the basis of their NPK content and application costs, with the assumption that arable farmers pay a share of 75% of both application costs and the value of NPK in mineral fertiliser form (pers. comm. H. Steinmann, Univ. of Göttingen). Hence, nutrients in organic fertilisers were set to be only slightly cheaper than in mineral fertilisers.

The nutrient (NPK) requirement at rotation level is the sum of requirements by selected cropping activities, given their relative N supply levels that best match the imposed conditions. P and K requirements were set according to Dutch fertilizer recommendations (de Haan & van Geel, 2013) and corresponded to N-defined yields. All nutrient requirements are to be met by the respective nutrient supplies from manures (short and long term), mineral fertilisers, crop residues and from the soil itself. Each crop could be grown at only one N supply level, but levels were allowed to differ between crops in the rotation. As stipulated by the Nitrates Directive, N input in the form of animal manures could not exceed 170 kg N ha^{-1} yr^{-1}. This restriction was not applied to compost-N. Agronomic P surplus at rotation level was restricted to zero, with a small margin (for numerical purposes) of ± 2.2 kg P ha^{-1} yr^{-1}, which is stricter than current legislation in the Netherlands.

2.4. Nitrogen and organic matter dynamics

N response curves for each crop (van Dijk et al., 2007) define the amount of plant available N required to reach a given target yield and corresponding N offtake. No N response curves were used for cover crops. Instead, here we defined one yield level only, with corresponding N requirement based on standard fertiliser recommendation. Van Dijk et al. (2007) based their curves on a large number of trials where yield and N offtake were measured in response to N fertiliser rate. We converted responses to fertiliser-N into responses to total N supply (plant available N), by estimating the sum of soil N supply (mineralisation) and atmospheric N in each of their trials as the observed N offtake in unfertilised plots divided by the apparent fertiliser-N recovery (ANR) at low N rate. Crops in our calculations must respond to total N supply rather than just fertiliser-N rate, because soil N supply (and hence supplemental fertiliser-N required to meet a given target yield) evolves as a function of farm configuration (model outcome) itself. A priori defined responses to just fertiliser-N would therefore not be consistent with solutions found. While NutMatch is not a dynamic simulation model, time can, therefore, not be ignored. Based on the above data set, we set *initial* soil N supply rate to 100 kg N ha^{-1} yr^{-1}, and considered it representative for intensively farmed soils in the case study region.

We distinguished N mineralised from organic materials applied within the first year of application ('first year mineralisation') from N released by the accumulated older organic materials. The 'first year mineralisation' from recent organic amendments was allocated to individual crops, while N mineralised from older materials was averaged over the total farm area. This approach eliminates the need to keep track of soil organic N pools per field, and is consistent with reality where crops and organic inputs usually rotate spatially over the farm area. In other words, soil N supply was calculated at the whole-farm level and assumed equal for all crops in the rotation.

Model outcomes refer to a time horizon of 25 years. This means that the resulting farm configuration represents the optimal solution (to the set of goals and constraints imposed) after maintaining this configuration during 25 years. Long-term effects of organic manures and crop residues are thus accounted for. Annual soil N supply after 25 years consists of N still released from the initial ($t = 0$) soil organic N pool, and N mineralised from organic manures and crop residues accumulated in the soil over the 25-year period since $t = 0$.

The build-up of soil carbon from organic amendments was calculated according to Yang (1996), and Yang & Janssen (1997, 2000), accounting for decomposition using specific decomposition parameters for different organic materials (see Supplementary Material). This monocomponent model is a simplified but practical approach to describe complex dynamic systems. Yang's model was deemed suitable to describe soil carbon accumulation over periods up to several decades, but less so for calculations over centuries (de Willigen et al., 2008). To assess the performance of a farm configuration in building up SOC, we distinguish newly formed SOC (SOC_{new}) from SOC already present at $t = 0$ (SOC_{ini}). SOC_{new} is all C that remains in the soil from organic amendments added during the 25 years since $t = 0$. During this period, SOC_{ini} continues to break down, but its amount and fate depend on preceding land use history, not on farm management as composed by the optimisation model. In presenting model outcomes, we only consider SOC_{new}, expressed as a mean formation rate across the 25 year period (kg C ha^{-1} yr^{-1}).

2.5. Calculation of N losses

Annual N loss from the soil-crop system was calculated at rotational level as annual throughput of mineral N in the soil minus annual N uptake in crops and soil surface ammonia loss. Throughput is the sum of N deposition, mineral N in applied organic and inorganic fertilisers, and total N mineralised from organic inputs and the soil organic N pool. Annual N uptake in crop products and residues was based on van Dijk et al. (2007). Soil surface ammonia loss was calculated via emission factors (de Haan & van Geel, 2013; Huijsmans & Hol, 2012; de Ruijter et al., 2013) that specify ammonia loss fractions from land application of organic fertilisers and from crop residues.

N not taken up by crops or lost as ammonia is subject to loss processes in the soil, notably denitrification and leaching. Leaching was calculated via a fixed leaching factor: the proportion of non-ammoniacal N loss that is actually leached as nitrate, as derived from long-term monitoring programmes on commercial farms in the region (Schröder et al., 2011; see Supplementary Material). We calculated nitrate concentration in upper groundwater (mg NO_3-N l^{-1}) from the amount of N leached (above), assuming this N is diluted in an average precipitation excess for this region of 346 mm (Schröder et al., 2011). Direct and indirect (off-farm) N_2O emissions from agricultural soils were quantified by emission factors (van der Hoek et al., 2007; see Supplementary Material). Direct N_2O emissions come from crop residues and applied fertilizers, indirect emissions from N first lost from our case study rotations as ammonia or nitrate. All emissions associated with imported mineral and organic fertilisers, hence occurring upstream of the rotations, were not accounted for.

