

Aridity index, soil erosion and climate drive no-till ecosystem services trade-off in Mediterranean arable land

Giorgio Baiamonte*, Luciano Gristina, Mario Minacapilli, Agata Novara

Department of Agricultural, Food and Forest Sciences (SAAF), University of Palermo, Viale delle Scienze, 90128 Palermo, Italy

*Corresponding author (giorgio.baiamonte@unipa.it)

Abstract

Despite the relevant literature on the advantages of no-till (*NT*) management, the world area under *NT* is only 10 % of the arable land, probably due to several limiting factors as climate, soil types, farming systems and yield. Soil conservation practices and particularly *NT* soil management are able to provide many ecosystem services (*ESs*). This paper suggests a framework to determine the area distribution of soil erosion control and food security *ESs* trade-offs furnished by *NT*, starting from the potential soil erosion and aridity index maps. The interaction between the potential soil erosion and the aridity index showed that different trade-offs and synergies of multiple *ESs* may occur and need to be interpreted. The yield success probability of *NT* with respect to *CT* ($P(NT)$) and soil erosion tolerance (*SET*) were considered to determine four classes of the *NT* adaptability index, which represent the *NT* ability to support multiple *ESs*, under the environmental condition. For 2020 as a baseline, and 2050 as a future scenario, in between latitudes 35° and 47°, an analysis was carried out, focusing on the arable lands of five European Mediterranean countries (France, Greece, Italy, Portugal and Spain), which are potentially cropped with durum wheat. Results showed that, for the

considered arable land area, under *NT* soil management and from a food security point of view, the potential durum wheat area, under climate change pressure, slightly increases, if residues are retained. On the contrary, if residues are removed, the potential durum wheat area is very limited for both the baseline and the 2050 scenario. The area distribution corresponding to the *NT* adaptability index classes could be suitable for stakeholders to modulate payments for agri-environmental measures, based on the actual provided *ESs*.

Keywords: Mediterranean climate, no-till, soil erosion, multifunctional agriculture, ecosystem services payment.

1. Introduction

The assessment and the improvement of ecosystem services (*ESs*) play an important role in the identification of strategies aimed at ensuring food security and environmental resources conservation. Because of evident linkages among different *ESs*, the improvement of one *ES* (e.g. the yield for food security) can determine the decline of the other one and, in the case of contrasting effects under the same driving forces, a trade-off may occur (e.g. climate condition and soil management). For example, trade-offs occur between water quality and agricultural production, when nitrogen is added to the soil, or when organic agriculture improves pollination, biological control and nutrient cycling, but generally decreases yield, contrasting world population growing needs (Lal, 2013). Therefore, understanding relationships between *ESs* and their trade-offs could be useful to estimate, in time and space, *ESs* benefits and also specific disservices (Bennett et al., 2009).

Generally, soil conservation practices and particularly no-till (*NT*) soil management can provide several kinds of *ESs*, such as carbon sequestration, biodiversity, elemental cycling, water provision (quality and quantity), soil conservation (structure and fertility), which are generally difficult to quantify, with exception of food security (e.g. by grain yield improvement) and ecological

benefits (Lal, 2013; Das et al., 2018). In the near future, under the effect of global climate change, the strategic goals of agronomic choices will address, besides to increase yield the yields, along with the stabilization and its maximization, depending on the possibility of the specific environments, including environmental protection (soil and water conservation, nitrate pollution, CO₂ sequestration, etc.).

In Mediterranean environments, the importance of soil moisture, with regard to management practices, has been assessed through reducing tillage, *NT* system and straw management (Kassam et al., 2012), especially in semi-arid regions that are characterized by low values of the aridity index (*AI*), as a consequence of low annual rainfall and high evapotranspiration demand (Greece, South Italy and Spain). Some studies showed that *NT* success probability compared to conventional tillage (*CT*) highly depends on *AI* (Gristina et al., 2018; Baiamonte et al., 2019; Bahri et al., 2019). In fact, when precipitation was less than 300 mm, during the durum wheat cycle (November–May) (De Vita et al., 2007), *NT* resulted in higher yields and water use efficiencies. *AI* is the most important factor influencing *NT* yield response (Ruisi et al., 2014; Pittelkow et al., 2015) that generally was found greater for *NT* than *CT* (Gristina et al., 2018), at low *AI* values (semi-arid environment).

As far as other *ESs* provided by *NT*, there is no evidence on soil organic carbon (*SOC*) sequestration capacity and greenhouse gases reduction (Powlson et al., 2014; Neufeldt et al., 2015). On the contrary, soil erosion processes are well controlled by *NT* in comparison to *CT* soil management, responding to environment conservation and food security, which are considered the most important and evaluable *ESs* (Lal, 2013). In a study carried out in the Upper Sangamon River Watershed (central Illinois), Lee et al. (2021) found that *NT* adoption determined a sediment reduction up to 76%, and indicated that *NT* was a key factor in reducing soil erosion. Moreover, in an arid environment, Bahri et al. (2019) also found that *NT* was more effective than *CT* in preventing water erosion, determining a reduction of soil loss from 4.6 t ha⁻¹y⁻¹ to 1.7 t ha⁻¹y⁻¹.

Specifically, considering yield and soil erosion controls as the main *ESs* provided by *NT*, we hypothesize a site-specific trade-off between the two identified *ESs*, because of the variability of

climate in space and time (influencing yield), and of soil morphology (influencing soil erosion). Although many research analysed the effect of *NT* on a single *ES*, there are no studies on the evaluation of interactions of soil erosion and durum wheat yield, driven by *AI*, neither small nor large scale. In this context, mapping the *ESs* provided by *NT* in the European Mediterranean regions could be useful to forecast *NT* application advantages, providing suitable areas of *NT* adoption in the present and in the near future (Bennett et al., 2009; Gristina et al., 2018; Baiamonte et al., 2019), which could be suitable for stakeholders to modulate payments for agri-environmental measures.

The main objective of this paper is to explore the *NT* effect on some *ESs*, stressing empirical patterns of trade-offs and synergies, across European Mediterranean regions. Specific aims are i) determining *NT* potentiality through the quantification of the suitable *NT* area in Europe, in 2020 and under a climate change scenario (2050); ii) evaluating yield improvement and soil erosion as *ESs* provided by durum wheat *NT* in relation to *AI*; iii) suggesting a conceptual criterion to diversify agri-environmental measures payment in relation to the effective provided *ESs*.

2. Materials and Methods

In order to define for durum wheat, the suitable *NT* areas in comparison to *CT* for Mediterranean environment, among the possible different *ESs*, and because of the aforementioned easy quantification, two provided *ESs* by *NT* were taken into account: i) soil erosion and ii) yield success probability ($P(NT)$) (Baiamonte et al., 2019). In this investigation, durum wheat monocropping and two residues management practices (removed and retained) were considered.

All input datasets used in this study were retrieved from geographic and meteorological databases (Table 1) that have been developed to support the scientific community, in between latitudes 35° and 47° and focusing on the arable lands of Mediterranean countries (France, Greece, Italy, Portugal and Spain), which are potentially cropped by durum wheat.

103 The Copernicus Global Land Service (CGLS) of the European Community was chosen to
 104 obtain the spatial distribution of arable land in the European countries of the Mediterranean area
 105 (Büttner and Kosztra, 2018). In particular, the 211 legend class of CGLS CORINE was used to map
 106 arable land with a resolution of 100 m x 100 m, as a reference map for the study area (Figure 1a). In
 107 this study, the reference map indicates the arable land area that could be potentially durum wheat
 108 cultivated. The European Soil Database (ESDB-ESDAC) (Figure 1b) was used as a source of high
 109 spatial resolution (100 m x 100 m) of soil erosion rates, derived by Panagos et al. (2012), by using
 110 the RUSLE (Revised Universal Soil Loss Equation) model (Renard et al., 1997). The historical
 111 climate database for Europe, ClimateEU, was used to retrieve two key meteorological variables
 112 (Hamann and Wang, 2005), the yearly rainfall, R_a (Figure 1c) and the yearly reference
 113 evapotranspiration, ET_0 , (Figure 1d) (Allen et al., 1998). These two latter were generated using a
 114 spatial resolution of 1 km x 1 km for the entire study area. All input data were managed using standard
 115 geographical information system (GIS) techniques. In particular, data was stored in a raster format
 116 (geotiff) and projected in a unique geographical system (ETRS89 Lambert Azimuthal Equal Area).
 117 Raster map-algebra techniques were used to overlap all raster data and to calculate the investigated
 118 outputs (Figure 2).

