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Research, part of a Special Feature on Ecosystem Services for Agroecological Transitions

The agrarian metabolism as a tool for assessing agrarian sustainability, and its application to Spanish agriculture (1960-2008)

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ABSTRACT. Agrarian metabolism applies the social metabolism framework to agriculture. It focuses on the study of the exchange of material and energy flows between a society and its environment for producing useful biomass. These flows must maintain the fund elements of the agroecosystem in sufficient quantity and of sufficient quality for them to continue providing ecosystem services. This methodology was applied to Spanish agriculture between 1960 and 2008, a period characterized by a deep process of intensification based on external inputs (EIs). We specifically focused on nitrogen (N), phosphorus (P), potassium (K), carbon (C), and energy flows, and on the three fund elements that they sustain such as soil, biodiversity, and woodland. The results show that the growing incorporation of EIs has broken the equilibrium between land and biomass uses required by traditional farming, lowering the density of internal energy loops. On cropland, the relative fall in unharvested biomass had a negative effect on both biodiversity and the soil, which reduced the replenishment of organic C between 1960 and 1990. The sharp increase in internal and external flows of biomass for animal feed hardly contributed to increasing soil organic carbon (SOC) between 1990 and 2008 because of the fact that these flows had increasingly lower C:N ratios. The massive importation of N in feed and mineral fertilizers (553 and 1150 Gg in 2000, respectively) increased the surplus and the losses of N, which in turn could have a negative impact on biodiversity, water, and the atmosphere. The scenario constructed without imported animal feed would allow a reduction in the environmental impacts related to the excess of N, with hardly any negative effect on SOC replenishment, and improving energy return rates in the form of total, unharvested, and accumulated phytomass.

Key Words: *agricultural intensification; agroecology; carbon and nutrient (N, P, K) balances; energy flows; social metabolism*

INTRODUCTION

Agrarian metabolism (AM) refers to the exchange of energy and materials between a given society and its agrarian environment. It arises from the application of the social metabolism framework (Haberl et al. 2004, Giampietro et al. 2009) to agriculture, which implies the modification of some conceptual and methodological aspects of this framework. AM specializes in generating biomass and ecosystem services for human purposes (Guzmán Casado and González de Molina 2017). It is essential to know whether that exchange is carried out sustainably. This entails adding some crucial aspects to the metabolic schema.

First, our AM proposal considers the agroecosystem as the system of reference, understanding that agroecosystems are dissipative structures designed and managed by the farmers, through which flows of energy and materials enter, exit, and recirculate. The agroecosystem receives from its environment, natural and/or social, the energy and material fluxes that allow it to generate order in the form of biomass and environmental services that are useful for humans. An agroecosystem is, therefore, the outcome of a social-ecological relationship (Guzmán Casado and González de Molina 2017).

Second, we should distinguish between agroecosystem flow and funds, in accordance with the proposals put forward by Georgescu-Roegen (1971) and emphasized by Giampietro et al. (2009). The ultimate aim of the economy is not the production and consumption of goods and services, but rather the reproduction and improvement of the set of processes required for the production and consumption of goods and services. This variation in the main aim of economic agrarian activity implies, from a biophysical perspective, transferring the focus away from the flow of energy and materials and onto fund elements. This shift in orientation allows us to evaluate whether flows of energy and materials into and out of the agrarian sector are capable of reproducing and even improving fund elements in successive production cycles. In other words, moving the focus of attention away from the volume of production and consumption of biomass toward sustainability to ascertain whether production and consumption can be maintained indefinitely and, by extension, the supply of ecosystem services, because these also depend on the state of the agroecosystem fund elements (Cornell 2010, Burkhard et al. 2011, Costanza 2012).

Third, from a metabolic point of view, we have to analyze the role played by energy flows within agroecosystems, which is a crucial element when it comes to evaluating their sustainable functioning (Guzmán Casado and González de Molina 2017). Indeed, the socio-metabolic approach to agriculture is usually based on the domestic extraction (DE) of biomass, leaving the structure and functioning of ecosystems to one side. However, from an agroecological perspective, the level and sustainability of DE also depends on the biomass that is not extracted and that, therefore, remains within ecosystems and is available to their other heterotrophic components. It is well known that the sustainable management of an agroecosystem depends on the levels of biodiversity and organic matter, the appropriate replenishment of soil fertility, and the possibilities of closing biogeochemical cycles on a local scale, among other factors (Gliessman 1998). This represents a cost, because a significant part of the biomass generated must recirculate to perform the basic productive and reproductive functions of the agroecosystem: seeds, animal labor,

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soil organic matter, biodiversity, and so forth. In accordance with the proposals of Ho and Ulanowicz (2005:41, 45) and later of Ho (2013), the sustainability of agroecosystems, therefore, correlates positively with the quantity and quality of its internal loops or cycles and, to that extent, with the energy flows that circulate within them and whose function is to reproduce the fund elements. In short, the maintenance of internal loops in agroecosystems is directly related to the use of a significant part of net primary production to fuel them. This has major implications when it comes to calculating net primary productivity (NPP), which must then be broken down into different categories according to its productive or reproductive functionality (Guzmán Casado and González de Molina 2017).

To this extent, an agroecosystem with fund elements that require the dissipation of low levels of energy for its maintenance by means of those recirculation processes in turn generates low entropy in the surrounding environment and minimizes the flows of external energy. In contrast, when the internal complexity of an agroecosystem is substantially reduced, and its internal loops diminished, it needs to import energy in the generation of internal order. In these cases, total entropy also increases, and sustainability of agroecosystem could be compromised.

In other words, when agroecosystem functioning and the maintenance of the fund elements are based on a high density of internal loops, the imported flow of energy is also minimal. At the other extreme, when a complex agroecosystem is simplified to the point that it hosts a monoculture, external energy flows must be increased significantly (Government Office for Science 2011:10). This means that the capacity of the agroecosystem to maintain the production of biomass in the long term, without increasing inputs of external energy, is the foremost expression of sustainable management (Guzmán Casado and González de Molina 2017). Both criteria, the capacity to reproduce the biophysical fund elements or not and to do so without increasing the use of external energy, have been chosen by us to measure the environmental damage caused by the industrialization of agriculture.

The evaluation of the maintenance of fund elements allows us to recognize which of them are undergoing processes of deterioration and which are being improved and restored. In addition, it allows us to predict whether the modification of a flow to improve a fund element will harm another fund element and the extent to which there can be compensations between both uses. This is so because flows of energy and materials are interconnected. This analysis may provide new understandings on setting the limits of what is possible in terms of the degree of sustainability of an agroecosystem for given agroclimatic and technological conditions. The proposal thus becomes an intentional and dynamic model that interrelates the different components of the agroecosystem within a changing context. In short, our proposal considers agroecosystem as a holon (Bland and Bell 2007) whose connections must necessarily be explored. As Koestler stated in his seminal formulation of holon: “Organisms and societies are multi-leveled hierarchies of semi-autonomous sub-wholes branching into sub-wholes of lower order and so on. The term holon has been introduced to refer to these intermediary entities which, relative to their subordinates in the hierarchy, function as self-contained wholes; relative to their superordinates as dependent parts. This dichotomy of wholeness and partness, of autonomy and dependence, is inherent in the concept of hierarchy order” (Koestler 1967:58). Therefore, the recognition of the complexity of agroecosystems as social-ecological systems led to postulating a multiscalar analysis because of the hierarchical structure of organization.

On the other hand, the state of an agroecosystem cannot be considered as the sum total of the states of its fund elements. Therefore, results obtained after analyzing the state of different fund elements need to be consistent with a robust indicator that evaluates the agroecosystem as a whole. We used the actual net primary production energy return on investment (NPPact EROI; Guzmán and González de Molina 2015, Galán et al. 2016, Guzmán Casado and González de Molina 2017, Guzmán et al. 2017) because it has been shown to be a very consistent indicator when applied to different case studies and, therefore, permits the necessary methodological triangulation to test the other indicators and obtain more robust results.