2.6. Set up of model runs

In a first optimisation cycle, trade-off curves were calculated showing how one objective variable is restricted by another one. Trade-off curves were calculated for each combination of four rotation and four nutrient source variants (Table 1), considering financial return vs. SOC_{new}, financial return versus either ammonia, nitrous oxide or nitrate loss and SOC_{new} versus either ammonia, nitrous oxide or nitrate loss. End points of each trade-off curve are defined by maximum values of desirable objective variables (financial return, SOC_{new}), and minimum values of undesirable objective variables (N losses). Intermediate points were calculated by maximising financial return or SOC_{new}, respectively, while stepwise tightening the restriction on ammonia, nitrous oxide and nitrate loss, respectively. For example, the trade-off curves of financial return and N_2O-N loss were calculated in a series of optimizations maximising financial return under the condition that N_2O-N emission per ha should not exceed 6, 5, 4 etc. kg per ha.

An objective variable can also be restricted by two other objective variables simultaneously. This was illustrated in a second series of optimisations, where financial return was maximised while stepwise tightening restrictions on maximum N_2O-N emission and minimum SOC_{new}. ("maximise financial return, while N_2O-N emission should not exceed 6, 5, 4 etc. kg per ha and SOC_{new} should be at least 1000, 900, 800 etc. kg per ha"). Through this procedure, the so-called trade-off surface is established. The trade-off surface is made up of binding solutions only, meaning that each co-ordinate on the trade-off surface is optimal, because none of the objective variables can be improved without sacrificing one of the others, hence without moving to another point on the surface. The outer boundaries of the surface are defined by the best attainable values of each single objective under stepwise tightened restrictions for the other objectives. Trade-off surfaces were also calculated for each of the four rotation and four nutrient source variants.

3. Results

In this section, we focus on results for rotations ROT1 and ROT2+. Designs of ROT1+ and ROT2 represented the stepwise transition from ROT1 to ROT2+, and their results are therefore intermediate of the results for the two extreme rotations.

3.1. Trade-off curves

3.1.1. Financial return versus SOC_{new}

Maximum financial return varied from 970 to 1400 euro per ha (Fig. 1). Maximum values for SOC_{new} ranged from 325 to 880 kg C per ha per year (Fig. 1), with rotation and nutrient source having major effects. Compared to using mineral fertilisers only, the use of cattle slurry and compost in both rotations added approximately 240 and 390 kg SOC_{new} per ha per year, respectively, while pig slurry added only 26 kg per ha per year (Fig. 1). Given that the use of organic fertilisers was limited by restrictions on P surplus (Section 2.3), considerable differences in the contribution of organic fertilisers to soil C inputs are explained by differences in their organic matter contents per kg P (Table 5 in Supplementary Material). Using different nutrient sources within rotation types did not result in drastic changes in financial return, so that organic fertilisers were largely interchangeable from a financial perspective.

Changes in design of ROT2+ compared to ROT1 added about 160 kg SOC_{new} per ha per year, but reduced financial return by about 340 euro per ha, irrespective of nutrient source (Fig. 1). The extra build-up of soil-C was mainly due to the increased winter wheat area, while the cover crops only had a modest effect (see Section 3.2). Income foregone in ROT2+ was 2.30-3.30 euro per kg SOC_{new} gained.

When either financial return or SOC_{new} is maximised, high N levels based on maximum use of organic fertilisers were selected. However, the solutions for maximum financial return and maximum SOC_{new} slightly differed in selected N levels. When maximizing SOC_{new}, all crops were supplied N at the highest defined rate, as this resulted in the highest carbon returns to the soil via crop residues. When maximizing financial return, winter wheat and sugar beet were not supplied N at the highest defined rate, hence carbon returned to the soil in crop residues was slightly below maximum values.

3.1.2. Financial return versus nitrogen losses

When financial return was at (near-)maximum values, N losses were also high (Fig. 2, Table 2). N losses were higher when organic fertilisers were used. Higher nitrate leaching was caused by increased N mineralisation outside the growing season, while higher emissions of nitrous oxide and ammonia were caused by higher N emission factors assumed for organic fertilisers (Supplementary Material). When nitrate leaching is not constrained, the annual soil N supply is highest when compost was used and lowest when only mineral fertilizers were used (Table 2), which was mirrored in, respectively, lowest and highest available N required from fertilisers.

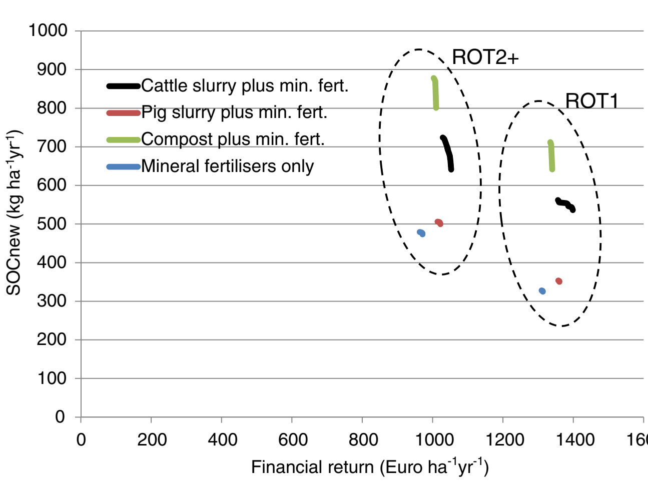

Fig. 1. Trade-off curves of financial return and SOC_{new} in ROT1 and ROT2+ under four nutrient source variants. End points of each curve correspond with maximum financial return and maximum SOC_{new}, respectively.

