

Chapter 5

Environmental Impacts of Spanish Agriculture's Industrialization



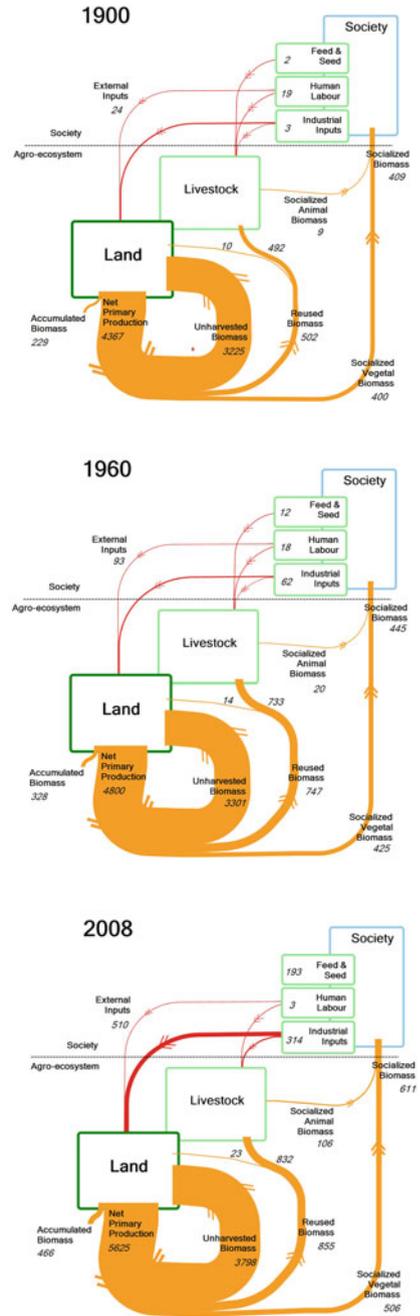
In this chapter, we focus on the changes to the functioning and structure of Spain's agroecosystem during the industrialization process in the twentieth century. Specifically, we aim at showing how changes in the quantity and quality of energy and material flows contributed to deteriorating the quality of the land fund element, that includes soil, biodiversity, water, etc., and that supports the provision of agroecosystemic services, among which biomass production. The degradation of the agroecosystem's biophysical structure is reflected in the progressive decline of energy returns in the form of biomass. We will also show how Spanish agricultural production had direct environmental impacts in remote regions of the planet due to the outsourcing of the land cost of food through massive imports of biomass from these regions. The methodology used for calculations is not described in this book but has been published in the following texts: García Ruiz et al. (2012), Soto et al. (2016), Guzmán and González de Molina (2017), Guzmán et al. (2017), Aguilera et al. (2018), Guzmán et al. (2018).

5.1 Functioning of the Agroecosystem

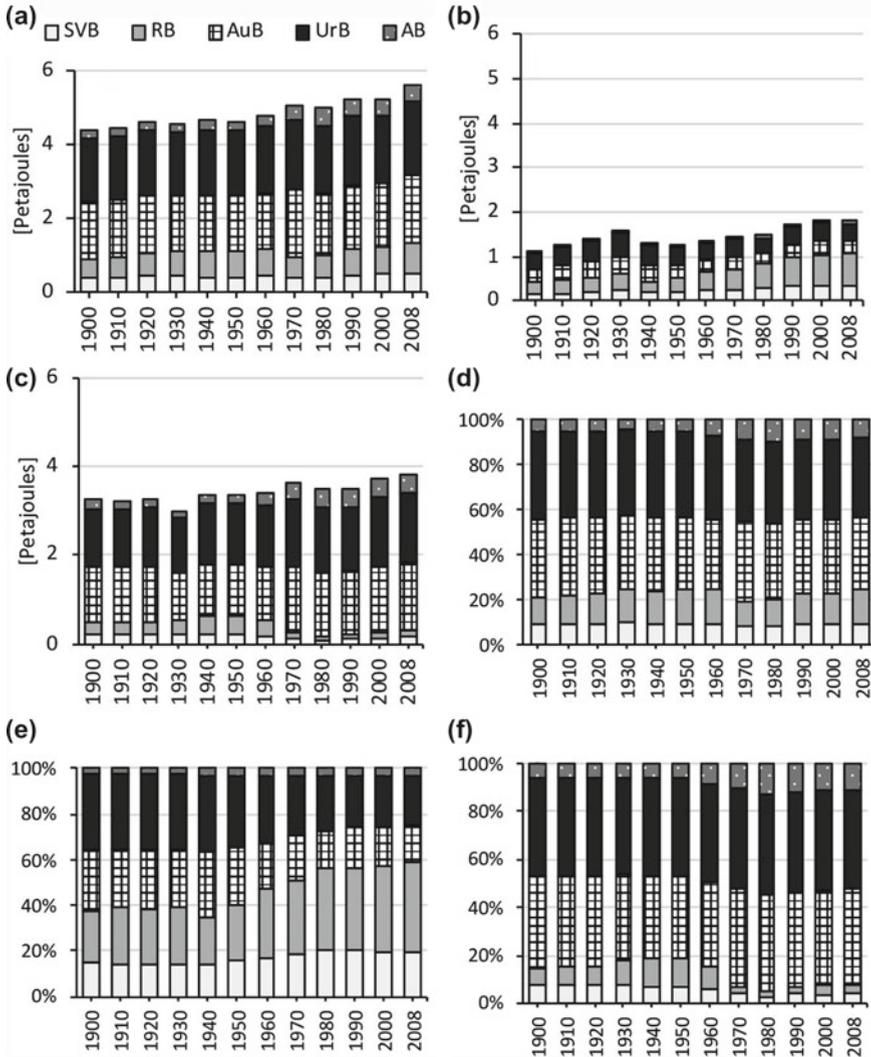
Chapter 3 showed how the intensification of Spanish agriculture was based on the increasing use of external inputs. The external energy invested in the agroecosystem over the 1900–1960 and 1960–2008 periods multiplied, respectively, by a factor of 3.9 and 5.5 (Graph 5.1 and Annex 3).

First, the increase of external inputs allowed to overcome limiting factors (e.g., nutrients, water) to some extent and ensure greater protection against heterotrophic organisms that translated into a higher NPP (Graph 5.1). However, this growth has been limited (10% from 1900 to 1960, and only 18% between 1960 and 2008), implying, as we shall see, a negative return on external energy invested to achieve this increase.

Graph 5.1 Energy flows in Spain's agroecosystem in 1900, 1960 and 2008, in petajoules



Second, ever-increasing external inputs led to modifications in the NPP's social use pattern (Graph 5.2). These changes are important because as we explained in Chaps. 1 and 2, the fundamental mechanism underlying agroecosystems is their use of energy in the form of biomass that is recirculated and stored within them.



Graph 5.2 The evolution of NPP according to its uses in absolute terms: Spanish agroecosystem (a), croplands (b), silvopastoral areas (c); and in relative terms: Spanish agroecosystem (d), croplands (e), silvopastoral areas (f), petajoules. **Note** SVB Socialized Vegetal Biomass; RB Reused Biomass; AuB Aerial unharvested Biomass; UrB Unharvested root Biomass; AB Accumulated Biomass

Therefore, a balance is achieved between the energy that is extracted (Socialized Vegetal Biomass (SVB) + Reused Biomass (RB), with respect to the non-extracted energy (Unharvested Biomass (UB) + Accumulated Biomass (AB)) that must be respected to ensure its long-term functioning.

In terms of energy, the share of biomass extracted and not extracted from croplands remained relatively constant between 1900 and 1950. Extracted biomass rose from 38 to 40% and, non-extracted biomass dropped from 62 to 60% over the period (except for 1940 when extraction fell to 35% due to the Civil War) (Graph 5.2b and e). In pasturelands, the tendency was similar. Between 1900 and 1950, extracted biomass increased from 15% in the first decades to 18% in the 1930–50 period (Graph 5.2c and f).

As from the 1950s, massive incorporations of external energy reinforced the process of substitution of internal flows, though differently in croplands and silvopastoral areas. This process deteriorated, as we will show later, the quality of the agroecosystem land fund element. Between 1950 and 2008, changes in RB were driven by big increases in livestock, mainly in monogastric animals (pigs and poultry), and the shift from extensive to intensive management (see Chap. 2 for more details). This profound change in the composition and management of livestock would not have been possible without massive imports of feed (mainly soybean and corn), that is difficult to produce in Spain for agroclimatic and economic reasons. As a result, pastures were partially abandoned. Meanwhile, increasing amounts of high-quality biomass (grains and fodder) from croplands were dedicated to livestock. In these lands, RB went from 25% of NPP in 1950, to 40% in 2008 in terms of energy. Meanwhile, in silvopastoral areas, it fell from 12 to 4% (Graph 5.2e and f). These land-use imbalances (intensification of agricultural land use and abandonment of pastures) also occurred in the interior's croplands. Cultivation areas were reduced during this period mainly because of the abandonment of non-irrigated lands that responded poorly to external inputs. Changes in the social use pattern of biomass were also visible mainly in the 80s and 90s through increasing practices of burning of straw and other crop residues that was no longer used to feed livestock.

