Suitability of operational N direct field emissions models to represent contrasting agricultural situations in agricultural LCA: review and prospectus

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Abstract

N biogeochemical flows and associated N losses exceed currently planetary boundaries and represent a major threat for sustainability. Measuring N losses is a resource-intensive endeavour, and not suitable for *ex-ante* assessments, thus modelling is a common approach for estimating N losses associated with agricultural scenarios (systems, practices, situations). The aim of this study is to review some of the N models commonly used for estimating direct field emissions of agricultural systems, and to assess their suitability to agricultural systems featuring organo-mineral or organic fertilisation, non-arable crops, or happening under tropical and sub-tropical conditions.

Simple N models were chosen based on their frequent use in LCA, following a literature review, including ecoinvent v3, Indigo-N v1/v2, AGRIBALYSE v1.2/v1.3, and the Mineral fertiliser equivalents (MFE) calculator. Model sets were contrasted, among them and with the dynamic crop model STICS, regarding their consideration of the biophysical processes determining N losses to the environment from agriculture, namely plant uptake, nitrification, denitrification, NH₃ volatilisation, NO₃ leaching, erosion and run-off, and N₂O emission to air; using four reference agricultural datasets. Models' consideration of management drivers such as crop rotations and the allocation of fertilisers and emissions among crops in a crop rotation, over-fertilisation and fertilisation technique, were also contrasted, as well as their management of the mineralisation of soil organic matter and organic fertilisers, and of drainage regimes. We highlighted the reasons for the differing model outputs.

Among these models, Indigo-N is the most data intensive, and ecoinvent the least. For the four agricultural datasets, the ecoinvent model predicted significantly lower values for NH₃ than the AGRIBALYSE and the STICS models. For N₂O, no significant differences were found among models. For NO₃, the ecoinvent and AGRIBALYSE models predicted significantly higher emissions than the STICS model, regardless of the fertilisation regime. For both emissions, values of Indigo-N were close to those of the STICS model. By analysing the reasons for such differences, and the underlying factors considered by models, a list of recommendations was produced regarding more accurate ways to model N losses (by including, for instance, the main drivers regulating emissions).

Keywords: agriculture, fertilisation, field emissions, nitrogen, organic, tropical

1 Introduction

1.1 Nitrogen modelling in agricultural LCA

Nitrogen is the main limiting factor for terrestrial and aquatic primary production, yet anthropogenic activities have altered the natural N cycle by massively increasing the flow of reactive nitrogen in the biosphere. This biogeochemical flow, as well as the global phosphorus flow and damage to genetic biodiversity, are considered to have exceeded the planetary boundaries (Steffen et al., 2015), with agriculture as a major contributor to the excess (Campbell et al., 2017). The production and use of agricultural fertilisers, together with symbiotic fixations due to human activities, represent an important part of those inputs to the environment. Those are sources of losses to the different environmental compartments, causing a series of impacts, the so called “nitrogen cascade” (Fowler et al., 2013; Galloway et al., 2003). This includes global (e.g. climate change) and local impacts (e.g. aquatic eutrophication, soil degradation). Understanding, quantifying and modelling these losses is thus an increasingly relevant research topic (Gao and Guo, 2014; Oenema et al., 2012; Yang et al., 2017). N losses, conditioned by both pedoclimatic conditions and agricultural strategies (e.g. rotations, fertilisation), predominantly take the form of ammonia (NH₃) volatilisation, nitrate (NO₃) leaching, nitrification-driven nitric oxide (NO_x) emission to air and denitrification-driven nitrous oxide (NO_x and N₂O) emissions to air (EMEP/EEA, 2016). Fertilisation strategies play a key role in N efficiency in agriculture, through unbalanced amounts exceeding crop requirements, time lag between fertilisation and crop uptake, and lack of emission mitigation management for some fertilising strategies, are leading yet manageable drivers of N losses (Padilla et al., 2018). Management of crop cover through rotation, catch crops, or intercropping to insure sufficient N uptake during drainage periods (e.g. winter in Europe, rainy seasons in the tropics) is another major driver (Abdalla et al., 2019).

Life Cycle Assessment (LCA) is widely used to estimate the environmental impacts of agricultural activities. Such assessment is based on life cycle inventories (i.e. resource consumption and emissions associated with a production system) (ISO, 2006), which include direct field emissions associated with fertilisation. Mineralisation, drainage, plant uptake, nitrification and denitrification, and volatilisation should be considered to estimate all N losses in agricultural LCA. Consideration of symbiotic fixation would be a plus, but seldom included. The most common approaches/concepts used to model these mechanisms are listed in the Supplementary Material (Table S1).

Measuring N losses is a resource-intensive endeavour, and not suitable for *ex-ante* assessments, thus modelling is a common approach for estimating N losses associated with agricultural scenarios (systems, practices, situations). Researchers in agricultural subjects use different types of models for estimating N losses according to their scientific questions, their level of familiarity with available models and agricultural systems studied, and their resource constraints (e.g. time, data). For instance, the modelling continuum relevant in LCA context in France is presented in Fig. 1. For example Brilli et al. (2017) reviewed very complex models, the International Soil Modeling Consortium website (<https://soil-modeling.org/resources-links/model-portal>) described a wide variety of agroecosystem models, and Jones et al. (2017) delivered a synthesis of agricultural systems modelling and modelling comparison/improvement initiatives.

Fig. 1. Modelling continuum for estimation of N emissions in the French LCA context

At both ends of the modelling continuum are "simple" models —i.e. empirical equations with or without parameters, usually based on regressions on emissions datasets (Brentrup et al., 2000; Koch and Salou, 2016; Nemecek and Schnetzer, 2012)— and "complex" simulation models —i.e. functional or mechanistic and dynamic biogeochemical/crop models (Addiscott and Wagenet, 1985; Brilli et al., 2017; Manzoni and Porporato, 2009)—. Another key dichotomy used to classify models is their mechanistic (generic) or functional (basic parameters for default conditions, adjusted by factors to other conditions) nature, where a trend towards the latter has been observed in the last decades (Cannavo et al., 2008). Other authors suggest that a mechanistic representation of biophysical processes should lead to a reduced number of analytical generalizable models, as opposite to a large number of situation-specific complex models (Manzoni and Porporato, 2009). It has also been noted that the mathematical features of models across the modelling continuum are more linked with models' fields of application than to their intended spatial and temporal scales of application (Cannavo et al., 2008; Manzoni and Porporato, 2009). A discussion on N models classification criteria, as well as on keywords associated with N models definition and classification, is presented in the Supplementary Material.

Output from mainly simple models requiring few and available input variables can be used to calculate environmental indicators (Buczko and Kuchenbuch, 2010). Such models are designed as "operational" in Bockstaller et al. (2015). Among them, pre-calculated emission factors (EF) for the different N emissions are used, especially by environmental researchers, but these factors are often generic and may not accurately represent the studied situation. EF are usually derived from simple, generic empiric models such as those proposed by the IPCC Guideline for National Greenhouse Gas Inventories (IPCC, 2006) and its 2019 update (Hergoualc'h et al., 2019). Nitrogen balances, among the most used nitrogen indicators (Bockstaller et al., 2015; Rasmussen et al., 2017) whose use is also recommended by the FAO and the OECD, are also suitable to predict

100 N field emissions, yet they have been suggested to be poor predictors of nitrate leaching risk, unless
101 considered at multi-annual temporal scales (Bockstaller et al., 2009). N-balances are computed at different
102 levels of aggregation (i.e. from the farm to the continent), based on empiric equations often calibrated for
103 specific conditions (Roy et al., 2003).

104 When N direct field emissions are the focus of research, LCA practitioners tend to use complex models (e.g.
105 soil-plant dynamic models) that provide detailed information and help interpret LCA results. Yet, such models
106 require relatively high time, data and knowledge, and thus are not widely used for agricultural LCA, but instead,
107 LCA practitioners typically use the most accessible models, in terms of data demand and ease of use, such as
108 those included in pre-defined model sets associated with databases like ecoinvent (Frischknecht et al., 2005),
109 the World Food LCA Database (Nemecek et al., 2014), the Agri-footprint database (Blonk Agri-footprint BV,
110 2014) and the French agricultural LCI database AGRIBALYSE (Colomb et al., 2015; Koch and Salou, 2016). More
111 often than not, LCA practitioners use default pre-calculated emission factors, such as those provided in
112 Albanito et al. (2017), Bouwman and van der Hoek (1997), Bouwman (1996), IFA/FAO (2001) and IPCC
113 (Hergoualc'h et al., 2019; IPCC, 2006). This strategy is in principle aligned with the nature of LCA, which aims at
114 **estimating** impacts, to be analysed in a comparative fashion (Bernstad and la Cour Jansen, 2012; Heijungs,
115 2021; Prado, 2018).

116 On the face of this situation, the aim of this study is to review simple N models used for estimating direct field
117 emissions of agricultural systems, and to assess their suitability to agricultural systems featuring contrasting
118 agricultural situations: organo-mineral or organic fertilisation, non-arable crops (e.g. perennials, vegetable
119 gardening, associated crops), or happening under tropical and sub-tropical conditions. To achieve it we
120 selected a set of models representing a broad gradient of complexity and approaches. Models were described
121 and their outputs associated with a set of example cropping systems compared to provide some information
122 about their relative sensitivity, although it is not possible to conclude on their predictive quality in the absence
123 of a sound set of measured emission data (Bockstaller et al., 2008; Buczko et al., 2010). Comparing outputs of
124 various N models has been recommended in Bockstaller et al. (2008) as a suitable model comparison strategy.
125 Ideally, the models' predictions should be compared to a dataset of field measurements of gaseous N
126 emissions and nitrate leaching, but such datasets are still rather rare.

127 On the base of said comparison, in contrast with key factors determining N emissions from agriculture,
128 recommendations were offered on the minimum requirements a model set would have to fulfil to accurately
129 represent N emissions from contrasting agricultural situations, in a context of LCA applications.

130 **1.2 General limitations of simple N emission models in agriculture**

131 Few models across the modelling continuum are able to model N dynamics across agricultural situations
132 (Cannavo et al., 2008). In the LCA context, N direct emission models commonly used, for instance those simple
133 models used by popular LCI databases such as ecoinvent v3.5 and AGRIBALYSE v1.3 (and earlier versions), are
134 predominantly representative of conventional fertilisation of field crops by synthetic fertilisers. These models
135 are not well adapted to the *modus operandi* of organic fertilisers, or to agricultural systems other than field
136 crops. Moreover, these models often disregard the fertilisation efficiency, that is to say, the effect on emission
137 intensity due to fertiliser inputs beyond the plant needs or after their peak absorption period, as well as the
138 position of a crop of interest within a crop rotation.

139 Various aspects challenge modelling of direct emissions from organic fertilisation in LCA. For instance, the
140 content and quality of nutrients in organic fertilisers is often unknown or very variable, especially the less

141 industrialised ones, such as digestates, composts, separated solid and liquid phases (of slurries, sludge and
142 digestates), and animal effluents. Moreover, organic fertilisers contain both organic and mineralised N, where
143 the organic fraction experience varying rates of mineralisation according with management and pedoclimatic
144 conditions. Several approaches have been developed to model N mineralisation of added organic matter (i.e.
145 agricultural residues, organic fertilisers) and soil organic matter (Benbi and Richter, 2002; Clivot et al., 2017;
146 Kwiatkowska-Malina, 2018; Manzoni and Porporato, 2009), including mineralisation kinetic curves (Doublet et
147 al., 2011; Morvan et al., 2006; Parnaudeau et al., 2006); yet simple N models often include pre-calculated
148 mineralisation factors representative of specific agricultural situations. Simple models and emission factors for
149 direct field emissions predominantly focus on conventional mineral fertilisation of field crops (Meier et al.,
150 2015). Furthermore, most LCA-oriented models focus on single crops rather than on crop rotations, which
151 consequently disregards the abovementioned delayed N (and C) dynamics of organic fertilisation and crop
152 residues left on the field.

153 The specificities of perennial crops are not captured by the most commonly used simple models such as those
154 used by ecoinvent —i.e. SALCA-N (Richner et al., 2014)—, nor by emission factors such as the popular ones
155 proposed by Bouwman and colleagues (Bouwman et al., 2002a, 2002b, 2002c; Stehfest and Bouwman, 2006).
156 These specificities include deep root system expansion, relatively high yields and low nutrient requirements,
157 and much longer rotation times, when compared with arable crops (Bessou et al., 2013; Cerutti et al., 2014). A
158 similar challenge applies to vegetable gardening, featuring much shorter rotation times, and associated crops
159 in the same field, where interactions among crops with different N absorption behaviours are not easy to
160 estimate (Perrin et al., 2014). Associated crops are seldom modelled in LCA, and their direct emissions are
161 complex to estimate, as crops are associated due to reinforcing mechanisms (including N absorption) which are
162 difficult to represent with simple models (Bessou et al., 2013).

163 Simple models and emission factors for direct field emissions are predominantly based on temperate weather
164 conditions. Only the IPCC (IPCC, 2006) and Bouwman and van der Hoek (1997) provide emission factors for
165 tropical and sub-tropical conditions, and for conventional field crops (Bessou et al., 2013). The draining
166 regimes, as well as other pedoclimatic conditions affecting these emissions, are different across agro-climatic
167 zones (van Wart et al., 2013). It has been suggested that IPCC-based results are flawed for N₂O emissions in
168 tropical environments (van Lent et al., 2015).

169 The practice of LCA in developing countries faces additional challenges than in developed ones (Basset-Mens et
170 al., 2018), including the paucity of background inventory data (Perrin et al., 2014), as well as the lack of reliable
171 statistics and adapted direct emission models. Most developing countries feature tropical and sub-tropical
172 conditions.

173 **2 Material and methods**

174 We performed a literature review to select (section 3.1) and investigate the known general limitations of
175 simple N models (section **Erreur ! Source du renvoi introuvable.**), to frame the specific limitations identified
176 during our data-based comparison of selected models (section 3.2), enabling us to provide N modelling
177 recommendations in an LCA context (section 0).

