



HAL
open science

Comparison of empirical and process-based modelling to quantify soil-supported ecosystem services on the Saclay plateau (France)

Pauline Choquet, Benoit Gabrielle, Maha Chalhoub, Joël Michelin, Ophélie Sauzet, Ottone Scammacca, Patricia Garnier, Philippe Baveye, David Montagne

► To cite this version:

Pauline Choquet, Benoit Gabrielle, Maha Chalhoub, Joël Michelin, Ophélie Sauzet, et al.. Comparison of empirical and process-based modelling to quantify soil-supported ecosystem services on the Saclay plateau (France). *Ecosystem Services*, 2021, 50, pp.101332. 10.1016/j.ecoser.2021.101332 . hal-03745779

HAL Id: hal-03745779

<https://brgm.hal.science/hal-03745779>

Submitted on 2 Aug 2023

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.



Distributed under a Creative Commons Attribution - NonCommercial 4.0 International License

COMPARISON OF EMPIRICAL AND PROCESS-BASED MODELLING TO QUANTIFY SOIL-SUPPORTED ECOSYSTEM SERVICES ON THE SACLAY PLATEAU (FRANCE).

Pauline Choquet^{a,b}

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

^bAgence de l'environnement et de la maîtrise de l'énergie (ADEME), 49000, Angers, France.

pauline.choquet2@gmail.com

Benoit Gabrielle^a

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

Benoit.gabrielle@agroparistech.fr

Maha Chalhoub^{a,c}

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

^cBRGM - Bureau de Recherches Géologiques et Minières (BRGM), 45000, Orléans, France.

M.Chalhoub@brgm.fr

Joël Michelin^a

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

Joel.michelin@agroparistech.fr

Ophélie Sauzet^{a,d}

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

^d University of Applied Science of Western Switzerland Hepia, Soils and Substrates Group, Institute Land-Nature-Environment, route de Presinge 150, 1254 Jussy, Geneva, Switzerland

ophelie.sauzet@hesge.ch

Ottone Scammacca^a

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

Ottone.scammacca@inrae.fr

Patricia Garnier^a

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

Patricia.garnier@inrae.fr

Philippe C. Baveye^a

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

baveye.rpi@gmail.com

David Montagne^a

^aUniversité Paris-Saclay, INRAE, AgroParisTech, UMR ECOSYS, 78850 Thiverval-Grignon, France.

David.montagne@agroparistech.fr

Corresponding author : David Montagne

Figure 1: Land-use map according to Corine digital land cover (2015) and Soil map of the Saclay plateau adapted from regional geographic soil database at 1 : 250 000 of the Ile-de-France region

Figure 2: Rescaled soil contribution to the provision of plant biomass using empirical, process-based modelling and direct measurement (data from Chalhoub et al., 2020) during the crop cycle 2017/2018. The errors bars correspond to the standard error observed between the 20 years of simulation for the process-based modelling. The process-based modelling for 2018 is also presented to allow direct comparison of direct measurement and process-based modelling for the same soil, the crop and crop management.

Figure 3: Rescaled soil contribution to the provision of water using empirical and process-based modelling and direct measurement (data from Chalhoub et al., 2020) during the crop cycle 2017/2018. The errors bars correspond to the standard error observed between the 20 years of simulation for the process-based modelling. The process-based modelling for 2018 is also presented to allow direct comparison of direct measurement and process-based modelling for the same soil, the crop, and crop management.

Figure 4: Rescaled soil contribution to the regulation of water quality using empirical and process-based modelling and direct measurement (data from Chalhoub et al., 2020) during the crop cycle 2017/2018. The errors bars correspond to the standard error observed between the 20 years of simulation for the process-based modelling. The process-based modelling for 2018 is also presented to allow direct comparison of direct measurement and process-based modelling for the same soil, the crop, and crop management.

Figure 5: Rescaled soil contribution to the regulation of global climate using empirical and process-based modelling.

Figure 6: Map on the Saclay plateau territory of the joint delivery of provisioning (biomass and water) and regulating (water quality and global climate) ecosystem services according to empirical (a) and process-based (b) modelling

1 I. Introduction

2 Since its inception in the 1950s until just a few years ago, the literature on nature's services, referred
3 to as "ecosystem services" (ES) after 1997, focused largely on defining concepts, identifying and
4 classifying services, and debating the extent to which a monetization of these services was feasible
5 (e.g., Westman, 1977; Gómez-Baggethun *et al.*, 2010; Baveye *et al.*, 2013, 2016, 2020; Baveye, 2014).
6 In the last few years, the emphasis has shifted to the operationalization of the concept (van Dijk *et*
7 *al.*, 2018), i.e., to use information about the ecological, sociocultural, or economic values of ES to
8 help implement various strategies for sustainable land, water, or urban management (Dick *et al.*,
9 2018, Harrison *et al.*, 2018). Many decision-support tools have emerged to fill the gap between
10 theoretical concepts, the biophysical assessment of ES, and their practical application for decision-
11 making. Bagstad and al. (2013) found and compared seventeen such decision-support tools, and the
12 list has steadily grown since (Grêt-Regamey and al. 2017).

13 The accuracy of most available tools and models, defined as the degree to which they reproduce
14 the real distribution and abundance of an ES (Schröter *et al.*, 2015), remains an object of debate at
15 this juncture. This is particularly the case when, beyond recognizing and increasing awareness of ES,
16 one attempts to capture their value as requested for priority setting (e.g., identifying areas critical for
17 ES provision) or instrument design (e.g., development of payment schemes for ES). One problematic
18 aspect of current tools is that due to the strong ecosystem bias that characterizes the bulk of the
19 research on ES, the soil component is often completely ignored (Baveye *et al.*, 2016, 2018). For
20 example, in the sizeable body of literature involving capacity matrices linking land use/land cover
21 types to ES bundles, soil heterogeneity inside land use/land cover patches is essentially overlooked
22 (Burkhard *et al.*, 2009; Jacobs *et al.*, 2015; Roche and Campagne, 2019). In the best cases, soils are
23 only poorly represented (Dominati *et al.*, 2016) using surface- or mean key physical, chemical or
24 biological properties as simplified proxies for soil ES (Rutgers *et al.*, 2012; van Wijnen *et al.*, 2012;
25 Hewitt *et al.*, 2015). Such strategies are reasonable for ES unrelated to soils (e.g., the service of
26 pollination). However, this is not the case for the numerous ES that are strongly linked to soil
27 processes and properties, as emphasized in several frameworks linking soil processes and properties,
28 associated soil-supported ES, and in some cases potential indicators for these services (Robinson *et*
29 *al.*, 2009; Dominati *et al.*, 2010; Adhikhari and Hartemink 2016; Buneman *et al.*, 2018; Vogel *et al.*,
30 2019; Fossey *et al.*, 2020). Indeed, different soils under the same land use sometimes evince
31 considerable differences in their ES levels and their economic values (Hewitt *et al.*, 2015; Dominati *et*
32 *al.*, 2016). Integration of a more realistic representation of soil diversity, complexity (e.g., depth-
33 distributions of soil properties), and soil spatial variation in soil-supported ES assessment thus
34 appears necessary to reduce the uncertainty attached to traditional assessment methods, and
35 ultimately to determine how land-use and land-management planning may be used to improve the

36 ES delivery according to stakeholders and policy-makers preferences (Schulp *et al.*, 2012; Lavorel at
37 *al.*, 2017; Muller *et al.*, 2020).

38 In this context, and if one considers that, to be most efficient, decision-support tools should embody
39 modelling in one way or another, the existence of a wide array of models of soil processes should be
40 viewed as encouraging. Among the large diversity of models that could be used to assess soil-
41 supported ES, some are based on empirical relationships that are either statistically established or
42 derived from expert knowledge (Debeljak *et al.*, 2019). The Storie Index (Storie, 1978) or the
43 Universal Soil Loss Equation (Wischmeier and Smith, 1978) are widely used examples that quantify
44 the provision of biomass or the regulation of soil erosion, respectively. These types of empirical
45 relationships generally assess the capacity of a soil to deliver a specific service more or less
46 independently of any explicit consideration of land use and land management (Greiner *et al.*, 2017).
47 The possibility to use empirical modelling for services strongly linked to biotic processes such as the
48 climate or water quality regulation services is however unclear. Estimation of biogeochemical and
49 biotic processes from classical soil data still remains an unresolved challenge (Van Looy *et al.*, 2017).

50 In contrast with empirical models, process-based models involve mechanistic representations of soil
51 processes. They have recently received considerable attention (Dominati *et al.*, 2016; Francesconi *et*
52 *al.*, 2016; Demestihis *et al.*, 2018; Ellili-Bargaoui *et al.*, 2021). They include explicitly site-specific
53 environmental factors such as climate, as well as temporal or spatial variations in land-use and land-
54 management (Greiner *et al.*, 2017). They are often viewed as potentially more accurate than
55 empirical models and applicable to a range of soil-supported services for which empirical models or
56 the possibility of direct measurements may be lacking at this point (Vereecken *et al.*, 2016). For
57 example, Dominati *et al.* (2014 and 2016) used a crop model to quantify food provision, flood
58 mitigation, nutrient filtering, organic carbon (OC) storage and greenhouse gas regulation under dairy
59 grazed pastures. The benefits of process-based modelling are however still uncertain as the potential
60 gains in accuracy linked to a more realistic representation of soil functioning are limited by the
61 availability, the spatial and temporal variability, or the reliability of the underlying data, and a
62 correspondingly limited capacity for independent application by most stakeholders (Bagstad *et al.*,
63 2013).

64 At this stage, the different modelling approaches, either empirical or process-based, that are
65 available to estimate the soil-supported ES in natural, forested, or cultivated areas have yet to be
66 compared with each other, and especially, to actual measurements of services (Dominati *et al.*, 2016,
67 Harrison *et al.*, 2018; Zulian *et al.*, 2018). In agreement with the general ES research, the lack of
68 direct measurements of soil-supported services is a critical issue (Remme *et al.*, 2014; Baveye *et al.*,
69 2016, Baveye 2017). Fortunately, progress is being achieved in this respect. A first set of field
70 measurements of soil-supported ES was obtained recently by Chalhoub *et al.* (2020), and the

71 resulting data can now be used to test the accuracy of empirical or process-based models. These
72 actual measurements are still limited in scope, since they concern only a small subset of all services,
73 but one might argue that they are an encouraging first step.

74 In this general context, the overall objective of the research described here is to evaluate our current
75 ability to assess and map two provisioning and two regulating soil-supported ES across a landscape
76 made of imbricated natural, forested, and cultivated areas. The former two ES relate to biomass and
77 water provision, and the latter two encompass regulation in terms of water quality and the global
78 climate. ES assessment is carried out under the constraints that it i) integrate a realistic
79 representation of soils, considered here as a mandatory step for an accurate assessment and
80 mapping of, at least, the soil-supported ES, and ii) is based only on easily-available data and models
81 in order to be routinely used in ES-based decision making. More specifically, two approaches of
82 increasing complexity (empirical versus process-based modelling) are adapted to the context of the
83 Saclay plateau (France) using the best available data and models. These two approaches are
84 compared for their ability i) to be close to the actual field measurements obtained by Chalhoub *et al.*,
85 (2020) in a cultivated soil (absolute accuracy), and; ii) to translate strong gradients of inherent and
86 manageable soil properties into differences in levels of the soil-supported ES (relative accuracy). This
87 research is then in line with ongoing efforts to better assess the influence of model choices on ES
88 estimates (Bagstad *et al.*, 2018)

89

90 **II. Material and methods**

91 II.1. Study area and soil data

92 II.1.1. The Saclay plateau

93 The Saclay plateau, of an area of 10,406 ha, is located in the Ile-de-France region, southwest of Paris
94 (Figure 1). Its geological stratification is typical of the Parisian Basin (Jamagne, 2011) with loess
95 deposits several meters deep overlaying a claybed rich in millstone called “argiles à Meulières”, a
96 sand deposit called “sables de Fontainebleau” of several tens of meters thick, and, finally, marl
97 deposits (Megnier, 1989). Since its artificial drainage through a complex network of swales at the
98 end of the 17th century, this previously marshy area evolved into a fertile plateau characterized by
99 dynamic agricultural activities (Tedesco *et al.*, 2017). Forests, contrastingly, dominate the
100 surrounding slopes and the valleys when not urbanized (Figure 1).

101

102 II.1.2. Soil data

103 The soil map presented in Figure 1 originates from the regional geographic soil database at
104 1 : 250 000 scale (Roque, 2004), called “Référentiels Régionaux Pédologiques” (RRP), which is the
105 most precise source of soil information in almost the whole continental France. In the RRP, soils are
106 mapped as a set of Soil Map Units (SMU), each one including one or several Soil Type Units (STU) that
107 are not spatially delineated but are described, quantified area-wise, and characterized in a soil
108 database. We took advantage of a soil-sampling program carried out on the Saclay plateau, to
109 describe and characterise the most abundant soil type in each SMU according to the specifications of
110 the RRP (INRA Infosol, 2005). Tables 1 and 2 provide a summary of, respectively, the main descriptive
111 and analytical soil properties considered in the assessment of soil-supported ES. Methods of soil
112 analysis are described in the supplementary material S1.

113 On nearly level top positions of the Saclay plateau, silty loam to silty clay loam Stagnic Luvisols
114 developed in thick loess deposits, with calcareous loess encountered between 100 and 120 cm depth
115 (SMU 29, Figure 1 and Table 1). When the thickness of the loess deposit decreases progressively
116 toward the borders of the Saclay plateau, thinner Stagnic Luvisols directly lie on the more or less
117 redistributed “argiles à Meulières” clay layer (SMU 30, Figure 1 and Table 1). Along the edge of the
118 Saclay plateau (SMU 31, Figure 1 and Table 1), Planosols developed in the superposition of a stony
119 and silty loamy slope deposit of 50 cm depth over the “argiles à Meulières” clay layer. These
120 Planosols on clay were formerly cultivated but most of them are currently under fallow. On the
121 forested medium slopes, Planosols are formed by sandy loam deposits several tens of centimetres
122 deep, rich in “sables de Fontainebleau” and in coarse fragments, overlying silty loamy deposits (SMU
123 33, Figure 1 and Table 1). Finally, Gleysols are encountered in the valleys (SMU 102, Figure 1 and
124 Table 1).

125 Overall, the five studied soils show very contrasted particle-size distribution, organic carbon (OC)
126 content, or cation exchange capacity (Table 2). These properties are sometimes considered as key
127 determinants of the levels of several soil-supported ES (Adhikari and Hartemink, 2016) and hence are
128 frequently used as proxies to assess, for example, the regulation of climate (Eigenbrod *et al.*, 2011;
129 Clec’h *et al.*, 2016) or the regulation of nutrients and contaminants (Calzolari *et al.*, 2016). A high
130 variability in the levels of such soil-supported services may thus be expected over the Saclay plateau,
131 which makes it very suitable to assess the ability of empirical and process-based modelling to
132 translate variability in soil properties into differences in the levels of soil-supported ES.

133

134 II.2. Quantification of soil-supported ES

135 II.2.1. Land use and management homogenization

136 No single empirical or process-based model encompassing a realistic representation of soils
137 processes and properties could be identified to model soil-supported ES across the Saclay plateau.
138 Models developed in recent years to quantify, map, and in some cases monetarily value ES are
139 generally based on lookup tables linking land cover and/or land uses to mean service values (Bagstad
140 *et al.*, 2013; Sharps *et al.*, 2017). They tend to be of little use to translate gradients in soil properties
141 into differences in levels of soil-supported ES. Dynamic global vegetation models are designed to
142 model primary production or water, carbon, and nitrogen cycles across land uses and may thus
143 appear as interesting candidates. These models, however, rely on very simplified and outdated
144 representations of soils properties and processes (Vereecken *et al.*, 2016; Dai *et al.*, 2019; Pontgraz
145 *et al.*, 2019). Most of the models involving a more realistic representation of soil processes and
146 properties are either focused on the modelling of crop growth in cultivated soils (e.g., STICS; Brisson
147 *et al.*, 2003) or on the modelling of specific soil processes like soil water (e.g. HYDRUS-1D; Simunek *et al.*,
148 2008) or soil carbon dynamic (e.g. ROTH-C; Coleman *et al.*, 1997). Whereas crop models may be
149 useful to assess multiple ES in cultivated soils (Dominati *et al.*, 2014, 2016; Demestihis *et al.*, 2018;
150 Ellili-Bargaoui *et al.*, 2021), they cannot deal with natural or forest vegetation. Contrastingly, specific
151 models are used over various land uses but for just a few ES (Francesconi *et al.*, 2016). Assessing and
152 mapping soil-supported ES over the Saclay plateau would thus require the use of numerous models,
153 when such models even exist, which is not systematically the case for forested soils.

