International trade in agriculture: land use changes, biodiversity and environmental sustainability

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International trade in agriculture: land use changes, biodiversity and environmental sustainability

Échanges internationaux en agriculture: changements d'utilisation des sols, biodiversité et durabilité environnementale

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CHAPTER I

INTRODUCTION

I.1 Motivation

Farming is essentially a resource extraction industry: it harvests renewable natural resources (water, genetic resources, natural fauna and flora. . .) produced by biological processes and uses them as intermediary consumption to produce crops and livestock. Farmers should be stewards of environmental quality since their productivity and well-being depends on the availability and quality of natural resources. However, the increased use of synthetic inputs as substitute for natural resources is lowering the marginal value of natural resources for agriculture and, thus, farmers' incentives to preserve them.[1]

Since the early 1960s, modern production techniques have been generating many negative environmental externalities. Currently, some 30% to 50% of Europe's surface waters are affected by pollution from nutrient and pesticides residues, mainly resulting from agriculture activities (EEA, 2012). Also, agriculture accounts for 10% to 12% of total global anthropogenic emissions of greenhouse gases (IPCC, 2014), [2] with application of fertilisers and organic manure responsible for 27% of agricultural emissions. Emissions from land use changes are counted separately: deforestation emits 3.8 billion tons of carbon dioxide equivalent per year and agriculture is estimated to be the driver of 80% of deforestation worldwide (Kissinger et al., 2012). Land use changes, combined with modern production techniques and the use of chemical inputs, cause severe biodiversity losses: 21% of the species in the International Union for the Conservation of Nature (IUCN) Red List are threatened by agriculture,[3] with sharp declines observed in birds, butterflies and pollinator insects, among others (EEA, 2010).

1 E.g., pesticide use more than tripled in the United States between 1960 and 1981, and herbicide use increased more than tenfold. Although after 1981, the quantities applied decreased, partly due to the introduction of new pesticides and insect resistant crops, pesticide use doubled between 1960 and 2008. However, the quantity that would have been applied had pesticide quality has remained constant over this period have been multiplied by 4.5.(Fernandez-Cornejo et al., 2014).

2 Agriculture emits between 5.2 bn and 5.8 bn tons of carbon dioxide equivalent per year.

3 The aim of the IUCN Red is to catalogue and highlight those plants and animals that are facing a higher risk of global extinction.

These environmental externalities are becoming limiting production factors and are engendering large economic and social costs. The costs generated by air and water pollution caused by nitrogen are now offsetting the benefits from the use of synthetic nitrogen fertilizers (Sutton et al., 2011b). The decline in pollinator species is causing agricultural production losses that have been estimated at several billion euros (Gallai et al., 2009, Allsopp et al., 2008).

To tackle the negative environmental impacts of agriculture, governments are using both quantity and price as planning instruments. Policies targeting prices can take the form of taxes on environmental damaging inputs, or subsidies for environmentally friendly measures while restrictions on specific inputs and requirements related to best-management practices are aimed at regulating quantities. These kinds of provisions are being included increasingly in agricultural policies. Since the 1990s, the European Union has issued several legally binding directives on habitat conservation, nitrogen use and pesticides while at the same time compensating farmers through the so-called agri-environmental schemes, for voluntary actions to protect the environment that go beyond standard practices. Under the 2014 reform, more than one-third of the European Common Agricultural Policy budget is dedicated to agri-environment-climate payments (EC, 2014). In the United States, in 2014, about 28% of the Farm Bill budget directly dedicated to farms (nutrition assistance excluded) is devoted to conservation programmes that finance measures to preserve or restore public and private land (USDA, 2013). In Brazil, a mix of public policy, monitoring systems and supply chain interventions has managed to slow the advance of the agricultural frontier in the Amazon forest by 70% between 2005 and 2013 (Nepstad et al., 2014, Macedo et al., 2012).

Since the 1960s, agriculture has also experienced the effects of globalisation. The share of agricultural and food production traded internationally rose from around one-ninth to some one-sixth of global production between 1961 and 2004, thanks to the decline in the costs of cross-border trade (Anderson, 2010). Agricultural trade has been affected by government policies: since the 1950s, agriculture has been highly protected in developed countries and heavily taxed in developing countries, and both groups of countries have used trade policies to stabilise their domestic markets. This has led to over-production in high income countries and to under-production in developing countries and, probably, less trade activity than would have occurred under free trade conditions. Nevertheless, during the past 25 years, both developed and developing countries have undertaken reform of their trade and price policies, resulting in the increased trade mentioned above. Current international agricultural markets, at least for some crops, are well developed and connected to other markets.

The coexistence of these two trends (environment degradation and globalisation), not specific to agriculture, has triggered many questions and much debate since the beginnings of the 1990s. Concerns have emerged also in political debates, such as those surrounding the North American Free Trade Agreement where the impact on Mexico's environment became a stumbling block, and the 1992 United Nations Conference on Environment and Development held in Rio de Janeiro. The economic literature on trade and the environment is rich and addresses mainly two categories of questions (Gallagher, 2008). One strand includes analyses of how trade affects the environment, distinguishing between direct and indirect effects. However, the direct effects have received less attention, and focus mainly on the introduction of invasive species and global shipping emissions. Indirect effects occur through three main channels: scale, composition and technique (Grossman and Krueger, 1991). Scale effects occur when trade increases the quantities produced, with resource depletion possibly related to these quantities. Trade can also change the mix of economic activities in a country, and generate composition effects if the new mix pollutes more or less. Changes in production technologies result in technique effects if the new technologies have a different impact on the environment compared to the old ones. Another literature strand focuses on the way environmental and trade policies interact. One of its main concerns is the way that increased economic activity induced by international trade can affect environmental policies. In other words, it considers the endogeneity of policies. Environmental policies may react to rising incomes and changing prices, which means that if trade changes incomes, this in turn changes environmental policies (Copeland and Taylor, 2004). Environmental policies can also affect competitiveness through taxes or restrictions. Therefore, they can be used strategically to gain market shares, in which case trade can result in ecological dumping.

I.2 Approach

The research described in this thesis contributes to the literature on trade and the environment by focusing specifically on the agricultural sector; most work in this area examines the manufacturing industries. Efforts have to be made to limit agricultural damage to the environment. Agriculture is one of the main sources of pollution and loss of biodiversity and its production must become both more sustainable and increasing. In parallel, the cost-effectiveness of environmental measures is one of the highest in agriculture, in particular in the context of climate change (Golub et al., 2009, Vermont and De Cara, 2010, De Cara and Jayet, 2011). The mechanisms linking agriculture to the environment and, therefore, agricultural trade to the environment, are more complex than the mechanisms involved in manufactured products because of the high dependence of agriculture on natural resources. This complexity is large enough to modify some standard results and, therefore, deserves some attention.

This thesis deals mainly with how environmental policies are affected by agricultural trade. Two different policies are studied: promotion of sustainable production techniques (Chapter II) and taxation on pesticides (Chapter III), and the underlying mechanisms (Chapter IV). The first part of the thesis considers a unilateral environmental policy and discusses the resulting leakages. What are the global indirect effects of a regional environmental policy on agriculture that is applied unilaterally? Do these indirect effects offset local positive impacts? The second part of the thesis analyses the case of bilateral policies, focusing on the strategic aspects of environmental policy in a trade context. Since reducing the stringency of environmental regulation could lead to increased competitiveness, it examines whether environmental policies applied to agriculture are more lenient under free trade.

I.2.1 Theoretical and empirical tools for global modelling

The trade literature considers agriculture mainly through institutional descriptions and empirical analysis, rather than new theoretical models (Karp and Perloff, 2002), and is linked closely to policy making. In line with this tradition, Chapter II develops an empirical analysis of the impacts of an agricultural policy aimed at improving agricultural sustainability. The policy of interest, a significant shift in European agriculture towards organic farming, is on the current policy agenda, and our main findings are relevant for policy-making. We attempt to gauge the extent to which local environmental benefits are offset by indirect effects in third countries. We first measure induced land use changes and derive some conclusions based on various criteria. Measuring land use changes requires an assessment of the changes in prices and the induced displacements of supply and demand in different regions. We use a specific version of a global dynamic general equilibrium model with an improved representation of land use (MIRAGE-BioF, Bouët et al., 2010, Laborde, 2011). We introduce a formal representation of organic production and a specific demand structure for organic products. The modelling of organic technology is grounded in micro data, which confirm that EU organic production yields are still lower than those obtained from conventional crops under actual farm conditions. After assessing land use changes in all agro-ecological zones at a global level, we deduce the corresponding changes in greenhouse gas emissions, fertiliser use and biodiversity, and try to find metrics to balance these indirect effects with the local benefits of greener European agriculture.

Another empirical tool, which relies on statistics and econometrics, is used in Chapter IV, not to analyse policy impacts but rather to explore one of the particular relationships between agriculture and the environment. We contribute to the literature by exploring the effects of crop biodiversity on productivity and farmers' exposure to risks, using a new database and grounding our approach in a theoretical model. The mechanisms at stake are identified and detailed, and investigated using a statistical approach.

However, empirical tools are not sufficient to study the question of agricultural environmental policies under trade. The interactions between agriculture and the environment are sufficiently distinctive to change the regular mechanisms linking trade and the environment, in which case, theoretical models are needed to understand these new relationships. In particular, the effects of crop biodiversity change the standard features of a trade model adding a new dimension to the composition effects of trade on the environment. Composition effects can have environmental impacts when the economy moves towards more (or less) polluting sectors (crops in the case of agriculture) because of the specialisation induced by trade and comparative advantages. Apart from the pollution intensity of crops, specialisation reduces crop biodiversity, inducing farmers to increase the quantities of pesticides used to combat the increased pest attacks. Then, even with a move towards cleaner crops, composition effects can result in environmental degradation. Chapter III uses a Ricardian standard trade model to explore these effects, taking account of crop biodiversity effects in production functions. These effects generate trade frictions, which are not new in trade models. What is new in our setup is that frictions are endogenous and depend on the cultivation intensity of crops. We derive some original conclusions on environmental policies, patterns of trade and production distributions.

I.2.2 A focus on the specific links between agriculture and environment

The contributions of this research are based mainly on a precise description of the way agriculture and the environment interact, and the impact of trade depending on this interaction. This involves an overlap between economic science and ecology and agronomy.

Chapter II relies on agronomy. We look at sustainable production techniques and organic farming in particular. Organic farming does not allow the use of synthetic fertilisers, pesticides and drugs, stocking rates are limited, and the feed purchases must obey strict rules. This results in a quite different production system from the conventional one. Soil fertility, nitrogen self-sufficiency and pest control are dealt with by crop rotation, natural predators, recycling of organic materials and different use of mechanical intervention. Thus, conversion to organic farming is more than a simple adaptation of previous practices, and often implies strategic redesign of the production system relying on new techniques and inputs (Stockdale et al., 2001). In economic terms, shifting from conventional to organic farming is not simply a matter of decreasing total factor productivity, it is rather a heterogeneous shock to factor productivity. It is not a matter of distance from the production frontier, but rather a matter of a different frontier (Mayen et al., 2010). Chapter II uses the shift to organic farming as an example of a unilaterally applied environmental policy and analyses resulting leakages. If this shift were considered simply as a decrease in total factor productivity rather than a change in production technology, the consequences for supply and, therefore, for the environment would probably be overestimated.

Chapter IV and III rely on a very robust stylised fact in ecology that the more diverse the cultivated crops, the lower the frequency of pest attacks. The underlying idea is that the greater the area dedicated to the same crop, the more pests will specialise on this crop (Pianka, 2011).[4] Agronomists and ecologists describe these mechanisms in detail, but economists have taken only scant account of their consequences for the way farmers manage productivity and production risks while decreasing their use of pesticides, and how trade and the environment interact. We investigate the issue of crop biodiversity and productivity empirically, and we detail the mechanisms at stake when considering environmental policies linked to crop biodiversity under trade using a theoretical trade model.

[4] This approach relies on island biogeography, where the crop is a sort of island for its specific parasites.

I.3 Organisation of the thesis

Chapter II: indirect effects of a greener European agriculture In the face of difficulties related to cooperation and formulation of international policy, countries may purse environmental policies unilaterally, which can result in leakages. The mechanisms underlying leakages have been studied in depth in relation to some policies, particularly taxes on greenhouse gas emissions (*inter alia* Monjon and Quirion, 2010, Dong and Walley, 2012, Fischer and Fox, 2012, Antimiani et al., 2013) and support for production of biofuels (*inter alia* Searchinger et al., 2008, Taheripour et al., 2010, Plevin et al., 2010, Laborde, 2011)), but totally ignored in relation to others such as the greening of agricultural production systems. Many recent agricultural policy reforms have tried to promote more environmentally friendly production techniques including organic farming. Organic farming is known for its local positive impacts on the environment, but it may also have indirect global impacts. Since organic yields are on average 25% lower than conventional yields (Seufert et al., 2012), a major shift to organic farming could result in a negative productivity shock, and market driven forces might generate incentives to produce more intensively in regions other than the one promoting organic farming. This could lead to new land being put under cultivation, including land in high natural value areas. If this occurs, do local benefits offset indirect negative global effects? We illustrate these possible effects by considering a partial, but significant shift in European agriculture to organic production. We represent organic production and demand formally in a global general equilibrium model with improved representation of land use. Land use changes and corresponding changes to greenhouse gas emissions, chemical inputs and biodiversity are assessed. The indirect negative effects seem generally to be limited, but this is not the case for greenhouse gas emissions. The aim of this research is not to discredit organic production techniques: considering the recent numbers on yield gaps, it is legitimate to investigate this question and especially since other environmental policies have been the subject of similar investigations. Furthermore, it is important to understand the key drivers of the mechanisms at stake in order to formulate policies that reduce indirect effects while enhancing local benefits.

Chapter III: agricultural trade and crop biodiversity effects The literature on trade and the environment mainly considers wildlife biodiversity. This is surprising since crop biodiversity also plays a key role in environment and agricultural production. On the one hand, it is a reservoir of genes where plant breeders can find traits useful for future selections.[5] On the other hand, crop diversity allows agro-ecosystem productivity to be maintained over a wide range of conditions (Tilman et al., 1996). In particular, crop biodiversity has an impact on pest activity and, consequently, on production levels, through the mechanisms described above. Chapter III analyses the interaction between crop biodiversity effects described in Chapter IV, and trade. As

[5] Part of the debate on the environmental impacts of the North Atlantic Free Trade Association (NAFTA) for Mexico, was on the potential loss of maize diversity. Mexico is the domestication region for maize, and Mexican farmers play a key role in maintaining the genetic diversity needed by future maize breeders.

soon as free trade is considered, a trade-off appears: is it better to specialise in more productive crops, gain from trade but lose crop biodiversity and face some production losses caused by pest attacks, or to maintain some crop biodiversity while losing competitiveness? To face pest attacks, farmers use pesticides. However, pesticides are harmful to the environment and human health and, therefore, are regulated by governments. Decision-makers face a second a trade-off: do they preserve the environment and consumers' health and accept loss of market share, or do they accept some environmental damage in order to be the most competitive on international markets? This question of the strategic use of environmental policy in the specific context of agriculture, is investigated using a theoretical Ricardian, two country and many goods setup, *à la* Dornbusch et al. (1977). Standard results on strategic environmental policies under trade are affected when crop biodiversity effects are taken into account. Once the mechanisms underlying the interaction between crop diversity, production level, environmental and trade policies are made explicit, some conclusions can be drawn about the impacts on food price distribution.