We have applied this proposal to Spanish agriculture that represents Mediterranean agroenvironmental conditions over the past 50 years. During this period, Spanish agriculture experienced a substantial intensification process based on the use of external inputs to a greater extent than other Mediterranean countries. This circumstance makes Spanish agriculture an optimal case study, because it provides diachronic scenarios with very different land use intensities, within these agroenvironmental conditions. In particular, our objectives have been as follows: (a) to quantify the biophysical flows, i.e., energy, macronutrients, and carbon, of Spanish agriculture between 1960 and 2008; (b) to test whether these flows are capable of reproducing and even improving fund elements, i.e., soil, biodiversity, and woodland, in successive production cycles; and (c) to demonstrate the usefulness of this methodological approach for the design of sustainable agroecosystems, through its application to a scenario in which specific flows have been modified.

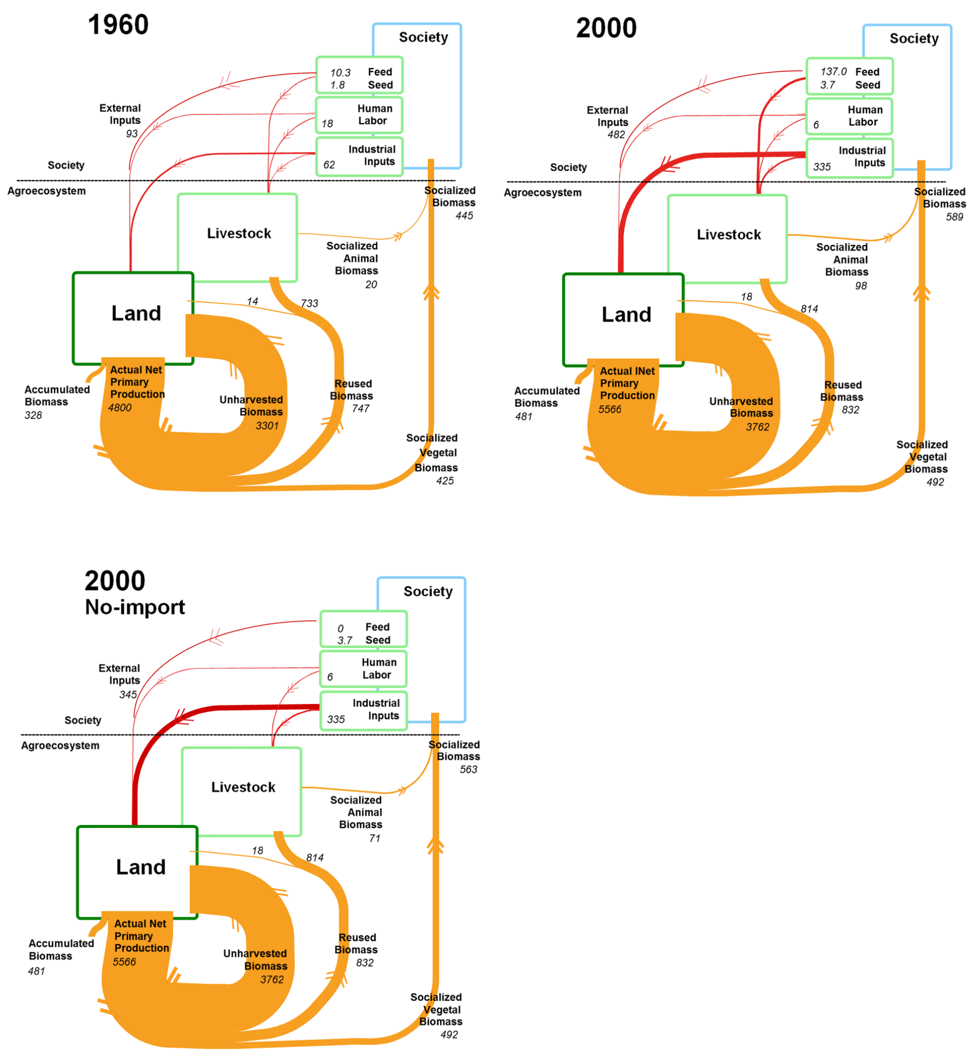
DATA COLLECTION, CONCEPTS, AND METHODS

Data collection

The main sources we used are the statistics provided by the Spanish government (MAGRAMA 2015). We have reconstructed the evolution of total biomass production in all Spanish land areas, excluding unproductive areas that remained practically constant throughout the period studied (see Table A1.1 in Appendix 1), and total inputs consumed (TIC) at six points over time between 1960 and 2008, using five-year averages to buffer year-on-year variability. The reconstruction of biomass production is described in detail in Soto et al. (2016) and Guzmán et al. (2017).

The exports and imports of biomass were calculated from foreign trade sources. For 1960 and 1990, we used the FAOSTAT database (FAO 2015). For the period from 2000 to 2008, we used the DATACOMEX database of Spanish overseas trade (MINECO 2015). The amounts of external inputs employed in Spanish agriculture during the period studied were mainly gathered directly from official statistics complemented by technical reports and research studies (Guzmán et al. 2017).

Fig. 1. Energy flows in Spanish agroecosystem in 1960, 2000, and no-import scenario (in PJ).



Concepts

NPPact is the net productivity that actually takes place in an existing ecosystem with human intervention, in contrast to that potential NPP that the ecosystem would achieve without human presence. It includes the root biomass as well as weeds.

Socialized vegetable biomass (SVB) is the phytomass that is directly appropriated by human society, considered as it is extracted from the agroecosystem, prior to its industrial processing. Socialized animal biomass (SAB) is the animal biomass, i.e., live weight of meat at the farm gate, milk, wool, and so forth, that is appropriated directly by society. The sum of SVB and SAB gives the socialized biomass (SB), which is the total biomass appropriated by society.

Reused biomass (RuB) is the phytomass that is intentionally restored to the agroecosystem by the farmer. Therefore, the RuB does not cross the boundary between the agroecosystem and society (see Fig. 1). This means that the phytomass is reincorporated into the agroecosystem by means of human labor and has an agronomic purpose that is recognized by the farmer, for example, to obtain a product or a service, i.e., seeds and animal feed for the supply of meat or milk. This category includes the biomass that is destroyed by fire, for example, stubble burning, because it involves conscious work and has an agronomic purpose. DE is the sum of SVB and RuB.

Unharvested biomass (UhB) is the phytomass that is restored to the agroecosystem by abandonment, without the pursuit of any specific aim, and without the investment of any human work, for example, litterfall and the root systems, except in crops where the root is harvested. UhB can be divided into aboveground unharvested biomass (AUhB) and belowground unharvested biomass (BUhB).

Accumulated biomass (AB) refers to the portion of phytomass that accumulates annually in the aerial structure and in the roots of perennial species, i.e., forest trees, woody crops, and shrubs. NPPact of agroecosystems is the sum of SVB + RuB+ UhB + AB.

External inputs (EIs) include human labor, as well as all of the inputs, i.e., fertilizer, pesticides, machinery, feed, and so forth, that originate outside the agroecosystem. They can be divided into industrial inputs, i.e., chemical fertilizers, machinery, and so forth, and nonindustrial inputs, i.e., phytomass, human labor, and so forth.

