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Soybean expansion and the challenge of the coexistence of agribusiness with local production and conservation initiatives: pesticides in a Ramsar site in Uruguay

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Summary

Soybean has undergone the greatest expansion of any global crop, fuelled by the emergence of herbicide-resistant crops. In Uruguay, soybean croplands have increased from virtually zero to more than 1 million ha in 20 years. Uruguay is also implementing its system of protected areas. Here, we assess the presence of pesticides within a Ramsar site and protected area, in a basin dominated by croplands. We consider pesticides as surrogates of the subtle impacts of agribusiness on conservation initiatives and other productions. Pesticides were found in soils, fishes and beehives, both within and around the protected area. Endosulfan was found in all matrices analysed (23 of 80 samples), while glyphosate (0–2.31 mg/kg) and aminomethylphosphonic acid (AMPA; 0–0.61 mg/kg) were found in all soil classes. The study also allowed for a retrospective evaluation of a recent policy banning endosulfan in Uruguay, suggesting that while the protected area has not been immune to the impacts of agribusiness on human health or biodiversity, limiting the use of pesticides reduces or avoids some of them. This has implications for the design of multifunctional landscapes and for the debate on land sharing versus land sparing.

Introduction

The increase in the human population is fuelling agricultural expansion and intensification throughout the world (Godfray et al. 2010, Smith et al. 2010, Foley et al. 2011). Land-use change is probably the main driver of global environmental change (Foley et al. 2005, Nelson et al. 2006), with the expansion of soybean crops being one of the most remarkable examples of recent and large-scale land conversion in order to increase commodities production (Pacheco 2012, WWF 2014, Gasparri et al. 2016). It is also a remarkable example of the socioecological impacts of these changes (WWF 2014, Richards et al. 2015).

Soybean has undergone the greatest expansion of any global crop, with its production growing from less than 30 million tonnes to more than 300 million tonnes in the last 50 years, currently covering an area over 100 million ha (Masuda & Goldsmith 2009, WWF 2014, FAO 2019). More than 90% of the world's soybean production comes from Brazil, the USA, Argentina, China, India and Paraguay, with Bolivia and Uruguay progressively becoming major players as well (WWF 2014). Growing demand from the European Union and China is the main driver of this expansion, with markets in Africa and the Middle East projected to expand rapidly in the next decade (WWF 2014).

This expansion has been fuelled by the emergence of herbicide-resistant genetically modified (GM) crops in the 1990s. This made it possible for farmers to use glyphosate, a broad-spectrum herbicide, in ways that were previously impossible (Benbrook 2012). GM crops provide farmers with a simple, flexible and forgiving weed-management system (Garrett et al. 2013). GM soybean cultivation often goes hand in hand with other techniques, such as cultivation in rows sown closer together, and this allows for agricultural systems geared primarily towards increasing efficiency in terms of production costs per unit of product. Typical features of these systems include large farm sizes, specific cropping patterns and tillage practices, use of machinery and agrochemicals (pesticides and fertilizers) and plant breeding (Bonny 2008).

The outcomes of this expansion are still under debate. On the one hand, it has had positive impacts on economic growth, expanding earnings for the state and municipalities, helping to create employment at the regional level, and to develop processing industries down the value chain. In countries such as Argentina and Uruguay, it became a motor for each state's economy, increasing the demand for services, housing and goods, and providing a source of investment capital to the non-agricultural sector. However, these large-scale and capital-intensive activities have also contributed to land concentration, favouring traders and industry owners and concentrating income in a small group of larger enterprises, with limited inclusion of smallholders in the value chains (Pacheco 2012, WWF 2014, Richards et al. 2015).

In addition, large areas of forest, grassland and savannah have been converted to agriculture, either directly or indirectly, as a result of the global boom in soybean production (Grau et al. 2005, Grecchi et al. 2014, WWF 2014, Caldas et al. 2015, Fehlenberg et al. 2017). In South America, the area of land devoted to soybean grew from 17 million ha in 1990 to 46 million ha in 2010, mainly on land converted from natural ecosystems (WWF 2014). Industrial-scale soy production requires a large supporting infrastructure, including transport links, processing mills and workers' facilities, which has led to further loss of natural ecosystems (WWF 2014).