Chapter IV: estimating the impact of crop biodiversity on agricultural productivity
Modern production techniques rely almost exclusively on genetically homogeneous populations, which are easier to manage since synthetic inputs disconnect agricultural systems from the variability of the environment. However, if the aim is to reduce the use of synthetic inputs to reduce environmental impacts, genetic diversity could be more to the forefront. This requires precise quantification of the impacts of crop biodiversity on agricultural productivity and on farmers' exposure to risk. Agronomists and biologists have detailed the interactions between plants and pests mainly at the individual level or field level. Chapter IV describes the fundamental mechanisms on which Chapter III relies. We contribute to the small body of economic literature on the subject (Smale et al., 1998, Di Falco and Chavas, 2006, 2009) by testing these interactions at the level of a country, South Africa. The dataset is original, very large and reliable since it is based on satellite imagery not local surveys. Its geographical scale allows investigation of spatial questions such as the size of the relevant perimeter to the biodiversity being considered. First, we build a theoretical model to analyse in detail the mechanisms linking biodiversity and agricultural productivity and their impacts on production levels and variability. Second, we quantify these effects using a reduced form approach. We confirm the significant and positive impact of crop biodiversity on productivity and on its variance.

CHAPTER II

HOW GREEN IS ORGANIC? INDIRECT ENVIRONMENTAL EFFECTS OF MAKING EU AGRICULTURE GREENER

While making it possible to feed the EU population, the production techniques adopted in European agriculture for the last fifty years have generated a number of negative externalities (EEA, 2010). In particular, agriculture is responsible for a large share of the pollution of European surface water, aquifers and coastal seas by nutrients and pesticide residues (EEA, 2012). The combination of modern production techniques, the use of chemical inputs, the draining of wetlands and the uprooting of hedges has led to dramatic erosion in farmland biodiversity, as testified by the sharp decline observed in farm bird and butterfly populations (EBCC, 2012).[1] Environmental damage is such that the long term sustainability of modern agriculture is now questioned, given the risk that soil erosion and compaction as well as the decline in pollinators become limiting production factors (Jones et al., 2012, Klein et al., 2007, Bauer and Wing, 2010).

These externalities now generate large economic costs. For example, estimates suggest that the social cost of nitrogen pollution could now match the actual economic contribution of fertilizers to agricultural output (Sutton et al., 2011a,b). Damages to populations of pollinating insects and to those species that control for pests (i.e. ladybugs, bats, birds) have now begun to cause production losses that are estimated to reach billions euros (Gallai et al., 2009, Allsopp et al., 2008, Boyles et al., 2011, Sumner and Boriss, 2006). Clearly, something must be done to make EU agricultural production more sustainable.

EU authorities have progressively included provisions to alleviate the negative environmental impact of agriculture in the Common Agricultural Policy (CAP). Because implementing the polluter-

Chapter written with Jean-Christophe Bureau.

[1] From 1980 common farmland bird populations fell by 50% in 30 years (EEA, 2010). The decline had slown down between the mid 1990s and mid 2000s - probably because of the compulsory set-aside of land between 1993 and 2008 that was part of the Common Agricultural Policy - but the decline seems to have resumed since 2008 (Jiguet et al., 2012). The population of some rather "common" species in grain growing areas (linett, common quail, skylark, etc.) has decreased by 30 to 70% in the last 20 years in a country like France. Figures are higher for species whose population was already low (e.g. 90% for the little bustard). Grassland butterfly populations have decreased by 60% since 1990 and the decline shows no sign of slowing down in the EU.

pays principle is difficult in the sector, characterized by both non point source pollution and a strong aversion of farmers to taxes, the EU has opted for a mix of regulatory measures and incentives. The former include legally binding directives on nitrate pollution and on habitats. However, member states tend to apply regulatory measures in a lenient way (e.g. nitrate and water directives). Incentives based measures include payments for voluntary adoption of environmentally friendly techniques, the Agri-Environmental Schemes (AES). AES are designed to compensate farmers for voluntary actions protecting the environment that go beyond standard practice. While these measures have had some success in protecting particular areas, the lack of zeal to implement them in some member states (which have to cofinance them) and their use as a way to subsidize farmers more than protect the environment in others, have limited their environmental impact. Because they require designing precise terms of reference, inspecting and controlling compliance, AES involve significant transaction costs. One cause is the asymmetric information on the level and cost of environmental effort, which generates informational rents and, in some cases, moral hazard. Overall, the cost-effectiveness of AES has been found to be limited (ECA, 2011).

Those CAP subsidies that provided direct incentives to produce intensively have progressively been replaced by a more production neutral Single Farm Payment (SFP). The latter has been made conditional to good environmental practices in addition to standard environmental legislation. However, the conditions attached to the SFP have remained lenient and hardly go beyond the respect of standard legislation. In addition, after 2006, the impact of these reforms has been dampened by the higher prices for agricultural products, driven by a growing demand, caused by population growth, change in diets and the increasing use of feedstocks for biofuels. High prices provide incentives to produce more intensively and to abandon voluntary conservation programs. As a result, the use of both pesticide and nitrogen, which were on decreasing trend in the 1990s and 2000s, have recently bounced back; and the decline in biodiversity has resumed at an alarming pace over the most recent years (Jiguet et al., 2012).

Under the 2014 reform, a third of direct payments (SFP) was converted into a "green payment" conditional to specific requirements, e.g. crop rotation obligations, a 5% limit to the loss of permanent grasslands, preservation of natural pastures and Ecological Focus Areas (EFAs), the latter intending to keep 5% of arable land coverage and in conditions favorable to preservation of biodiversity (possibly 7% in the future) (Matthews, 2013). However, the requirements have been considerably watered down by successive amendments by the European Parliament and Council. Because many farmers are now exempted, and because the measures have been made more flexible (from the possibility to grow nitrogen fixing crops on EFAs to the recognition of potentially lenient national certification schemes as equivalent to EFAs), observers believe that the regulation will have very limited impact and that most EU farmers will qualify for the green payments without changing anything to their routine (Pe'er et al., 2014). The new framework nevertheless allows for a large subsidiarity and those member states that want to be serious about greening their agricultural policy can use many options to do so.[2]

Many options have been proposed to make EU agriculture environmentally friendlier. Here, we focus on the conversion of a share of EU agricultural production to organic techniques. Encouraging organic farming is increasingly seen by environmental groups as a way to overcome the poor record of the "greening" of the CAP direct payments. Organic agriculture, which partly funds itself through higher output prices, is also seen as a cost effective environmental measure by governments. Initiatives to encourage organic production[3] have resulted in a rapid expansion of organic agriculture in the EU. The EU-27 area under organic farming (including fully converted and under conversion areas) increased by 48.5% between 2005 and 2011, with an average annual increase by 6.8%. This is a rapid growth, even if organic production accounted only for 5.5% of the EU total used area in 2011.[4]

A move to organic production in EU agriculture would have several benefits. It is well-accepted that the requirements for organic certification correspond to genuine environment-friendly farming practices (EC, 2004, EEA and UNEP, 2007). Organic production was found to result in higher soil organic matter content, less nitrogen and phosphorus losses, less N_2O emissions, lower eutrophication potential, and a positive impact on biodiversity (Tuomisto et al., 2012, Mondelaers et al., 2009, Schneider et al., 2014). Organic certification is governed by clear, verifiable rules that leave little place for moral hazard, unlike the *ad hoc* definition of terms of reference for AES schemes that often lead to windfall gains and information rents. Certification procedures have been successfully tested over several decades and work well.[5] In addition, the terms of reference include provisions that match the demand for a whole bundle of attributes, from pesticide free food, to nitrogen runoff controlled production and animal welfare requirements. This bunching of attributes makes the organic certification attractive to a large set of consumers who have a variety of environmental, health and ethical concerns.

However, at a large scale, organic production might have some undesirable effects which have so far been poorly documented. In particular, there is a risk of global market effects that can be paralleled with the well-known carbon leakage, i.e. the risk that local regulations induce production displacement and more pollution abroad, resulting in a a less positive global environmental balance than expected. Clearly, these indirect effects need to be assessed and, if they are significant, accounted for. A major issue is whether indirect Greenhouse Gas (hereafter GHG) emissions,

2 Member state policies show contrasted options, with Wales using almost all possible options to make the agricultural policy greener, while Slovakia doing the opposite, for example.

3 Even if the 2004 EU action plan for organic farming has not led to setting a EU-wide figure, many member states have set national targets. France set a target of 20% of arable land grown organic in 2020, even though it is unlikely to match this goal; Austria set a 20% target in 2013, Slovenia in 2015. Ireland set less ambitious targets for 2012. See the European Commission funded research project "Development of criteria and procedures for the evaluation of the European Action Plan for Organic Agriculture", http//:www.orgap.org.

4 Authors' calculations based on figures from Eurostat database on "certified organic crop area by crop products"(food_in_porg1), retrieved on June 2014.

5 Organic production is covered by EU Regulations EC/834/2007 and EC/967/2008 that lay out rules on organic farming practices.

biodiversity losses and water pollution in other regions offset some of the local benefits of converting EU agriculture to organic production.

II.1 The mechanisms at stake

II.1.1 dLUC and iLUC effects

The literature on the indirect effects of making European agriculture environmentally friendlier is limited. European Commission's impact assessments of making agricultural policy greener tend to focus on the intra-EU environmental effects (EC, 2011). Lambin and Meyfroidt (2011) have provided useful insights on global cascade effects of local land use changes. Cantore (2012) has focused particularly on changes in EU agricultural policies. Some authors have recently attempted to measure indirect GHG impact of land set-aside in the EU (Pelikan et al., 2014). In spite of these pioneering efforts, no full assessment of the indirect environmental effects of changes in EU agricultural policy has yet been completed. However, an extensive body of literature has been developed to investigate the indirect energy balance and GHG emissions of biofuels. We build on this literature to assess indirect effects of organic agriculture.

Since the pioneering work of Searchinger et al. (2008), it has been argued that indirect effects should be taken into account when assessing the GHG impact of substituting biofuels for fossil fuels. This involves assessing changes in the use of land driven by biofuel production. In particular, several approaches have been developed to identify the way EU and U.S. biofuel policies modify price vectors and equilibria in agricultural and food markets (Taheripour et al., 2010, Havlìk et al., 2011, Laborde, 2011). Indeed, since the mid 2000s, EU policies have encouraged the use of biodiesel (made from vegetable oil) and ethanol (made from corn, wheat and sugar beets) through tax rebates and blending mandates. To illustrate the LUC effects, consider the case of biodiesel. Public support to biodiesel has boosted demand for rapeseed and sunflower oil from the energy sector, resulting in changes in prices, supply and land use. One typically distinguishes direct LUC effects (dLUC) and indirect ones (iLUC). A dLUC corresponds to the replacement *ceteris paribus* of one hectare of (say) cereals by one hectare of rapeseed for biodiesel production in the EU. However, this is not the only effect taking place. Other indirect LUCs occur as the result of the impact of biofuels expansion on prices. Indeed, the channeling of EU rapeseed in the energy market induces changes in relative prices and substitutions both on the supply and demand sides. EU consumers substitute other vegetable oils for rapeseed oil, that has become more expensive. The EU food industry typically imports more palm, soybean and canola oil. The demand for such oils spills across markets and countries and induces extra supply. In some cases this lead to expand agricultural production in the EU but also overseas, including in previously uncultivated areas such as savanahs and forests. For example, greater use of biofuels in the EU may have contributed to deforestation for growing crops (e.g. soybean in Brazil), to the draining of peatlands (e.g. palm oil in Indonesia) or the ploughing of natural pastures (e.g. soybean in Urugay, canola in Canada). This may have indirect consequences in terms of GHG emissions, and potentially offset part of the GHG savings generated by using biodiesel rather than fossil fuel in the EU. The magnitude of these effects is nevertheless still very controversial (De Cara et al., 2012). One reason is that the different indirect effects are complex and sometimes opposite. For example, blending mandates that have led to higher biodiesel consumption have led to EU to import more palm oil and soybean oil. But EU biodiesel production made more rapeseed cake (a byproduct of biodiesel) available for the livestock sector, hence contributing to lower imports of soybean cakes from South America.

The mechanisms described in the case of biofuels are not limited to the channeling of feedstocks to the energy market. Should the development of organic production in Europe result in land use changes, similar mechanisms might be at work. This is particularly the case if a shift to organic production results in a lower output per hectare in the EU. In such a case, a large shift of EU agriculture towards organic production could lead to cross-market and cross-country land displacement effects. If a consequence is to bring some previously uncultivated land into production, the shift to more organic production in the EU could have negative side effects in other countries, in particular in the area of biodiversity loss, pesticide and fertilizer pollution and GHG emission. In theory, it is even possible that the decline in EU production is such that it requires importing food from countries that expand their agricultural production by deforesting or turning into industrial plantations some biodiversity rich habitats.

Such harmful consequences are not inevitable, though. There is still a large scope for improvement in organic yields, for example, since research in organic agriculture (and more generally agroecology and other low chemical inputs technolgies) has been poorly financed during the last decades. Price changes may drive some substitutions that dampen the impact of dLUC and iLUC. Some rebound effects may also offset some of the dLUCs and iLUCs. Assume, for example, that price effects drive shifts to more vegetarian diets, requiring less land to grow feedstuffs. Indirect negative environmental effects would be small. The environmental effect of land displacement could also be limited if expansion of production took place on unused degraded land rather than high natural value areas. In brief, the overall consequences are therefore ambiguous and there is a need for quantification of the various mechanisms at stake.

II.1.2 Key determinants of indirect externalities

A key variable in the mechanism described above is the change in yields that would take place in the EU. If yields decrease significantly following a conversion to organic agriculture, the Net Displacement Factor (NDF)[6] will be large, hence potentially large indirect externalities (GHG emission, biodiversity erosion, water pollution and depletion, etc.). There is evidence that organic agriculture can reach high yields. This has been observed in experimental conditions and in traditional forms of agriculture (e.g. Caribbean or East Asian traditional agrosystems). However,

[6] The NDF is the ratio of hectares of (a) land brought into crop production anywhere in the world to replace land used for organic crops to (b) hectares dedicated to additional organic crops (Plevin et al., 2010).

high yield organic agriculture often requires a large amount of labor. Given the relatively high cost of labor that prevails in most EU member states, in practice, a shift to organic production is likely to results in lower aggregate yields, as shown by several estimates.[7] Note, however, that while the gap yield observed in farm surveys is large, this may also be due to external factors, reflecting for example that organic farms are located in less fertile areas.

Another key variable is the spatial allocation of organic production. This issue is linked to the policy instrument used. For instance, assume that EU governments set global targets (e.g. a compulsory acreage in organic agriculture or a compulsory ratio of agricultural land grown organic). Farmers will first convert land to organic farming in areas where both the yield gap and the costs of conversion are lower. If most of the shift to organic takes place on already extensive grazing areas or on traditional orchards, vineyards or olive plantations, the production impact will be limited. So would be the iLUC effects. The shift to organic production may also be larger in those sectors where the price gap between organic and standard product is wider, e.g. fruits rather than rapeseed. Obviously, if other instruments such as subsidies, are used to reach certain organic targets in particular regions, the land allocation, price and supply response effects will be different. So will be the dLUC and iLUC, as well as the related externalities.

Supply response of the non organic technology is also a key determinant of indirect externalities. If some of agricultural output is converted to organic, the resulting changes in prices may provide incentive to intensify production on those fields that are left to non-organic production.[8] A consequence might be more pollution in the non-organic areas, but a lower iLUC effect, meaning that a trade-off between local and foreign environmental damage exists.

