Contribution of the land use allocation model for agroecosystems: the case of Torrecchia Vecchia.

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In this study, we developed a bio-economic model coupling land use and ecosystem services to investigate the role of forests on a broad set of ecosystem services, including carbon sequestration, soil quality and biodiversity. As a case study, the model was calibrated with economic, agronomic and ecological data from the Torrecchia Vecchia agroecosystem in Italy. In our analysis of optimal land use allocation, the results showed that diversified land use is required to provide a good balance between provisioning and non-provisioning ecosystem services. More specifically, the development of woodlands alongside farming activities had a positive impact on the soil quality score and on landscape heterogeneity, which is a proxy for ecosystem function and resilience. These findings demonstrate that the inclusion of woodlands can alleviate the trade-offs between provisioning and non-provisioning services as they can generate profit while allowing for better soil quality and biodiversity relative to more intensive land use. The study also confirms that a landscape-scale method can be used to investigate agroecosystem management problems when spatially explicit data is not available.

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Abstract

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Keywords: Optimization under constraints, Viability, Land use, Ecosystem services, Tipping points

1. Introduction

Agricultural landscapes and ecosystems have changed dramatically over the last decades (Hendrickx *et al.* 2007, Klijn 2004, Robinson & Sutherland 2002, Stoate *et al.* 2001). The intensification of agriculture resulting from increasing demand due to human population growth during the 20th century has been identified as a main driver of this transformation (Erisman *et al.* 2008, Garratt *et al.* 2018, Tilman & Clark 2014, Wall *et al.* 2015). In addition to increased inputs, this intensification has resulted in massive changes in land use, including the replacement of natural areas such as forests, wetlands and natural grasslands by croplands (Rees 2017). Such land use change has led both to a reduction of semi-natural habitats and their fragmentation into habitat patches (Geertsema & FJJA Bianchi 2017, Lindenmayer *et al.* 2012), eroding ecosystem function and services (Bengtsson *et al.* 2005, Foley *et al.* 2005). To tackle this concerning trend, a new paradigm has emerged: the sustainable intensification of agriculture (SIA) (Conway 1999, Godfray & Garnett 2014, Struik *et al.* 2014). The goal of SIA is to design resilient agroecosystems that not only rely on but contribute to several ecosystem services (defined as benefits that humans freely gain from the natural environment and from properly functioning ecosystems in the Millennium Ecosystem Assessment) (MEA 2005), repositioning agriculture from its current role as one of the world's largest drivers of environmental damage to a major contributor to sustainability (Foley *et al.* 2005, Rockström *et al.* 2017). However, the implementation of this concept greatly depends on the definition of sustainability. If sustainability is defined as maintaining the production level of a set of ecosystem services, the challenge of SIA can be interpreted as finding the trade-offs and synergies between a range of ecosystem services. Diversifying agricultural land use might offer interesting perspectives, since agricultural diversification has a strong positive impact on biodiversity through the creation of habitat and resource heterogeneity. The positive effect of diversification on biodiversity has been experimentally identified for different types of land use: for crop landscapes in Laiolo (2005), for grasslands in Robinson & Sutherland (2002), and at a broader scale by Mouysset *et al.* (2013), Quijas *et al.* (2010), Worm *et al.* (2006). Two types of land use are of particular interest for agricultural diversification: grasslands and forests. Grasslands have been frequently investigated in recent decades following the introduction of agro-environmental measures in public policy. Their positive impacts on biodiversity and ecosystem services have been demonstrated in many studies (Alavalapati *et al.* 2002, Mouysset *et al.* 2011, Shi & Gill 2005). However, the impact of grasslands remains limited (Pe'er *et al.* 2014). In this context, agricultural policymakers have started to consider forests as another potential lever for sustainable agriculture (Mouysset *et al.* 2019). For example, a new EU forestry strategy adopted in 2013 (Commission *et al.* 2013) stresses the importance of taking into account biodiversity and forest management in the environmental objectives of agricultural policy.