Usually, EROIs in agriculture have been used to measure the "energy cost" (Scheidel and Sorman 2012) of net biomass produced for appropriation by society (Martinez Alier 2011), whether in the form of foodstuffs, raw materials, or biofuels. However, it has recently been proposed to broaden its application to the assessment of agrarian sustainability. The EROIs developed to this end have been named "agroecological EROIs," and they estimate the return on the energy invested by society in the form of biomass flows that sustain agroecosystem fund elements. NPPact EROI, biodiversity EROI, and woodening EROI are agroecological EROIs. The methodology for agroecological EROI calculations was developed in Guzmán Casado and González de Molina (2017) and Guzmán et al. (2017).

NPPact EROI (Eq. 1) estimates the real productive capacity of the agroecosystem, whatever the origin of the energy it receives, i.e., solar for the biomass or fossil for an important portion of the EIs. This is an indicator that provides integrative information on the state of the agroecosystem, beyond the particular situation of each fund element. A decreasing trend in NPPact EROI values of an agroecosystem over time indicates degradation of the productive capacity.

NPPact EROI = NPPact/TIC (Eq. 1)

Whereas TIC is calculated as RuB + UhB + EI. Biodiversity EROI provides useful information on the extent to which energy invested in the agroecosystem contributes to sustaining food chains of heterotrophic species, e.g., Arthropoda, Mammalia, and so forth, and is calculated according to Eq. 2. The relationship between energy flows and biodiversity has been proposed by ecologists based on empirical studies showing that ecosystems with larger amounts of energy entering the food web will be able to support longer food chains and hence greater biodiversity (Thompson et al. 2012). In the particular case of agroecosystems, different authors have found that the increase in forage resources is one of the drivers of the biodiversity increase associated with the conversion of conventional farms into organic farms in the present (Döring and Kromp 2003, Gabriel et al. 2013). A decreasing trend of this EROI indicates a deterioration of the biodiversity fund element.

Biodiversity EROI = UhB/TIC (Eq. 2)

Woodening EROI estimates whether the energy added to the system is contributing to the storing of energy in the system as AB. A decreasing trend in this EROI indicates a deterioration of the woodland fund element. This EROI is calculated as follows:

Woodening EROI = AB/TIC (Eq. 3)

Methods

Functioning of the agroecosystem
The calculation in energy terms of NPPact and its components, and of the EIs, can be found in detail in Guzmán Casado and González de Molina (2017) and Guzmán et al. (2017).

State of the fund elements
1. Soil: The maintenance of this fund element encompasses mainly two processes: the replenishment of nutrients (nitrogen [N], phosphorus [P], and potassium [K]) and organic carbon. The first is based on flows of materials that might come from fossil fuels, such as mineral fertilizers, or through solar energy sources, such as manure, legumes, green manure, and so forth. The latter requires only flows from solar energy sources (biomass).

Replenishment of soil fertility (nitrogen [N], phosphorus [P], and potassium [K]): N, P, and K budgets for the agricultural sector, i.e., cropland and pastureland, of the whole of Spain were performed every 10 years from 1960 to 2008. To this end, N, P, and K inputs and outputs in pastureland and cropland were accounted for. Input data of N, P, and K through a variety of synthetic fertilizers were obtained from the International Fertilizer Industry Association (https://www.fertilizer.org/). To estimate N input through atmospheric wet deposition, we followed the approach of Garcia-Ruiz et al. (2012), and we took into account the mean annual rainfall and nitrate and ammonium concentration of the rainwater of 17 weather stations distributed throughout Spain. N inputs via irrigation were estimated taking into account irrigation volume for agriculture and the concentration of available N in river water (water database of the European Environment Agency [EEA]; http://www.eea.europa.eu/data-and-maps). N fixation by the 28 main Spanish N-fixing crops was estimated. Fixed N was calculated from the total N biomass produced, including the belowground and aboveground unharvested part, and assuming that 60% (grand mean from the review of Gathumbi et al. [2002]) of that was fixed from the atmosphere. N fixation in pastureland was estimated from the mean annual aboveground biomass productivity of pastureland in the wet, dry, and semiarid regions of Spain and taking into account that legumes compromise 8.5% of that biomass on average (MMARM 2010). Manure N, P, and K production was estimated based on the balance between N, P, and K ingestion and production. N, P, and K ingestion was calculated from the available feed, i.e., part of the harvest toward the livestock, imported feed, and grazing, and N contained in gross livestock production was estimated based on livestock dry matter production, multiplied by ratios of live weight to marketable products and N contents from Bodirsky et al. (2012).

The main outputs of N, P, and K from the cropland were as harvests, whereas in the case of pastures they were as grazed biomass. With respect to N, denitrification, NH_3 volatilization, and nitrate leaching were also accounted for. We estimated N, P, and K output in cropland by harvest for main cereals (18 cereal crops), grain legumes (16), potatoes, vegetables (39 crops), vineyards (for dessert and wine grapes), tree and fruit crops (40), olive orchards (for olive oil and table olives), textile and oil crops (17), sugarcane, miscellaneous crops (17), flavoring crops (4), and artificial pasture and forage crops (38) throughout the period studied from the yearbooks and reports of the Spanish Agriculture Ministry. This information together with N, P, and K contents of the edible harvest (U.S. Department of Agriculture [USDA] National Nutrient Database for Standard Reference and USDA Natural Resources Conservation Service; https://ndb.nal.usda.gov/ndb/search/list) was used to estimate the N, P, and K outputs of the harvest. Outputs of N, P, and K for pastureland were estimated from the grazed biomass taking into account the grazing livestock size and mean N, P, and K content of the grazed biomass. N lost by denitrification was mainly estimated from the synthetic N and manure N applied to cropland, or from excreted N on pastureland, taking into account Intergovernmental Panel on Climate Change (IPCC 2006) emission factors. NH_3 volatilization was estimated from the *EMEP/CORINAIR Emission Inventory Guidebook* (EEA 2007), which provided default NH_3 emission factors for different chemical N fertilizers and manure N, and excreted N (for pastureland) once they have been applied. N nitrate lost through leaching was estimated from IPCC (2006) default values for synthetic N and manure N or excreted N application rates.

Replenishment of organic carbon: The soil organic carbon (SOC) model is an adaptation of the Henin-Dupuis model, with one active pool of soil organic matter, containing 58% C (Mann 1986), which is mineralized at a constant rate each year and is replenished by inputs of carbon, each one with a specific humification coefficient. We did not attempt to estimate C sequestration rates in each time frame but preferred to assess the C balance in each time frame by calculating SOC stock at equilibrium in each land use category and for the whole territory. In this way, the uncertainty associated with the estimation of SOC stocks at a given time point, which is dependent on previous history, is avoided, while it still allows a fair comparison of the contribution of each management and land use arrangement to C stocks in the different time frames. We assumed that the active SOC mineralization rate was 1% on rain-fed soils and 2% on irrigated soils, based on Saña et al. (1996) and Sofo et al. (2005). We applied average values of C retention per unit C input (humification coefficients). Biomass production data, expressed as dry matter, were estimated as described in Soto et al. (2016). The carbon content of plant residues and external inputs were taken from Carranca et al. (2009), Boiffin et al. (1986), Rahn and Lillywhite (2002), Bilandzija et al. (2012), and Ono et al. (2009). Humification coefficients of herbaceous crop residues (average 12.5% for cereals and 8% for legumes) were based on a meta-analysis of carbon sequestration in Mediterranean soils (Aguilera et al. 2013) and other sources (Boiffin et al. 1986, Kätterer et al. 2011). In the case of woody crop residues, humification coefficients (36% on average) were based on data from Sofo et al. (2005) and Repullo et al. (2012). For external organic inputs (mainly manure), we used a humification coefficient of 31%, which is the median value (N = 25) of the percentage of C input contributing to net C sequestration in external organic input categories in Aguilera et al. (2013), and it also matches the data from Kätterer et al. (2011) and Andrén and Kätterer (1997). For roots, we assumed a humification coefficient of 39%, based on Kätterer et al. (2011).