Nitrate concentrations at maximum financial return ranged between 18 and 21 mg NO_3-N per litre (Fig. 2a, Table 2). Restricting nitrate leaching initially had only a moderate effect on financial return. This is explained by a rather weak response of yield to reduced N in the upper end of the response curves and because it initially sufficed to reduce N levels of economically low-yielding crops only. When further tightening the restriction on nitrate leaching, the response of yield to reduced N becomes steeper and N application rates in high-yielding crops also had to be reduced.

If the norm of the Nitrates Directive was to be met (11.3 mg NO_3-N per litre), N levels were strongly reduced, resulting in reduced yields and reduced financial returns (Fig. 2a, Table 2). Reductions in N available to crops from fertilisers ranged from 22% to 55%, depending on rotation and nutrient source. Meeting the Nitrates Directive norm reduced financial return by 100-150 euro per ha in ROT1 and by 200-300 euro per ha in ROT2+ (Table 2), or 15 and 30 euro per mg NO_3-N reduced, respectively. Relative income loss in ROT2+ (20-25%) was higher than in ROT1 (10%), because in NutMatch only one N level was defined for cover crops, implying that N supply to these crops could not be reduced. Meeting the 11.3 mg standard in ROT2+ therefore required extra reductions of N supply to financially rewarding crops.

By substituting mineral fertilisers for animal slurries, ammonia emissions could be reduced without dramatically affecting financial returns (Fig. 2b). Hence, when restricting ammonia loss to 1 kg NH_3-N per ha in ROT1 and 2.5 kg NH_3-N in ROT2+, fertilisation was entirely based on mineral fertilisers, so that remaining ammonia emission was from crop residues only. In ROT1, ammonia emission could be further reduced than in ROT2+, as in ROT1 ammonia emitting crop residues from cover crops were absent. When unrestricted, ammonia emission reached particularly high values in ROT1 when cattle slurry was applied. This is explained by the spring application of cattle slurry in winter wheat using shallow injection, a technique that is associated with higher ammonia emission than standard injection (Supplementary Material).

Nitrous oxide emissions at maximum financial return ranged from 3.1 to 5.8 kg N_2O-N per ha per year (Fig. 2c). Compared to using mineral fertiliser only, the use of cattle slurry, pig slurry and compost in ROT1 increased nitrous oxide emissions by 2.4, 1.3 and 1.0 kg N_2O-N per ha per year, respectively. Similar to nitrate leaching and for similar reasons, restricting nitrous oxide emission initially had only a moderate effect on financial yield. When restrictions became tighter, the use of organic fertilisers was strongly reduced, explained by their fourfold higher emission factor than that of mineral N fertiliser. Further restricting nitrous oxide emission, organic fertilisers were not applied anymore, so that differences between the nutrient source variants disappeared.

3.1.3. SOC_{new} versus N losses

(Near-)maximum values for SOC_{new} were attained when crops are grown at high N levels and when organic fertilisers were used at maximum levels. High SOC_{new} values were hence associated with high N losses (Fig. 3). The levelling off of the curves can be explained by a higher proportion of applied N that is lost, due to a diminishing N recovery by crops with increasing N input. Relationships between SOC_{new} on the one hand and NH_3 and N_2O emission on the other are more or less linear. Nitrate concentration in groundwater, however, could be considerably reduced without affecting SOC_{new}. When maximizing SOC_{new} while tightening the restriction on nitrate concentration, the use of organic fertilisers remained high at first, while mineral N input was