Conversely, a smaller share of biomass was abandoned on croplands. UB went from 57% of NPP to 38% (Graph 5.2e), a clear sign that soil fertility had become reliant on mineral fertilizers, to the detriment of organic matter. The use of herbicides also explains this relative fall. In silvopastoral areas, however, UB and AB increased by 6 and 5 percentage points, respectively, as a result of abandonment (Graph 5.2f).

As a result of the processes mentioned above, in terms of energy, Socialized biomass (SB) grew by 5.5% between 1900 and 1950, SVB accounted for an increase of 4.4% and Socialized Animal Biomass (SAB) an increase of 54.1%. Subsequently, SB growth accelerated, increasing by 41.5% between 1950 and 2008, mainly boosted by a SAB increase of 636%. The SVB grew barely by 21% over the period, as a result of a 75% increase in the crops SVB and a 26% drop in woodland SVB. The latter decline was due to the replacement of firewood by fossil fuels in households. The increase in cultivated land SVB (75%) was higher than that recorded for those lands' NPP (43%) (Guzmán et al. 2017). The increase in the harvest share of cereal varieties introduced by the Green Revolution contributed to the disparity, significantly

increasing grain yields without any major changes to total aerial biomass (Austin et al. 1980; Vita et al. 2007; Sánchez García et al. 2013). The changes in the plant biomass distribution pattern were driven by the loss of agricultural residues' functions of feeding livestock and replacing soil fertility, in turn, caused by external energy imports.

Third, external input imports made it possible to simplify rotations and replace legumes, which were no longer essential for incorporating nitrogen into the agroecosystem. From 1960 to 2000, the surface area devoted to legumes fell from 1.4 to 0.55 million hectares (Mha). Consequently, biological fixation N accounted for 28% of the entries in 1960, and only 11% in the year 2000 (Guzmán et al. 2018). To summarize, the intensification of agricultural land in Spain unfolded based on the substitution of internal energy loops (particularly through the relative decline of UB, and the marginalization of legumes), with external energy inputs, increasing the generated entropy and, as we will see later, degrading the land fund element and its components.

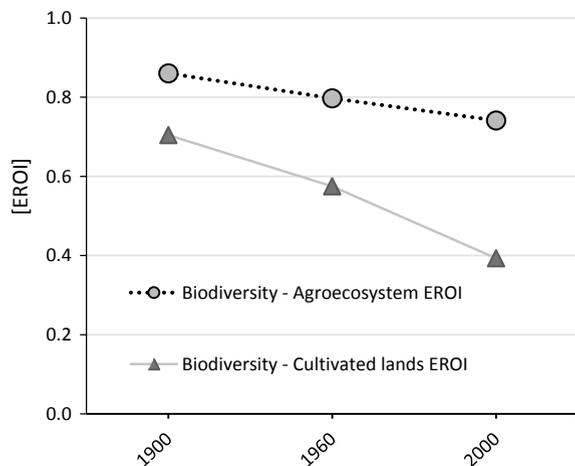
5.2 The Energy Efficiency of Agricultural Production

The total energy consumed to sustain the functioning of agroecosystems not only includes external flows but also the internal biomass flows that are reinvested in the agroecosystem (unharvested biomass plus reused biomass) (Guzmán and González de Molina 2015; Tello et al. 2016; Guzmán and González de Molina 2017). In terms of energy, the flows necessary for the functioning of Spain's agroecosystem rose from 3761 PJ in 1900 to 5163 PJ in 2008 (Graph 5.1). This represents a 37.3% increase in energy consumption. However, NPP only achieved an increase of 28.8% (Graph 5.2). This energy return was low despite the significantly higher amount of energy invested and also despite an increase in irrigated areas that rose from 0.8 Mha in 1900 to 3.3 Mha in 2008. Access to additional water flows is key to increasing plant production in semi-arid regions such as the Mediterranean. Therefore, it would be expected that the combined additional contributions of energy and water would have been synergistic and increased NPP further. However, this did not occur because the decoupling between internal energy and material flows led to the degradation of the land fund element, as we will see later, resulting in a negative return of the invested energy. We call this rate of return on energy investment NPP-EROI. The NPP-EROI (Net Primary Productivity-Energy Return on Investment) is part of the battery of agroecological indicators of energy efficiency (Agroecological EROIs) that we advanced in other studies to estimate the return of energy invested by society, in terms of biomass flows that support the agroecosystem's fund elements. In these texts, we defended that the indicator of energy efficiency $NPP-EROI = NPP / (RB + UB + \text{external inputs})$ systematically expresses the evolution of the quality of the agroecosystem's fund elements, to the extent that the agroecosystem's total biomass production is included in the numerator—not only the production share that is of interest to society—and all the energy consumed in the production process is present

in the denominator. The degradation processes affecting the fund elements (e.g. salinization, soil erosion, genetic erosion, etc.) must be compensated by the incorporation of increasing amounts of energy to alleviate the agroecosystems' loss of productive capacity. Therefore, the negative evolution of this indicator indicates the agroecosystem's degradation (Guzmán and González de Molina 2015; Guzmán and González de Molina 2017; Guzmán et al. 2017; Guzmán et al. 2018). In the case of Spain, it remained stable at 1.16 between 1900 and 1960, when it began to fall gradually reaching 1.09 in 2008 (Guzmán et al. 2017; Guzmán and González de Molina 2017).

Another interesting agroecological EROI is the so-called Biodiversity-EROI (= $UB/(RB + UB + \text{External inputs})$) that defines the return on energy invested in the agroecosystem in the form of available phytomass for wild heterotrophic species (UB). The relationships between energy flows and biodiversity have been put forward by ecologists, based on concrete studies that show that systems with higher amounts of energy entering the food web can support longer food chains and, therefore, increased biodiversity (Thompson et al. 2012). The EROI of Spanish agriculture decreased more slowly in the first half of the century and fell more sharply in the second half, especially as of 1970 (Guzmán et al. 2017). As we explained earlier, this decline was encouraged by the increase in biomass extraction for human and animal food, and by harmful practices such as the burning of crop residues and the use of herbicides on cultivated land. If we calculate the Biodiversity-EROI exclusively for croplands, the drop is much bigger (Graph 5.3). Therefore, the heterotrophic species associated with agricultural areas were the first to be affected by the relative scarcity of available phytomass. This effect has not been compensated by the abandonment of pastures and woodlands. In summary, the agroecosystem's dissociation between intensive production areas and abandoned and/or protected areas (e.g., 40.5% of total forest areas of Spain are protected, according to MAGRAMA 2014) has not achieved a significant increase in trophic energy available for transfer from plants to other levels in the food webs.

Graph 5.3 Biodiversity-EROI in the Spanish agroecosystem and in cultivated lands



According to the supporters of “land sparing” (see Phalan et al. 2011 as an example), the intensification of agriculture based on the incorporation of external inputs to create areas free from human intervention is an appropriate way of maintaining biodiversity. Our results put this stance into question. Land sparing was implemented in the second half of the twentieth century in Spain, and there seems to have been no benefits for wildlife, as we will see later.

It is worth noting that the sharp fall in cultivated land UB also brought about a drastic reduction in organic matter soil inputs, negatively affecting the quality of this fund element. To evaluate these effects, we chose to use material flows, specifically carbon, as exposed in the next section.

Lastly, we will describe the evolution of the relationship between socialized biomass and external inputs, called “net efficiency” or “external final EROI” ($EFEROI = SB/\text{external inputs}$). This ratio constitutes the most widely used energy efficiency indicator in agricultural studies; it has the advantage of being comparable with other studies, although it also has the serious drawback of considering the agroecosystem as a black box because it does not take internal flows into account (Guzmán and González de Molina 2015; Tello et al. 2016; Guzmán and González de Molina 2017). Its use is, therefore, more economic than agroecological. The evolution of this indicator is even more baffling. It fell from 17.3 in 1900, to 4.8 in 1960 and 1.2 in 2008 (Guzmán et al. 2017). These figures are similar to those obtained by Carpintero and Naredo (2006) for Spanish agriculture, i.e., 6.10 in 1950–51 and 1.27 in 1999–2000.

To summarize, the energy efficiency of Spanish agriculture followed a downward trend, with serious economic and environmental consequences: dependence on inputs from outside the sector increased to the detriment of internal biomass flows that feed the components of the land fund element.

5.3 State of the Components of the Land Fund Element

The changes described in the system’s functioning modify the state of the land fund element’s components, as explained below.

