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ORIGINAL ARTICLE

Spanish agriculture from 1900 to 2008: a long-term perspective on agroecosystem energy from an agroecological approach

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Abstract According to the agroecological approach, energy analyses applied to agriculture should provide information about the structure and functions of the agroecosystem; in other words, about the maintenance of its fund elements, which sustain the flow of ecosystem services. To this end, we have employed a methodological proposal that adds agroecological EROIs to the existing economic EROIs. This methodology is applied here for the first time at the country level, and over a long-term historical period. The Spanish agroforestry sector, which is representative of Mediterranean agroclimatic conditions, has been studied on a decadal basis from 1900 to 2008, fully spanning its process of industrialization and modernization. The results show the loss of energy efficiency brought about by the industrialization of Spanish agriculture. The economic EROIs (FEROI, EFEROI and IFEROI) fell by 42, 93 and 12%, respectively. The shift towards livestock production and the dramatic increase in industrial inputs are the causes of this decline. With regard to agroecological EROIs, NPPact EROI and Biodiversity EROI fell by 6 and 15%, respectively. This suggests that the fund elements are being degraded and alerts us to low returns to nature in the form of un-harvested biomass available to aboveground and underground wildlife. Finally, Woodening EROI increased by 48%. Sixty percentage of this increment was due to the growth of woodland in areas freed from agricultural activities. However, this change in land use was partly due to feed imports from third countries where deforestation processes may well be taking place, an effect that has not been considered in the analysis.

Keywords Social metabolism · Ecosystem services · Land sharing · Land sparing · Land use change · EROI

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Introduction

Energy balances applied to agriculture commonly adopt an input–output approach, with the aim of assessing external energy invested per unit of energy contained in the product (food, fibre, wood) that leaves the system and is available for society. This approach is necessary but insufficient, since it inevitably conceals the internal agroecological functioning of farm systems within a black box (Tello et al. 2016). It is thus necessary to change this approach in order to reach a better understanding of the energy functioning of agroecosystems and provide keys to improving their sustainability (Guzmán and González de Molina 2015).

An EROI is an indicator that measures the efficiency of energy production and, as such, provides information to support decision-making on this vital aspect of the functioning of productive activities. For example, it has been used for some time on the conversion of oil and other primary energy sources into fuel and other energy products, attempting to measure the efficiency of the process (Giampietro et al. 2010; Hall 2011; Mulder and Hagens 2008). It is a significant instrument in “energy analysis” or “net energy analysis” (Hall et al. 2009). It is strictly economic in its way of reasoning and is based on the same valuation criteria as monetary investments, that is, on unidimensional cost–benefit analysis. It provides a numerical indicator that can be quickly and easily used for a comparison with other similar energy processes, in both space and time (Murphy et al. 2011).

When applied to agriculture, it measures the amount of energy invested in order to obtain one unit of energy in the form of biomass. Put more simply, we could say that an EROI in agriculture measures 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. This indicator is particularly important in the context of industrialized agriculture, which directly and indirectly uses large amounts of external energy and which faces the challenge of reducing its energy costs and GHG emissions. Given that the endosomatic metabolism of people and the production of raw materials, which are difficult to produce synthetically, can only be satisfied through the production of biomass, the efficiency of energy usage in agriculture has become a fundamental question (Tello et al. 2016).

However, energy efficiency cannot be reduced to a single number or a single criterion for analysis, as emphasized by Giampietro et al. (2010), especially when applied to agriculture. Furthermore, EROIs can also be more than a mere indicator of energy efficiency. If designed appropriately, EROIs can, in effect, become a measurement of metabolic efficiency, that is, of the exchange of energy between an agrarian system and the environment, and of whether that metabolic exchange is sustainable over time. This paper considers EROIs that look beyond the social benefits offered by investing more energy in agriculture. This requires us to recognize that it is necessary not only to invest energy in the production of biomass that is useful to society or to the farmer, but also to invest energy in the maintenance of the agroecosystem so that it can continue to produce biomass under the best possible conditions. As we shall see, the key lies in considering not only the energy cost of producing socially useful biomass, but also the cost of maintaining the ecosystem services provided by an agroecosystem: that cost does not end with the re-use of seeds or the production of animal feed (which corresponds only to the supply services provided by agroecosystems), but also extends to the maintenance of the rest of the ecosystem services (nutrient recycling, biological pest control, soil conservation, etc.). It is, therefore, necessary to adopt an agroecological focus. Agroecology operates at the interface between nature and society, taking the whole of the agroecosystem as its unit of analysis. The aim of this approach is to increase agrarian sustainability (Noorgard 1987; Gliessman 1998; Guzmán et al. 1999). Therefore, the development of methodological tools with an agroecological approach can contribute to a better understanding of the energy functioning of agroecosystems and can provide keys to improving their sustainability. In spite of this, little effort has been made to date to develop this potential (Guzmán and González de Molina 2015).

Based on the fund-flow approach, put forward by Georgescu-Roegen (1971), the fund elements of agroecosystems generate flows of ecosystem services, part of which are used for their own renovation (Folke et al. 2011; Millennium Ecosystem Assessment 2005). According to Schröter et al. (2014), every agroecosystem has a specific capacity to provide these services, depending on their soil and climate conditions. Nevertheless, agroecosystems are also dependent on human management, which affect the quantity and quality of fund elements, and therefore the rate at which they provide services also depends on how they are managed (Spangenberg et al. 2014). An adequate provision of services will depend on the overall health of the agroecosystem, that is, on the sustainability of its fund elements (Cornell 2010; Costanza 2012). Conversely, the degradation of the fund elements within an agroecosystem can reduce its supply of ecosystem services (Burkhard et al. 2011).