In this study, we developed a bio-economic model to explore different SIA scenarios based on land use allocation that included croplands, grasslands or woodlands. We assessed the performance of each according to a range of ecosystem services as recommended in Clark (1982), Doyen (2018), Wätzold *et al.* (2006). We captured the ecosystem services through land use/land cover (LULC) metrics (Kassawmar *et al.* 2018), landscape metrics and economic metrics. Of these, LULC metrics are relevant for approximating carbon sequestration and agricultural production (Schulp *et al.* 2008), while landscape-structure metrics can be used as a proxy for ecosystem diversity and complexity (Schulp & Alkemade 2011, Van Berkel & Verburg 2014, Van Oudenhoven *et al.* 2012, Willemen *et al.* 2008). An assessment of the agro-environmental sustainability of SIA scenarios should include cost-benefit (Janssen *et al.* 2018), cost-effectiveness (Holzkämper & Seppelt 2007, Kimball *et al.* 2015, Mouysset *et al.* 2014) and viability analyses (De Groot & Hein 2007, Doyen *et al.* 2017, Mouysset *et al.* 2013, Sabatier *et al.* 2010). However, in a context in which monetarization of ecosystem services remains controversial, cost-benefit analysis is difficult to achieve. We thus focused on a double analysis coupling cost-effectiveness and viability. Nevertheless, it is interesting to note that there is a direct link between the strength of the constraint used in cost-effectiveness (i.e. optimization under constraints) and viability analyses (i.e. allocations satisfying a set of constraints) and the implicit price of the ecosystem service at stake in the constraint. More specifically, a sensitivity analysis could provide relevant information about the implicit prices of ecosystem services and the social optimum. Our bio-economic model coupled land use allocation and biodiversity indices to investigate the impact of different types of land use on ecosystem services (Nalle *et al.* 2004). As a case study, it was calibrated with economic, agronomic and ecological data from the Torrecchia Vecchia agroecosystem in Italy. This landscape includes a combination of cattle raising, croplands and managed woodland. The results showed trade-offs between selected ecosystem services and economic profits, and indicated that diversified land use, especially the inclusion of woodlands, provides a good balance between provisioning and non-provisioning ecosystem services.

2. The bio-economic model

2.1. Context

The model we developed considered an agricultural landscape that is managed by a farmer. The farmer selects an agricultural strategy, which includes the proportion of the total agricultural area A_i allocated to different land uses i, and the livestock unit[1] V. The land use distribution at landscape scale is characterized by the vector $A = [A_1, \ldots A_i, \ldots, A_n]$. The different land uses i generate unitary profits Π_i and the cattle activity generates a profit from livestock Π_v. The profit at the landscape scale Π_L is computed as the sum of the unitary profits of all land uses and livestock minus indirect costs $\overline{c_L}$ arising from administrative expenses:

$$\Pi_L = \sum_{i=1} \Pi_i A_i + \Pi_v V - \overline{c_L} \tag{1}$$

In this model, unitary profits are assumed to be deterministic. Besides these profits, the agricultural strategy impacts the set of ecosystem services provided by the landscape. Specifically, variations in land use allocation impact the provision $P_{s,L}$ of ecosystem services s at landscape scale L. Taken together, the ecosystem service provision $P_{s,L}$ and the profit Π_L constitute different landscape scores S_L which are taken into account by the farmer to determine his/her land use allocation as depicted by figure 1.

We explored different agricultural strategies: optimal strategies, cost-effectiveness strategies and viable strategies. The rules governing a farmer's decisions are described in the following subsections.

[1]The livestock unit (LSU) is a reference unit for the aggregation of livestock from various species and age via the use of specific coefficients established on the basis of the nutritional requirement. The reference unit is the grazing equivalent of one adult dairy cow.

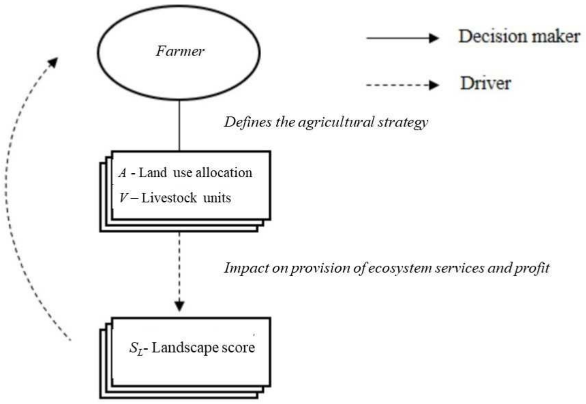


Figure 1: The bio-economic model. The farmer defines the agricultural strategy by allocating the land use vector (A) and a number of livestock units (V). In a feedback loop, the agricultural strategy impacts the value of the landscape scores (S_L).

2.2. Optimal scenarios

In an optimal scenario, the aim was to determine optimal land use allocation regarding a particular landscape score, which can either be the provision of one ecosystem service (*ie* $S_L = P_{s,L}$) or the economic profit (*ie* $S_L = \Pi_L$). The maximization program was defined as follows:

$$S_L^* = \max_{A,V} S_L(A, V) \tag{2}$$

Under agronomic constraints:

$$Tec(A, V) \geq 0 \tag{3}$$

These technical constraints represented agronomic characteristics specific to the case study (they are detailed in section 3.1). For example, they could take into account the minimum surface area required for mechanized farming equipment, the relationship between livestock units and the surface area of pastures and hay fields, or the relationship between the surface area of tree-cutting and logging rotation, etc. These optimal scenarios indicated the highest scores achievable in the landscape regarding each criteria.