5.3.1 Soil

- (a) Replacement of edaphic macronutrients (N, P, and K) closing the nutrient cycle at the agroecosystem scale

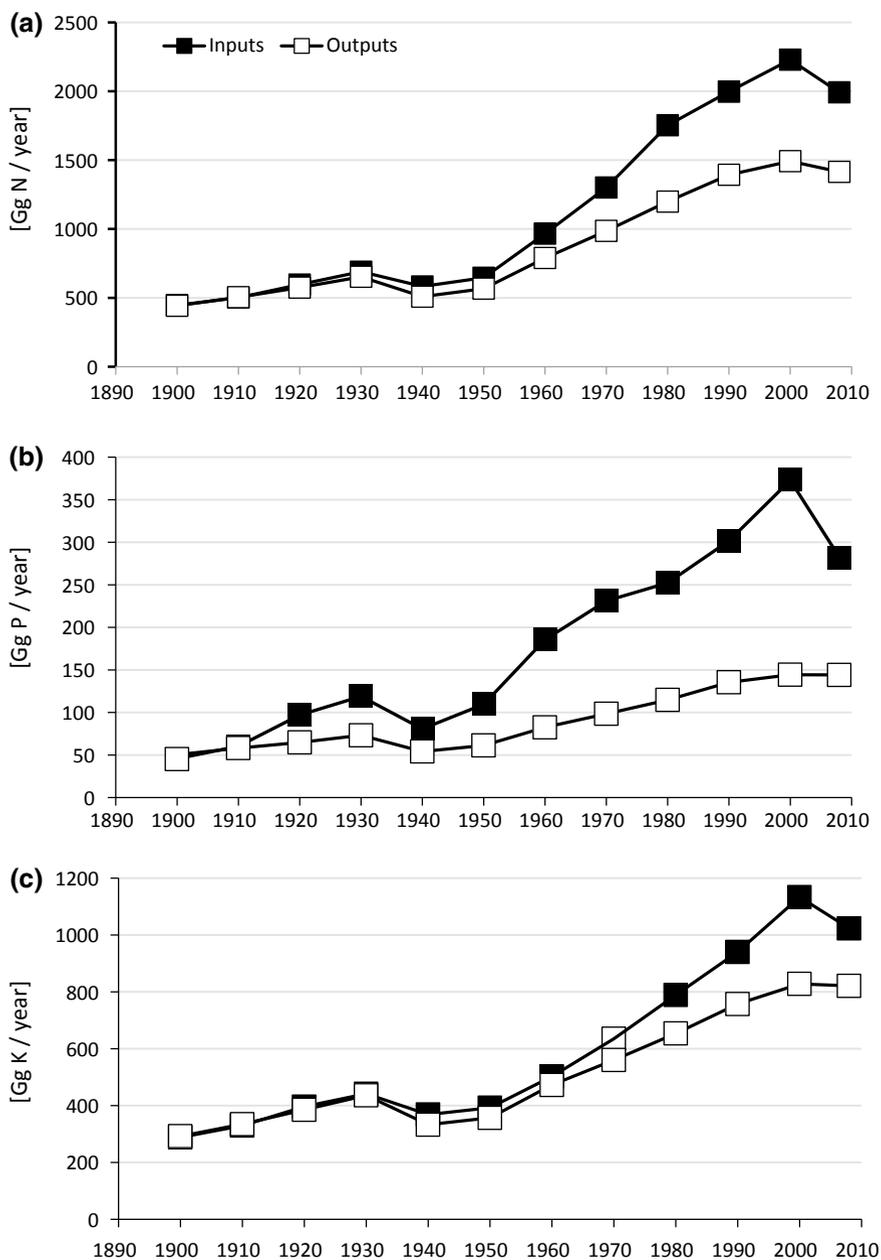
Throughout the twentieth century, changes in the amount of nutrients in the land’s soil, resulting from the balance between inputs and outputs, followed three trends based on an intricate set of socio-economic and political factors.

Until 1950, the balances of N and P (inputs minus outputs) were fairly balanced (from $+0.23 \text{ kg N ha}^{-1} \text{ year}^{-1}$ and $-0.28 \text{ kg P ha}^{-1} \text{ year}^{-1}$ in 1900, up to $3.9 \text{ kg N ha}^{-1} \text{ year}^{-1}$ and $2.4 \text{ kg P ha}^{-1} \text{ year}^{-1}$ in 1950) in cultivated fields (Graph 5.4). In the case of K, it was negative during the first 20 years of the twentieth century ($-0.26 \text{ kg K ha}^{-1} \text{ year}^{-1}$ on average) and was slightly positive until 1950 ($1.1 \text{ kg K ha}^{-1} \text{ year}^{-1}$) (Graph 5.4). These balanced or slightly positive balances in the mid-twentieth century became markedly positive from the 1960s, especially in the case of nitrogen, with average annual increases of 0.79, 0.20 and 0.38 kg of N, P and K ha^{-1} until 2000 (Graph 5.4). In 2000, the surplus entering annually in the fields was $40.3 \text{ kg N ha}^{-1}$, $12.5 \text{ kg P ha}^{-1}$ and $16.7 \text{ kg K ha}^{-1}$. This value in the case of N is lower than that provided by Leip et al. (2011) who found a positive annual balance for N of $50 \text{ kg N ha}^{-1} \text{ year}^{-1}$ for Spain during the 2001–2003 period, slightly lower than the European average (EU27). However, the values were similar to those quantified for the set of herbaceous crops ($39.7 \text{ kg N ha}^{-1} \text{ year}^{-1}$) and woody crops ($41.1 \text{ kg N ha}^{-1} \text{ year}^{-1}$) for Spain in 2011 (MAGRAMA 2013).

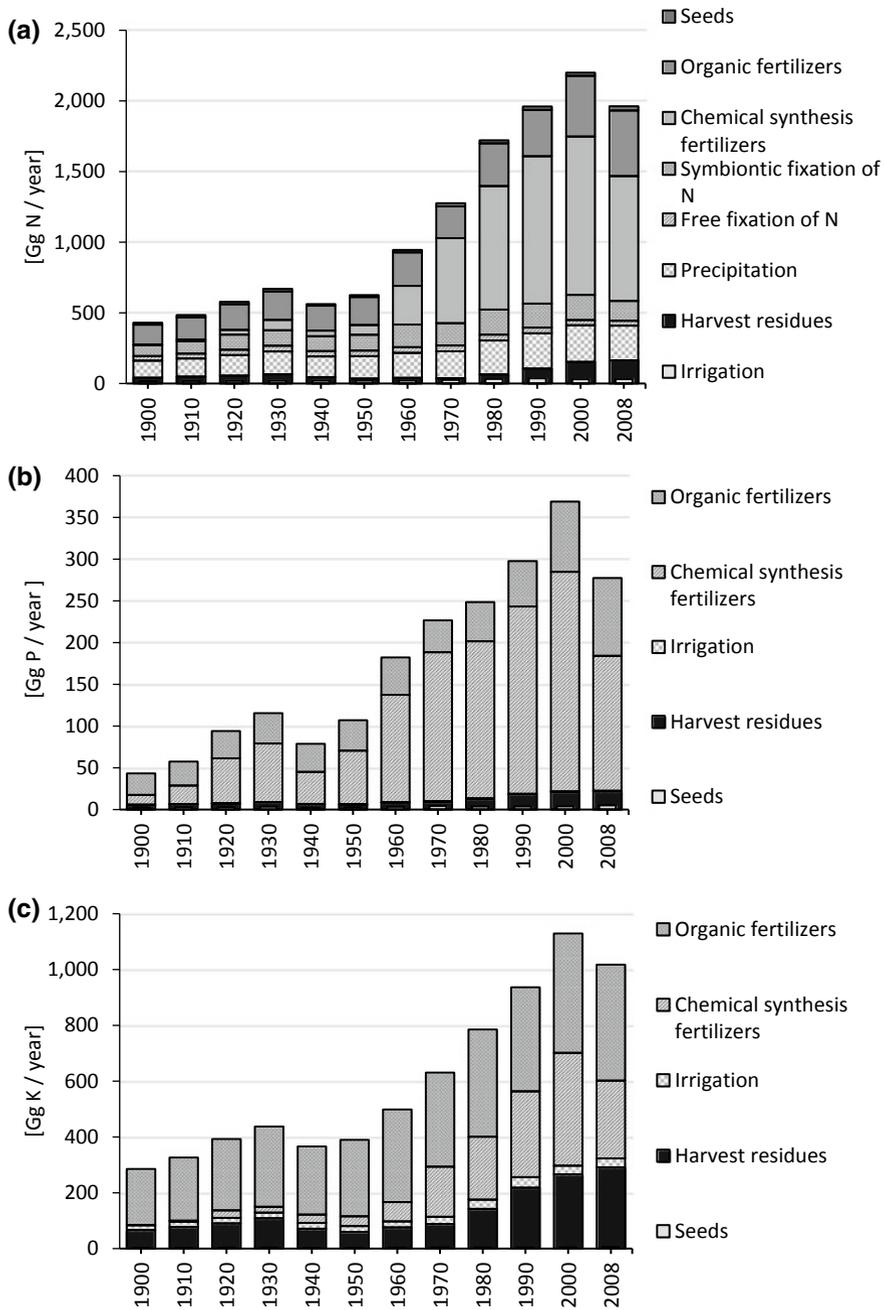
The patterns of change in the nutrient balance were mainly due to changes in the amount and input entry routes of N, P and K. During the first third of the twentieth century, the annual rate of N, P, and K inputs increased slightly (Graph 5.5). In the case of N, between 57.3 and 63.4% of annual inputs during this period corresponded to natural (precipitation and fixation of atmospheric N) and recycled (crop residues) inputs on the production site itself, while the contribution of natural and recycled inputs of potassium was low (20.0–25.5%) or very low in the case of phosphorus (10.3–17.7%). Manure constituted between 41.6 and 56.0% and 65.2–69.6% of the entries of P and K, respectively. During this period, annual inputs from synthetic fertilizers in the case of N and especially of P, as well as rising production levels led indirectly to input increases in the form of crop residues and manure (Graph 5.5).