Finally, there is a potential trade-off between land use changes inside and outside EU, depending on the ability to expand agricultural land in EU member states. The expansion of agricultural areas as a response to environmental measures has been found to be significant in some cases.[9] There is little arable land that has not been cultivated in the EU. However, in some new member states, some agricultural land was abandoned in the 2000s. Putting it back into production may

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[7] The larger amount of labor per hectare in organic farms is observed in most member states, Spain being an exception. This may nevertheless reflect smaller structures. Lower yields are also observed by EC (2013), which finds that wheat yields in organic farms are less than half of those in conventional farms in Germany and France. Overall, Seufert et al. (2012) find in their meta-analysis an average organic-to-conventional yield ratio of 0.75 (0.89 for oilseeds, 0.74 for cereals, 0.63 for vegetables). The meta-analysis conducted by Tuomisto et al. (2012) finds a similar aggregate figure, with a standard deviation of 17%. The largest yield gap with conventional production is for winter wheat (yield ratio of 0.62), but the gap is lower for some productions (e.g. vegetables). Ponti et al. (2012) find that organic yields are on average 80% of conventional yields, with possible substantial variations (standard deviation of 21%). Niggli et al. (2008) find large gaps for yields in wheat in countries where production is more fertilizer intensive (organic yield for wheat reaches on average only 50% of the conventional yield in France but 88% in Italy). They also find large gaps for potatoes in Austria and Germany. They find lower yields gaps for barley, oilseeds and pulses across Europe.

[8] When mandatory set-aside was introduced in the early 1990s in the EU, production became more capital and chemical input intensive in the land that was not left idle, so that the overall output reduction of the set-aside was lower than initially expected (Herlihy and Madell, 1994)

[9] Wu et al. (2001) show that one of the consequences of price changes induced by the U.S. conservation reserve program is to put in production additional amounts on land that was idle. Wu (2000) estimates that about 20 acres were brought into crop production for every 100 put out of production under the program.

limit some of the iLUC effects in non-EU countries. The trade-off in the localisation of the impacts of land use changes, especially on biodiversity, also depends on the ability of countries to build and enforce regulations to protect high natural value areas.

II.2 Methodology

A significant shift of EU agriculture to organic production involves several general equilibrium effects. First, if partial productivity of land goes down, this will affect supply and demand in interrelated markets, with substitutions between commodities. Second, the changes in demand for intermediate consumption (fertilizers and pesticides) and other production factors (land, labor, capital) have price effects that spill over the entire economy. Third, the change in prices between organic and conventional products induces changes in relative demand. Overall, changes in relative prices, production and trade depend on substitution and complementarity between inputs (organic agriculture could require less treatments but more mechanical intervention e.g. for weeding, for example), between outputs and within final consumption. These effects must be taken into account, and one should consider the new economic equilibrium in order to carry out a comparison of environmental effects relative to the initial equilibrium. This is the spirit of what is sometimes called a "systemic" or "system-level" Life Cycle Analysis (LCA) approach (Hendrickson et al., 2006). A way to do so is to construct a General Equilibrium (GE) model that makes it possible to assess the impact of a sectoral EU policy on other markets and in third countries, accounting for a series of domino effects. Even though they require a number of simplifying assumptions, GE approaches are useful if one wants to compare a reference situation to a counterfactual one where the EU would have converted part of its agriculture into organic. GE modeling makes it possible to account for market adjustments, supply and demand reactions in the different parts of the world. These important issues cannot be dealt with using traditional LCA.

Simulations rely on the framework developed under MIRAGE, a global multi-country, multi-sector recursive dynamic general equilibrium model (Bchir et al., 2002, Decreux and Valin, 2007, Laborde and Valin, 2012). Building on this framework, a specific version that distinguishes land allocation (MIRAGE-BioF) was developed to assess the LUC impacts of some specific policies, in particular the biofuel policies, by a team of modellers from various institutions.[10] The representation of land market relies on a nested structure of constant elasticities of transformation (CET), so as to account for the various possibilities of land substitution and expansion in each agro-ecological zone within each country. Land is classified as managed (cropland, pasture, managed forest), with an economic return, or as unmanaged. When the price of a crop increases, more land is allocated to this crop. The new land is taken from the other categories of managed land, according to their

[10] See Laborde and Valin (2012) for details. David Laborde and Antoine Bouët at the International Food Policy Research Institute and Hugo Valin, at the time at the French Institut National de la Recherche Agronomique and then at the International Institute for Applied System Analysis, were the main contributors and kindly allowed us to build on their work.

price, following the CET specifications. When the possibilities of substitution become smaller, cropland prices increase and some land expansion occurs. This expansion is endogenously calculated according to both an historical trend (to track non agricultural price driven land use changes such as urbanization) and an elasticitity of land expansion. This elasticity decreases linearly: the more land is cultivated, the lower is the elasticity. Marginal productivity of newly cultivated land is lower than the productivity of initial cropland. Yield responds to an increase in the use of fertilizers, with a maximum yield set for each crop and an asymptotic convergence with this cap.[11] Apart from land and agricultural markets, a detailed database has been built, introducing new sectors and re-calibrating production technologies in order to ensure the consistency between values and volumes. This is of particular importance for our simulations, since physical linkages and substitutions play a critical role in the assessment of demand displacements and land use change.

Building on the land use version of MIRAGE (MIRAGE-BioF), we constructed specific demand and supply systems to explicitly represent the organic agricultural sector.[12] Organic technology was introduced for selected arable crops. This required constructing a consistent set of social accounts that makes possible to represent the economic situation of organic products and the linkages with the production and consumption of non organic products, as well as the linkages with the other sectors. Because of data availability, organic technology was introduced only for maize, rapeseed, sunflower and wheat in the model. We lacked reliable data for a satisfactory calibration of organic technology in fruits, vegetables and other crop sectors as well as in the grassfed livestock sector.[13] That is, we assume here our scenario regarding a shift to organic only impacts the livestock sector through the consumption of organic cereals and protein concentrates.

Characterization of the production technology in the organic sector runs into the lack of consensus regarding the yield gap with conventional agriculture. While some studies show that in experimental conditions it is possible for organic farming to reach yields that are similar to conventional production, it often depends on the ability to use more labor. And for certain agricultural productions, giving up chemical inputs (often used as a way to reduce production risk) generates a more variable output. In a dynamic framework, uncertainty also prevails on the ability of improving yields in organic production. We calibrated the production technology and the yield changes when shifting from standard to organic production on microeconomic data. We used data from the European Farm Accountancy Data Network (FADN), which includes an identifier variable for organic farms (starting in 2000) (Offermann and Lampkin, 2005).[14] The average organic to conventional ratio

[11] For more details on yield responses, land extension and expansion in MIRAGE-BioF, see Al-Riffai et al. (2010)

[12] Note that the use of a specific representation of the organic production technology, instead of a shock on factor productivity, is a key specification here. Indeed, it has been shown that the organic and conventional production technologies are not homogeneous (Mayen et al., 2010).

[13] In the European Union, grassland, arable crops and permanent crops represent respectively 44%, 42% and 10% of the total organic area (Lernoud and Willer, 2014). 15% of the total organic area is devoted to cereals (mainly wheat, oats and barley) and 1.5% to oilseeds (mainly sunflower) (Bio, 2013). The main products sold on organic market, according to their value, are fruits and vegetables, followed by milk and dairy products and bread and bakery products (Schaack et al., 2014).

[14] The Farm Accountancy Data Nerwork is a system of sample surveys conducted yearly to collect data on European farms. Surveys cover only farms exceeding a minimum economic size since the aim of the FADN is to register

for the arable crops resulting from these data and included in the simulation is 0.68, i.e. lower than the average value of 0.74 found by Seufert et al. (2012) for cereals. This figure of 0.68 may reflect spurious correlations. Organic farms tend to be located in less fertile areas. There may be a self-selection procedure in shifting to organic where yields are already low. The various conditions for getting subsidies under the EU less-favored areas and rural development programs, which many organic farms are eligible to, might also act as a yield limiting factors, regardless of the technology (EC, 2013). In Spain, for example, where organic production tends to take place on large farms in plains, the yield gap in the wheat production is smaller than in other EU member states. The figure of 0.68 should be considered as a lower bound if one wants to assess yield differences due only to the technology. We therefore carry out simulations with alternative assumptions on the yield gap (provided in the appendix). However, we chose to stick with what micro-data tells us in our core simulation.

To calibrate the area dedicated to organic crops, we used data from Eurostat, which reports areas under organic practices for different crops in each member state. The land supply representation of MIRAGE-Biof was amended to differentiate between organic and conventional land for each crop, adding an extra substitution level in the nested structure. In our calibration of the demand system, organic and conventional crop productions are aggregated in a virtual good that is used by other sectors, by final domestic consumers and by foreign demand. Few econometric studies deal with the elasticity of demand for organic products, and they focus on very specific products aimed at final consumption (Thompson, 1998, Glaser and Thompson, 1999, 2000, Wier et al., 2001, Lin et al., 2009). All studies find higher own-price elasticities for organic products than for conventional products (1.3 to 12 times larger according to products and studies). In the absence of precise estimates, we converted this differential in own-price elasticities between organic and conventional products by using a larger substitution elasticity between organic and conventional products for the same commodity than between different commodities. Because we used elasticities of substitution set somewhat arbitrarily larger (5 times larger), we carry out sensitivity analysis on the simulations, reported in the appendix.

II.2.1 Simulations

Starting from the reference year 2008, we implement a baseline scenario (hereafter *BASE*), extended through 2020, in which we reproduce the present rate of adoption of organic farming in the European maize, sunflower, rapeseed and wheat sectors.[15] Social accounting matrices are calibrated on 2004

activities of professional farms in each member state. This can induce a bias in results on organic farms. Most importantly, FADN samples are built to be representative of different European regions, of different economic sizes and of different types of farming, but not of organic farming in particular. Then, samples of organic farms are not stable across time and are sometimes too small to allow the publication of results. Therefore data on organic farming have to be used and analysed with care. Nevertheless, FADN is the only source of harmonized micro-economic agricultural data on organic farms across European Union.

[15] In 2011, in the European Union, the share of organic area in the total cultivated area of maize, rapeseed, sunflower and wheat were respectively 0.8%, 0.4%, 1.5% and 1.6% (authors' calculations based on Eurostat data)

data, and the year 2008 was constructed as a pre-experiment that reproduces the actual 2008 situation. The policies implemented in 2008 and those already decided are included in both *ORG* and *BASE* scenarios, including the CAP and biofuel policies, following Laborde and Valin (2012). Demand for the considered organic products is calibrated using Eurostat data and market share estimate of 2% in 2008 (Agence Bio, 2012, Willer, 2012). After 2008, we assume a slight increase in consumers' preferences for organic products, with a market share for organic products reaching 5% by 2020.[16]

Simulations of a policy change scenario (hereafter *ORG*) relative to this baseline include policies that increase the share of organic agriculture in the EU. The *ORG* scenario is such that the EU requires 20% of area cultivated for maize, rapeseed, sunflower and wheat to be under organic farming by 2020. The rate of 20% corresponds to a target in the organic action plans released by several European member states (see above). We implement the mandate progressively from 2008 to 2020, in each agro-ecological zone of EU, with no geographical reallocation of the organic mandate possible. The 20% target is calculated as the ratio of the total organic land over the total land (conventional and organic) dedicated to the crops under consideration. Consumers' preferences are the same as in the *BASE* scenario. To reach the policy target, the consumption of organic products is subsidized under the *ORG* scenario. In order to ensure the budget neutrality of the policy, we assume that an equal cost tax is added on the total (conventional and organic) consumption of the considered crops, following Kretschmer and Peterson (2010).

II.3 Results

II.3.1 Supply and demand effects

An EU-wide mandate on the share of organic land has large effects on production. Organic output is multiplied by almost 12 by 2020 under the *ORG* scenario, relative to the *BASE* scenario. Such a large increase may rise some technical obstacles, such as mobilizing large quantities of natural fertilizer, in particular animal or green manure. Because of the lack of reliable data to model substitutions between chemical and organic fertilizers, we assume that that resources will not be limiting factors so as to allow the expansion of organic production without major changes in the overall farming system. Note that potential constraints on fertilizing capacity would mostly translate into lower yields, and that the yield gap is subject to detailed sensitivity analysis (see the appendix). More generally, we adopt a simplified view of the conversion from conventional to organic farming. We consider changes in the production technology of some crops and do not take into account

[16] In the EU, the value of organic consumption (for all products) increased by 5% between 2008 and 2009 and by 6.2% between 2009 and 2010. In France, the share of organic consumption has been multiplied by 1.8 between 2007 and 2010. In 2010, the most developed organic market was in Danemark, where organic products accounted for 7.2% of the total food consumption (Willer, 2012). Using data from various sources (Eurostat data on food enterprises and Agence Bio, 2012), we estimate that the average market share for organic products in the EU was around 2% in 2010.

changes induced in the rest of the production system (changes in the crop patterns across time and space, increase in the use of mechanical techniques for weeding...) nor the changes in supply and demand of inputs and crops they cause.

The combination of production changes and consumption subsidies modifies the relative prices of organic and conventional products. The demand for organic products increases, while the demand for conventional crops decreases. Production follows, through the expansion of the area dedicated to organic crops (+ 8.1 million hectares) as well as the increase in the use of other inputs. Under the *ORG* scenario, a decrease in the land used for conventional crops we focus on (maize, rapeseed, sunflower, wheat) goes with a decrease in the use of intermediate inputs on non organic land. The total (conventional + organic) area dedicated to each crop is almost similar between the *ORG* and *BASE* scenarios in 2020, as shown in Table II.1. Under the *ORG* scenario, the organic area expands and conventional crops area contracts. Overall, land use changes within the EU mostly involve substitutions between conventional and organic crops as a response to higher feedstock prices.

Table II.1. Cultivated area (1 000 ha) for selected crops in EU27

	2008	2020	
Crop	BASE	BASE	ORG
Maize	9 921	9 833	9 819
Conventional	9 865	9 714	8 186
Organic	55	119	1 633
Rapeseed	4 665	4 563	4 541
Conventional	4 647	4 532	4 168
Organic	18	31	374
Sunflower	3 651	3 565	3 569
Conventional	3 618	3 497	2 633
Organic	32	68	936
Wheat	26 498	26 139	26 197
Conventional	26 153	25 601	20 276
Organic	345	538	5 920
Total organic	450	756	8 863

Because of the large gap between organic yields and the ones in conventional agriculture, the impacts of the *ORG* scenario on prices, trade and EU land use are significant. EU production of wheat decreases by 9.7 million tons and EU maize production by 3.5 million tons compared to the *BASE* scenario in 2020. This corresponds to a shift of 5.4 million hectares of wheat and 1.4 million hectares of maize to organic. As a consequence, world prices are affected, albeit in an uneven way. The increase in world prices is larger for wheat (Table II.2). In the new equilibrium, the decline in EU arable crops production is not fully offset by an increase in production in the rest of the world. In the oilseeds and maize sector, there is more adjustment in the demand by food and energy industries, in particular because of a decline in the use of the feedstock for bioenergy purpose. For wheat, the adjustment takes place mostly in the feed sector, with a change in the input mix for feed concentrates.