2.3. Multi-criteria scenarios

The multi-criteria scenario maximized a score $\underset{\sim}{S}{}_L^m$ subject to a constraint based on another score S_L. Typically, a multi-criteria scenario might maximize the profit subject to a constraint based on an ecosystem service score (*ie* $\underset{\sim}{S}{}_L^m = \Pi_L$ and $S_L = P_{s,L}$), or it could maximize one ecosystem service score with a constraint based on another ecosystem service score. The maximization program was defined as follows:

$$S_L^{m*} = \max_{A,V} S_L^m(A, V) \quad (4)$$

Under constraints:

$$S_L \geq \omega S_L^* \quad (5)$$

and

$$Tec(A, V) \geq 0 \quad (6)$$

where ω represented the strength of the constraint. Typically, different values of ω between 0 and 1 were tested. The constraint was based on the maximum achievable score S_L^* which was determined through optimal scenarios. By combining the two scores, these scenarios were a first step towards a multi-criteria approach.

2.4. Viable scenarios

To deepen the multi-criteria analysis, we investigated viable scenarios. The viability analysis selected land use allocations that ensured landscape scores were above pre-identified thresholds. The viable land use allocations $Viab = (A, V)$ were defined with the following constraints:

$$S_L \geq \omega S_L^* \quad (7)$$

and

$$Tec(A, V) \geq 0 \quad (8)$$

The constraint (7) was built for all criteria S_L at stake in the landscape (typically, all the ecosystem service provision indicators and the profit). Similar to the multi-criteria scenarios, the parameter ω represented the strength of the constraint based on the maximal score (achieved in the optimal scenario S_L^*).

2.5. Numerical Solution

The optimal, multi-criteria and viability scenarios were handled using MATLAB. For the optimal and multi-criteria scenarios, we used MATLAB's Nonlinear Programming solver, and its function 'fmincon' with an interior-point optimization algorithm. For the viable scenarios, allocations were randomly generated using the RND function in VBA, before being filtered to take into account only the allocations respecting the technical constraints.

3. Case study

3.1. Context

The bio-economic model was calibrated with data from Torrecchia Vecchia, a diversified landscape extending over 510 hectares and located 50 km from Rome, Italy (figure 2). The landscape is managed by a single farmer. However, this farmer is required to adhere to the environmental standards of the Torrecchia Vecchia Foundation. These two stakeholders negotiate to decide which agricultural strategies will be implemented on the land, according to the profitability objectives of the farmer and the environmental requirements of the foundation. The parameter ω in the constraints in the multi-criteria and viable scenarios can be interpreted as a proxy for this negotiation.

(a) Location of the farm (circled in red).

(b) Aerial view (from Google Earth).

Figure 2: Map and aerial view of the case study.

In this landscape, we defined seven types of land use ($A = [A_1, A_2, A_3, A_4, A_5, A_6, A_7]$) which were classified in three main categories: crops, cattle raising and woodland. There were three types of crop land use ($i = 1, 2, 3$) as crop rotation in the case study is organized around a three-field system. There were also three types of cattle raising land use, with one area for the production of fava bean-type forage crops ($i = 4$), one area fand one area for pasture land and an area for pasture land ($i = 6$). An additional land use ($i = 7$) was dedicated to a conservatively managed forest, with a predominance of oak (Quercus cerris L.). Table 1 shows the land use distribution in the year of the study (2019).

The unitary profits of these activities are shown in Table 2.

The technical constraints $Tec(A, V)$ represent agronomic specificity related to the case study. We considered a set of five agronomic constraints. The first (equation 9) was related to the impossibility of using agricultural machines in small surface areas. It set a minimal surface area A_{min} for all types of land use:

$$A_i \geq A_{min} \tag{9}$$

The second constraint (equation 10) corresponded to the minimum surface area for on-site forage production to be allocated in relation to the number of livestock units:

$$A_4 \geq fa_1.V \tag{10}$$

Land use	Land use category	Variable	Value	Unit
Cereal 1	Crops	A_1	36	Hectares
Cereal 2	Crops	A_2	36	Hectares
Legume	Crops	A_3	36	Hectares
Forage crop	Cattle raising	A_4	7	Hectares
Hay	Cattle raising	A_5	64	Hectares
Pasture land	Cattle raising	A_6	130	Hectares
Woodland	Woodland	A_7	201	Hectares
Number of cattle	Cattle raising	V	129	Livestock Units

Table 1: Land use distribution in Torrecchia Vecchia in the year of the study (2019), characterized by the vector $A = [A_1, A_2, A_3, A_4, A_5, A_6, A_7]$, and the number of livestock units V.

Parameters	Value	unit
Π_1	664.7	euros/ha
Π_2	664.7	euros/ha
Π_3	664.7	euros/ha
Π_4	109.8	euros/ha
Π_5	343.4	euros/ha
Π_6	100.0	euros/ha
Π_7	56.7	euros/ha
Π_v	377.0	euros/V
$\overline{c_L}$	100000	euros

Table 2: Unitary profits of the different land uses ($\Pi_1, \Pi_2, \ldots \Pi_7$) and the livestock units (Π_v), and the value of indirect costs ($\overline{c_L}$).