The fund elements of an agroecosystem require a specific amount of energy for reproduction and maintenance, which can only partially be replaced by external energy. For example, only biomass can feed the food chains that sustain the life within the soil and the general biodiversity of the agroecosystem. Hence, the degree to which society appropriates biomass is considered a biodiversity pressure indicator, which must be complemented by others such as EROIs or nutrient balances (Firbank et al. 2008; Haberl et al. 2013: 39). In other words, the fund elements of an agroecosystem, upon which an adequate supply of ecosystem services depends (among them the production of socially useful biomass such as food, fuel and fibres), are maintained or improved by means of adequate biomass flows (Smil 2013; Tittonell et al. 2012).

When an economic approach is taken to agrarian energy analysis, often only cultivated plants are taken into account and, among these, the aerial part of the plants, ignoring the adventitious plants, the root biomass and, very often, crop residues. An agroecological approach should take into account all the biomass produced within the limits of the agroecosystem, that is, all the Net primary production (NPP). From this perspective, it is also necessary to consider potentials trade-offs between competing uses of biomass (Tittonell et al. 2012).

These peculiarities of the throughput of energy in agroecosystems can be captured by an EROI if it is calculated in accordance with agroecological criteria. At the same time, an EROI of this type could be a means of measuring environmental quality or the state of the agroecosystem and its sustainability (Murphy et al. 2011). Therefore, the objective of agroecological EROIs is to ascertain whether a given agroecosystem is capable of maintaining its ecosystem services (sustainability) or whether it degrades them, requiring increasing amounts of external energy in order to compensate for the loss, albeit only partially (Guzmán and González de Molina 2015). In this paper, we take this novel approach to the question of energy efficiency in agroecosystems, which is "complementary" to traditional methods and which aims to bring an agroecological perspective to energy analysis.

We have applied this proposal to Spanish agriculture that represents Mediterranean agroenvironmental conditions, over the last hundred years. During the twentieth century, Spanish agriculture experienced a strong intensification process based in the use of external inputs, to a greater extent than other Mediterranean countries. This circumstance makes Spanish agriculture an optimal case study, as it provides diachronic scenarios with very different land use intensities, within these agroenvironmental conditions. This issue has been raised by various authors when evaluating the state of fund elements and the ecosystem services of agroecosystems (Berlin and Uhlin 2004; Tuomisto et al. 2012). The history of this process can be divided into four different periods. The first corresponds to the first third of the twentieth century, when agrarian production grew by 52%, in terms of fresh matter. In this period, Spanish agriculture started an incipient process of integration in the international markets, but the process ended abruptly with the civil war (1936–1939). In the second period (1936–1960), civil war and autarky policy of Franco regime stopped short such trends and the country got into an international isolation in terms of trade. International isolation and the ill-advised economic policy of the regime led to a reduction in agrarian production, which would only be resolved after the 1960s, once the "green revolution" had begun. In the third period (1960–1986) yields multiplied thanks to the use of the complete package of the green revolution. During the fourth 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 those products with a higher demand in the European Union (olive oil, fruits and vegetables). In parallel, the less productive lands were abandoned (generally grain cropland devoted to feed use, and pastureland), while high-protein feed imports skyrocketed (Soto et al. 2016).

Data collection, concepts and methods

Data collection

The main sources used in our study are the statistics provided by the Spanish government, with different quality and frequency from 1900 to 2008, to maintain coherence over the entire period, since FAOSTAT does not provide data between 1900 and 1960 (FAO 2015). We have reconstructed the evolution of total biomass production in all Spanish land areas (excluding unproductive areas which remained practically constant throughout the period studied; see "Supplementary data") and total inputs consumed at twelve points over time between 1900 and 2008, using 5-year averages to buffer year-to-year variability.

The reconstruction of biomass production and the sources employed are described in detail in Soto et al. (2016). In short, we employed yields, crop areas and livestock production data from statistical sources (e.g. GEHR 1991; Carreras and Tafunell 2005). Annual data for total Spanish agricultural production are available from 1929 onwards.[1]

We then calculated the amounts of agricultural residues using converters (Guzmán et al. 2014).

Based on land uses reconstructed using the sources cited previously, the production of pastureland and fallow land was calculated. The production of timber and wood was estimated as described in Infante-Amate et al. (2014).

The exports and imports of biomass were calculated from foreign trade sources. Between 2000 and 2008, we used the DATACOMEX database of Spanish overseas trade (MINECO 2015a). For 1960 and 1990, we used the FAOSTAT database (FAO 2015). Lastly, for the period from 1900 to 1950, we used overseas trade statistics for Spain.[2]

The amounts of external inputs employed in Spanish agriculture during the period studied were mainly gathered directly from official statistics (see footnote 1) complemented by technical reports and research studies, and assuming constant growth rates for the missing years. Data from the Spanish Agrarian yearbooks include fertilizers from 1933 onwards, tractors and other farm machinery from 1955 onwards, fuels in 1960, pesticides from 1933 onwards, and greenhouses, tunnels and mulched surface

[1] The Spanish Agrarian Yearbooks are available online (MAGRAMA 2015b).

[2] The original Trade Yearbooks are available online (MINECO 2015b).

areas from 1975 onwards. Pesticide data from 1950 to 1980 were expressed in the statistics on a monetary basis, and we converted these to weightings using deflation data from Carreras and Tafunell (2005). Fertilizers in 1900–1922 were estimated from the data compiled by Gallego Martínez (1986) and Mateu Tortosa (2013). Fuel consumption data in 1950 and 1990–2008 were taken from the statistics (MI 1961, MINETUR 2015) and in 1970–1980 from FAO (2015). Fuel consumption from 1900 to 1940 was estimated based on the installed capacity of the machinery. Electricity consumption data in 1950 were taken from INE (1960), in 1960 from Carpintero and Naredo (2006), in 1970–1980 from FAOSTAT (FAO 2015) and from 1990 onwards from MINETUR (2015). Electricity consumption before 1950 was estimated assuming that agricultural electricity represented the same share of total Spanish electricity consumption as in 1950. Data from Corominas (2010) were used to take into account upstream electricity consumption in irrigation. Surface areas represented by each type of irrigation were taken from MAGRAMA (2015a). Official machinery data in the first half of the twentieth century were complemented with data from Martínez Ruiz (2000). We considered 97% of greenhouses to be of "Almeria vineyard type" and 3% to be of "Glass greenhouse type" (MAGRAMA 2008).