178 **2.1 Criteria for model selection**

179 We established criteria for selecting simple N models, as well as a strategy for an objective and comprehensive
180 comparison.

Simple N models to be tested were chosen based on their applicability in LCA, following a literature review. For instance, French researchers apply LCA to many different agricultural systems, including organic, gardening, perennial, and tropical ones, and have produced several methodological proposals and case studies regarding direct field emissions estimation, be it specific equations or combinations and adaptations of existing models (e.g. Bockstaller and Girardin 2010; Bellon-Maurel et al. 2015; Koch and Salou 2016; Brockmann et al. 2018). We privileged models and model sets used in the European and French research environment (both in European and non-European contexts), as it is one of the more prolific communities in agricultural LCA, as represented for instance in the international LCA Food conferences (<https://www6.inra.fr/lcafoodconferencearchives/>).

We consider emission factors to be, by definition, less representative of a particular agricultural situation (an agricultural system under given pedoclimatic conditions) than the outcomes of a simple model or model set that captures the main determinants of emissions, and whose inputs include parameters that can be calibrated to the given situation or to a similar one. Pre-calculated emission factors, notably those proposed by Bouwman and colleagues (Bouwman and van der Hoek, 1997; Bouwman, 1996; Bouwman et al., 2002c, 2002a, 2002b; L. Bouwman et al., 2013; Lex Bouwman et al., 2013; Stehfest and Bouwman, 2006), are widely used in agricultural LCA. Nonetheless, as suggested in Goglio et al. (2015), simple models should be preferred to fixed emission factors for soil organic C modelling in agricultural LCA, because they allow for a better adaptation to specific conditions. Therefore, generic emission factors were excluded from this review, except for comparison purposes.

2.2 Model (outputs) comparison strategy

Selected models were fed with agricultural and pedo-climatic data from four reference agricultural datasets (see section 2.3), and ran to obtain predictions of N emissions. For model-estimated parameters such as plant N uptake, we always retained the agricultural datasets data. Results of simple models were compared with outputs from the complex dynamic model STICS (Brisson et al., 2003), and the resulting differences analysed. STICS was retained for a direct comparison of simple models with a complex simulation model, which takes into consideration more parameters and mechanisms of emissions than simple models. Moreover, simulation models are better equipped to represent dynamic of emissions, and also the cumulative effect of repeated inputs of organic matter, a key mechanism associated with organic fertilisation (Constantin et al., 2012, 2010). The level of predictive error associated with STICS has been computed for a wide range of systems and pedo-climatic conditions, and determined to be, in decreasing order of relative importance, more prevalent for nitrate leaching, plant biomass, N uptake, and soil water (Coucheney et al., 2015). All retained simple models were implemented in Excel, and fed with the experimental datasets. In the case of multi-annual datasets, we retained average annual values to feed the simple models, while presented STICS results consist of the mean of annual outputs.

Disaggregation of mineral- and organic-fertilised subsystems was possible for all agricultural datasets. A few measurements were available for nitrate losses in the Senegal and Reunion Island sites, which were used as reference points to assess the quality of model predictions, beyond their sensitivity. Measurements were made with lysimetric plates at 40 and 100 cm, respectively.

A 3-way ANOVA and post-hoc Tukey's tests (Piepho, 2018) were firstly performed to assess the effect of three factors and their interactions on N emissions across models: the type of N emission considered (*Emission*), the study site (*Site*) and the fertilisation regime (*Ferti*).

222 Outputs from selected models were then compared, per specific emission, across agricultural datasets (study
223 sites) after normalisation because the emissions simulated by the models were not in the same scales for the
224 different sites and were therefore normalised by the average method (**Erreur ! Source du renvoi introuvable.**)
225 to enable the comparison between models across sites.

$$x' = \frac{x - \text{mean}(x)}{\text{max}(x) - \text{min}(x)} \quad \text{Eq. 1}$$

226 where x' is the normalised emission, x is the output model, $\text{mean}(x)$ is the average value of the different
227 models, $\text{min}(x)$ and $\text{max}(x)$ are the minimum and maximum values of the different models in each site.

228 A second 3-way ANOVA and corresponding post-hoc Tukey's tests were then performed to conduct pairwise
229 comparison across models. The three factors tested were the model itself (*Model*), *Emission* and *Ferti*. The
230 normality of the residues was checked prior to the statistical analyses and p-values were corrected with the
231 Benjamini-Hochberg procedure (<https://stat.ethz.ch/R-manual/R-devel/library/stats/html/p.adjust.html>) to
232 reduce the false discovery rate. The significance threshold was fixed to 5%. Data were processed using the R
233 software (R Core Team, 2020).

234 **2.3 Agricultural datasets for model comparison**

235 We used reference agricultural datasets to test the models, and highlighted the reasons for their differing
236 results. Datasets used for model comparison include one for field crops in France, two for market vegetable
237 gardening in Benin and Senegal, and one for sugarcane in Reunion Island. Such variety permits to capture
238 differing agricultural systems under very contrasting pedo-climatic conditions: temperate, tropical wet
239 (continental and islander) and tropical dry. All datasets feature data for mineral and organic fertilisation. The
240 main pedoclimatic conditions of all four sites are synthesised in Table 1. Common total and mineral N contents
241 for organic fertilisers, as detailed in (Galland et al., 2020), were retained across sites to reduce parameter
242 uncertainty.

243 **Table 1. Pedoclimatic conditions in the sites where the agricultural activities represented by the reference datasets take**
 244 **place**

Key features	Feucherolles, France	South Benin ^a	Sangalkam, Senegal	Reunion Island
Soil texture	Silty	Sandy	Sandy	Clayey
Soil type (FAO/IIASA, 2009)	Luvisol	Arenosol	Arenosol	Nitisol (Ferralsol)
Total topsoil C (%)	1.10	0.70	0.64	1.86
Total topsoil N (%)	0.11	0.05	0.06	0.16
Topsoil clay fraction (%)	16.12	13.00	9.12	43.30
Topsoil pH	7.34	6.02	6.61	6.10
N in Soil Organic Matter (kg N/ha)	4 997	1948	2 689	6 720
Global agro-ecological zone (IIASA/FAO, 2012)	Temperate oceanic forest	Tropical rainforest	Tropical shrubland	Tropical mountain system
Average annual precipitation	583	1101	424	2 665
Average annual temperature (°C)	10.7	25	26.5	25

^a Average of 12 sites (Perrin, 2013)

245 **2.3.1 Temperate field crop: maize in central France**

246 The first dataset used for comparisons comes from the long-term field experiment QualiAgro
 247 (<https://www6.inra.fr/qualiagro/>), corresponding to a field trial located on the Plateau des Alluets le Roi,
 248 Feucherolles, about 20 km west of Paris, France. QualiAgro is part of the SOERE-PRO network (System of
 249 Observations, Experiments and Environmental Research on Organic Residual Waste, <https://si-pro.fr/>).

250 The trial consists of a maize-wheat rotation in the period 1998-2013, fertilised with the mineral fertiliser Urea
 251 Ammonium Nitrate solution (aka “Solution 390”, a liquid mixture of urea and ammonia nitrate, featuring 30% N
 252 in the form of 25% N-NO₃, 25% N-NH₄ and 50% N-NH₃) at two mineral fertilisation rates: minimal and optimal;
 253 and amended with four different organic products (cattle manure, compost of organic waste, compost of
 254 sludge and green waste, and compost of green waste). The experimental setup is described in Cambier et al.
 255 (2014) and Bourdat-Deschamps et al. (2017), and both annual fertiliser inputs (for the optimal mineral fertiliser
 256 rates) and resulting crop yields depicted in Table 2. Plant uptake was estimated between 150 and 188 kg N/ha
 257 (according with the fertilisation scenario, which includes a control mineral fertiliser-only scenario).

258 **Table 2. Fertiliser treatments for the central France maize-wheat dataset (1998-2013)**

Crop	year	Average of 4 organo-mineral treatments			Mineral treatment	
		Organic fertilisers (kg N/ha)	Solution 390 (kg mineral N/ha)	Yield (kg/ha)	Solution 390 (kg mineral N/ha)	Yield (kg/ha)
wheat	1998-1999	-	-	-	-	-
maize	1999-2000	294	79	11 274	79	7 608
wheat	2000-2001	-	102	7 991	51	7 902
maize	2001-2002	335	68	11 497	68	11 076
wheat	2002-2003	-	124	7 196	62	6 501
maize	2003-2004	352	50.8	11 700	50.8	9 647

wheat	2004-2005	-	122	8 642	61.5	7 614
maize	2005-2006	312	51.5	8 234	51.5	6 152
wheat	2006-2007	-	121.1	7 341	60.3	5 698
barley	2007-2007	326	82.3	9 760	82.3	7 426
maize	2008-2009	330	-	8 395	108	8 453
wheat	2009-2010	-	173.5	8 189	110	7 702
maize	2010-2011	315	12.5	6 722	136	6 965
wheat	2011-2012	-	199	6 254	99	5 339
maize	2012-2013	287	-	8 210	110	8 457
Annual average		170	79	8 094	101.8	7 103
No irrigation; fertilisers spread by broadcaster, with soil incorporation; rooting depth: 1.8 m						

2.3.2 Tropical wet garden crop: tomato in south Benin

The second dataset represents off-season (i.e. grown during the dry season, irrigated, featuring low yields) field tomato production in south Benin during 2011-2012, as described in Perrin (2013), who used STICS to estimate N emissions from 12 different systems (Table 3). In average, these tomato systems received 448.7 kg N/ha, 337.4 kg N/ha of which from poultry manure, resulting in a yield of 5 092 kg FM/ha. Plant uptake was estimated at 200 kg N/ha.

For this dataset, as a STICS-based comparison device, emission factors computed with STICS as presented in Perrin (2013) were retained —expressed as a function of total N inputs—: N₂O = 0.6%, NO₃ = 10% (range 0 to 52%) and NH₃ = 10% (range 0 to 37%). NO_x emissions are not originally computed by Indigo-N v2.70 or STICS, but Perrin (2013) estimated an emission factor based on total N inputs (for the specific conditions of her study, to complement her STICS results).

Table 3. Fertiliser treatments of the south Benin off-season tomato dataset (2011-2012)

Fertiliser (kg N/ha)	Average of 8 organo-mineral treatments	Average of 4 mineral treatments	Weighted average of all treatments
Urea (46% N)	27.8	65.3	40.3
NPK (16-16-16)	87.6	38.0	71.1
Dried poultry droppings (0.5% N)	506.1	0	337.4
Total	621.5	103.3	448.7
Yields (t FM/ha)	4.2	6.8	5.1
Irrigation: 500 mm/ha in average; fertilisers spread by hand, without soil incorporation; rooting depth: 0.5 m; FM = fresh mass			

2.3.3 Tropical dry garden crop: market vegetables in north-west Senegal

The third dataset includes historical data (2016-2018) from the experimental site set up in 2016 by the Laboratoire Mixte International Intensification Ecologique des Sols Cultivés en Afrique de l'Ouest (LMI IE SOL), in Sangalkam, near Dakar, Senegal, in the context of the SOERE-PRO network. The area, neighbouring the Niayes coastal strip, is semi-arid.

The experimental design features a total randomised set-up, with 16 m² plots and three replicates per fertilisation treatment (Table 4). Three organic fertilisers are studied, at two applied doses representing 100 and 200% of the recommended dose of mineral fertilisers: poultry litter (210 kg N/ha), sewage sludge (122 kg

281 N/ha), and agricultural digestate (103 kg N/ha). The crops consist of rotations of lettuce-carrot-tomato. The
 282 fertilisation and other agricultural practices are considered as representative of peri-urban market vegetable
 283 gardening in the greater Dakar area. The cumulative plant uptake by this rotation was estimated at 350 kg
 284 N/ha, but 593 kg N/ha were added per year (considering only the mean of all treatments furnishing 100% of
 285 fertiliser needs of the rotation), 197 kg N/ha of which were furnished by mineral fertilisers.

286 **Table 4. Fertiliser treatments of the Sangalkam market garden vegetables dataset (2017-2018)**

Fertiliser (kg N/ha)	Treatment 1 (organo-mineral)	Treatment 2 (organo-mineral)	Treatment 3 (organo-mineral)	Weighted average of all treatments
Urea (46% N)	233.7	153.3	45.5	197.0
Limed sewage sludge (1% N)	262.6			152.0
Digestate of cattle manure (0.5% N)		506.0		172.4
Dried poultry droppings (0.5% N)			157.1	71.3
Total	496.3	659.3	202.6	592.7
Annual yields of the rotation (t FM/ha)	21.3	12.4	18.5	18.7
Irrigation: 1305 mm/ha in average; fertilisers spread by hand, with soil incorporation; rooting depth: 0.5 m; FM = fresh mass				

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288 **2.3.4 Tropical wet field crop: sugarcane in Reunion Island**

289 The fourth dataset includes data (2017-2018) from the experimental site set up in 2014 by the Recycling and
 290 risk research unit of CIRAD in La Mare, near Saint-Denis in Reunion Island, France (20°54'12.2"S, 55°31'46.6"E).
 291 The experimental trial took place in a highly monitored site belonging to the SOERE-PRO network, designed to
 292 investigate the long-term impact of organic fertilisation on the different compartments of the sugarcane
 293 agroecosystem. The trial was planted in March 2014 with one sugarcane variety (R579) and a 1.5 m row-
 294 spacing. The trial was irrigated throughout the crop cycle (29 mm/week) except for the last two months before
 295 harvest. The trial consisted of six treatments, each with a different fertiliser, which were repeated in 5 blocks.
 296 Each plot made up of six sugarcane rows of 28 m, constituting a total plot area of 250 m². The data used in the
 297 present study were obtained from three distinct fertilisation treatments (Table 5) according with the
 298 dominating source of nutrients: urea, sewage sludge and swine slurry (the last two complemented with urea
 299 applications). In average, 152.2 kg N/ha were furnished, 54.7 kg N/ha of which by mineral fertilisers, to satisfy a
 300 plant uptake of 150 kg N/ha, resulting in a yield of 36 t/ha.