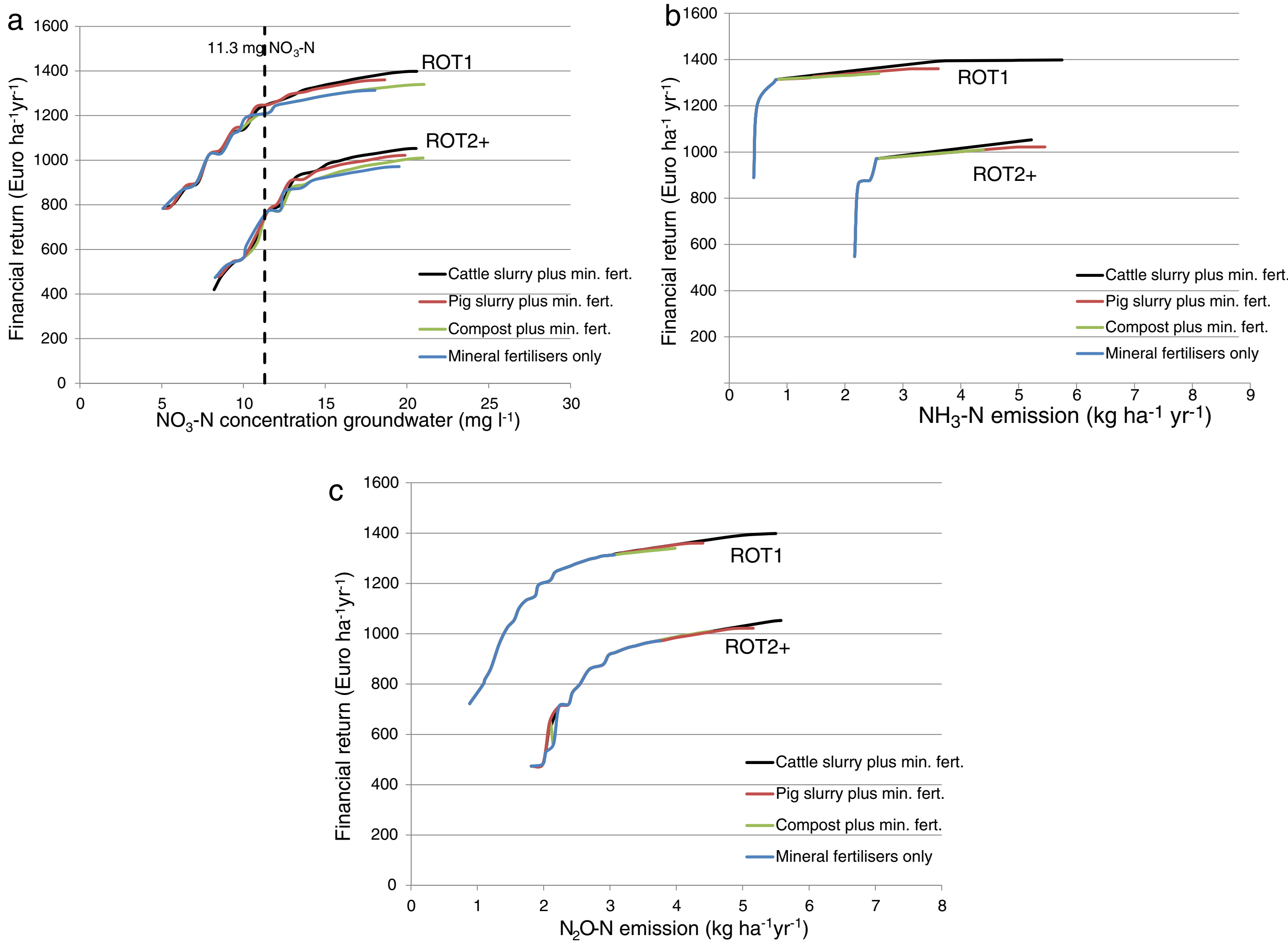

Fig. 2. a-c: Trade-off curves of N losses and financial return in ROT1 and ROT2+ under four nutrient source variants. End points of each curve correspond with minimum N loss and maximum financial return, respectively.

reduced. This strategy slightly affected carbon input via crop residues, but left the more significant supply of carbon via organic fertilisers unaffected. However, when nitrate concentration was restricted to the norm set in the Nitrates Directive, the use of organic fertilisers had to be reduced, minimizing N loss due to N mineralisation outside the growing season. Hence, the use of cattle slurry was reduced by more than half in ROT1 and even further in ROT2+. The use of pig slurry was not reduced as strongly, which is attributable to its high content of readily available mineral N. Due to the severely constrained use of organic fertilisers, maximum SOC_{new} values were considerably reduced when nitrate loss was restricted.

Expressing SOC_{new} and N_2O emission both in CO_2-equivalents (1 kg SOC_{new} = 3.7 kg CO_2-equivalents, 1 kg N_2O-N = 487 kg CO_2-equivalents) it can be calculated, from a GHG emission perspective, that the maximum acceptable increase in N_2O-N emission is 0.75 kg per 100 kg extra SOC_{new}. This break-even line is shown in Fig. 3c, with sections of trade-off curves below the line indicating a net increase in emission of CO_2-equivalents and sections of trade-off curves above the line a net decrease. Only the compost nutrient variant curves were entirely above the break-even line, i.e. SOC_{new} more than compensated N_2O emission in all cases. All other curves were below the break-even line when approaching higher values of SOC_{new}. Based on the slopes of trade-off curves in Fig. 3c, increases in N_2O-N emission ranged from 0.61 to 3.38 kg per 100 kg extra SOC_{new} (Table 3), with compost resulting in the lowest increase and pig slurry in the highest.

3.2. Net GHG emission as affected by rotation and nutrient source variants

Differences in SOC_{new} and N_2O emissions between rotation and nutrient source variants can be used to quantify net GHG emission effects of changing from one variant to another. For example, the net GHG emission effect of introducing fertilised cover crops on half of the cultivated area can be quantified by comparing SOC_{new} and N_2O emission between ROT2 and ROT2+. With compost as organic fertiliser and maximizing financial return, the additional SOC_{new} due to cover crops in ROT2+ was 167 kg CO_2-equivalents per ha per year (Table 4). This amount, however, did not compensate for additional N_2O emission in ROT2+ (239 kg CO_2-equivalents per ha per year), so that there was a net increase of 71 kg CO_2-equivalents per ha per year due to the cultivation of cover crops in ROT2+. Using other organic fertilisers, a similar result was found. The net GHG emission effect of doubling the winter wheat area was quantified by comparing SOC_{new} and N_2O emission between ROT1 and ROT2. With again compost as organic fertiliser, additional SOC_{new} and N_2O emission in ROT2 were 416 and 36 kg CO_2-equivalents per ha per year, respectively (Table 4), hence a net decrease in GHG emission of 380 kg CO_2-equivalents per ha per year. Comparing results for ROT1 and ROT2+, the effect of both interventions combined was a net decrease in GHG emission of 310 kg CO_2-equivalents per ha per year.

Optimizations with and without the application of organic fertilisers allow quantification of the net GHG emission effect resulting from the

Table 2
NPK supply from mineral and organic fertilisers, N losses, NO_3-N concentration in groundwater, soil N supply, financial return and SOC_{new} in ROT1 and ROT2 + under four nutrient source variants, maximising financial return with and without the restriction that the Nitrates Directive norm (11.3 mg NO_3-N per litre) has to be met. All data pertain to an average ha in the rotation and are expressed in kg N/P/K/C ha^{-1} yr^{-1}, except nitrate-N concentration (mg NO_3-N l^{-1}) and financial return (Euro ha^{-1} yr^{-1}).