Concepts and methods

Concepts and components of Net primary productivity

Net primary productivity (NPP) is the amount of energy actually incorporated into plant tissues as the result of the opposed processes of photosynthesis and respiration. NPP may refer to the net productivity of an ecosystem that would be in place in the absence of humans (potential NPP, *NPPpot*) or to the net productivity that actually remains in an existing ecosystem (actual NPP, *NPPact*) (Haberl et al. 2007). Most estimates of NPP only consider aerial biomass, which has drawn criticism (Smil 2013) due to the relevant role of root biomass in the maintenance of complex soil food chains and in soil organic matter dynamics (Kätterer et al. 2011). Therefore, we have taken root biomass into account in this study.

Agricultural statistics usually focus on the harvested portion of NPPact. Therefore, the remaining components have to be estimated, for which there are different approaches. In this paper, we have applied three of them: (1) using algorithms, which take into account variations in vegetation and soil and climatic conditions; (2) using conversion factors, which allow NPPact to be estimated from harvested biomass and which take into account the changes occurring over the period studied; and (3) extrapolating from other studies with different agroclimatic and management contexts.

NPPact of agroecosystems can be broken down into the following portions, according to their fate (Guzmán and González de Molina 2015):

Socialized Vegetable Biomass (SVB): this is the phytomass that is directly appropriated by human society, considered as it is extracted from the agroecosystem, prior to its industrial processing.

In the same way, *Socialized Animal Biomass* (SAB) is the animal biomass (live weight of meat at the farm-gate, milk, wool…) that is appropriated directly by society. Obviously, SAB is not part of NPP. The sum of SVB and SAB gives the *Socialized Biomass*, which is the total biomass appropriated by society (SB).

The concept of SB does not imply the existence of an economic exchange in monetary terms. That is to say, SB includes all of the biomass (food, fibre, timber, firewood, etc.) that is self-consumed or exchanged by barter. There may also be biomass outputs from an agroecosystem involving monetary exchange but not considered SB. This would be the case of biomass that leaves the agroecosystem but which is not destined for society, but rather to sustain the functions of another agroecosystem. For example, hay sold as feed for the livestock of another producer, the sale of working animals, etc.

Recycling Biomass (*RcB*): this is the phytomass that is reincorporated into the agroecosystem, including seeds and vegetative reproduction organs and the phytomass recycled through livestock farming or through wild heterotrophic organisms. From society's perspective, the *RcB* can be divided into two portions:

Reused Biomass (*RuB*): this is the portion that is intentionally returned to the agroecosystem by farmer. This means that the phytomass is reincorporated into the agroecosystem by means of human labour and has a agronomic purpose that is recognized by farmer, for example, in order to obtain a product or a service (animal feed for the supply of meat or milk). This category includes the biomass that is destroyed by fire (for example, stubble burning) since it involves conscious work and has an agronomic purpose;

Un-harvested Biomass (*UhB*): this is the phytomass that is returned 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 Un-harvested Biomass* (*AUhB*) and *Belowground Un-harvested Biomass* (*BUhB*).

Accumulated Biomass (*AB*): this refers to the portion of phytomass that accumulates annually in the aerial structure (trunk and crown) and in the roots of perennial species.

All of these portions of the NPPact have been taken into account. The calculation method used for cropland, grassland and woodland is detailed below.

Figure 1 outlines the main biomass flows considered.

Calculation of NPPact

Cropland NPPact Cropland NPPact is the biomass of crops and also of associated weeds. To calculate the aerial biomass of weeds in traditional agricultural systems, we have used data from contemporary Organic Farming trials and for more recent periods, data from conventional agriculture (Guzmán et al. 2014).

The NPPact of crops is obtained using conversion factors, which allow it to be calculated from crop production (*SVB*), which is the most commonly available data in historical sources. The conversion factors were obtained from an extensive literature review (Guzmán et al. 2014). They include harvest indexes and root/shoot ratios to calculate the total biomass (aboveground + belowground biomass), dry matter coefficients to convert the fresh biomass into dry biomass, and energy coefficients to convert the biomass into gross energy. For some coefficients, like the harvest index, we can expect changes over time in some crops. In those cases, we have also provided information for pre-industrial time periods in some crops (cereals).

Grassland NPPact Grassland NPP has been collated from studies conducted in Spain and based on different grassland types and climatic conditions (CIFA 2007; Correal et al. 2007; Hernández Díaz-Ambrona et al. 2008; Robles 2008; San Miguel 2009). The productivity of root biomass was calculated using conversion factors (Guzmán et al. 2014).

Woodland NPPact Wood production (fuel wood and timber for society + aboveground wood accumulated on trees) calculations are described in Infante-Amate et al. (2014). By applying partition coefficients, we calculated the remaining components of the NPP, including the annual production of leaf biomass and reproductive structures, along with the root biomass. The proportion of root biomass that is accumulated and recycled every year in the soil was calculated taking into account the root/shoot ratio of the adult holm oak (0.84) and pine (0.3). Basic data regarding these transformations can be seen in Almoguera Millán (2007) and CMAOT (2014). The conversion factors for biomass into gross energy can be found in Guzmán et al. (2014).

Calculation of external inputs (EI)

EI include human labour, as well as all of the inputs (fertilizer, pesticides, machinery, feed…) that originate outside the agroecosystem. They can be divided into industrial inputs (chemical fertilizers, machinery, etc.) and non-industrial inputs (biomass, human labour, etc.). The allocation of energy to each type of input is summarized below.