154 To limit the risk of inconsistencies due to the use of different models over contrasted land uses, soil-
155 supported ES were assessed in each soil under a similar succession of winter rapeseed and soft
156 winter wheat including those found in the natural and forested areas (SMU 33, 102, and to a lower
157 extent 31). The assessment and the mapping of ES across a complex landscape consequently turn
158 into a more common land evaluation exercise, a topic that has been dealt with in detail in the
159 literature (Dominati *et al.* 2016). Large numbers of tools and models exist in that context, specifically
160 developed for cultivated soils.

161 The succession of soft winter wheat following winter rapeseed was chosen because winter wheat is
162 the main crop on the Saclay Plateau (Tedesco *et al.*, 2017) and more largely in France
163 (FranceAgriMer, 2012), and because rapeseed is a very common previous crop for winter wheat
164 (FranceAgriMer, 2012). The winter wheat and rapeseed crop management is described in Table 3.
165 The total N inputs, supposed to be exclusively in mineral form, were assumed to be, respectively, 200
166 kgN ha⁻¹ and 168 kgN ha⁻¹ for winter rapeseed and soft winter wheat. The direct measurements of
167 Chalhoub *et al.*, (2020) were performed on the 2017-2018 winter wheat crop cycle with a similar
168 crop management but slightly higher N inputs (251.3 kgN ha⁻¹).

169

170 II.2.2. Selection of services and metrics

171 The selection of services was limited to soil-supported ES that are relevant on the Saclay plateau and
172 for which both empirical and process-based models, able to quantify the chosen indicators as a
173 function of basic soil properties, are available. As a result of these limitations, four services were
174 selected (Table 3): 1) “plant biomass provisioning”, 2) “water provisioning”, 3) “water quality
175 regulation”, and finally 4) “climate regulation”. Despite being limited, the selection of ES that was
176 made still manifests appreciable diversity (Greiner *et al.*, 2017; Grêt-Regamey *et al.*, 2017) since it
177 includes provisioning and regulating services as well as biotic (biomass provisioning), abiotic (water
178 provisioning) or mixed (water quality and climate regulation) services.

179 Given the simplified and homogenized land use we consider, the provision of plant biomass is
180 defined as the ecological contribution to the growth of cultivated crops that can be harvested and
181 used for food, non-nutritional, or energy production (Haines-Young and Potschin, 2018). It is
182 classically quantified through annual yields (Remme *et al.*, 2014; Ungaro *et al.*, 2014; Crouzat *et al.*,
183 2015; Rinot *et al.*, 2019). Crop yield however clearly involves various capital and labor inputs so that
184 its use as an indicator is controversial (Boyd and Banzhaf, 2007; Remme *et al.*, 2014). Annual yields
185 were nevertheless selected as the metric for the provision of plant biomass for a number of reasons.
186 First, it is practically difficult (Haines-Young and Potschin, 2018), if not impossible (Barot *et al.*, 2017;
187 Costanza *et al.*, 2017), to disaggregate the contributions that the ecological and economic production
188 systems make to cultivated biomass provisioning ES. Furthermore, when implicitly in empirical-, or
189 explicitly in process-based modelling, agricultural practices are kept constant, it seems reasonable to
190 consider that the variability of yields among soils may be used as a measure of the variability of the
191 soil contribution to plant biomass provision. Finally, the development of empirical and process-based
192 models dedicated to the quantification of the soil contribution to yields has a long history in soil
193 science.

194 Water provisioning was defined following Therond *et al.* (2017) and Haines-Young and Potschin
195 (2018) as the water that is routed to surface water or groundwater bodies through lateral surface
196 and subsurface flows or through deep infiltration. This corresponds to the water that is not directly
197 transpired by the vegetation but remains available for drinking or non-drinking purposes, even if it
198 may be at some distant location and at a much later time. The water yield defined as the difference
199 between precipitation and evapotranspiration is a classical indicator for this service (Sharps *et al.*, *et*
200 *al.*, 2017; Therond *et al.*, 2017). However, this indicator does not identify the fractions of the water
201 that end up in the groundwater or in surface waters. Here, we opted to rely on the quantity of water
202 that infiltrates through the soil until a depth of one meter and the amount of runoff to distinguish
203 the soil contribution to services specifically provided by groundwater and to services specifically

204 provided by surface water (O'Farrell *et al.*, 2010). As almost no runoff was observed over the Saclay
205 plateau, only the infiltration until a depth of one meter was computed and used as indicator for
206 water provisioning (Table 4). It is however likely that only a small part of the amount of water
207 infiltrated at one meter depth participated to the groundwater recharge for various reasons
208 including redirections of the water flows above the "argiles à Meulières" layer or interception of the
209 water flow by artificial subsurface drainage networks (Chalhoub *et al.*, 2019).

210 The regulation of water quality was defined as the ability of soils to filtrate or degrade contaminants
211 and nutrients (Haines-Young and Potschin, 2018). The application of synthetic fertilizers on
212 agricultural fields results in large inputs of nitrogen (N), some of which undoubtedly ending up in the
213 groundwater (Tedesco *et al.*, 2017). It makes sense to concentrate on the water quality aspects
214 linked to N. In that context, the regulation of water quality was quantified by the amount of non-
215 leached N (Table 4), i.e., the N used by crops, stored in the soil or volatilized (Dominati *et al.*, 2014;
216 Therond *et al.*, 2017).

217 The contribution of soils to the climate regulation was defined by the reduction of greenhouse gas
218 concentrations in the atmosphere (Haines-Young and Potschin, 2018) as a result of OC storage in soil,
219 as well as the reduction of emissions of N₂O and CH₄ from soils. It is quantified by OC storage in soils
220 (0-30 cm depth) according to Dominati *et al.* (2014) or Therond *et al.* (2017) rather than by OC stocks
221 as usually done, e.g., by Ungaro *et al.* (2010), in order to be closer to the service definition.
222 Attenuation of N₂O and CH₄ emissions have not been integrated in a global balance of greenhouse
223 gas emissions due to a lack of appropriate empirical or process-based models to quantify it.

224

225 II.2.3. Quantification by empirical modelling

226 Empirical or statistical models tend to be strongly biased toward the pedoclimatic context for which
227 they were developed or calibrated, and their efficiency rapidly drops outside of this particular
228 context (Van Looy *et al.*, 2017). This is particularly critical for models developed for the traditional
229 assessment of soil capability that postulate rather than explicitly integrate climatic or land use and
230 management conditions. To increase the assessment accuracy, the choice of models is then limited
231 to the ones that have been developed, calibrated and, when possible, successfully tested for rainfed
232 winter wheat cropping with high levels of management in temperate zones and for pedological
233 conditions as close as possible to those observed on the Saclay Plateau.

234

235 II.2.3.1. Provision of plant biomass

236 The Muencheberg Soil Quality Rating (M-SQR) expert-based tool was selected (Table 4) to assess the
237 soil's suitability to grow wheat using soil descriptive and analytical properties (Mueller *et al.*, 2007).
238 This tool was tested favorably in various pedoclimatic contexts and for many grain crops (Mueller *et*

239 *al.*, 2010, 2013) including rainfed winter wheat in temperate zones (Abdollahi *et al.*, 2015). It
240 generates scores between 1 and 100 from the values of eight basic indicators, namely soil substrate,
241 depth of the A-horizon, topsoil structure, subsoil compaction, rooting depth, profile available water,
242 wetness and ponding, and finally slope and relief. Single basic indicators are evaluated using
243 matching tables, empirically weighted, and summed to obtain the M-SQR basic score. Although it is
244 possible to add hazard indicators, only the basic indicators were used in this study.

245

246 II.2.3.2. Provision of water

247 Only one model, called SENSIB in Table 4 according to Aveline *et al.* (2009), developed for soils of the
248 great Paris Basin could be found (Cam *et al.*, 1996). Similarly to the M-SQR, SENSIB assesses the soil
249 sensitivity to infiltration by scoring and summing six criteria (i.e., topsoil texture, presence of an
250 impermeable layer, soil depth, profile available water, soil profile permeability, and velocity of water
251 flow), through lookup tables based on nine descriptive or analytical soil properties. This model has
252 not been formally assessed and is not widely known. Cam *et al.* (1996) however tested it on various
253 soil types and demonstrated that the indicator provides coherent distinctions between soils with low,
254 moderate, and high capacity for deep infiltration. One weakness of this expert-based scoring is that it
255 relies on soil properties alone but does not take into account climatic or vegetation characteristics
256 (Aveline *et al.*, 2009). This does not particularly matter in this study as empirical modelling is carried
257 out under climatic conditions over the Saclay plateau that are both spatially homogeneous and
258 roughly similar to those considered in the development of Cam *et al.*'s (1996) model. This model
259 however assesses the soil inherent capability to contribute to service provision rather than the soil
260 performance under a specific use.

261

262 II.2.3.3. Regulation of water quality

263 The regulation of water quality is assessed using the composite expert-based model MERLIN v2
264 (Aimon-Marie *et al.*, 2001) developed to evaluate the risk of nitrate leaching, given that no empirical
265 model able to assess the proportion of the applied and produced N retained in the soil or the soil-
266 plant system could be found. MERLIN has been chosen because it was developed for an area of the
267 great Paris Basin similar to the Saclay site from a pedological point of view. MERLIN is easy to use,
268 and its suitability has been successfully tested over a wide range of soil types and crops, including
269 155 cases of wheat cultivation around Paris (Aveline *et al.*, 2009). MERLIN relies on the aggregation
270 of three sub-indicators that evaluate respectively the soil capacity for deep infiltration according to
271 Cam *et al.* (1996), the N balance between plant requirements and N inputs, and finally the N uptake
272 capacity of the potential soil cover before the beginning of the drainage period (Aimon-Marie *et al.*,

273 2001). As MERLIN was designed to assess the risk of leaching rather than the level of non-leached N,
274 the three classes, namely low, intermediate, and high risk of leaching were inverted to obtain three
275 classes of non-leached N: sufficient (value 3), insufficient (value 2), and highly insufficient (value 1).
276 Whereas MERLIN takes land use, land management, and level of inputs explicitly into account, this is
277 not the case for climatic characteristics.

278

279 II.2.3.4. Regulation of climate

280 The capacity of soils to store OC was assessed through the “carbon saturation” approach (Table 4),
281 which estimates the OC pool that is associated with the fine soil fraction and can consequently be
282 protected for a sufficiently long residence time to act positively on climate mitigation (Hassink,
283 1997). Although criticized (e.g., Barré *et al.*, 2017), this approach was preferred to data-driven
284 approaches because it proved successful when applied to French agricultural topsoils (Angers *et al.*,
285 2011) and because the latter require a large and generally unavailable amount of measured OC
286 stocks (Calzolari *et al.*, 2016; Barré *et al.*, 2017) . The OC saturation, i.e., the inherent maximum stock
287 of stable OC, was calculated according to the following equation (Hassink, 1997):

$$288 \quad C_{\text{sat}} = 4.09 + 0.37(\text{Clay} + \text{fSilt}) \quad (1)$$

289 where C_{sat} is the C saturation expressed in gC kg^{-1} ; Clay and fSilt are respectively the clay (0–2 μm
290 particles) and the fine silt (2–20 μm particles) content in $\text{g } 100\text{g}^{-1}$.

291

292 II.2.4. Quantification by process-based modelling

293 Yields, water flows at one meter-depth, non-leached N, and finally OC storage (0-30cm) were
294 modelled using the CERES-EGC crop model (Gabrielle *et al.*, 2006) This model was chosen because it
295 has been tested favorably for various pedoclimatic and agricultural conditions, as well as extensively
296 used for crop modelling in temperate climate and for soils similar to those of the Saclay plateau.

297 CERES-EGC runs on a daily time step and is composed of three main modules. A physical module
298 simulates the transfer of heat, water, and solutes in soils as well as plant water uptake and
299 evapotranspiration in relation with climatic conditions. Water infiltrates down the soil profile
300 following a tipping-bucket approach and may be redistributed upwards after evapotranspiration has
301 dried some soil layers. The soil OC turnover is modelled in the plough layer using the sub-model
302 NCSOIL that comprises six soil OC pools with specific turnover rates: pool 0, the “zymogenous
303 biomass”; pool I, the “microbial biomass”; pool II, the “mineralisable soil OC”; pool III, “highly-
304 humified OC”; pool “Exogenous OC,” and pool plant residues (Noirot-Cosson, 2016). Finally, crop
305 growth and development are modelled through net photosynthesis. A more precise description of

306 CERES-EGC and of NCSOIL may be found in Jones *et al.*, (1986), Gabrielle *et al.* (1995, 2006), and
307 Noirot-Cosson *et al.* (2016).

308 Model inputs include soil properties and external drivers such as climate and management practices.
309 Soils are described over a one meter-depth as a superposition of layers of roughly ten centimeters
310 following the soil horizonation (Table 1). Each of the soil layers is characterized by its particle-size
311 distribution, bulk density, pH, C and total N contents, as well as C/N ratios (Table 2), from which
312 water retention and hydraulic conductivity characteristics are computed by means of pedo-transfer
313 functions (Supplementary Material S2). Roots are distributed throughout the profile depth
314 proportionally to root density (Table 1). Finally, CERES-EGC was a priori parametrized according to
315 Gabrielle *et al.* (1995).

316 In order to test CERES-EGC, the four studied ecosystem services were first computed in a thick Luvisol
317 over loess deposit (SMU 29) for the winter wheat crop cycle 2017-2018 and compared with the direct
318 measurements of Chalhoub *et al.* (2020) performed in a similar, artificially drained soil, for the same
319 crop and the same crop cycle 2017-2018 and for a very similar crop management. The water
320 artificially redirected to nearby streams in this particular situation was assumed to be a good
321 estimate for the amount of water infiltrated at one meter-depth in undrained soils as the subsurface
322 drains were implanted at one meter-depth in the plot studied by Chalhoub *et al.* (2020). CERES-EGC
323 was subsequently run for twenty consecutive years (1998-2018) using climate data from a nearby
324 weather station (1 km away from the experimental plot). Quantitative yearly outputs computed from
325 daily soil water and nutrient budget were averaged on the ten modelled winter wheat crop cycles
326 and used to quantify the services of plant biomass provisioning, water provisioning, and water
327 quality regulation. OC fluxes were cumulated over the twenty years of modelling to quantify the
328 service of climate regulation (Table 4).

329

330 II.3. Spatial co-occurrence of soil-supported ES

331 The spatial dimension of soil-supported ES delivery resulting from empirical modelling on the one
332 hand and from process-based modelling on the other hand is analyzed through a static spatial co-
333 occurrence approach (Obiang Ndong *et al.*, 2020). Such an approach quantifies the spatial overlap of
334 multiple ES rather than the interactions between services in the sense that the drivers and
335 mechanisms underlying the interactions are not explicitly examined (Obiang Ndong *et al.*, 2020). It is
336 however suitable for identifying areas with specific ES-related issues (e.g., the so-called hotspots and
337 coldspots), which is a common and important step in dealing with ES bundles (Mouchet *et al.*, 2014;
338 Obiang Ndong, 2020). The levels of each soil-supported service obtained respectively from empirical
339 and process-based modelling were first rescaled between 0 and 1. To do that, the minimum value is
340 subtracted from each level, and the result is then divided by the difference between the maximum

341 and the minimum values. The minima and maxima used to rescale the soil-supported services levels
342 (Table 5) were either the lowest and the highest scores of each specific tool (MSQR, SENSIB, MERLIN)
343 or the lowest and the highest levels observed in France. The scores for each service were eventually
344 summed to obtain a final score between 0 and 4. Such strategy implicitly gives the same weight to
345 the different services.

346

347 **III. Results**

348 **III.1. Provisioning services**

349 **III.1.1. Plant biomass**

350 According to the empirical model, the Luvisol on loess (SMU 29) has the highest potential yield
351 (Figure 2). This yield, classified as “good”, but not “very good”, according to the M-SQR (Mueller *et*
352 *al.*, 2007) is coherent with the harvested yield of 8.5 t ha⁻¹ in 2018 (Table 6) considering that wheat
353 yields frequently exceed 10 t ha⁻¹ in France (Table 5). The potential yield of this soil is indeed limited
354 by subsoil compaction, the low concentration of OC in the topsoil, and the periodic hydromorphy
355 below 60 cm (Tables 1 and 2). The potential yield is minimal in the Planosol on clay (SMU 31) and in
356 the Gleysol (SMU 102). In these two soils, potential yields are constrained by a strong hydromorphy
357 from the surface downward, by a strong textural gradient above 80 cm (deducted by the M-SQR) and
358 by the clayey texture of subsoil horizons limiting the rooting depth and the available water capacity
359 (Table 1). Finally, the high stoniness of the surface horizons (Table 1) further limits the potential yield
360 in the Planosol on clay. The Luvisol on clay (SMU 30) and the Planosol (SMU 33) show intermediate
361 potential yields (Figure 2) because of periodic hydromorphy occurring below 30 cm depth (Table 1).
362 As secondary reasons, the Luvisol on clay is sharp-edged and low in OC, and the Planosol has a
363 sandy surface horizon. On flat plateau positions, the M-SQR scores (Figure 2, Table 6) decrease from
364 the center (SMU 29) to the edge of the plateau as the thickness of the soil layers developed from
365 loess material decreases (SMU 30) and finally vanishes (SMU 31).