301 **Table 5. Fertiliser treatments of the Reunion Island sugarcane dataset (2017-2018)**

Fertiliser (kg N/ha)	Treatment 1 (mineral)	Treatment 2 (organo-mineral)	Treatment 3 (organo-mineral)	Weighted average of all treatments
Urea (46% N)	71.8	47.6	44.6	54.7
Limed sewage sludge (1% N)		24.0		8.0
Swine slurry (0.4% N)			268.5	89.5
Total	71.8	71.6	313.1	152.2
Yields (t FM/ha)	91.0	110.0	94.0	99.3
No irrigation; fertilisers spread by broadcaster, with soil incorporation; rooting depth: 1.0 m; FM = fresh mass				

302

3 Results and discussion

3.1 Selected simple models and their features

Several simple models are contrasted in Table 6: ecoinvent v3 (an international model widely used in LCA), World Food LCA database v3 (a European model, heavily based on ecoinvent), Indigo-N v1/v2 and AGRIBALYSE v1.2/v1.3 (French models used in French LCA research and case studies), Calculateur AzoteViti and Mineral fertiliser equivalents (MFE) calculator (recent French research models), and FAO N-balances (international models with tropical calibration, used in FAO case studies). These models, among others, are used by LCA practitioners to complete their agricultural life cycle inventories. Model sets were contrasted regarding their consideration of the biophysical processes determining N losses to the environment from agriculture, namely plant uptake, NH_3 volatilisation, NO_3 leaching, N transfer by erosion and run-off, N_2O emissions by nitrification ($\text{NH}_4 \rightarrow \text{NO}_3$) or denitrification ($\text{NO}_3 \rightarrow \text{N}_2$). Models' consideration of management drivers such as rotation over-fertilisation and fertilisation technique were also contrasted, as well as regarding their management of the mineralisation of soil organic matter (SOM) and organic matter provided by fertilisers, drainage regimes, and the allocation of fertilisers (and thus of emissions, mainly by leaching) among crops in a crop rotation.

Table 6. Direct emission model sets used in France

	International model sets		French model sets		French research models		Other international approaches	
Features	ecoinvent v3	World Food LCA database v3	Indigo-N v1/v2	AGRIBALYSE v1.2/v1.3	Calculteur AzoteViti	Mineral fertiliser equivalents (MFE) calculator	FAO N-balances (plot/farm scale only)	Pre-calculated emission factors
Source	Nemecek and Schnetzer (2012)	Nemecek et al. (2015)	Bockstaller and Girardin (2010)	Koch and Salou (2015, 2016)	Bellon-Maurel et al. (2015)	Brockmann et al. (2018)	Roy et al. (2003)	Various (e.g. Bouwman et al., 2002c)
Geographical validity	Switzerland, Europe, Global (SQCB)	Global (main food-exporting countries)	France	France, a few tropical	France	Denmark, France, Germany, Netherlands, Poland	Calibrated for Africa, but global applicability	Variable, but mainly global
Crops covered	Field crops	Field crops, grasslands	Field crops, grasslands	Field crops, grasslands, vegetables, rice, fruits	Grape vines	Crop-independent	Field crops	Field crops, other crop types
Types of fertilisers	Mineral, manure, sugarcane vinasse	Mineral	Mineral, certain organic	Mineral, certain organic	Mineral, most organic	Mineral, most organic	Mineral, most organic	Mainly mineral
Timescale	Annual	Annual	Roughly annual ⁱ	Roughly annual ⁱ	Annual	Annual and long-term	Annual	Annual
Physical scale	Plot, farm (AGRAMMON)	Plot	Plot, farm	Plot	Plot	Plot	Plot, farm	Any
N uptake by plants (plant requirements)	Pre-calculated factors based on a combination of STICS (Brisson et al., 2003) and factors from Flisch et al. (2009)	See ecoinvent v3	N uptake coefficients per crop type and sowing date, based on plant needs	N uptake coefficients per crop type (for SQCB only)	Computed from the N needs for grape production, from literature	Not considered	NUTMON model ^h : millet, sorghum, maize, rice, wheat, and other crops	Not considered
N mineralisation of added organic matter	Not explicitly considered	Fixed factors with corrections (EMEP/EEA 2013 model). See NH ₃	Mineralisation factors for harvest residues, minimum value depending on	Research mineralisation kinetic curves used only for allocation of fertilisation	AZOBIL equation (Machet et al., 1990) modified by a monthly soil moisture curve	Plant available nitrogen (PAN) mineralisation factors (WEF, 2005)	Fixed factors of various origins (literature, NUTMON model)	Implicit

Soil mineral N	Not explicitly considered	Not explicitly considered	soil and its increase due to over-fertilisation Minimum value depending on soil and increase due to over-fertilisation	Not explicitly considered	Not explicitly considered	Not explicitly considered	Not explicitly considered	Not explicitly considered
N mineralisation of soil organic matter (SOM)	Fixed factors with correction factors (SALCA-NO3 and SQCB-NO3 models). See NO ₃	See ecoinvent v3	Mineralisation equation (Taureau et al., 1996) ^a	Leaching models retained do not require mineralisation	Implicit in the N balance approach used (Kücke and Kleeberg, 1997)	Deliberately set to 0	Same as previous	Implicit
NH ₃ volatilisation model	AGRAMMON model tier 3 ^b (https://www.agrammon.ch/) for emissions from leaf surface, mineral fertilisers, manure and vinasse. Emission factors for manure management (Menzi et al., 1997).	EMEP/EEA 2013 tier 2 ^c (EMEP/EEA, 2013)	Volatilisation coefficient from literature, per type of fertiliser, the limestone content, the time of year, and the soil tillage	EMEP/EEA 2009 tier 2 (EMEP/EEA, 2009) for organic fertilisers EMEP/CORINAIR 2006 tier 2 (EMEP/CORINAIR, 2006) for mineral fertilisers	Volatilisation coefficients from literature: NH ₄ assumed to follow an exponential decay with half-life of 12 h. Affected by rain.	Same as ecoinvent 3 for organic fertilisers EMEP/EEA 2013 tier 2 for mineral fertilisers	All N gaseous emissions are considered together, as N Empiric equations from the NUTMON model	Emission factors for chemical fertilisers, for developed and developing countries (Bouwman and van der Hoek, 1997)
N ₂ O emission model	IPCC 2006 tier 1 ^g (De Klein et al., 2006; IPCC, 2006), for direct (mineral and organic fertilisers, and crop residues) and indirect emissions (from NO ₃ leached)	See ecoinvent v3	Empiric equation (Bouwman, 1996) with denitrification computed from IPCC 1997: 1.25%	IPCC 2006 tier 1	empiric equation (Bouwman, 1996) with denitrification computed from IPCC 2006: 1%	IPCC 2006 tier 1	See NH3	Factors and empiric equation for N ₂ O and NO (Bouwman et al., 2002a, 2002c; IFA/FAO, 2001) Emission factors for tropical and sub-tropical

NO _x emission model	Fixed factor for NO _x emissions from N ₂ O (from a personal communication)	Fixed factors for mineral and organic fertilisers from EMEP/EEA 2013	excluded	EMEP/EEA 2009 tier 1	excluded	EMEP/EEA 2013 tier 2	See NH ₃	N ₂ O, per continent, country, crop type and fertiliser type (Albanito et al., 2017) See N ₂ O
NO ₃ leaching model	For Europe: SALCA-NO ₃ ^d (Richner et al., 2014). For other countries: SQCB-NO ₃ ^e (Faist Emmenegger et al., 2009), an adaptation of the de Willigen (2000) model (Roy et al., 2003). Drainage not considered by either model.	See ecoinvent v3	Empiric equation explicitly including drainage (Burns, 1976) modified (Laurent and Castillon, 1987), for post-fertilisation, using N absorption curves. COMIFER ^{a,f} (COMIFER, 2001) for winter drainage, based on N-balances.	ARVALIS method (Tailleur et al., 2012) for field crops. DEAC (Cariolle, 2002) for grassland. SQCB-NO ₃ for perennials and vegetables. IPCC 2006 tier 1 (De Klein et al., 2006) for tropical. Only DEAC and ARVALIS method include drainage.	Empiric equation (Burns, 1975). Calculated from N budget after deducting NH ₃ and N ₂ O emissions. Monthly timestep.	Empiric equations from NUTMON model (Roy et al., 2003), distinguishing background nitrate emissions from SOM N. Drainage not considered.	Empiric equations from the NUTMON model.	NO ₃ leaching factors are often computed with dynamic models (e.g. Groenendijk et al. 2005; Kasper et al. 2019) or measurements fitted to regression models (e.g. Vázquez et al. 2005; Bruun et al. 2006). Some models include drainage.
Nitrification (NH ₄ --> NO ₃)	Not considered	Not considered	Not explicitly considered	Not explicitly considered	Not considered	Not explicitly considered	Not explicitly considered	Not explicitly considered
Denitrification (NO ₃ --> N ₂ O)	Implicitly considered (NO _x), not considered (SALCA-NO ₃)	See ecoinvent v3	1.25% of remaining N after volatilisation (IPCC 1997)	Not explicitly considered	1% of remaining N after volatilisation (IPCC 2006) (De Klein et al., 2006)	Not explicitly considered	Not explicitly considered	Not explicitly considered

Mineral fertiliser equivalents	not considered	not considered	MFE coefficients per crop type	not considered	not considered	Calculated with the PAN formula	not considered	N/A
Consideration of over/under fertilisation	not considered, SQCB-NO3 seems to be calibrated for adequate fertilisation	not considered	Calculation of an increase in soil mineral N after harvest due to over-fertilisation following Machet et al. (1997)	not considered	not considered	not considered	not considered	Assumes adequate fertilisation
Required input data	Target crop, N inputs	Target crop, N inputs	Target crop, next crop, specific dates, recommended and actual N inputs	Target crop, N inputs, French region	Monthly climate data, soil data, N inputs	Fertiliser application data	Detailed farm operation data	N/A

^a Current version of the method: COMIFER (2013). ^b AGRAMMON parameters: total ammonia nitrogen (TAN) in fertilisers, emission rates, various corrections factors related with management. ^c EMEP/EEA 2013 tier 2 parameters: TAN in fertilisers, amount of mineral fertilisers, emission factors per soil pH, correction factors (application method, application time and season). ^d SALCA-NO3 parameters: N mineralisation from the SOM per month, N uptake by vegetation (if any) per month, N input from the spreading of fertiliser, soil depth. Assumes a priori a soil with 15% clay and 2% humus, but modifiable. ^e SQCB-NO3 parameters: precipitation and irrigation, clay content, rooting depth, N in fertilisers, N in organic matter, N uptake by plants. ^f COMIFER N-balances consider residues, residues from over-fertilisation, mineralisation of crop residues and humus, mineral and organic N. ^g IPCC 2006 tier 1 parameters: total N in fertilisers, N in crop residues, N from mineralisation of SOM, NH₃ losses, NO₂ losses, NO₃ losses. ^h The NUTMON (Nutrient Monitoring for Tropical Farming Systems) model is not available online, as it has been replaced by the MonQI (Monitoring for Quality Improvement) model (<https://www.monqi.org/>). ⁱ i.e. current and next crops.

319 3.1.1 International models

320 The ecoinvent database v3 (Nemecek and Schnetzer, 2012) retains the AGRAMMON model
321 (<https://www.agrammon.ch/>), calibrated for Swiss conditions, for NH₃ volatilisation of both mineral and
322 organic fertilisers. For NO₃ leaching, ecoinvent v3 retains the SQCB-NO3 model (Faist Emmenegger et al., 2009),
323 which is based on a widely used (including by FAO) regression model by de Willigen (2000), which in turn is
324 based on NUTMON data (Roy et al., 2003). The Nutrient Monitoring for Tropical Farming Systems (NUTMON)
325 model, was calibrated for tropical conditions. It is currently obsolete and has been replaced and extended by
326 the Monitoring for Quality Improvement (MonQI) model (<https://www.monqi.org/>). The models used in
327 ecoinvent v3 are claimed to have global applicability, but the analysis of certain modelling elements suggests it
328 is an exaggerated claim. For instance, very few added organic matter used as fertiliser are represented
329 (manure, sewage sludge and sugarcane vinasse only). Moreover, the model set does not explicitly compute
330 mineral fertiliser equivalents for these organic fertilisers, nor does it consider over-fertilisation. A similar
331 statement can be made on the nitrate leaching model proposed in Brentrup et al. (2000).

332 The World Food LCA database v3 (Nemecek et al., 2015) uses the same approach as ecoinvent v3 for N
333 emissions, except that it retains the EMEP/EEA (2013) tier 2 model (EMEP/EEA, 2013) for NH₃ volatilisation.
334 This guideline/model set proposes volatilisation factors for mineral fertilisers and manure only. No results were
335 computed for this model, as the modelling principles are virtually identical to ecoinvent's. The most recent 3.5
336 version of the database maintains the 3.0 version model selection (Nemecek et al., 2020), but updating
337 EMEP/EEA tier 2 to the 2016 version (EMEP/EEA, 2016).

338 The Agri-footprint database model set was not retained because it systematically and exclusively uses IPCC (De
339 Klein et al., 2006; IPCC, 2006) simple models.

340 The FAO N-balance approach (Roy et al., 2003), at the plot and farm scales, are tailored to tropical conditions,
341 as they heavily rely on the NUTMON model. All gaseous emissions are considered aggregated, and fixed N
342 mineralisation factors considered. No results were computed for this model.

343 3.1.2 French models

344 AGRIBALYSE v1.3 (Koch and Salou, 2016) proposes a combination of models, some of which are tailored to
345 French conditions. The overall modelling strategy is coherent and comprehensive, yet outdated models were
346 retained for NH₃ volatilisation —EMEP/EEA 2009 (EMEP/EEA, 2009) and EMEP/CORINAIR 2006
347 (EMEP/CORINAIR, 2006)—, while various models are used for NO₃ leaching, according to the type of crop,
348 including semi-qualitative ones, namely SQCB-NO3, factors from IPCC 2006 (De Klein et al., 2006, Table 11.3),
349 and the method described in Tailleur et al. (2012).