2. Biodiversity: The maintenance of this fund element was evaluated by the biodiversity EROI (see Guzmán Casado and González de Molina 2017, Guzmán et al. 2017).

3. Woodland: The maintenance of this fund element was evaluated by the woodening EROI (see Guzmán Casado and González de Molina 2017, Guzmán et al. 2017).

Scenario without imported feed

AM could be used as a systemic sustainability evaluation tool. It allows us to know the consequences for the functioning of agroecosystems and the quality of fund elements of any alteration of energy and material flows. As an example of this aspect, we have constructed a scenario of the year 2000, but without imported feed. The year 2000 was chosen because it marked the change of century and was clearly a precrisis period.

To this end, the metabolizable energy represented by animal feed was quantified, and the part of the livestock population that actually consumed it was subsequently eliminated. The impact of this elimination on biomass, carbon, and nutrient flows (N, P, and K), as well as on the state of the fund elements, was then quantified.

RESULTS

Functioning of the agroecosystem

The beginning of the 1960s was a key period after which the modernization of Spanish agriculture accelerated. A first period, 1960-1986, can be distinguished, in which crop yields grew constantly as a result of the use of the complete package of the green revolution. In a second period, 1986-2008, the intensification process of Spanish agriculture continued, but its evolution was shaped by Spain's incorporation into the European Economic Community (1986). During this stage, Spanish agriculture became specialized in products with a higher demand in the European Union, i.e., olive oil, fruit, and vegetables. In parallel, low-productivity land was abandoned, generally grain cropland devoted to feed use and pastureland, whereas high-protein feed imports skyrocketed (Soto et al. 2016).

Between 1960 and 2008, the intensification of Spanish agriculture was based on the increase in EIs, and this meant multiplying the external energy invested in the agroecosystem by 5.5. This increase can be subdivided into several items. Although human labor input decreased to one-fifth, industrial inputs were multiplied by 5, rising from 62 to 314 PJ, and imported biomass, mainly from Latin America, rose from 12 to 193 PJ (an increase of 1508%; Fig. 1). Of this, 85% and 97% corresponded to feed, respectively, and the rest to seeds (Table A1.2 in Appendix 1).

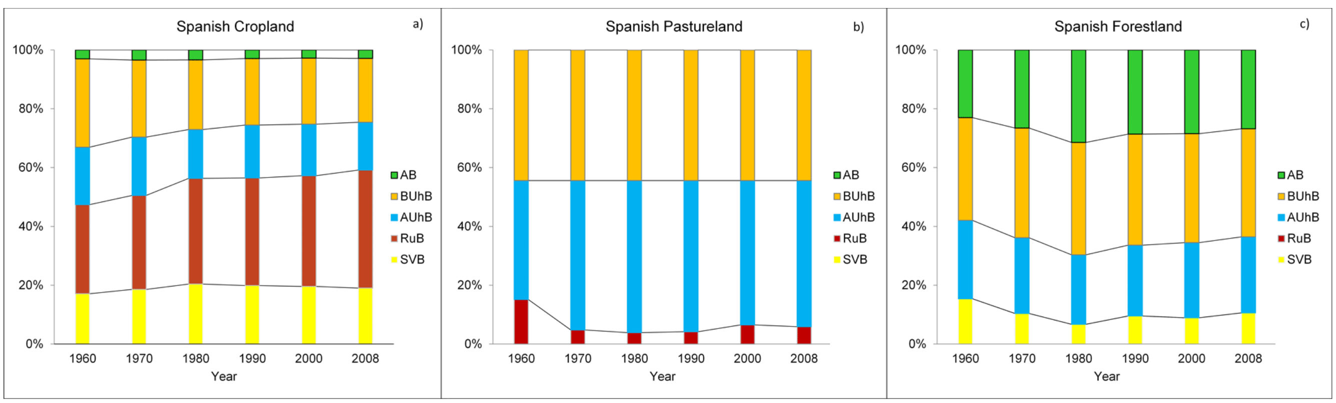
Of the industrial inputs, energy used for crop protection grew the most during the period studied, by a factor of 35.2. It was followed by traction energy (which multiplied by 10.2), irrigation energy (by 7.7), and mineral fertilizers (by 2). This moderate increase in fertilizers was for two reasons. First, mineral fertilizers were an early addition to the Spanish agroecosystem. In fact, in 1960 they represented 62% of the energy of industrial inputs (Table A1.2 in Appendix 1). Second, this modest growth must be linked with a phenomenon inherent to semiarid agroecosystems typical of the Mediterranean: the lack of rainfall means that the application of more fertilizer is of limited use in terms of increasing NPPact in the absence of optimum hydric conditions. For this reason, its growth is also related to the rise in irrigation energy (24% of industrial input in 2008) and in the irrigated land area, which rose by 82% between 1960 and 2008. In energy terms, the introduction of mechanical technologies played a greater role, now accounting for 41% of industrial inputs (Table A1.2 in Appendix 1).

As a consequence of the increase in EIs, some limiting factors were overcome to a certain extent, e.g., nutrients, water, and so forth, and afforded greater protection against heterotrophic organisms, which translated into a greater NPPact (Fig. 1).

Table 1. Actual net primary productivity, socialized animal biomass, external inputs, and agroecological energy return on investment (EROI) for the whole Spanish territory (in PJ).

	1960	1970	1980	1990	2000	2008	No import
Actual net primary productivity *(NPPact)* (a + c + d + e)	4800.0	5073.0	4986.9	5215.9	5566.5	5625.2	5566.5
Socialized vegetable biomass (*SVB*)(a)	425.2	412.0	396.4	478.2	491.6	505.7	491.6
Socialized vegetable biomass (cropland)	234.6	264.7	300.1	341.3	356.5	341.3	356.5
Socialized vegetable biomass (forestland)	190.6	147.3	96.2	136.9	135.0	164.4	135.0
Socialized animal biomass (*SAB*)(b)	20.2	35.8	54.9	72.6	97.7	105.9	97.7
Socialized biomass (*SB*)(a + b)	445.3	447.8	451.2	550.8	589.3	611.5	589.3
Reused biomass (*RuB*)(c)	746.7	563.4	609.6	714.9	832.0	854.7	832.0
Unharvested biomass (*UhB*)(d)	3300.6	3671.3	3480.5	3560.7	3762.3	3798.4	3762.3
Aboveground unharvested biomass (*AUhB*)	1482.7	1778.6	1659.2	1713.1	1800.5	1825.4	1800.5
Belowground unharvested Biomass (*BUhB*)	1817.9	1892.6	1821.3	1847.7	1961.8	1973.0	1961.8
Accumulated biomass (*AB*)(e)	327.6	426.2	500.4	462.1	480.5	466.5	480.5
External inputs (*EI*)	92.7	248.8	401.9	352.4	482.1	510.3	345.1
NPPact EROI = *NPPact*/(*RuB* + *UhB* + *EI*)	1.16	1.13	1.11	1.13	1.10	1.09	1.13
Biodiversity EROI = *UhB*/(*RuB* + *UhB* + *EI*)	0.80	0.82	0.77	0.77	0.74	0.74	0.76
Woodening EROI = *AB*/(*RuB* + *UhB* + *EI*)	0.08	0.10	0.11	0.10	0.09	0.09	0.10

Fig. 2. Evolution of actual net primary production (in PJ) by its use in relative terms in Spanish cropland (a), Spanish pastureland (b), and Spanish forestland (c). AB, accumulated biomass; AUhB, aboveground unharvested biomass; BUhB, belowground unharvested biomass; RuB, reused biomass; SVB, socialized vegetable biomass.



However, this growth was very moderate (17%) and involved a negative return on the total energy invested to obtain that increase (NPPact EROI), which meant a certain degree of degradation of the Spanish agroecosystem (Table 1).