Table II.2. Changes in market balance (1 000 tons) in the ORG scenario, with respect to the reference scenario *REF*

	EU27	Brazil	USA	IndoMalay	RoW	World	World prices
Maize							
Supply	-3 453	630	606	57	991	-1 169	+2.3%
Final demand	-86	-4	-2	-32	-533	-657	
Livestock demand	114	-172	-40	22	527	451	
Other demand	-233	-38	-384	-2	-305	-963	
Rapeseed							
Supply	-874	0	-1	0	-8	-882	+1.6%
Final demand	-0	-0	-1	-0	-0	-1	
Livestock demand	20	0	1	0	26	47	
Other demand	-622	-0	-1	-0	-305	-927	
Sunflower							
Supply	-294	1	-0	0	-81	-375	+1.4%
Final demand	-0	-0	-0	-0	-0	-0	
Livestock demand	36	0	3	0	42	81	
Other demand	-191	1	-4	0	-261	-455	
Wheat							
Supply	-9 671	81	176	0	2 341	-7 073	+3.3%
Final demand	-157	-3	-3	-0	-297	-460	
Livestock demand	-1 859	-5	-229	-92	-3 345	-5 530	
Other demand	-746	-98	-16	-3	-221	-1 084	

Note: Demand includes demand from final consumers (*Final demand*), intermediate consumption by the feedstock industry (*Livestock demand*) and intermediate consumptions by other sectors (*Other demand*), including biofuel production. Global markets are balanced.

As a consequence of the decrease in the quantities produced, the change in prices and the demand displacement, the EU trade balance for feedstocks deteriorates, as shown in Table II.3. The net exports of maize and wheat decrease by 18% and 25% respectively. The production of rapeseed and sunflower oil also experience a slight decreases and their imports rise. The demand for these oils is partially displaced by palm and soybean oils imports. The ranking between regions exporting to Europe (and to which Europe exports) remains roughly the same as shown in Table II.4. The ORG scenario results in larger imports of maize from Brazil, slighlty larger palm oil from Malaysia and Indonesia, and wheat from Russia, Canada and the U.S. EU exports of wheat and maize are also affected (Table II.5).

Table II.3. EU production and trade in 2020 (1 000 tons)

	BASE				ORG			
	Prod.	Exp.	Imp.	N. trade	Prod.	Exp.	Imp.	N. trade
Maize	73 516	6 784	11 221	-4 437	70 063	5 536	13 221	-7 685
Rapeseed	20 551	605	3 064	-2 460	19 678	524	3 256	-2 732
Sunflower	7 579	315	1 633	-1 318	7 285	277	1 734	-1 457
Wheat	151 898	19 880	5 898	13 982	142 227	14 843	7 770	7 073
Soybeans	1 180	198	27 362	-27 164	1 175	194	28 077	-27 883
Palm oil	15	13	3 194	-3 181	15	14	3 264	-3 250
Rapeseed oil	7 402	116	218	-102	7178	109	220	-112
Soybean oil	3 177	760	1 575	-815	3 317	781	1 635	-854
Sunflower oil	2 709	222	846	-624	2 636	209	856	-647

Note: Table report production (Prod.), exports (Exp.), imports (Imp.) and net trade (N. trade) for European Union, in the baseline and in the *ORG* scenario.

II.3.2 Land use changes

Changes in demand and production induce changes in land use across regions. In addition to the direct conversion of arable land to organic (i.e. the dLUC effect here), changes in relative prices cascade across sectors and countries, through substitutions and supply response elasticities. A new equilibrium is such that, globally, matching the target of 20% of maize, rapeseed, sunflower and wheat acreage converted to organic production in the EU requires around 600 000 ha of land to be converted to crop production worldwide, as shown in Table II.6. Analysis of detailed data shows that, at the world level, the main effects are the conversion of pasture to arable crops (323 000 ha); the conversion of forests (166 000 ha including some 10 000 ha of primary forest) to agriculture; and the conversion of previously uncultivated savanahs (59 000 ha). The most affected country is Brazil, where cropland increases by 223 000 hectares at the expenses of pastures (-169 000 hectares) and forest (-47 000 hectares). The aggregate "Rest of the World" region also shows significant changes in land use. A major component is the increase in land devoted to wheat in Russia and neighboring countries, whose cereal production benefits from the reduction of EU output (see Table II.2).

The net displacement factor (NDF), as defined in Plevin et al. (2010) is found to be 0.074. This means that, overall, the changes in relative prices induce 0.074 hectare of additional cropland anywhere in the world for each additional hectare of organic maize, rapeseed, sunflower or wheat in Europe. This value is lower than other estimates found in the literature for other policies: Laborde and Valin (2012) find a NDF value around 0.2 for the European biofuel policy and Plevin et al. (2010) consider values in the range from 0.25 to 0.8 for the expansion of the U.S. corn production for ethanol. The conversion to organic production only affect the average yield, while biofuel policies take feedstock land out of food production (even though byproducts of biofuel production used in the animal feed sector partially offset this effect and reduce the NDF).

Our estimates nevertheless suggest that the LUC changes induced by a shift of EU production

Table II.4. EU imports in 2020 (1 000 tons), by exporter

	2008	2020		
Crop and exporter	REF	REF	ORG	% Variation
Maize				
Brazil	4 749	9 146	10 709	17.1
RoW	3 374	1 776	2 153	21.2
USA	815	299	360	20.2
World	8 938	11 221	13 221	17.8
Rapeseed				
RoW	2 790	3 047	3 237	6.3
USA	13	18	19	6.1
World	2 803	3 064	3 256	6.3
Sunflower				
RoW	1 036	1 339	1 434	7.1
USA	339	294	300	1.8
World	1 375	1 633	1 734	6.2
Wheat				
Brazil	468	705	906	28.5
RoW	4 505	3 506	4 704	34.2
USA	2 275	1 687	2 160	28.1
World	7 248	5 898	7 770	31.7
Palm oil				
Brazil	13	25	25	1.1
IndoMalay	2 384	2 532	2 588	2.2
RoW	358	637	651	2.2
World	2 755	3 194	3 264	2.2
Rapeseed oil				
RoW	127	183	184	0.9
USA	36	35	36	1.1
World	163	218	220	0.9
Soybean oil				
Brazil	404	553	565	2.1
IndoMalay	0	0	0	7.5
RoW	542	1 013	1 061	4.8
USA	11	9	9	1.2
World	957	1 575	1 635	3.8
Sunflower oil				
Brazil	3	4	4	2.5
IndoMalay	0	0	0	3.0
RoW	609	823	833	1.1
USA	19	19	19	0.6
World	631	846	856	1.1

Table II.5. EU exports in 2020 (1 000 tons), by importer

	2008	2020		
Crop and exporter	REF	REF	ORG	% Variation
Maize				
Brazil	0	0	0	−18.6
IndoMalay	15	29	22	−22.5
RoW	1 631	4 768	3 854	−19.2
USA	513	1 987	1 660	−16.5
World	2 159	6 784	5 536	−18.4
Rapeseed				
Brazil	2	3	3	−11.1
IndoMalay	0	0	0	−16.4
RoW	264	601	521	−13.3
World	266	605	524	−13.3
Sunflower				
RoW	154	308	271	−12.1
USA	3	7	6	−10.7
World	157	315	277	−12.0
Wheat				
Brazil	17	29	21	−27.9
IndoMalay	46	80	56	−29.8
RoW	9 910	19 746	14 748	−25.3
USA	18	26	18	−29.5
World	9 991	19 880	14 843	−25.3
Palm oil				
RoW	10	12	13	1.9
USA	1	1	1	3.0
World	10	13	14	2.0
Rapeseed oil				
Brazil	0	0	0	−7.1
IndoMalay	3	3	3	−7.1
RoW	128	111	104	−6.3
USA	2	2	2	−6.3
World	132	116	109	−6.3
Soybean oil				
Brazil	0	1	1	2.6
RoW	683	750	770	2.7
USA	8	10	10	3.6
World	692	760	781	2.8
Sunflower oil				
Brazil	0	0	0	−6.5
IndoMalay	2	2	2	−6.8
RoW	192	215	201	−6.2
USA	5	6	5	−5.5
World	199	222	209	−6.2

to organic are too large to be ignored in a comprehensive LCA. We now turn to a more detailed analysis of the overall environmental balance of a shift to organic agriculture.

II.4 Global environmental impacts of organic agriculture

II.4.1 The direct environmental effects of shifting to organic production in the EU

The conversion of 20% of EU arable crops to organic results in significant environmental benefits in the EU. It is well accepted that organic farming has fewer negative environmental impacts than conventional agriculture per unit of area. In particular, studies for Europe show that organic farms use lower nutrient inputs and supply them in more stable forms (Guyomard, 2013). This results in higher soil organic matter and lower contamination of soils and waters by nitrogen and phosphorus.[17] LCAs show that organic agriculture involves lower energy consumption and GHG emissions (Nemecek et al., 2011, Guyomard, 2013). For example, estimates suggests that, over 100 years, in the United Kingdom, 1 kilogram of organic wheat makes it possible to saves about 18 grams of CO_2 equivalent emitted with respect to the same production of conventional wheat (Williams et al., 2006); in Danemark and throughout, it saves about 43 grams of CO_2 equivalent (Nielsen et al., 2003).[18]

If we look now at ecological impacts, it is well-accepted that organic farming benefits to farmland biodiversity. Tuck et al. (2014) find that, on average, organic farming increases species richness by about 30% relative to conventional farming. Organic farming seems also to have a positive impact on species abundance (Bengtsson et al., 2005), even if results are highly variable between studies and species.[19] The main channels are that conventional agriculture damages ecosystems through the use of synthetic pesticides and fertilizers, which induce changes in the chemical composition of soil that are detrimental to soil fauna, insects and vertebrates. With organic farming, crop rotations are more diversified and tend to respect more semi-natural habitats (Burel and Garnier, 2008).

How much are these positive direct environmental effects offset by the indirect ones? Clearly, the results above suggest that a significant increase in the land dedicated to organic farming in the EU has sizable indirect effects in terms of land use change both inside and outside the EU. We use several indicators to assess the induced impact on ecosystems, i.e. fertilizer use, GHG emissions and biodiversity indexes.

17 Note, however that lower yields make the environmental benefits per unit of output less obvious than when measured on a per hectare basis. For example, when measuring nutrient leaching per unit of product, Mondelaers et al. (2009) find no significant difference between organic and conventional farming while Tuomisto et al. (2012) find higher nitrogen leaching in organic farms.

18 The number of LCAs on organic crops in Europe is very limited. References are more numerous on other regions (Meisterling et al., 2009, Pelletier et al., 2008, Wood et al., 2006), but agronomic conditions for the cultivation of conventional crops differ a lot between the EU and the U.S. or Canada, making the extrapolation of American results to Europe difficult.

19 Abundance refers to the relative representation of one specie, usually measured as the number of individuals, while richness refers to the number of different species represented, regardless of their abundance.

Table II.6. Land use changes in the *ORG* scenario

	2008	2020	
Region and land type	Area (10^6 ha)	Area increase (10^3 ha)	Carbon emissions ($MtoeCO_2$)
Brazil			
Cropland	67	223	
Pasture	136	−169	
SavnGrasslnd	133	−6	19
Other	39	−0	
Managed forest	19	−45	13
Primary forest	448	−2	1
EU27			
Cropland	93	20	
Pasture	69	6	
SavnGrasslnd	20	−2	2
Other	52	−2	
Managed forest	148	−22	5
Primary forest	7		
Indonesia and Malaysia			
Cropland	42	16	
Pasture	3	−5	
SavnGrasslnd	9	−0	2
Other	50	−0	4
Managed forest	6	−11	4
Primary forest	98	−0	0
Rest of the World			
Cropland	956	314	
Pasture	974	−146	
SavnGrasslnd	2 985	−47	26
Other	2 555	−50	
Managed forest	507	−64	14
Primary forest	2 386	−7	1
USA			
Cropland	100	28	
Pasture	62	−9	
SavnGrasslnd	268	−4	2
Other	147	−2	
Managed forest	142	−14	3
Primary forest	161	0	−0
World			
Cropland	1 258	602	
Pasture	1 244	−323	
SavnGrasslnd	3 415	−59	50
Other	2 844	−54	4
Managed forest	822	−156	38
Primary forest	3 101	−10	2
Total			95

Note: We report (i) areas of land use categories in 2008 (first column – areas are the same in our two scenarios in 2008, see II.2.1), (ii) changes with respect to this initial area occurring in the *ORG* scenario, by 2020 (second column) and (iii) carbon emissions caused by deforestation and cultivation of previously uncultivated land (see Annex A.2.3).

II.4.2 The indirect impacts on fertilizer use

According to our simulations, the extension of organic farming in the *ORG* scenario leads to an overall decrease in use of fertilizers of -2.4% in the EU, relative to the *BASE* scenario. This corresponds to the net effect of not using synthetic fertilizers on the (now larger) area grown organic, minus the increase in fertilizer use as a response to higher prices in the non organic production. From that point of view, the shift to organic contributes to reduce some of the problems described by Sutton et al. (2011b,a), who report that half of the nitrogen added to European farm fields end up in water and air pollution, resulting, among other things, in algal development and biodiversity loss. Price changes also induce a slight increase in the use of fertilizers in rest of the world, under the *ORG* scenario. The consequence of the EU shift in to organic is *ceteris paribus*, an increase in fertilizer use of +0.35% in the U.S., +0.10% in Brazil and +0.03% in the rest of the world.[20]

These estimates suggest that the indirect effects partially, but not totally, offset the decrease in fertilizer use in the EU. Altogether, the shift of 20% of arable crops to organic in the EU results in a global decrease in the use of fertilizer of 0.35% at the world level. It is noteworthy that the EU shift to organic results may worsen the negative externalities of fertilizer use in the U.S. where a "dead zone" in the Gulf of Mexico is fueled by excess nutriments carried by the Mississipi river (Kling et al., 2014). Even though the increase in U.S. fertilizer use is modest, our estimate illustrate that there might be "leakage" effects between policies that benefit the environment in the EU and have negative consequences elsewhere, unless appropriate policies are also implemented there.

II.4.3 The overall impacts on GHG emissions

This "leakage" effect is an issue which one is perhaps more familiar with when dealing with GHG emissions. The EU experience suggests that if a given country takes drastic measures to curb CO_2 emissions in the industrial sector while others do not, part of the pollution move abroad, because of imports and relocation of polluting industries.

Is a similar "leakage" in GHG emissions important in the case of organic agriculture? Nielsen et al. (2003) and Williams et al. (2006) report figures showing that GHG emissions are lower when the crop is grown organic. This is mostly due to the emissions saved by not using fertilizers and pesticides. Fertilizers have a large fossil energy content given the intensive use of natural gas to

[20] The figure for the U.S. seems surprisingly large, given that our simulation only deals with a very partial shift to organic agriculture in the EU. However, it is consistent with elasticities found in the literature. Our simulations correspond to elasticities of fertilizer demand relative to the price of maize, soybeans and wheat ranging from 0.35 to 0.9, and to an aggregate elasticity for all crops of 0.45. These estimates are in the same range of those found by Choi and Helmberger (1993). Keeney and Hertel (2009), Hertel et al. (2013) use an estimate of 0.25 (derived from the results of Choi and Helmberger (1993)) for the demand of fertilizer relative to corn price, a figure that is also used by the California Air Resources Board (Babcock et al., 2011). Results by Goodwin et al. (2013) are in a similar range.Figures from the U.S. Department of Agriculture (http://www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx#26730) also give orders of magnitude that are consistent with our results. It is noteworthy that, between 2005,when the Renewable Fuel Standard was first implemented in the U.S., and 2012 the quantity of nitrogen used increased by 4% (the nitrogen applied to corn increased by 10%).

synthetize nitrogen. LCAs also show that there are significant emissions due to the need to transport bulk materials from large scale production units (Nemecek et al., 2011).