Industrial inputs In this study, the energy allocated to industrial inputs is embodied energy. In other words, the sum of the gross energy of the input plus the energy requirements for production and delivery of the input. The

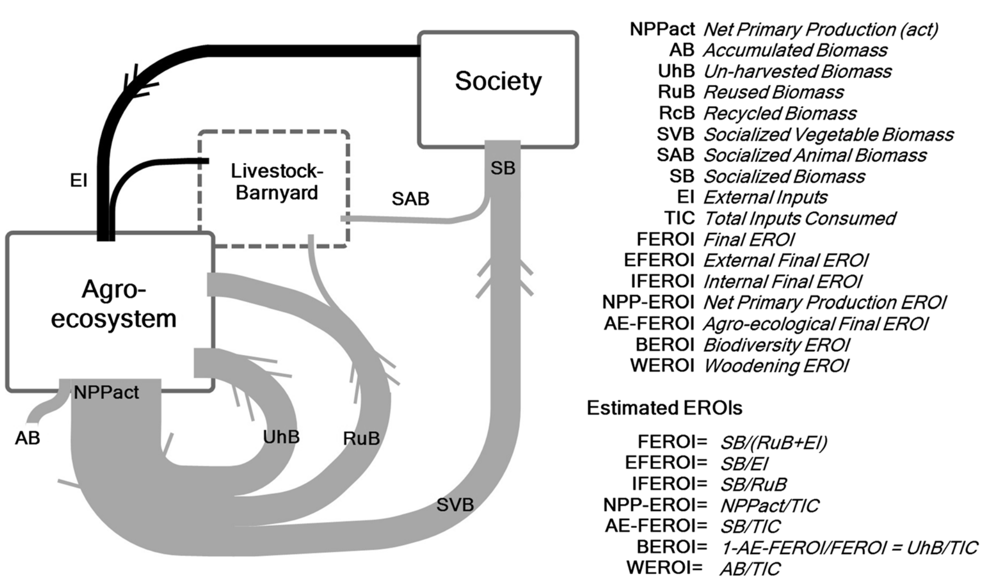

Fig. 1 Schematic representation of the energy flows and EROIs considered

embodied energy of industrial inputs evolved over time, as the energy efficiency of the production and delivery of the inputs changed. Therefore, in order to be rigorous when calculating EROIs from a historical perspective, we have drafted a working paper that develops embodied energy (and its components) in agricultural inputs for the period 1900–2010 (Aguilera et al. 2015), where we also include theoretical and methodological considerations. As the data in Aguilera et al. (2015) are shown on a decadal basis, and the time points do not always match ours, we estimated the missing values through linear interpolation. To estimate the energy embodied in machinery, we took into account the installed power (MW) of the machinery, the years of manufacture of the machinery mix and the actual replacement rate in the studied year (estimated based on yearly census registrations and removals). The energy embodied in electricity was estimated taking into account the Spanish electricity mix and the efficiency of Spanish thermal power plants (Bartolomé-Rodríguez 2007; UNESA 2005; MINETUR 2016; REE 2011), complemented with fuel embodied energy data from Aguilera et al. (2015).

Non-industrial inputs We estimated the energy in human labour as dietary energy consumption (2.2 MJ/h) (Fluck 1992). This method for accounting for energy in human labour does not include the energy required to produce the food consumed by the labour (embodied energy) (see a discussion in Aguilera et al. 2015). This avoids a problem of circular referencing or double counting, which can stem from this method, since the product (food) is used as an (important) input of the system.

The energy in the net imported biomass (such as seeds and feed) is the gross energy of the different products, calculated using conversion factors (Guzmán et al. 2014). The cost of transport was added to this (Aguilera et al. 2015). The energy required to produce the biomass was not considered, to avoid problems of double counting, since this cost should be attributed to the agroecosystems of origin.

Calculation of the EROIs

Proposed EROIs from an economic perspective Economic EROIs inform us of the return on energy intentionally invested by society in agroecosystems. The proposed EROIs are (Tello et al. 2016; Guzmán and González de Molina 2015):

$$\text{Final EROI} = \text{SB}/(\text{RuB} + \text{EI}),$$

where $\text{SB} = \text{SVB} + \text{SAB}$Final EROI (FEROI) tells us about the return on the energy investment made by society. It can be broken down into two elements: External Final EROI (EFEROI) and Internal Final EROI (IFEROI):

$$\text{EFEROI} = \text{SB}/\text{EI}$$

EFEROI relates *EI* to the final output crossing the agroecosystem boundaries

$$\text{IFEROI} = \text{SB}/\text{RuB}$$

IFEROI refers to the efficiency with which intentionally recycled biomass is transformed into a product that is useful to society.

Proposed EROI from an agroecological perspective Agroecological EROIs inform us of the actual productivity of the agroecosystem, not just the portion that is socialized. Furthermore, they inform us of the reinvestment made in the fund elements, that is, in the structure of the agroecosystem that provides basic ecosystem services. We have calculated four EROIs from an agroecological perspective (modified from Guzmán and González de Molina 2015):

$$\text{NPPact EROI} = \text{NPPact}/\text{Total inputs consumed},$$

Total inputs consumed *(TIC)* being $= \text{RcB} + \text{EI} = \text{RuB} + \text{UhB} + \text{EI}$NPPact EROI refers to the actual productive capacity of the agroecosystem, whatever the origin of the energy it receives (solar for the biomass or fossil for an important portion of the *EI*).

$$\text{Agroecological Final EROI (AE} - \text{FEROI)} = \text{SB}/\text{TIC}$$

This EROI gives a more exact idea of the total energy required to obtain SB. From an agroecological point of view, the relationship between this indicator and the FEROI is of great interest.

$$\text{Biodiversity EROI} = 1 - \frac{\text{AE} - \text{FEROI}}{\text{FEROI}} = \text{UhB}/\text{TIC}$$

It can reach a minimum of 0, when all of the recycled biomass is reused, indicating agroecosystems with very significant human intervention, and a maximum value of 1 when there are no external inputs and no biomass is reused by society. This would be true of natural ecosystems without human intervention.