In addition, the growing use of EIs made it possible to modify the pattern of the social use of NPPact (Fig. 2). Basically, the RuB and UhB on cropland and pastureland were affected. The changes in RuB were driven by the large increase in the livestock population, mainly monogastric animals (porcine and avian), and the change from extensive to intensive farming (Soto et al. 2016). This profound change in the composition and management of livestock would not have been possible without the massive importation of feed, mainly soya and maize, which, for agroclimatic and economic reasons, was difficult to produce in Spain. In consequence, pastureland was partly abandoned. In parallel, growing amounts of high-quality biomass (grain and forage) from cropland have been devoted to livestock. On cropland, the RuB rose from 30% of NPPact in 1960 to 40% in 2008. Meanwhile, on pastureland, it fell from 12% to 6% (Fig. 2). These imbalances in the intensity of land use, i.e., sharp intensification versus abandonment, were also seen inside the cropland. Over this period, the area of cropland fell by 3.1 Mha (15% of the total) mainly because of the abandonment of rain-fed land with little response to EIs (Soto et al. 2016). The change in the pattern of social use of the biomass is also expressed in the increase in the burning of straw and other crop residues, mainly in the 1980s and 1990s, which were no longer used to feed livestock. Between 1960 and 1990, biomass burning in cropland rose from 0.6 to 3.6 Mt. This trend has been attenuated over the past 2 decades because of public policies addressed to restrict crop residue burning. Biomass burning fell from 3.6 to 1.3 Mt dry matter between 1990 and 2008.

Inversely, a smaller proportion of biomass is abandoned on cropland. The UhB fell from 50% of NPPact to 38%. In relative

Table 2. Nutrient balance (N, P, and K) for the different land uses and for the whole Spanish territory, 1960-2008 and no-import scenario.

	1960	1970	1980	1990	2000	2008	No import	1960	1970	1980	1990	2000	2008	No-import
	Gg N							kg N ha^{-1}						
Cropland	169	316	561	617	787	625	618	8.3	15.1	27.4	30.6	43.0	36.2	33.8
Grassland	125	231	281	331	366	357	309	7.8	15.4	19.1	23.0	23.5	24.1	19.9
Total	293	547	842	948	1152	982	927							
	Gg P							kg P ha^{-1}						
Cropland	116	152	162	194	265	176	226	5.7	7.3	7.9	9.6	14.5	10.2	12.4
Grassland	-1	6	7	15	16	17	9	-0,1	0.4	0.5	1.0	1.0	1.2	0.6
Total	115	159	170	209	281	193	235							
	Gg K							kg K ha^{-1}						
Cropland	-108	-69	-40	55	214	142	106	-5.3	-3.3	-2.0	2.7	11.7	8.2	5.8
Grassland	-132	-7	5	47	22	29	-11	-8.3	-0.5	0.3	3.3	1.4	2.0	-0.7
Total	-240	-76	-35	102	236	170	95							

terms, the reduction in BUhB was especially dramatic, falling by 9% (Fig. 2). This is a clear indication that the fertility of the soil has become the responsibility of mineral fertilizers, to the detriment of organic material. The use of herbicides is the main cause of this relative fall. On pastureland, however, UhB rose by 9% as a result of abandonment.

As a consequence of these processes, SB grew by 37% between 1960 and 2008, mainly because of the 425% increase in animal biomass. SVB grew by only 19%, as the result of a 45% increase in SVB produced from crops but a fall of 14% in forest SVB (Table 1). The latter was reduced because of the substitution of firewood by fossil fuels in the home. On forestland, the lower extraction of firewood has contributed to an increase in AB, which we have quantified at 45% (Table A1.3 in Appendix 1). The increase in cropland SVB (45%) was greater than that seen in cropland NPPact (30%). The increase in the harvest index of the varieties introduced in the green revolution contributed to this gap. The loss of the use of agrarian residue as animal feed and for the replenishment of soil fertility, as a result of the importation of external energy, is the driver of this modification in patterns of plant biomass partitioning.

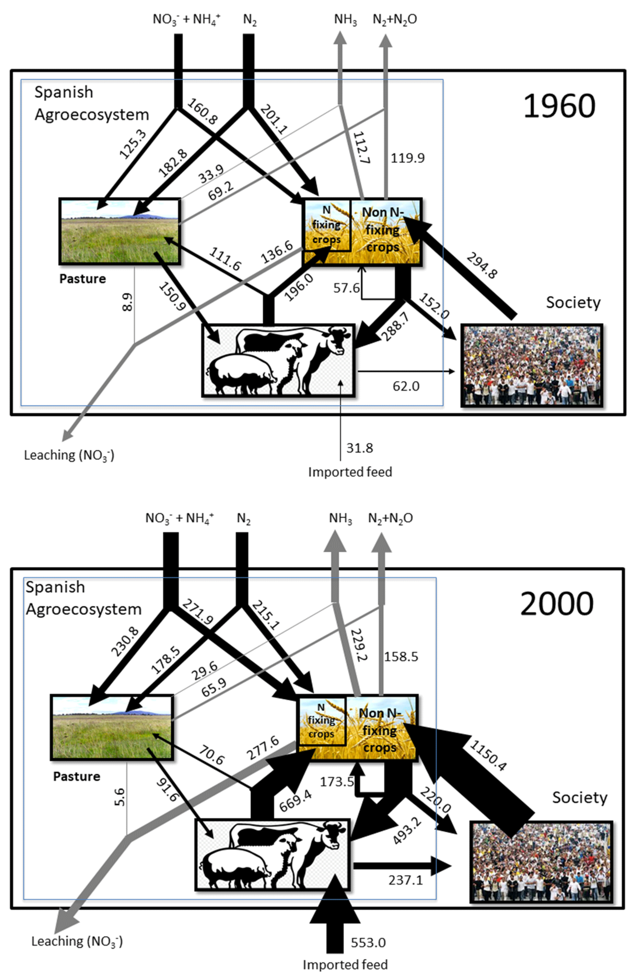
Finally, the import of EIs allowed rotation to be simplified and the substitution of legumes, which were no longer essential to incorporate nitrogen into the agroecosystem. Between 1960 and 2000, the crop area of legumes fell from 1.4 to 0.6 Mha. In parallel, N input by biological N fixation went from 28% of inputs to only 11% in the year 2000, indicating a change in the agricultural model based on the application of EIs and not on a model reliant on biological N sources (Fig. 3).

State of the fund elements

Soil

Replenishment of soil fertility (N, P, and K) closing nutrient cycles on an agroecosystem scale: Between 1960 and 2008, Spain converted a relatively balanced equilibrium in N and P (≈ +8.3 kg N ha^{-1} yr^{-1} and ≈ +5.7 kg P ha^{-1} yr^{-1}) or a slightly deficit balance for K (≈ -5.3 kg N ha^{-1} yr^{-1}) on cropland into a significant surplus, and this was particularly true for N (≈ +43.0 kg N ha^{-1} yr^{-1} in 2000; Table 2). The pattern of the increasingly positive balance from

Fig. 3. Nitrogen flows (Gg) in Spanish agroecosystem in 1960 and 2000.



1960 onward was mainly because of the 3.2, 1.3, and 4.0 times

Table 3. Soil organic carbon stocks at equilibrium for the different land uses and for the whole Spanish territory, 1960-2008 and no-import scenario.