During the 1940s, the annual input of nutrients decreased compared to the previous decade (Graph 5.5), mainly due to a decrease in inputs from crop residues, due, in turn, to lower production levels associated with a decrease in N and P inputs from synthesis fertilizers. From 1960 to 2000, annual inputs of N, P, and K, respectively, multiplied 2.3, 2.0, and 2.25 times (Graph 5.5). This increase has been mainly due to the increase of inputs through (i) synthetic chemical fertilizers, which in 2000 were 4.1, 2.0, and 6.0 times higher than in 1960 for N, P, and K, respectively; (ii) organic fertilizers, which increased between 1.3 and 1.9 over the same period; and (iii) crop residues that in 2000 were 3.8, 2.5 and 3.5 times higher than those of 1960 for N, P, and K, respectively. Worthy of note, the annual entries of N by biological fixation in 2000 were only 7% higher than those of 1960, revealing that the agricultural model changed from one based on inputs by biological fixation and manure to a model relying on synthetic fertilizers.

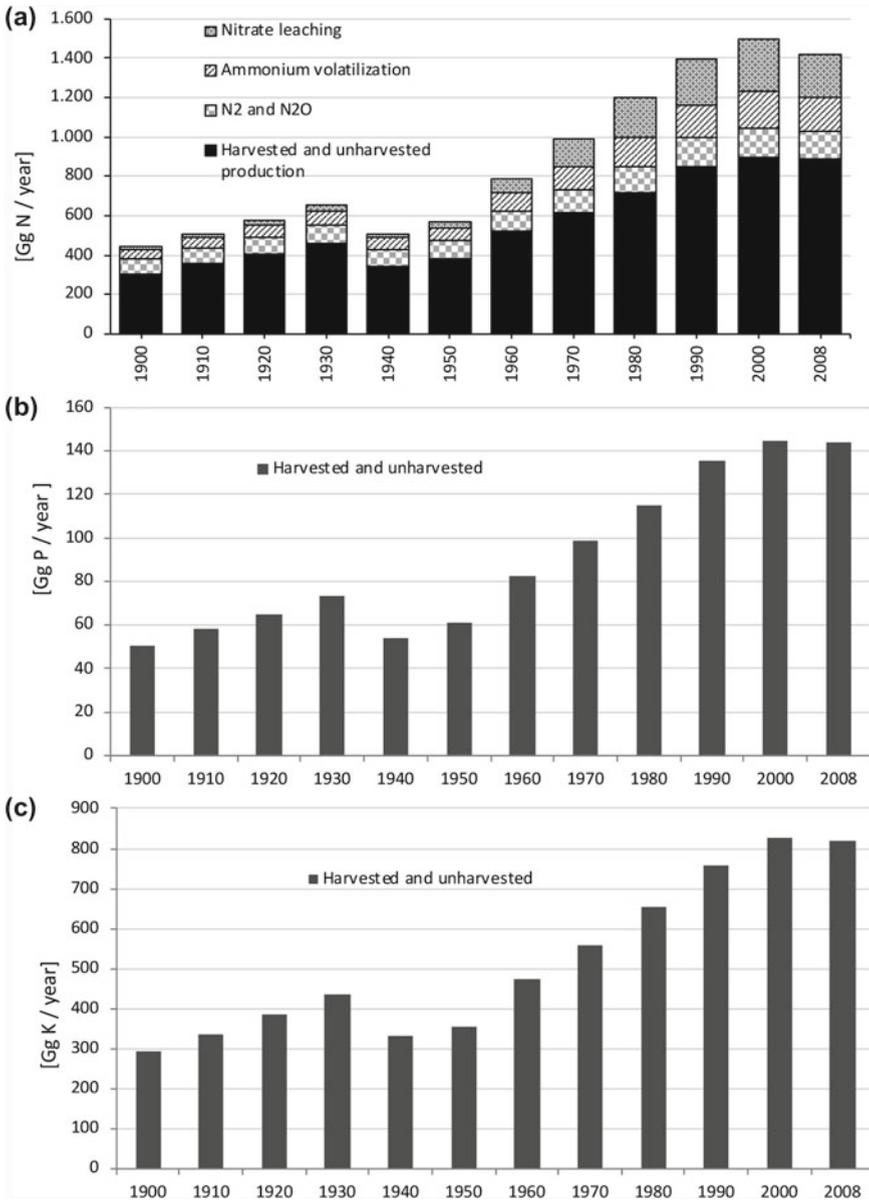
The notable increase in N, P, and K inputs in Spanish lands, which were 4.2, 5.8, and 3.4 times higher for N, P, and K in 2008 than in 1900, did not occur simultaneously to outputs due to aerial biomass production. During the study period, the amount of N, P and K in the produced biomass multiplied between 2.8 and 2.9 times, but at different rates over the three clearly identified periods (Graph 5.6).



Graph 5.4 Inputs and outputs of nitrogen (a), phosphorus (b) and potassium (c) in Spanish cultivated lands from 1900 to 2008, in gigagrams of N, P and K. The balance is equal to the difference between the inputs and outputs



Graph 5.5 Annual inputs of N (a), P (b) and K (c) through different entry routes in cultivated lands over the 1900 to 2008 period in Spain, in gigagrams of N, P and K per year



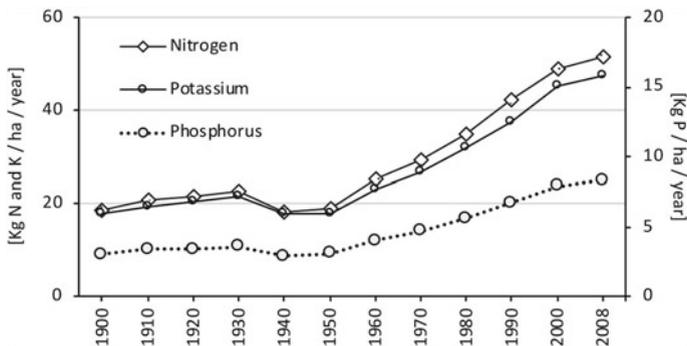
Graph 5.6 Annual outputs of N, P and K via different routes into the cultivated lands during the 1900 to 2008 period in Spain, in gigagrams of N, P and K per year

At the end of the first third of the century, the outputs of N, P, and K were 45.2–49.2% higher than those recorded for 1900, and between 69.2% and 70.9% corresponded, for the case of N, to outputs in the form of aerial biomass, the main destination being animal feed (Graph 5.6). Indeed, between 61.4 and 63.7%, 54.7–57.7% and 75.7–77.5% of N, P and K produced in the cultivated lands were aimed at stock feed. During this period, 13.6, 11.0, and 23.1% of N, P, and K were harvest residues that remained in the field.