This expansion has been accompanied by an exponential increase in the introduction of pesticides into landscapes. In Uruguay, the importation of herbicides multiplied more than two-fold between 2003 and 2010, and the importation of the main insecticides used in soybean production increased nearly 20-fold between 2001 and 2010 (Oyhantçabal & Narbondo 2011). The use of agrochemicals is one of the main environmental threats linked to soybean production, causing soil contamination as well as impacts on water quality, biodiversity and human health (WWF 2014). Both the acute and long-term, low-dose exposure to some of these products are increasingly linked to human health effects, such as immune suppression, endocrine disruption, neurological dysfunctions, reproductive abnormalities and cancer. Pesticides can be toxic to a host of non-target organisms, including mammals, birds, amphibians, fish, beneficial insects and plants. Insecticides are generally the most acutely toxic class of pesticides, but herbicides can also pose significant risks to non-target organisms (Aktar et al. 2009, Saunders et al. 2012, Myers et al. 2016).

Largely because of these changes, Uruguay has been identified as one of the countries that will suffer the greatest loss in terrestrial biodiversity in the coming decades as a consequence of crop expansion and intensification (Zabel et al. 2019). In Uruguay, soybean croplands have increased from virtually zero to more than 1 million ha in the last 20 years (Oyhantçabal & Narbondo 2011, Redo et al. 2012, Alvarez et al. 2015, FAO 2019). At the same time, the country is implementing its National System of Protected Areas (SNAP, Spanish acronym). The system currently covers *c.* 1% of the land area (Di Minin et al. 2017). With over 90% of the country suitable for agricultural production, the SNAP faces the challenge of promoting biodiversity conservation in productive landscapes. This provides a unique opportunity for exploring ways to articulate national policies on agricultural production, land planning and biodiversity conservation, and to understand and make explicit some of the conflicts among these policies. Experience on the application of landscape and socioecological approaches in the implementation of protected areas (Palomo et al. 2014) can thus be gained as a means to promote sustainable development and to mainstream biodiversity conservation (Redford et al. 2015).

As an input for the design of the SNAP, Di Minin et al. (2017) identified key areas for biodiversity retention at the national scale that minimize conflicts with other land uses. Under a range of scenarios aimed at minimizing conflicts between biodiversity conservation, agriculture production, afforestation and opportunity costs for conservation, 12.6% of the country was consistently identified as a conservation priority (Di Minin et al. 2017). Yet, whether these areas will have the capability to protect key biodiversity and local productions from the impact of large-scale, technologically intensive activities has not yet been evaluated. Here, we assess the presence of pesticides in one of these areas. We use pesticides as a surrogate for some of the subtle, non-evident impacts of these activities on biodiversity and local productions. Despite the existence of a large literature on the direct and indirect impacts of various types of pesticides beyond the areas where they are applied, this is one of the first studies to simultaneously analyse the presence of the pesticides regularly used in soybean production in multiple environmental matrices.

Specifically, we analyse the presence of pesticides in and around a national park and Ramsar site. We assess the presence of the pesticides in three different environmental matrices within the protected area and compare these values with those found in croplands and tree plantations nearby. We discuss the impacts that these pesticides may have on the viability of some conservation targets of the national park and on the health and productive activities of local communities living in its surroundings. Finally, we discuss the implications of our findings for the promotion of biodiversity conservation policies in a country with an economy that is highly dependent of the exploitation of natural resources, as well as for the coexistence of large-scale and capital-intensive agribusiness with other economic activities dependent on natural resources.

Methods

Study area

The study was conducted in the Ramsar site and National Park Esteros de Farrapos e Islas del Río Uruguay (PNEFIRU, Spanish acronym) and its basin (Fig. 1), covering a total area of 58 949 ha (17 496 ha within the Ramsar site). This basin is characterized by intensive land use, with 27% of the basin covered in 2010 by rain-fed crops (mainly GM soybean–wheat rotation) and 20% by *Eucalyptus* spp. plantations (mainly for cellulose pulp production). Two towns concentrate most of human population: Nuevo Berlín (south of PNEFIRU, 2450 inhabitants) and San Javier (north of PNEFIRU, 1781 inhabitants). Beekeeping and small-scale fishing are important sources of jobs (Ríos et al. 2010).