366 The level of plant biomass provision modelled with CERES-EGC shows a pattern that is very similar to
367 that estimated by empirical modelling (Figure 2). CERES-EGC yields are relatively high in the Luvisol
368 on loess and almost identical to the measured ones for the year 2018 (Figure 2). Yields decrease in
369 the Luvisol on clay and further in the Planosol on clay as the clayey layer approaches the surface and
370 limits the rooting depth (Table 1). The Planosol exhibits yields similar to those observed in for the
371 Luvisol on loess, despite the sandy texture of its surface horizons and the periodic hydromorphy
372 observed above 50 cm depth. Finally, very low yields are modelled with CERES-EGC in the Gleysol.

373 When rescaled from 0 to 1, the yields tend to be lower for the process-based modelling than for the
374 empirical modelling except in the two Planosols (SMU 31 and 33) where similar yields are observed

375 (Figure 2). Yields in the two Planosols are potentially underestimated by the empirical modelling,
376 overestimated by the process-based modelling, or both.

377

378 III.1.2. Blue water

379 The water layer infiltrated at one meter depth, estimated with the empirical model, is arbitrarily high
380 (value 85 in Table 6) in the Gleysol, due to the proximity of the associated alluvial groundwater in the
381 valley (Figure 3). The infiltration is also high for the Planosol (SMU 33), characterized by its low
382 retention due to a mostly sandy texture (Table 2). The slight decrease of infiltration from the
383 Planosol to the Luvisol on loess is consistent with higher water retention as a result of a silty to silty-
384 clay texture. Finally, the low amounts of infiltrated water in the Luvisol on clay and the Planosol on
385 clay are coherent with the presence of clayey deep soil horizons (Table 2), which act like a hydraulic
386 barrier and foster lateral runoffs to the detriment of deep infiltration. The depth at which this barrier
387 appears (80 cm in the Luvisol on clay and 60 cm in the Planosol on clay, Table 2) seems to have no
388 effect on the amount of infiltrated water (Figure 3).

389 The water layer modeled with the process-based model CERES-EGC in 2018 is similar to the
390 measured one (Figure 3). Process-based modeling estimates that a similar quantity of water drains
391 from all of the studied soils (Figure 3), despite the hydraulic barrier in the Luvisol on clay and in the
392 Planosol on clay (but not in other soils), or despite very contrasted soil textures, either sandy in the
393 Planosol, silty in the Luvisol on loess or clayey in the Gleysol.

394 When rescaled from 0 to 1, the empirical model output gives higher levels of water provisioning than
395 does the process-based modelling (Figure 3). The differences are particularly pronounced in the
396 Gleysol, in the Planosol and in the Luvisol on loess.

397

398 III.2. Regulating services

399 III.2.1. Water quality

400 According to the empirical model MERLIN, all soils offer a highly insufficient level of non-leached N
401 (value 1 in Table 6), i.e., a null value when rescaled from 0 to 1 (Figure 4). Indeed, significant amounts
402 of N remain available for leaching as a result of i) N inputs exceeding real crop requirements, and ii)
403 the absence of intermediate crops that are likely to trap mineralized N during the Summer or the
404 beginning of the Fall.

405 In good agreement with empirical modelling, process-based modelling also shows that N retention is
406 insufficient to entirely avoid N leaching from all the studied soils (Table 6, Figure 4). However, with
407 non-leached N, ranging from a minimum of 80% in the Planosol to more than 90% of total N inputs in
408 the four other soils (Table 6), the process-based modelling gives a relatively high level of water
409 quality regulation for all soils when rescaled from 0 to 1 (Figure 4). In the Luvisol on loess, the level of

410 the modelled non-leached N is very close to the measured one (Figure 4). With reference to the
411 Luvisol on loess, the service of water quality regulation is only slightly higher in the Luvisol on clay, in
412 the Planosol on clay, and in the Gleysol but it drops in the Planosol, where this service is minimal.
413 This last soil shows a similar amount of infiltrating water but also the highest soil OC content (Table
414 2). Mineralization then produces higher amounts of N available for leaching than in the Luvisol on
415 loess, relatively poor in OC.

416

417 III.2.2. Global Climate

418 When quantified with the empirical model of Hassink *et al.* (1997), the service of global climate
419 regulation is relatively high in the clayey topsoil of the Gleysol, intermediate in the silt-loamy topsoils
420 of the Luvisol on loess, the Luvisol on clay and the Planosol on clay and finally low in the sandy
421 topsoil of the Planosol (Figure 5).

422 All of the five studied soils experience significant OC storage as a result of 20 years of a simplified
423 rapeseed/wheat crop rotation without any other organic amendments than rapeseed residue returns
424 (Figure 5, Table 6). The modelled OC storage is maximum in the currently (SMU 29 and 30) and
425 formerly (SMU 31) cultivated soils showing topsoil OC contents lower than one percent (Table 2). The
426 mean annual OC storage slightly decreases in the Gleysol and further decreases in the Planosol: the
427 two forested soils (Table 1) with topsoil OC contents reaching respectively 2 and 5 percent (Table 2).
428 When rescaled from 0 to 1, the levels of services quantified with the process-based model are higher
429 than those quantified with the empirical model except in the Gleysol, in which the level of service is
430 independent of the model used.

431

432 III.3. Spatial co-occurrence of soil-supported services

433 The soils of the Saclay plateau show a joint supply of provisioning and regulating services above the
434 mean, except in the Planosol (Figure 6b), when the assessment is performed with the process-based
435 model. The same soils however exhibit a joint soil services supply that is below average when
436 estimated through empirical modelling (Figure 6a). Under the simplified crop rotation, the studied
437 soils may thus be considered as relatively good to relatively poor services providers depending on the
438 methodology used. The higher scores obtained with the process-based modelling appear to be
439 related to the very high scores obtained for regulating services (Figures 4 and 5).

440 Soil ranking by ES level varies whether empirical or process-based modelling is performed. Indeed,
441 the joint supply of ES decreases in the order Gleysol (SMU 102), Luvisol on loess (SMU 29), Luvisol on
442 clay (SMU 30), Planosol (SMU 33) and finally Planosol on clay (SMU 31) according to the empirical
443 modelling. At the same time, it decreases in the order Luvisol on loess (SMU 29), Luvisol on clay (SMU
444 30), Planosol on clay (SMU 31), Gleysol (SMU 102) and finally Planosol (SMU 33) according to the

445 process-based modelling. The best supplier of services under crop is the Luvisol on loess according to
446 empirical modelling but the Gleysol according to process-based modelling. Furthermore, the Luvisol
447 on clay and the Planosol on clay are respectively ranked among the first and the last ES suppliers,
448 respectively by the process-based and the empirical modellings.

449

450 **IV. Discussion**

451 IV.1. Performance of empirical models

452 The M-SQR empirical model successfully distinguished currently forested soils like the Gleysol and
453 the Planosol with yield potentials too low to have been (recently) cultivated, or soils that have been
454 abandoned in the past years like the Planosol on clay, from the two currently cultivated Luvisols
455 (SMU 29 and 30). The empirical model appears moreover sufficiently sensitive to quantify small
456 changes in yield potential between various cultivated soils. An example is the decrease in yield
457 potential from the Luvisol on loess to the Luvisol on clay resulting from a decreasing rooting depth
458 and an increasing intensity of hydromorphy. Finally, when calibrated with reference to yields
459 commonly observed in cultivated soils in France (Table 5), the yield estimated by the M-SQR
460 empirical model is satisfactorily close to the measured one (Figure 2). Since the M-SQR is able to
461 assess potential yields not only relatively among various soils but also absolutely, it appears to be an
462 efficient tool to quantify the biomass provisioning ES, as already suggested by Mueller *et al.* (2010,
463 2013). Its efficiency may be explained in two different ways. First, the development of an empirical
464 model dedicated to the assessment of the soil suitability to yield main cereal crops under conditions
465 of high-level management from inherent soil properties relies on a rich and long history dating back
466 at least to the 30s with the development of the Storie Index (Storie, 1978). Moreover, in conditions
467 of high-level management without irrigation, potential yields are chiefly influenced by the soil
468 available water capacity, which may be reasonably well assessed from classically available inherent
469 physicochemical soil properties (Van Looy *et al.*, 2017).

470 Regarding water provisioning, the empirical assessment is consistent with what could be
471 expected from the soil pedological class and characteristics. It limits deep infiltration in case of i) high
472 water retention in deep fine-textured soils like the silt-loam to silty-clay loam Luvisol on loess or ii)
473 lateral water flows promoted by the more or less impermeable clay bed observed in the Luvisol on
474 clay and in the Planosol on clay, as typically described in Planosols (IUSS Working Group WRB, 2015).
475 The empirical model considers implicitly that at least one part of such lateral water flows is
476 redistributed towards the valley bottoms where it infiltrates preferentially at the landscape level.
477 This justifies the high level of water provisioning of the valley soils (here the Gleysol). In such 1D
478 model, the amount of water redistributed at the landscape scale is however only arbitrary but not
479 quantitatively based (Cam *et al.*, 1996). These empirical assessments are moreover independent of

480 climate characteristics (Aveline *et al.*, 2009). When used over the Paris basin, the empirical scores
481 consequently range from no infiltration (value of 0) to infiltration of all the available water (value of
482 100). The maximum amount of water available in the Luvisol on loess defined by the difference
483 between precipitations (around 625 mm) and evapotranspiration (around 450 mm) was in the range
484 of 200 mm for the crop cycle 2017-2018 (Chalhoub *et al.*, 2020). Inversely, the measured and
485 process-based modelled infiltration are rescaled in Figure 3 for a considerably larger range of climate
486 and infiltration as observed in France with a maximum of 1,000 mm per year observed in sandy soils
487 under mountainous climate (Table 5). Such discrepancy induces the systematic overestimation of the
488 absolute level of water provisioning estimated empirically compared to the process-based modelling
489 outcomes (Figure 3). Although the empirical scoring of water provisioning must not be interpreted
490 absolutely due to some arbitrary scores and to its implicit climatic dependency, it might however be
491 used for the relative comparison of soils under similar climatic conditions.

492 Field measurements (Chalhoub *et al.*, 2020) as well as the predictions of the empirical and process-
493 based models agreed on the fact that the level of water quality regulation was insufficient to
494 completely avoid N leaching from all of the studied soils. This observation suggests that MERLIN is at
495 least effective in identifying such situations, as demonstrated by Aveline *et al.* (2009). However,
496 MERLIN does not quantify differences in the regulation of water quality among the five studied soils
497 (Figure 4). The Luvisol on loess presents a higher yield potential so higher N consumption by crops
498 than in the Gleysol (Figure 2). On the contrary, lower infiltration is observed in the Luvisol on loess
499 than in the Gleysol (Figure 3). Consequently, for similar N inputs, the Luvisol on loess should logically
500 offer a higher service, using N to grow crops and with a minimal N transport towards the
501 groundwater, than the Gleysol. Since this tool has been created to assess the risk of nitrate leaching
502 in order to protect water resources (Aveline *et al.*, 2009), it seems logical that MERLIN is more
503 sensitive to the excess of the capacity of the soil-plant system to regulate water quality, than to the
504 magnitude of this excess.

505 By construction, C saturation (Hassink, 1997) is proportional to the content in clay plus fine silt-
506 sized particles. As a result, soils with finer texture are supposed to show a higher contribution to
507 global climate regulation (Figure 5). In a certain way, this agrees with measured OC stocks (data not
508 shown) except in the Planosol, which has a higher OC stock than the two Luvisols (SMU 29 and 30)
509 and the Planosol on clay. The topsoil OC contents (Table 2) in the silty-clay Luvisol on loess, Luvisol on
510 clay and Planosol as well as in the clayey Gleysol, are lower than the saturation limit (Table 6),
511 suggesting that these soils are far from saturation. Intensely cultivated soils such as the two Luvisols
512 are indeed commonly unsaturated (Angers *et al.*, 2011). By contrast, it is unlikely that OC content
513 further increases in the Gleysol that is already forested and periodically waterlogged at the surface
514 (Table 1). Carbon saturation is indeed rarely reached in fine-textured soils (Angers *et al.*, 2011),

515 suggesting that OC saturation, and thus, the level of global climate regulation, are likely
516 overestimated in such clayey soils. Finally, the OC content in the topsoil of the forested and sandy
517 Planosol is considerably higher than the saturation limit (Tables 2 and 6). According to the estimation
518 of the stable OC proposed by Angers *et al.* (2011), the Planosol is already oversaturated in carbon, as
519 it is frequently observed for acid sandy soils (Angers *et al.*, 2011). The Planosol is consequently more
520 prone to carbon emissions than to additional carbon sequestration. Although OC saturation is very
521 easy to use and seems helpful to rank various soils according to their inherent capability of climate
522 regulation, it may hide considerable differences in current OC stocks and in additional soil OC
523 sequestration potential as a result of contrasted land uses or agricultural practices among soils
524 (Wiesmeier *et al.*, 2014; Chen *et al.*, 2018). The carbon saturation deficit, calculated as the difference
525 between the intrinsic soil potential to stabilize OC and the actual stocks of OC in the fine soil fraction,
526 should better reflect the legacy of land-uses and agricultural practices (Chen *et al.*, 2018).
527 Information about current OC stocks in the fine soil fractions is unfortunately poorly available at this
528 stage (Chen *et al.*, 2018).

529 To conclude this section, only the contribution of soils to biomass provision and its variability among
530 soils can be quantified absolutely using empirical modelling with a relatively reasonable level of
531 confidence. The quantification of the provision of water and of the regulation of global climate is
532 indeed only relative, which may nevertheless be sufficient to rank various soils according to potential
533 service levels, as postulated by Greiner *et al.* (2017) for empirical modelling. Finally, the assessment
534 of the contribution of soils to the regulation of the water quality is limited to the identification of
535 soils with an “insufficient” level of service under cropping with relatively high levels of N inputs.

536

537 IV.2. Performance of the process-based model

538 The process-based model was able to quantify with a very good accuracy the provision of biomass
539 and water, as well as the regulation of water quality (Figures 2, 3 and 4) in the cultivated, deep, freely
540 drained, and silty Luvisol on loess, i.e., a soil similar to those for which CERES-EGC was developed and
541 calibrated (Gabrielle *et al.*, 1995). The OC storage (Figure 5) cannot be compared to experimental
542 data as OC content was not measured in the field twenty years ago. The OC storage in intensely
543 cultivated soils may be surprising considering the very low OC contents currently observed in the
544 surface soil horizons of the two cultivated Luvisols (Table 2). It is however consistent with OC budgets
545 driven by i) relatively high yields leading to high inputs of crop residues and notably of rapeseed
546 residues that are not exported (Table 3) and by ii) low carbon emission as mineralization is
547 proportional to the very low OC content observed in the Luvisol on loess. Although potentially far
548 from the field truth, associated with a more complex and diverse crop rotation, it is likely that
549 modelled OC storage is consistent with our oversimplified crop rotation (Noirot-Cosson *et al.*, 2016).

550 The efficiency of CERES-EGC to quantify soil services in the Luvisol on loess is likely shared by the
551 Luvisol on clay. The two Luvisols (SMU 29 and 30) are indeed morphologically and physico-chemically
552 very similar (Tables 1 and 2), and show very closed contributions for the different soil services
553 (Figures 2, 3, 4, 5).

554 Yields in the Planosol on clay, in the Planosol, and in the Gleysol are mainly limited by
555 waterlogging (Table 1). If process-based modelled yields are lower in the Planosol on clay and in the
556 Gleysol than in the Luvisol on loess, the Planosol contrastingly shows similarly high yields (Figure 2).
557 The Planosol on clay and the Gleysol indeed suffer from a poor water available capacity due to both
558 shallow rooting depth and to clayey subsoil horizons (Table 1). This is however not the case in the
559 Planosol (SMU 33). CERES-EGC yields seem thus limited by insufficient water availability rather than
560 by direct retroactions between soil waterlogging and the growth or functioning of crop roots. The
561 decreasing yields with increasing waterlogging observed in the Planosol on clay and in the Gleysol are
562 hence mostly incidental but it does not have to be that way as shown by the Planosol.