350 Indigo-N v2, a subset of the Indigo environmental assessment method (Bockstaller et al., 2008; Bockstaller and
351 Girardin, 2010), consists of a combination of simple models for each different N emission, around an annual
352 mass balance of nutrients allocated to a crop location. The model relies on mineralisation factors, correction
353 factors for management, and empiric equations. It is calibrated to field crops and prairies under temperate
354 conditions. It is the only model accounting for the effects of over- and under-fertilisation on nutrient emissions.
355 The main originality of this model is the calculation of NO₃ losses. Without being a complex soil-plant dynamic
356 model, Indigo-N addresses effects of climatic and soil conditions, fertilisation in regard with the plant's needs
357 or some management practices (e.g. soil management between crops, the following crop). Here the 2.70
358 version of Indigo-N was implemented in Excel for the comparison.

359 The approach and associated Excel tool “Calculateur AzoteViti” (Bellon-Maurel et al., 2015), tailored to
360 viticulture, implements models and a modelling approach similar to those in Indigo-N. Consequently, no
361 separate results are presented for this model.

362 The approach and associated Excel tool “Mineral fertiliser equivalent (MFE) calculator” (Brockmann et al.,
363 2018), is tailored to temperate conditions (a list of European countries is parameterised). Its models to
364 estimate N mineralisation of added organic matter, and NH₃ volatilisation (a combination of EMEP/EEA 2013
365 tier 2 and AGRAMMON), are flexible as to represent contrasting agricultural situations. Nonetheless its use of
366 the obsolete NUTMON model for NO₃ leaching, as well as its reliance on country-specific data to compute
367 emission factors, limit its suitability to represent contrasting agricultural situations. Moreover, the outputs
368 from this model are not directly comparable with those of Indigo-N and other models, because it considers
369 organic fertilisation only. Moreover, it considers nitrate losses after fertilisation but not as a result of the whole
370 crop cycle. The model was retained nonetheless because it represents a useful approach for organic
371 agriculture.

372 **3.1.3 Model data requirements**

373 The retained model sets (ecoinvent, AGRIBALYSE, MFE and Indigo-N) have different input data requirements,
374 further detailed and contrasted with those of STICS in Table 7. . MFE does not require any data beyond the
375 fractioning of N in the fertiliser and basic knowledge of the fertilisation mechanism, as it is based on emission
376 and operational correction factors. Indigo-N features similar data requirements (at a larger time resolution)
377 than complex models such as STICS, whose simulations are based on very detailed data files for soil, crop, and
378 (daily) weather. The basic data for non-expert use are quite similar. Pedo-transfer functions are available in
379 STICS to estimate parameters that are less often measured (or not measurable). What changes a lot is the
380 interface that makes Indigo easy to use, and the level of expertise to properly interpret the results.

381 **Table 7. Data requirements of selected N emission model sets used in France, compared with those of a complex dynamic system (STICS)**

Data	ecoinvent v3	Indigo-N v1/v2	AGRIBALYSE v1.2/v1.3	STICS v8.5 *
Weather data	<ul style="list-style-type: none"> Annual rain and irrigation (mm) 	Data provided for France, but needing adaptation for other geographies: <ul style="list-style-type: none"> Mean annual temperature (°C) Drainage after winter runoff (January-March) (mm) Drainage after spring runoff (April-June) (mm) Winter drainage (mm) Inter-annual frequency of drainage after winter runoff (fraction ≤ 1) Inter-annual frequency of spring winter drainage (fraction ≤ 1) Excess mineralisation during drainage period (%) 	<ul style="list-style-type: none"> Duration of draining period (days) Drained surface (%) 	<ul style="list-style-type: none"> Irrigation (yes/no) Detailed daily weather data (temperature, rain, etc)
Soil data	<ul style="list-style-type: none"> Rooting depth (m) Clay content (%) 	<ul style="list-style-type: none"> Texture (list provided) Clay content (%) Soil depth class (list provided) Soil organic matter content (%) Soil pebble content (%) Soil limestone content (%) Soil status as hydromorphic and humiferous (yes/no) 	<ul style="list-style-type: none"> Texture (list provided) Rooting depth (cm) Soil pebble content (%) Soil organic matter content (%) 	<ul style="list-style-type: none"> Texture (list provided) Soil pebble content (%) Soil organic matter content (%) pH Soil capacity C/N Soil density by horizon Permanent wilting point Etc...
Land preparation		<ul style="list-style-type: none"> Type of soil labour (list provided) Frequency of organic matter inputs (list provided) Frequency of burial of crop residues (list provided) Reversal of previous-year prairies (yes/no) Reversal of previous-year fallows (yes/no) 	<ul style="list-style-type: none"> Frequency of organic matter inputs (yes/no) Season of organic fertilisation (list provided) 	<ul style="list-style-type: none"> Type of soil labour (list provided) Dates of all soil labour Active/inert fractions of SOC, which correspond to all Indigo-N parameters in this category

Crop	<ul style="list-style-type: none"> N uptake by crop (kg/ha) 	<ul style="list-style-type: none"> Crop (list provided) Previous crop (list provided) Sowing date Harvest date Expected yield (kg/ha) Recommended N inputs (alternative calculation is provided if unknown) Fate of crop residues (list provided) Date of residues burial Irrigation (yes/no) Irrigation mode (list provided) 	<ul style="list-style-type: none"> Crop (list provided) Sowing date Harvest date Expected yield (kg/ha) Co-product yield (kg/ha) 	<ul style="list-style-type: none"> Crop (list provided) Sowing date Harvest method Harvest date
Intermediate and next crop		<ul style="list-style-type: none"> Intermediate crop (list provided) Sowing date of intermediate crop Next crop (list provided) Sowing date of next crop 	<ul style="list-style-type: none"> Intermediate crop (list provided) Sowing date of intermediate crop Next crop (list provided) Sowing date of next crop 	<ul style="list-style-type: none"> The rotation definition informs whether there is an intermediate crop
Fertilisation	<ul style="list-style-type: none"> Amount and N content of fertilisers (kg) TAN content of organic fertilisers (%) 	<ul style="list-style-type: none"> Fertiliser (list provided) Quantity (kg, t, m3) Date of input Localised input (yes/no) Burial of input within 24 h (yes/no) 	<ul style="list-style-type: none"> Fertiliser (list provided) Quantity (kg, t, m3) 	<ul style="list-style-type: none"> Fertiliser (list provided) Quantity (kg, t, m3) % of N-NH₄ % of dry matter % of C Date of input and associated soil labour
Background data provided	<ul style="list-style-type: none"> Correction factors for application of slurry and manure Coefficient of NH₃ volatilisation (mineral fertilisers) FAO eco-zones and their assigned carbon content and annual precipitation USDA soil orders and their assigned clay contents Crops and their rooting depth as assumed for calculations Crops and their nitrogen uptake as assumed for 	<ul style="list-style-type: none"> Minimal mineral N in soil, per soil type Soil useful reserve, per soil texture and soil depth class Number of days after which a crop reaches 50% of N uptake, per crop N absorption values until the onset of winter, per crop Proportion of N mineralised from crop residues N content of fertilisers Percentage of mineralisable N in organic fertilisers Coefficient of organic fertiliser equivalence, per organic fertiliser 	<ul style="list-style-type: none"> NPK content of fertilisers TAN of organic fertilisers Coefficient of NH₃ volatilisation (mineral fertilisers) and TAN-based coefficient of NH₃ volatilisation (organic fertilisers) Default factors for estimation of N added to soils from crop residues Coefficient of N allocation from organic fertilisers to crops, per fertiliser and season NPK content in exported crops ARVALIS data for estimation of 	<ul style="list-style-type: none"> Detailed soil, crop, and weather data files (mainly for field crops)

calculations	• Coefficient of NH_3 volatilisation, affected	leaching (see Table 1)
• NO_x emission coefficient from	by burial and chalk content of soil	• Default N_2O emission factors
N_2O		from managed soils
-

* Required input data for STICS represents minimal user-modified inputs, assuming the vast majority of data needs is fulfilled from provided data files for soils, crops, weather, etc.

TAN: total ammonia nitrogen

3.2 Comparison of simple model outputs and specific model limitations

As shown in Fig. 2 and Fig. 3, the N outputs estimated with the different models were related to the type of emission considered and influenced both by the study site and the fertilisation regime. Predicted N emissions are indeed significantly higher in wet study sites, regardless of the type of emissions considered, as supported by the lack of interaction effect between the factors *Site* and *Emission* (see p-values and further details in the Supplementary Material, Table S2). The fertilisation regime significantly influenced the model outputs by doubling the N emissions in the two wettest sites (Benin and Reunion Island) when supplied with organic fertilisers as compared to mineral fertilisation.

As expected, the 3-way ANOVA showed no significant effect of fertilisation regime or emission type as a result of the normalisation procedure. There were no interaction effect of *Model* and fertilisation regime, showing that the ability of models to predict N emissions were not significantly affected by the fertilisation regime. A significant interaction effect between *Emission* and *Model* was however found, indicating that the effect of the model depended on the type of N emission considered (see p-values and further details in the Supplementary Material, Table S3).

Table 8. Predicted values across models for all treatments together (different lowercase letters indicate significant difference at $p < 0.05$ on normalised outputs)

N flow	ecoinvent		AGRIBALYSE		MFE		Indigo-N		STICS	
NH ₃	11	a	38	b	28	ab	27	ab	25	b
N ₂ O	3.1	a	3.1	a	3.1	a	3.2	a	3.9	a
NO ₃	54	b	43	b	51	ab	17	ab	12	a

398

For NH₃, the ecoinvent model predicted significantly lower values than the AGRIBALYSE and the STICS models (Table 8). The ecoinvent outputs were systematically at the lowest level while the AGRIBALYSE NH₃ outputs were particularly high in situations of organic fertilisation and in the Senegal site. The NH₃ estimations for these two models highly relied on emission coefficients from EMEP/EEA, IPCC and Bouwman and colleagues publications (see Table 6), which are intended to have a global validity, but which fail to accurately represent emissions in contrasted tropical wet and dry climates. The ecoinvent methodology, in particular, deploys the AGRAMMON model for volatilisation. It can also be noted that AGRIBALYSE applies a volatilisation factor to total ammonia nitrogen (TAN), while ecoinvent applies it to total N. MFE, Indigo-N and STICS models were closer despite punctual divergences although they do not use at all the same calculation.

For N₂O emissions there were no significant differences across models although the outputs of STICS appeared slightly higher than the other models (Table 8), especially in situations with organic fertilisation (Figure 2). Gaseous emissions are predicted across retained models via linear regressions that include parameters such as total N and total ammonia nitrogen (TAN) inputs, as well as emission factors from EMEP/EEA, IPCC and Bouwman and colleagues publications (see Table 6). Ecoinvent and AGRIBALYSE used the same calculation method and other methods used emissions factors which seem to be close to those of ecoinvent and AGRIBALYSE. Denitrification and N₂O emissions are calculated by the STICS model according to the NOE model (Hénault et al., 2005) that considers edaphic parameters such as temperature, water field pore space, soil pH and mineral N availability.

417 For NO_x emissions, AGRIBALYSE always showed higher value than while MFE yielded results between those
418 from AGRIBALYSE and ecoinvent for both French sites (QualiAgro and Reunion Island), and similar results to
419 AGRIBALYSE for African situations (Fig. 2c).

420 Regarding NO₃, the ecoinvent and AGRIBALYSE models predicted significantly higher emissions than STICS,
421 regardless of the fertilisation regime. NO₃ emissions were rather high for ecoinvent, AGRIBALYSE and MFE,
422 especially in tropical conditions for organic fertilisation (Fig. 3), and lower for Indigo-N and STICS models which
423 are in line with the two available measured values. Thus, ecoinvent, AGRIBALYSE (for tropical conditions) and
424 MFE seem to overestimate NO₃ leaching. They take into account mainly precipitation, irrigation, rooting depth,
425 soil texture and fertiliser inputs, but overlook other factors like evapotranspiration in the calculation of
426 drainage. For instance, the nitrate model used by ecoinvent for non-European contexts and AGRIBALYSE for
427 French vegetables (SQCB-NO₃) consists of a regression equation calculating NO₃ leaching in function of
428 precipitation + irrigation, rooting depth, clay content, N in soil organic matter, fertiliser amount and crop
429 uptake. In cases where N inputs are below plant requirements, SQCB may yield negative results, and when
430 such inputs are beyond plant needs, predicted leaching soars. The reason for such behaviour is that the model
431 consists of a linear regression calibrated to specific conditions, whose validity is not global, as it seems to
432 exclude situations where crops requirements were not exactly met. High level of precipitation is commonly
433 observed in tropical wet conditions, potentially leading to high leaching output in emission models that not
434 take into consideration evapotranspiration. Indigo-N includes potential evapotranspiration in leaching
435 predictions. STICS computes actual evapotranspiration by taking into account the climatic conditions, the soil
436 water status and the crop physiological state (Constantin et al., 2015; Coucheney et al., 2015). Neither are
437 tackled by the first group of models others factors such as N adsorption and immobilisation, which reduce
438 available NO₃ in the soil, what can explain overestimation of NO₃ leaching. Furthermore, estimating the N
439 fraction of organic fertiliser available for the crop and likely to be lost remained complicated for these models
440 under tropical conditions (heat, wind, moisture) in which the mineral fraction can be rapidly lost through
441 volatilisation and where, conversely, the mineralisation of the organic fraction can be greatly accelerated as
442 compared to temperate conditions (Wetselaar and Ganry, 1982).