Taking into account the indirect effects modifies the GHG balance of organic agriculture. Indeed, the shift to organic under the ORG scenario induces price changes, themselves leading to use more intermediate inputs in conventional productions. In addition, in the new equilibrium that prevails after a policy change that rises organic production, 1 kilogram of conventional wheat is replaced by some organic wheat, but there are also many production displacements, including more production outside the EU. Direct and indirect LUCs must also be taken into account, in particular because the land use changes that lead to convert peatlands or primary forests to cropland can generate high levels of GHG emissions per hectare.

When typical GHG emission values are applied to the different land use changes in each agro-ecological zone, we find hat, in 2020, the *ORG* scenario produces around 31 million tons of organic crops more than under the *BASE* scenario, while it generates the emission of 95 million tons of CO_2 equivalent. This suggests that, when cumulated over 100 years, 1 kilogram of organic cereal would induce emissions of around 3 kilograms of CO_2 equivalent because of land use change. This is more than 70 times larger than the direct CO_2 equivalent that Nielsen et al. (2003) find to be saved by organic production methods. That is, in spite of the many uncertainties, it seems highly plausible that the iLUC effects of organic agriculture induce negative consequences that exceed the direct benefits, as far as GHG emissions are concerned.

II.4.4 Impacts on biodiversity at the global scale

Arguably, reducing GHG emissions is not a major motivation for shifting to organic agriculture. In addition to reducing pollution by pesticides and fertilizers, a key objective is often to slow down the decline in biodiversity thanks to production systems that are more integrated and diversified.

As far as biodiversity is concerned, few scientific results hold independently from local conditions and species. Population dynamics depend on multiple factors, face complex non-linearities, path dependency and tipping points. The impact of a shift to organic agriculture on biodiversity depends on the shape of the so-called density dependence function. That is, the relation between the intensity of agricultural production and biodiversity indicators can be highly convex (so that a limited intensity of agriculture already results in large biodiversity losses) or concave (in which case low impact on biodiversity is maintained until a high level of intensity is reached). Some species, especially farmland specialists, also need some degree of agricultural activity to maintain their habitats but suffer from high levels of intensification, hence an inverted U-shaped intensity function. The consequences of a change in farming practices are therefore very different across situations. However, for a large number of species (those with a convex or U-shaped density dependence function), there is no doubt that a 20% shift to organic will have significant positive consequences in the EU.

Consider now the indirect effects. The dLUC and iLUC impacts of a significant shift to organic agriculture induce changes that may have negative consequences for biotopes. The figures quoted above regarding the NDF and the deforestation effects caused by iLUC appear minor, but some of them affect areas with a high level of biodiversity. Comparing the local benefits and potential losses overseas in terms of biodiversity is a dubious exercise. It would require balancing loss in biodiversity in different habitats, some of them primary forests, others anthropized agricultural landscapes. Here, we use several indicators as an attempt to gauge the net effects.

Literature in ecology largely uses the species-area relationship to estimate species losses caused by deforestation (May, 2000). A rather robust stylized fact in ecology is the positive relationship between the number of species and the size of the area in which they are found. The species-area relationship characterizes this fact with a formula where the number of species is proportional to the area rised to an exponent (usually a number between 0.2 and 0.3). Applying this relationship, we can compute the relative change in biodiversity caused by the variation in forest area. The formulation of the species-area relationship is $S = cA^z$, with S the number of species, A the area, c and z two parameters, with $z = 0.25$ following Rosenzweig (2004). The relative variation in the number of forest (primary + managed) species between the baseline and the policy scenario is then given by $S_{ORG}/S_{BASE} = \exp[z(lnA_{ORG} - lnA_{BASE})]$. Applying it to our simulation outputs, we find that the larger relative biodiversity losses occur in Brazil and in the IndoMalaysia region (Indonesia + Malaysia) and they amount to 0.0028% and 0.0032% with respect to the baseline. Clearly, these figures are very small compared to the uncertainties that result from the many assumptions made regarding the density dependence function and those required to construct the GE model that provides the LUC results. They suggest that the biodiversity losses induced by the indirect effects are minor, mainly because of the small value of the NDF.

Other indicators have been developed in the framework of LCAs to describe the potential damages caused to ecosystems by land use. They are often referred to as *characterization factors*. For our purpose, these indicators need (i) to have a global scope and (ii) to differentiate the impacts of different land use intensities. The Mean Species Abundance (MSA) is one of the few indicators meeting both criteria. It describes biodiversity as the remaining mean species abundance of original species relative to their abundance in primary vegetation, which is assumed to be the reference habitat. Alkemade et al. (2009) and Alkemade et al. (2013) compute the MSAs for different land use categories from a meta-analysis of peer-reviewed literature. In particular, they find that low-input agriculture has a MSA of 0.3 (the original species abundance under low input agriculture is 30% of their abundance under primary habitat, i.e. primary vegetation), while intensive agriculture has a MSA of 0.1 (the MSA of livestock grazing and man-made pasture is respectively 0.7 and 0.3 and forest plantations have a MSA of 0.2). We compute the global MSA corresponding to the land use changes in 2020 under both the *ORG* and the *BASE* scenarios adapting the methodology proposed by Alkemade et al. (2009) for missing data (see the appendix A.3). Our synthetic indicator is the area weighted mean of the MSA values of each land use. We find that the MSA is 0.016% higher in the *ORG* scenario that in the *BASE* scenario by 2020. That is, the MSA indicator suggests that the gains in biodiversity obtained by expanding organic agriculture in the EU is large (the MSA in Europe increases by 1.29% in the *ORG* scenario with respect to the *BASE* scenario) compared to the losses elsewhere in the world induced by LUC.

II.4.5 Accounting for the vulnerability of specific ecosystems

The positive biodiversity effect of a shift to organic agriculture in the EU as measured by the MSA nevertheless leaves important issues unanswered. Irreversibility and uniqueness are ill-measured by the MSA, since every hectare has the same weight, regardless of species' endemism and degree of threat. Forests in Brazil, Indonesia and Malaysia gather 12.5% of the endangered species living in forests threatened by agricultural activities, according to the International Union for Conservation of Nature and Natural Resources.[21] They also concentrate 3 of the 25 world biodiversity hotspots according to Myers et al. (2000)'s definition of a hotspot, i.e. an area "featuring exceptional concentrations of endemic species and experiencing exceptional loss of habitats". The endemics plant and vertebrate species of Brazil's Atlantic Forest and Cerrado and Indonesia and Malaysia's Sundaland represent 9.2% and 5.1% of all plant and vertebrate species world-wide.

There is no satisfactory method to balance a biodiverisity gain on EU farmland and a potential loss of biodiversity in a tropical hotspot. As a crude benchmark, we can use Schmidt (2008)'s proposal to average species richness with an indicator of ecosystem vulnerability. The idea is that when only a small fraction of an ecosystem is left, a further transformation will have a greater impact on species than if the ecosystem had its initial size. Following the United Nations guidelines (Koellner et al., 2013), Schmidt (2008) distinguishes 3 main stages in land use changes: *land transformation* is a brutal change due to human activities (such as deforestation) that make land suitable for economic purposes; it is followed by *land occupation*, the new use for which the land has been transformed; and finally, when the land is no more used, there is *land restoration*. Schmidt (2008) characterizes the number of species that are lost or gained due to each of these three stages, with regard to the reference habitat (natural forest). Schmidt focuses only on two regions, Northern Europe (Danemark) and Indonesia and Malaysia. He uses a metric based on weighted species richness on a standard area of $100m^2$. His calculation suggest that the transformation of 1 hectare of intensive arable land into extensive arable land (which shows similarity with conversion from conventional to organic farming practices) in Danemark results in a gain of 5. While the transformation of 1 hectare of natural forest into oil palm plantation in Malaysia and Indonesia induces a loss of 6978. Transformation of particular natural habitats into palm plantations may therefore generate considerable biodiversity losses. Besides transformation, cultivation of one hectare of palm oil during one year has an impact factor of 28. On the other hand, the changes from intensive cultivation of cereals to extensive practice in Denmark allows for a gain of 15 every year (extensive cereal production has an impact of 20 while intensive cereal production has an impact of

21 Figures from the IUCN Red list on April 2014, http://www.iucnredlist.org/

35). Clearly, Schmidt's work cannot be extrapolated to all the regions included in our simulations. But his result suggests that any potential iLUC effect that leads to the conversion of primary forest in tropical hotspots (this conversion amounts to 10 000 hectares in our simulations) will result in a considerable biodiversity debt, while annual biodiversity gains would cumulate over time on the 8.1 million hectares that shifted to organic production in the EU.

Another (controversial) approach to compare gains and losses of biodiversity across the world would be to compare the economic values of the affected ecosystems and species and of the services they provide. It leads to weighting differently the different taxa and species, which is hardly satisfactory from an ecology standpoint. However, the monetary metric can be seen as reflecting a social preference for, say, the existence value of dolphins compared to ticks, as they enter in an individual utility function. Unfortunately, empirical economic valuation of biodiversity are so far partial and lack robustness, as illustrated by the varisous economic valuations of Amazonian hotspots.[22] Given the lack of consensus on monetary valuation, it is probably vain to use such values to compare a biodiversity gain in the EU farmland to a loss of biodiversity in tropical forests. However, Guéant et al. (2012) show that because the economic valuation of an environmental good increases when it becomes rare, it is justified to overweigh in a spectacular way the probabilities of the events associated with bad environmental outcomes such as the irreversible destruction of a rare habitat. Their demonstration suggests that if protecting biodiversity in the EU resulted in threatening some particular forms of biodiversity elsewhere due to iLUC effects, a large weight should be given to the potentially negative impact of iLUCs.

II.5 Conclusion

II.5.1 Organic agriculture and indirect effects

In the EU, the long term environmental damages of intensive agriculture have long been neglected. They are now becoming apparent with, in particular, a dramatic degradation of habitats and widespread water pollution. The loss of productive ecosystem services is now translating into actual economic costs and future generations will pay dearly for the current excess nutriments, long lasting pollutant residues and degraded pollination capacity. From that perspective, a major shift to organic agriculture is sometimes seen as a way to preserve natural capital and future production potential. Most of the scientific literature indeed concludes to significant benefits of organic agriculture in terms of chemical pollution, biodiversity, and to a lesser extent, GHG emissions.

[22] Simpson et al. (1996) use an approach based on bioprospection for pharmaceutical research and find a value of USD 2.59 per hectare for the Western Amazonian biodiversity hotspot. Rausser and Small (2000) adapt this approach to take into account information on functional biodiversity and find much higher values, reaching USD 1 043 US dollars per hectare for the Western Amazon. Costello and Ward (2006) revisits these two papers and explains why results differ so much. Kassar and Lasserre (2004) argue that substituability across species does not reduce their value (as it is the case in the models of Simpson et al. (1996) and Rausser and Small (2000)) but, in a context of uncertainty about the future needs of research, it may on the contrary generate higher conservation incentives.

However, most of the assessments of the impacts of organic agriculture have so far neglected some indirect effects, and in particular those induced by a change in relative prices. Here, we add to the standard environmental analysis an economic component that makes it possible to assess some LUC effects, including those that result from a new market equilibrium at the world level. Should the EU convert a large scale of its arable crops to organic, our results suggest that price changes would provide new incentives to intensify production on that land that remains in non organic agriculture. There would also be indirect effects in third countries, such as putting more land into production and using more fertilizers abroad, due to cascade effects across markets and regions.

Overall, few of these indirect effects are large enough to significantly offset the environmental benefits of shifting to organic agriculture. However, they should be accounted for in LCAs and sustainability impact assessments. In particular, the case of GHG emissions is illustrative since it is likely that the (negative) indirect emissions caused by dLUC and iLUC offset the direct emission savings made possible by organic agriculture. In the area of fertilizers pollution, the direct environmental benefits of shifting to organic agriculture are large. Indirect effects remain limited but price effects resulting in a (slightly) higher use of fertilizers in third countries such as the U.S. could worsen worrying local environmental damages. Regarding the net effect on biodiversity, the indirect land use changes are limited in our simulation of a shift of 20% of arable crops to organic in the EU. Such a shift would only have a very small impact on deforestation, the draining of peatland and the conversion of savanahs in third countries, which are driven by much more powerful factors. However, *ceteris paribus*, because of tipping points and irreversibilities, even a slight iLUC impact can generate damages to specific biodiversity hotspots.

Two important points should be kept in mind when questioning the overall environmental impact of organic agriculture. The first one is that the land use change effects we describe are dependent on the environmental regime in place in other countries. Should efforts to protect primary forests in tropical countries, and to restrict conversion of prairies and grassland in arable crops, be successful, adverse effects would be limited. Our point that iLUC changes outside the EU should be considered when designing EU environmental policy is, from that point of view, debatable. One could argue that the EU cannot be taken for responsible for the lack of appropriate environmental policies outside its borders.

Second, there is no reason to single out organic agriculture for its dLUC and iLUC effects. For example, the use of feedstocks for the production of bioenergy in the EU has much larger LUC effects than the potential 20% shift of EU agriculture to organic.[23] A large variety of consumption patterns have large LUC effects and can be questioned from an environmental standpoint: the

[23] Estimates with the model used in this paper suggest that the EU biofuel policy (as implemented on the basis of the 27 National Renewable Energy Action Plans, i.e. 27.5 Mtoe of biofuel incorporated in 2020, 72% of biodiesel and 28% of bioethanol following the scenario modeled by Laborde and Valin, 2012) results in an increase of cropland of 2.7 million hectares, a decrease in pasture of 0.36 million hectares, a decrease in savanah and natural grasslands of 1.8 million hectares and a decrease in forest of 0.5 million hectares. These figures are much larger than those in Table II.6, i.e. an increase in cropland of 602 000 hectares and a decrease in savanah and natural grassland of 59 000 ha, and forest of 166 000 ha under our simulation of a 20% shift to organic agriculture

large quantities of food wasted participates to large iLUCs; so do large food intakes which cause obesity in a growing number of countries; iLUC effects can also be attributed to meat consumption compared to a more vegetarian diet ; and to the use of detergents, soaps and cosmetics, which use considerable quantities of vegetable oil, including palm oil; garments and apparels require land for growing cotton that largely competes with other uses, including natural habitats in countries such as India; etc. In many of these areas there is a large potential for reduction that would not involve large costs. By contrast, capping organic agriculture would require giving up well-established local environmental benefits.

II.5.2 What lessons for EU policies?

While they should not be used to point a finger at organic agriculture, our findings rise several issues. First, our results suggest that it is useful to have comprehensive environmental assessments of any policy reform. From that point of view, the LCAs and sustainability impact assessments that are implemented are often partial. Before concluding that such or such environmental policy is desirable, one should look also at the indirect effects. This requires assessing the price changes that are induced and the cross market and cross countries effects. Even though the methods are clearly imprecise and require many assumptions, it is the only way to identify and measure the potential leakages of a particular policy.

Second, regarding EU agricultural policy, our findings suggest that the indirect effects of any environmental policy can be significant. In particular, for organic agriculture to be a route for an environmentally friendlier agriculture, it is necessary to improve yields. In real life situations, there is still a large gap in yields with conventional agriculture. Little priority has been given to research on organic agriculture over the last fifty years, and there is certainly a way for innovation to reduce this yield gap. However, a matter of concern is the non-separability in the technology: in practice, higher yields in organic agriculture often require a large amount of labor, which is expensive in the EU. Even if technical changes lifted obstacles for a large yield increase, economic factors might not warrant it unless ambitious Pigouvian policies that modify relative input prices are implemented. The strong opposition to using biotechnology in organic agriculture may also be a double edged sword if, in the future such a technology focused more on improving yields without recourse to chemical inputs.