563 The clay bed observed in the Planosol on clay does not result in a decrease in the amount of water
564 infiltrated at one meter-depth (Figure 3) as expected from soil morphology. CERES-EGC does indeed
565 not model lateral subsurface water flow (Gabrielle *et al.*, 1995). The hydraulic conductivity is
566 moreover averaged from 30 to 100 cm-depth mixing artificially clayey horizon with low hydraulic
567 conductivity with silty horizon of high hydraulic conductivity and overestimating the subsoil hydraulic
568 conductivity (Table 2, supplementary material S2). As a result, clayey subsoils appear able to merely
569 slow down but not to stop infiltration and redistribute water laterally. Therefore, the amount of
570 water infiltrated in the Planosol on clay, and more generally in strongly texture-differentiated soils
571 are overestimated with CERES-EGC. Contrastingly, as water does not accumulate in lowland soils as
572 the Gleysol (SMU 102), or the Planosol (SMU 33) to a lesser extent, the amount of infiltrated water in
573 such soils is conversely underestimated.

574 In comparison to Luvisols (SMU 29 and 30), the Planosol stores less carbon (Figure 4) for similar
575 yields (Figure 2) likely as a result of higher CO₂ emissions due to higher OC contents. The Gleysol
576 contrastingly stores more carbon despite lower yields as a result of CO₂ emissions highly limited by its
577 clayey texture. The variability in OC storage (Figure 5) seems, at first sight, consistent with the spatial
578 heterogeneity of soil properties. It is however very unlikely that the Planosol and the Gleysol,
579 currently forested, continue to store carbon when cultivated, as cultivation is known to induce
580 decreases in carbon stocks (Balesdent *et al.*, 1998). OC budgets are moreover strongly dependent on
581 OC inputs and consequently on yields estimates that were unfortunately found to be highly
582 questionable at least in the Planosol and Gleysol.

583 Estimates of N cycling and leaching are unlikely to be very reliable given the difficulties encountered
584 in the assessment of yields, infiltration, as well as soil organic matter mineralization in the Planosol
585 on clay, in the Planosol and in the Gleysol.

586 To conclude this section, the process-based model is particularly effective to quantify services in
587 deep, homogeneous, and well-drained soils for which crop models have been developed and
588 calibrated. Crop models are unsurprisingly inadequate to quantify soil-supported services in marginal
589 soils as frequently observed under forest. For various reasons including intense waterlogging,
590 extreme acidity, or strong texture differentiation, such soils are indeed clearly out of the range of
591 validity of most crop models.

592

593 IV.3. Guidance in the assessment of soil-supported ES and potential uses

594 No single empirical or process-based model integrating a realistic representation of soils, for example
595 in terms of the depth-discretization of main inherent and manageable soil properties, could be
596 identified to quantify soil-supported ES for land uses ranging from forest to crop cultivation.
597 Integration of a realistic representation of soils in soil-supported ES assessment and mapping is thus
598 only possible with reference to a cultivated use and for a limited set of crops and cropping practices.
599 Such an assessment, which is basically a quantitative land suitability exercise, goes however largely
600 beyond classical land evaluation (Dominati *et al.*, 2016). Indeed, it provides quantitative rather than
601 qualitative estimates of soil performance under cultivation and it addresses not only food
602 provisioning, as is classically the case, but multiple soil-supported ES. It is thus an interesting way to
603 balance biomass provisioning with other soil-supported ES under a particular use, chosen for its
604 representativeness or its interest for stakeholders.

605 With reference to the common agricultural practices considered in this research, the levels of
606 the soil-supported ES vary significantly according to soil types and properties (Figures 2 to 5). Part of
607 this variability in ES levels is coherent with the present variability in land uses and it may then be
608 captured by a land use/land cover assessment approach. This is for example the case of the
609 cultivated Luvisols that unsurprisingly show a higher level of biomass production than the forested
610 Planosol and Gleysol (Figure 2). A significant variability is however observed between soils with a
611 similar land use (Figure 6). For instance, OC saturation ranges from less than 10 g kg⁻¹ in the forested
612 Planosol to about 30 g kg⁻¹ in the forested Gleysol. The biomass provision in presently- or recently-
613 cultivated soils similarly ranges from good levels (M-SQR scores between 60 and 80) to poor levels
614 (M-SQR scores between 20 and 40). This clearly demonstrates that the explicit integration of soil
615 processes and properties within the ES assessment and the translation of the spatial variability of soil
616 properties into levels of soil-supported ES, will increase our knowledge on the spatial delivery of soil-

617 supported ES inside patches with similar land cover and land use, and decrease the uncertainty
618 associated with purely land use/land cover assessment methods (Roche and Campagne, 2019).
619 For the four-soil supported ES considered in this study, process-based models seem to be the best
620 solution for arable soils. Such an approach, limited to cultivated soils for which process-based models
621 have been historically developed, has been implemented for example by Therond *et al.* (2017) at the
622 French national scale, or by Ellili-Bargaoui *et al.* (2021) for 64 cultivated soils across the Brittany
623 region of France. It is particularly helpful to reveal the contribution of various combinations of land
624 management and soil types to multiple ES (Therond *et al.*, 2017; Ellili-Bargaoui *et al.*, 2021), to
625 quantify the interactions among soil-supported services (Ellili-Bargaoui *et al.*, 2021) or finally to
626 optimize land management for specific or multiple outcomes identified by stakeholders (Dominati *et*
627 *al.*, 2016; Demestihis *et al.*, 2018). However, approaches limited to cultivated soils that consider
628 forested and other marginal lands as “blank” areas, will miss the provision of ES by forested soils in
629 spite of the fact that such soils may be critical areas for the delivery of soil-supported ES, such as the
630 Gleysol in this study (Fig. 6), or may show high levels for a particular ES, like water provisioning in
631 SMU 33 (Fig.3). Moreover, further research is necessary to better characterize the range of cultivated
632 soils over which soil-supported services may be quantified by these models, and the associated
633 uncertainty, since all cultivated soils cannot be considered as deep, homogeneous, and well-drained.
634 By contrast, the empirical modelling approach appears at this stage to be the only one that may be
635 used for a wider variety of land uses and, hence, for the comparison of several land planning
636 scenarios. However, the availability of empirical models for different pedoclimatic conditions is very
637 uneven among the different ES: relatively high for the provision of biomass, low for water
638 provisioning, and finally almost nonexistent insofar as soil biological functioning is primarily
639 concerned as it is the case for C and N cycling. In such cases, the adaptation of empirical models not
640 specifically designed to quantify the targeted services, although informative, may be incompletely
641 satisfactory. The empirical model MERLIN, for example, is efficient at identifying situations with a
642 potential impact on the environment (i.e., leached N) in accordance with its initial design, but turns
643 out to be unable to accurately assess the amount of non-leached N, which is crucial for quantifying
644 regulating services.

645 The simultaneous use of a wide portfolio of models or model types chosen to cover all soil type/land
646 use combinations as well as the diversity of soil-supported ES should overcome the limitations
647 attached to each of the empirical and process-based models and reduce the uncertainty arising from
648 the use of one single type assessment. Such option, if feasible, would however inevitably increase i)
649 the need for input data and consequently the complexity of the assessment, and ii) the risk of
650 inconsistencies between the various models, as often observed in comparative studies (Bagstad *et*
651 *al.*, 2013; Vorstius and Spray, 2015; Sharps *et al.*, 2017). This latter point is confirmed here by the

652 significantly method-dependent assessment of the soil-supported services, despite all the efforts
653 made to propose two approaches differing only by their increasing complexity (Figure 6).
654 Finally, steps such as i) the construction of input data sets, ii) the selection of documented and
655 adapted empirical models for a particular pedoclimatic context or the appropriation and running of
656 more complex process-based models, and iii) the interpretation of service assessments considering
657 the limits and uncertainties embedded in each of the empirical or process-based approaches requires
658 significant technical, human and financial resources. Consequently, most local stakeholders are far
659 from managing an operational assessment of the contribution of soils to ES. Furthermore, every
660 assessment will ultimately remain strongly influenced by numerous methodological choices. If
661 sufficiently informed, such choices are helpful to increase the fit-for-purpose (Schroter *et al.*, 2015)
662 but, if not, they may have unexpected implications on ES levels, ES bundles (not explicitly addressed
663 in this study) and finally ES-based decision-making (Bagstad *et al.*, 2018). Besides testing the
664 potential outputs of ES assessment regardless of the methodology used, a similar attention at least
665 should be devoted to the continuous improvement of methodologies, for example by integrating a
666 more realistic representation of soil processes and properties and by comparing ES assessment with
667 field data at least in experimental stations.

668

669 V. Conclusion

670 The primary objective of the present article was to examine how a realistic representation of soil
671 properties and processes could be taken into account when assessing the levels of ES these soils
672 provide, and to appraise the degree of accuracy that can be achieved using only easily-available data
673 and existing models. One of the key results of the research described in this article is that we could
674 not find a single empirical or process-based model that could both integrate a realistic
675 representation of soils and account for land uses ranging from forest to cultivation in the
676 quantification of soil-supported ES. At this stage, integrating a complex representation of soils in the
677 assessment of soil-supported ES may only be performed with reference to a cultivated use and for a
678 limited set of crops and cropping practices, which is basically a quantitative land suitability exercise.
679 Such an assessment goes however largely beyond classical land evaluation. It is indeed an interesting
680 way to balance biomass provisioning with other soil-supported services under a particular use,
681 chosen for its representativeness or its interest for stakeholders.

682 To integrate a complex representation of soils and ultimately increase the accuracy in ES assessment
683 and mapping , we urge the research community to pursue its efforts towards i) acquiring
684 environmental data, from basic soil properties to field measurements of the soil-supported ES, ii)
685 developing empirical models specifically designed to assess soil functioning in various pedological
686 contexts, and more specifically the geopedological controls on C and N cycling, and iii) integrating in

687 crop or land surface models a more precise soil discretization in depth, a greater diversity of soil
688 processes including lateral (subsurface) water flows, waterlogging and more complex interactions
689 between soil properties and soil processes like mineralization or root growth. The ability of both the
690 empirical and process-based models to assess the biomass provision from basic soil properties clearly
691 suggests that such a path is feasible, efficient, and probably the only way to operationalize the ES
692 concept.

693

694

695 **References**

696 Abdollahi, L., Hansen, E.M., Rickson, R.J., Munkholm, L.J., 2015. Overall assessment of soil quality on
697 humid sandy loams: Effects of location, rotation and tillage. *Soil Tillage Res.* 145, 29–36.
698 <https://doi.org/10.1016/j.still.2014.08.009>

699 Adhikari, K., Hartemink, A.E., 2016. Linking soils to ecosystem services — A global review. *Geoderma*
700 262, 101–111. <https://doi.org/10.1016/j.geoderma.2015.08.009>

701 Agreste, 2006. ADEME - Site Bilans GES [WWW Document]. URL [http://www.bilans-
702 ges.ademe.fr/fr/accueil/documentation-gene/index/page/Statistiques_territoriales](http://www.bilans-ges.ademe.fr/fr/accueil/documentation-gene/index/page/Statistiques_territoriales)
703 (accessed 11.18.19).

704 Aimon-Marie, F., Angevin F., Guichard L., 2001. MERLIN : une méthode agronomique pour apprécier
705 les risques de pollution diffuse par les nitrates d'origine agricole. *Agrotransfert Lusignan* 27 p.

706 Angers, D.A., Arrouays, D., Saby, N.P.A., Walter, C., 2011. Estimating and mapping the carbon
707 saturation deficit of French agricultural topsoils. *Soil Use Manag.* 27, 448–452.
708 <https://doi.org/10.1111/j.1475-2743.2011.00366.x>

709 Arvalis – Institut du végétal, 2019. Céréales à paille, Variétés et interventions d'automne, Synthèse
710 nationale 2019. Choisir et Décider.

711 Arvalis - Institut du végétal, 2014a. Planter son blé tendre dans de bonnes conditions en AB
712 [WWW Document]. URL [https://www.arvalis-infos.fr/semer-du-ble-tendre-en-agriculture-
713 biologique-@/view-14692-arvarticle.html](https://www.arvalis-infos.fr/semer-du-ble-tendre-en-agriculture-biologique-@/view-14692-arvarticle.html) (accessed 11.19.19).

714 Arvalis - Institut du végétal, 2014b. Stratégies de fertilisation du blé tendre : fractionner les apports
715 [WWW Document]. URL [https://www.arvalis-infos.fr/fractionner-l-azote-en-trois-apports-
716 @/view-12293-arvarticle.html](https://www.arvalis-infos.fr/fractionner-l-azote-en-trois-apports-@/view-12293-arvarticle.html) (accessed 11.19.19).

717 Aveline, A., Rousseau, M.L., Guichard, L., Laurent, M., Bockstaller, C., 2009. Evaluating an
718 environmental indicator: Case study of MERLIN, a method for assessing the risk of nitrate
719 leaching. *Agric. Syst.* 100, 22–30. <https://doi.org/10.1016/j.agsy.2008.12.001>

720 Bagstad, K.J., Cohen, E., Ancona, Z.H., McNulty, S.G., Sun, G., 2018. The sensitivity of ecosystem
721 service models to choices of input data and spatial resolution. *Applied Geography* 93, 25–36.
722 <https://doi.org/10.1016/j.apgeog.2018.02.005>

723 Bagstad, K.J., Semmens, D.J., Waage, S., Winthrop, R., 2013. A comparative assessment of decision-
724 support tools for ecosystem services quantification and valuation. *Ecosyst. Serv.* 5, 27–39.
725 <https://doi.org/10.1016/j.ecoser.2013.07.004>

726 Balesdent, J., Besnard, E., Arrouays, D., Chenu, C., 1998. The dynamics of carbon in particle size
727 fractions of soil in a forest cultivated sequence. *Plant Soil* 201, 49–57.
728 <https://doi.org/10.1023/A:1004337314970>

729 Barot, S., Yé, L., Abbadie, L., Blouin, M., Frascaria-Lacoste, N., 2017. Ecosystem services must tackle
730 anthropized ecosystems and ecological engineering. *Ecol. Eng.* 99, 486–495.
731 <https://doi.org/10.1016/j.ecoleng.2016.11.071>

732 Barré, P., Angers, D.A., Basile-Doelsch, I., Bispo, A., Cécillon, L., Chenu, C., Chevallier, T., Derrien, D.,
733 Eglin, T.K., Pellerin, S., 2017. Ideas and perspectives: Can we use the soil carbon saturation

734 deficit to quantitatively assess the soil carbon storage potential, or should we explore other
735 strategies? *Biogeosciences Discuss.* 1–12. <https://doi.org/10.5194/bg-2017-395>

736 Baveye, P.C., 2017. Quantification of ecosystem services: Beyond all the “guesstimates”, how do we
737 get real data? *Ecosyst. Serv.* 24, 47–49. <https://doi.org/10.1016/j.ecoser.2017.02.006>

738 Baveye, P.C., 2014. Ecological economic perspective in environmental practice: Much-needed
739 common sense amid overwhelming market rhetoric. *Environ. Pract.* 16, 246–248.
740 <https://doi.org/10.1017/S1466046614000179>

741 Baveye, P.C., Baveye, J., Gowdy, J., 2016. Soil “Ecosystem” Services and Natural Capital: Critical
742 Appraisal of Research on Uncertain Ground. *Front. Environ. Sci.* 4.
743 <https://doi.org/10.3389/fenvs.2016.00041>

744 Baveye, P.C., Baveye, J., Gowdy, J., 2013. Monetary valuation of ecosystem services: It matters to get
745 the timeline right. *Ecol. Econ.* 95, 231–235. <https://doi.org/10.1016/j.ecolecon.2013.09.009>

746 Baveye, P.C., Chalhoub, M., Choquet, P., Montagne, D., 2018. Is the Focus on “Ecosystems” a Liability
747 in the Research on Nature’s Services? *Front. Ecol. Evol.* 6.
748 <https://doi.org/10.3389/fevo.2018.00226>

749 Baveye, P.C., Schnee, L.S., Boivin, P., Laba, M., Radulovich, R. 2020. Soil organic matter research and
750 climate change: Merely re-storing carbon versus restoring soil functions. *Front. Environ.*
751 *Sci.* 8:579904. <https://doi.org/10.3389/fenvs.2020.579904>

752 Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental
753 accounting units. *Ecol. Econ., Ecological Economics of Coastal Disasters* 63, 616–626.
754 <https://doi.org/10.1016/j.ecolecon.2007.01.002>

755 Brisson, N., Gary, C., Justes, E., Roche, R., Mary, B., Ripoche, D., Zimmer, D., Sierra, J., Bertuzzi, P.,
756 Burger, P., Bussi re, F., Cabidoche, Y.M., Cellier, P., Debaeke, P., Gaudill re, J.P., H nault, C.,
757 Maraux, F., Seguin, B., Sinoquet, H., 2003. An overview of the crop model stics. *European*
758 *Journal of Agronomy* 18(3), 309–332. [https://doi.org/10.1016/S1161-0301\(02\)00110-7](https://doi.org/10.1016/S1161-0301(02)00110-7)

759 Bunemann, E.K., Bongiorno, G., Bai, Z.G., Creamer, R.E., De Deyn, G., de Goede, R., Fleskens, L.,
760 Geissen, V., Kuyper, T.W., Mader, P., Pulleman, M., Sukkel, W., van Groenigen, J.W.,
761 Brussaard, L., 2018. Soil quality - A critical review. *Soil Biology & Biochemistry* 120, 105–125.
762 <https://doi.org/10.1016/j.soilbio.2018.01.030>

763 Burkhard, B., Kroll, F., M ller, F., 2009. Landscapes’ Capacities to Provide Ecosystem Services – a
764 Concept for Land-Cover Based Assessments. *Landsc. Online* 1–22.
765 <https://doi.org/10.3097/LO.200915>

766 Calzolari, C., Ungaro, F., Filippi, N., Guermandi, M., Malucelli, F., Marchi, N., Staffilani, F., Tarocco, P.,
767 2016. A methodological framework to assess the multiple contributions of soils to ecosystem
768 services delivery at regional scale. *Geoderma* 261, 190–203.
769 <https://doi.org/10.1016/j.geoderma.2015.07.013>

770 Cam C., Froger D., Moulin J., Rassineux J., Servant J. 1996. Repr sentation cartographique de la
771 sensibilit  des sols   l’infiltration hydrique verticale : carte th matique   l’infiltration
772 verticale. * tude et Gestion des Sols*, 97–112.