443 All studied models are sensitive to N inputs, but the ecoinvent and Indigo-N models for nitrate leaching are
444 highly sensitive to drainage. A variation of plus or minus 10% produced a higher variation of NO₃ leaching in
445 tropical wet contexts (Fig. 4; see more details in the Supplementary Material, Table S4 and Table S5). Thus,
446 drivers of drainage, namely soil organic carbon and clay content, rooting depth, and water inputs through
447 precipitation and irrigation, should be carefully considered.

448

450 **Fig. 2. Estimation of N gaseous direct field emissions across sites and models: A) fertilisation treatments dominated by organic inputs, B) mineral fertilisation treatments**
451 **equivalent to organic ones; 1) ammonia, 2) nitrous oxide, 3) nitrogen oxide (NO + NO₂)**

452 **Fig. 3. Estimation of nitrate direct field emissions across sites and models: A) fertilisation treatments dominated by organic inputs, B) mineral fertilisation treatments**
453 **equivalent to organic ones. Reference values based on averaged lysimetric measurements.**

454 **Fig. 4. Sensitivity of ecoinvent and Indigo-N models to a 10% change in precipitation, irrigation and drainage parameters affecting NO₃ leaching predictions for A) fertilisation**
455 **treatments dominated by organic inputs, B) mineral fertilisation treatments equivalent to organic ones**
456

457

458 **3.3 Recommendations for N modelling under various agricultural situations**

459 Based on the models we compared, and additional knowledge we have from other models and approaches, we
460 discuss here the principles that should guide future models, in such a way that these future models would be
461 better adapted to organic fertilisation, non-field crops (vegetable, perennial crops and grasslands), and varied
462 pedo-climatic conditions.

463 A guiding principle of these recommendations was that a balance is sought between simplicity (e.g. data
464 requirements) and comprehensiveness (e.g. consideration of key determinants —mechanisms, drivers— of
465 emissions) (Bockstaller et al., 2015). The ideal N model for LCA should be as simple as possible and as complex
466 as necessary.

467 **3.3.1 Allocation of N inputs among crops in a rotation and long terms effects of organic matter inputs**

468 All N inputs, be it organic fertilisers or crop residues returned to soil, should be allocated among the successive
469 crops in a rotation.

470 The consideration of crop rotation in LCA has been amply discussed in the literature (e.g. van Zeijts et al. 1999;
471 Goglio et al. 2017), and specific approaches have been implemented in French agricultural LCA databases (Koch
472 and Salou, 2016; Wilfart et al., 2016). It is a consensual conclusion that added nutrients and their associated
473 environmental impacts should be transferred from the crop where they occur to other crops in the rotation,
474 but the basis for such allocation are not always agreed upon. For instance, fertiliser inputs and their direct
475 emissions could be allocated evenly among all crops in the rotation, or weighted by some criteria such as
476 individual crop requirements. Nevertheless, organic inputs may increase the soil organic matter content and
477 thus increase the N mineralisation rate, which is the case for organic fertilisers (Noirot-Cosson et al., 2016;
478 Obriot et al., 2016) as well as for crop or catch crop residues (Constantin et al., 2012, 2010; Tribouillois et al.,
479 2016). Furthermore, the mineralisation of input organic N does not meet the crop period and can last more
480 than one crop period. Such dynamics can only be handled by a dynamic model such as STICS. Thus, a
481 compromise has to be found between this need of modelling dynamics and simplicity of representation of
482 process and data parsimony.

483 **3.3.2 Mineralisation of N in added organic matter (organic fertiliser and crop residues)**

484 The estimation of added organic matter mineralisation in form of organic fertilizer or from incorporation of
485 crop residue is relevant for computing over-fertilisation and related emissions. The N supply by organic
486 fertiliser is often expressed as mineral fertiliser equivalents (MFE) (e.g. Brockmann et al., 2018). MFE is a
487 measure of the capability of organic fertilisers to substitute mineral ones, based on their content of mineralised
488 and rapidly mineralisable N. The model used in Brockmann et al. (2018), based on the Plant Available Nitrogen
489 calculation (WEF, 2005), estimates MFE from mineralisation rates (k_{min}) affecting added organic matter, the
490 mineral N content of all fertilisers (as $N-NH_4$ and $N-NO_3$), and the N emissions (NO_3 and NH_3 losses) from all
491 organic fertilisers. Used k_{min} were obtained, as pre-calculated factors, from literature (Sullivan, 2008; WEF,
492 2005). Brockmann et al. (2018) calculated “first year” and “long term” MFE, based respectively on short- and
493 long-term N mineralisation rates. An alternative approach to MFE is for instance the coefficient of equivalence
494 of effective mineral N in fertilisers (K_{eq}N), which represents the ratio between the amount of N provided by a
495 synthetic mineral fertiliser and the total amount of N provided by an organic source which allows the same N
496 absorption by the crop (COMIFER, 2013).

For contrasting agricultural situations, we suggest the use of mineralisation kinetic curves based on moisture- and temperature-normalised days to determine k_{min} , instead of pre-calculated mineralisation factors. We assumed that this approach, created for temperate conditions, is also valid for tropical ones, as supported by the study of Sierra et al. (2010) for carbon mineralisation under maize and banana. These curves would inform the availability of mineral N to crops, and thus the risk of emissions, once faced with time-specific N needs of the crops (e.g. N absorption curves associated with plant development) and drainage events (e.g. associated with precipitation and irrigation). The use of normalised time allows analysing the role of soil properties on mineralisation, in isolation from climatic factors (confounding factors). Mineralisation kinetic curves for various organic inputs to agriculture occurring under contrasting agricultural situations are available in the literature, for instance, for the vast majority of organic residual fertilisers used in France (Bouthier et al., 2009; Houot et al., 2015), including animal effluents (Morvan et al., 2006) and agro-industrial wastewaters (Parnaudeau et al., 2006); for European catch crop residues (Justes et al., 2009), aboveground crop residues and green manures (Machet et al., 2017), root residues and green manures (Chaves et al., 2004), agricultural composts (Amlinger et al., 2003), and a large variety of plant materials (Jensen et al., 2005); as well as for African leguminous cover crops (Baijukya et al., 2006) and Brazilian root, stem and leaf residues (Abiven et al., 2005).

The Nicolardot et al. (2001) model (later included in STICS as a “decomposition sub-model”), is a dynamic mineralisation model based on the C:N ratio of crop residues and requiring fitting of initial parameters. Once integrated into STICS, there is no more fitting to be done, as it retains the default settings set in Nicolardot et al. (2001) complemented with settings from Justes et al. (2009).

3.3.3 Mineralisation of N in soil organic matter

We propose the empiric equations proposed in Clivot et al. (2017), which are based on 65 field experiments in France, where mineralisation is predicted from soil parameters. Among the various models proposed, the “soil-history-biological” model explains 77% of the computed potential net N mineralisation rate variance. Under this model, the N mineralisation rate of SOM is determined, in descending order of importance, by soil organic N, soil C:N ratio, edaphic factors (clay and CaCO_3 content, pH), the effect of returning crop residues to soil, and the activity of soil microorganisms.

This equation should be tested and re-parameterised under tropical conditions to ensure its validity. Tropical soils regularly have very acidic pH values that strongly influence the results obtained with the Clivot equation. An important challenge also concerns the validity of the functional relationship established in temperate conditions between mineralisation and soil clay content stabilising SOM. Clay mineralogy (Motavalli et al., 1995) as well as a higher degree of humification of SOM in tropical soils (Grisi et al., 1998) are supposed to modify the relation.

3.3.4 Consideration of over/under fertilisation

The excess of fertilisation is one of the major component of nitrate leaching. In the Indigo-N model, two hypotheses justify changes in N emissions due to over- and under-fertilisation: that N inputs beyond the optimal dose required by crops entails increased N leaching; and that inputs below the crop requirements do not prevent N leaching in a linear manner. This is due to the minimum amount of mineral nitrogen at harvest, available for instance in COMIFER (2013). The model thus calculates an increase of leachable N consisting of either zero (under under-fertilisation) or 50% (Machet et al., 1997) of the difference between total inputs (minus losses by volatilisation and leaching) and the theoretical optimal dose (COMIFER, 2013). This simplified

537 way to cope with more complex relations between over-fertilisation and increase of soil mineral nitrogen at
538 harvest (ten Berge, 2002) seems to be an acceptable compromise.

539 **3.3.5 NH₃ volatilisation**

540 As volatilisation happens rapidly after fertiliser application (Sommer et al., 2004), it should be deducted from
541 the computation of the other emission pathways.

542 The EMEP/EEA (2016) tier 2 model (EMEP/EEA 2016, Chapter 3.D - Crop production and agricultural soils) uses
543 emission factors, for mineral fertilisers, for various pedoclimatic conditions (i.e. discriminated by temperature
544 and pH). Tier 1 proposes emission factors for a few organic matter commonly added as fertilisers/amendments,
545 namely sewage sludge and animal effluents. In principle, tier 2 is deemed suitable for contrasting agricultural
546 situations, but correction factors should be applied to better account for agricultural management and climatic
547 conditions. Correction factors are proposed, for instance, for soil preparation and irrigation (Bockstaller and
548 Girardin, 2010), and for spreading technology, incorporation into soil, and seasonality (Brockmann et al., 2018;
549 Nemecek and Schnetzer, 2012).

550 The AGRAMMON model (<https://www.agrammon.ch/>) tier 3 proposes emission factors for organic residues
551 added as fertilisers: animal effluents and sugarcane vinasse (Nemecek and Schnetzer, 2012), as well as
552 compost, digestate and sewage sludge (Brockmann et al., 2018). These factors are expressed as emission rates
553 of the available mineral N (TAN) in the added organic matter. The model includes correction factors for
554 spreading technology, incorporation into soil, and seasonality.

555 Both models are complementary, and a combination of them with correction factors would be suitable to
556 estimate NH₃ volatilisation for contrasting agricultural situations. Both models lack specific factors for industrial
557 organic fertilisers (e.g. manure composts enriched with N-rich materials such as pressed cake and rendered
558 animal products), which should be found in other sources.

559 **3.3.6 N₂O emissions**

560 The widely used IPCC (2006) tier 1 (De Klein et al., 2006, Chapter 11 - N₂O emissions from managed soils, and
561 CO₂ emissions from lime and urea application) features emission factors for direct N₂O emissions for various
562 soils and added organic matter, based on literature. It proposes as well emission factors for indirect N₂O
563 emissions due to volatilised and re-deposited N (NH₃, NO_x), as well as to N lost to leaching and runoff. As
564 various pedoclimatic conditions, crops and added organic matter are considered by specific emission factors,
565 tier 1 is deemed suitable for contrasting agricultural situations, but correction factors should be applied to
566 better account for agricultural management and climatic conditions. Correction factors are proposed, for
567 instance, for soil type, incorporation into soil, and irrigation (Bockstaller and Girardin, 2010).

568 No model includes denitrification N₂ losses as an N-balance element, although it may represent a non-
569 negligible amount of nitrogen (Mathieu et al., 2006; Saggar et al., 2013). The empirical equation in Le Gall et al.
570 (2014) is one possible solution to calculate it in simple way.

571 **3.3.7 NO_x emissions**

572 The EMEP/EEA (2016) tier 1 model (Chapter 3.D - Crop production and agricultural soils) uses emission factors
573 for various added organic matter derived from Stehfest and Bouwman (2006). These emission factors are
574 based on a dataset of observations from global agricultural systems, representing 10 climate classes, multiple
575 soil types; organic, organo-mineral and mineral fertilisation; and a variety of field and non-field crops and

576 perennials. In principle, tier 2 is deemed suitable for contrasting agricultural situations. AGRIBALYSE, for
577 instance, retained the EMEP/EEA approach, while ecoinvent uses a single fixed emission factor.

578 **3.3.8 NO₃ leaching**

579 As shown for AGRIBALYSE in Table 6, no single approach seems suitable to represent contrasting agricultural
580 situations, mainly because most models are rather simple models adapted to specific situations. Observed
581 discrepancies among models outputs with the few measured values for the tropical site of the study were
582 observed for the simplest models, namely ecoinvent, AGRIBALYSE and MFE. This shows the limits of
583 approaches based on an emission coefficient function depending on climatic conditions, be it a dry/wet
584 differentiation (AGRIBALYSE), a single regression equation (ecoinvent), or even a more elaborated approach
585 using correction factors (MFE). The approach implemented in the Indigo-N model deserves more attention
586 since it considers processes, though in a simplified way. It computes post-fertilisation leaching and post-
587 harvest leaching (associated with draining events, which under temperate conditions correspond to the winter
588 period). For post-fertilisation leaching, it combines Burns leaching coefficients with a correction factor that
589 associates the timing of fertilisation with that of maximum N uptake by crops, according with plant uptake
590 curves for different crops (from literature). For post-harvest leaching, it combines Burns leaching coefficients
591 with post-harvest N-balances, which take into consideration mineralisation of added organic matter (crop
592 residues, organic fertilisers), mineralisation of SOM, intermediate crops, mineral N inputs, and increased N
593 losses due to over-fertilisation. This last part was inspired from the COMIFER approach behind the AGRIBALYSE
594 model for temperate situations (COMIFER, 2013; Taureau et al., 1996).

595 Such a formalism, despite being originally designed for field crops in temperate climate only, allows a great
596 flexibility for representing different drainage regimes (e.g. winter rains in temperate climates, rainy seasons in
597 tropical climates) and agricultural systems featuring different cycle lengths (e.g. vegetable cycles of <2 months
598 vs. fruit tree cycles of several years). An adaptation and enhancement of this approach, suitable for contrasting
599 agricultural conditions, would compute leaching coefficients associated with drainage regimes and soil
600 characteristics, along the duration of a crop or crop rotation. Mineralisation of organic nitrogen from input and
601 its cumulative effects should be better represented without demanding additional data.