	1960	1970	1980	1990	2000	2008	No import
				Tg C			
Cropland	754	638	604	638	687	675	632
Grassland	1415	1333	1230	1244	1422	1450	1422
Woodland	745	891	866	873	930	951	930
Total	2914	2862	2700	2756	3040	3076	2984
				Mg C ha^{-1}			
Cropland	36.9	30.6	29.5	31.7	37.6	39.1	34.6
Grassland	88.9	88.7	83.6	86.5	91.5	92.7	91.5
Woodland	74.4	81.9	74.8	71.7	74.3	70.7	74.3
Weighted average	57.7	56.7	53.5	54.6	60.2	60.9	59.1

increase in the application of N, P, and K in mineral fertilizers together with that applied as manure (Table A1.4 in Appendix 1). The increase in availability of imported manure boosted N, P, and K inputs on cropland, which multiplied by between 1.83 and 2.25 during that period. These increases in nutrient inputs on cropland did not see a concomitant increase in harvested nutrients as they only doubled during that period. Therefore, over the period studied, N, P, and K use efficiencies decreased from 0.54 to 0.39, 0.42 to 0.35, and 1.23 to 0.79, respectively.

The decreasing trend in the N efficiency use of Spanish cropland over the period studied was concomitant with an increase in N losses per kilogram of harvested N. Certainly, although in 1960 0.5 kg N was lost for each kilogram of harvested N, this increased by 50% in 2000 (0.73 kg N loss kg^{-1} harvested N). N_2O emissions, which, in magnitude, are considered to be the third largest greenhouse gas contribution to stratospheric ozone layer depletion, increased by 2.7 during the study period, and figures were 2.0 and 3.0 for NH_3 emissions and nitrate leaching on cropland (Table A1.5 in Appendix 1).

Replenishment of organic carbon: In both ecosystems and agroecosystems, the main driver of SOC is biomass input (Rodríguez-Martín et al. 2016). The fundamental difference is that, in agroecosystems, the magnitude of this input is both directly and indirectly conditioned by the farming method: directly, because many farming practices, such as residue burning, organic fertilization, use of herbicides, and so forth, intentionally modify the magnitude of the input; and indirectly, because the NPPact is affected by farming methods, to the extent that it affects the state of the fund elements and/or modifies the availability of limiting factors. Therefore, the SOC balance is the result of different processes that are, at times, contradictory.

Table 3 shows equilibrium SOC stocks in each time frame. Given that there are significant differences in SOC stocks between land uses, part of the change in total C stocks shown in Table 3 was because of land use changes, and part of it was because of changes in equilibrium C stocks in each type of land use. There was a marked drop in equilibrium SOC stocks in the period 1960-1980. This occurred particularly in cropland, and it was mainly because of the expansion of residue burning practices, herbicides, and tillage during this period, as well as the consolidation of varietal change. As from 1990, there was a recovery in cropland equilibrium SOC stocks, which surpassed the 1960 levels in 2000. This was attributable to, first, the restriction of crop residue burning; second, the increase in manure inputs because of the continued expansion of the livestock population; and, finally, the increase in unharvested residue, as a consequence of the greater NPPact and the decrease in the grazing of residue.

Biodiversity

Biodiversity EROI decreased by 8%, indicating a decrease in UhB in relation to TIC, which entails a lower level of relative energy availability for wild heterotrophic organisms (Table 1). The change in the pattern of biomass use on cropland, mentioned previously, was responsible for this fall. The sharp increase in the amount of energy invested in cropland in the studied period increased NPPact by 30% in absolute terms, but the increase in productivity was invested mainly in RuB, which grew by 73% (Table A1.3 in Appendix 1). The shift toward animal feed production had a negative impact on biodiversity. This effect was not compensated by the abandonment of pastureland and forestland. In short, the disassociation of the agroecosystem in areas of intensive production and abandoned and/or protected areas has not brought about a significant increase in the trophic energy available for transfer from plants to other levels in the food webs.

Woodland

Woodening EROI grew by 14% between 1960 and 2008. That is, the growth rate of AB was higher than that of the energy invested in the agroecosystem. AB rose from 327.6 to 466.5 PJ. Of this growth (138.9 PJ), approximately 7% corresponded to the expansion of woody crops (mainly almond and olive groves; 10.2 PJ). An additional 19% (26.2 PJ) was because of the lower extraction of forestland SVB (Table 1). The remaining 74% of AB growth was because of the growth of forestland in areas freed from agricultural activities.

Scenario without imported feed

In the early 21st century, imported feed was 38% of EIs and 21% of N inputs into the agroecosystem. Given its dimension, the alteration of this flow could substantially modify the functioning of the agroecosystem and the state of the fund elements. However, with respect to the functioning of the agroecosystem, because imported feed was fed to livestock, its elimination only directly affected the flow of SAB. Indirectly, it could affect the NPPact and its components bringing changes in the quality of the fund elements.

Table 1 shows that the SAB would be reduced by 27%. This fall would mainly affect food produced from monogastric animals (pigs and poultry), because they are the main consumers of this feed. Given that the animals are poor energy converters, the impact on SB as a whole would be low (approximately 4%).

Table 2 shows that the excess of N per hectare would be reduced by 21% on cropland and by a little less (15%) on pastureland. This fall would bring with it a reduction of losses of 4%, 6%, and 7.5% because of denitrification, volatilization, and leaching, respectively.

However, reduction in the livestock population in the "no feed import" scenario would not have a major effect on equilibrium SOC stocks on cropland (Table 3). This is because of the fact that this reduction is mainly focused on animal species whose manure is often managed in liquid forms, with a low C:N ratio.

Finally, the agroecological EROIs calculated show a slight increase (Table 1). This is a consequence of the reduction of EIs and the maintenance of NPPact, because the elimination of imported feed does not cause any deterioration of the soil, and may even improve the state of other fund elements, as it would reduce N losses.

DISCUSSION

The pattern of agrarian intensification based on external energy, in comparison to intensification based on the increase of the density of low-entropy internal loops, became generalized in parallel to the implementation of the green revolution worldwide (González de Molina and Guzmán 2017). The case of Spain is no different from that of other countries. However, the high weighting of imported feed in EIs is less common. In fact, globally, Spain forms part of the leading group of countries in net nutrient, especially nitrogen, imports in the form of feed (Lassaletta, Billen, Grizzetti, et al. 2014), with much higher figures than neighboring countries such as France, where imported N used for animal feed in 2006 was 21% of the total nitrogen consumed by animals (Le Nöe et al. 2017), in comparison with Spain's 47% in 2000. In contrast, other European countries, such as the Netherlands, Denmark, and Belgium, share Spain's intensive livestock farming methods, sustained by the import of large amounts of feed (Lassaletta, Billen, Romero, et al. 2014).

EIs have broken the relative equilibrium in the land and biomass uses required by traditional farming. In short, the shift toward DE (SVB + RuB) versus UhB is based on several strategies. First, industrialized farming uses herbicides and/or intensive tillage to prevent the growth of phytomass that farmers are not interested in appropriating. Second, modern varieties, mainly cereals, have been selected to increase the harvest index (Sanchez-Garcia et al. 2013). Finally, the role of biomass in the maintenance of soil fertility was replaced, to a certain extent, by mineral fertilizers and pesticides, whose most extreme manifestation was the increase in crop residue burning. This loss of functionality allows us not only to break the balance between the different uses of biomass within a given space, e.g., cropland, but also to interrupt or reduce the biomass and nutrient flows between territories with different synchronic or diachronic uses, e.g., pastureland to cropland or between legumes and nonlegumes. The transfer of nutrients from pastureland and dry farming areas devoted to animal feed to more productive areas of cropland was not uncommon in traditional Mediterranean agriculture (Cussó et al. 2006, Guzmán Casado and González de Molina 2009). The process of intensification not only interrupted this dynamic, but also converted pastureland into the net recipient of nutrients throughout the importation of animal feed (Fig. 3).