	ROT1								ROT2 +							
	Cattle slurry		Pig slurry		Compost		Mineral fertiliser		Cattle slurry		Pig slurry		Compost		Mineral fertiliser	
Objective	Max fin.		Max fin.		Max fin.		Max fin.		Max fin.		Max fin,		Max fin.		Max fin.	
Restriction	-	NO_3-N <11.3	-	NO_3-N <11.3	-	NO_3-N <11.3	-	NO_3-N <11.3	-	NO_3-N <11.3	-	NO_3-N <11.3	-	NO_3-N <11.3	-	NO_3-N <11.3
N																
Mineral fertiliser	80	57	108	43	134	126	189	126	104	75	114	49	132	92	199	92
Organic fertiliser	140	71	88	78	97	0	0	0	112	22	91	48	100	0	0	0
Total N	220	128	197	122	231	126	189	126	216	97	205	97	232	92	199	92
Plant available N from fertilisers	171	104	184	110	161	126	189	126	177	89	192	89	161	92	199	92
P																
Mineral fertiliser	2	8	0	0	0	20	21	20	4	13	0	4	0	16	22	16
Organic fertiliser	22	11	25	22	21	0	0	0	18	4	26	14	22	0	0	0
Total P	24	20	25	22	21	20	21	20	22	17	26	18	22	16	22	16
K																
Mineral fertiliser	0	72	105	103	94	158	165	158	0	88	70	82	59	114	132	114
Organic fertiliser	166	84	60	53	71	0	0	0	132	26	62	33	74	0	0	0
Total K	166	156	165	156	165	158	165	158	132	114	132	115	132	114	132	114
N losses																
NO_3-N	71	39	65	37	73	39	62	39	71	39	69	39	73	38	68	38
NO_3-N concentration	20.6	11.3	18.7	10.7	21.1	11.3	18.1	11.3	20.6	11.3	19.9	11.3	21.0	10.9	19.5	10.9
NH_3-N	5.9	2.4	3.7	3.2	2.6	0.7	0.8	0.7	5.3	2.8	5.6	3.8	4.5	2.3	2.6	2.3
N_2O-N direct	3.7	2.1	2.8	2.2	2.2	1.1	1.5	1.1	3.8	1.6	3.4	2.0	2.7	1.3	2.0	1.3
N_2O-N indirect	1.8	1.0	1.6	1.0	1.9	1.0	1.6	1.0	1.8	1.0	1.8	1.0	1.9	1.0	1.7	1.0
Financial return	1398	1243	1360	1239	1340	1208	1313	1208	1053	725	1022	736	1010	711	971	711
Annual soil N supply	96	78	75	70	107	65	69	65	109	80	94	80	127	76	88	76
SOC_{new}	536	405	350	321	642	300	325	300	641	413	500	394	801	380	743	380

use of organic fertilisers. For ROT1, only when using compost, additional SOC_{new} outweighed additional N_2O emissions (Table 4).

3.3. Trade-off surfaces

Trade-off surfaces of ROT1 and ROT2 + and nutrient source variants involving compost and cattle slurry are given in Fig. 4, with x,y-coordinates connected by labelled iso-financial return lines. Trade-off surfaces of the nutrient source variants involving pig slurry and mineral fertilisers only were too compressed to plot in a readable way (Supplementary Material) and are hence not shown. Note that the trade-off surfaces differ from the trade-off curves in Figs. 2c and 3c in that in the surfaces the objective variable 'financial return' is restricted by two other objective variables (N_2O-N emission and SOC_{new}) simultaneously instead of one (Section 2.6). The surfaces show that SOC_{new} and financial return were highest when restrictions on N_2O emission were lenient. Tightening the restriction on N_2O emission reduced both SOC_{new} and financial return. The shapes of the surfaces exemplify that reducing N_2O emission on the one hand and adding carbon to the soil and generating income on the other are conflicting goals. Only the trade-off surfaces involving compost were almost entirely above the break-even line, i.e. SOC_{new} outweighed N_2O emission in all cases when expressed in CO_2-equivalents, indicating a net decrease in GHG emission. Trade-off surfaces of the other nutrient source variants were partly above and partly below the break-even line, the latter indicating a net increase in GHG emissions.

4. Discussion

4.1. Trade-off between SOC accumulation, N losses and GHG emissions

In the introduction section of this paper, we briefly highlighted a number of limitations related to soil C sequestration, including saturation of the carbon sink, the reversibility of C sequestration and 'leakage' issues (Smith, 2012). The results of our study suggest that, in addition, trade-offs between C inputs and N losses can be substantial, and in situations where increases in soil C are associated with increases in emissions of greenhouse gases (notably N_2O) or other pollutants (NO_3, NH_3), these trade-offs should be made explicit. Although these trade-offs have been widely addressed in other scientific publications in a qualitative way as well, there are very few papers that do so in a quantitative manner. In quantitative terms, the most important N loss pathway in our calculations is through leaching, with about 70 kg NO_3-N per ha lost when maximizing income and 40 when the Nitrates Directive norm has to be met (Table 2). Compared to leaching loss, losses of N_2O and NH_3 are relatively small (< 6 kg N per ha in all cases).

Based on modelling, Ryals et al. (2015) quantified the effects of different compost amendment scenarios to grassland on both soil C storage and GHG emissions. They found that increased GHG emissions, particularly direct soil N_2O emissions and indirect N_2O emissions through NO_3-N leaching, partially offset C sequestration benefits of compost additions. However, all modelled scenarios resulted in a net GHG sink in the soil that persisted for several decades, indicating that compost additions to grassland have potential to contribute to climate change mitigation. In our study, we found a similar result for compost additions to arable land. Based on a literature review and modelling, Conant et al. (2005) quantified the effects of a broader set of grassland management options on C sequestration and GHG emissions. Their results showed that changes in soil C and N stocks due to changed grassland management were tightly linked, i.e. in most cases either both increased or both decreased. The study further demonstrated that even when improved grassland management practices result in considerable rates of C and N sequestration, changes in N_2O fluxes can offset a substantial portion of C sequestration gains. We are not aware of similar studies quantifying trade-offs between maximizing C sequestration and minimizing N_2O and other N emissions in an arable context. The lack of