Sampling and analytical procedures

The data analysed here were collected between 2009 and 2010. The environmental matrices analysed were selected in order to provide information on the potential impacts of pesticides in the main local productions and the conservation targets of PNEFIRU. They include: (1) freshwater fish species captured by local fishermen during their daily activities; (2) honey and wax produced by active beehives from local apiaries; and (3) soils from GM soybean crops, *Eucalyptus* spp. plantations and natural areas within the Ramsar site. Samples were also taken from mass die-off events from beehives and freshwater fish recorded during the study period.

Samples were taken during the soybean crop pesticide application period: from late austral spring through the summer. From February to April 2010, 27 samples of muscle of 8 species of fish weighing 10–20 g were taken from the dorsal area. Eleven 80 g samples of honey and ten 5 g samples of wax were taken in December 2009, and another 11 in February 2010. In addition, between February and April 2010, three beehive and one fish mass die-off events were recorded. From these events, samples of dead bees, wax and dead fishes were taken. Finally, 15 soil samples were taken from December 2009 to March 2010: five from soybean croplands, five from *Eucalyptus* spp. plantations and five from natural areas inside PNEFIRU.

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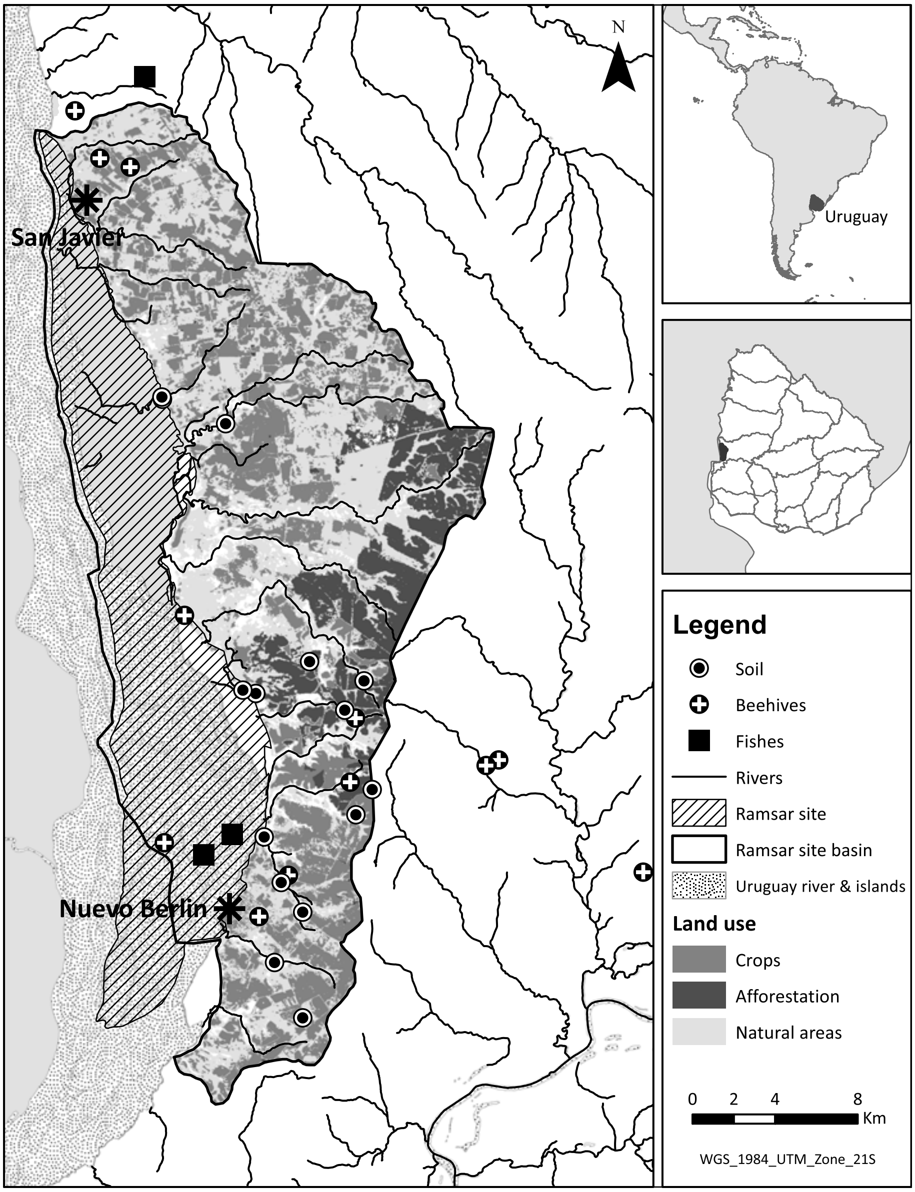

Fig. 1. Study area with the limits of the Ramsar site Esteros de Farrapos e Islas del Río Uruguay, its basin and the two main towns of San Javier and Nuevo Berlín, land uses and sampling sites.