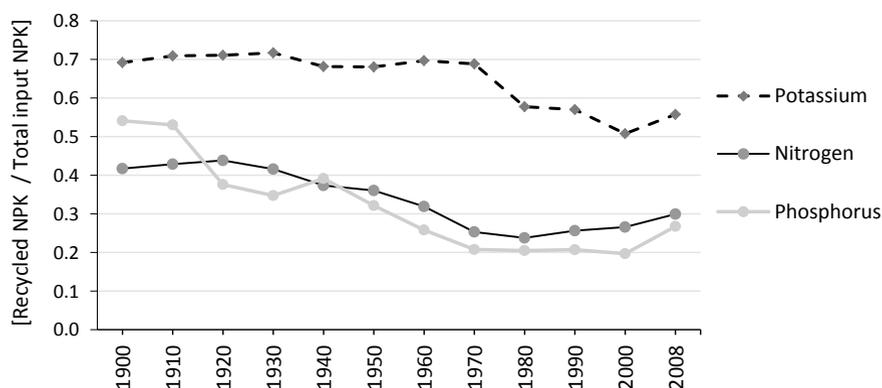
As in the case of inputs, there was a turning point in the 1940s when N, P and K outputs, dropped compared to the end of the first third of the century, mainly due to declining production (Graph 5.6). Nutrient outputs increased markedly from the 1960s and in 2000, N, P, and K were 89.0, 74.9, and 75.2% higher than those estimated in 1960. In 1900, the total annual outputs of N, P, and K were 26.8 kg N ha⁻¹, 3.05 kg P ha⁻¹ and 17.7 kg K ha⁻¹, while they were 81.9 kg N ha⁻¹, 8.3 kg P ha⁻¹ and 47.5 kg K ha⁻¹ in 2008. The productivity of N, P, and K in the form of aerial biomass rose slightly during the first third of the century, decreased during the 40s, and increased steadily after the 60s (Graph 5.7). In 2008, productivity had multiplied twofold compared to 1960 and multiplied 2.68–2.77 times compared to 1900, for N, P, and K.

The share of recycled nutrients—i.e., harvest and manure residues exclusively from livestock feed produced in crop fields—in nutrient inputs, tended to decrease throughout the study period (Graph 5.8). In the cases of N and P, this share decreased at a fairly similar rate until the end of the 1950s, after which its decline accelerated and in 2008, the recycling of N and P compared to total inputs was approximately half that of 1900. In the case of K, there was a sharp drop from the 1970s, mainly due to the increase in K inputs in the form of synthetic fertilizers.

During the first third of the twentieth century, annual net transfer of N, P, and K from pastures to cultivated lands, in the form of manure that was collected while the livestock was stabled, was more or less constant (Graph 5.9). It increased during the 1940s and part of the 1950s (Graph 5.9). From the 60s, this pattern was reversed and



Graph 5.7 Productivity in terms of N, P and K in the aerial biomass produced in Spanish cultivated lands over the 1900 to 2008 period, in kilograms of N, P and K per hectare and per year



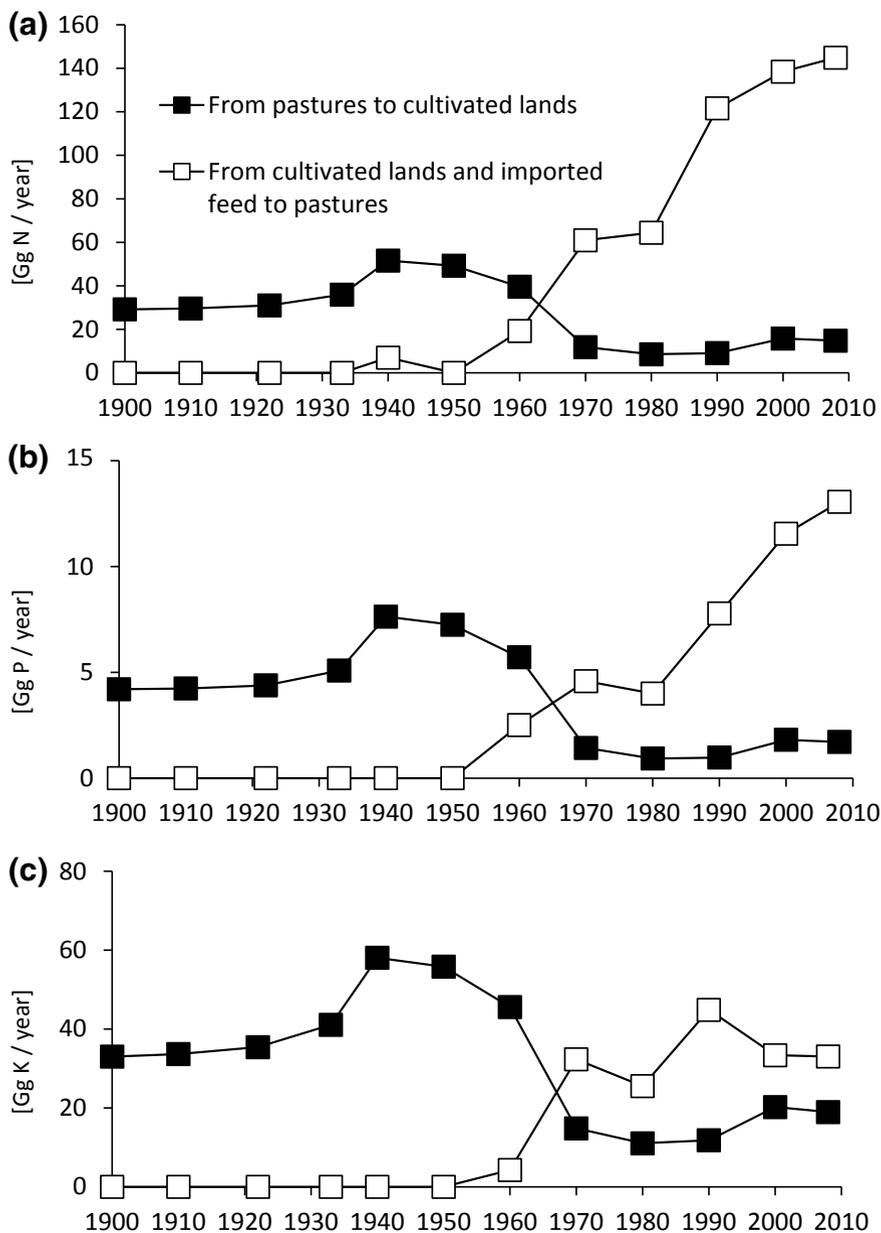
Graph 5.8 Share, out of 1, of the recycling of N, P and K of the total input in Spanish cultivated lands over the 1900 to 2008 period

in 2008 there was a net transfer of N, P, and K from cultivated lands to pastures in the form of excretions originating in food from the land—imported or not.

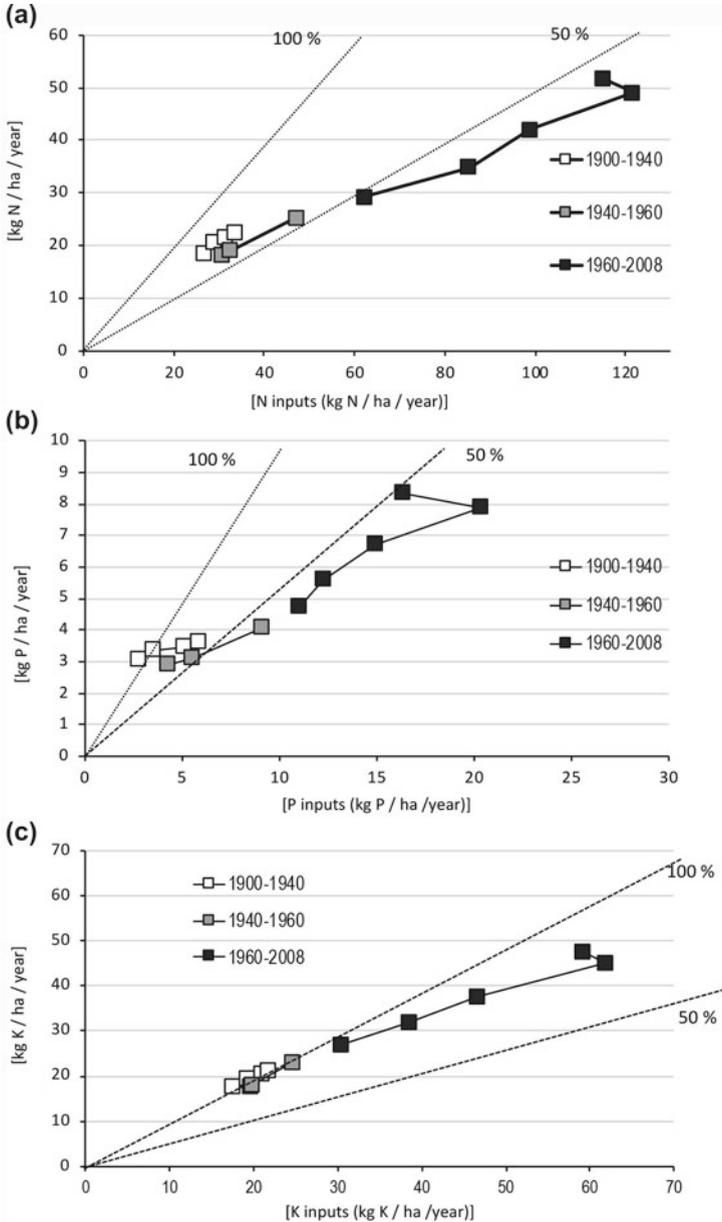
The gap in quantitative terms between total inputs and aerial biomass outputs led to drops in the efficiency of use of N, P, and K and rising losses of N to the atmosphere and other ecosystems with potential negative effects. The efficiency of use of N, P, and K (NPK contained in the aerial NPP divided by the total number of NPK inputs) shows a clear pattern related to the notable increase in nutrient inputs via synthetic fertilizers and manure produced from imported food since the last third of the twentieth century. The efficiencies of N use at the beginning of the twentieth century were above 66% and dropped to 40% in 2000 (Graph 5.10). In the case of P, the efficiency fell from 110% in 1900 to 38% in 2000, and in the case of K, that has always been higher than N and P, it ranged between 90 and 100% during the first half of the twentieth century, declining as of 1960 to reach 73% in 2000. For all three macronutrients, use efficiencies increased in 2008, mainly due to a decrease in inputs through synthetic fertilizers, despite the fact that productivity did not fall.