119

120 **2.1. Aridity Index, NT yield probability and Soil erosion**

121

122 Meteorological input data retrieved from the ClimateEU database was used to obtain the spatial
 123 distribution of AI (UNEP, 1992) defined as:

124

$$125 \quad AI = \frac{R_a}{ET_0} \quad (1)$$

126

127 where R_a (mm) is the yearly rainfall and ET_0 (mm) is the yearly reference evapotranspiration (Allen
 128 et al., 1998). For both variables (R_a and ET_0), in order to take into account two Mediterranean

climates, 2020 as a baseline, and the 2050 temporal projection were used, by considering the PRISM (Parameter Regression of Independent Slopes Model) interpolation method (Knutti et al., 2013).

The probability of *NT* success, $P(NT)$, was calculated according to Toliver et al. (2012) using the following relationship:

$$P(NT) = \frac{\exp(\text{Logit } NT)}{1 + \exp(\text{Logit } NT)} \quad (2)$$

where $P(NT)$ is equal to 0.5 when the *CT* yield success probability equals the *NT* probability. According to Gristina et al. (2018), the Logit equation (Eq. 2) can be expressed as a function of *AI* as:

$$\text{Logit } NT = a + b \text{ } RM + c \text{ } AI \quad (3)$$

where *Logit NT* is the success of *NT* (*Logit NT* = 0 when *CT* yields more than *NT*, *Logit NT* = 1 when *CT* yields less than *NT*); *RM* is a dichotomous variable indicating the residue management (1 = residue retained, 0 = residue removed); *a*, *b* and *c* are numerical coefficients equal to 0.884, 0.506 and -2.20, respectively. For $P(NT) = 0.5$, the corresponding *AI* threshold, *TAI*, in two cropping systems, monocropping residue removed and crop rotation residue retained, equals 0.45 and 0.625, respectively. Further details of the shortly summarized procedure can be found in Gristina et al. (2018) and in Baiamonte et al. (2019).

For the same investigated Mediterranean countries, the ESB/ESDAC soil erosion database (Panagos et al., 2012) was used to map the spatial distributions of soil erosion over the arable lands (Figure 1a). Using standard GIS techniques and starting from the European map of RUSLE soil erosion, the corresponding spatial distribution of the study area (Figure 1b), overlapping the arable lands, was derived. The availability of different temporal climatic projections inside the ESB/ESDAC

154 database made it possible to consider the soil erosion estimation for both the baseline 2020 and the
155 2050 scenario.

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157 **2.2. No-till adaptability index**

158

159 Introducing two threshold values, one for $P(NT)$ and one for soil erosion tolerance (SET),
160 determined four classes of the NT adaptability index (Figure 2), which represent the NT ability to
161 support one or multiple ES s, under the same driving forces. Threshold value $P(NT) = 0.5$ and $SET =$
162 $3 \text{ t ha}^{-1} \text{ y}^{-1}$ (Bazzoffi, 2009) were established, respectively.

163 The flowchart plotted in Figure 2 describes the methodology applied in this study, whereas
164 Table 2 reports the NT adaptability index, corresponding to each provided ES . In particular, the four
165 considered NT adaptability index classes were: i) areas where CT yields more than NT and limited
166 risk, or no risk, of soil erosion occur; ii) areas where NT provides only food security ecosystem
167 service, iii) areas where NT provides only soil erosion control ecosystem service; and iv) areas where
168 NT provides both the two proposed ecosystem services (NT soil erosion control and food security).

169

170 **3. Results**

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172 For the considered countries, Figure 3 shows the spatial distribution of the aridity index (AI),
173 both in 2020 and in the simulated 2050 scenario. In the right, the corresponding area distribution is
174 graphed, which is characterized by a bimodal shape. Simulated data for the 2050 scenario shows
175 negligible differences in AI with respect to 2020, with some decreases for South Italy, Spain and
176 Greece. Lower AI values mainly cover South Italy, Spain, and North Greece. For Spain, Italy and
177 Greece, the areas characterized by AI values lower than 0.625 (TAI for durum wheat monocropping
178 residue retained), increased from 126576 km^2 in 2020 to 134547 km^2 in 2050, and for AI values lower

179 than 0.45 (TAI for durum wheat monocropping residue removed), from 73432 km² in 2020 to 88098
180 km² in 2050.

181 Figure 4a and Figure 4b show the soil erosion spatial distribution and their frequencies in 2020
182 and 2050, respectively. On average, soil erosion is characterized by slight temporal variability in the
183 study area, with mean values 3.9 t ha⁻¹ y⁻¹, for 2020, and 4.2 t ha⁻¹ y⁻¹, for 2050. Despite the slight
184 differences in the mean values of soil erosion, it is interesting that, at regional scale, areas
185 characterized by values less than SET (3 t ha⁻¹ y⁻¹), decrease from 2020 to 2050, for France (-5 %),
186 Greece (-21 %), Portugal (-19 %) and Spain (-8 %), whereas increases for Italy (24 %), where soil
187 erosion less than SET results 36964 km², in 2020, and 45711 km², in 2050. Moreover, significant
188 variations of soil erosion occur in Italy, where soil erosion decreases moving from the South to the
189 North, because of the lower rainfall in the North than in the South area.

190 Figure 5 illustrates the $P(NT)$ spatial distributions calculated according to Eq. (2) and Eq. (3).
191 As can be observed, for 2020 (a,b), the NT success probability, $P(NT)$, strongly depends on residues
192 management, ranging on average from 41 % to 54 % for removed and retained residues, respectively,
193 whereas the 2050 scenario (c,d) shows a slight increase of $P(NT)$ with respect to 2020. France and
194 Italy have the highest benefit when residues are retained in the soil and in relation to the different
195 climate.

196 Areas with the lowest AI values (Spain and Greece, see Figure 3) showed a limited increase
197 in the case of both residue management and future scenario analysis. Generally, for any country, the
198 highest areas characterized by $P(NT)$ less than the threshold ($P(NT) = 0.5$) were detected for France,
199 Spain and Italy, especially for residue removed.

200 For both residue retained and residue removed management practices, France and Portugal
201 showed no significant differences (> 5%) in $P(NT)$ area distribution, between 2020 and 2050.
202 Whereas, when adopting residue removed management practice, Greece, Italy and Spain area
203 differences vary from 28 % to 43 %, from 8 % to 13 %, and from 20 to 19 %, respectively, than
204 residue retained.

205 For the food security *ES*, advantages of *NT* adoption arise for *AI* values lower than 0.625, for
206 residue retained management practice, and for *AI* values lower than 0.45, for residue removed. These
207 values denote that *TAI* represents the threshold of *NT* success probability ($P(NT) = 0.5$) for yield, thus
208 for $P(NT) < 0.5$, *NT* management provides a real contribution only for soil erosion control.

209 For the study-case, Figure 6 illustrates the spatial distributions of the *NT* adaptability index, for
210 durum wheat mono-cropping systems, in relation to residues management and climate change
211 projection (left panels). This analysis was thus obtained by following the methodology illustrated in
212 Figure 2, i.e. by considering the indicated threshold values for both $P(NT)$ and soil erosion, $P(NT) =$
213 0.5 and $SET = 3 \text{ t ha}^{-1} \text{ y}^{-1}$, respectively.

214 Similarly to Figure 5, Figure 6 plots the area distribution of the *NT* adaptability index in relation
215 to the *AI* (right panels). The difference between these distributions mainly depends on the
216 management practice rather than on the temporal projection. Moreover, Figure 6 shows that in 2020
217 (a,b) and for residual retained management practice, *CT* productive capacity is absent below the
218 threshold value, $AI = 0.45$. The two climatic contexts show an increase in the area suitable for *NT*,
219 especially when residues are removed. The residue removed management doubles the area where *CT*
220 yield is higher than *NT*, for both 2020 and 2050 (Table 3). In the five considered countries (95% of
221 the durum wheat area in Europe), removing residues determine about 60,000 km² of *CT* cultivation
222 increase, excluding France, where the area remains constant. Climate contexts (2020 and 2050)
223 substantially do not modify areas suitable for *CT*.