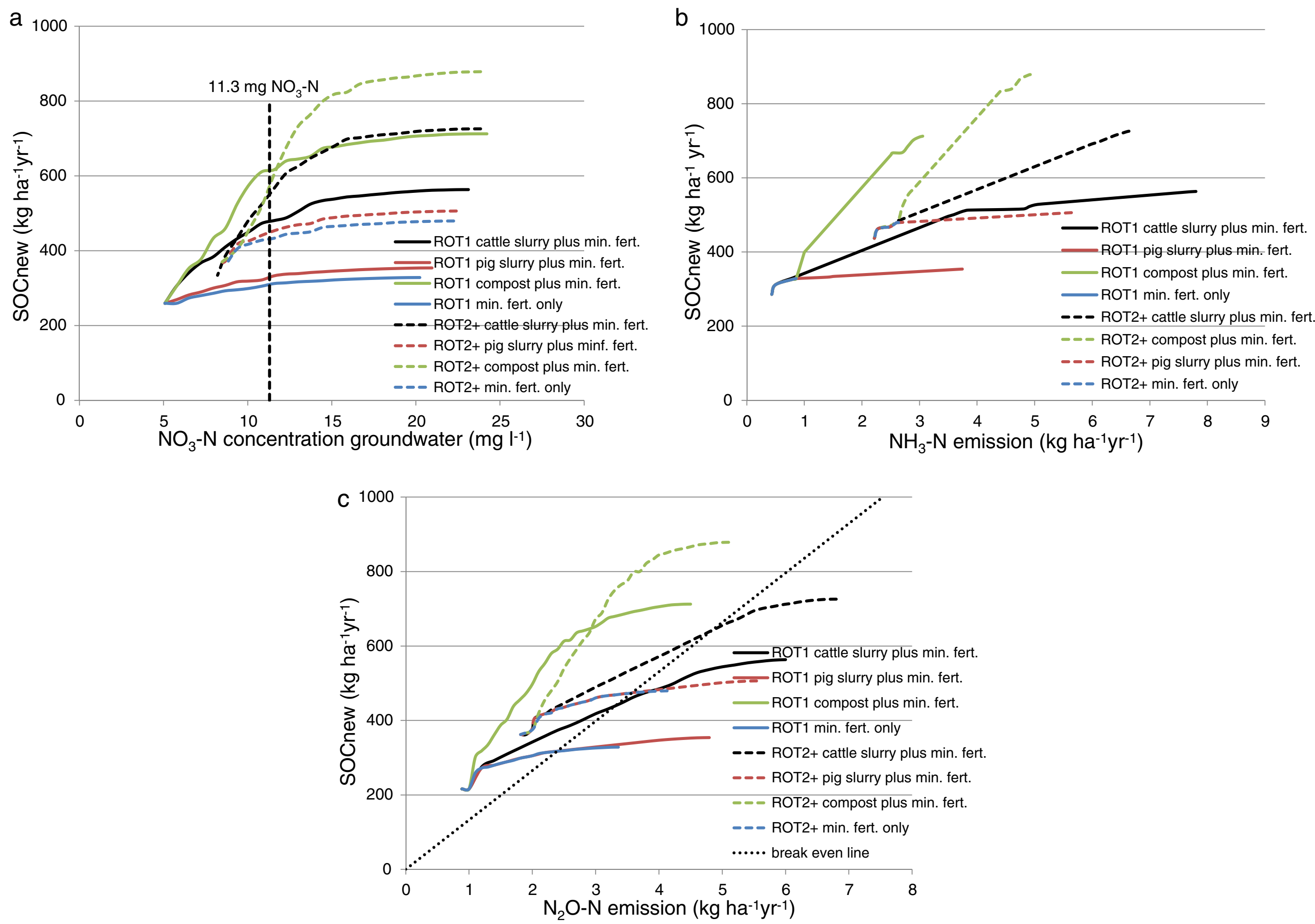

Fig. 3. a-c: Trade-off curves of N losses and SOC_{new} in ROT1 and ROT2+ under four nutrient source variants. End points of each curve correspond with minimum N loss and maximum SOC_{new}, respectively. The dotted line in curve (c) is, from a GHG emission perspective, the break-even line between the build-up of soil C and N_2O-N emission, with sections of trade-off curves below the line indicating a net increase in emission of CO_2-equivalents and sections of trade-off curves above the line a net decrease.

more quantitative studies in this field is probably the result of major knowledge gaps. According to Paustian et al. (2016) the implementation of soil-based GHG mitigation activities is still at an early stage, with accurately quantifying reductions and emissions remaining a substantial challenge. Referring to N_2O emissions, these authors conclude that high temporal and spatial variability make predictions of changes in N_2O fluxes in response to management surprisingly difficult. Particularly lacking are empirical data for multi-intervention strategies that may interact in unexpected ways.

There currently exists much policy interest in enhancing carbon sequestration in agricultural soils for climate change mitigation and crop production purposes (e.g. the "4‰ Initiative", http://newsroom.unfccc.int/media/408539/4-per-1000-initiative.pdf). One important lesson to be learned from our study is that this *may* be a counterproductive strategy for mitigation purposes, very much depending on the way in which this enhanced carbon sequestration is accomplished. Based on our results, effective strategies in crop rotations are the use of compost in fertilisation and increasing the area of crops that return a large residue to the soil.

Table 3
Increase in N_2O-N emission per 100 kg extra SOC_{new} for each nutrient source variant in ROT1 and ROT2+ (kg N_2O-N per 100 kg SOC_{new}).