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; Rundlöf et al. 2008, Gabriel et al. 2013). From this perspective, we consider that Biodiversity EROI provides useful information on the extent to which energy invested in the agroecosystem contributes to sustaining food chains of heterotrophic species.

Biodiversity EROI also allows us to explore the hypothesis of land sparing versus land sharing from the perspective of energy, since it links the productivity of the system with the biomass available for wild heterotrophic species. The availability of phytomass is necessary to sustain complex food chains of heterotrophic species, but on its own it is not sufficient. Other factors, such as the absence of biocides and the presence of a diverse territorial matrix, are also pillars that sustain biodiversity in agroecosystems. The absence of biocides is an inherent characteristic of traditional agriculture. Likewise, other research shows that traditional agriculture generated complex territorial organizational matrices which sustained high biodiversity levels (Gliessmann 1998; Guzmán and González de Molina 2009; Perfecto and Vandermeer 2010; Marull et al. 2015). It therefore remains for us to ascertain whether or not these types of agriculture are able to free up greater proportions of phytomass than industrialized agriculture, and this is what the Biodiversity EROI allows us to do. Some authors state that the intensification of agricultural production, through external energy inputs, will free up land (land sparing) for the recuperation of wild biodiversity (Phalan et al. 2011). This theory has been discussed by other authors such as Perfecto and Vandermeer (2010), Phelps et al. (2013), and Tscharntke et al. (2012). Land sparing for biodiversity can have several meanings. It can be understood as the liberation of phytomass, as proposed with the Biodiversity EROI indicator, but it could also be understood as the liberation of physical space, for example, through the conversion of cropland or pastureland to woodland. The application of the following EROI (Woodening EROI) allows us to look in greater depth at the latter aspect.

Woodening EROI informs us whether the energy added to the system is contributing to the storing of energy in the system as AB. AB can be considered a fund element related to the ecosystem services provided by forests, but not only by them. Biomass can also be accumulated in cropland or grassland (hedgerows, shade trees), providing ecosystem services for agrarian activity. It is also accumulated in living tissues of woody crops, growing when there is an expansion of these crops. In all cases, biomass accumulation contributes to carbon sequestration. This is a novel type of EROI, not shown in Guzmán and González de Molina (2015).

$$\text{Woodening EROI} = \text{AB}/\text{TIC}$$

Results and discussion

Evolution of NPPact and external inputs

Figure 2 shows that the industrialization process of Spanish agriculture yielded an uneven growth pattern among the different components of NPPact and EI. NPPact increased by only 29% between 1900 and 2008. This growth is the result of very divergent, even opposed, behaviours of their components: SVB from cropland doubled, while that from forestland decreased by 30% due to the replacement of firewood with fossil fuels, mainly from 1960 onwards, when the industrialization process became more evident. RuB grew 70% during the period studied. Its evolution can be divided in two stages: from 1900 to 1960 it grew, linked to growth in the number of draught animals employed, which had not yet been replaced by self-propelled machinery. From 1960 to 1970, it dropped sharply due to the rapid replacement of animals with machines, and from 1970 onwards it grew again due to the continued growth of the livestock herd (Soto et al. 2016). Livestock numbers rocketed as of 1960, but the composition also drastically changed. Numbers of species that were best suited to Mediterranean pastureland, such as sheep and goats, did not increase. Equine species, used fundamentally as working animals, practically disappeared. However,

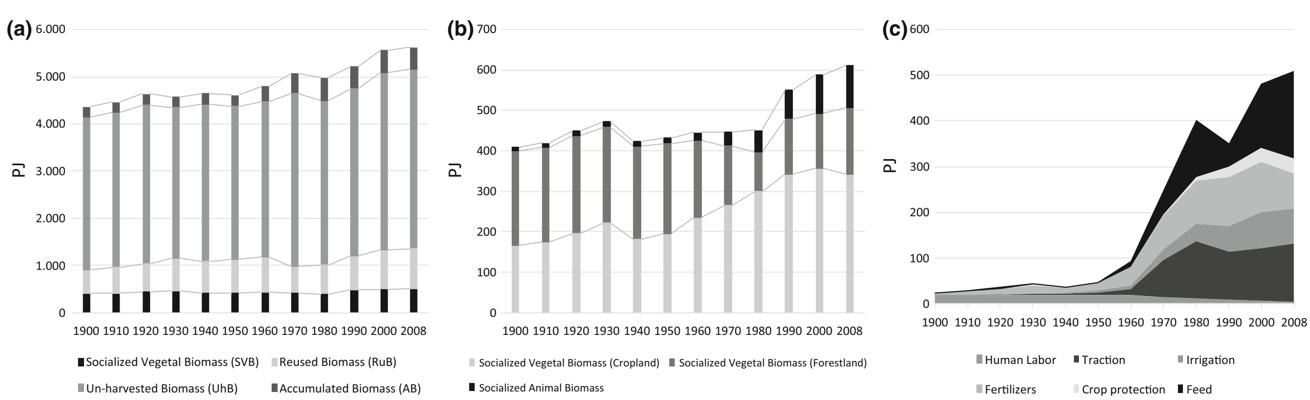

Fig. 2 Evolution of NPPact (**a**), socialized biomass (**b**) and external inputs (**c**) (PJ)

monogastric animals (pigs and fowl) and cattle, particularly dairy cattle, grew exponentially. These species are largely fed using commercial feed and fodder, a large proportion of which is imported. These changes have been accompanied by a profound modification in patterns of food consumption within Spain in recent decades. The Mediterranean diet has been replaced by an animal-based diet, with meat and dairy becoming increasingly central (Alexandratos 2006; Infante-Amate and González de Molina 2013; Lassaletta et al. 2014).