The third constraint (equation 11) corresponded to the relationship between the surface area of hay production and pasture land and the number of cattle:

$$A_5 + A_6 \geq fa_2.V \tag{11}$$

The fourth constraint (equation 12) corresponded to the minimal pasture land to be allocated related to the number of cattle:

$$A_6 \geq fa_3.V \tag{12}$$

And the fifth constraint (equation 13) was related to the woodland management decided by the farmer. In the case study, very conservative forest management was implemented, in which the yearly cut surface area b_1 was rotated to allow the forest time to reach its targeted maturity b_2:

$$A_7 \geq b_1.b_2 \tag{13}$$

The coefficient values are provided in appendix 6.2.

3.2. Soil Quality

We then used the model to investigate two types of ecosystem services. We selected soil quality (SQ) as the first ecosystem service ($s = 1$). This is defined by the ability of a soil to perform essential functions (Garrigues *et al.* 2012): both physical (e.g. penetration and storage of water, support for plants) and biochemical (e.g. water quality, nutrient cycle regulation – carbon, nitrogen, oxygen, etc.). More broadly, soil services include carbon storage and gas regulation; soil can have a strong mitigating effect on climate change (Bommarco *et al.* 2013, Lal 2005). Indeed, carbon sequestration of atmospheric CO2 in the soil is considered a possible way to address global warming (Arrouays *et al.* 2002), due to the high stability of carbon in humus and soil organic matter (SOM). It has been estimated that 80% of global terrestrial carbon is stored in soil. Declining soil quality (Pimentel *et al.* 1995) as a result of unsustainable agricultural practices through processes such as salinization, acidification and erosion (Bommarco *et al.* 2013) is thus likely to have global consequences. Intensively managed agroecosystems appear to become less functionally efficient, mainly because of the reduction in ecosystem services provided by soils (Pan *et al.* 2009). Due to the correlation between a soil's organic matter and its ecosystem services, SOM/humus content is often used as a proxy for soil services (Magdoff & Weil 2004). A greater amount of humus is generally associated with better soil quality and better physical and biochemical soil services (Reboul 1977, Waksman 1936). For each type of land use, we computed the marginal performance of soil quality using data from the scientific literature, comparing the mineralization rate (d = destruction of SOM) to the humification rate (h = creation of humus), adjusted to match the reality of the case study (details in appendix 6.1). Several factors were taken into account, such as the cultivation depth (Mary & Guérif 1994), the ratio of limestone (Marin-Laflèche & Rémy 1974), the ratio of clay (Boiffin *et al.* 1986), the climate, and the average biomass production. The soil quality score was then computed as a sum of the marginal performance of each land use ($P_{1,L} = SQ$) (equation 14):

$$P_{1,L} = \sum_{i=1} (h_i - d_i).A_i \tag{14}$$

3.3. Landscape heterogeneity

We selected landscape heterogeneity (HE) as the second ecosystem service (s = 2). Landscape heterogeneity represents the degree of complexity of the spatial arrangement of a given landscape, both in terms of the diversity and the structure of land use. There is a positive relationship between biodiversity and heterogeneity (Norderhaug *et al.* 2000, Pino *et al.* 2000) : the latter notably plays a crucial role in the former by providing a variety of habitats for plants and animals (Ricketts 2001, Wethered & Lawes 2003). Heterogeneity is considered a main driver in supporting many species in agroecosystems (Tscharntke *et al.* 2012) and may also encourage the persistence of species that require different habitats during their lifecycle or throughout the year (Benton *et al.* 2003, Chamberlain & Gregory 1999). In agroecosystems, enhancing heterogeneity to increase biodiversity is also correlated to better natural management of pests (Bianchi *et al.* 2006). Heterogeneity seems to be a condition for the proper functioning of ecosystems, given that the resilience of an ecosystem (its ability to maintain its function after disturbance) depends on the heterogeneity of the functional capabilities of its species (Valencia *et al.* 2015). For this study, we used the compositional heterogeneity ($P_{2,L} = HE$) of the landscape as a proxy for biodiversity. We computed the compositional heterogeneity with the Shannon index of land use. This is a popular measure of diversity that increases with the number of cover types (Shannon & Weaver 1949) and is often used in the literature on land use allocation models as a proxy for ecosystem services related to the benefits of biodiversity (Lichtenstein & Montgomery 2003). The Shannon index of land use was computed as below (equation 15):

$$P_{2,L} = -\sum_{i=1}^{n} \frac{A_i}{\sum_{i=1}^{n} A_i} \cdot \ln\left(\frac{A_i}{\sum_{i=1}^{n} A_i}\right) \tag{15}$$