224 For each investigated country, the standardization of the areas of *NT* adaptability index classes,
225 shows that removing residues determines an increase of the area suitable for *CT* (Figure 7). In fact,
226 with exception of France, where the *CT* area remains constant between the two residue management,
227 residues removed determines an increase up to 50 % of the areas devoted to *CT*. Removing residues
228 also allows (both in 2020 and 2050) uniformity of the *NT* adaptability index classes, with exception
229 of Spain and Greece. Only in these two countries, an area lower than 20 % and 5 %, respectively,

remains suitable for *NT* aimed at food security. A small reduction of the area suitable for *CT* occurs in the 2050 scenario due to the projected aridity index variation.

For the five considered countries, for the two residue management practices and the two contexts (2020 and 2050), Figure 8 plots the area distribution as a function of the *AI*, for the *NT* adaptability index classes. The figure shows that unimodal distributions can be observed for France, Portugal and Spain, whereas bimodal or multimodal distributions correspond to Greece and Italy, reflecting the high climatic variability of Greece and Italy than the other countries.

Differently from Table 3, where the *AI* effect was not evidenced, the area distributions plotted in Figure 8 made it possible to determine, for each country, the *AI* corresponding range. Thus, these *AI* ranges refer to the area that could potentially benefit from the different *ESs*, described by the adaptability index class. As an example, for Spain, the peak area corresponds to $AI = 0.3$ (*NT* food security), for France to $AI = 0.67$ (*NT* soil erosion), and for Italy, according to the multimodal area distribution, to $0.4 < AI < 0.9$ (*NT* soil erosion). Interestingly, residue retained management benefit appears slight only for Spain and Italy (*NT* soil erosion), limiting the latter to low *AI* values ($AI < 0.5$). Generally, with respect to the whole area of investigated arable land (223,319 km²), if residues are retained the potential durum wheat area is about 116,000 km² and under climate change pressure will be 123,000 km². Whereas, the potential durum wheat areas are only 33,000 km² and 44,000 km², in 2020 and 2050, respectively, if residues are removed.

4. Discussion

Several reasons are supporting *NT* adoption and they can be summarized as follows: i) better farm income; ii) flexible technical possibilities (sowing, fertilizer application, weed control, etc.); iii) soil erosion control; iv) better nutrient use efficiency; v) better water-use efficiency; vi) equal or increasing yields and greater yield stability in dryland areas.

In the near future, under the effect of global climate change, results of this work show that the

suitable area for *NT* adoption will increase, suggesting *NT* management as the most appropriate adaptation strategy for agricultural production systems, and not a strategic management tool aimed at mitigation. In fact, compared to ploughing, carbon sequestration does not increase under *NT* and may vary widely with crop, soil and climate, which are not fully established yet. Climatic change may compromise food security in an increasing food demand scenario. Uncertainty in seasonal climatic trends may determine an increase in risk and uncertainty in the primary production, but also may determine changing areas of traditional cultivation and/or its expansion or reduction. In other words, *NT* increases the resilience of agricultural production and is a new emergence to ensure food security in the near future, and at the same time to improve agriculture (Bahri et al., 2019). Both environmental and economic factors may be a driving force for future *NT* adoption, provided *NT* is actually supported by government environmental policies. Further quantification of all the environmental impacts of *NT* is needed, particularly with respect to the influence of climate forcing, water pollution and soil carbon erosion.

Despite the several benefits of *NT* adoption, it is not widespread, because its efficacy is a site-specific policy, and often its application is not economically affordable, in comparison to traditional management. For these reasons, payment levels should be set sufficiently high to attract farmers and to avoid over-compensation. This requires appropriate analysis of efficient payments that have to be proportional to acquired benefits (environmental, economics, etc.), being strongly dependent on the climatic characteristics, as the above findings showed.

The economic gain obtained by food production is of interest for the society and for the farmers, while the soil protection through erosion control, in the short run, is not perceived as an important benefit by farmers, but as a cost. Erosion control, which is important for maintaining ecosystem health, is therefore an *ES* that should be promoted for the interest of the whole community (Rouabhi et al., 2019). In light of this, the approach for designing ecosystem service payments should maximize the efficiency of public spending of environmental policies, and avoids the risk of inequity in the case of multiple *ESs* (Galati et al., 2015, 2016). In fact, according to Bannet et al. (2009), the

282 driver effect on multiple *ESs* can be independent or shared, and their interactions can be weak or
283 strong, and unidirectional or bidirectional, thus they need to be addressed.

284 Results demonstrated that the provided *ESs* relationships, considering specific soil management
285 (i.e. *NT* management adoption), depend on environmental factors (aridity index) and *ESs* thresholds
286 values, determining trade-off conditions.

287 Figure 9 illustrates a schematic representation of synergy and trade-off of *ESs*, and the suggested
288 approaches for agri-environmental payment and soil management adoption. For example, *NT*
289 adoption has an impact in opposite direction on both *ESs*, e.g. food security and erosion risk (red and
290 green arrow in Figure 9, respectively), when soil erosion risk is high and the *AI* is also high, while
291 the impact is in the same positive direction if the *AI* is low (Figure 9) and the impact of *NT* provides
292 erosion control benefits in areas where *SE* risk is high. Knowledge on multiple services relationships,
293 but also the quantification of trade-offs, lead to an efficient application of Agri-Environmental
294 Measures (*AEM*) and payment ecosystem services (*PES*), for both farmers and environment (social
295 cost) (European Commission, 2005).

296 At the regional level, an approach for designing incentives based on the provided multiple *ESs*
297 was proposed. This approach considers both a constant (K_{NT}) and a variable aliquot ($Q \cdot SE$), as
298 displayed in Figure 9. The constant aliquot, K_{NT} , should be provided to farmers in order to encourage
299 the application of *NT* practice, in areas where $P(NT)$ is higher than the threshold, or in the case of not
300 synergic (trade-off) ecosystem services; whereas the variable aliquot, $Q \cdot SE$, should be provided for
301 each Mg ha^{-1} of the potential saved soil erosion (*SE*, Figure 9), with Q the unit cost of soil loss.

302 The introduction of the constant aliquot is considered helpful, although the farmers have a high
303 probability to increase the yield using *CT* in the area with high *AI*. In fact, this constant aliquot is
304 necessary to encourage farmers to abandon the traditional management or to change the farm
305 equipment useful to apply the new management.

306 The variable aliquot of the payment is addressed only for farms with high erosion risk (higher
307 than the *SE* threshold) and it is a function of the effective *ES* provided by *NT* application to soil

erosion control. No incentives will be provided for farms where *CT* management is suggested (Figure 9).

5. Conclusion

Despite the relevant literature on the advantages of *NT* management, the world area under *NT* is only 10 % of arable land (Kassam et al., 2019). Reasons for the limited *NT* adoption depend on several factors and particularly on climate, soil types, farming systems, production means availability (*NT* seeding machine, herbicides, etc.), competing uses of crop residues for fodder and fuel, as well for the more skill and experience needed for *NT* adoption.

Evidence of this work showed that, with respect to the considered arable land, a durum wheat area almost equal to 50% was suitable for food security, if residues are retained, and a small increase in the suitable area was observed in the near future (2050); whereas a relatively small potential durum wheat areas was detected, if residues are removed in both 2020 and 2050. The use of *NT* in sloping areas contributes to providing another *ES*, thanks to the reduction in soil erosion.

Considering the overall benefits of *NT* soil management, incentives should be provided, designing payments through the exact knowledge of benefits (environmental – soil erosion control and income – food security) that society and/or farmers may have, by adopting *NT* soil management. The effectiveness of *NT* soil management on *ESs* mainly depends on climatic condition but to a lesser extent also on other site-specific conditions and other key drivers (e.g., soil properties, irrigation techniques, and fertilizer applications). Therefore, the proposed approach aimed at achieving high efficiency of public spending, differentiating the incentives in relation to the real benefit provided by each site. In any case, the payment approach introduced in this study aims at encouraging the adoption of *NT* also when a trade-off between *ESs* occurs (i.e. high *AI* and *SE*).