The decrease in N use efficiency in Spanish agricultural land took place at the same time as the absolute losses of N and those relative to the N produced in the aerial biomass, especially as of the 1960s. In 2000, the annual amount of lost N ($30.3 \text{ kg N ha}^{-1} \text{ year}^{-1}$ on average) was 3.7 times higher than in 1900 ($8.3 \text{ kg N ha}^{-1} \text{ year}^{-1}$), although the largest annual increases were observed since the 1960s (Graph 5.11). In 1900, 0.41 kg N was lost for each kg of N produced, and this value increased by 59% in 2000 (0.66 kg N per kg of N produced) (Graph 5.11).

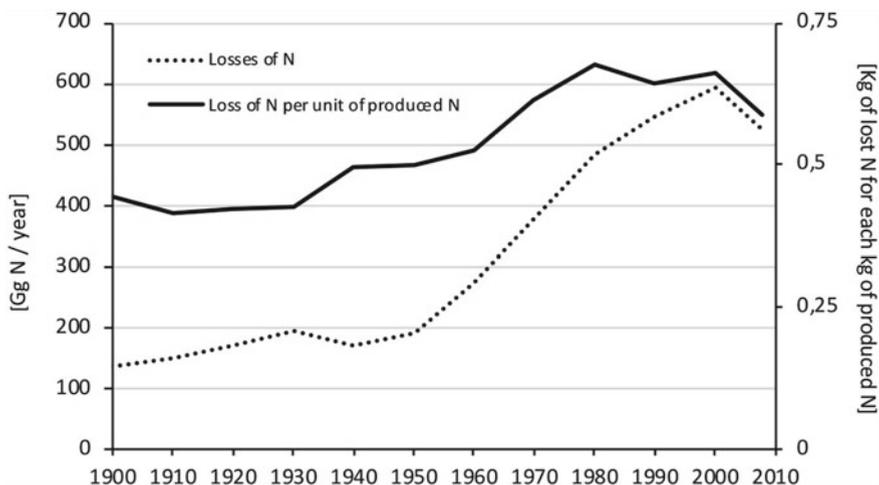
These losses of N are linked to environmental impacts that have been described in detail (Erisman et al. 2013). The N cycle in agroecosystems has indeed many escape routes because nitrate is a very mobile nutrient, and non-retained nitrogen in ecosystems, mainly in the form of organic nitrogen, can be transferred to the atmosphere, in the form of N_2O , N_2 , and NH_3 or to other aquatic and terrestrial ecosystems in the form of nitrate, where it contributes to a large amount of adverse



Graph 5.9 Annual transfer of N(a), P(b) and K(c) from the pastures to cultivated lands and from the cultivated lands and imported feed through excretions over the 1900–2008 period in Spain, in gigagrams of N, P and K per year



Graph 5.10 Relationship between the productivity in terms of N(a), P(b), and K(c) and the annual inputs of N, P, and K in Spanish cultivated lands. The dashed lines reflect the 100% and 50% use efficiencies of the three macronutrients



Graph 5.11 Annual losses of N (Gg N, in the form of N_2 and N_2O , volatilization of ammonium and leaching of nitrate) and losses relative to produced N ($\text{kg N lost kg N produced}^{-1}$) of crops

effects. Nitrate lost in agroecosystems can become a pollutant in surface waterways, encouraging eutrophication processes that result in a loss of biodiversity and loss in water quality, and in aquifers. Relatively high levels of nitrate ($>25 \text{ mg NO}_3^- \text{ l}^{-1}$) in drinking water have been linked to the incidence of colon cancer (Grizzetti et al. 2011). Moreover, nitrate can be lost in agroecosystems in the form of N_2O , which is the third greenhouse gas and contributes to the destruction of the ozone layer (Ravishankara et al. 2009) therefore, to climate change (Galloway et al. 2003; Galloway et al. 2008). Finally, ammonium can be transformed into ammonia gas that, in high concentrations, can be toxic to plants (Nordin et al. 2011) and is considered an atmospheric pollutant (Sutton and Fowler 2002).

5.3.2 Replacement of Organic Carbon

In both ecosystems and agroecosystems, the main driver of soil organic carbon (SOC) is the input of biomass (Aguilera et al. 2013). The fundamental difference is that in agroecosystems, the size of this entry is conditioned by the cultivation method both directly and indirectly. It is conditioned directly because a number of management practices (burning waste, organic fertilization, application of herbicides, etc.) intentionally modify the size of this entry; and it is conditioned indirectly because the NPP is affected by farming methods, to the extent that they affect the state of the fund elements and/or modify the availability of limiting factors. On the other hand, management practices also affect SOC because of impacts on the mineralization of organic matter, thus altering the system's output. For example, irrigation stimulates

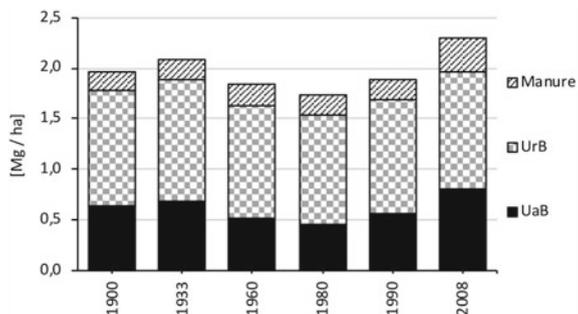
microbial activity and therefore mineralization and tillage can generate a provisional increase in mineralization, while it can also move carbon to deeper layers, where it is more protected. Therefore, the SOC balance is the result of distinct and sometimes opposite processes.

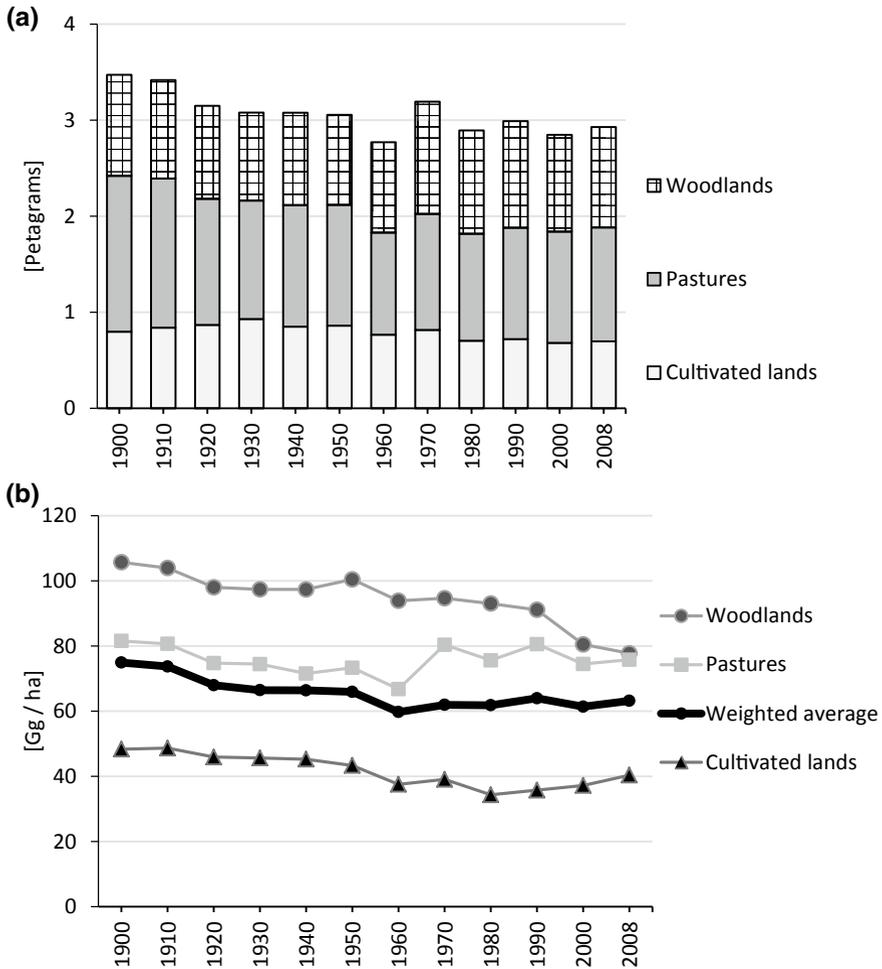
In the case of Spain, it is worth asking to what extent and when UB entries to the soil have been compensated by external biomass imports. Graph 5.12 shows that in cultivated lands, carbon inputs in the soil reached their maximum levels in the first third of the twentieth century, after which they shrunk and reached their minimum levels in the 1980s. A progressive recovery is currently underway related to a certain increase in UB and the greater availability of manure due to continually expanding livestock. The UB increase results from bigger restrictions to stubble burning and to an increase in residue production that has accompanied the expansion of irrigation and the drop in grazing on agricultural land over the last decade. The drop in carbon entry during the twentieth century helps to explain why half of Spain’s agricultural land currently has an organic carbon content of less than 1% (Rodríguez-Martín et al. 2009).