With respect to the changes in biomass use patterns and agroecological EROIs, it is not possible to compare with other cases on a national scale, given the novelty of the methodology used. However, at the crop scale, the study of coffee in Costa Rica (1935 and 2005) reveals that intensification based on EIs gave rise to similar changes. The intensification of the coffee agroecosystem was accompanied by a small increase (8%) in NPPact, which gave rise to a negative NPPact EROI return, and a sharp increase (27%) in RuB and AB (95%), resulting in a notable increase in woodening EROI (50%). In line with what has happened in Spain, the increase in such uses reduced the amount of biomass abandoned in the coffee fields (UhB; -3%), reducing the biodiversity EROI by 26% (Infante-Amate et al. 2017). On a local scale, these changes in biomass use patterns were also documented in Santa Fe (Granada, Spain) between 1934 and 1997. This municipality in the southeast of Spain represents one of the earliest and most profoundly intensified areas of the country (Guzmán Casado and González de Molina 2017). In this municipality, the trend of these EROIs was similar to what we found for Spain, with the exception of the NPPact EROI. In Santa Fe, the NPPact EROI grew between 1934 and 1997, because of the sharp increase in water consumption and the expansion of crops with higher biomass production, such as poplar (Guzmán Casado and González de Molina 2017).

With regard to N fluxes, these results are similar to those of Leip et al. (2011) who found an N surplus for Spain in the period 2001-2003 of about 50 kg N ha^{-1} yr^{-1}, slightly lower than the mean for the European Union-27 (EU-27). Our data on annual anthropogenic N inflows in 1960 (326 Gg N yr^{-1}; chemical fertilizer plus imported N feed) and 2000 (1703 Gg N yr^{-1}; Fig. 3) were similar to those calculated by Lassaletta, Billen, Romero et al. (2014) for the periods of 1961-1964 (405 Gg N yr^{-1}) and 2000-2009 (1558 Gg N yr^{-1}). In short, the Spanish agroecosystem went from a balanced equilibrium of macronutrients in 1960, in which 68%, 41%, and 82% of N, P, and K flows took place within the agricultural sectors, to a very imbalanced situation in 2000, when only 34%, 13%, and 34%, of the N, P, and K flows were provided by the agroecosystem itself. As well as growing dependence and lower efficiency, this imbalance led to significant losses of N, which, because it is a mobile element, is not stored in the agroecosystem. This nutrient dissipates into the environment and cascades through air, water, and terrestrial ecosystems where it contributes to a multitude of effects, including adverse impacts on human health, ecosystem services, and climate change (Galloway et al. 2003, Erisman et al. 2013). NH_3 volatilized from cropland in 2008 (185 Gg N) was higher than the value for the same year estimated by the official Spanish N balance (MAPAMA 2017), but lower than the value estimated by Sanz-Cobena et al. (2014) using a more detailed approach. NH_3 has a short atmospheric lifetime and is usually deposited near its source, contributing to the eutrophication of natural waters (Grizzetti et al. 2011), increased N input into natural terrestrial ecosystems, causing biogeochemical imbalances (van Herk et al. 2003), and increased susceptibility to stress and changes in soil and plant communities (Dise et al. 2011). Nitrate leaching is related to the eutrophication of ground and surface watercourses and estuaries (Fowler et al. 2013) and poses a recognized risk to human health. With regard to biodiversity, high N availability generally causes biodiversity decline in both terrestrial and aquatic ecosystems. The most widely reported consequences of available N enrichment in terrestrial ecosystems are declines in plant species richness and evenness (Suding et al. 2005, Bobbink et al. 2010). The proposed mechanism of a reduction in plant biodiversity driven by an excess of N availability includes ammonium toxicity, acidification, light exclusion, increased susceptibility to secondary stress factors, and changes to plant-soil feedback. The reduction of species biodiversity in aquatic ecosystems because of an excess of available N is related to eutrophication and water hypoxia driven by the high productivity of low diverse algal blooms promoted by high available N and P (Sala et al. 2000). Therefore, although this surplus does not affect the quality of the soil, it does negatively affect other fund elements such as water, biodiversity, and the atmosphere.

Regarding carbon, our estimations of SOC stocks at equilibrium are in line with published empirical studies reporting SOC stocks for the different land uses in Spain. Our estimation of SOC stocks in croplands of 34.5 Mg C ha^{-1} on average for the period 1960-2008 is somewhat lower than the 43.5 Mg C ha^{-1} estimated by Rodríguez-Martín et al. (2016) in a comprehensive assessment of SOC in Spanish croplands. In the case of grassland and forestland, our average values of 92.7 and 70.7 Mg C ha^{-1}, respectively, are between the values reported by Rodríguez-Martín et al. (2016) of 64.1 and 69.3, respectively, and those reported by Doblas-Miranda et al. (2013) of 103.0 and 101.6, respectively.

The relatively low values in cropland indicate that agricultural land is on the threshold of degradation (Romanyà et al. 2007, Rodríguez-Martín et al. 2016). In theory, the increase in cropland NPPact resulting from intensification, together with the massive importation of feed, could have brought about a substantial increase in the return of organic carbon to the soil. However, the imbalance between biomass uses, depressing UhB, and the preferential use of feed for pigs and poultry prevented this from occurring.

On the other hand, in terms of SOC stock at equilibrium with respect to the recycled biomass in the cropland (cropland RuB + cropland UhB + imported feed consumed by the livestock on the cropland), the evolution was markedly negative. It fell from 685 g C MJ^{-1} in 1960 to 440 g C MJ^{-1} in 2008. In other words, it was necessary to recycle 56% more biomass to obtain the same SOC stocks in cropland. From the perspective of climate change mitigation, clearly the current biomass management strategy is not adequate.

In summary, the alteration of the functioning of the agroecosystem because of the intensification of EIs has directly encouraged the degradation of the soil and of biodiversity and, indirectly, of water, biodiversity and the atmosphere. The fall in the NPPact EROI reflects this deterioration, with cropland being the space most affected. Biodiversity could be especially harmed by the synergistic interaction between biomass and N fluxes. Furthermore, the increasing use of pesticides, whose effects we have not considered, would be added to this situation.

Woodland is the only fund element that evolves favorably, mainly as a result of the increase in AB in the forest. However, when examined in greater detail, it can be seen that the improvement occurred between 1960 and 1980. From then onward, this EROI began to see a significant fall. Once the process of substituting firewood as fuel was completed, i.e., a social change unconnected with the management of the agroecosystem, and the rate of change of the use of agricultural soil toward woody crops, which could not be indefinite, slowed down, the rate of growth in the amount of energy invested was much higher than that of AB. This occurred despite the fact that the continuous intensification of Spanish cropland meant that 3.5 Mha had been abandoned since 1980 (Table A1.1 in Appendix 1). This territory could theoretically be transformed into forest or shrubland, increasing the AB. However, the abandoned semiarid farmland was often close to degradation thresholds, which, in borderline agroclimatic conditions (semiarid, dry climate), do not recover but trigger the desertification of the territory (Loveland and Webb 2003, Romanyà et al. 2007). On the other hand, the liberation of Spanish territory for reforestation was because of the import of feed, mainly from Latin America (Lassaletta, Billen, Romero, et al. 2014), where it has possibly contributed to deforestation (Bettwy 2006).

From the social perspective, the return in the shape of SB cannot justify the unsustainable process of intensification that has been applied. In Spain, the most important driver of this shift was not the increase in the human population, but the rapid change in dietary patterns, which evolved from a typical Mediterranean diet to an animal protein–rich diet. The increase in the share of animal protein has been more intense, from 37% in the 1960s to 65% today (Lassaletta, Billen, Romero, et al. 2014). This is significantly above World Health Organization recommendations (WHO 2007).