Furthermore, this approach could be strategic support in the case of interactions of regional and local actors and stakeholders to design and implement *AEM* programs, making also

334 possible flexibility in the application of national programs to local conditions and guaranteeing a more
335 targeted and efficient payment system.

336 In the near future, when the economic benefits at the farm level will be determined, this
337 procedure could be useful to check and implement other provided *ESs* not easy to be estimated.
338 However, this will determine further synergies and trade-offs increasing the complexity of the
339 analysis to be performed as well as the choice of the best management practice.

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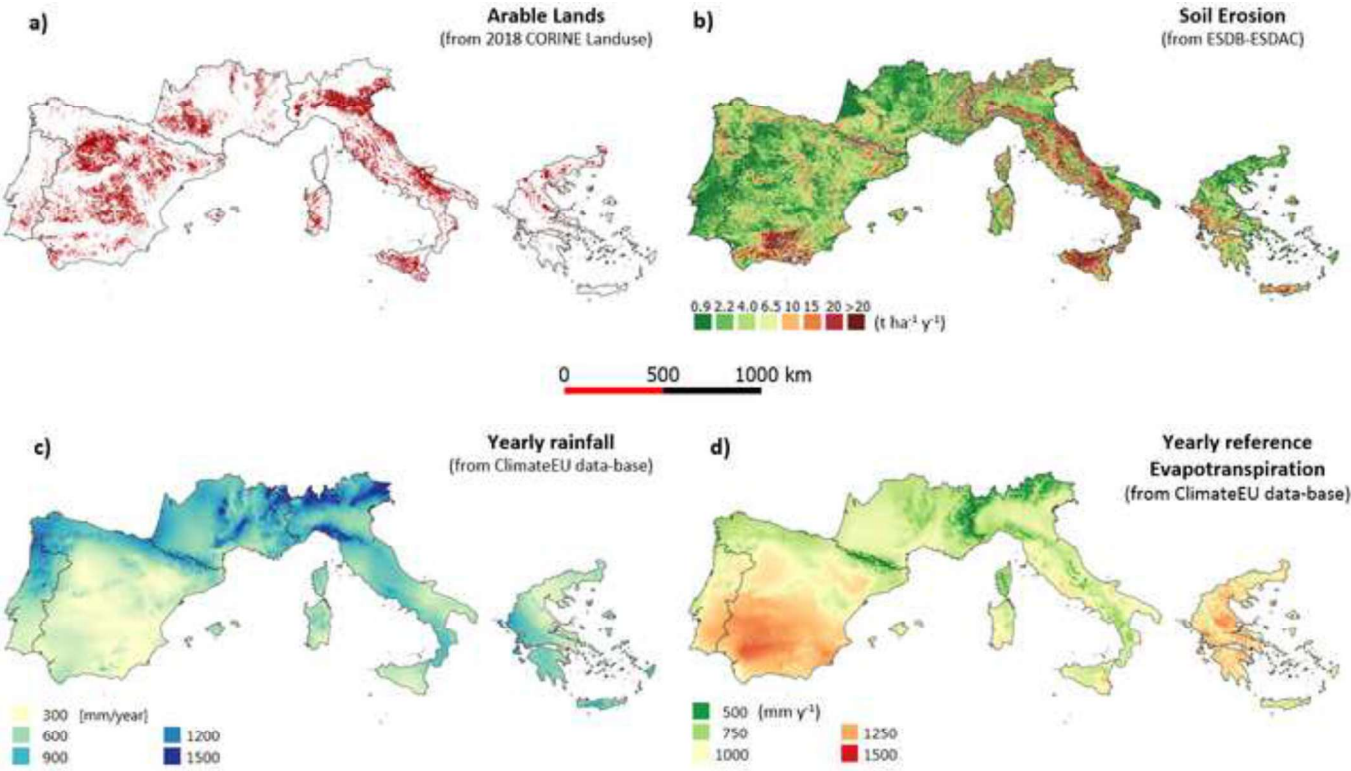
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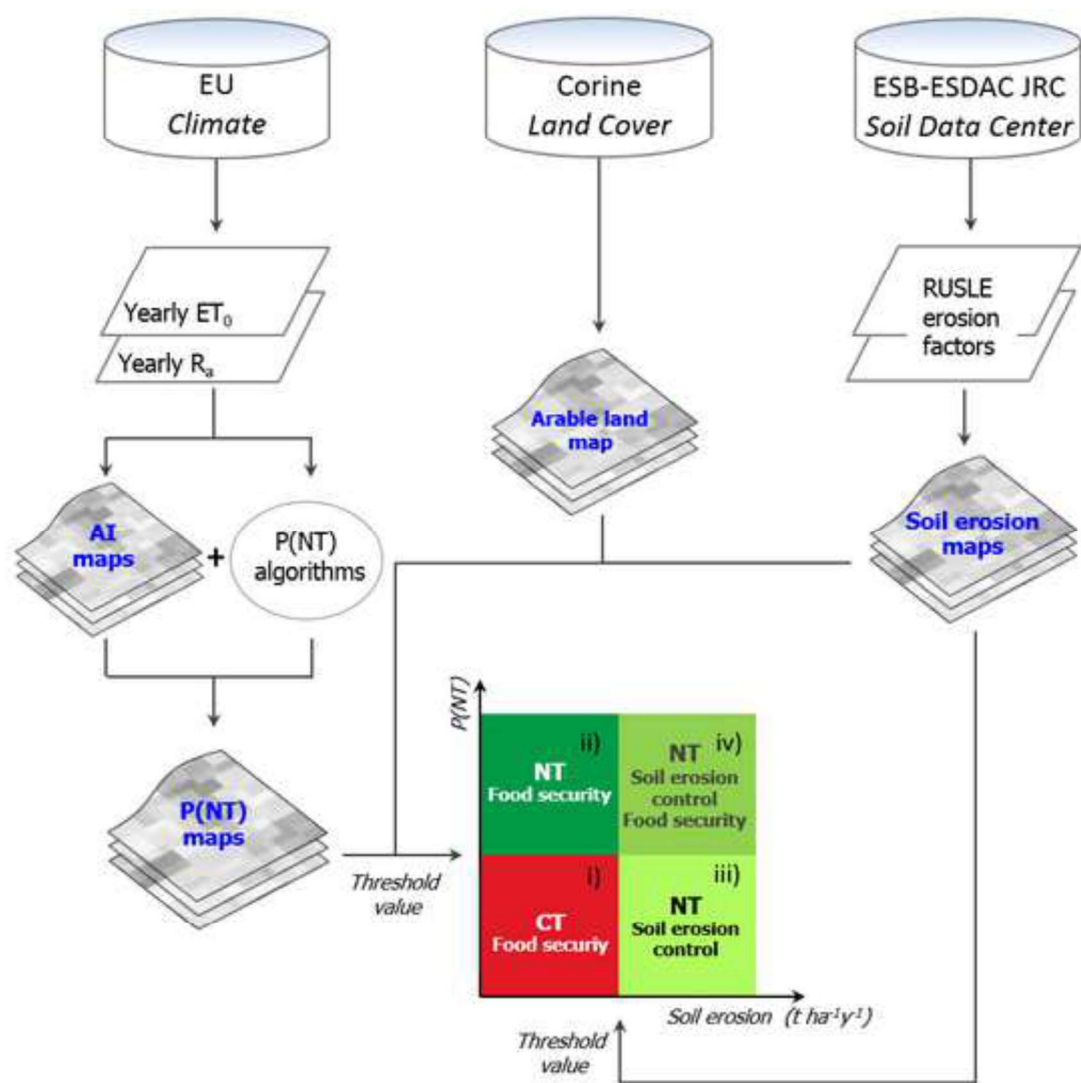
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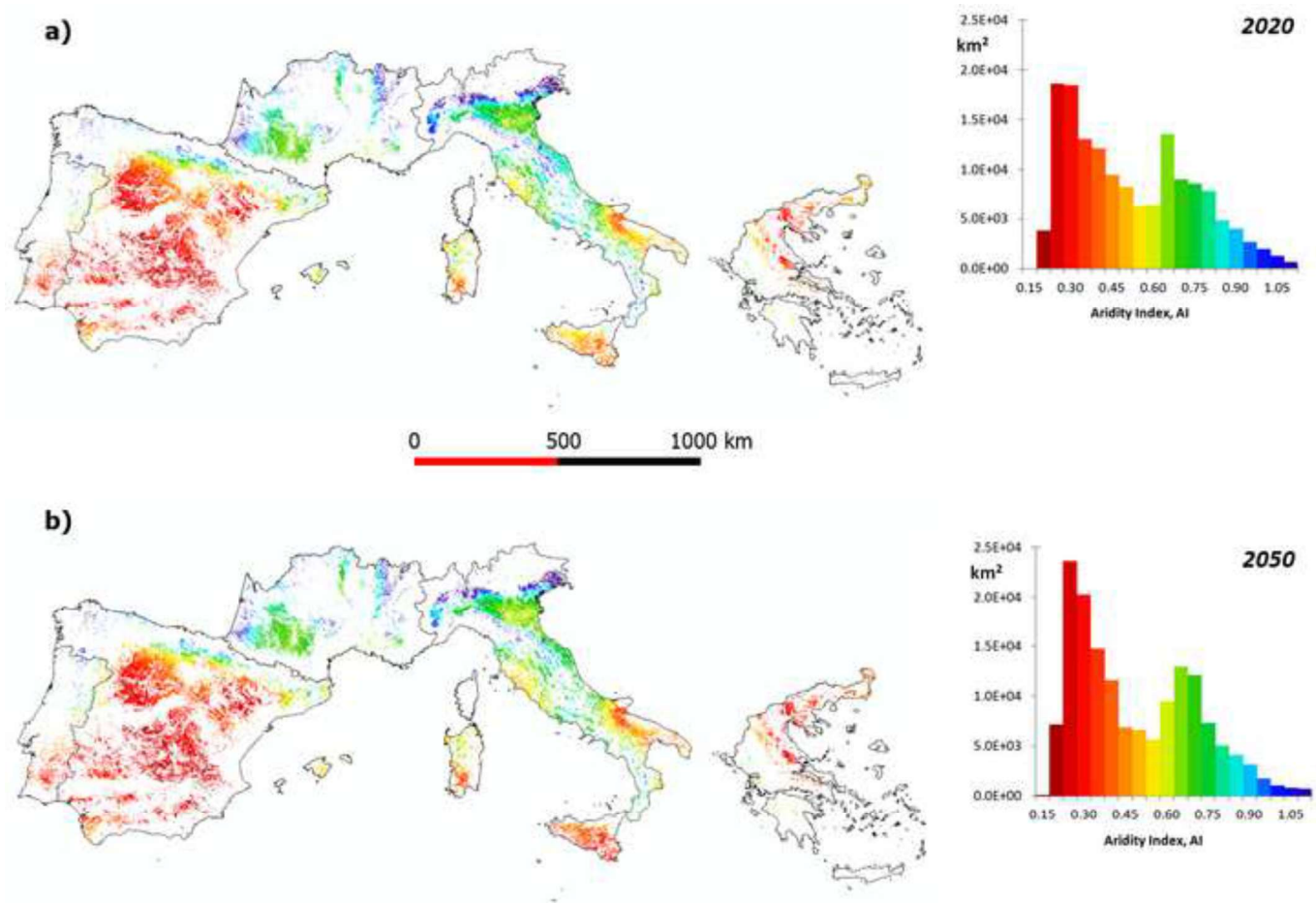
Figure_1a-d