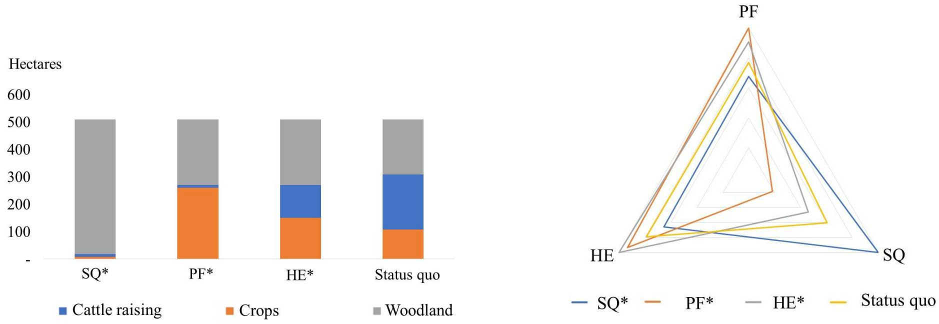
4. Results

4.1. Optimal scenarios

The figure 3 presents the results from modelling the optimal scenarios for maximizing profit (PF^*), for maximizing soil quality (SQ^*) and for maximizing landscape heterogeneity (HE^*). The graphs show the land use allocation by category (fig. 3a) and the landscape scores (fig. 3b) for the three different scenarios, compared with the current allocation and scores (status quo). The three scenarios show contrasting land use allocation. In particular, there is a significant difference between the land use pattern that maximizes soil quality and the one that maximizes profit. The former is dominated by woodlands, while the latter has an increased surface area of crops and reduces woodlands to the minimum allowed by the agro-technical constraints. The scenario maximizing landscape heterogeneity has a more balanced distribution of land use. Consistent with these contrasting allocations, fig. 3b depicts a strong trade-off between profit and soil quality.

4.2. Multi-criteria scenarios

Figure 4 depicts the change in optimal profits under different SQ and HE constraints (fig. 4a and fig. 4c respectively) and associated land use allocations (fig. 4b and fig. 4d respectively). The scenarios maximizing profit under soil quality constraints reveal a concave trade-off between the two scores (profit and soil quality). A marginal increase in soil quality generates strong profit losses after a tipping point (about $\omega = 0.7$ corresponding to $SQ = 259 t/humus/year$). These two trends are explained by land use patterns: for $\omega < 0.7$, increasing soil quality relies on replacing crops with land for raising cattle; however, after this tipping point, woodland is part of the optimal land use. The Pareto frontier between the two scores provides a marginal rate of transformation between soil



(a) Land use allocations (b) Landscape scores

Figure 3: Land use allocations and scores from the optimal scenarios for soil quality $SQ*$, profit $PF*$ and heterogeneity $HE*$. The yellow line (status quo) corresponds to current land use distribution.

quality and profit. Before the tipping point, the marginal rate of transformation of soil quality is -93 euros for an additional t/humus/year, it then gradually decreases after the tipping point from -450 euros to -952 euros. In contrast, scenarios maximizing profit under heterogeneity constraints (fig. 4c) exhibit minor antagonism between these two scores. The level of profit is only impacted by high heterogeneity demands ($\omega > 0.8$). After this threshold, the marginal rate of transformation of heterogeneity for an additional unit of the Shannon index rapidly rises from -33k euros to -165k euros.

Figure 5 presents the change in optimal HE and SQ under different soil quality and heterogeneity constraints and associated land use allocations. The results show a concave trade-off between soil quality and heterogeneity characterized by two successive land use trends: an increase in the heterogeneity constraint up to $\omega = 0.7$ corresponds to a progressive substitution of woodlands by land for cattle, while beyond the tipping point ($\omega > 0.7$) an increase in heterogeneity demands stimulates the development of crops. Before the tipping point, the marginal rate of transformation of heterogeneity (one additional unit of the Shannon index) gradually decreases the soil quality from -35 to -134 t/humus/year. After the tipping point, this transformation rate decreases more rapidly to reach -1640 t/humus/year.