Finally, UhB grew by just 17%, while AB doubled, as a result of the lower pressure on forestland starting in 1960, and of the reduction in pastureland area (−23%) and cropland area in recent times (−17% since 1970), which led to 33% growth in forestland areas from 1900 to 2008. More details about the evolution of land uses, NPPact and livestock are provided in Soto et al. (2016).

EI increased 20-fold. The use of synthetic chemical fertilizers increased substantially, although their participation in the total *EI* was relatively low (15%) in 2008 (Fig. 2c). This modest percentage must be linked with a phenomenon inherent to semi-arid agroecosystems typical of the Mediterranean: the lack of rainfall means that the application of more fertilizer is of limited utility in terms of increasing NPPact due to the absence of optimum hydric conditions. In energy terms, the introduction of mechanical technologies has played a greater role, now accounting for 25% of *EI*. However, the importing of animal feed saw the biggest growth, now representing 37% of *EI* (Fig. 2c). As we will see, this fact, together with the growing importance of RuB, has important implications for energy efficiency in Spanish agroecosystems.

EROIs of Spanish agriculture from an economic perspective

As we have seen, in the last half century Spanish society has invested a considerable amount of energy to obtain a supply of biomass destined to feed an increasingly large animal component. *SVB* from cropland has doubled in energy terms, but *SAB* has by far increased the most: over 11-fold (Fig. 2b). In other words, the rise in productivity achieved by Spanish agroecosystems between 1960 and 2008 was largely invested in producing food for livestock. The well-known inefficiency of converting plant biomass into animal biomass has been transferred to the whole of Spain's agrarian sector, as shown by the proposed economic EROIs.

FEROI has fallen significantly (over 40%) (Fig. 3a). In terms of the energy invested by society, traditional organic agriculture was more efficient than industrial agriculture. If we look at the temporal evolution of this indicator, we see a major decline in FEROI between 1900 and 1970, and stabilization in the last 40 years. This stabilization is due to complementary factors such as continued increases in the efficiency of external input manufacture (Aguilera et al. 2015); the partial outsourcing of livestock maintenance costs to foreign agroecosystems; the increase in irrigated land area (68% between 1970 and 2008), a key factor in increasing phytomass growth in semi-arid areas typical of Spain; and the continued expansion of the olive grove, which generates two products (wood and oil) of high calorific value.

As expected, EFEROI has dropped even more dramatically, from a production of 17.3 joules of SB for every joule invested from outside the sector, to 9.2 in 1950 and 1.2 in 2008 (Fig. 3a). The key to increasing yield per land area unit and increasing agrarian production has been the massive increase in energy incorporated into production (Smil 2013), coming from fossil fuels and biomass. The important depressor effect on EFEROI brought about by importing external biomass inputs into agroecosystems has been demonstrated in other case studies (González de Molina and Guzmán 2006; Guzmán and Alonso 2008). The behaviour of FEROI and EFEROI has borne out the findings of two other case studies conducted at a local level in

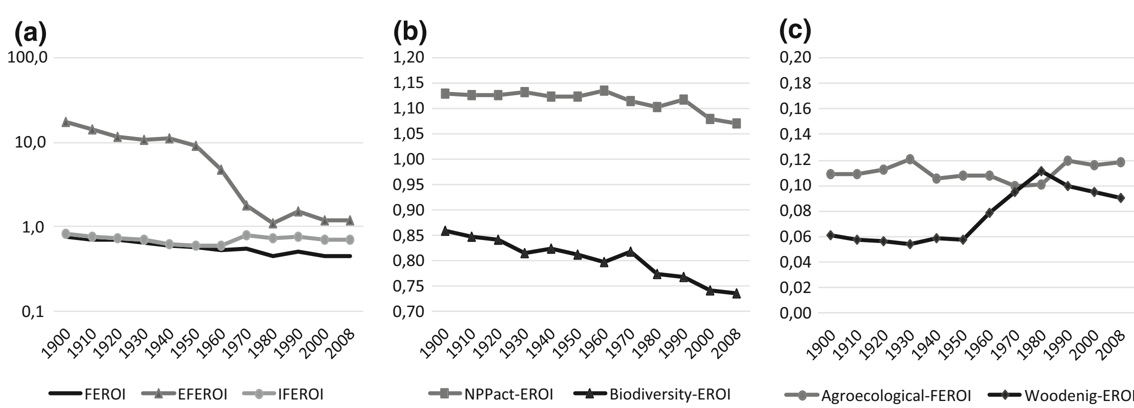

Fig. 3 Evolution of economic and agroecological EROIs of Spanish agriculture

the north-east (Vallès County, c.1860 and 1999) and south-east (municipality of Santa Fe, c. 1904 and 1997) of Spain (respectively, Tello et al. 2016; Guzmán and González de Molina 2015).

Ostensibly, traditional organic agriculture had to invest a huge amount of biomass in order to make its own reproduction possible, but in the case of Spain, IFEROI is even lower for industrial agriculture than for traditional organic agriculture, having fallen by 12% since 1900. This latter phenomenon is to some extent unexpected. Traditional organic agriculture, given its high land cost to replenish fertility and produce the energy required for the production process (e.g. to feed working animals), is assumed to be more inefficient than industrial agriculture in terms of the investment of internal energy (Guzmán and González de Molina 2009). In theory, the availability of external inputs should save on the amount of land required for production, or should decrease the investment of *RuB* (Guzmán et al. 2011). As a consequence, IFEROI and even FEROI should increase. In fact, the increase in IFEROI throughout the twentieth century is reported in the aforementioned case studies (Guzmán and González de Molina 2015; Tello et al. 2016), where the opposite process was observed, in other words, a process of agriculturalization. However, in the case of Spain, this has not occurred. This is due to the fact that the increase in productivity achieved in recent decades has largely been invested in feeding livestock.