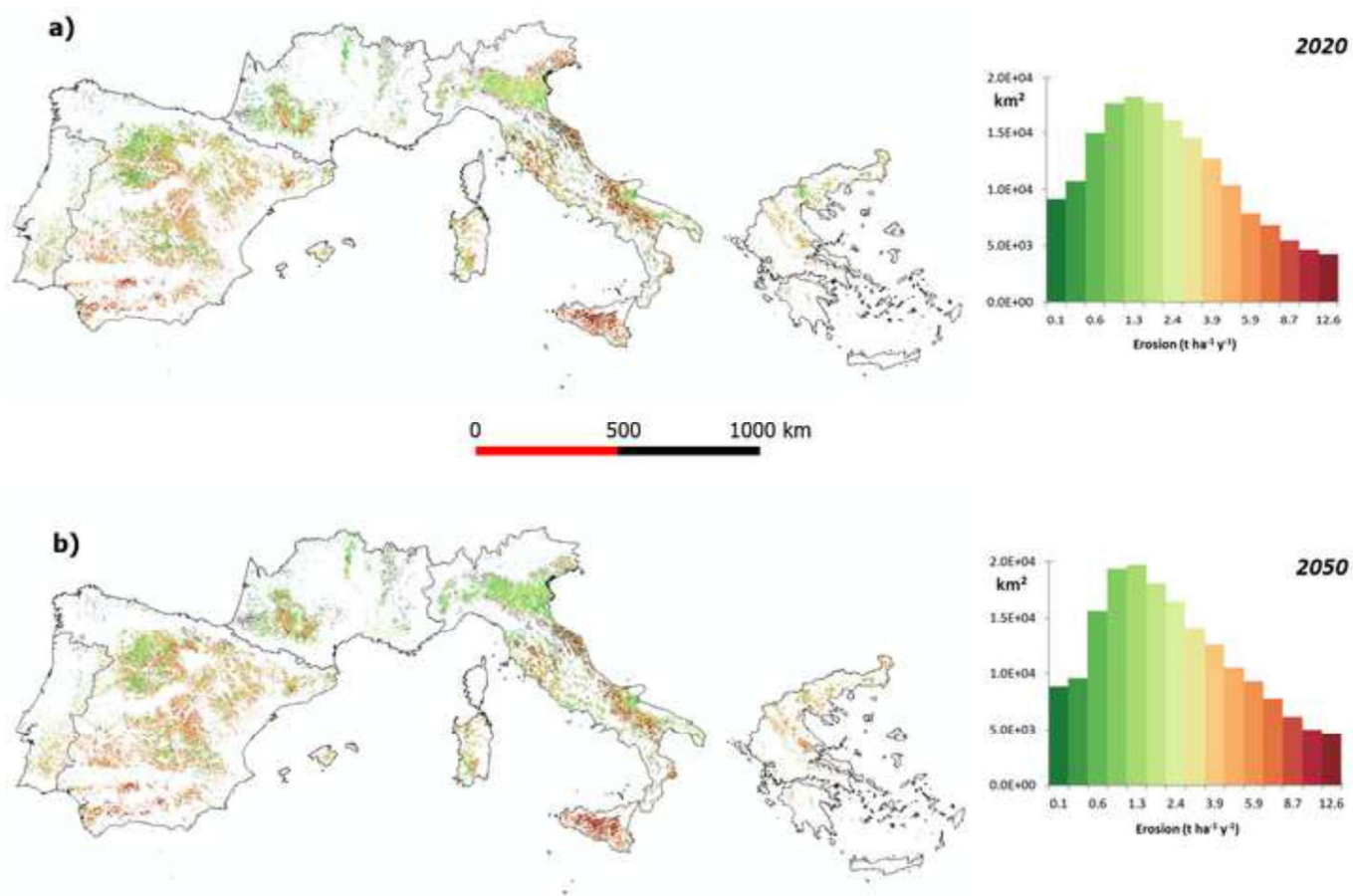
Figure_2



Figure_3a-b

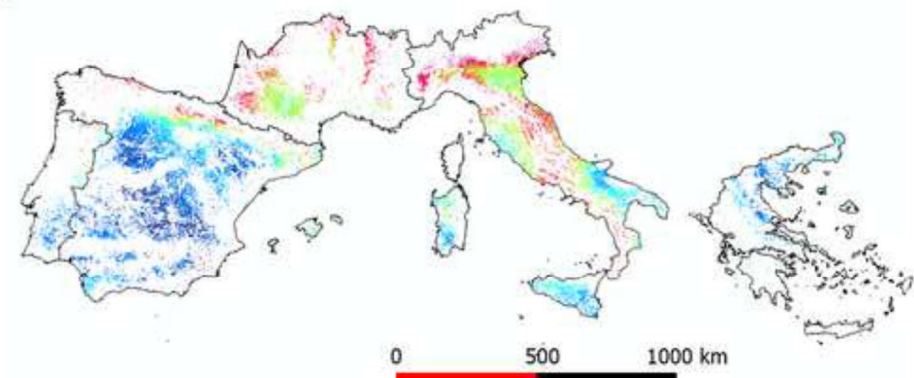


Figure_4a-b

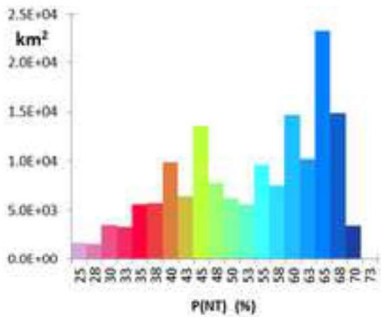


Figure_5a-b

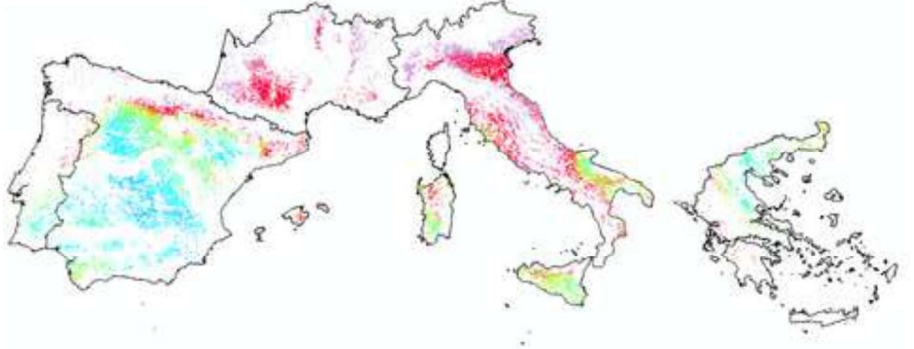
a) Residues Retained



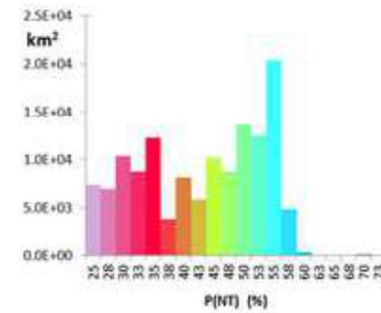
2020



b) Residues Removed