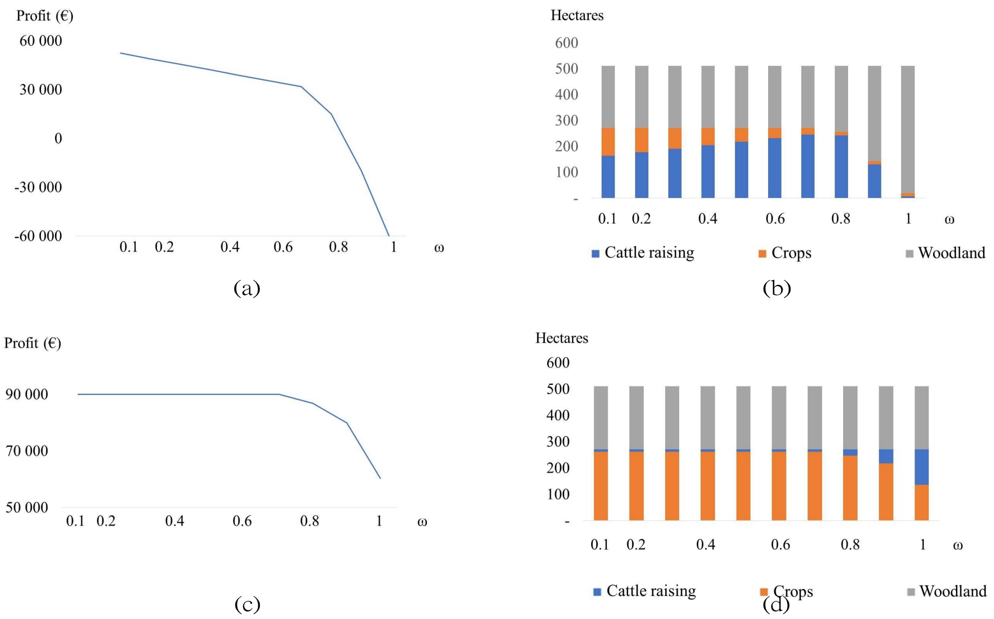


Figure 4: Changes in optimal profits PF under different soil quality SQ (a) and heterogeneity HE (c) constraints and associated land use allocations (b and d, respectively).

4.3. *Viable scenarios*

Figure 6 represents the three-dimensional scores (soil quality SQ, heterogeneity HE and profit PF) of non-viable and viable scenarios (in red and green, respectively) for different values of ω. The figure 6a, b, c highlight in green the viable scenarios when $\omega = 0$ ($\omega = 0.2$, $\omega = 0.4$ respectively). Since some scores can be negative, the no-constraint scenarios are different from the $\omega = 0$ viable scenarios. Of the 7115 possible scenarios, only 1359 were viable when considering $\omega = 0$, decreasing to 246 viable scenarios when considering $\omega = 0.2$, and only one viable scenario when considering $\omega = 0.4$. This scenario achieved the highest scores for all services simultaneously, corresponding to a balanced allocation that included woodlands and cattle raising (53 hectares of crops, 216 hectares for cattle raising with 136 livestock units, and 241 hectares of woodlands). Figure 6 also shows that the reduction in viable solutions due to the increase of ω occurs around solutions above the tipping points $0.7SQ^*$ and $0.8HE^*$ identified in the multi-criteria scenarios. This means that solutions satisfying the highest viability constraints are solutions that avoid high transformation rates.

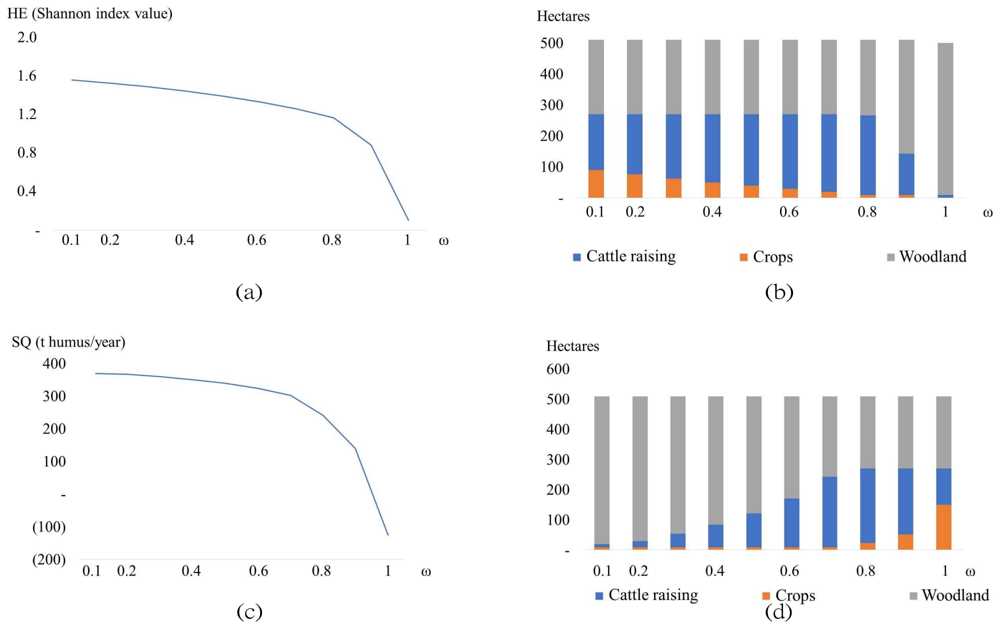
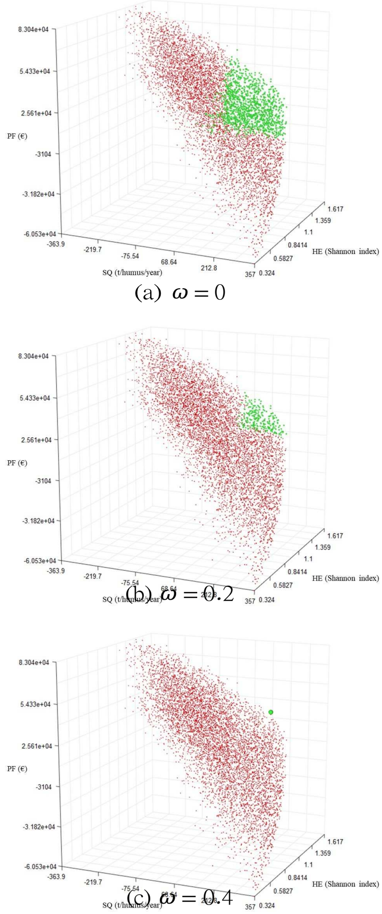


Figure 5: Changes in optimal heterogeneity HE and soil quality SQ under different soil quality (a) and heterogeneity (c) constraints and associated land use allocations (b and d, respectively).