602 **4 Conclusion**

603 A set of operational models across the N modelling continuum used in LCA to assess impacts due to nitrogen
604 losses were compared each other and to a complex model integrating a lot of processes like STICS. The
605 theoretical analysis and their implementation on four very contrasted sites, temperate and tropical showed
606 several shortcomings of such models. Although the comparison was limited to four sites with two fertiliser
607 regimes, the contrasted situations and especially the implementation under tropical conditions, made possible
608 to highlight important discrepancies among models highlighting their limitations. For nitrate leaching,
609 especially, models based on simplification excluding major drivers, e.g. using emission coefficients or
610 regression equations failed (and in general, fail) to yield sound results. This can be explained by their
611 limitations, including the poor integration of organic fertiliser and crop residues, the lack of consideration of
612 some processes like N₂ emissions, fertiliser surplus, etc. We provide recommendations for building a model
613 that more accurately represents the mode of action of organic fertilisers and considers the pedo-climatic
614 conditions prevalent beyond temperate conditions. The approach of the Indigo-N model, designed for
615 temperate conditions, could be a solid basis for the perspective when developing a model which would
616 implement the recommendations' derived from our model analysis and comparison. Lastly, we compared here

617 four LCA models and an agri-environmental model. One step further would be to integrate into the comparison
618 more simple models such as those listed the review by Buczko and Kuchenbuch (2010).

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626 **References**

- 627 Abdalla, M., Hastings, A., Cheng, K., Yue, Q., Chadwick, D., Espenberg, M., Truu, J., Rees, R.M., Smith, P., 2019.
628 A critical review of the impacts of cover crops on nitrogen leaching, net greenhouse gas balance and crop
629 productivity. *Glob. Chang. Biol.* 25, 2530–2543. <https://doi.org/10.1111/gcb.14644>
- 630 Abiven, S., Recous, S., Reyes, V., 2005. Mineralisation of C and N from root, stem and leaf residues in soil and
631 role of their biochemical quality. *Biol. Fertil. Soils* 42, 119–128. [https://doi.org/10.1007/s00374-005-0006-](https://doi.org/10.1007/s00374-005-0006-0)
632 0
- 633 Addiscott, T.M., Wagenet, R.J., 1985. Concepts of solute leaching in soils: a review of modelling approaches. *J.*
634 *Soil Sci.* 36, 411–424. <https://doi.org/10.1111/j.1365-2389.1985.tb00347.x>
- 635 Albanito, F., Lebender, U., Cornulier, T., Sapkota, T.B., Brentrup, F., Stirling, C., Hillier, J., 2017. Direct nitrous
636 oxide emissions from tropical and sub-tropical agricultural systems - A review and modelling of emission
637 factors. *Sci. Rep.* 7, 1–12. <https://doi.org/10.1038/srep44235>
- 638 Amlinger, F., Götz, B., Dreher, P., Geszti, J., 2003. Nitrogen in biowaste and yard waste compost: dynamics of
639 mobilisation and availability — a review. *Eur. J. of Soil Biol.* 39, 107–116. [https://doi.org/10.1016/S1164-](https://doi.org/10.1016/S1164-5563(03)00026-8)
640 5563(03)00026-8
- 641 Baijukya, F.P., Ridder, N. De, Giller, K.E., 2006. Nitrogen release from decomposing residues of leguminous
642 cover crops and their effect on maize yield on depleted soils of Bukoba District , Tanzania. *Plant Soil* 279,
643 77–93. <https://doi.org/10.1007/s11104-005-2504-0>
- 644 Basset-Mens, C., Acosta-Alba, I., Avadí, A., Bessou, C., Biard, Y., Feschet, P., Perret, S., Tran, T., Vayssières, J.,
645 Vigne, M., 2018. Towards specific guidelines for applying LCA in South contexts, in: The 11th International
646 Conference on Life Cycle Assessment in the Agri-Food Sector. 17 - 19 October 2018, Bangkok, Thailand.
- 647 Bellon-Maurel, V., Peters, G.M., Clermidy, S., Frizarin, G., Sinfort, C., Ojeda, H., Roux, P., Short, M.D., 2015.
648 Streamlining life cycle inventory data generation in agriculture using traceability data and information and
649 communication technologies – part II: application to viticulture. *J. Clean. Prod.* 87, 119–129.
650 <https://doi.org/10.1016/j.jclepro.2014.09.095>
- 651 Benbi, D.K., Richter, J., 2002. A critical review of some approaches to modelling nitrogen mineralization. *Biol.*
652 *Fertil. Soils* 35, 168–183. <https://doi.org/10.1007/s00374-002-0456-6>
- 653 Bernstad, A., la Cour Jansen, J., 2012. Review of comparative LCAs of food waste management systems--current
654 status and potential improvements. *Waste Manag.* 32, 2439–55.
655 <https://doi.org/10.1016/j.wasman.2012.07.023>
- 656 Bessou, C., Basset-Mens, C., Tran, T., Benoist, A., 2013. LCA applied to perennial cropping systems: A review
657 focused on the farm stage. *Int. J. Life Cycle Assess.* 18, 340–361. [https://doi.org/10.1007/s11367-012-](https://doi.org/10.1007/s11367-012-0502-z)
658 0502-z

659 Blonk Agri-footprint BV, 2014. Agri-footprint. Description of data. Gouda: Blonk Agri-footprint BV.

660 Bockstaller, C., Feschet, P., Angevin, F., 2015. Issues in evaluating sustainability of farming systems with
661 indicators. OCL - Oilseeds fats 22. <https://doi.org/10.1051/ocl/2014052>

662 Bockstaller, C., Girardin, P., 2010. Mode de calcul des indicateurs agri-environnementaux de la methode
663 Indigo®. Colmar: INRA.

664 Bockstaller, C., Guichard, L., Keichinger, O., Girardin, P., Galan, M.-B., Gaillard, G., 2009. Comparison of
665 methods to assess the sustainability of agricultural systems. A review. Agron. Sustain. Dev. 29, 223–235.
666 <https://doi.org/10.1051/agro:2008058>

667 Bockstaller, C., Guichard, L., Makowski, D., Aveline, A., Girardin, P., Plantureux, S., 2008. Agri-environmental
668 indicators to assess cropping and farming systems. A review. Agron. Sustain. Dev. 28, 139–149.
669 <https://doi.org/10.1051/agro:2007052>

670 Bourdat-Deschamps, M., Ferhi, S., Bernet, N., Feder, F., Crouzet, O., Patureau, D., Montenach, D., Moussard,
671 G.D., Mercier, V., Benoit, P., Houot, S., 2017. Fate and impacts of pharmaceuticals and personal care
672 products after repeated applications of organic waste products in long-term field experiments. Sci. Total
673 Environ. 607–608, 271–280. <https://doi.org/10.1016/j.scitotenv.2017.06.240>

674 Bouthier, A., Trochard, R., Parnaudeau, V., 2009. Cinétique de minéralisation nette de l’azote organique des
675 produits résiduels organiques à court terme in situ et en conditions contrôlées, in: 9e Renc. Fertilisation
676 Raisonnée et de l’analyse de La Terre, Comifer-Gemas, Blois. p. 6.

677 Bouwman, A., van der Hoek, K., 1997. Scenarios of animal waste production and fertilizer use and associated
678 ammonia emission for the developing countries. Atmos. Environ. 31, 4095–4102.
679 [https://doi.org/10.1016/S1352-2310\(97\)00288-4](https://doi.org/10.1016/S1352-2310(97)00288-4)

680 Bouwman, A.F., 1996. Direct emission of nitrous oxide from agricultural soils. Nutr. Cycl. Agroecosystems 46,
681 53–70. <https://doi.org/10.1007/BF00210224>

682 Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002a. Emissions of N₂O and NO from fertilized fields: Summary
683 of available measurement data. Global Biogeochem. Cycles 16, 6-1-6–13.
684 <https://doi.org/10.1029/2001GB001811>

685 Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002b. Estimation of global NH₃ volatilization loss from
686 synthetic fertilizers and animal manure applied to arable lands and grasslands. Global Biogeochem. Cycles
687 16, 8-1-8–14. <https://doi.org/10.1029/2000GB001389>

688 Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002c. Modeling global annual N₂O and NO emissions from
689 fertilized fields. Global Biogeochem. Cycles 16, 28-1-28–9. <https://doi.org/10.1029/2001GB001812>

690 Bouwman, Lex, Goldewijk, K.K., Hoek, K.W. Van Der, Beusen, A.H.W., Vuuren, D.P. Van, Willems, J., Rufino,
691 M.C., Stehfest, E., 2013. Correction for “Exploring global changes in nitrogen and phosphorus cycles in
692 agriculture induced by livestock production over the 1900–2050 period,” PNAS 110, 21195–21196.
693 <https://doi.org/10.1073/pnas.1206191109>

694 Bouwman, L., Goldewijk, K.K., Van Der Hoek, K.W., Beusen, A.H.W., Van Vuuren, D.P., Willems, J., Rufino, M.C.,
695 Stehfest, E., 2013. Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by
696 livestock production over the 1900–2050 period. Proc. Natl. Acad. Sci. 110, 20882–20887.
697 <https://doi.org/10.1073/pnas.1012878108>

698 Brentrup, F., Kiisters, J., Lammel, J., Kuhlmann, H., 2000. Methods to Estimate On-Field Nitrogen Emissions
699 from Crop Production as an Input to LCA Studies in the Agricultural Sector. Int. J. Life Cycle Assess. 5, 349–
700 357.

701 Brilli, L., Bechini, L., Bindi, M., Carozzi, M., Cavalli, D., Conant, R., Dorich, C.D., Doro, L., Ehrhardt, F., Farina, R.,

702 Ferrise, R., Fitton, N., Francaviglia, R., Grace, P., Iocola, I., Klumpp, K., Léonard, J., Martin, R., Massad, R.S.,
703 Recous, S., Seddaiu, G., Sharp, J., Smith, P., Smith, W.N., Soussana, J.F., Bellocchi, G., 2017. Review and
704 analysis of strengths and weaknesses of agro-ecosystem models for simulating C and N fluxes. *Sci. Total*
705 *Environ.* 598, 445–470. <https://doi.org/10.1016/j.scitotenv.2017.03.208>

706 Brisson, N., Gary, C., Justes, E., Roche, R., Mary, B., Ripoche, D., Zimmerb, D., Sierra, J., Bertuzzi, P., Burger,
707 P., Bussire, F., Cabidoche, Y.M., Cellier, P., Debaeke, P., Gaudillre, J.P., Hnault, C., Maraux, F., Seguin,
708 B., Sinoquet, H., 2003. An overview of the crop model STICS. *Eur. J. Agron.* 18, 309–332.
709 [https://doi.org/10.1016/S1161-0301\(02\)00110-7](https://doi.org/10.1016/S1161-0301(02)00110-7)

710 Brockmann, D., Pradel, M., Hlias, A., 2018. Agricultural use of organic residues in life cycle assessment:
711 Current practices and proposal for the computation of field emissions and of the nitrogen mineral
712 fertilizer equivalent. *Resour. Conserv. Recycl.* 133, 50–62.
713 <https://doi.org/10.1016/j.resconrec.2018.01.034>

714 Bruun, S., Hansen, T.L., Christensen, T.H., Magid, J., Jensen, L.S., 2006. Application of processed organic
715 municipal solid waste on agricultural land - A scenario analysis. *Environ. Model. Assess.* 11, 251–265.
716 <https://doi.org/10.1007/s10666-005-9028-0>

717 Buczeko, U., Kuchenbuch, R.O., 2010. Environmental indicators to assess the risk of diffuse nitrogen losses from
718 agriculture. *Environ. Manage.* 45, 1201–1222. <https://doi.org/10.1007/s00267-010-9448-8>

719 Buczeko, U., Kuchenbuch, R.O., Lennartz, B., 2010. Assessment of the predictive quality of simple indicator
720 approaches for nitrate leaching from agricultural fields. *J. Environ. Manage.* 91, 1305–1315.
721 <https://doi.org/10.1016/j.jenvman.2010.02.007>

722 Burns, I.G., 1976. Equations to predict the leaching of nitrate uniformly incorporated to a known depth or
723 uniformly distributed throughout a soil profile. *J. Agric. Sci.* 86, 305–313.
724 <https://doi.org/10.1017/S0021859600054769>

725 Burns, I.G., 1975. An equation to predict the leaching of surface-applied nitrate. *J. Agric. Sci.* 85, 443–454.
726 <https://doi.org/10.1017/S0021859600062328>

727 Cambier, P., Pot, V., Mercier, V., Michaud, A., Benoit, P., Revallier, A., Houot, S., 2014. Impact of long-term
728 organic residue recycling in agriculture on soil solution composition and trace metal leaching in soils. *Sci.*
729 *Total Environ.* 499, 560–573. <https://doi.org/10.1016/j.scitotenv.2014.06.105>

730 Campbell, B.M., Beare, D.J., Bennett, E.M., Hall-spencer, J.M., Ingram, J.S.I., Jaramillo, F., 2017. Agriculture
731 production as a major driver of the Earth system exceeding planetary boundaries. *Ecol. Soc.* 22.

732 Cannavo, P., Recous, S., Parnaudeau, V., Reau, R., 2008. Modeling N Dynamics to Assess Environmental Impacts
733 of Cropped Soils. *Adv. Agron.* 97, 131–174. [https://doi.org/10.1016/S0065-2113\(07\)00004-1](https://doi.org/10.1016/S0065-2113(07)00004-1)

734 Cariolle, M., 2002. Deac-azote : un outil pour diagnostiquer le lessivage d’azote l’chelle de l’exploitation
735 agricole de polyculture, in: *Proceedings of the 65th IRB Congress*, 13– 14 Fvrier 2002, Bruxelles. pp. 67–
736 74.