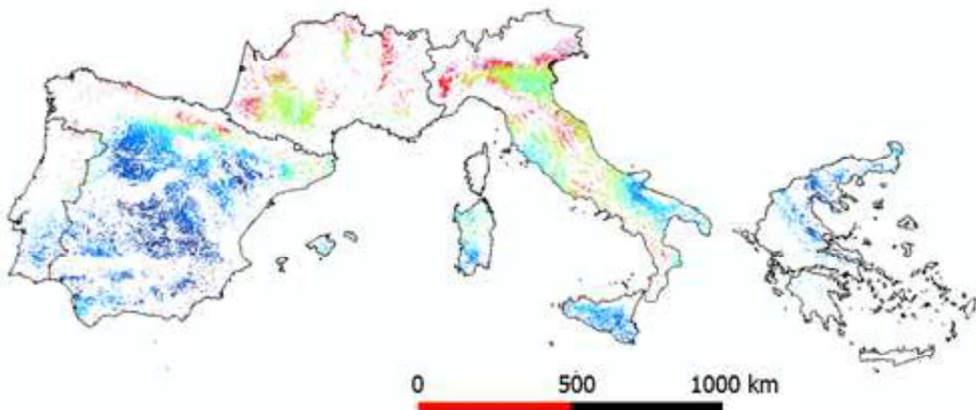


2020

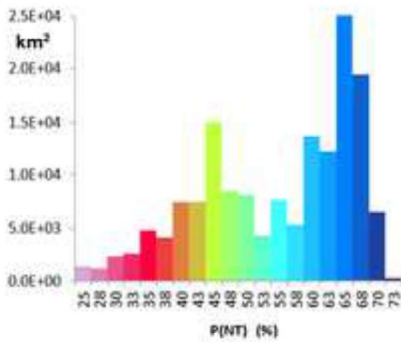


Figure_5cd_REV2

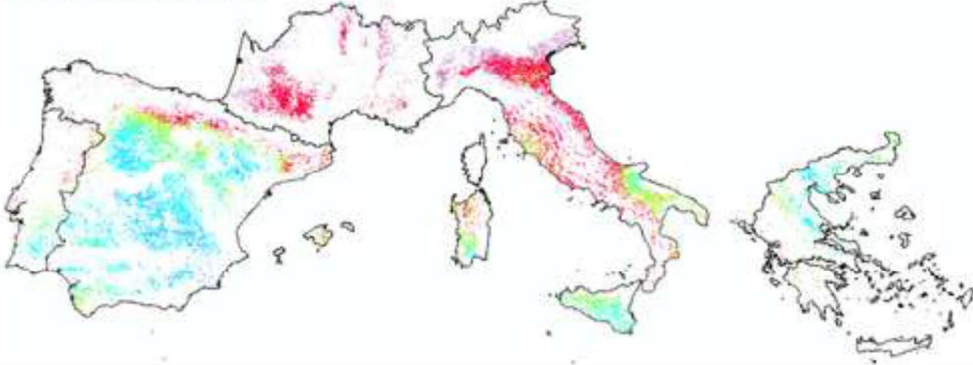
c) Residues Retained



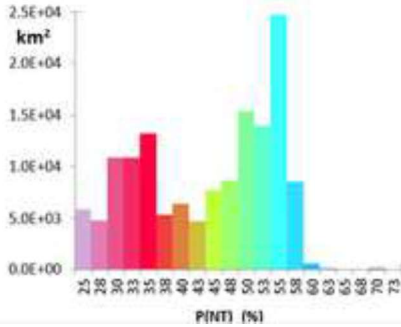
2050



d) Residues Removed

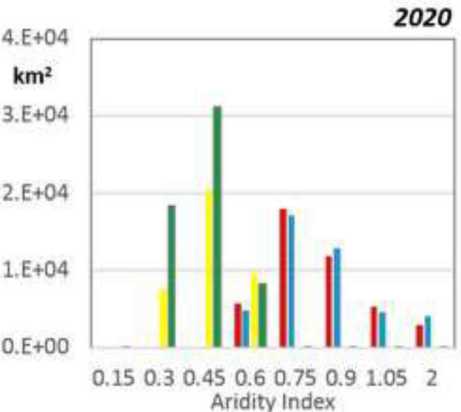
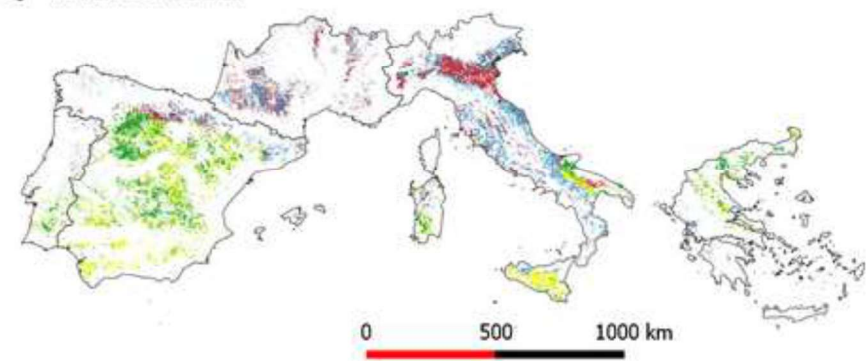