5. Discussion

5.1. Bioeconomic modeling for sustainable intensification of agriculture

The bio-economic model developed in this study offers a framework for policymakers aiming to improve the sustainability of farming. While the calibration is based on a specific case study, the model is easily applicable to other contexts. The parameters and technical constraints can be adjusted to fit the agro-ecological characteristics of other agroecosystems. Calibrated to specific contexts, the framework is able to provide the quantitative analyses that are crucial to help stakeholders in decision-making. However, it should be kept in mind that the model results are not exact predictions of future bio-economic states, but quantitative trends indicating the ecosystem function of land use change decisions they might implement. The first contribution of this framework regards its ability to show trade-offs between ecosystem services in agricultural landscapes (Foley *et al.* 2005, Guerry *et al.* 2015, Lawler *et al.* 2014, Maass *et al.* 2005, Pereira *et al.* 2005). Specifically, based on our case study, the results confirm that agricultural intensification mainly driven by crops leads to a loss of heterogeneity and soil fertility in agricultural landscapes (Dalal *et al.* 1991, Geri *et al.* 2010). The second contribution is the model's ability to show the trade-offs between non-provisioning ecosystem services and profit. By



(a) $\omega = 0$

(b) $\omega = 0.2$

(c) $\omega = 0.4$

Figure 6: The three-dimensional scores (soil quality SQ, heterogeneity HE and profit PF) of non-viable and viable scenarios (in red and green, respectively) for different values of ω.

identifying Pareto frontiers, our bio-economic analysis was able to characterize crucial tipping points in transformation rates. The third contribution concerns the viability analysis, which is an interesting complement to the traditional optimal approach (Doyen & Martinet 2012). By identifying scenarios below the Pareto frontier but above crucial thresholds, it becomes possible to add additional criteria into the analysis or to include new stakeholders with different priorities. Implementing a viability approach in a bio-economic model offers a flexible framework to investigate issues around SIA. While the multi-criteria scenarios can be used to have a better understanding of the trade-offs between ecosystem services and identify crucial tipping points, viability scenarios are key to determine the best land uses strategy based on their priorities in terms of ES. Lastly, this model was able to successfully couple land use allocation with ecological and economic indices with a minimal amount of data. We used scalarization methods, which are useful for including stakeholders' preferences (Kaim *et al.* 2018) and facilitate the resolution of multi-objective optimization problems, by defining only one objective function and adding additional objectives as constraints (Ehrgott 2005). In the context of on-going advances in multi-criteria research and the development of guidelines for selecting the appropriate resolution for land use questions, a key contribution of this framework is its ability to provide a landscape-scale method for agroecosystem management when data resolution is not sufficient to obtain a spatially explicit solution (as detailed in Briner *et al.* (2012)).

5.2. Woodland as a key to sustainability in agroecosystems

A central finding of our study is that integrating woodlands in an agroecosystem can be an effective way to achieve sustainable ecosystem service provision. The inclusion of woodlands can alleviate the trade-offs between provisioning and non-provisioning services, as they generate profits while allowing better soil quality and biodiversity relative to more intensified land use. This result stems from the fact that forests have a significant impact on the production of humus and SOM. As a consequence, the integration of woodlands has a strong positive influence on the soil quality score at the landscape level. However, this depends heavily on forest management. Different approaches to forest management have highly variable impacts on ecosystems and ecosystem services (Cao *et al.* 2011, Costanza 2000, Drever *et al.* 2006, Hynynen *et al.* 2005, Triviño *et al.* 2015). In our specific case study, forest management is low intensity, resulting in high soil quality scores. So our findings are consistent with studies advocating low-intensity silvicultural practices to promote non-provisioning ecosystem services (Drever *et al.* 2006, Seely *et al.* 2002). While the positive impact of conservatively managed woodlands on soil quality is not surprising, our results highlighted a second benefit of introducing woodlands in agroecosystems: their positive influence on landscape heterogeneity. This is due to the inherent technical constraints of this type of woodland management (i.e. the slow growth cycle of forest biomass and planned rotation of tree-cutting support the sustainability of woodlands over time). In the context of SIA, improving landscape heterogeneity is an important advantage as this is considered a main driver in supporting diverse species in agroecosystems and plays a major role in healthy ecosystem functioning (Tscharntke *et al.* 2012, Valencia *et al.* 2015). Moreover, landscape heterogeneity may support ecosystem resilience, especially in a context of climate change (Malika *et al.* 2009, Thuiller *et al.* 2005). Given the numerous positive impacts of woodlands on agroecosystems, their introduction should be promoted by agricultural policymakers in order to support the sustainability of agroecosystems (Mouysset *et al.* 2019).