773 Carpentier, M., 2012. Mise en place du colza : Apr s les semis, assurer la protection de la culture
774 [WWW Document]. *Terre-Net*. URL [https://www.terre-net.fr/observatoire-technique-](https://www.terre-net.fr/observatoire-technique-culture-culturale/strategie-technique-culture/article/apres-les-semis-assurer-la-protection-de-la-culture-217-82760.html)
775 [culture-culturale/strategie-technique-culture/article/apres-les-semis-assurer-la-](https://www.terre-net.fr/observatoire-technique-culture-culturale/strategie-technique-culture/article/apres-les-semis-assurer-la-protection-de-la-culture-217-82760.html)
776 [culture-217-82760.html](https://www.terre-net.fr/observatoire-technique-culture-culturale/strategie-technique-culture/article/apres-les-semis-assurer-la-protection-de-la-culture-217-82760.html) (accessed 3.3.20).

777 Chalhoub, M., Gabrielle, B., Tournebize, J., Chaumont, C., Maugis, P., Girardin, C., Montagne, D.,
778 Baveye, P.C., Garnier, P., 2020. Direct measurement of selected soil services in a drained
779 agricultural field: Methodology development and case study in Saclay (France). *Ecosyst. Serv.*
780 42, 101088. <https://doi.org/10.1016/j.ecoser.2020.101088>

781 Chambre d’agriculture IdF, 2016. Bl  tendre : choix vari tal, implantation, d sherbage d’automne,
782 lutte contre les ravageurs d’automne, itin raire int gr  sur bl .

783 Chen, S.C., Martin, M.P., Saby, N.P.A., Walter, C., Angers, D.A., Arrouays, D., 2018a. Fine resolution
784 map of top- and subsoil carbon sequestration potential in France. *Science of the Total*
785 *Environment* 630, 389–400. <https://doi.org/10.1016/j.scitotenv.2018.02.209>

- 786 Clec'h, S.L., Oszwald, J., Decaens, T., Desjardins, T., Dufour, S., Grimaldi, M., Jegou, N., Lavelle, P.,
787 2016. Mapping multiple ecosystem services indicators: Toward an objective-oriented
788 approach. *Ecol. Indic.* 69, 508–521. <https://doi.org/10.1016/j.ecolind.2016.05.021>
- 789 Coleman, K., Jenkinson, D.S., Crocker, G.J., Grace, P.R., Klír, J., Körschens, M., Poulton, P.R., Richter,
790 D.D., 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-
791 26.3. *Geoderma* 81(1), 29-44. [https://doi.org/10.1016/S0016-7061\(97\)00079-7](https://doi.org/10.1016/S0016-7061(97)00079-7)
- 792 Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., Grasso, M.,
793 2017. Twenty years of ecosystem services: How far have we come and how far do we still
794 need to go? *Ecosyst. Serv.* 28, 1–16. <https://doi.org/10.1016/j.ecoser.2017.09.008>
- 795 Crouzat, E., Mouchet, M., Turkelboom, F., Byczek, C., Meersmans, J., Berger, F., Verkerk, P.J., Lavorel,
796 S., 2015. Assessing bundles of ecosystem services from regional to landscape scale: insights
797 from the French Alps. *J. Appl. Ecol.* 52, 1145–1155. <https://doi.org/10.1111/1365-2664.12502>
- 799 Dai, Y., Shangguan, W., Wei, N., Xin, Q., Yuan, H., Zhang, S., Liu, S., Lu, X., Wang, D., Yan, F., 2019. A
800 review of the global soil property maps for Earth system models. *Soil* 5(2), 137-158.
801 <https://doi.org/10.5194/soil-5-137-2019>
- 802 Debeljak, M., Trajanov, A., Kuzmanovski, V., Schröder, J., Sandén, T., Spiegel, H., Wall, D.P., Van de
803 Broek, M., Rutgers, M., Bampa, F., Creamer, R.E., Henriksen, C.B., 2019. A Field-Scale
804 Decision Support System for Assessment and Management of Soil Functions. *Frontiers in*
805 *Environmental Science* 7(115). <https://doi.org/10.3389/fenvs.2019.00115>
- 806 Demestihias, C., Plénet, D., Génard, M., Garcia de Cortazar-Atauri, I., Launay, M., Ripoche, D.,
807 Beaudoin, N., Simon, S., Charreyron, M., Raynal, C., Lescourret, F., 2018. Analyzing ecosystem
808 services in apple orchards using the STICS model. *Eur. J. Agron.* 94, 108–119.
809 <https://doi.org/10.1016/j.eja.2018.01.009>
- 810 Dick, J., Turkelboom, F., Woods, H., Iniesta-Arandia, I., Primmer, E., Saarela, S.-R., Bezák, P., Mederly,
811 P., Leone, M., Verheyden, W., Kelemen, E., Hauck, J., Andrews, C., Antunes, P., Aszalós, R.,
812 Baró, F., Barton, D.N., Berry, P., Bugter, R., Carvalho, L., Czúcz, B., Dunford, R., Garcia Blanco,
813 G., Geamăna, N., Giucă, R., Grizzetti, B., Izakovičová, Z., Kertész, M., Kopperoinen, L.,
814 Langemeyer, J., Montenegro Lapola, D., Liqueste, C., Luque, S., Martínez Pastur, G., Martin-
815 Lopez, B., Mukhopadhyay, R., Niemela, J., Odee, D., Peri, P.L., Pinho, P., Patrício-Roberto,
816 G.B., Preda, E., Priess, J., Röckmann, C., Santos, R., Silaghi, D., Smith, R., Vădineanu, A., van
817 der Wal, J.T., Arany, I., Badea, O., Bela, G., Boros, E., Bucur, M., Blumentrath, S., Calvache,
818 M., Carmen, E., Clemente, P., Fernandes, J., Ferraz, D., Fongar, C., García-Llorente, M.,
819 Gómez-Baggethun, E., Gundersen, V., Haavardsholm, O., Kalóczkai, Á., Khalalwe, T., Kiss, G.,
820 Köhler, B., Lazányi, O., Lellei-Kovács, E., Lichungu, R., Lindhjem, H., Magare, C., Mustajoki, J.,
821 Ndege, C., Nowell, M., Nuss Girona, S., Ochieng, J., Often, A., Palomo, I., Pataki, G., Reinvang,
822 R., Rusch, G., Saarikoski, H., Smith, A., Soy Massoni, E., Stange, E., Vågnes Traaholt, N., Vári,
823 Á., Verweij, P., Vikström, S., Yli-Pelkonen, V., Zulian, G., 2018. Stakeholders' perspectives on
824 the operationalisation of the ecosystem service concept: Results from 27 case studies.
825 *Ecosystem Services* 29, 552-565. <https://doi.org/10.1016/j.ecoser.2017.09.015>
- 826 Dominati, E., Patterson, M., Mackay, A., 2010. A framework for classifying and quantifying the natural
827 capital and ecosystem services of soils. *Ecological Economics* 69(9),
828 <https://doi.org/10.1016/j.ecolecon.2010.05.002>
- 829 Dominati, E., Mackay, A., Green, S., Patterson, M., 2014. A soil change-based methodology for the
830 quantification and valuation of ecosystem services from agro-ecosystems: A case study of
831 pastoral agriculture in New Zealand. *Ecol. Econ.* 100, 119–129.
832 <https://doi.org/10.1016/j.ecolecon.2014.02.008>
- 833 Dominati, E.J., Mackay, A.D., Bouma, J., Green, S., 2016. An Ecosystems Approach to Quantify Soil
834 Performance for Multiple Outcomes: The Future of Land Evaluation? *Soil Science Society of*
835 *America Journal* 80(2), 438-449. <https://doi.org/10.2136/sssaj2015.07.0266>

836 Eigenbrod, F., Bell, V.A., Davies, H.N., Heinemeyer, A., Armsworth, P.R., Gaston, K.J., 2011. The
837 impact of projected increases in urbanization on ecosystem services. *Proc. R. Soc. B Biol. Sci.*
838 278, 3201–3208. <https://doi.org/10.1098/rspb.2010.2754>

839 Ellili-Bargaoui, Y., Walter, C., Lemerrier, B., Michot, D., 2021. Assessment of six soil ecosystem
840 services by coupling simulation modelling and field measurement of soil properties. *Ecological Indicators* 121, 107211. <https://doi.org/10.1016/j.ecolind.2020.107211>

841 Fossey, M., Angers, D., Bustany, C., Cudennec, C., Durand, P., Gascuel-Oudou, C., Jaffrezic, A., Pérès,
842 G., Besse, C., Walter, C., 2020. A Framework to Consider Soil Ecosystem Services in Territorial
843 Planning. *Frontiers in Environmental Science* 8(28).
844 <https://doi.org/10.3389/fenvs.2020.00028>

845 FranceAgriMer, 2012. Itinéraires techniques du blé tendre dans cinq départements du Grand Bassin
846 Parisien 1995/2010.

847 Francesconi, W., Srinivasan, R., Pérez-Miñana, E., Willcock, S.P., Quintero, M., 2016. Using the Soil
848 and Water Assessment Tool (SWAT) to model ecosystem services: A systematic review. *Journal of Hydrology* 535, 625-636. <https://doi.org/10.1016/j.jhydrol.2016.01.034>

849 Gabrielle, B., Laville, P., Duval, O., Nicoulaud, B., Germon, J.C., Hénault, C., 2006. Process-based
850 modeling of nitrous oxide emissions from wheat-cropped soils at the subregional scale:
851 regional N₂O emissions from arable soils. *Glob. Biogeochem. Cycles* 20, n/a-n/a.
852 <https://doi.org/10.1029/2006GB002686>

853 Gabrielle, B., Menasseri, S., Houot, S., 1995. Analysis and field evaluation of the Ceres models water
854 balance component. *Soil Sci. Soc. Am. J.* 59, 1403-1412.
855 <https://doi.org/10.2136/sssaj1995.03615995005900050029x>

856 Gómez-Baggethun, E., de Groot, R., Lomas, P.L., Montes, C., 2010. The history of ecosystem services
857 in economic theory and practice: From early notions to markets and payment schemes. *Ecol.*
858 *Econ.* 69, 1209–1218. <https://doi.org/10.1016/j.ecolecon.2009.11.007>

859 Greiner, L., Keller, A., Grêt-Regamey, A., Papritz, A., 2017. Soil function assessment: review of
860 methods for quantifying the contributions of soils to ecosystem services. *Land Use Policy* 69,
861 224–237. <https://doi.org/10.1016/j.landusepol.2017.06.025>

862 Grêt-Regamey, A., Sirén, E., Brunner, S.H., Weibel, B., 2017. Review of decision support tools to
863 operationalize the ecosystem services concept. *Ecosyst. Serv.* 26, 306–315.
864 <https://doi.org/10.1016/j.ecoser.2016.10.012>

865 Haines-Young, R., Potschin, M., 2018. Common International Classification of Ecosystem Services
866 (CICES) V5.1 Guidance on the Application of the Revised Structure. Fabis Consulting Ltd.

867 Harrison, P.A., Dunford, R., Barton, D.N., Kelemen, E., Martín-López, B., Norton, L., Termansen, M.,
868 Saarikoski, H., Hendriks, K., Gómez-Baggethun, E., Czúcz, B., García-Llorente, M., Howard, D.,
869 Jacobs, S., Karlsen, M., Kopperoinen, L., Madsen, A., Rusch, G., van Eupen, M., Verweij, P.,
870 Smith, R., Tuomasjukka, D., Zulian, G., 2018. Selecting methods for ecosystem service
871 assessment: A decision tree approach. *Ecosystem Services* 29, 481-498.
872 <https://doi.org/10.1016/j.ecoser.2017.09.016>

873 Hassink, J., 1997. The capacity of soils to preserve organic C and N by their association with clay and
874 silt particles. *Plant Soil* 191, 77–87. <https://doi.org/10.1023/A:1004213929699>

875 Hewitt, A., Dominati, E., Webb, T., Cuthill, T., 2015. Soil natural capital quantification by the stock
876 adequacy method. *Geoderma* 241, 107-114.
877 <https://doi.org/10.1016/j.geoderma.2014.11.014>

878 INRA Infosol, 2005. Référentiel Régional Pédologique: Cahier des Clauses Techniques et Générales.

879 IUSS Working Group WRB, 2015. International soil classification system for naming soils and creating
880 legends for soil maps (World Reference Base for Soil Resources 2014, update 2015), World
881 Soil Resources Reports No. 106. FAO, Rome.