737 Cerutti, A.K., Beccaro, G.L., Bruun, S., Bosco, S., Donno, D., Notarnicola, B., Bounous, G., 2014. Life cycle
738 assessment application in the fruit sector: State of the art and recommendations for environmental
739 declarations of fruit products. *J. Clean. Prod.* 73, 125–135. <https://doi.org/10.1016/j.jclepro.2013.09.017>

740 Chaves, B., Neve, S. De, Hofman, G., Boeckx, P., Cleemput, O. Van, 2004. Nitrogen mineralization of vegetable
741 root residues and green manures as related to their (bio) chemical composition. *Eur. J. Agron.* 21, 161–
742 170. <https://doi.org/10.1016/j.eja.2003.07.001>

743 Clivot, H., Mary, B., Val, M., Cohan, J.P., Champolivier, L., Piraux, F., Laurent, F., Justes, E., 2017. Quantifying in
744 situ and modeling net nitrogen mineralization from soil organic matter in arable cropping systems. *Soil*
745 *Biol. Biochem.* 111, 44–59. <https://doi.org/10.1016/j.soilbio.2017.03.010>

Colomb, V., Amar, S.A., Mens, C.B., Gac, A., Gaillard, G., Koch, P., Mousset, J., Salou, T., Tailleur, A., Werf, H.M.G. van der, 2015. AGRIBALYSE, the French LCI database for agricultural products: high quality data for producers and environmental labelling. OCL - Oilseeds Fats, Crop. Lipids 22, D104. <https://doi.org/10.1051/ocl/20140047>

COMIFER, 2013. Calcul de la fertilisation azotée - Cultures annuelles et prairies. COMIFER- Comité Français d'Étude et de Développement de la Fertilisation Raisonnée, Groupe Azote.

COMIFER, 2001. Lessivage des nitrates en systèmes de cultures annuelles. Diagnostic du risque et proposition de gestion de l'interculture. COMIFER- Comité Français d'Étude et de Développement de la Fertilisation Raisonnée, Groupe Azote.

Constantin, J., Beaudoin, N., Launay, M., Duval, J., Mary, B., 2012. Long-term nitrogen dynamics in various catch crop scenarios: Test and simulations with STICS model in a temperate climate. Agric. Ecosyst. Environ. 147, 36–46. <https://doi.org/10.1016/j.agee.2011.06.006>

Constantin, J., Mary, B., Laurent, F., Aubrion, G., Fontaine, A., Kerveillant, P., Beaudoin, N., 2010. Effects of catch crops, no till and reduced nitrogen fertilization on nitrogen leaching and balance in three long-term experiments. Agric. Ecosyst. Environ. 135, 268–278. <https://doi.org/10.1016/j.agee.2009.10.005>

Constantin, J., Willaume, M., Murgue, C., Lacroix, B., Therond, O., 2015. The soil-crop models STICS and AqYield predict yield and soil water content for irrigated crops equally well with limited data. Agric. For. Meteorol. 206, 55–68. <https://doi.org/10.1016/j.agrformet.2015.02.011>

Coucheney, E., Buis, S., Launay, M., Constantin, J., Mary, B., García de Cortázar-Atauri, I., Ripoche, D., Beaudoin, N., Ruget, F., Andrianarisoa, K.S., Le Bas, C., Justes, E., Léonard, J., 2015. Accuracy, robustness and behavior of the STICS soil-crop model for plant, water and nitrogen outputs: Evaluation over a wide range of agro-environmental conditions in France. Environ. Model. Softw. 64, 177–190. <https://doi.org/10.1016/j.envsoft.2014.11.024>

De Klein, C., Novoa, R.S.A., Ogle, S., Smith, K.A., Rochette, P., Wirth, T.C., McConkey, B.G., Mosier, A., Rypdal, K., 2006. Chapter 11: N₂O Emissions from Managed Soils, and CO₂ Emissions from Lime and Urea Application, 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel on Climate Change (IPCC).

de Willigen, P., 2000. An analysis of the calculation of leaching and denitrification losses as practised in the NUTMON approach. Rep. 18. Wageningen, Netherlands, Plant Res. Int.

Doublet, J., Francou, C., Poitrenaud, M., Houot, S., 2011. Influence of bulking agents on organic matter evolution during sewage sludge composting; consequences on compost organic matter stability and N availability. Bioresour. Technol. 102, 1298–1307. <https://doi.org/10.1016/j.biortech.2010.08.065>

EMEP/CORINAIR, 2006. Air pollutant emission inventory guidebook, Technical report No 11/2006. European Environment Agency (EEA), Copenhagen, Denmark.

EMEP/EEA, 2016. EMEP/EEA air pollutant emission inventory guidebook 2016: Technical guidance to prepare national emission inventories. EEA Rep. No 21/2016 1–76. <https://doi.org/10.1158/1078-0432.CCR-08-2545>

EMEP/EEA, 2013. EMEP/EEA air pollutant emission inventory guidebook 2013: Technical guidance to prepare national emission inventories, EEA Technical report No. 12/2013. European Environment Agency (EEA), Copenhagen, Denmark. <https://doi.org/10.2800/92722>

EMEP/EEA, 2009. Air pollutant emission inventory guidebook, Technical report No 9/2009. European Environment Agency (EEA), Copenhagen, Denmark.

Faist Emmenegger, M., Reinhard, J., Zah, R., 2009. Sustainability Quick Check for Biofuels - intermediate background report. With contributions from T. Ziep, R. Weichbrodt, Prof. Dr. V. Wohlgemuth, FHTW

790 Berlin and A. Roches, R. Freiermuth Knuchel, Dr. G. Gaillard. Agroscope Reckenholz-Tänikon. Dübendorf.

791 FAO/IIASA, 2009. Harmonized World Soil Database (version 1.2), FAO, Rome, Italy and IIASA, Laxenburg,
792 Austria. FAO, Rome, Italy and IIASA, Laxenburg, Austria.

793 Flisch, R., Sinaj, S., Charles, R., Richner, W., 2009. GRUDAF 2009 - Grundlagen für die Düngung im Acker und
794 Futterbau. *Agrarforschung* 16, 97.

795 Fowler, D., Coyle, M., Skiba, U., Sutton, M.A., Cape, J.N., Reis, S., Sheppard, L.J., Jenkins, A., Grizzetti, B.,
796 Galloway, J.N., Vitousek, P., Leach, A., Bouwman, A.F., Butterbach-Bahl, K., Dentener, F., Stevenson, D.,
797 Amann, M., Voss, M., 2013. The global nitrogen cycle in the twenty-first century. *Philos. Trans. R. Soc. B*
798 *Biol. Sci.* 368. <https://doi.org/10.1098/rstb.2013.0164>

799 Frischknecht, R., Jungbluth, N., Althaus, H.-J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hirschler, R., Nemecek,
800 T., Rebitzer, G., Spielmann, M., 2005. The ecoinvent Database: Overview and Methodological Framework.
801 *Int. J. Life Cycle Assess.* 10, 3–9. <https://doi.org/10.1065/lca2004.10.181.1>

802 Galland, V., Avadí, A., Bockstaller, C., 2020. Data to inform the modelling of direct nitrogen field emissions from
803 global agriculture. *Data Br.*

804 Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B., Cosby, B.J., 2003. The
805 Nitrogen Cascade. *Bioscience* 53, 341. [https://doi.org/10.1641/0006-3568\(2003\)053\[0341:tnc\]2.0.co;2](https://doi.org/10.1641/0006-3568(2003)053[0341:tnc]2.0.co;2)

806 Gao, W., Guo, H.C., 2014. Nitrogen research at watershed scale: A bibliometric analysis during 1959–2011.
807 *Scientometrics* 99, 737–753. <https://doi.org/10.1007/s11192-014-1240-8>

808 Goglio, P., Brankatschk, G., Knudsen, M.T., Williams, A.G., Nemecek, T., 2017. Addressing crop interactions
809 within cropping systems in LCA. *Int. J. Life Cycle Assess.* 23, 1–9. <https://doi.org/10.1007/s11367-017-1393-9>

810

811 Goglio, P., Smith, W.N., Grant, B.B., Desjardins, R.L., McConkey, B.G., Campbell, C.A., Nemecek, T., 2015.
812 Accounting for soil carbon changes in agricultural life cycle assessment (LCA): A review. *J. Clean. Prod.*
813 104, 23–39. <https://doi.org/10.1016/j.jclepro.2015.05.040>

814 Grisi, B., Grace, C., Brookes, P.C., Benedetti, A., Dell'Abate, M.T., 1998. Temperature effects on organic matter
815 and microbial biomass dynamics in temperate and tropical soils. *Soil Biol. Biochem.* 30, 1309–1315.
816 [https://doi.org/10.1016/S0038-0717\(98\)00016-9](https://doi.org/10.1016/S0038-0717(98)00016-9)

817 Groenendijk, P., Renaud, L.V., Roelsma, J., 2005. Prediction of Nitrogen and Phosphorus leaching to
818 groundwater and surface waters. Process descriptions of the ANIMO4.0 model, Alterra–Report 983.
819 Alterra, Wageningen.

820 Heijungs, R., 2021. Selecting the best product alternative in a sea of uncertainty. *Int. J. Life Cycle Assess.*
821 <https://doi.org/10.1007/s11367-020-01851-4>

822 Hénault, C., Bizouard, F., Laville, P., Gabrielle, B., Nicoullaud, B., Germon, J.C., Cellier, P., 2005. Predicting in situ
823 soil N₂O emission using NOE algorithm and soil database. *Glob. Chang. Biol.* 11, 115–127.
824 <https://doi.org/10.1111/j.1365-2486.2004.00879.x>

825 Hergoualc'h, K., Akiyama, H., Bernoux, M., Chirinda, N., Prado, A. del, Kasimir, Å., MacDonald, J.D., Ogle, S.M.,
826 Regina, K., Weerden, T.J. van der, 2019. Chapter 11: N₂O Emissions from Managed Soils, and CO₂
827 Emissions from Lime and Urea Application, 2019 Refinement to the 2006 IPCC Guidelines for National
828 Greenhouse Gas Inventories. Intergovernmental Panel on Climate Change (IPCC).

829 Houot, S., Pierre, P., Decoopman, B., Trochard, R., Gennen, J., Luxen, P., 2015. Minéralisation de produits
830 résiduels organiques : des sources d'azote variées. *Fourrages* 224, 257–264.

831 IFA/FAO, 2001. Global estimates of gaseous emissions of NH₃, NO and N₂O from agricultural land. Rome,
832 International Fertilizer Industry Association and Food and Agriculture Organization of the United Nations.

833 IIASA/FAO, 2012. Global Agro-ecological Zones (GAEZ v3.0). IIASA, Laxenburg, Austria and FAO, Rome, Italy.

834 IPCC, 2006. Volume 4. Agriculture, forestry and other land use, 2006 IPCC Guidelines for National Greenhouse
835 Gas Inventories. Intergovernmental Panel on Climate Change, Prepared by the National Greenhouse Gas
836 Inventories Programme.

837 ISO, 2006. ISO 14040 Environmental management — Life cycle assessment — Principles and framework. The
838 International Standards Organisation. <https://doi.org/10.1136/bmj.332.7550.1107>

839 Jensen, L.S., Salo, T., Palmason, F., Breland, T.A., Henriksen, T.M., Stenberg, B., Pedersen, A., Lundstro, C., 2005.
840 Influence of biochemical quality on C and N mineralisation from a broad variety of plant materials in soil.
841 *Plants Soil* 273, 307–326. <https://doi.org/10.1007/s11104-004-8128-y>

842 Jones, J.W., Antle, J.M., Basso, B., Boote, K.J., Conant, R.T., Foster, I., Godfray, H.C.J., Herrero, M., Howitt, R.E.,
843 Janssen, S., Keating, B.A., Munoz-Carpena, R., Porter, C.H., Rosenzweig, C., Wheeler, T.R., 2017. Brief
844 history of agricultural systems modeling. *Agric. Syst.* 155, 240–254.
845 <https://doi.org/10.1016/j.agry.2016.05.014>

846 Justes, E., Mary, B., Nicolardot, B., 2009. Quantifying and modelling C and N mineralization kinetics of catch
847 crop residues in soil : parameterization of the residue decomposition module of STICS model for mature
848 and non mature residues. *Plant Soil* 325, 171–185. <https://doi.org/10.1007/s11104-009-9966-4>

849 Kasper, M., Foldal, C., Kitzler, B., Haas, E., Strauss, P., Eder, A., Zechmeister-Boltenstern, S., Amon, B., 2019. N₂
850 O emissions and NO₃– leaching from two contrasting regions in Austria and influence of soil, crops and
851 climate: a modelling approach. *Nutr. Cycl. Agroecosystems* 113, 95–111. [https://doi.org/10.1007/s10705-](https://doi.org/10.1007/s10705-018-9965-z)
852 [018-9965-z](https://doi.org/10.1007/s10705-018-9965-z)

853 Koch, P., Salou, T., 2016. AGRIBALYSE[®] : Rapport Méthodologique - Version 1.3. ART, INRA, ADEME.

854 Koch, P., Salou, T., 2015. AGRIBALYSE[®] : METHODOLOGY Version 1.2. Ed. ADEME, Angers, France.

855 Kücke, M., Kleeberg, P., 1997. Nitrogen balance and soil nitrogen dynamics in two areas with different soil,
856 climatic and cropping conditions. *Eur. J. Agron.* 6, 89–100. [https://doi.org/10.1016/S1161-0301\(96\)02027-](https://doi.org/10.1016/S1161-0301(96)02027-8)
857 [8](https://doi.org/10.1016/S1161-0301(96)02027-8)

858 Kwiatkowska-Malina, J., 2018. Qualitative and quantitative soil organic matter estimation for sustainable soil
859 management. *J. Soils Sediments* 18, 2801–2812. <https://doi.org/10.1007/s11368-017-1891-1>

860 Laurent, F., Castillon, P., 1987. Le reliquat azoté sortie hiver. *Perspect. Agric.* 47–57.

861 Le Gall, C., Jeuffroy, M.H., Hénault, C., Python, Y., Cohan, J.P., Parnaudeau, V., Mary, B., Compere, P., Tristant,
862 D., Duval, R., Cellier, P., 2014. Analyser et estimer les émissions de N₂O dans les systèmes de grandes
863 cultures français. *Innov. Agron.* 34, 367–378.