Third, adverse indirect effects potentially affect all policies that aim at protecting the environment if they involve a large NDF. This rises the question of targeting organic agriculture (or any other environmentally friendly policy) where it maximizes environmental benefits. Policies that organize a "zoning" of European agriculture, e.g. impose a a drastic ban on the use of chemicals in areas that match water caption perimeters, or drastic measures to protect biodiversity in high natural value areas, have been proposed in the past (Mahé and Ortalo-Magné, 2001). Separating the territory between areas to be protected and others to be left to intensive agriculture is at odds with the European approach of rural development (Cochrane and Wojan, 2008). However, zoning relates to the debate on *land sparing vs land sharing* which has resurfaced in the EU, given the recent findings on the relative environmental benefits of both policies (Hodgson et al., 2010, Gabriel et al., 2013, Phalan et al., 2011, 2014). The need to conciliate the protection of biodiversity and higher production makes it worth to investigate more thoroughly the pros and cons of zoning.

CHAPTER III

AGRICULTURAL TRADE, BIODIVERSITY EFFECTS AND FOOD PRICE VOLATILITY

World prices of major agricultural commodities are subject to busts and booms that regularly trigger worries about the functioning of world markets and raise questions about the mechanisms underlying the trend of food prices and their volatility.[1] Price variations are not specific to agricultural markets but the volatility of agricultural commodities (and particularly food commodities) is higher than that of manufactures (Jacks et al., 2011).

Various causes for price fluctuations have been proposed, but economic models usually neglect stochastic elements that may affect production processes. While this is probably innocuous for industrial goods, which are rarely subject to hazardous conditions, it is more questionable regarding agricultural products. Indeed, agricultural goods face a large range of potential threats that may severely undercut farms' production. Two types of factors may interfere with the agricultural production: Biotic factors, called pests, which are harmful organisms, either animal pests (insects, rodents, birds, worms, etc), pathogens (virus, bacteria, fungi, etc), or weeds, and abiotic factors such as water stress, temperature, irradiance and nutrient supply that are often related to weather conditions (Oerke, 2006). Many studies show that crop losses due to pests are critical.[2]

Chapter written with Jean-Marc Bourgeon.

[1] One recent example of public concern was given during the G-20 leaders summit meeting of November 2010: The participants requested several international institutions "to develop options on how to better mitigate and manage the risks associated with the price volatility of food and other agriculture commodities, without distorting market behavior, ultimately to protect the most vulnerable." See Gilbert and Morgan (2010) and Wright (2011) for overviews on food price volatility and examinations of the causes of the recent spikes.

[2] Oerke (2006) estimates that the potential losses due to pests (i.e., without crop protection of any kind) between 2001 and 2003 for wheat, rice, maize, soybeans and cotton were respectively 49.8%, 77%, 68.5%, 60% and 80% of the potential yield. Savary et al. (2000) also finds significant losses: combined average level of injuries led to a decrease of 37.2% in rice yield (if injuries were considered individually, the cumulative yield loss would have been 63.4%). Fernandez-Cornejo et al. (1998) compile similar data on previous periods and find that expected losses relative to current or potential yield from insects and diseases range between 2 to 26% and that losses from weeds range between 0% and 50%.

Our analysis highlights the importance of the stochastic elements affecting agricultural production in determining the pattern of food trade, and particularly the biotic ones that relate the biodiversity of cultivated crops to their productivity and create what are dubbed *biodiversity effects* in the following. Because farmland has specificities that differ from one country to the other, trade must result in specialized international crop productions according to the implied competitive advantages. In this Ricardian setup, opening to trade should not help much to smooth out production risks since the world is exposed to the idiosyncratic risk of the only country producing each crop. However, we show that biodiversity effects create a production externality that impedes productivity and results in an incomplete specialization at equilibrium under free trade. To reduce the productive loss due to pests, farmers have recourse to agrochemicals (pesticides, fungicides, herbicides and the like),[3] but these products also have damaging side-effects on the environment and human health that spur governmental intervention.[4] However, restrictions on the use of pesticides may impede the competitiveness of agriculture on international markets. Governments, when designing these regulations, thus face a tradeoff: either they contain the externality on consumers and tolerate a high level of negative production externality, which diminishes the competitiveness of their agricultural sector, or they are more permissive in the utilization of pesticides to enhance the productivity of their agricultural sector to the detriment of their consumers. We show that when governments use these regulations strategically to enhance the comparative advantages of their agriculture, trade results in an environmental "dumping" situation where regulations are more lenient than they otherwise would be. Nevertheless, restrictions on pesticides under free trade are generally more stringent than under autarky. The resulting effect of trade on crop price distributions is ambiguous: because of more restrictive policies and the intensification of production, the expected market price of crops produced exclusively by one country is increased for its domestic consumers. For crops produced by both countries, the change in price depends on the share of farmland devoted to these crops compared to under autarky. Trade also unambiguously increases the production volatility of crops produced by both countries. Some specialized productions (the ones for which comparative advantages are large) could see a reduction in their volatility, but that supposes very

3 Practices controlling biotic factors include physical (mechanical weeding or cultivation…), biological (crop rotations, cultivar choice, predators…) and chemical measures. They allow a reduction of losses, but do not totally avoid them. In the case of wheat, they reduce potential losses of 50% to average actual losses of about 29% (from 14% in Northwest Europe up to 35% and more in Central Africa and Southeast Asia). Actual losses for soybean average around 26%. The order of magnitude of losses for maize and rice is greater, with actual losses of respectively 40% and 37% (Oerke, 2006).

4 Pimentel (2005) reports more than 26 million cases worldwide of non-fatal pesticide poisoning and approximately 220 000 fatalities. He estimates that the effects of pesticides on human health cost about 1.2 billion US$ per year in the United States. Mammals and birds are also affected. Farmland bird population decreased by 25% in France between 1989 and 2009 (Jiguet et al., 2012), and a sharp decline was also observed in the whole EU during the same period (EEA, 2010). Pesticides also contaminate water and soils and significantly affect water species both locally and regionally (Beketov et al., 2013). Many countries have adopted regulations that forbid the most harmful molecules, set provisions on the use and storage of pesticides, and intend to promote their sustainable use. For example, only a fourth of the previously marketed molecules passed the safety assessment made within the European review process ended in 2009 (EC, 2009). The use of pesticides seems to follow a decreasing trend: in Europe, quantities sold decreased by 24% between 2001 and 2010 (ECP, 2013).

small biodiversity effects.

To investigate the importance of biodiversity effects on the pattern of trade and on price volatility, we re-examine the standard Ricardian model of trade as developed by Dornbusch et al. (DFS, 1977), which considers the impact of trade on two economies that produce a continuum of goods. The productive efficiencies of these economies determine their comparative advantages, resulting in a division of the production into two specific ranges of products, one for each country. Here, we consider two-sector economies, an industrial/service sector that produces a homogeneous good with equal productivity in the two countries (which serves as the numeraire), and an agricultural sector that produces a range of goods with different potential yields, the effective crop yields depending on biotic and abiotic factors. Many agro-ecology studies show that genetic diversity within and among crop species improves the resilience of agricultural ecosystems to adverse climatic conditions and harmful organisms. Crop biodiversity reduction, observed in many modern agricultural systems, increases the vulnerability of crops to these stresses.[5] To account for these relationships in a tractable way, we follow Weitzman (2000): the larger the share of farmland dedicated to a crop, the more its parasitic species proliferate, and thus the more fields of that crop are at risk of being wiped out.[6] More specifically, farming a given plot of land is a Bernoulli trial in which the probability of success depends on the way land is farmed in the country: the larger the share of farmland devoted to a particular crop, the higher the probability that the plots of that crop are destroyed by biotic or abiotic factors. At the country level, these biodiversity effects translate into crop production that follows normal distributions parameterized by farming intensities. Pesticides have a positive impact at both the individual level (direct effect) and regional level (indirect, cross-externality effect): by destroying pests on her field, a farmer improves the resilience of her plot and reduces the risk borne by the adjacent fields. This cross-externality between fields of the same crop thus attenuates the negative biodiversity effects. We abstract from risk-aversion by assuming that farmers and consumers are risk-neutral.[7] Although by assumption potential crop yields are different, consumer preferences and the other characteristics of the two economies are such that the autarky case displays symmetric situations: the agricultural revenue, land rent and environmental tax are at the same levels; only crop prices differ. We then consider the free trade situation and show that biodiversity effects result in an incomplete specialization of the countries. The intuition is as follows: without biodiversity effects, free trade results in the same specialization as described by DFS, a

[5] This is demonstrated in Tilman et al. (2005) using simple ecological models that describe the positive influence of diversity on the biomass produced by both natural and managed (i.e. agricultural) ecosystems, and corroborated by empirical results detailed in Tilman and Downing (1994) and Tilman et al. (1996). See also Smale et al. (1998), Di Falco and Perrings (2005) or Di Falco and Chavas (2006) for econometric investigations.

[6] Weitzman (2000) makes an analogy between parasite-host relationships and the species-area curve that originally applies to islands: the bigger the size of an island, the more species will be located there. He compares the total biomass of a uniform crop to an island in a sea of other biomass. A large literature in ecology uses the species-area curve which is empirically robust not only for islands but for uniform regions more generally (May, 2000, Garcia Martin and Goldenfeld, 2006, Drakare et al., 2006, Plotkin et al., 2000, Storch et al., 2012).

[7] Risk-aversion would induce government to reduce specialization under trade to diversify production risks among countries as demonstrated by Gaisford and Ivus (2014). Abstracting from this diversification effect allows us to isolate the impact of biodiversity on specialization.

threshold crop delimiting the production range specific to each country. With biodiversity effects, such a clear-cut situation with one threshold crop can no longer exist because the intensification of farming reduces expected crop yields: an increase in acreage devoted to a crop reduces its expected yield and, symmetrically, a diminution of its acreage increases it. As a result, the two countries produce a whole range of crops delimited by two threshold crops, their shares of production evolving according to the relative yields. This range of crops depends on the environmental taxes, which also determine the country's share of the worldwide agricultural revenue. Although we consider competitive economies and free trade, the free trade outcome is the result of a strategic game played by the two countries similar to the "strategic trade policy" problem first investigated by Brander and Spencer (1985) (see Helpman and Krugman, 1989, for a detailed analysis), where environmental taxes are used as a substitute for trade policies.[8] This is the case with or without biodiversity effects, since pesticides have a direct effect on production, but distinguishing between these two cases allows us to detail the biodiversity effects more thoroughly. Two countervailing effects are at work when accounting for the biodiversity effects: as specialization increases the negative production externality, governments are induced to lower the tax on pesticides, but since the externality limits trade specialization, the effect of the tax on prices concerns a reduced set of crops, which induces governments to increase the tax.

To solve this game when biodiversity effects are at work, and to compare its equilibrium to the autarky case, we focus on a family of relative potential yield functions that allows for symmetric equilibria. We also characterize a fictitious non-strategic situation where the governments are unaware that regulations have an impact on their revenues. Compared to this non-strategic situation, we show that environmental taxes are lower at the equilibrium of the strategic policy game. Hence, to give a competitive advantage to their farmers, the governments are permissive regarding the use of pesticides. That leniency depends on the way the relative potential yield evolves along the range of crops: the more rapidly the productivity differential increases, the tighter the regulation.

However, we also show that environmental taxes are generally above the autarky level. Indeed, under autarky, governments are willing to accept a more intensive use of pesticides because they allow a decrease in the prices of all crops sold in the country. Domestic consumers bear all the costs and reap all the benefits of the agrochemical use. This is no longer true under free trade because of the (partial) specialization of each country. Domestic agricultural production concerns only a reduced range of products, and some of these crops are also produced abroad: prices of these goods are less reactive to the domestic regulation.[9]

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8 It is thus a "strategic environmental policy game" as analyzed by Barrett (1994) in an oligopoly setup à la Brander and Spencer (1985) where firms pollute and governments are prevented from using trade policy instruments. He shows that market share rivalry results in weak environmental standards. Trade policies in the DFS setup have been analyzed in Itoh and Kiyono (1987), Opp (2010) and Costinot et al. (2013). The latter study shows that the optimal trade policy is a mix of uniform import tariffs and of export subsidies that are weakly decreasing with respect to comparative advantage.

9 Markusen et al. (1995) and Kennedy (1994) obtain comparable results in an imperfect competition framework. Compared to the market share rivalry analyzed by Barrett (1994), which tends to lower environmental tax

Finally, the distribution of crop production allows us to characterize the market price distribution in each case. Approximations of the expected value, of the standard deviation and of confidence intervals for the market prices are derived using the variation coefficient (the ratio of the standard deviation over the expected value) of the crop production. At a country level, because shocks affecting plots are independent, an increase in a crop's share of farmland creates a "scale effect" that counteracts the biodiversity effect of intensification, resulting in a decrease in production volatility. However, at the worldwide level, the discrepancy in the countries' farmland productivity creates a "yield effect" that increases volatility. We show that compared to the volatility of the aggregate production under autarky, trade increases the volatility of crops produced by both countries as well as some of the specialized productions: the production volatility only of those crops for which countries have high comparative advantages could be reduced by trade. However, if biodiversity effects are large, the volatility of all crops is increased.

Models of food trade incorporating production shocks generally assume that the distributions of idiosyncratic shocks are not affected by trade policies and that free trade allows the diversification of risk. This is the case e.g. in Newbery and Stiglitz (1984) which shows that when financial markets are missing, production choices of risk-averse farmers under free trade are different from those made under autarky and may result in welfare losses. These models assume that the potential productivity of farmland is the same whatever the country. We take the opposite viewpoint and adopt the Ricardian setup of DFS to account for farmland discrepancies between countries. This setup allows us to describe how idiosyncratic shocks evolve with trade. The literature of the effect of volatility in Ricardian setups dates back to Turnovsky (1974) and is mostly focused on asserting the impact of risk aversion on the pattern of trade.[10] Recent Ricardian models of trade involving more than two countries (Eaton and Kortum, 2002, Costinot and Donaldson, 2012) incorporate a stochastic component together with trade costs to specify the pattern of trade and avoid complete specialization. These components are not related to the production process (they are "exogenous" and of no use in explaining price volatility). Our setup offers such features endogenously: biotic and abiotic factors affect stochastically production which generates price volatility and causes productivity losses that prevent complete specialization at equilibrium.

The rest of this article is organized as follows: the next section details how we integrate biodiversity effects in the agricultural production process. We derive the autarky situation and show that biodiversity effects result in an incomplete specialization under free trade. Section 3 is devoted to the tax policy under free trade. Strategic and non strategic tax policies are derived to analyze the impact of the biodiversity effects on public policy. Their implications on price volatility

policies below Pigovian levels, they show that there is a "pollution shifting" effect that works in the opposite direction when the competition takes place on domestic markets: although under autarky an environmental tax increase is detrimental for consumption, this is not the case when domestic consumers have access to the goods produced in the foreign country. Governments are thus induced to (unilaterally) increase their environmental tax.

[10] This is the case e.g. in Gaisford and Ivus (2014) which introduces stochastic shocks to foreign technologies in a DFS setup. In their model, given high risk, risk aversion and no domestic and international risk sharing, tariff protection encourages the sectoral diversification of domestic production and increases expected welfare.

are exposed in section 4. We discuss the impacts of trade on fertilizer use in section 5. The last section concludes.