882 Jacobs, S., Burkhard, B., Van Daele, T., Staes, J., Schneiders, A., 2015. ‘The Matrix Reloaded’: A review
883 of expert knowledge use for mapping ecosystem services. *Ecol. Model., Use of ecological
884 indicators in models* 295, 21–30. <https://doi.org/10.1016/j.ecolmodel.2014.08.024>

- 887 Jahn, R., Blume, H.P., Asio, V.B., Spaargaren, O., Schad, P., 2006. Guidelines for soil description, 4th
888 edition. FAO, Rome.
- 889 Jamagne, M., 2011. Grands paysages pédologiques de France, Quae. ed.
- 890 Jones, C.A., Kiniry, J.R., Dyke, P.T., 1986. CERES-Maize: a simulation model of maize growth and
891 development. Texas AandM University Press.
- 892 Lavorel, S., Bayer, A., Bondeau, A., Lautenbach, S., Ruiz-Frau, A., Schulp, N., Seppelt, R., Verburg, P.,
893 Teeffelen, A.v., Vannier, C., Arneth, A., Cramer, W., Marba, N., 2017. Pathways to bridge the
894 biophysical realism gap in ecosystem services mapping approaches. *Ecological Indicators* 74,
895 241-260. <https://doi.org/10.1016/j.ecolind.2016.11.015>
- 896 Megnien, 1989. Notice explicative, Carte géol. France (1/50 000), feuille CORBEIL-ESSONES (219).
- 897 Mouchet, M.A., Lamarque, P., Martin-Lopez, B., Crouzat, E., Gos, P., Byczek, C., Lavorel, S., 2014. An
898 interdisciplinary methodological guide for quantifying associations between ecosystem
899 services. *Global Environmental Change-Human and Policy Dimensions* 28, 298-308.
900 <https://doi.org/10.1016/j.gloenvcha.2014.07.012>
- 901 Mueller, L., Schindler, U., Behrendt, A., Eulenstein, F., Dannowski, R., 2007. The Muencheberg Soil
902 Quality Rating (SQR).
- 903 Mueller, L., Schindler, U., Shepherd, T.G., Ball, B.C., Smolentseva, E., Hennings, V., Schad, P.,
904 Behrendt, A., Helming, K., Eulenstein, F., 2010. Assessing agricultural soil quality on a global
905 scale. 19th World Congr. Soil Sci. Soil Solut. Chang. World Aust. 4.
- 906 Mueller, L., Shepherd, G., Schindler, U., Ball, B.C., Munkholm, L.J., Hennings, V., Smolentseva, E.,
907 Rukhovic, O., Lukin, S., Hu, C., 2013. Evaluation of soil structure in the framework of an
908 overall soil quality rating. *Soil Tillage Res.*, Special issue: Applications of Visual Soil Evaluation
909 127, 74–84. <https://doi.org/10.1016/j.still.2012.03.002>
- 910 Müller, F., Bicking, S., Ahrendt, K., Kinh Bac, D., Blindow, I., Fürst, C., Haase, P., Kruse, M., Kruse, T.,
911 Ma, L., Perennes, M., Ruljevic, I., Schernewski, G., Schimming, C.-G., Schneiders, A., Schubert,
912 H., Schumacher, n., Tappeiner, U., Wangai, P., Windhorst, W., Zeleny, J., 2020. Assessing
913 ecosystem service potentials to evaluate terrestrial, coastal and marine ecosystem types in
914 Northern Germany – An expert-based matrix approach. *Ecological Indicators* 112, 106116.
915 <https://doi.org/10.1016/j.ecolind.2020.106116>
- 916 Noirot-Cosson, 2016. Optimisation de l’insertion des Produits Résiduaire Organiques dans les
917 systèmes de cultures d’un territoire francilien: évolution des stocks de carbone organique
918 des sols et substitution des engrais minéraux.
- 919 Noirot-Cosson, P.E., Vaudour, E., Gilliot, J.M., Gabrielle, B., Houot, S., 2016. Modelling the long-term
920 effect of urban waste compost applications on carbon and nitrogen dynamics in temperate
921 cropland. *Soil Biol. Biochem.* 94, 138–153. <https://doi.org/10.1016/j.soilbio.2015.11.014>
- 922 Obiang Ndong, G., Therond, O., Cousin, I., 2020. Analysis of relationships between ecosystem
923 services: A generic classification and review of the literature. *Ecosystem Services* 43, 101120.
924 <https://doi.org/10.1016/j.ecoser.2020.101120>
- 925 O’Farrell, P.J., Reyers, B., Le Maitre, D.C., Milton, S.J., Egoh, B., Maherry, A., Colvin, C., Atkinson, D.,
926 De Lange, W., Blignaut, J.N., Cowling, R.M., 2010. Multi-functional landscapes in semi arid
927 environments: implications for biodiversity and ecosystem services. *Landsc. Ecol.* 25, 1231–
928 1246. <https://doi.org/10.1007/s10980-010-9495-9>
- 929 Pongratz, J., Dolman, H., Don, A., Erb, K.-H., Fuchs, R., Herold, M., Jones, C., Kuemmerle, T., Luysaert,
930 S., Meyfroidt, P., Naudts, K., 2018. Models meet data: Challenges and opportunities in
931 implementing land management in Earth system models. *Glob. Change Biol.* 24(4), 1470-
932 1487. <https://doi.org/10.1111/gcb.13988>
- 933 Remme, R.P., Schröter, M., Hein, L., 2014. Developing spatial biophysical accounting for multiple
934 ecosystem services. *Ecosyst. Serv.* 10, 6–18. <https://doi.org/10.1016/j.ecoser.2014.07.006>
- 935 Robinson, D.A., Lebron, I., Vereecken, H., 2009. On the definition of the natural capital of soils: A
936 framework for description, evaluation, and monitoring. *Soil Science Society of America*
937 *Journal* 73(6), 1904-1911. <https://doi.org/10.2136/sssaj2008.0332>

938 Roche, P.K., Campagne, C.S., 2019. Are expert-based ecosystem services scores related to biophysical
939 quantitative estimates? *Ecol. Indic.* 106, 105421.
940 <https://doi.org/10.1016/j.ecolind.2019.05.052>

941 Roque, J., 2004. Référentiel régional pédologique de l'Île-de-France à 1/250 000, INRA. ed, Régions
942 naturelles, pédopaysages et sols.

943 Schröter, M., Remme, R.P., Sumarga, E., Barton, D.N., Hein, L., 2015. Lessons learned for spatial
944 modelling of ecosystem services in support of ecosystem accounting. *Ecosystem Services* 13,
945 64-69. <https://doi.org/10.1016/j.ecoser.2014.07.003>

946 Schulp, C.J.E., Alkemade, R., Klein Goldewijk, K., Petz, K., 2012. Mapping ecosystem functions and
947 services in Eastern Europe using global-scale data sets. *International Journal of Biodiversity
948 Science, Ecosystem Services & Management* 8(1-2), 156-168.
949 <https://doi.org/10.1080/21513732.2011.645880>

950 Sharps, K., Masante, D., Thomas, A., Jackson, B., Redhead, J., May, L., Prosser, H., Cosby, B., Emmett,
951 B., Jones, L., 2017. Comparing strengths and weaknesses of three ecosystem services
952 modelling tools in a diverse UK river catchment. *Sci. Total Environ.* 584-585, 118-130.
953 <https://doi.org/10.1016/j.scitotenv.2016.12.160>

954 Rutgers, M., van Wijnen, H.J., Schouten, A.J., Mulder, C., Kuiten, A.M.P., Brussaard, L., Breure, A.M.,
955 2012. A method to assess ecosystem services developed from soil attributes with
956 stakeholders and data of four arable farms. *Science of the Total Environment* 415, 39-48.
957 <https://doi.org/10.1016/j.scitotenv.2011.04.041>

958 Šimůnek, J., van Genuchten, M.T., Šejna, M., 2008. Development and Applications of the HYDRUS and
959 STANMOD Software Packages and Related Codes. *Vadose Zone Journal* 7(2), 587-600.
960 <https://doi.org/10.2136/vzj2007.0077>

961 Storie, D. of A.S.U. of C., 1978. Storie Index Soil Rating.

962 Tedesco, C., Petit, C., Billen, G., Garnier, J., Personne, E., 2017. Potential for recoupling production
963 and consumption in peri-urban territories: The case-study of the Saclay plateau near Paris,
964 France. *Food Policy* 69, 35-45. <https://doi.org/10.1016/j.foodpol.2017.03.006>

965 Terres Inovia, 2019. Guide de culture : colza.

966 Therond, O., Tichit, M., Tibi, A., Accatino, F., L., B.-D., Bockstaller, C., Bohan, D., Bonaudo, T., Boval,
967 M., Cahuzac, E., Casellas, E., Chauvel, B., Choler, P., Constantin, J., Cousin, I., Daroussin, J.,
968 David, M., Delacote, P., Derocles, S., De Sousa, L., Domingues Santos, J.P., Dross, C., Duru, M.,
969 Eugene, M., Fontaine, C., Garcia, B., Geijzendorffer, I., Girardin, A., Graux, A.-I., Jouven, M.,
970 Langlois, B., Le Bas, C., Le Bissonnais, Y., Lelievre, V., Lifran, R., Maigne, E., Martin, G., Martin,
971 R., Martin-Laurent, F., Martinet, V., McLaughlin, O., Meillet, A., Mignolet, C., Mouchet, M.,
972 Nozières-Petit, M.-O., Ostermann, O.P., Paracchini, M.L., Pellerin, S., Peyraud, J.-L., Petit-
973 Michaut, S., Picaud, C., Plantureux, S., Pomeon, T., Porcher, E., Puech, T., Puillet, L.,
974 Rambonilaza, T., Raynal, H., Resmond, R., Ripoché, D., Ruget, F., Rulleau, B., Rusch, A., Salles,
975 J.-M., Sauvans, D., Schott, C., Tardieu, L., 2017. Volet "écosystèmes agricoles" de l'Evaluation
976 Française des Ecosystèmes et des Services Écosystémiques. Rapport d'étude. INRA, 966 pp.

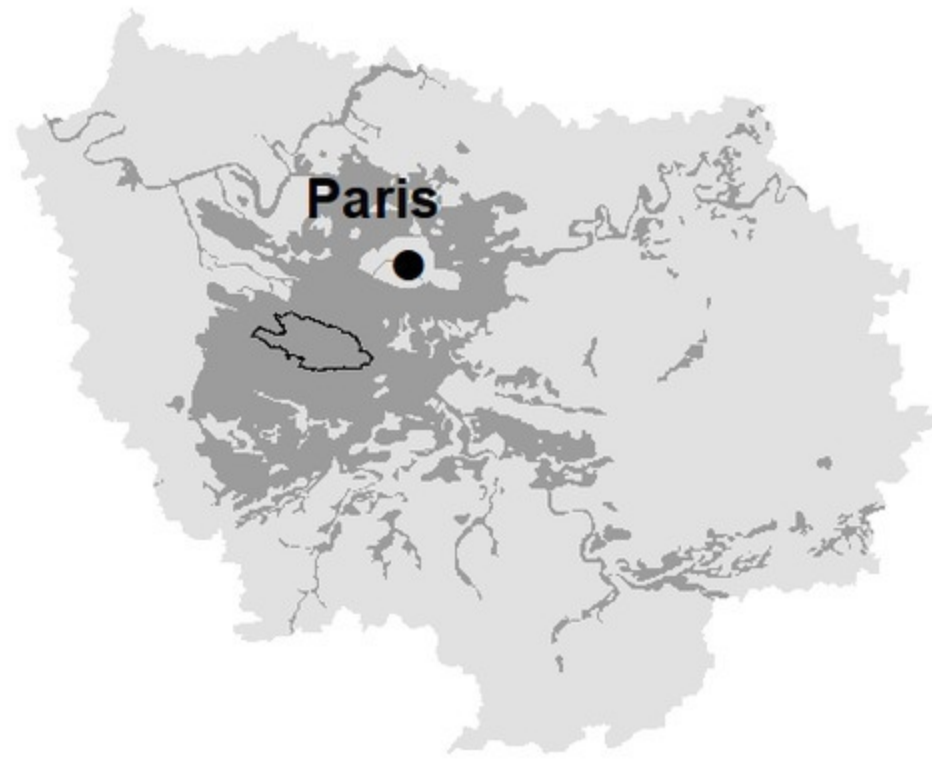
977 Ungaro, F., Staffilani, F., Tarocco, P., 2010. Assessing and mapping topsoil organic carbon stock at
978 regional scale: A scorpan kriging approach conditional on soil map delineations and land use.
979 *Land Degrad. Dev.* 21, 565-581. <https://doi.org/10.1002/ldr.998>

980 Ungaro, F., Zasada, I., Pierr, A., 2014. Mapping landscape services, spatial synergies and trade-offs. A
981 case study using variogram models and geostatistical simulations in an agrarian landscape in
982 North-East Germany. *Ecol. Indic.* 46, 367-378. <https://doi.org/10.1016/j.ecolind.2014.06.039>

983 Vaisman, Institut d'aménagement et d'urbanisme de la région d'Île-de-France, Agence régionale de
984 l'environnement et des nouvelles énergies (Île-de-France), 2018. La paille: concurrences et
985 complémentarités des usages du gisement agricole en Île-de-France. IAU Île-de-France, Paris.

986 van Dijk, J., Dick, J., Harrison, P., Jax, K., Saarikoski, H., Furman, E., 2018. Editorial: Operationalisation
987 of natural capital and ecosystem services – Special issue. *Ecosyst. Serv.* 29, 411-414.
988 <https://doi.org/10.1016/j.ecoser.2017.11.013>

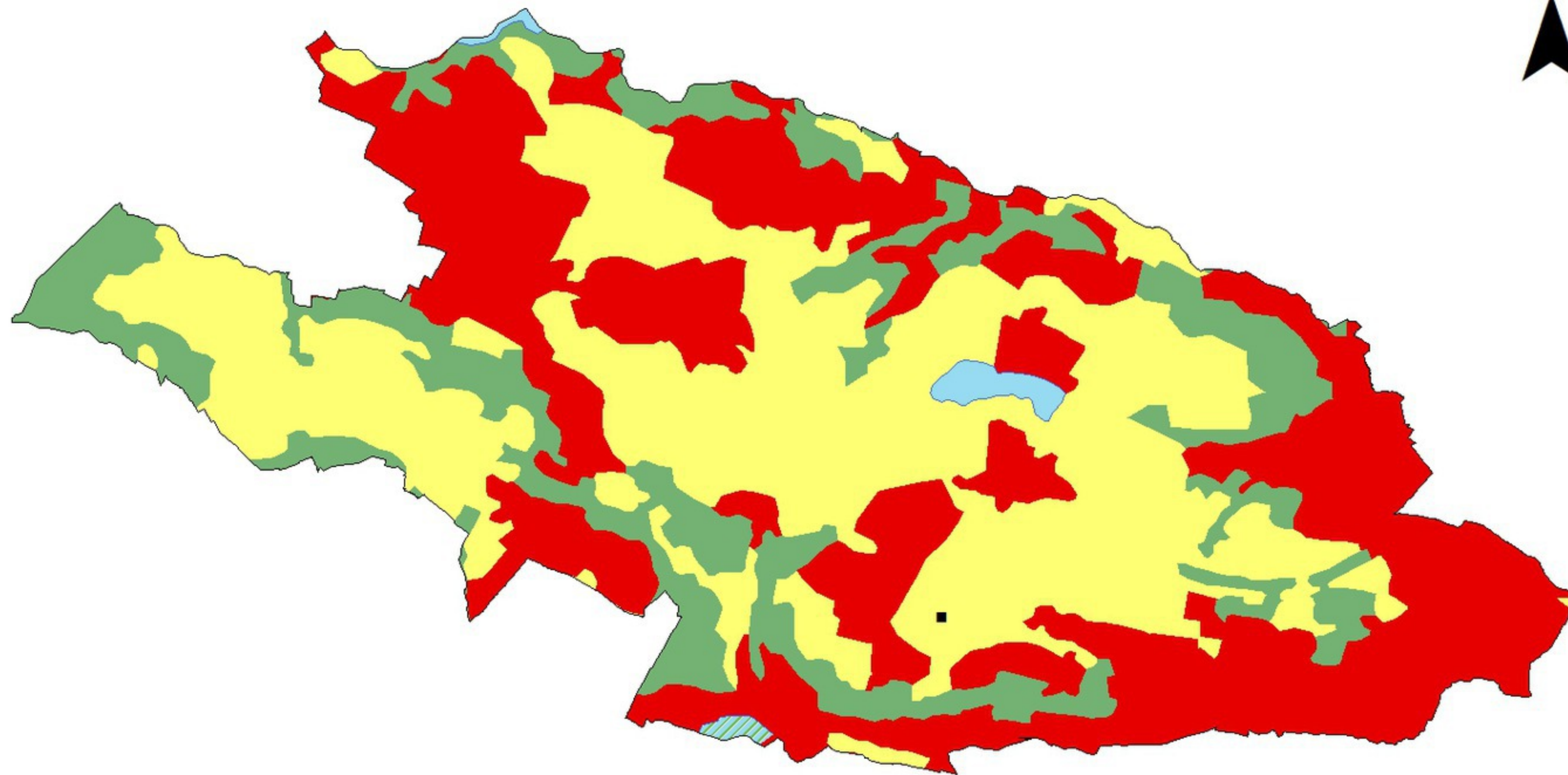
- 989 Van Looy, K., Bouma, J., Herbst, M., Koestel, J., Minasny, B., Mishra, U., Montzka, C., Nemes, A.,
990 Pachepsky, Y.A., Padarian, J., Schaap, M.G., Tóth, B., Verhoef, A., Vanderborght, J., van der
991 Ploeg, M.J., Weihermüller, L., Zacharias, S., Zhang, Y., Vereecken, H., 2017. Pedotransfer
992 Functions in Earth System Science: Challenges and Perspectives: PTFs in Earth system science
993 perspective. *Rev. Geophys.* 55, 1199–1256. <https://doi.org/10.1002/2017RG000581>
- 994 van Wijnen, H.J., Rutgers, M., Schouten, A.J., Mulder, C., de Zwart, D., Breure, A.M., 2012. How to
995 calculate the spatial distribution of ecosystem services - Natural attenuation as example from
996 The Netherlands. *Science of the Total Environment* 415, 49-55.
997 <https://doi.org/10.1016/j.scitotenv.2011.05.058>
- 998 Vereecken, H., Schnepf, A., Hopmans, J.W., Javaux, M., Or, D., Roose, T., Vanderborght, J., Young,
999 M.H., Amelung, W., Aitkenhead, M., Allison, S.D., Assouline, S., Baveye, P., Berli, M.,
1000 Brüggemann, N., Finke, P., Flury, M., Gaiser, T., Govers, G., Ghezzehei, T., Hallett, P.,
1001 Hendricks Franssen, H.J., Heppell, J., Horn, R., Huisman, J.A., Jacques, D., Jonard, F., Kollet, S.,
1002 Lafolie, F., Lamorski, K., Leitner, D., McBratney, A., Minasny, B., Montzka, C., Nowak, W.,
1003 Pachepsky, Y., Padarian, J., Romano, N., Roth, K., Rothfuss, Y., Rowe, E.C., Schwen, A.,
1004 Šimůnek, J., Tiktak, A., Van Dam, J., van der Zee, S.E.A.T.M., Vogel, H.J., Vrugt, J.A., Wöhling,
1005 T., Young, I.M., 2016. Modeling Soil Processes: Review, Key Challenges, and New
1006 Perspectives. *Vadose Zone J.* 15, 0. <https://doi.org/10.2136/vzj2015.09.0131>
- 1007 Vogel, H.-J., Eberhardt, E., Franko, U., Lang, B., Ließ, M., Weller, U., Wiesmeier, M., Wollschläger, U.
1008 2019. Quantitative evaluation of soil functions: Potential and state. *Front. Environ. Sci.* 7:164.
1009 <https://doi.org/10.3389/fenvs.2019.00164>
- 1010 Vorstius, A.C., Spray, C.J., 2015. A comparison of ecosystem services mapping tools for their potential
1011 to support planning and decision-making on a local scale. *Ecosyst. Serv.* 15, 75–83.
1012 <https://doi.org/10.1016/j.ecoser.2015.07.007>
- 1013 Westman, W.E., 1977. How Much Are Nature's Services Worth? *Science* 197, 960–964.
1014 <https://doi.org/10.1126/science.197.4307.960>
- 1015 Wiesmeier, M., Hübner, R., Spörlein, P., Geuß, U., Hangen, E., Reischl, A., Schilling, B., von Lützw,
1016 M., Kögel-Knabner, I., 2014. Carbon sequestration potential of soils in southeast Germany
1017 derived from stable soil organic carbon saturation. *Glob. Change Biol.* 20(2), 653-665.
1018 <https://doi.org/10.1111/gcb.12384>
- 1019 Wischmeier and Smith, 1978. Predicting rainfall erosion losses – a guide to conservation planning
1020 (Agriculture Handbook No. No. 537). U.S. Department of Agriculture.
- 1021 Zank, B., Bagstad, K.J., Voigt, B., Villa, F., 2016. Modeling the effects of urban expansion on natural
1022 capital stocks and ecosystem service flows: A case study in the Puget Sound, Washington,
1023 USA. *Landsc. Urban Plan.* 149, 31–42. <https://doi.org/10.1016/j.landurbplan.2016.01.004>
- 1024 Zulian, G., Stange, E., Woods, H., Carvalho, L., Dick, J., Andrews, C., Baró, F., Vizcaino, P., Barton, D.N.,
1025 Nowel, M., Rusch, G.M., Autunes, P., Fernandes, J., Ferraz, D., Ferreira dos Santos, R.,
1026 Aszalós, R., Arany, I., Czúcz, B., Priess, J.A., Hoyer, C., Bürger-Patricio, G., Lapola, D., Mederly,
1027 P., Halabuk, A., Bezak, P., Kopperoinen, L., Viinikka, A., 2018. Practical application of spatial
1028 ecosystem service models to aid decision support. *Ecosystem Services* 29, 465-480.
1029 <https://doi.org/10.1016/j.ecoser.2017.11.005>