A list of pesticides used in soybean production was compiled with information provided by the national cluster of organisms and companies involved in soybean production in Uruguay (http://mto.org.uy). Based on this list and on the analytic capacities of the laboratories involved in the analysis, we ended up with a different list of pesticides to be analysed for each matrix (Table 1).

Soil samples were analysed in the Laboratory of the Ministry of Cattle, Agriculture and Fisheries of Uruguay (Montevideo, Uruguay). Glyphosate and aminomethylphosphonic acid (AMPA) were analysed using derivatization with fluorenylmethyloxycarbonyl chloride (FMOC), while chlorpyrifos ethyl, α-endosulfan, β-endosulfan, endosulfan sulphate (reported together as total endosulfan), cypermethrin and fipronil were analysed following the method of Luke et al. (1981). Fish, bees, wax and honey were analysed using gas chromatography–mass spectrometry by Applica GmbH (Bremen, Germany), an ISO 17025-certified laboratory, part of the Intertek Group. Prior to analyses, soil samples were stored in plastic bags at 4°C. Bees, honey and wax were stored in paper bags, plastic bags and plastic jars, respectively, and kept at room temperature. Fish samples were packed in aluminium paper and stored in dry ice.

Results

In 34 of the 80 samples (42.5%), at least one of the pesticides was found. Endosulfan was found in fish, wax and soil (Tables 2–4). In fish samples, endosulfan was found in 50.0% of the species and in 44.4% of the individuals analysed, with recorded concentrations ranging from 0.009 to 0.052 mg/kg during regular sampling (Table 2). In dead fishes from the mass die-off event, concentrations were significantly raised ($t_{3,12} = 8.208$, $p < 0.001$) to values between 0.418 and 1.180 mg/kg. It was also found in one sample of wax, soil samples from soybean croplands, and wax and dead bees from the three bee mass die-off events (Tables 3 & 4).

Fishes showed different patterns of pesticides presence, not clearly linked to diet or migratory behaviour (Table 2). Endosulfan was found in *Hoplias malabaricus* (piscivore and non-migratory), *Pimelodus maculatus* (omnivore and short-distance migratory),

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Table 1. Pesticides regularly used in the soybean crops analysed in this study. Analytical limits provided by the laboratories differed.

Pesticide	LQ (mg/kg) and analytical method		
	Soils	Fishes	Bees, wax and honey
Glyphosate	0.03[a]	Not analysed	Not analysed
AMPA	0.03[a]	Not analysed	Not analysed
Chlorpiryfos	0.02[b]	Not analysed	Not analysed
α-endosulfan	0.02[b]	0.01[c]	0.01[c]
β-endosulfan	0.02[b]	0.01[c]	0.02[c]
Endosulfan sulphate	0.02[b]	0.01[c]	0.01[c]
Fipronil	0.02[b]	Not analysed	0.01[c]
Cypermethrin	0.10[b]	Not analysed	0.02[c]
λ-cyhalothrin	Not analysed	Not analysed	0.01[c]

[a]Derivatization with fluorenylmethyloxycarbonyl chloride (FMOC).
[b]Luke et al. (1981).
[c]Gas chromatography–mass spectrometry.
AMPA = aminomethylphosphonic acid; LQ = limit of quantification.

Table 2. Pesticides (mg/kg, mean ± SD (where n > 1)) detected in fishes in the Ramsar site Esteros de Farrapos e Islas del Río Uruguay and surroundings.

Fish species	Samples (n)	Positive samples	Total endosulfan
Hoplias malabaricus	6	5	0.019 ± 0.018
Pimelodus maculatus	3	3	0.028 ± 0.015
Prochilodus lineatus	3	3	0.021 ± 0.002
Megaleporinus obtusidens	3	1	0.011
Luciopimelodus pati	3	0	ND
Salminus brasiliensis	3	0	ND
Pseudoplatystoma corruscans	3	0	ND
Iheringichthys labrosus	3	0	ND
Fish mass die-off event	3	3	0.819 ± 0.383

ND = not detected.