III.1 The model

Consider two countries (Home and Foreign) whose economies are composed of two sectors; industry and agriculture. Our focus being on agriculture, the industrial/service sector is summarized by a constant return to scale production technology that allows production of one item with one unit of labor. The industrial good serves as the numeraire which implies that the wage in these economies is equal to 1. The agricultural sector produces a continuum of goods indexed by $z \in [0,1]$ using three factors: land, labor and agrochemicals (pesticides, herbicides, fungicides and the like) directed to control pests and dubbed "pesticides" in the following.[11] Both countries are endowed with total labor force and farmland denoted by L and N for Home, and L^* and N^* for Foreign (asterisks are used throughout the paper to refer to the foreign country). As farming one plot requires one unit of labor, industry employs $L - N$ workers.

Technical coefficients in agriculture differ from one crop to the other, and from one country to the other. More precisely, absent production externality and adverse meteorological or biological events, this mere combination of one unit of labor with one unit of land produces $\bar{a}(z)$ good z in Home and $\bar{a}^*(z)$ in Foreign. The crops are ranked in decreasing order for Home: the relative crop yield $A(z) \equiv \bar{a}^*(z)/\bar{a}(z)$ satisfies $A'(z) > 0$, $A(0) < 1$ and $A(1) > 1$. Hence, on the basis of these differences in potential yield, Home is more efficient producing goods belonging to $[0, z_s)$ and Foreign over $(z_s, 1]$ where $z_s = A^{-1}(1)$. However, the production of crops is affected by various factors such as weather conditions (droughts or hail) and potential lethal strains (pests, pathogens), resulting in land productivity lower than its potential. We suppose in the following that a unit plot of crop z is destroyed with probability $1 - \psi(z)$ from one (or several) adverse conditions and that, otherwise, with probability $\psi(z)$ the crop survives and the plot produces the yield $\bar{a}(z)$ independently of the fate of the other plots in the region.[12] Here, $\psi(z)$ is a survival probability that depends on many factors, among which a biodiversity factor that evolves with the way land is farmed in the country, and more specifically, the intensity of each crop cultivation as measured by the proportion of land devoted to it, i.e. $B(z) \equiv N(z)/N$ where $N(z)$ is the number of crop z plots.[13] The biodiversity effect we consider relates positively the intensity of a crop with the probability that pests affect each individual plot of this crop if farmers do not take appropriate actions to diminish their exposure. This biodiversity effect thus corresponds to a negative production externality between farmers of

[11] In order to streamline the analysis, we don't consider fertilizers. They are discussed section 5.

[12] Pests and/or meteorological events do not necessarily totally destroy a plot, but rather affect the quantity of biomass produced. This assumption allows for tractability, our random variable being the number of harvested plots rather than the share of harvested biomass.

[13] By assuming independence between plots of the same country, we abstract from macroeconomic effects other than the biodiversity ones. Also, to simplify our setup, we abstract from potential positive or negative spillovers across different varieties of crops.

the same crop, which engenders a decreasing returns to scale production technology in farming: the more land (and labor) that is devoted to a crop at the national level, the lower its expected yield per acre.

However, to counteract the impact of external events on her plot, a farmer can avail herself of a large range of chemicals. All these individual actions put together may in turn have a positive impact on the survival probability of plots by impeding the pest infection in the region, which limits the negative externality due to the intensity of the cultivation. To synthesize these elements in a manageable way, we assume that at the farmer i's level, the survival probability of her plot is given by

$$\psi(z, \pi_i, \bar{\pi}(z), B(z)) = \frac{\mu_0 \eta(\pi_i, z)}{1 + \hat{\kappa}(\bar{\pi}, z) B(z)} \tag{III.1}$$

where π_i is the quantity of pesticides used by farmer i, and $\bar{\pi}(z)$ the average quantity of pesticides used by the other farmers of crop z in the country. Here, $\eta(\pi_i, z)$ corresponds to the impact of farmer i's pesticides on the resilience of her plot, with $\eta(0, z) = 1, \eta'_z(0, z) > 0, \eta''_{zz} < 0$, and $\mu_0[1 + \hat{\kappa}(\bar{\pi}, z)B(z)]^{-1}$ the survival probability of her plot if she does not use pesticides. This probability is nevertheless affected by a cross-external positive impact of the other farmers' treatments, with $\hat{\kappa}(\bar{\pi}, z) > 0$ and $\hat{\kappa}'_{\bar{\pi}}(\bar{\pi}, z) < 0$. Hence, the expected resilience of individual plots decreases with B, the intensity of the crop, but increases with the level of pesticides used $\bar{\pi}$. In the following, we consider the convenient specifications $\eta(\pi_i, z) = e^{\pi_i(\theta(z) - \pi_i/2)}$ where $\theta(z)$ corresponds to the level of pesticides that maximizes the expected yield of the crop, $\hat{\kappa}(\bar{\pi}, z) = e^{(\theta(z) - \bar{\pi})^2/2} \kappa(z)$ where $\kappa(z) \equiv \kappa_0 e^{-\theta(z)^2/2}$ is the lowest cross externality factor for crop z, when all crop-z farmers use pesticide level $\theta(z)$, and $e^{-(\theta(z) - \bar{\pi})^2/2}$ is the negative effect on the crop's resilience induced by an average use of pesticides $\bar{\pi} \leq \theta(z)$. We assume that μ_0, κ_0 and $\theta(z)$ are the same in both countries and such that $\psi(\cdot) \leq 1$.[14]

Because of the externality due to pesticides (on human health and the environment), governments restrict their use. To ease the exposition and simplify the following derivations, we model the governmental policy as a tax which results in a pesticide's price τ, and we suppose that the governments complement this tax policy with a subsidy that allows farmers to cope with the environmental regulation: at equilibrium, it is financially neutral for farmers.[15]

We assume that farmers are risk-neutral and may only farm one crop (the one they want) on one unit plot. Given her crop choice, farmer i chooses the intensity of the chemical treatment of her

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[14] Parameter μ_0 may be considered as resulting from factors beyond the control of the collective action of farmers, such as meteorological events, which may differ from one crop to the other and from one country to the other, i.e. we may rather have two functions $\mu(z)$ and $\mu^*(z)$ taking different values. However, as competitive advantages are assessed by comparing $a(z) = \mu_0 \bar{a}(z) e^{\theta(z)^2/2}$ to its foreign counterpart, assuming that it is a parameter is innocuous.

[15] This tax-neutral policy is equivalent to implementing a restriction on pesticide use, i.e. a quota policy where farmers of crop z are allowed to spread only $\pi \leq \pi(z)$ on their fields. τ then corresponds to the "shadow price" of the pesticides quota and is the same for all farmers.

field in order to reach expected income

$$r_i(z) \equiv \max_{\pi} \mathrm{E}[\tilde{p}(z)\tilde{y}_i(z)] - \tau\pi + T(z) - c$$

where $\tilde{y}_i(z)$ is her stochastic production level, either equal to 0 or $\bar{a}(z)$, $T(z)$ is the subsidy for crop z, which is considered as a lump sum payment to farmer i, and c the other input costs, i.e. the sum of the wage (one unit of labor is necessary to farm a plot of land) and of the rental price of land, which is the same whatever crop is farmed. Denote by $y(z) \equiv \mathrm{E}[\tilde{y}(z)]$ and by $p(z) \equiv \mathrm{E}[\tilde{p}(z)]$ the expected production of crop z and the corresponding expected market price, and define the crop z reference price as[16]

$$\bar{p}(z) \equiv \frac{\mathrm{E}[\tilde{p}(z)\tilde{y}(z)]}{y(z)} = p(z) + \frac{\mathrm{cov}(\tilde{p}(z), \tilde{y}(z))}{y(z)}. \tag{III.2}$$

This reference price corresponds to the per unit average price of crop z for farmers. As $\mathrm{cov}(\tilde{p}(z), \tilde{y}(z)) < 0$, it is lower than the expected market price due to the correlation between total production $\tilde{y}(z)$ and market price $\tilde{p}(z)$. With atomistic individuals (N is so large that the yield of a single plot has a negligible effect on the market price), the crop z farmer's program can be rewritten as

$$r(z) = \max_{\pi} \bar{a}(z)\psi(z, \pi, \bar{\pi}(z), B(z))\bar{p}(z) - \tau\pi + T(z) - c.$$

The first-order condition leads to an optimal level of pesticides $\pi(z)$ that satisfies

$$\eta'_{\pi}(\pi(z), z) = \tau \frac{1 + \hat{\kappa}(\pi(z), z)B(z)}{\mu_0 \bar{a}(z)\bar{p}(z)}$$

at the symmetric Nash equilibrium between crop z farmers. As the subsidy allows farmers to break even, we have $T(z) = \tau\pi(z)$.[17] Competition in the economy (notably on the land market) leads to $r(z) = 0$ for all z at general equilibrium, which gives using $\eta'_{\pi}(\pi, z) = [\theta(z) - \pi]\eta(\pi, z)$,

$$\pi(z) = \theta(z) - \frac{\tau}{c}. \tag{III.3}$$

Competition also implies that the reference price $\bar{p}(z)$ corresponds to the farmers' break-even price: we have

$$\bar{a}(z)\psi(z)\bar{p}(z) = c \tag{III.4}$$

where $\psi(z) \equiv \psi(z, \pi(z), \pi(z), B(z))$, which leads to

$$\mathrm{E}[\tilde{p}(z)\tilde{y}(z)] = \bar{p}(z)\bar{a}(z)\psi(z)NB(z) = cNB(z), \tag{III.5}$$

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16 This definition holds whatever the case at hand, autarky or free trade, with $\tilde{y}(z)$ and $y(z)$ replaced by $\tilde{y}(z) + \tilde{y}^*(z)$ and $y(z) + y^*(z)$ in the latter case.

17 For the sake of simplicity, we consider neither the production nor the market of agrochemicals in the following. Implicitly, farmers thus are "endowed" with a large stock of agrochemicals that farming does not exhaust, leading to prices equal to 0.

i.e., the expected value of the crop z domestic production is equal to the sum of the wages and the land value involved in its farming.

Defining the tax index $t \equiv \exp[(\tau/c)^2/2] = \exp\{-[\theta(z) - \pi(z)]^2/2\}$ where $\pi(z)$ is given by (III.3), which measures the negative effect of the restricted use of pesticides on the crop's resilience (it diminishes with $\pi(z)$ and is equal to 1 when $\pi(z) = \theta(z)$), we get

$$\psi(z) = \frac{\mu_0 \exp[\theta(z)^2/2]}{t[1 + t\kappa(z)B(z)]}. \tag{III.6}$$

Denoting the crop z maximum expected yield $a(z) \equiv \bar{a}(z)\mu_0 \exp[\theta(z)^2/2]$, we get the convenient expressions

$$\bar{p}(z) = \frac{ct[1 + t\kappa(z)B(z)]}{a(z)} \tag{III.7}$$

and

$$y(z) = \frac{a(z)NB(z)}{t[1 + t\kappa(z)B(z)]} \tag{III.8}$$

for the crop z reference price and domestic expected production level. Production of crop z decreases with t because of two effects: the corresponding reduction in the use of pesticides has a direct negative impact on the productivity of each plot but also an indirect negative cross-externality effect between plots.

The representative individuals of the two countries share the same preferences over the goods. Their preferences are given by the Cobb-Douglas utility function

$$U = b \ln x_I + (1 - b) \int_0^1 \alpha(z) \ln \tilde{x}(z) dz - hZ$$

where $\int_0^1 \alpha(z)dz = 1$ and $h > 0$. The first two terms correspond to the utility derived from the consumption of industrial and agricultural goods respectively, while the last term corresponds to the disutility of the environmental damages caused by a domestic use of

$$Z = N \int_0^1 B(z)\pi(z)dz$$

pesticides by farmers. The corresponding demand for the industrial good is $x_I = bR$ where R is the revenue per capita. The rest of the revenue, $(1 - b)R$, is spent on agricultural goods with demand for crop z given by

$$\tilde{x}(z) = \frac{\alpha(z)(1 - b)R}{\tilde{p}(z)} \tag{III.9}$$

where $\tilde{p}(z)$ depends on the realized production level $\tilde{y}(z)$ and $\alpha(z)$ is the share of the food spending devoted to crop z. We assume that consumers are risk-neutral and thus evaluate their *ex ante* welfare at the average consumption level of crop z, $x(z) \equiv \mathrm{E}[\tilde{x}(z)]$. Using (III.9), the corresponding reference price is $\mathrm{E}[1/\tilde{p}(z)]^{-1}$. At market equilibrium under autarky (the same reasoning applies under free trade), as $L\tilde{x} = \tilde{y}$, we obtain from definition (III.2) and the fact that $L\tilde{x}\tilde{p} = \alpha(z)(1-b)LR$ that $\bar{p}(z) = \mathrm{E}[1/\tilde{p}(z)]^{-1}$, i.e. the consumers' and producers' reference prices are the same. The representative consumer's indirect utility function can be written as[18]

$$V(R, Z) = \ln(R) - (1-b)\int_0^1 \alpha(z)\ln\bar{p}(z)dz - hZ. \tag{III.10}$$

where $R = (L - N + cN)/L$ since there is no profit at equilibrium.

The government determines the optimal policy by maximizing this utility, taking account of the relationship between pesticides and the value of land.

III.1.1 Equilibrium under autarky

The agricultural production cost under autarky c_A (the sum of the rental price of the agricultural land and of the wage), is derived from the equilibrium condition on the market for industrial goods.[19] Indeed, due to the constant returns to scale in the industrial sector, the total spending on industrial products must be equal to the total production cost at equilibrium, i.e.,

$$bLR = b[Nc_A + L - N] = L - N$$

which gives $c_A = (\ell - 1)(1-b)/b$ where $\ell \equiv L/N > 1$. Consequently, the value of land is positive if $N < (1-b)L$, a condition assumed to hold in the following. Observe that this value depends neither on the use of pesticides nor on the crops' prices and is the same in both countries in spite of their differences in terms of crop yields. This is due to the Cobb-Douglas preferences (in addition to the constant productivity in the industrial sector). Total domestic revenue is given by

$$LR = Nc_A + L - N = \frac{N(\ell - 1)}{b}$$

for both countries. Market equilibrium on the crop z market implies that total expenses are equal to total production cost, i.e.

$$\alpha(z)(1-b)LR = NB(z)c_A$$

which gives $B(z) = \alpha(z)$. Using (III.3) and $\tau/c = \sqrt{2\ln t}$, we get

$$Z = N\int_0^1 \alpha(z)\theta(z)dz - N\sqrt{2\ln t}.$$

The optimal level of pesticide (or, equivalently, the optimal pesticides tax index) is determined

18 Up to a constant given by $b\ln(b) + (1-b)\int_0^1 \alpha(z)\ln\alpha(z)dz$.
19 Subscript "A" indexes equilibrium values under autarky.

by maximizing the utility of the representative consumer (III.10) which reduces to

$$\min_t (1-b)\int_0^1 \alpha(z)\ln\{t[1+t\kappa(z)\alpha(z)]\}dz + hZ,$$

a program that applies to both countries. We obtain that the optimal tax index under autarky, t_A, solves

$$\sqrt{2\ln t_A}\left[1+\int_0^1 \frac{t_A\kappa(z)\alpha(z)^2}{1+t_A\kappa(z)\alpha(z)}dz\right] = \frac{Nh}{1-b}. \tag{III.11}$$

As $\kappa(z) = \kappa_0 \exp\left[-\theta(z)^2/2\right]$, the optimal tax decreases when κ_0 increases, with a maximum given by $\tau_A = (L-N)h/b$.