Graph 5.13 shows the stocks of equilibrium SOC in each period, after calculating the balance between inputs and outputs. As shown, there are significant differences in SOC stocks per hectare between the different land uses (Graph 5.13b). Therefore, changes in the total stocks of C (Graph 5.13a) were partly due to changes in land use, and partly due to the evolution of equilibrium C stocks for each type of land use. The highest levels of equilibrium SOC were reached at the beginning of the twentieth century, due, on the one hand, to relatively high levels of C inputs and, on the other, to relatively low mineralization rates because of lower average temperatures, to still reduced irrigated land surface areas, and to a good vegetation cover in woody crops (Aguilera et al. 2018). The equilibrium SOC began to fall in 1920, first because of expanding cultivated areas to the detriment of pastures, and from the 50s onwards, due to ever more widespread burning practices, herbicides, and the reduction of harvest indexes due to varietal changes (Graph 5.13b).

As of 1990, stocks of equilibrium SOC of agricultural lands recovered but did not attain the levels of the early twentieth century. The slight increase is due to the increase in C entries, described above. Despite this, the levels were below their potential with respect to current levels of productivity and are not sufficient to maintain the

Graph 5.12 Evolution of annual soil carbon entries in agricultural land in the twentieth century, Mg/ha/year. Note: Unharvested aerial Biomass (UaB); Unharvested root Biomass (UrB)





Graph 5.13 Evolution of stocks of equilibrium organic soil carbon for different land uses and for the agroecosystem as a whole, 1900–2008, petagrams, total (a) and per hectare (b)

SOC in a context of increased mineralization because of rising average temperatures, associated with climate change, of extensive irrigated areas, and scarce plant cover in woody crops. The effect of climate change in recent decades has also been visible in pastures and woodlands: while it stabilizes equilibrium levels in a context of increased inputs in pastures, it leads to lower equilibrium SOC levels in a context of stable inputs in woodlands (Graph 5.13b).

In terms of the stock of equilibrium SOC with respect to the total biomass recycled in cultivated lands (UB + RB + imported feed consumed by livestock in cultivated lands), the evolution is markedly negative. This indicator falls from 844 g C MJ⁻¹ in 1950 to 451 g C MJ⁻¹ in the year 2000. In other words, it is necessary to recycle 87%

more biomass to obtain the same stock of SOC in agricultural lands. The current biomass management strategy is clearly inadequate in relation to climate change mitigation. Monogastric livestock, whose manure, often handled in liquid form, has a low C: N ratio contributes to this low efficiency, in addition to greater mineralization due to climatic change and the expansion of irrigation.

The equilibrium SOC data presented correspond to hypothetical values and not true estimates of actual content at a given time. We carried out these estimates for agricultural lands in Aguilera et al. (2018), obtaining very similar values to those estimated by Rodríguez-Martín et al. (2016) in their exhaustive study of the SOC in Spanish soils based on field measurements. We do not dispose of any similar estimate for pastures and woodlands, but we can use the average equilibrium SOC values over the period studied as a guideline, i.e., 75.8 and 94.5 Mg C ha⁻¹ respectively. These values are situated 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 agricultural lands indicate that these soils are at a degradation threshold (Romanyà et al. 2007; Rodríguez-Martín et al. 2016). The increase in NPP resulting from intensification, together with massive feed imports, could have theoretically led to a substantial increase in the return of organic carbon to the soil, thus enabling to face increased mineralization brought about by climate change and the spread of irrigation. However, the breakdown of the balance between the uses of biomass and the preferential use of feed to nourish poultry and pigs prevents this from happening.

5.3.3 *Biodiversity*

Non-agricultural biodiversity is another fund element to have been seriously harmed by Spanish agriculture's metabolic transformation. According to the "country profile" developed for Spain by the "Convention on Biological Diversity", Spanish biodiversity has "dropped significantly" in recent decades, and 40–60% of the evaluated species have been included in some threatened category (CBD 2017). There are multiple causes. According to the Evaluation of Spain's Ecosystems of the Millennium report, the greatest threats to biodiversity include the expansion of intensive agriculture, urbanization and habitat fragmentation caused by the increase in linear transport infrastructures (EEME 2011, pp. 52). The Observatory on Sustainability also establishes a strong relationship between the intensification of agriculture and the loss of biodiversity in Spain (OS, 2016).

The impact of intensive agriculture on non-agricultural biodiversity is of a strong and complex nature. First, contamination by fertilizers and pesticides has been found to be a major problem, and it goes against the European Union's clearly downward trend (OS 2016). In this book, we described the evolution of N flows from agriculture that is dissipated in the atmosphere and waters, affecting terrestrial and aquatic

biodiversity. We also presented the increasing size, in terms of energy, of the flows of pesticides that have been poured into the agroecosystem (see Chap. 4).

Second, several authors have emphasized how the industrialization of traditional agriculture has damaged biological diversity due to the loss of landscape heterogeneity at multiple spatial and temporal levels (Benton et al. 2003; Firbank et al. 2008; Lindborg and Eriksson 2004; Perfecto and Vandermeer 2010; Schuch et al. 2012; Vos and Meekes 1999). In other case studies, we showed how the local generation of energy and material flows that sustain organic-based agriculture is reflected in the landscape, leading to complex land matrices (Guzmán and González de Molina 2009; Garzón et al. 2011) that favor biodiversity. Conversely, simplified landscapes proper to industrialized agriculture harm biodiversity (Marull et al. 2015).

Third, we showed how agricultural industrialization has generated a major imbalance in the uses of phytomass in cultivated lands, which further weakens the conservation of non-agricultural biodiversity. If we consider threatened bird species as an indicator of biodiversity, 17.5% of them are associated with pseudo-stem cereals and 5% with high diversity agricultural areas (orchards, irrigated tree crops, etc.) (EEME 2011, pp. 80). MAPAMA (2013) thus shows that between 1998 and 2012, the trend for birds associated with the agricultural medium, expressed as % change in populations, was 4.8% in tree crops and 25% in cereal crops. The cereal crop group does not use fertilizers and pesticides particularly intensively. However, the fiasco of unharvested biomass reached a peak in the case of cereals. Modern varieties have little straw, are mostly low in size and of short cycle, the use of herbicides and the burning of stubbles mean these crops leave hardly any useful biomass for heterotrophic species, which affects the size of the populations they can sustain and the trophic chains they are part of. This is the case of populations of birds of prey, such as the Lesser Kestrel. The decline of these birds is linked to the fact that they need to invest much greater efforts to catch their prey (arthropods and small vertebrates) in the cereal fields since they have been modernized (Ministry of the Environment 2004). For other birds linked to cereals, UB also offers a refuge for them to reproduce themselves. On this subject, the reader can consult the actions recommended in the "Life Project" for the conservation of steppe birds issued by the Ministry of Environment of Andalusia (2003).