Megaleporinus obtusidens (omnivore and migratory) and *Prochilodus lineatus* (detritivore and migratory). No pesticides were detected in samples from *Iheringichthys labrosus* (invertivore and non-migratory), *Luciopimelodus pati* (piscivore and migratory) and *Pseudoplatystoma corruscans* (piscivore and migratory).

No pesticides were found in honey samples either (Table 3). A wax sample from a mass die-off event registered in April 2009 contained endosulfan (0.063 mg/kg) and cypermethrin (0.023 mg/kg). Two further mass die-off events were registered in March and April 2010. Dead bees analysed contained λ-cyhalothrin (0.3 mg/kg, March samples) and endosulfan (0.25 and 0.028 mg/kg, values of March and April samples, respectively).

In soil samples taken from soybean crops, glyphosate (1 of 5 samples), AMPA (2 of 5), endosulfan (4 of 5) and chlorpyrifos (5 of 5) were recorded. In samples taken from tree plantations, glyphosate (5 of 5) and AMPA (3 of 5) were recorded, while glyphosate (3 of 5) and chlorpyrifos (3 of 5) were found in samples from the Ramsar site (Table 4). Glyphosate had not been used in the tree plantations sampled for at least 3 years and had never been used inside the Ramsar site.

Discussion

Studies on the impacts of pesticides and agribusiness on protected areas are scarce (e.g., Kaiser 2011, Quinete et al. 2013), and the present study is one of the first to simultaneously analyse the presence of the pesticides regularly used in soybean crops in multiple environmental matrices. Pesticides were found in every matrix analysed, including soils, fishes and beehives, in areas where the pesticides are often applied, but also in natural areas several kilometres away from these sites, and in productive areas where its presence was not expected, given their reported time of residence. This highlights the risks of assuming that residence and degradation times in natural conditions correspond to those reported from laboratory analysis.

Five out of the six pesticides (and derivatives) regularly used in the soybean crops assessed were detected in at least one of the samples analysed. Endosulfan was the pesticide found in the largest number of environmental matrices (fishes, wax, bees and soils), while fipronil was not detected in any sample. Glyphosate and AMPA were found in all the soil classes analysed (crops, afforestation and natural areas). Particularly worrying are the high levels of endosulfan detected in fishes of commercial value that are regularly consumed by local people. These can be sources of chronic exposure of humans to these substances, with effects on the health of rural populations, a process that is critical to understand but yet largely ignored both in Uruguay and elsewhere (Booth et al. 2015, Landrigan & Benbrook 2015). This highlights the need for comprehensive indicators of human health risks from exposure to agrochemicals, and from fish consumption in particular (Pérez-Parada et al. 2018).

In terms of the impact of pesticides on fish populations and hence on aquatic communities, fish contaminated with pesticides have a lower capacity to withstand sudden environmental changes such as abrupt changes in temperature (Vardia & Durve 1981, Patra et al. 2007). This is relevant in the current context of climate change and variability. In Uruguay, mass mortalities are usually considered as consequences of changes in water temperature or the effects of parasites, but concentration of pesticides in fishes is rarely considered as a probable cause (Franco Teixeira-de-Mello, personal observation).

A similar pattern was observed in relation to honey production. In samples from the mass die-off events, traces of endosulfan, λ-cyhalothrin and cypermethrin were found. Beekeeping is an economic activity that has been strongly impacted in Uruguay by the undesirable effects of pesticides. In recent years, exports to Germany have been rejected due to the presence of traces of glyphosate or its derivatives in honey, or due to the presence of transgenic soybean pollen. This has had a disproportionate impact on the smallest honey producers, with the number of beekeepers declining from a peak of nearly 3500 to less than 2200 in the last decade (Antúnez et al. 2013, 2017, CHDA 2018).

These results point to the difficulties of accurately assessing some of the impacts of agricultural expansion on human wellbeing. They also highlight the disparities in the ways in which these benefits and costs are distributed among stakeholders, with some of the most vulnerable sectors of the population bearing a disproportionate part of the burden (Urcola et al. 2015, Ezquerro-Cañete 2016). In terms of spatial and land planning, these results corroborate the incompatibility of some land uses and hence the need to articulate sectoral policies in order to ensure coherence when they are finally translated into actions on the ground (Cumming & Spiesman 2006, King et al. 2013).