EROIs of Spanish agriculture from an agroecological perspective

However, the loss of efficiency in industrial agriculture with regard to traditional Spanish organic agriculture might have additional causes. Agroecological EROIs allow us to detect whether the degradation of fund elements is undermining the productivity of agroecosystems. NPPact EROI remained steady up until the 1960s. However, after that point it fell by 6%, coinciding with the industrialization of Spanish agriculture (Fig. 3b). This decline occurred in spite of the injection of energy and water received and more efficient manufacturing of industrial inputs. In a semi-arid country such as Spain, the 82% increase in irrigated land area between 1960 and 2008, combined with the growth of received energy (24.7% more between 1960 and 2008) should have had the opposite effect. However, high rates of erosion (Gómez and Giráldez 2008; Vanwalleghem et al. 2011), the decrease in organic soil matter (Romanyà et al. 2007; Rodríguez Martín et al. 2016), salinization and the overexploitation of water resources (European Commission 2013), and the loss of agrarian biodiversity (Garrido 2012; MAPA 1995) are responsible for this decline. Ultimately, the deterioration of fund elements (soil, water, biodiversity), caused by industrial agriculture itself, is taking its toll.

Biodiversity EROI decreased by 15%, indicating a decrease in UhB in relation to TIC, which entails a lower level of relative energy availability for wild heterotroph organisms, particularly on cropland (Fig. 4), where a major decline was observed for UhB, both below and above ground. The fall in BUhB would help to explain why half of the agricultural land in Spain currently has an organic carbon content of less than 1% (Rodríguez Martín et al. 2009, 2016). The drop of AUhB on cropland is in line with the relationship between the changes undergone by Spanish agriculture in the twentieth century and the declining state of biodiversity in the country (MAPAMA 2011).

In other words, the shift towards feed production, a fundamental component of RuB, has a negative impact on biodiversity. This effect would not be compensated by the abandonment of Pastureland and Woodland (Fig. 4), questioning the strategy of land sparing. The disassociation of the agroecosystem in areas of intensive production and abandoned and/or protected areas (e.g. 40% of Spain's total forest areas are protected, according to MAGRAMA, 2014) has not brought about a significant increase in the trophic energy available for transfer from plants to other levels in the food webs.

This argument adds to those put forward in other research, showing that the intensification of agriculture has led to biodiversity losses owing to the loss of ecological heterogeneity at multiple spatial and temporal levels (Benton et al. 2003; Firbank et al. 2008; Guzmán and González de Molina 2009; Lindborg and Eriksson 2004; Marull et al. 2015; Vos and Meekes 1999). Furthermore, it supports the strategy of land sharing, at least in countries where traditional agriculture has played a major role in shaping the landscape (Barral et al. 2015; Ramankutty and Rhemtulla 2012; Wehrden et al. 2014).

AE-FEROI has grown by 9% (Fig. 3c). This increase might explain the fairly widespread notion that Spanish agriculture significantly increased productivity over the course of the twentieth Century. However, as shown here, in reality total productivity did not grow; rather, growth was achieved in the part of production appropriated by society in relation to TIC. This indicator can be broken down into three different components. We have analysed two of them (EFEROI and IFEROI) above, showing a drop in both of them. Therefore, in order to increase AE-FEROI, the third component must increase (SB/UhB). Indeed, this relationship increased by 27% from 1900 to 2008. Therefore, this improvement was due to the fact that industrialized farming has significantly reduced the un-harvested part of NPPact.

Finally, Woodening EROI remained stable up to 1950. It then grew by 93% from 1950 to 1980, surpassing even the output for society (Fig. 3c), and from 1980 to 2008 it decreased by 19%. Hence, in energy terms, there has been strong growth in TIC, far higher than the growth in AB since 1980. At first glance, this reforestation process would