Nutrient source variant/rotation	ROT1	ROT2+
Cattle slurry	1.47	1.32
Pig slurry	3.38	3.15
Compost	0.73	0.61
Mineral fertiliser only	2.51	2.33

Apart from the above questions concerning the net benefits in terms of GHG emissions, there remain good reasons for adding organic materials to soils, for instance to maintain organic matter levels in the soil and for nutrient recycling purposes (Powlson et al., 2011b). This is possibly to result in improved soil functioning and more efficient crop production in terms of input use per unit output, hence potentially yielding indirect climate benefits. A key question then is what is the origin of these organic materials and how much GHG emissions have occurred to produce these materials. Whilst the application of animal slurries and other organic fertilisers on arable farms results in substantial savings in mineral fertiliser imports on these farms, the production of these organic fertilisers also generated GHG emissions and a range of other environmental impacts. Savings in mineral fertilisers on arable farms through imported organic fertilisers are thus based on pollution swapping. A more complete account of climate effects of management interventions on arable farms can be obtained in NutMatch by factoring in upstream GHG emissions of all external inputs and GHG emissions associated with fuel use required for crop cultivation, including cover crops. In such an approach, linear programming is combined with life cycle analysis (LCA) frameworks. A modelling approach combining optimization and LCA has been developed by Glithero et al. (2012) to evaluate biofuel feedstock production at farm level.

Doubling the winter wheat area combined with the cultivation of cover crops to increase SOC accumulation resulted in a financial trade-off of 2.30-3.30 euro per kg SOC_{new} gained. This is a much higher price than the price currently prevailing in the European carbon emission

Table 4
Additional SOC_{new} and N_2O emissions resulting from soil fertility management interventions when maximizing financial return compared to a reference.

Management intervention	Reference	Additional SOC_{new}		Additional N_2O-N		Net GHG emission effect
		kg C ha^{-1} yr^{-1}	kg CO_2-eqv ha^{-1} yr^{-1}	kg N_2O-N ha^{-1} yr^{-1}	kg CO_2-eqv ha^{-1} yr^{-1}	kg CO_2-eqv ha^{-1} yr^{-1}
Rotation design						
ROT1+: Cover crop on 25% of area	ROT1	23	84	0.30	144	+60
ROT2+: Cover crop on 50% of area	ROT2	46	167	0.49	239	+71
ROT2: Doubling of winter wheat area	ROT1	114	416	0.07	36	-380
Nutrient source						
ROT1: Cattle slurry	Mineral fertilisers only	211	773	2.43	1185	+412
ROT1: Pig slurry	Mineral fertilisers only	25	93	1.33	650	+556
ROT1: Compost	Mineral fertilisers only	317	1159	0.92	446	-713

trading system (about 0.01 euro kg^{-1} C), suggesting that this management intervention is far from competitive. Using different nutrient sources within rotation types had little effect on financial return, so that, within rotation types, there was no trade-off between gains in SOC_{new} and financial return (Fig. 1). The lack of a trade-off is the consequence of the method used to calculate prices of organic fertilisers (Section 2.3), which were slightly cheaper than mineral fertilisers. In order to be competitive with mineral fertilisers, market prices of organic fertilisers may in practice be considerably lower than the prices we used, especially in regions with high animal densities. In such regions, animal slurries are considered wastes, and their prices may even be negative, i.e. arable farmers are paid if they use slurries to fertilise their crops. Compared to animal slurries, compost is generally the more expensive fertiliser. If slurries are to be replaced by compost to increase soil-C build-up, the resulting compost-based SOC gain will be associated with a financial trade-off.