These results also evidence the difficulties of reconciling biodiversity conservation initiatives and local productions with the expansion of agribusiness. Coexistence does not seem to be a realistic alternative, and it seems inevitable that this productive model will have significant impacts on the health of ecosystems and

Table 3. Pesticides (mg/kg, mean ± SD (where n > 1)) detected in beehives in the Ramsar site Esteros de Farrapos e Islas del Río Uruguay and surroundings.

Beehive material	Samples (n)	Positive samples	Total endosulfan	λ-cyhalothrin	Cypermethrin	Fipronil
Honey	11	0	ND	ND	ND	ND
Wax (December)	10	0	ND	ND	ND	ND
Wax (February)	11	0–1	0.014	ND	ND	ND
Wax mass die-off event	1	0–1	0.063	ND	0.023	ND
Bees mass die-off event	2	0–2	0.139 ± 0.157	0.3	ND	ND

ND = not detected.

Table 4. Pesticides (mg/kg, mean ± SD (where n > 1)) detected in soils in the Ramsar site Esteros de Farrapos e Islas del Río Uruguay and surroundings.

Soil	Samples (n)	Positive samples	Total endosulfan	Cypermethrin	Glyphosate	AMPA
Soybean crop	5	0–4	0.620 ± 1.127	ND	0.03	0.085 ± 0.064
Afforestation	5	0–5	ND	ND	0.290 ± 0.208	0.320 ± 0.265
Natural vegetation cover	5	0–3	ND	ND	0.123 ± 0.065	ND

AMPA = aminomethylphosphonic acid; ND = not detected.

people (Ezquerro-Cañete 2016, Phélinas & Choumert 2017). When it comes to designing productive policies, it seems crucial to develop mechanisms explicitly aimed at minimizing and reversing the negative effects of agricultural expansion or intensification, such as through the introduction of limitations on the pesticides used and the ways in which they are applied (Dudley et al. 2017).

After our sampling, the use of endosulfan was banned in Uruguay. Comparing the results of this study with information on pesticides after the banning provided a unique opportunity for evaluating the impact of this kind of policy. A more recent study of 72 pesticides, including endosulfan, in fishes from the same region of the country that we analyse here found no trace of endosulfan (Ernst et al. 2018). This is an encouraging result, which suggests that timely policies to mitigate some of the negative impacts of agricultural expansion may have significant positive impacts on species, ecosystems, local productions and human health.

However, the creation of protected areas does not seem to avoid some of the most pervasive impacts of agribusiness on human health or biodiversity. Observations from other protected areas around the world seem to reinforce this idea (Kaiser 2011, Quinete et al. 2013). This has implications for the design of multifunctional landscapes (Groot et al. 2007, Lovell & Johnston 2009, Renting et al. 2009, Selman 2009, Tittonell 2014) and the debate on land sharing versus land sparing (Fischer et al. 2014, Mertz & Mertens 2017): while land sparing does not ensure the protection of some valuable assets (e.g., biodiversity and health), it seems some land uses cannot simply coexist.

We draw two main conclusions from this study. Firstly, for the design of protected areas and land planning for multifunctional landscapes, activities in the surroundings of priority areas for conservation may ultimately impair the fulfilment of conservation objectives within those areas. Planning land uses around these areas should be an integral part of the process of designing and planning these areas' management. This is not new, but reinforces the recommendation that where the impacts of activities from outside protected areas cannot be minimized, other conservation strategies might have to be considered instead. Problems of a regional nature have to be tackled at the appropriate scales. It is important that the whole problem is recognized and that management efforts do not focus on only a small piece of it (Cumming & Spiesman 2006).

Secondly, for the design and assessment of the impacts of agriculture policies, it is crucial to bear in mind that agribusiness has negative impacts on other productive activities dependent on natural resources, the health of local residents and the environment, and that these will persist unless specific actions and policies are enacted to avoid them. In the specific case of pesticides, these can have negative spill-over effects far from the areas where they are actually applied. In order to reduce their impacts on other activities, segregation between land uses must be carefully analysed. It should also be considered that limiting the use of pesticides may also have positive impacts soon after implementation, as is suggested by our results. Consequently, this should be amongst the first strategies to consider (King et al. 2013). In any case, permanent monitoring of their effects on health, the environment and other productive activities should be an integral part of any development strategy aimed at promoting agribusiness.

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