2050

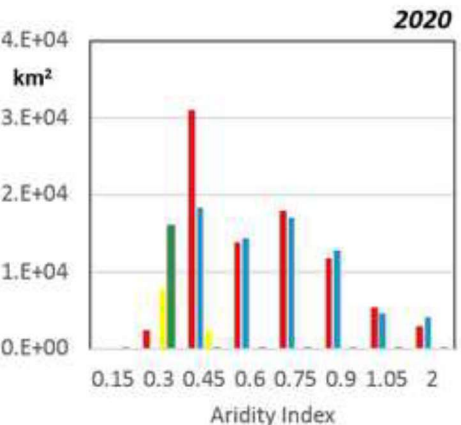
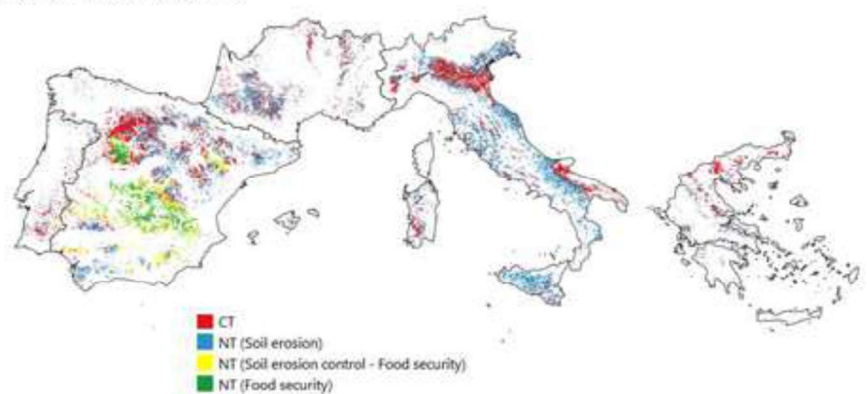


Figure_6a-b

a) Residues Retained

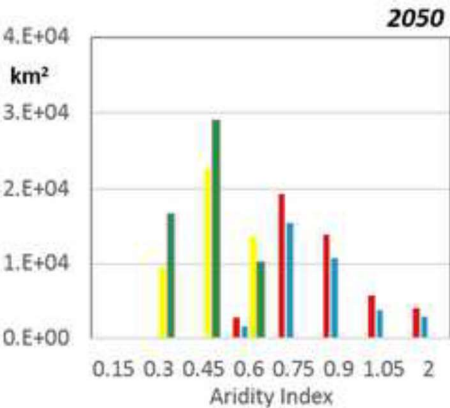
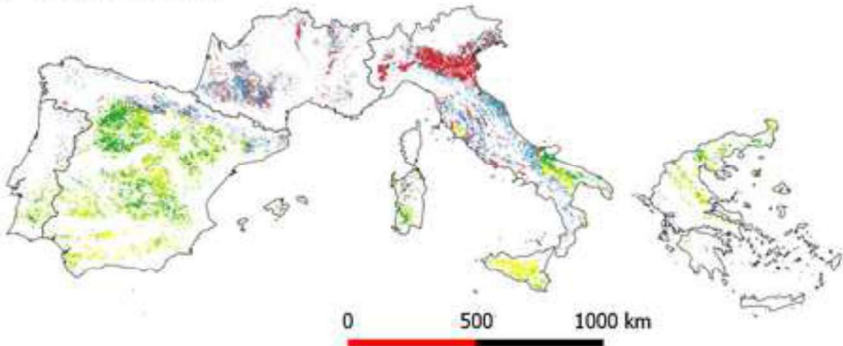


b) Residues Removed

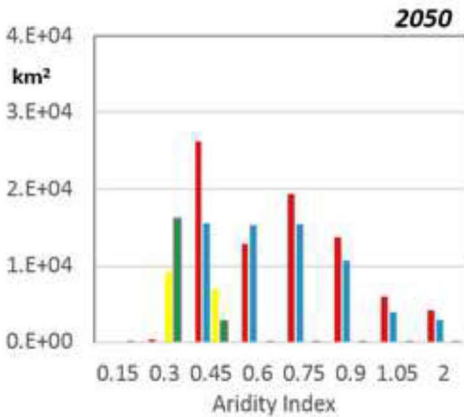
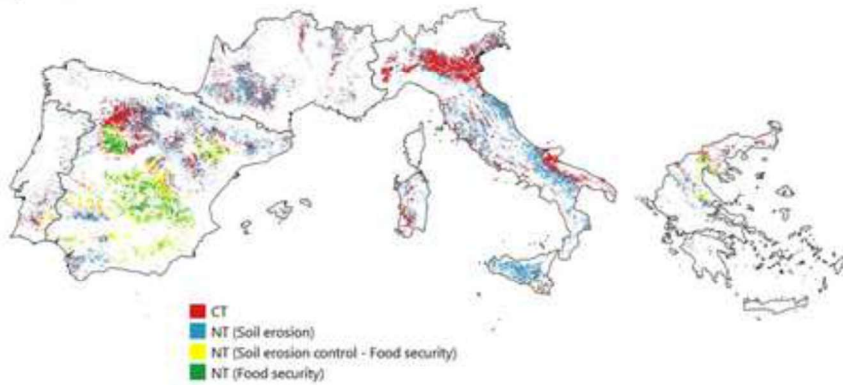