864 Machet, J.-M., Dubrulle, P., Damay, N., Duval, R., Julien, J.-L., Recous, S., 2017. A Dynamic Decision-Making Tool
865 for Calculating the Optimal Rates of N Application for 40 Annual Crops While Minimising the Residual
866 Level of Mineral N at Harvest. *Agronomy* 7, 73. <https://doi.org/10.3390/agronomy7040073>

867 Machet, J.M., Dubrulle, P., Louis, P., 1990. AZOBIL: a computer program for fertilizer N recommendations based
868 on a predictive balance sheet method, in: *Proceedings of the First Congress of the European Society of*
869 *Agronomy* (p. 21). Paris, FRA (1990-12-05 - 1990-12-07).

870 Machet, J.M., Laurent, F., Chapot, J.Y., Dore, T., Dulout, A., 1997. Maîtrise de l'azote dans les intercultures et
871 les jachères, in: Lemaire, G., Nicolardot, B. (Eds.), *Maîtrise de l'azote Dans Les Agrosystèmes: Les*
872 *Colloques de l'INRA*. Reims: INRA, pp. 271–288.

873 Manzoni, S., Porporato, A., 2009. Soil carbon and nitrogen mineralization: Theory and models across scales. *Soil*
874 *Biol. Biochem.* 41, 1355–1379. <https://doi.org/10.1016/j.soilbio.2009.02.031>

875 Mathieu, O., Lévêque, J., Hénault, C., Milloux, M.J., Bizouard, F., Andreux, F., 2006. Emissions and spatial
876 variability of N₂O, N₂ and nitrous oxide mole fraction at the field scale, revealed with 15N isotopic
877 techniques. *Soil Biol. Biochem.* 38, 941–951. <https://doi.org/10.1016/j.soilbio.2005.08.010>

878 Meier, M.S., Stoessel, F., Jungbluth, N., Juraske, R., Schader, C., Stolze, M., 2015. Environmental impacts of
879 organic and conventional agricultural products - Are the differences captured by life cycle assessment? *J.*
880 *Environ. Manage.* 149, 193–208. <https://doi.org/10.1016/j.jenvman.2014.10.006>

881 Menzi, H., Katz, P., Fahrni, M., Keller, M., 1997. Ammonia emissions following the application of solid manure
882 to grassland, in: *Gaseous Nitrogen Emissions from Grasslands* (Eds. Jarvis, S. and Pain, B.). CAB
883 International, Oxon, UK, pp. 265–274.

884 Morvan, T., Nicolardot, B., Péan, L., 2006. Biochemical composition and kinetics of C and N mineralization of
885 animal wastes: A typological approach. *Biol. Fertil. Soils* 42, 513–522. [https://doi.org/10.1007/s00374-](https://doi.org/10.1007/s00374-005-0045-6)
886 005-0045-6

887 Motavalli, P.P., Palm, C.A., Elliott, E.T., Frey, S.D., Smithson, P.C., 1995. Nitrogen Mineralization in Humid
888 Tropical Forest Soils: Mineralogy, Texture, and Measured Nitrogen Fractions. *Soil Sci. Soc. Am. J.* 59,
889 1168–1175. <https://doi.org/10.2136/sssaj1995.03615995005900040032x>

890 Nemecek, T., Bengoa, X., Lansche, J., Mouron, P., Rossi, V., Humbert, S., 2015. World Food LCA Database:
891 Methodological Guidelines for the Life Cycle Inventory of Agricultural Products. Version 3.0.

892 Nemecek, T., Bengoa, X., Rossi, V., Humbert, S., 2014. World Food LCA Database: Methodological Guidelines for
893 the Life Cycle Inventory of Agricultural Products. Version 2.0 79.

894 Nemecek, T., Bengoa, X., Rossi, V., Humbert, S., Lansche, J., Mouron, P., 2020. World Food LCA Database:
895 Methodological guidelines for the life cycle inventory of agricultural products. Version 3.5. Agroscope and
896 Quantis.

897 Nemecek, T., Schnetzer, J., 2012. Methods of assessment of direct field emissions for LCIs of agricultural
898 production systems. Data v3.0, Agroscope Reckenholz-Tanikon Research station.

899 Nicolardot, B., Recous, S., Mary, B., 2001. Simulation of C and N mineralisation during crop residue
900 decomposition: A simple dynamic model based on the C:N ratio of the residues. *Plant Soil* 228, 83–103.

901 Noirot-Cosson, P.E., Vaudour, E., Gilliot, J.M., Gabrielle, B., Houot, S., 2016. Modelling the long-term effect of
902 urban waste compost applications on carbon and nitrogen dynamics in temperate cropland. *Soil Biol.*
903 *Biochem.* 94, 138–153. <https://doi.org/10.1016/j.soilbio.2015.11.014>

904 Obriot, F., Stauffer, M., Goubard, Y., Revallier, A., Vieublé-Gonod, L., Houot, S., 2016. Effects of repeated
905 organic amendment applications on soil and crop qualities. *Acta Hortic.* 1146, 87–96.
906 <https://doi.org/10.17660/ActaHortic.2016.1146.11>

907 Oenema, O., Velthof, G., Amann, M., Klimont, Z., Winiwarter, W., 2012. Emissions from agriculture and their
908 control potentials, TSAP Report #3, Version 1.0, DG-Environment of the European Commission.

909 Padilla, F.M., Gallardo, M., Manzano-Agugliaro, F., 2018. Global trends in nitrate leaching research in the 1960–
910 2017 period. *Sci. Total Environ.* 643, 400–413. <https://doi.org/10.1016/j.scitotenv.2018.06.215>

911 Parnaudeau, V., Nicolardot, B., Robert, P., Alavoine, G., Pagès, J., Duchiron, F., 2006. Organic matter
912 characteristics of food processing industry wastewaters affecting their C and N mineralization in soil
913 incubation. *Bioresour. Technol.* 97, 1284–1295. <https://doi.org/10.1016/j.biortech.2005.05.023>

914 Perrin, A., 2013. Evaluation environnementale des systèmes agricoles urbains en Afrique de l’Ouest :
915 Implications de la diversité des pratiques et de la variabilité des émissions d’azote dans l’Analyse du Cycle
916 de Vie de la tomate au Bénin. PhD thesis. Sciences agricoles. AgroParisTech, 2013. Français.

917 Perrin, A., Basset-Mens, C., Gabrielle, B., 2014. Life cycle assessment of vegetable products: A review focusing

on cropping systems diversity and the estimation of field emissions. *Int. J. Life Cycle Assess.* 19, 1247–1263. <https://doi.org/10.1007/s11367-014-0724-3>

Piepho, H.P., 2018. Letters in mean comparisons: What they do and don't mean. *Agron. J.* 110, 431–434. <https://doi.org/10.2134/agronj2017.10.0580>

Prado, V., 2018. Interpretation of comparative LCAs: external normalization and a method of mutual differences 2018–2029. <https://doi.org/10.1007/s11367-017-1281-3>

R Core Team, 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria [WWW Document]. URL <http://www.r-project.org/index.html>

Rasmussen, L.V., Bierbaum, R., Oldekop, J.A., Agrawal, A., 2017. Bridging the practitioner-researcher divide : Indicators to track environmental , economic , and sociocultural sustainability of agricultural commodity production. *Glob. Environ. Chang.* 42, 33–46. <https://doi.org/10.1016/j.gloenvcha.2016.12.001>

Richner, W., Oberholzer, H.-R., Freiermuth, R., Huguenin, O., Ott, S., Nemecek, T., 2014. Modell zur Beurteilung der Nitrat- auswaschung in Ökobilanzen - SALCA-NO₃, Agroscope.

Roy, R.N., Misra, R.V., Lesschen, J.P., Smaling, E.M., 2003. Assessment of soil nutrient balance. Approaches and methodologies, *FAO Fertiliser and Plant Nutrition Bulletin* 14. Rome, Food and Agriculture Organization of the United Nations.

Saggar, S., Jha, N., Deslippe, J., Bolan, N.S., Luo, J., Giltrap, D.L., Kim, D.G., Zaman, M., Tillman, R.W., 2013. Denitrification and N₂O: N₂ production in temperate grasslands: Processes, measurements, modelling and mitigating negative impacts. *Sci. Total Environ.* 465, 173–195. <https://doi.org/10.1016/j.scitotenv.2012.11.050>

Sierra, J., Brisson, N., Ripoche, D., Déqué, M., 2010. Modelling the impact of thermal adaptation of soil microorganisms and crop system on the dynamics of organic matter in a tropical soil under a climate change scenario. *Ecol. Modell.* 221, 2850–2858. <https://doi.org/10.1016/j.ecolmodel.2010.08.031>

Sommer, S.G., Schjoerring, J.K., Denmead, O.T., 2004. Ammonia Emission from Mineral Fertilizers and Fertilized Crops. *Adv. Agron.* 82, 557–622. [https://doi.org/10.1016/s0065-2113\(03\)82008-4](https://doi.org/10.1016/s0065-2113(03)82008-4)

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., Vries, W. De, Wit, C.A. De, Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sörlin, S., 2015. Planetary boundaries: Guiding changing planet. *Science* (80-.). 347. <https://doi.org/10.1126/science.1259855>

Stehfest, E., Bouwman, L., 2006. N₂O and NO emission from agricultural fields and soils under natural vegetation: Summarizing available measurement data and modeling of global annual emissions. *Nutr. Cycl. Agroecosystems* 74, 207–228. <https://doi.org/10.1007/s10705-006-9000-7>

Sullivan, D.M., 2008. Estimating Plant-available Nitrogen from Manure, Oregon State University, Extension Catalog.

Tailleur, A., Cohan, J., Laurent, F., Lellahi, A., 2012. A simple model to assess nitrate leaching from annual crops for life cycle assessment at different spatial scales, in: Corson M.S., van Der Werf H.M.G. (Eds), *Proceedings of the 8th International Conference on Life Cycle Assessment in the Agri-Food Sector (LCA Food 2012)*, 1-4 October 2012, Saint-Malo, France. INRA, Rennes France. pp. 903–904.

Taureau, J.C., Gitton, C., Laurent, F., Machet, J.M., Plas, D., 1996. Calcul de la fertilisation azotée des cultures annuelles. Paris: COMIFER.

ten Berge, H.F.M., 2002. A review of potential indicators for nitrate loss from cropping and farming systems in the Netherlands, Report 31. Plant Research International B.V., Wageningen.

Tribouillois, H., Cohan, J.P., Justes, E., 2016. Cover crop mixtures including legume produce ecosystem services

961 of nitrate capture and green manuring: assessment combining experimentation and modelling. *Plant Soil*
962 401, 347–364. <https://doi.org/10.1007/s11104-015-2734-8>

963 van Lent, J., Hergoualc’h, K., Verchot, L. V., 2015. Reviews and syntheses: Soil N₂O and NO emissions from land
964 use and land-use change in the tropics and subtropics: A meta-analysis. *Biogeosciences* 12, 7299–7313.
965 <https://doi.org/10.5194/bg-12-7299-2015>

966 van Wart, J., van Bussel, L.G.J., Wolf, J., Licker, R., Grassini, P., Nelson, A., Boogaard, H., Gerber, J., Mueller,
967 N.D., Claessens, L., van Ittersum, M.K., Cassman, K.G., 2013. Use of agro-climatic zones to upscale
968 simulated crop yield potential. *F. Crop. Res.* 143, 44–55. <https://doi.org/10.1016/j.fcr.2012.11.023>

969 van Zeijts, H., Leneman, H., Wegener Sleswijk, A., 1999. Fitting fertilisation in LCA: allocation to crops in a
970 cropping plan. *J. Clean. Prod.* 7, 69–74. [https://doi.org/10.1016/S0959-6526\(98\)00040-7](https://doi.org/10.1016/S0959-6526(98)00040-7)

971 Vázquez, N., Pardo, A., Suso, M.L., Quemada, M., 2005. A methodology for measuring drainage and nitrate
972 leaching in unevenly irrigated vegetable crops. *Plant Soil* 269, 297–308. [https://doi.org/10.1007/s11104-](https://doi.org/10.1007/s11104-004-0630-8)
973 004-0630-8

974 WEF, 2005. National Manual of Good Practice for Biosolids. Alexandria, VA, USA: Water Environment
975 Federation.

976 Wetselaar, R., Ganry, F., 1982. Nitrogen balance in tropical agrosystems. *Micobiology Trop. soils plant Product.*
977 1–35. https://doi.org/10.1007/978-94-009-7529-3_1

978 Wilfart, A., Espagnol, S., Dauguet, S., Tailleur, A., Gac, A., Garcia-Launay, F., 2016. ECOALIM: a dataset of
979 environmental impacts of feed ingredients used in French animal production. *PLoS One* 11, 17.
980 <https://doi.org/10.5061/dryad.14km1>

981 Yang, B., Huang, K., Sun, D., Zhang, Y., 2017. Mapping the scientific research on non-point source pollution: a
982 bibliometric analysis. *Environ. Sci. Pollut. Res.* 24, 4352–4366. [https://doi.org/10.1007/s11356-016-8130-](https://doi.org/10.1007/s11356-016-8130-y)
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985 **Figure captions**

- 986 Fig. 1. Modelling continuum for estimation of N emissions in the French LCA context
- 987 Fig. 2. Estimation of N gaseous direct field emissions across sites and models: A) fertilisation treatments
988 dominated by organic inputs, B) mineral fertilisation treatments equivalent to organic ones; 1) ammonia, 2)
989 nitrous oxide, 3) nitrogen oxide (NO + NO₂)
- 990 Fig. 3. Estimation of nitrate direct field emissions across sites and models: A) fertilisation treatments dominated
991 by organic inputs, B) mineral fertilisation treatments equivalent to organic ones. Reference values based on
992 averaged lysimetric measurements
- 993 Fig. 4. Sensitivity of ecoinvent and Indigo-N models to a 10% change in precipitation, irrigation and drainage
994 parameters affecting NO₃ leaching predictions for A) fertilisation treatments dominated by organic inputs, B)
995 mineral fertilisation treatments equivalent to organic ones