The reduction of SAB by 27% in the scenario without imported feed would bring an improvement in public health, a reduction in environmental impacts related to excessive nitrogen, with hardly any trade-off with the replenishment of SOC, and with an improvement in energy return rates in the form of total, unharvested, and accumulated phytomass.

CONCLUSIONS

The application of the metabolic approach to Spanish agriculture has shown that the intensification model based on EIs not only generates alterations in the relationship between input and output flows of energy and materials, which is how intensification has usually been evaluated, but also brings about significant changes in the internal functioning of the agroecosystem. In short, the reduction of the density of internal loops generated by new technologies has made the agroecosystem more dependent on external inputs, especially those from fossil fuels. These changes are perfectly captured by the AM framework.

A second conclusion is that the study of energy and material fluxes, which constitute the basis of social metabolism, allow us to explore the environmental impacts of agricultural industrialization, which had been out of the scope of metabolic methodologies so far. Our proposal on AM, in this sense, constitutes an innovation, incorporating tools that allow an assessment of the environmental impacts of a given type of management or a particular metabolic arrangement. In the Spanish case, the results show the deterioration of two of the fund elements studied: soil and biodiversity. This deterioration undermines the productive capacity of the agroecosystem, which is reflected by the NPPact EROI.

Third, given that the provision of ecosystem services by agroecosystems depends on the state of their fund elements, we can conclude that AM, through the assessment of the state of fund elements, allows us to inquire about the capacity of agroecosystems for provisioning ecosystem services. As we have demonstrated for the Spanish case, the deterioration of fund elements has reduced the provision of ecosystem services such as unpolluted water, carbon sequestration, or those derived from biodiversity, such as pest and disease control, which translates into a continued growth in pesticide use.

Finally, we consider that these results highlight the importance of choosing a correct strategy for agroecological intensification. The results obtained indicate that it is not simply a question of producing more biomass, but also of establishing the appropriate balance between different uses of that biomass and even between different animal species.

Responses to this article can be read online at:
http://www.ecologyandsociety.org/issues/responses.php/9773

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Appendix 1

Biophysical macro magnitudes of Spanish agriculture (1960-2008 and Scenario without imported feed)

Table A1.1. Land uses evolution (Mha), Spain

	1960	1970	1980	1990	2000	2008
Cropland	20	21	20	20	18	17
Closed Forest	5	6	7	7	7	8
Coppice	5	5	5	5	5	5
Dehesa	3	4	4	4	4	4
Pastures and Shrubland	13	11	11	11	12	11
Unproductive	4	4	4	4	4	4
Total	50	50	50	50	50	50

Table A1.2. External inputs evolution (TJ), Spain

	1960	1970	1980	1990	2000	2008	No-import
External Inputs (EI)(a+e)	92,690	248,832	401,945	352,368	482,101	510,260	345,112
Non-industrial Inputs (a) = b+c+d	30,505	68,799	137,283	61,149	146,955	196,434	9,966
Feed (b)	10,289	53,403	124,365	51,822	136,989	187,842	0
Seed (c)	1,785	0	557	0	3,674	5,335	3,674
Human Labor (d)	18,431	15,396	12,361	9,327	6,292	3,257	6,292
Industrial Inputs (e) = f+g+h+i	62,185	180,033	264,662	291,219	335,146	313,826	335,146
Traction (f)	12,764	81,696	124,785	103,296	116,318	129,666	116,318
Irrigation (g)	9,769	20,712	37,435	58,565	77,110	75,407	77,110
Fertilizers (h)	38,768	75,241	93,842	104,988	112,039	77,618	112,039
Crop protection (i)	884	2,385	8,599	24,369	29,679	31,135	29,679

Table A1.3. NPPact evolution in cropland, pastureland and forestland (TJ), Spain

	Cropland					
	1960	1970	1980	1990	2000	2008
NPPact	1,371,287	1,426,425	1,471,174	1,715,584	1,821,935	1,793,445
Socialized Vegetable Biomass (SVB)	234,616	264,716	300,139	341,307	356,544	341,303
Reused Biomass (RuB)	415,610	456,273	528,539	628,004	686,282	719,606
Aboveground Un-harvested Biomass (AUhB)	267,260	283,029	244,192	307,763	319,000	292,004
Belowground Un-harvested Biomass (BUhB)	411,991	373,179	348,298	388,219	409,708	388,481
Accumulated Biomass (AB)	41,811	49,227	50,006	50,291	50,401	52,050
	Pastureland					
	1960	1970	1980	1990	2000	2008
NPPact	2,188,886	2,227,802	2,084,407	2,061,839	2,234,369	2,284,455
Socialized Vegetable Biomass (SVB)	0	0	0	0	0	0
Reused Biomass (RuB)	331,060	107,174	81,060	86,875	145,724	135,058
Aboveground Un-harvested Biomass (AUhB)	884,988	1,130,494	1,076,944	1,058,591	1,095,592	1,134,084
Belowground Un-harvested Biomass (BUhB)	972,838	990,134	926,403	916,373	993,053	1,015,313
Accumulated Biomass (AB)	0	0	0	0	0	0

Forestland						
	1960	1970	1980	1990	2000	2008
NPPact	1,239,886	1,418,681	1,431,276	1,438,500	1,510,151	1,547,289
Socialized Vegetable Biomass (SVB)	190,569	147,312	96,234	136,899	135,020	164,358
Reused Biomass (RuB)	0	0	0	0	0	0
Aboveground Un-harvested Biomass (AUhB)	330,472	365,113	338,098	346,726	385,922	399,289
Belowground Un-harvested Biomass (BUhB)	433,070	529,328	546,562	543,066	559,067	569,212
Accumulated Biomass (AB)	285,775	376,927	450,383	411,809	430,142	414,430

Table A1.4. Inputs of N, P and K in Spanish cropland and pastureland (Gg)

	Crop residues			Rainfall	Free living N fixation	Symbiotic N fixation	Irrigation			Mineral fertilizers			Manure/Excretions			Seeds		
	N	P	K	N	N	N	N	P	K	N	P	K	N	P	K	N	P	K
								Cropland										
1960	40	8	76	168	41	160	21	0.3	22	274	129	69	237	58	194	17	3	4
1970	37	9	88	189	42	157	25	0.4	26	602	179	180	227	58	192	23	4	5
1980	64	13	142	245	41	178	32	0.5	33	874	188	225	305	72	208	21	4	4
1990	106	18	218	261	40	170	36	0.5	38	1043	225	307	331	82	245	24	4	4
2000	153	21	266	266	37	179	30	0.5	31	1120	263	404	490	121	337	23	4	4
2008	164	22	292	248	35	140	30	0.5	37	884	162	279	528	131	355	30	5	6
No-import	153	21	266	255	37	179	30	0.5	31	1120	263	404	333	82	229	23	4	4
								Pastureland										
1960				131	64	119							219	30	154			
1970				136	60	113							121	17	88			
1980				176	59	110							108	15	76			
1990				186	58	108							169	23	123			
2000				226	62	116							219	30	153			
2008				213	59	111							221	31	152			
No-import				216	62	116							172	24	120			

Table A1.5. Outputs of N, P and K in Spanish cropland and pastureland (Gg)

	Harvest/Grazing			Denitrification	NH_3 volatilization	Leaching
	N	P	K	N	N	N
Cropland						
1960	517	83	473	107	92	74
1970	611	98	560	122	115	139
1980	716	115	654	137	148	199
1990	849	136	757	145	165	235
2000	898	144	828	150	199	263
2008	891	144	821	138	185	219
No-import	898	144	828	143	172	251
Pastureland						
1960	273	32	286	74	44	18
1970	90	10	94	66	33	10
1980	68	8	71	64	32	9
1990	73	8	76	65	37	13
2000	125	14	131	72	43	18
2008	117	14	123	69	42	18
No-import	125	14	131	70	39	14