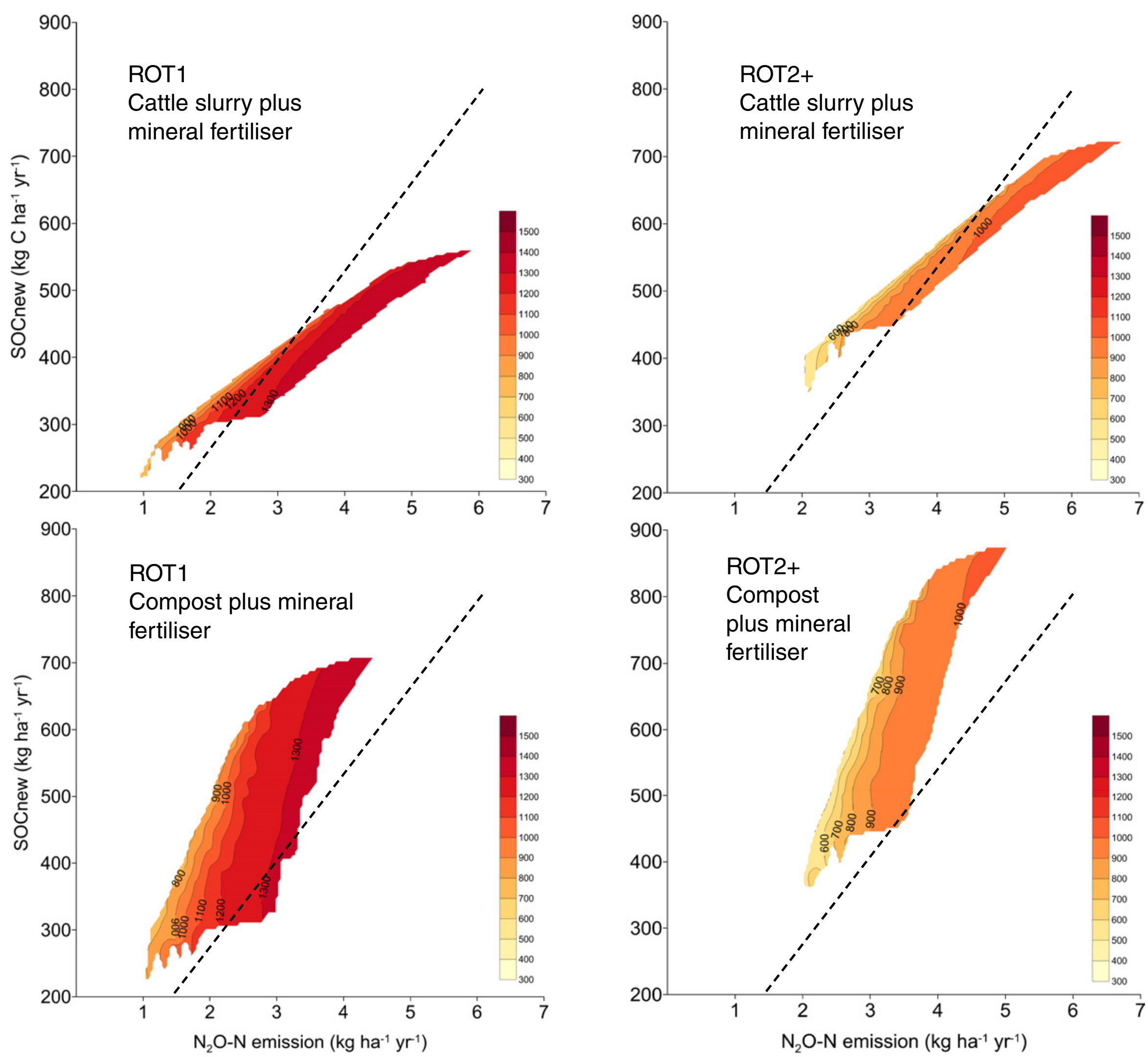

Fig. 4. Trade-off surfaces of ROT1 (left) and ROT2+ (right) under nutrient source variants involving cattle slurry and compost as determined by maximisation of financial return. Red to yellow colours represent a decrease in financial return (Euro ha^{-1} yr^{-1}) while gradually tightening restrictions on maximum N_2O emission and minimum SOC_{new}. The dotted line is, from a GHG emission perspective, the break-even line indicating the minimum value of SOC_{new} required to compensate for N_2O emission.

4.2. Model evaluation

Modelling, in principle, allows a transparent and consistent evaluation of a large number and wide diversity of farming systems. The modelling approach used in this study integrates and synthesises a large number of data from very different sources and as such contributes to bridging the gap between basic and applied sciences and integrating the bio-physical and socio-economic sciences. However, owing to the very different data sources, it is hardly possible to validate the model. Therefore, rather than making predictions, the main purpose of the model was to explicate synergies and trade-offs, contributing to informed decision making and setting the research agenda (van Ittersum et al., 1998).

N_2O emissions were quantified on the basis of simple emission factors for different sources of N, assuming a linear increase with increasing N application of each source. In our optimizations, emissions ranged from 6.8 to 0.9 kg N_2O-N per ha per year, of which 4.6 and 0.4 kg were direct emissions. Emissions of similar magnitude have been measured in experiments (Bell et al., 2015; Ball et al., 2014), but with large variations between sites and years. Results of some experiments suggest that N_2O emissions increase exponentially with increasing N application rates (e.g. Hoben et al., 2011), while other experiments found a less than linear increase (Bell et al., 2015). Yet, other experimental work suggests that the use of cattle slurry, compost and other organic amendments is, at least in some years, associated with increased N_2O emissions compared to a control based on mineral fertilisers (Ball et al., 2014; Jones et al., 2007). For crop residues, Chen et al. (2013) conclude that N_2O emissions are at least similar if not greater than those of mineral fertilisers, suggesting that they could play a role beyond their N content in N_2O production. Enhanced N_2O production from organic amendments could be due to their stimulating effect on microbial respiration, thus enhancing oxygen depletion and promoting denitrification in anaerobic conditions. This would justify the greater emission factors used for organic amendments in our study. Currently our scientific understanding of N_2O emissions is insufficient to quantify N_2O emissions as depending on N source, N level, soil characteristics and local climatic and weather conditions with more precision (Paustian et al., 2016).

Similar to N_2O emissions, model calculations on soil C build-up are also uncertain. De Willigen et al. (2008) showed that different models used to predict the effects of management interventions on organic matter contents result in widely differing outcomes. Differences arise from different conceptual approaches followed in these models and from differences in time scales considered (from decades to centuries). De Willigen et al. (2008) considered the model we used suitable for calculations about expected changes in the short term up to 25 years.

5. Conclusions

The bio-economic farm model NutMatch was successfully used to explore the effects of rotation design and the use of inorganic fertiliser and different types of organic fertilizers on SOC accumulation, N losses and financial returns for an arable farming system in the Netherlands. At rotation level, crop residues, cattle slurry and compost each substantially contribute to SOC accumulation (range 200-450 kg C ha^{-1} yr^{-1}), while contributions of pig slurry and cover crops are small (20-50 kg C ha^{-1} yr^{-1}). The use of compost and pig slurry resulted in increases of 0.61-0.73 and 3.15-3.38 kg N_2O-N per 100 kg extra SOC accumulated, respectively, with the other fertilizers taking an intermediate position. The maximum permissible increase from a GHG emission perspective is 0.75 kg N_2O-N per 100 kg extra SOC accumulated, which was only met by compost. From a greenhouse gas emission perspective, soil-C gained via the use of animal slurries was hence entirely offset by increased N_2O emissions. Doubling the winter wheat area combined with the cultivation of cover crops to increase SOC accumulation resulted in a net GHG emission benefit, but was associated with a financial trade-off of 2.30-3.30 euro per kg SOC_{new} gained. This is a much higher price than the price currently prevailing in the European carbon emission trading system. Trade-offs between C inputs and emissions of greenhouse gases (notably N_2O) or other pollutants (NO_3, NH_3) can be substantial. Identifying these trade-offs is relevant for decision makers, but unfortunately there is still a lack of scientific understanding to accurately quantify them in carbon sequestration studies in agriculture.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.agsy.2016.09.013.

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