Paris



N

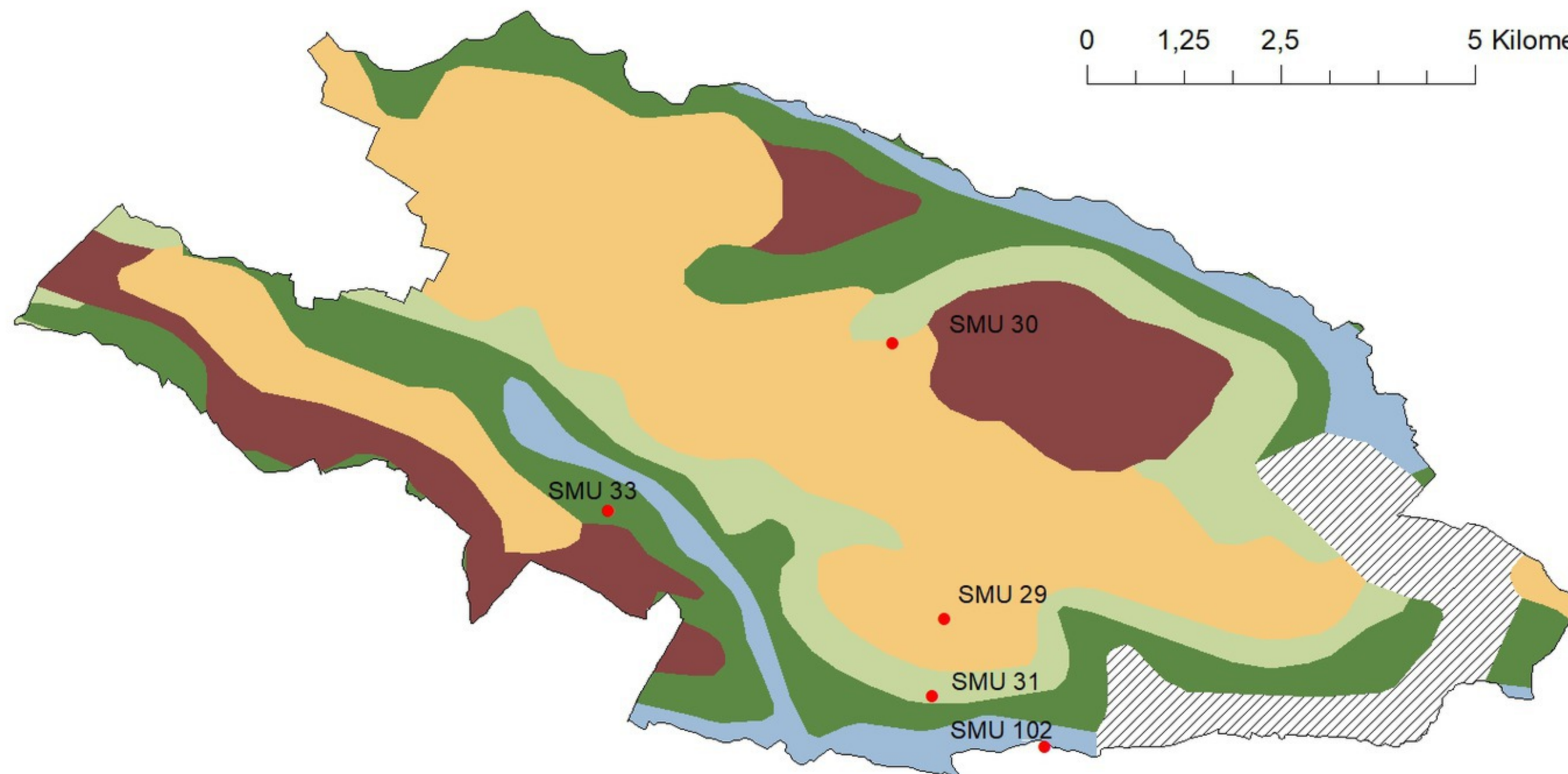


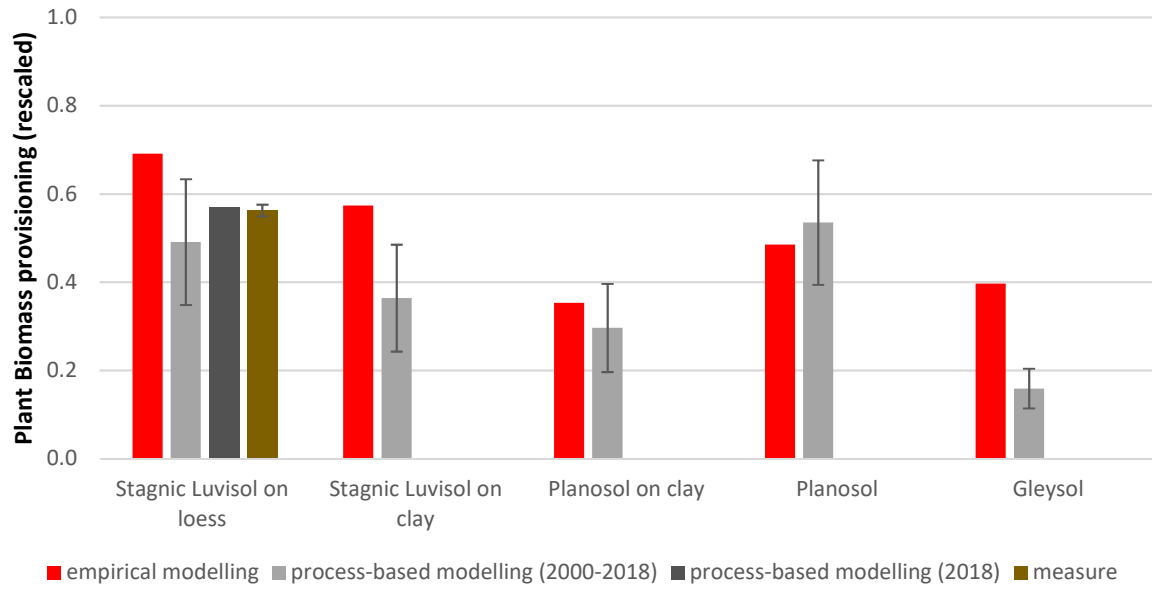
Legend

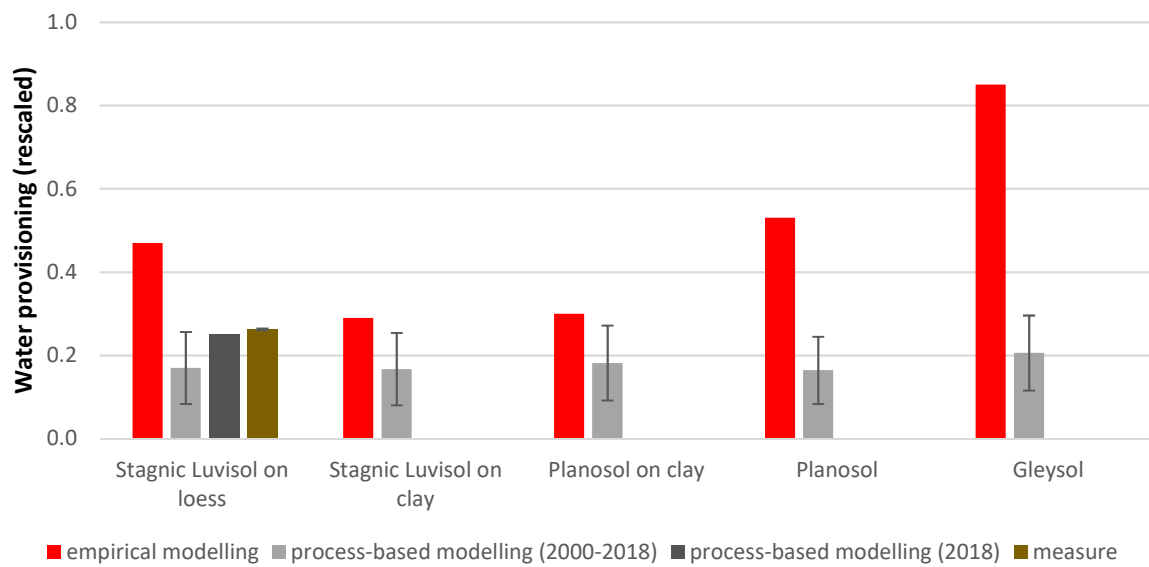
- Ile-de-France region
- The Saclay plateau
- Typical soils of Saclay
- Field site
- Agricultural areas
- Artificial surfaces
- Forest and semi natural areas
- Water bodies
- Wetlands

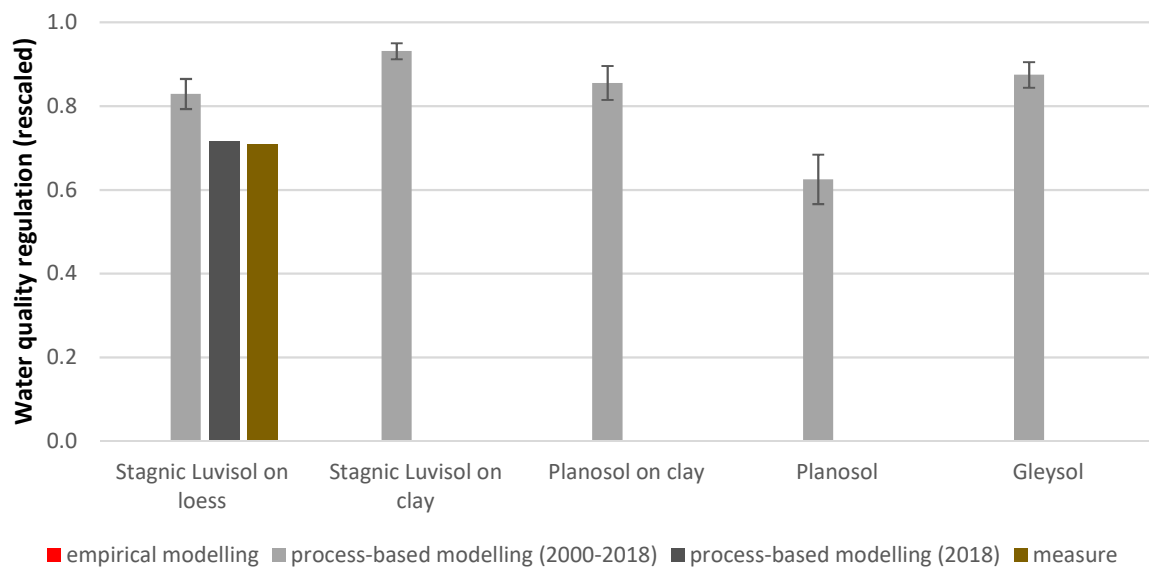
Soil Map Units

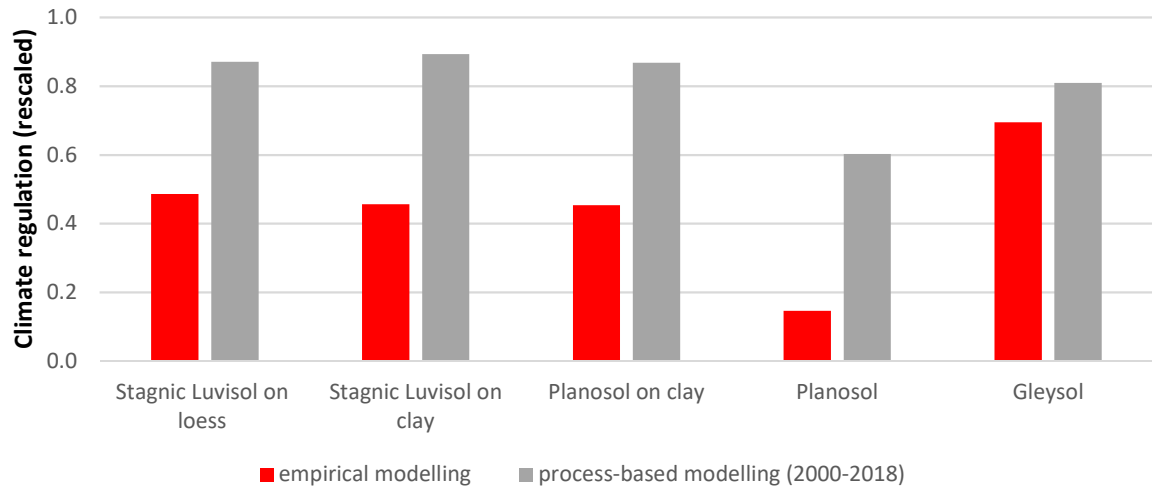
- Stagnic Luvisol on loess (SMU 29)
- Stagnic Luvisol on clay (SMU 30)
- Planosol on clay (SMU 31)
- Planosol (SMU 33)
- Gleysol (SMU 102)
- Urbanized areas
- Pit representative of the Soil Map Unit