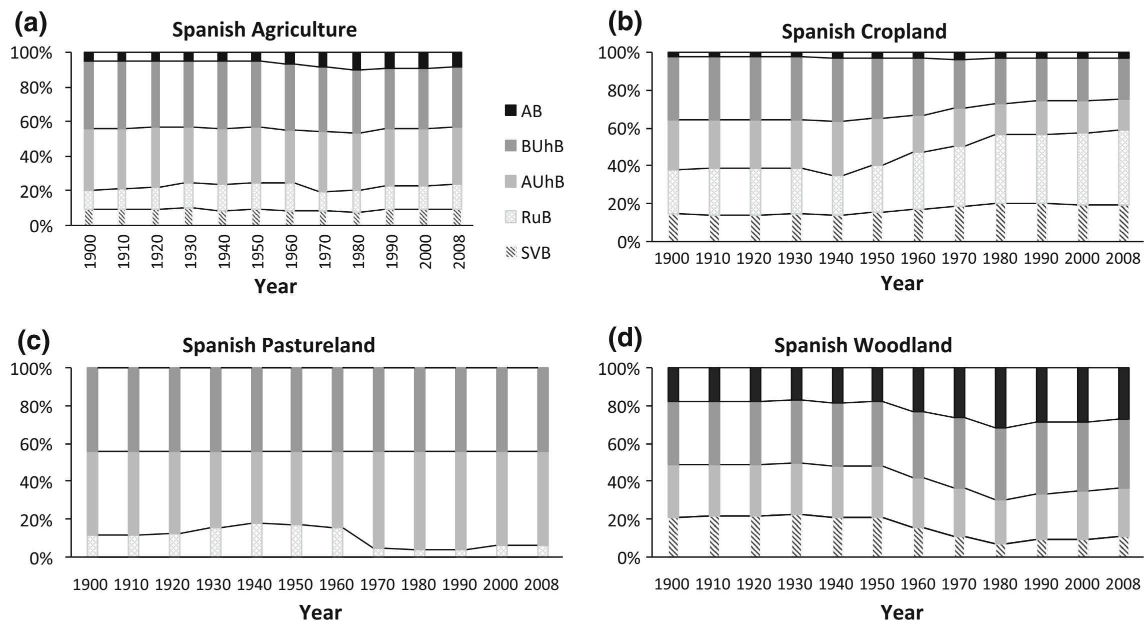

Fig. 4 Evolution of NPPact (TJ) by its use in relative terms in **a** Spanish Agriculture, **b** Spanish Cropland, **c** Spanish Pastureland and **d** Spanish Woodland. *Note*: *SVB* socialized vegetable biomass, *RuB* reused biomass, *AUhB* aboveground un-harvested biomass, *BUhB* belowground un-harvested biomass, *AB* accumulated biomass

support a *land sparing* strategy, as agricultural modernization would have allowed for growth in areas of forestland. However, these Woodening EROI values require contextualization. AB increased from 229,401 TJ in 1900 to 466,480 TJ in 2008. Of this growth (237,079 TJ), approximately 10% corresponded to the expansion of woody crops, mainly olive groves (23,079 TJ). An additional 30% was due to the substitution of firewood by fossil fuels. The remaining 60% of AB growth was due to the growth of forestland in areas freed from agricultural activities. On the other hand, the growth in forest area was due to the abandonment of land uses (pastureland and rainfed land) devoted to producing animal feed, which were massively substituted by RuB from intensively managed cropland (with its effect on Biodiversity EROI described above) and by feed imports (Fig. 2c), mainly from Latin America (Soto et al. 2016). Moreover, it is likely that the partial outsourcing of the land cost of Spanish livestock production to third countries might have caused major deforestation there, a process that we have not studied in this paper. In fact, estimated GHG emissions from deforestation (LULUC emissions) caused by Spanish feed imports in 2004 ranged between 20 and 64 Tg CO_2-eq in three different scenarios, which can be compared to an estimated emission from the Spanish livestock sector (excluding LULUC emissions) of 48 Tg CO_2-eq (Leip et al. 2010). In this regard, some studies show a close relationship between soy production and deforestation in Latin America (Gasparri et al. 2013). A large proportion of the soy imported by Spain for animal feed is grown in this region (Lassaletta et al. 2014). Deducting the *dis-accumulated* biomass in other agroecosystems would probably yield negative results, but this issue should be addressed by further research.

Agroecological EROIs (NPPact EROI, AE-FEROI and Biodiversity EROI) have only been previously applied to the case of Santa Fe (Guzmán and González de Molina 2015). In this municipality, the trend of these EROIs is the same as we found here for Spain, with the exception of NPPact EROI. In Santa Fe, NPPact EROI grew between 1904 and 1997, due to the continued increase in water consumption. Water is a peculiar input in terms of energy, because its gross energy content is "0" and, therefore, it does not have a direct repercussion on EI. It only has an indirect impact, as EI encompasses the energy costs of the impulsion and infrastructure required for irrigation. However, in semi-arid climates, the availability of water is essential to produce biomass. Therefore, the large proportion of the territory devoted in 1997 to a high biomass-yielding crop (irrigated black poplar) was compensating for the decline in NPPact EROI in the rest of the territory (Guzmán and González de Molina 2015).

Conclusions

The industrialization of Spanish agriculture allowed the biomass allocated to society (SB) to grow by 49% in the period studied, especially livestock biomass (1034%). On the one hand, this growth was based on the injection of large quantities of external energy in the form of fossil inputs and biomass from 1960 onwards. Animal feed imports have been essential to sustain a model of intensive livestock farming that is decoupled from the territory, leading to the abandonment of pastureland and rainfed land. On the other hand, the growth in SB is a result of the increase in the proportion of NPPact appropriated by society. In fact, the proportion represented by RuB has increased from 11 to 15% of NPPact, mainly destined to animal feed.

Both processes—the shift towards livestock production and the dramatic increase in industrial inputs—are the causes of the drop in economic EROIs of Spanish agriculture. From a social point of view, the return was highest in the early twentieth century, when considering the total energy invested by society (FEROI fell from 0.78 in 1900 to 0.45 in 2008) or the external or internal inputs separately (EFEROI fell from 17.3 to 1.2 and IFEROI from 0.82 to 0.72 in the same period). In short, Spanish society has obtained decreasing returns on the energy invested throughout the process of agrarian industrialization.

The application of the proposed agroecological EROIs to the case study has informed us of processes which affect the fund elements of agroecosystems and their capacity to generate flows of ecosystem services. The 6% drop in NPPact EROI from 1960 to 2008, when the industrialization process was taking place, suggests that the fund elements are being degraded. This drop takes place although the irrigated land area (a key factor for the increase in NPP in semi-arid areas) is almost doubling, and external energy is being injected on a massive scale. In fact, agroecological EROIs tell us of other key processes that undermine the sustainability of the agroecosystem. The Biodiversity EROI ratio alerts us to the low return to nature in the form of UhB available to aboveground and underground wildlife, especially on cropland. This low return on cropland is not compensated by the abandonment of pastureland and forestland, questioning the hypothesis of *land sparing* for the purpose of sustaining biodiversity, rather than *land sharing*. Furthermore, cropland soil suffered a drastic reduction of biomass inputs, entailing negative impacts on soil quality (Romanyà et al. 2007; Rodríguez Martín et al. 2009).

Finally, the growth in forestland area and the loss of forest functionality have allowed a certain increase of accumulated biomass in this space (48%). However, analysed globally, Woodening EROI shows that this structural improvement in the agroecosystem through reforestation was due, to a large extent, to biomass imports from third countries, where it was probably contributing to the increase in deforestation. This phenomenon should be considered in further analyses.

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