It is very difficult to discern the isolated effects of each of these processes on declining biodiversity. Interactions and synergies are likely to exist between them. In recent years, the conversion of farms from Industrialized Agriculture (also called conventional agriculture) to Organic Farms (managed without agrochemicals and using organic fertilization making them similar to traditional farms) provide us with keys to build a deeper understanding of the interactions. These studies have proliferated in recent years and various available meta-analyses show that converting to organic farming improves biodiversity in cultivated areas (Bengtsson et al. 2005; Hole et al. 2005; Norton et al. 2009; Tuck et al. 2014; Gomiero 2015). Only some of the studies collected in the meta-analyses assess the specific causes of this rise in biodiversity. Some studies show that the main causes consist of a greater complexity of the landscape and ecological connectivity and the reductions in the pressure of biocides. However, in recent years, several authors have found that the increase of forage

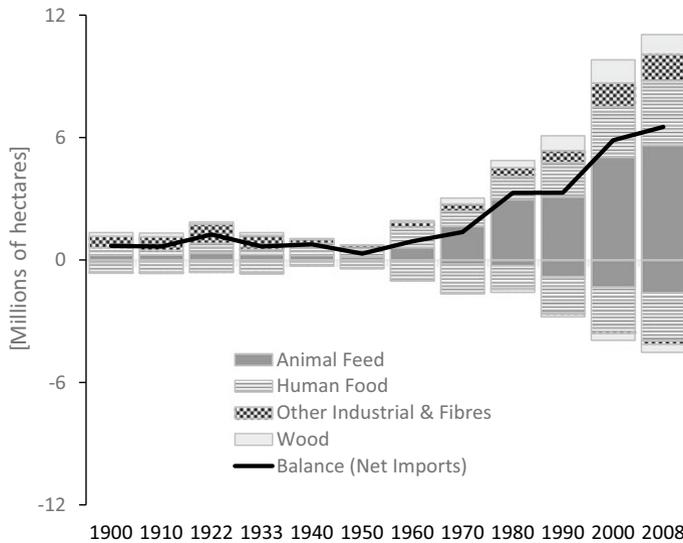
resources (availability of phytomass) for heterotrophic species is one of the drivers of this increase in biodiversity. For example, Rundlöf et al. (2008) state that the two main favoring factors of bumblebees in organic farms, compared to conventional ones, are the prohibition of agrochemicals and the provision of additional fodder resources. Döring and Kromp (2003), in their literature review on carabid beetles in conventional versus organic agriculture, found that in most cases, the species was richer in organic farms, because ecologically managed fields provided larger food supplies for herbivorous fauna. Wickramasinghe et al. (2004) found that organic management was beneficial for bats both through the provision of more structured habitats, as well as greater food resources (insect prey). Gabriel et al. (2013) showed that differences in the biodiversity of different species (bumblebees, bees, butterflies, epigeous arthropods) between organic and conventional fields could be due to the fact that UB grass was more abundant in organic farms and, consequently, so was arthropod biodiversity.

5.4 A Diet Rich in Food of Animal Origin: The Outsourcing of Its Land Costs

To finish, we must point out that the negative effects on agroecosystems' fund elements are not limited to Spain's territory: they have also been partly outsourced abroad. While the effects of bad management affected local areas in traditional agriculture, with rise of international trade, these effects have become globalized.

To study this impacts Infante-Amate et al. (2018) estimated land embodied in Spain's biomass trade, i.e., the land required to produce the biomass exported and imported by Spain. Graph 5.14 shows the evolution and composition of land embodied in both imports and exports of biomass. Both of them depict strong growth in the period analyzed, mostly concentrated from the 1960s onwards. In fact, between 1900 and 1933, the land traded remained stable, while between ca. 1940 and 1960 both imports and exports declined. Land embodied in imports multiplied 8-fold over the whole period analyzed, from 1.3 million hectares (Mha) in 1900 to 1.9 Mha in 1960 and 11.0 Mha in 2008. Similarly, land embodied in exports increased 7-fold, from 0.7 to 1.0 Mha and 4.5 Mha, respectively. When distinguishing the final uses of traded land, feed appears as the main product traded, mainly as of 1960 in the case of imports. Its share in land embodied in imports rose from 0.6 Mha in 1960 to 5.6 Mha in 2008. This result is in relation to nutritional transition already explained in other parts of this book. At the beginning of the twentieth century, fibers and industrial products accounted for 41% of total land embodied in imports, while they dropped to 11.9% by 2008.

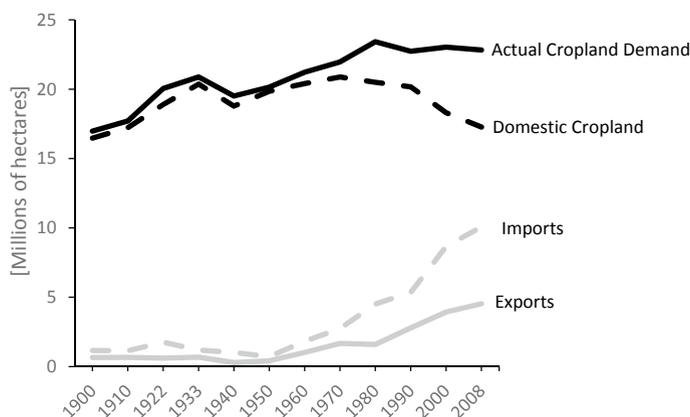
Net flows depict, therefore, a clear pattern of historical external dependence, which was especially evidenced from the 1960s. In all (twelve) benchmark years analyzed, Spain was a net importer of land, although the magnitude varied over the course of the period studied. In 1900, net imports accounted for 0.7 Mha or 372 m²/inhab, while the



Graph 5.14 Land embodied in biomass imports (positive values) and exports (negative values), in millions of hectares. *Source* Infante-Amate et al. (2018)

figure was 9 and 4 times higher in 2008, respectively (9.1 Mha or 1954 m²/inhab). The minimum value was found in 1950, due to the particular historical context related to the Spanish Civil War and the Autocratic policies developed under Franco's dictatorship, as discussed in Chap. 2.

In the case of cropland products, Infante-Amate et al. (2018) also provide evidence on embodied land in consumption activities, not only in trade. Knowing the actual cropland occupied in Spain (domestic cropland) and having estimated cropland embodied both in imports and exports, authors were able to quantify land requirements related to cropland-based biomass consumption. Cropland does not reveal any abrupt changes over the last century in Spain, moving from 16.5 Mha 1900 to 17.3 Mha in 2008, and peaking at 20.9 Mha in 1970. However, actual cropland use, considering the effect of land embodied in trade, shows more significant changes, mainly from 1960. In the early stages of the century, cropland use was basically explained by domestic cropland, due to low international trade levels. Actual cropland requirements were mainly met with domestic resources. Actual cropland use accounted for 17.1 Mha in 1900, a similar figure to domestic cropland. As trade gained importance, the gap between domestic cropland and actual cropland requirements increased (Graph 5.15). Today, actual cropland use (22.8 Mha) is 1.3 times higher than the cropland area occupied within the country (17.3 Mha). In the 1960s cropland embodied in imports amounted 1.9 Mha while cropland embodied in exports was 1.0 Mha. Net imports barely represented 4% of actual cropland demand and cropland embodied in imports equalled only 11.6% of domestic cropland, i.e., cropland occupied in third countries was only one tenth of land occupied for cropland



Graph 5.15 Actual cropland demand, domestic cropland, imports and exports of land, in millions of hectares. *Source* Infante-Amate et al. (2018)

within the country. Only five decades later, in 2008, net imports of cropland grew 7-fold, amounting to 37.7% of actual requirements, and total cropland imports made up 64.0% of domestic cropland. Given that a significant part of domestic cropland is devoted to exports (26.2% in 2008), today, Spanish inhabitants occupy a similar amount of cropland domestically as they do overseas to meet their cropland needs.

The expansion of the farming frontier represents one of the most pressing problems at a global level (Foley et al. 2005; Rockström et al. 2009), associated with deforestation (Kaplan et al. 2005; DeFries et al. 2010), which in turn leads to serious environment problems such as increased climate change (Cramer et al. 2004), losses of biodiversity (Pereira et al. 2012), and the alteration of the nitrogen cycle (Austin et al. 2006; Billen et al. 2014). In this chapter, we have explained the impacts of land intensification in Spain. Nevertheless, many other problems associated with the expansion of farmland and its intensification have taken place outside of Spain, and especially in highly sensitive areas from an environmental perspective. Spain's main imports of biomass are grain and oilseeds for animal feed (see also Lassaletta et al. 2014b; Soto et al. 2016), and the majority of these imports come from countries in which export agriculture generates serious social and environmental problems. According to Kastner et al. (2014), the main two exporters of agricultural land to Spain in 2009 were Brazil and Argentina, with just over one million hectares exported in each case. Many studies have indicated that agro-exports in these two countries are responsible for processes of deforestation and agrarian intensification (e.g., Kastensen et al. 2013; Lassaletta et al. 2014) that, as well as serious environmental problems, generate serious social problems that often lead to violent conflict, displacements and the breaking down of traditional communities (Hecht and Corckburn 2010; van Solinge 2010; Mayer et al. 2015). Recently, exports from Eastern Europe have grown in importance. Currently, Ukraine, Romania, and Bulgaria allocated almost 1.5 Mha to exporting agrarian products to Spain. Agro-exports in these

regions are also a growing cause for concern in terms of the socio-political and environmental problems (Visser and Spoor 2011) associated with them. In short, imports of biomass not only involve hidden land flows but also other impacts such as the loss of biodiversity, pollution, precarious labor, etc.

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