Figure_6c-d

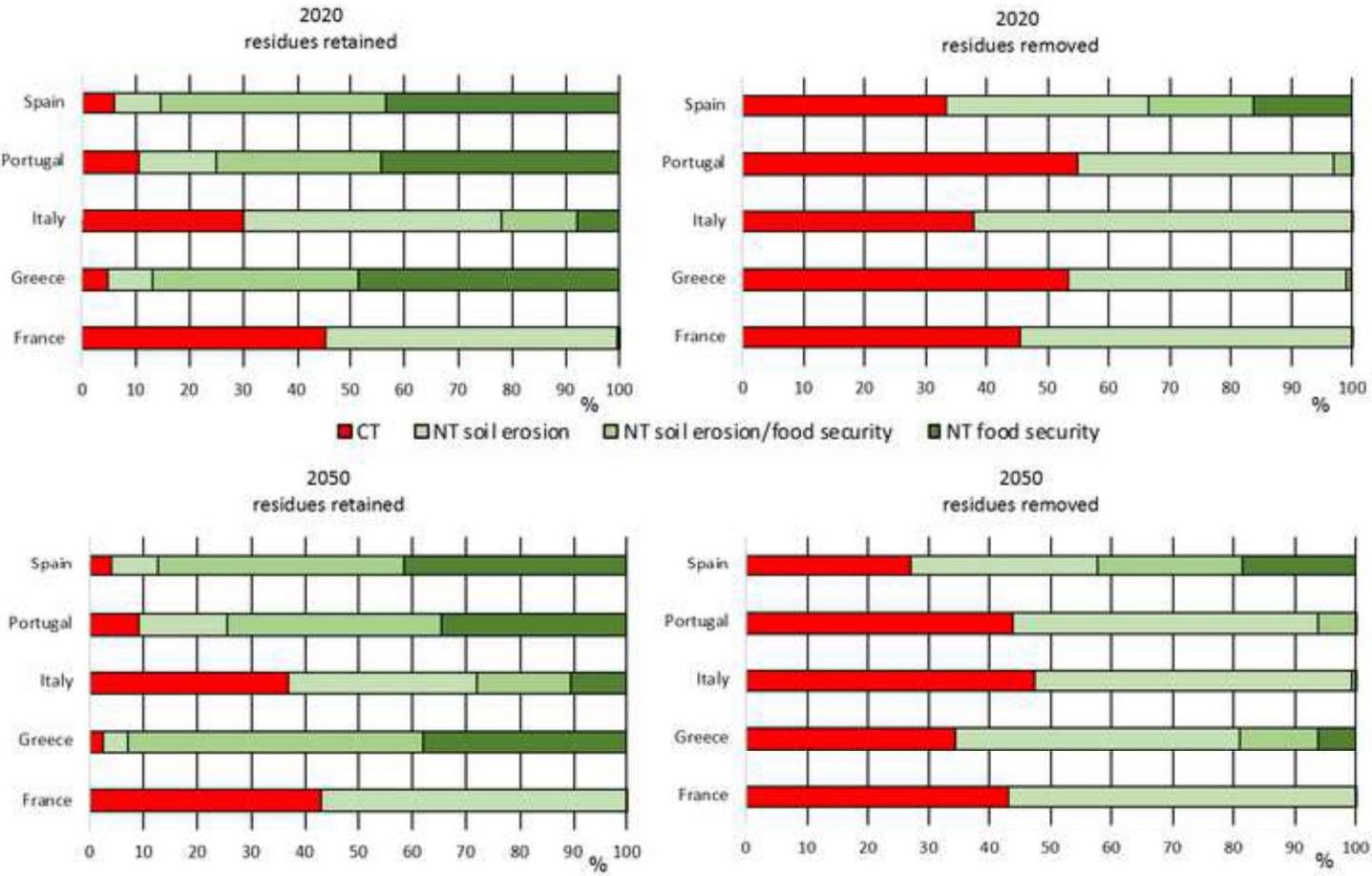
c) Residues Retained



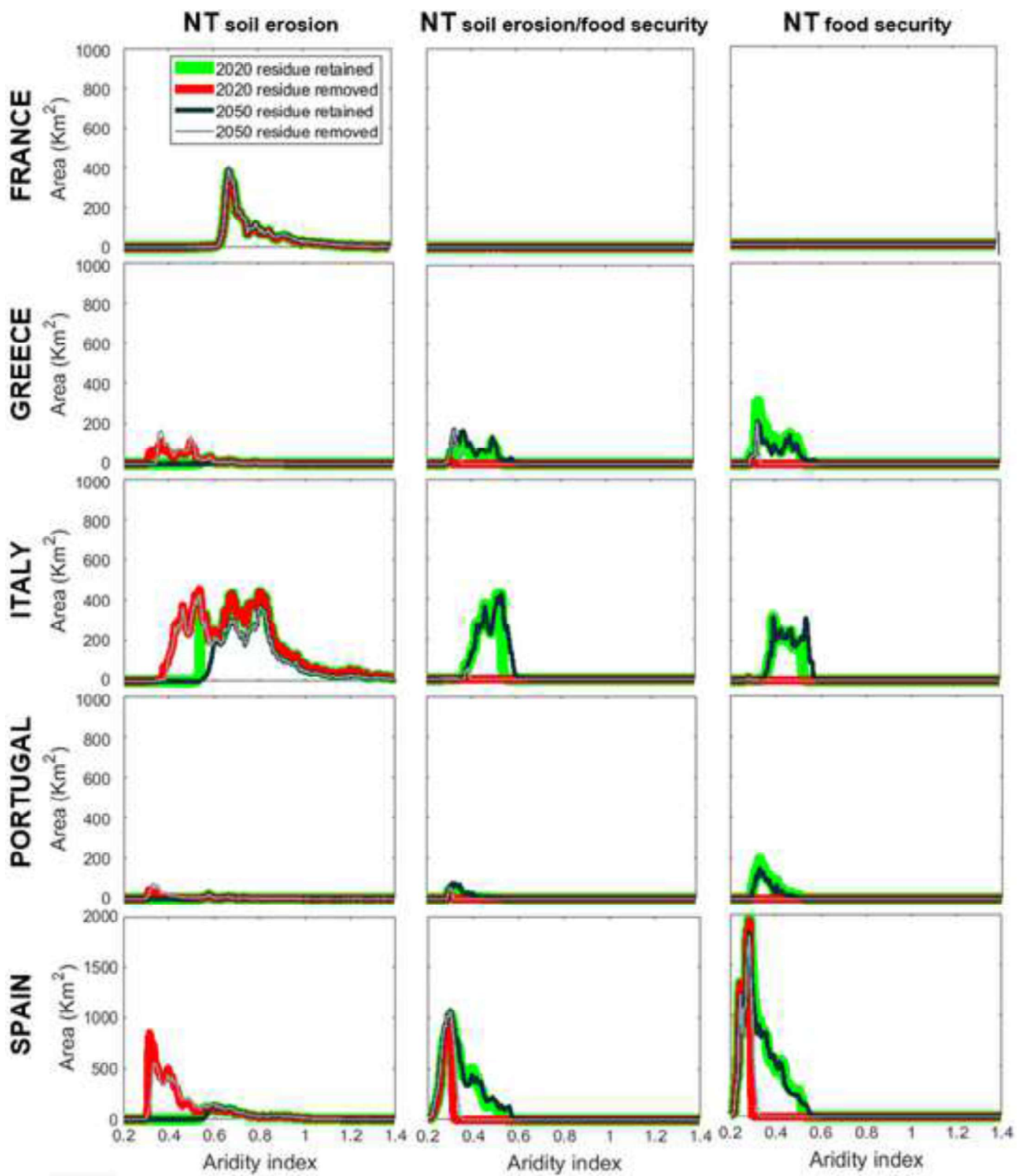
d) Residues Removed



Figure_7



Figure_8



Figure_9_REV2

Positive effect	Negative effect	Environmental factors	Payment for multiple ES		Suggested management
  		<i>AI</i> high <i>SE</i> high	Constant K_{NT}	Real benefit ($Q * SE$)	no-till
		<i>AI</i> low <i>SE</i> high			
		<i>AI</i> low <i>SE</i> low			
		<i>AI</i> high <i>SE</i> low	No payment		CT



 erosion control



 food security

Figures captions

Figure 1 – Input raster layers used for the investigation.

Figure 2 – Schematic flowchart of the proposed methodology. The figure also illustrates the four ecosystems services corresponding to no-till yield success probability, $P(NT)$, and soil erosion tolerance, SET .

Figure 3 – Spatial distribution of Aridity index in 2020 (a) and in 2050 scenario (b). In the right, frequencies distribution area.

Figure 4 – Spatial distribution of soil erosion in 2020 (a) and in 2050 scenario (b), for Corine Arable Lands. In the right, the corresponding frequencies.

Figure 5 – Spatial distribution of no-till yield success probability, $P(NT)$, in relation to residues management, in 2020 (a, b) and in 2050 scenario (c, d). In the right, the corresponding frequencies.

Figure 6 – Spatial distribution of no-till adaptability index, in relation to residues management and climate change, in 2020 (a, b) and in 2050 scenario (c, d). In the right, the corresponding frequencies.

Figure 7 – Area contribution (%) of conventional tillage (CT) and no-till (NT), for the considered countries, in 2020 and in 2050 scenario. Red bar refers to CT green bar refers to NT , according to the provided ecosystem services (light green - soil erosion control; medium green - soil erosion control/food security; dark green - food security).

Figure 8 – Area distribution versus the aridity index (AI), for the different residue management and for the baseline (2020) and for the 2050 scenario, for the considered ecosystem services (soil erosion control, soil erosion control/food security and food security).

Figure 9 – No-till (NT) driver effects of the considered ecosystem services, ESs , (green arrow = erosion control, red arrow = food security), and schematic representation of their synergies and trade-offs, suggesting agri-environmental payment and soil management adoption (SE = soil erosion; AI = Aridity index; K_{NT} = constant; Q = the unit cost of soil loss).

Table 1 – Summary and description of input databases.

Description	Source	Data type and resolution	References
Arable lands	CORINE (Copernicus CGLS)	Raster 100 x 100 m	Büttner and Kosztra (2018)
Meteorology	ClimateEU	Raster 1000 x 1000 m	Wang et al. (2006)
Soil Erosion	(ESDB-ESDAC)	Raster 100 x 100 m	Panagos et al. (2012)

Table 2 – No till adaptability index.

EC_s	Soil erosion (t ha ⁻¹)	P(NT)	Suggested tillage practice	Provided ecosystem service
i)	Low (≤ 3)	Low (≤ 0.5)	CT	food security
ii)	Low (≤ 3)	High (> 0.5)	NT	food security
iii)	High (> 3)	Low (≤ 0.5)	NT	soil erosion control
iv)	High (> 3)	High (> 0.5)	NT	food security and soil erosion control

Table 3 – Areas (km²) for the four provided Ecosystem services, EC_s , in the two considered climatic scenarios and residue management.

year	Region	residue retained				residue removed			
		CT	(soil erosion)	(soil erosion - food security)	(food security)	CT	(soil erosion)	(soil erosion - food security)	(food security)
2020	France	12476	14935	54	35	12500	14977	12	11
	Greece	563	981	4461	5711	6245	5336	105	30
	Italy	23915	38160	11163	6201	30086	49274	48	30
	Portugal	699	979	2058	2983	3681	2822	215	1
	Spain	5946	8313	40981	42705	32708	32476	16818	15943
	Total	43599	63368	58717	57635	85220	104885	17198	16015
2050	France	11803	15644	36	18	11813	15664	15	8
	Greece	291	524	6432	4469	4029	5443	1513	731
	Italy	29368	27851	13919	8303	37634	41385	383	37
	Portugal	603	1114	2671	2331	2931	3369	416	2
	Spain	3859	8329	45025	40731	26355	29996	23358	18235
	Total	45924	53462	68083	55852	82762	95857	25685	19013