5.3. Perspectives

This bio-economic model showed that in our case study diversified land use – particularly the development of woodlands alongside farming activities – had a positive impact on the soil quality score and on landscape heterogeneity, which is a proxy for ecosystem function and resilience. In future studies, the model's perspectives could be broadened to consider other aspects. For example, an important characteristic of agroecosystems is uncertainty. Climate or market uncertainties can be determining factors in the decisions of stakeholders. Thus it would be very valuable to extend this bio-economic framework in order to include an examination of this. Several studies have highlighted the positive impact of forests faced with ecological uncertainties (Gunderson *et al.* 2002, Holling 1973, Loreau *et al.* 2001, Tilman *et al.* 2001), so an analysis of uncertainty might confirm the positive role of forests and land use diversification. Another crucial characteristic of agroecosystems relates to the spatial aspects and connectivity of the system (Plantinga 2015), and future work may include spatiality and temporality to deepen the analysis of the impacts of land use decisions on the evolution of trade-offs and synergies over time. However, aspatial models are considered as relevant tools as they can be more easily replicated and compared (Nelson *et al.* 2008), and policymakers can use them to immediately evaluate the impact of their land use decisions on several ecosystem services. In our case study, the low-intensity woodland management practices in place had a significant impact on the results. It would be interesting to adapt the model to a context with higher woodland profitability. Many studies have identified woodland management practices that associate high timber production and biodiversity conservation. In a land sparing perspective, it would also be interesting to adapt this model to include two types of woodland management, one focusing on timber production and the other on biodiversity conservation, to evaluate the impact of these two management practices on sustainability (Jensen & Skovsgaard 2009, Rieger & Wilhelm 2013).

6. Appendix

6.1. Details on soil quality

Soil quality scores were computed based on data from the literature, adjusted by experts to reflect the reality of the case study. Specifically, soil quality marginal performances for each land use were computed using data comparing the mineralization rate d corresponding to SOM destruction, to the humification rate h corresponding to the creation of humus. In the model, we used the formula from Mary & Guérif (1994) for the mineralization rate, which takes into account the clay rate A and the limestone rate C:

$$d = 1200.temp./((200 + A).(200 + 0.3.C)) * P \tag{16}$$

The temperature factor $temp$ depended on the average temperature T° and was computed as follows:

$$temp = 0.2(T^\circ - 5) \tag{17}$$

Data from the literature was adjusted and included in the model (Bockstaller & Girardin 2008), based on several factors such as the maximum ploughing depth *pr*, the cultivation system (Mary & Guérif 1994) and frequency of the incorporation of organic matter and crop residues *fr*, irrigation *ir*, and the tillage strategy *ts*:

$$P = pr.fr.ir.ts \tag{18}$$

Experts adjusted the data to take into account the reality of the case study regarding the intensity of ploughing, the poor management of crop residues (systematic withdrawal) and the use of manure. The same process was applied to humus destruction rates for land use related to cattle raising, such as hay production and pasture surface area. The humification rate h was estimated with the average production of dry matter and roots using data from the literature (Leclerc 2001), adjusted to match the case study. Similarly, for woodlands, humic assessment took into account the production of dry matter and humification rates based on the literature. The mineralization rate was computed following the previously described process and adjusted by experts to match the case study, particularly regarding the slow growth of Quercus cerris L., the shallow soil depth and the removal of crop residues. Once coefficients h and d were determined for all land uses, the humic assessment was computed to take into account an estimated land mass fixed at c.4200 t/ha and a stock of humus fixed at 2% = c.84t/ha. The results can be interpreted as an equivalent for ton of humus produced/hectare.

6.2. Parameters

Parameters	Value	unit
n	7	land uses
$h_1 - d_1$	-2.2	t/humus/year/ha
$h_2 - d_2$	-2.2	t/humus/year/ha
$h_3 - d_3$	-2.2	t/humus/year/ha
$h_4 - d_4$	-2.2	t/humus/year/ha
$h_5 - d_5$	0.5	t/humus/year/ha
$h_6 - d_6$	0.5	t/humus/year/ha
$h_7 - d_7$	0.8	t/humus/year/ha
$SQ*$	369.9	t/humus/year
$HE*$	1.62	Shannon index value
$PF*$	89888.6	euro
A_{max}	510	ha
A_{min}	3	ha
fa_1	0.05	ha/V
fa_2	1.5	ha/V
fa_3	1	ha/V
b_1	8	ha/year
b_2	30	years

Table 3: Parameters related to landscape scores computation, technical constraints and maximal landscape score values derived from the optimal scenarios

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