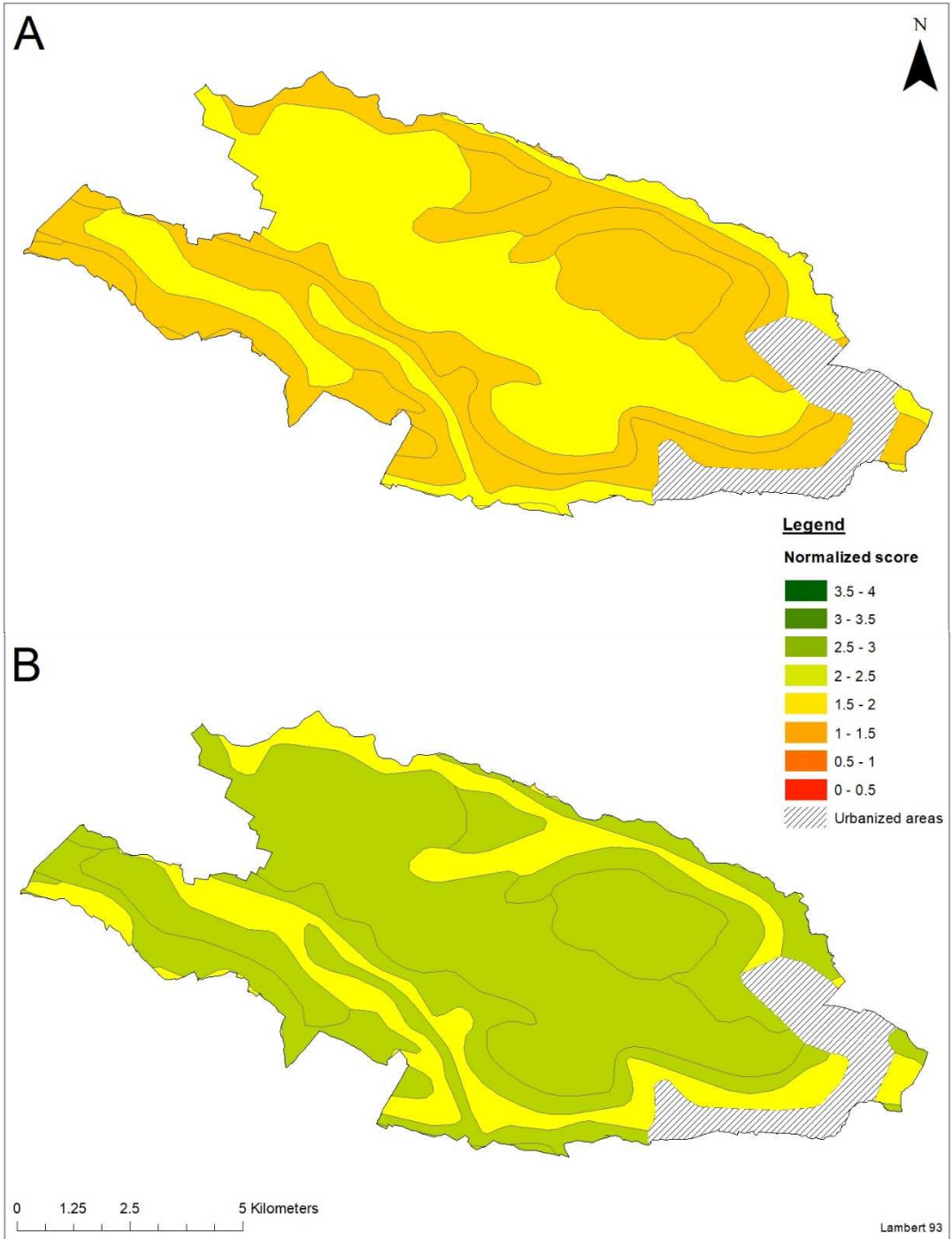


Table 1: Main soil descriptive properties^a

Table 2: Main soil analytical properties

Table 3: Crops and crops management

Table 4: Soil supported ecosystem services associated to indicators and tools

Table 5: Absolute minimum and maximum values per service used for results rescaling (national French scale)

Table 6: Assessment of the soil contribution to ecosystem services from empirical and process-based modelling

Soil identification ^b			General Information			Soil description								
SMU	STU	Name	Slope (%)	Land Use	Parent Material	Horizon	Depth (cm)	Texture ^c	Coarse fragments (%)	Colour (moist)	Mottling (%)	Structure	Clay Coatings (%)	Roots (<2mm)
29	85	Stagnic Luvisol on loess	0.5-1.0	Crop	Loess	Ap	0 - 42	SiL	0 - 2	10 YR 44	0	Crumbly	0	20 - 50
					Loess	Bt	42 - 60	SiCL	0 - 2	10 YR 56	0	Angular blocky	> 80	1 - 20
					Loess	Btg1	60 - 90	SiCL	0 - 2	10 YR 56	5 - 15	Prismatic	40 - 80	1 - 20
					Loess	Btg2	90 - 100	SiCL	0 - 2	10 YR 54	15 - 40	Prismatic	15 - 40	1 - 20
					Loess	BtgC	100 - 115	SiL	0 - 2	10 YR 54	15 - 40	Massive	2 - 5	1 - 20
					Loess	Ckg	115 - 135	SiL	0 - 2	10 YR 56	15 - 40	Massive	2 - 5	0
30	86	Stagnic Luvisol on clay	1.0-2.0	Crop	Loess	Ap	0 - 35	SiL	0 - 2	10 YR 34	0	Subangular blocky	0	1 - 20
					Loess	Bt	35 - 50	SiCL	0 - 2	10 YR 44	2 - 5	Angular blocky	15 - 40	1 - 20
					Loess	Btg	50 - 80	SiCL	0 - 2	10 YR 44	5 - 15	Angular blocky	40 - 80	1 - 20
					Slope deposits	2Bt	80 - 120	SiC	15 - 40	10 YR 46	2 - 5	Massive	40 - 80	1 - 20
31	90	Planosol on clay	1.0-2.0	Fallow	Slope deposits	ApEg	0 - 40	SiL	15 - 40	10 YR 53	5 - 15	Massive	0	20 - 50
					Slope deposits	Eg	40 - 60	L	5 - 15	10 YR 64	40 - 80	Massive	0	20 - 50
					Clay	2Cg1	60 - 90	C	5 - 15	7,5 YR 56	15 - 40	Prismatic	0	1 - 20
					Clay	2Cg2	90 - 120	CL	15 - 40	10 YR 58	15 - 40	Prismatic	0	0
33	47	Planosol	5-10	Forest	Slope deposits	A1	0 - 5	SL	2 - 5	10 YR 22	0	Single grain	0	20 - 50
					Slope deposits	A2	5 - 15	SL	2 - 5	10 YR 43	0	Single grain	0	1 - 20
					Slope deposits	Bw	15 - 35	SL	2 - 5	10 YR44	0	Single grain	0	1 - 20
					Slope deposits	C	35 - 45	SL	2 - 5	10 YR 44	0	Single grain	0	1 - 20
					Slope deposits	2Cg1	45 - 70	SiL	2 - 5	10 YR 46	5 - 15	Massive	0	1 - 20
					Slope deposits	2Cg2	70 - 120	SiL	0 - 2	10 YR 56	> 40	Massive	0	1 - 20
102	56	Gleysol	0.5-1.0	Forest	Alluvial deposits	A	0 - 7	CL	0 - 2	10 YR 21	0	Subangular blocky	0	1 - 20
					Alluvial deposits	Bg1	7 - 20	CL	0 - 2	10 YR 41	5 - 15	Prismatic	0	1 - 20
					Alluvial deposits	Bg2	20 - 35	SiC	0 - 2	10 YR 51	5 - 15	Prismatic	0	1 - 20
					Alluvial deposits	Bg3	35 - 55	SiCL	0 - 2	10 YR 51	15 - 40	Prismatic	0	1 - 20
					Alluvial deposits	Bg4	55 - 80	SiL	0 - 2	10 YR 51	> 40	Massive	0	1 - 20
					Alluvial deposits	Cr	80 - 110	SL	0 - 2	10 YR 61	2 - 5	Massive	0	1 - 20

^a Soil description according to the terminology of the fourth edition of the guidelines for soil description (Jahn et al., 2006).

^b According to the regional geographic soil database at 1 : 250 000 of Ile-de-France (Roque, 2004). SMU and STU are respectively referring to Soil Map Units and Soil Typological Units.

^c Abbreviations: S, Sand; LS, Loamy Sand; SL, Sandy Loam; SiL, Silt Loam; SiCL, Silty Clay Loam; CL, Clay Loam; L, Loam; SiC, Silty Clay; C, Clay.

Soil identification and horizonation ^a					Particle Size distribution (g.kg ⁻¹) ^b					Bulk density (g.cm ⁻³)	pH (H ₂ O)	CaCO ₃ (g.kg ⁻¹)	OC ^c (g.kg ⁻¹)	Total N (g.kg ⁻¹)	CEC (cmol ⁺ .kg ⁻¹)
SMU	STU	Name	Horizon	Depth (cm)	cl	fsi	csi	fs	cs						
29	85	Stagnic Luvisol on loess	Ap	0 - 42	187	299	451	46	17	1.41	7.6	< 1	8.91	0.94	11.20
			Bt	42 - 60	280	289	383	35	13	1.51	7.7	< 1	4.41	0.57	14.60
			Btg1	60 - 90	298	283	377	33	9	1.54	7.9	< 1	2.89	0.39	16.00
			Btg2	90 - 100	282	246	442	24	6	1.56	8.0	< 1	2.02	0.28	16.20
			BtgC	100 - 115	243	278	451	25	3	1.59	8.1	< 1	1.64	0.23	15.10
			Ckg	115 - 135	204	293	462	19	22	1.58	8.6	106	1.59	0.20	12.80
30	86	Stagnic Luvisol on clay	Ap	0 - 35	191	265	446	65	33	1.51	6.8	< 1	8.64	0.90	10.70
			Bt	35 - 50	284	256	406	41	13	1.54	7.0	< 1	3.25	0.44	14.20
			Btg	50 - 80	290	264	384	47	15	1.59	7.4	< 1	2.39	0.34	16.30
			2Bt	80 - 120	386	222	264	71	57	1.70	7.3	< 1	1.73	0.28	22.00
31	90	Planosol on clay	ApEg	0 - 40	198	255	283	133	131	1.54	6.1	< 1	9.94	0.63	7.71
			Eg	40 - 60	257	261	257	137	88	1.58	7.1	< 1	4.56	0.30	9.24
			2Cg1	60 - 90	527	139	116	101	117	1.57	7.1	< 1	3.03	0.18	23.90
			2Cg2	90 - 120	378	114	125	181	202	1.57	5.1	< 1	1.77		14.50
33	47	Planosol	A1	0 - 5	73	71	107	389	360	0.98	5.5	< 1	51.40	2.89	9.42
			A2	5 - 15	59	78	126	384	353	1.43	4.9	< 1	11.10	0.82	2.22
			Bw	15 - 35	66	87	128	372	347	1.39	4.9	< 1	5.82	0.47	1.57
			C	35 - 45	78	108	164	345	305	1.39	5.1	< 1	3.11	0.28	2.16
			2Cg1	45 - 70	151	220	361	180	88	1.58	5.2	< 1	2.29	0.29	4.75
			2Cg2	70 - 120	143	193	356	195	130	1.58	5.3	< 1	1.68	0.19	4.60
102	56	Gleysol	A	0 - 7	393	242	150	171	44	1.32	6.0	< 1	19.38	1.04	29.60
			Bg1	7 - 20	393	242	150	171	44	1.32	6.0	< 1	19.38	1.04	29.60
			Bg2	20 - 35	527	286	123	41	23	1.57	6.0	< 1	11.01	1.07	28.90
			Bg3	35 - 55	421	340	153	73	13	1.57	6.6	< 1	6.44	0.53	22.30
			Bg4	55 - 80	247	177	194	327	55	1.59	6.8	< 1	5.50	0.39	14.60
			Cr	80 - 110	151	94	110	462	183	1.58	6.4	< 1	3.36	0.18	8.27

^a According to the regional geographic soil database at 1 : 250 000 of Ile-de-France (Roque, 2004). SMU and STU are respectively referring to Soil Map Units and Soil Typological Units.

^b Abbreviations: cl, clay (< 2 µm); fsi, fine silt (2 – 20 µm); csi, coarse silt (20-50 µm); fs, fine sand (50 – 200 µm); cs, coarse sand (200 – 2000 µm).

^c Organic Carbon (OC).

			Winter Rapeseed	Source	Winter wheat	Source
Sowing	Cultivar		Magelan	CERES-EGC database	Roche	CERES-EGC database
	Day of year		246	Terres Inovia, 2019	290	Chambre d'agriculture IdF, 2016
	Density (grain.m ⁻²)		115	Carpentier, 2012	262.3	Chambre d'agriculture IdF, 2016
	Depth (cm)		2.5	Terres Inovia, 2019	2.5	Arvalis – Institut du végétal, 2014a
N inputs	1 st Input	Day of year	62	Terres Inovia, 2019	82	Arvalis – Institut du végétal, 2014b; Interviews
		Amount (kgN.ha ⁻¹)	79	Agrete, 2006	80	Agrete, 2006; Interviews
	2 nd Input	Day of year	84	Terres Inovia, 2019	109	Arvalis – Institut du végétal, 2014b; Interviews
		Amount (kgN.ha ⁻¹)	89	Agrete, 2006	80	Agrete, 2006; Interviews
	3 rd Input	Day of year			137	Arvalis – Institut du végétal, 2014b; Interviews
		Amount (kgN.ha ⁻¹)			40	Agrete, 2006; Interviews
Residues			Incorporation	Terres Inovia, 2019	Exportation	Vaisman et al., 2018; Interviews
Tillage			Yes (30 cm)	Terres Inovia, 2019	Yes (30 cm)	FranceAgriMer, 2012

ES Families	Soil-Based ES	Indicators	Modellings					
			Empirical	Units	Reference	Process-based	Units	Reference
Provisioning	Plant biomass	Yield	Expert-based scoring M-SQR	X	Mueller et al., 2007	CERES-EGC model	t ha ⁻¹ yr ⁻¹	Jones et al., 1986
	Water	Infiltrated water (1m depth)	Expert-based scoring SENSIB	X	Cam et al., 1996	CERES-EGC model	mm yr ⁻¹	Jones et al., 1986
Regulating	Water quality	Non-leached nitrogen	Expert-based scoring MERLIN v2.	X	Aimon-Marie et al., 2001	CERES-EGC model	kgN ha ⁻¹ yr ⁻¹	Jones et al., 1986
	Climate	Carbon (C) storage	Pedotransfer function C saturation	gC kg ⁻¹	Hassink, 1997	CERES-EGC model	tC ha ⁻¹ yr ⁻¹	Jones et al., 1986

ES Families	Soil-Based ES	Modellings					
		Empirical	Units	Source	Process-based	Units	Source
Provisioning	Plant biomass	[0 ; 100]	X	Mueller et al., 2007	[4 ; 12]	t ha ⁻¹ yr ⁻¹	Arvalis – Institut du végétal, 2019
	Water	[0 ; 100]	X	Cam et al., 1996	[55 ; 1119]	mm yr ⁻¹	Therond et al., 2017
Regulating	Water quality	[1 ; 3]	X	Aimon-Marie et al., 2001	[55 ; 100]	%	Therond et al., 2017
	Climate	[4 ; 40]	gC kg ⁻¹	Angers et al., 2011	[-0.4 ; 0.4]	tC ha ⁻¹ yr ⁻¹	Therond et al., 2017

Soil-supported ES	Model	Stagnic Luvisol on loess	Stagnic Luvisol on clay	Planosol on clay	Planosol	Gleysol
Plant biomass	M-SQR	69	57	35	49	40
	CERES-EGC 2000-2018 (t ha ⁻¹ yr ⁻¹)	7.9 (±2.3)	6.9 (±2.3)	6.6 (±2.1)	8.3 (±2.2)	5.3 (±1.5)
	CERES-EGC 2018 (t ha ⁻¹ yr ⁻¹)	8.6				
	Measure (t ha ⁻¹ yr ⁻¹)	8.5				
Water	SENSIB	47	29	30	53	85
	CERES-EGC 2000-2018 (mm yr ⁻¹)	236 (±120)	233 (±121)	280 (±120)	230 (±113)	274 (±120)
	CERES-EGC 2018 (mm yr ⁻¹)	322				
	Measure (mm yr ⁻¹)	334				
Water quality	MERLIN v2.	1	1	1	1	1
	CERES-EGC 2000-2018 (%)	92 (±4)	97 (±2)	95 (±2)	83 (±8)	94 (±3)
	CERES-EGC 2018 (%)	87				
	Measure (%)	87				
Climate	C _{sat} (gC kg ⁻¹)	22.1	21.0	20.9	9.5	29.8
	CERES-EGC 2000-2018 (tC ha ⁻¹ yr ⁻¹)	0.297	0.314	0.302	0.082	0.247