While acreage and pesticide levels are the same in both countries, the average production of each is different because of the differences in crop yields. The revenue being the same in both countries, crop demands are identical but because average production levels are different, break-even prices are also different.

III.1.2 Free trade equilibrium

First consider the equilibrium on the market for the industrial good: the condition of equalization of total spending with the total production cost is given by

$$b(Nc + L - N + N^*c^* + L^* - N^*) = L - N + L^* - N^*$$

where $L = L^*$ and $N = N^*$. We obtain $c + c^* = 2(\ell-1)(1-b)/b$, hence that the worldwide agricultural revenue is the same as under autarky. This is also the case for the total revenue, given by

$$LR + L^*R^* = N[c + c^* + 2(\ell-1)] = \frac{2N(\ell-1)}{b}.$$

As under autarky, the share of the agricultural sector of this revenue is unchanged, given by $1-b$. For Home, it results in a per-individual revenue given by

$$R = (\ell-1)\left[1 + 2q\frac{1-b}{b}\right], \tag{III.12}$$

which depends on the fraction $q \equiv c/(c+c^*)$ obtained at equilibrium. This share depends on crop yields and on the environmental tax policies implemented in each country.

In addition to tax policies, competitive advantages also depend on biodiversity effects that evolve with the way land is farmed. Indeed, for a given tax level, because of the production externality effect, the higher the intensity of the farming of a crop (i.e. the more land is devoted to that crop), the lower the average productivity of the land: intensification undermines the competitive advantages apparent under autarky. More precisely, if crop z is produced by Home only, the market equilibrium condition implies that worldwide expenses on crop z are equal to total production cost, i.e.,

$$2\alpha(z)(\ell-1)N\frac{1-b}{b}=NB(z)c$$

which can be written as $\alpha(z)/q = B(z)$. Opening to trade could thus correspond to a large increase of the acreage devoted to that crop: e.g., if $q = 1/2$, the total farmland doubles which may seriously impair Home's land productivity for crop z. On the other hand, crop z is produced by both countries if their break-even prices are equal, or using (III.7), if

$$A(z)=\frac{c^*}{c}\frac{t^*}{t}\frac{1+t^*\kappa(z)B^*(z)}{1+t\kappa(z)B(z)}. \tag{III.13}$$

At market equilibrium, worldwide expenditure on crop z is equal to the sum of production costs of the two countries:

$$2\alpha(z)N(\ell-1)\frac{1-b}{b}=cNB(z)+c^*N^*B^*(z),$$

which can also be written as

$$\alpha(z)=qB(z)+q^*B^*(z). \tag{III.14}$$

Crop z is thus produced by both countries if there exist $B(z) > 0$ and $B^*(z) > 0$ which solve (III.13) and (III.14). As stated formally in the following proposition, this is true for a whole range of crops. More precisely,

Proposition 1. *Specialization is incomplete under free trade: Assuming κ_0 is not too large, both countries produce crops belonging to $(\underline{z}, \overline{z})$, $0 \le \underline{z} < \overline{z} \le 1$ satisfying*

$$A(\bar{z})=\frac{t^*}{t}\frac{q^*+t^*\kappa(\bar{z})\alpha(\bar{z})}{q} \tag{III.15}$$

and

$$A(\underline{z})=\frac{t^*}{t}\frac{q^*}{q+t\kappa(\underline{z})\alpha(\underline{z})}. \tag{III.16}$$

The intensity of these crops is given by

$$B(z)=\chi(z)\frac{1-q\phi(z)}{q} \tag{III.17}$$

where

$$\phi(z)\equiv\frac{1+A(z)t/t^*}{1+\alpha(z)t^*\kappa(z)}, \tag{III.18}$$

$$\chi(z)\equiv\frac{1+t^*\alpha(z)\kappa(z)}{t\kappa(z)[A(z)t/t^*+t^*/t]} \tag{III.19}$$

for Home and symmetric expressions hold for Foreign (with $A(z)$ replaced by $1/A(z)$). Crops belonging to $[0, \underline{z}]$ ($[\bar{z}, 1]$) are produced by Home (Foreign) only, with intensity $B(z) = \alpha(z)/q$.

Proof: see the appendix.

Using (III.15) and (III.16) with $\kappa_0 = 0$, we end up with $A(\bar{z}) = A(\underline{z}) = (q^*/q)(t^*/t)$ and thus a unique threshold index and complete specialization. With $\kappa_0 > 0$, we have $A(\underline{z}) < A(\bar{z})$ and since A is strictly increasing, $\underline{z} < \bar{z}$: albeit technical differences exist between the two countries, comparative advantages are trimmed by the negative externality that affects national production of each country.

Using (III.8) and (III.17), we obtain that Home's expected production of crop $z \in (\underline{z}, \bar{z})$ is

$$y_T(z) = Na(z)\frac{q^* + \alpha(z)\kappa(z)t^* - qA(z)t/t^*}{t\kappa(z)[qt^* + q^*t + tt^*\alpha(z)\kappa(z)]}.$$

Using the symmetric expression for Foreign, we obtain that total expected production under free trade is given by

$$\begin{aligned} y_T^W &= N\frac{\alpha(z)[a^*(z)t_T/t_T^* + a(z)t_T^*/t_T]}{qt_T^* + q^*t_T + \alpha(z)t_Tt_T^*\kappa(z)} \qquad \text{(III.20)} \\ &= y_A(z)\frac{t_A}{t_T}\frac{1+\alpha(z)t_A\kappa(z)}{1+\alpha(z)t_T^*\kappa(z) - q(1-t_T^*/t_T)} + y_A^*(z)\frac{t_A}{t_T^*}\frac{1+\alpha(z)t_A\kappa(z)}{1+\alpha(z)t_T\kappa(z) - q^*(1-t_T/t_T^*)}. \end{aligned}$$

Observe that with unchanged tax index ($t_T = t_A$) in both countries, we would have $y_T^W = y_A^W$. More generally, compared to the production of these crops under autarky, trade operates as if two distortive effects were at work on domestic productions: a direct tax effect and a cross-externality effect. The first one, given by the ratio t_A/t_T, measures the impact of the change in the environmental policy on the use of pesticides. The second one also reflects the impact of the policy change on the production externality, but it is not a simple ratio: there is a correcting term given by $q(1 - t_T^*/t_T)$ which also depends on the relative wealth.

For these crops produced by both countries, the break-even price is

$$\bar{p}_m(z) = \frac{2(1-b)(\ell-1)}{ba(z)}\frac{q(t^*-t) + t[1+\alpha(z)t^*\kappa(z)]}{t^*/t + A(z)t/t^*}. \qquad \text{(III.21)}$$

For the other crops, the corresponding expected productions and break-even prices are given by

$$y(z) = \frac{a(z)N\alpha(z)}{t[q + t\kappa(z)\alpha(z)]}$$

and

$$\bar{p}_s(z) = \frac{2(\ell-1)(1-b)t[q + t\kappa(z)\alpha(z)]}{ba(z)} \qquad \text{(III.22)}$$

for all $z \leq \underline{z}$ and by symmetric expressions for all crops $z \geq \bar{z}$.

From these preliminary results, it is possible to isolate a first effect due to the production externality by supposing that the environmental taxes and the agricultural revenues of the two countries are unchanged at equilibrium.

Lemma 1. *Suppose that $t_T = t_T^* = t_A$ and $q = 1/2$ at equilibrium under free trade. Then, the break-even prices of the crops produced by only one country are higher for this country than under autarky and are reduced for the other country. For crops produced by both countries, the price for Home (Foreign) is lower (larger) for all crops $z \in (A^{-1}(1), \bar{z}]$ and larger (lower) for all $z \in [\underline{z}, A^{-1}(1))$.*

Proof: see the appendix.

These increases in break-even prices affecting crops produced more intensively under trade than under autarky are the consequences of the cross-externality effects between plots: while the input prices do not change by assumption, the marginal production cost increases, whereas in DFS it is constant.

Of course, environmental taxes and the sharing of the total agricultural revenue evolve with international trade. Using (III.3), we obtain that the level of pesticides under free trade is given by

$$Z_T = \frac{N}{q}\left\{\int_0^{\underline{z}} \alpha(z)\theta(z)dz + \int_{\underline{z}}^{\overline{z}} \chi(z)[1-q\phi(z)]\theta(z)dz\right\} - N\sqrt{2\ln t} \tag{III.23}$$

which depends on the crops farmed at equilibrium and on how the worldwide agricultural revenue is shared.

To determine this share, we can use the fact that the domestic revenues come from the sale of the goods produced nationally.[20] On interval $[0, \underline{z}]$, all revenues spent are collected by Home, while it is only a share $s(z)$ of them on $[\underline{z}, \overline{z}]$. We thus have

$$LR = L - N + (1-b)(LR + L^*R^*)\left\{\int_0^{\underline{z}} \alpha(z)dz + \int_{\underline{z}}^{\overline{z}} s(z)\alpha(z)dz\right\}$$

which simplifies to

$$q = \int_0^{\underline{z}} \alpha(z)dz + \int_{\underline{z}}^{\overline{z}} s(z)\alpha(z)dz \tag{III.24}$$

where $s(z) \equiv y_T(z)/y_T^W(z)$ corresponds to Home's share of expected production relative to the total. Using (III.5), (III.14) and (III.17), we get

$$s(z) = \frac{qB(z)}{qB(z) + q^*B^*(z)} = \frac{qB(z)}{\alpha(z)} = \frac{[1-q\phi(z)]\chi(z)}{\alpha(z)} \tag{III.25}$$

and thus

$$q = \frac{\int_0^{\underline{z}} \alpha(z)dz + \int_{\underline{z}}^{\overline{z}} \chi(z)dz}{1 + \int_{\underline{z}}^{\overline{z}} \phi(z)\chi(z)dz}. \tag{III.26}$$

20 The same expression can be derived using the equilibrium condition on the land market.

III.2 Environmental tax policy and trade

The free trade equilibrium depends on the environmental tax policies (hence t and t^*) that determine the sharing of the worldwide agricultural revenue between the countries. We model this problem as a Nash equilibrium of a two-stage game where Home and Foreign governments choose simultaneously their tax policies in the first stage, knowing that in the second stage farmers decide which crops to sow and how much pesticide to use. Home's government problem when defining its tax policy corresponds to the following program:

$$\max_t \ln(R)-(1-b)\left\{\int_0^{\underline{z}} \alpha(z)\ln \bar{p}_s(z)dz + \int_{\underline{z}}^{\overline{z}} \alpha(z)\ln \bar{p}_m(z)dz + \int_{\overline{z}}^{1} \alpha(z)\ln \bar{p}_s^*(z)dz\right\}-hZ \quad \text{(III.27)}$$

where $\bar{p}_s^*(z)$ is given by (III.22) with t and q replaced by t^* and $q^*=1-q$. The optimal environmental policy resulting from this program depends on t^*: maximizing (III.27) gives Home's best-response to Foreign's policy t^*. We suppose that Foreign's government is in the symmetric situation, i.e. that opening the borders to trade results in a Nash equilibrium of a game where both governments act simultaneously in a strategic way. For the sake of argument, we consider two cases in the following: one in which governments ignore the relationship between their tax policies and their share of the worldwide agricultural revenue, called "non-strategic" free trade equilibrium, and the more realistic case where they reckon that q and thus R depend on t and t^*, which leads to trade equilibrium dubbed "strategic". One may easily show that q is negatively related to t and positively related to t^*. Indeed, a total differentiation of (III.24) yields, using $s(\overline{z})=0$ and $s(\underline{z})=1$,

$$\frac{dq}{dt}=\frac{\int_{\underline{z}}^{\overline{z}}[B(z)(d\chi(z)/dt)/\chi(z)-\chi(z)(d\phi(z)/dt)]dz}{1+\int_{\underline{z}}^{\overline{z}}\phi(z)\chi(z)dz} \quad \text{(III.28)}$$

where it is straightforward from (III.19) and (III.18) that $d\chi(z)/dt<0$ and $d\phi(z)/dt>0$. Hence, we have $dq/dt<0$ and since $q+q^*=1-b$, $dq^*/dt=-dq/dt>0$. Because of the strategic substitutability of the environmental taxes, pesticides are used more intensively at the strategic trade equilibrium than when governments act non strategically. To detail this competition effect and assess its interaction with the biodiversity externality that affects production, we consider in the following the case where $\alpha(z)=1$ and $\theta(z)=\theta$ for all z which implies that $\kappa(z)=\kappa_0\exp(-\theta^2/2)\equiv\kappa$ for all z. Hence, neither the demand nor the externality on consumers' utility distinguishes crops, so we have $Z=N\theta-N\sqrt{2\ln t}$ whatever the case at hand. We also suppose that $A(z)$ allows us to obtain symmetric equilibria so that $q=1/2=z_s$ at equilibrium. We analyze the two types of free trade equilibria (non-strategic and strategic) assuming first that there is no biodiversity effect, i.e. $\kappa=0$. While there are no cross-externality effects between fields of the same crop, farmers still have an incentive to spread pesticides on their plots to increase their expected yield. We then introduce the negative production externality ($\kappa>0$), which induces decreasing returns to scale in the agricultural sector at the national level.

III.2.1 Trade without biodiversity effect

When $\kappa = 0$, we obtain using (III.11), $(2\ln t_A)^{1/2} = \tau_A/c_A$ and $c_A = (\ell - 1)(1-b)/b$, that the environmental tax under autarky is given by $\tau_A = (L-N)h/b$. Under free trade, each country specializes on one segment of the range of crops delimited by threshold z_s which satisfies $A(z_s) = (q^*t^*)/(qt)$ using (III.13). Equilibrium on the land market, $\int_0^{z_s} B(z)dz = \int_0^{z_s}(1/q)dz = 1$, leads to $q = z_s$: Home's share of the worldwide agricultural revenue is equal to the range of crops produced domestically. Consequently z_s solves $\xi(z_s) = t^*/t$ where $\xi(z) \equiv A(z)z/(1-z)$ is strictly increasing. In the non-strategic situation, governments do not take into account the effect of their environmental taxes on the sharing of the agricultural revenue. The effect of the tax policies on z_s, q and R are neglected when solving (III.27). The problem simplifies to

$$\min_t (1-b)q\ln t - hN\sqrt{2\ln t}$$

where q is considered as a constant. The first-order condition leads to an optimal tax index that solves

$$\sqrt{2\ln t} = \frac{Nh}{q(1-b)}. \tag{III.29}$$

Using $q = z_s$, we get $t = \exp\left\{(Nh)^2[(1-b)z_s]^{-2}/2\right\}$ and thus a threshold crop that solves

$$\xi(z_s) = \exp\left\{\frac{(Nh)^2(1-2z_s)}{2[(1-b)z_s(1-z_s)]^2}\right\}.$$

As $\sqrt{2\ln t} = \tau/c$ and $c = 2qc_A$, (III.29) allows us to obtain $\tau = 2\tau_A$ whatever the country's share of the worldwide agricultural revenue. Stated formally:

Proposition 2. *Suppose that there are no biodiversity effects. Then, at the non-strategic trade equilibrium, the environmental tax is doubled compared to under autarky.*

The intuition is as follows. The environmental policy affects only crops produced domestically. As the range of domestic products is smaller under free trade than under autarky, the impact of the environmental policy on consumer welfare is reduced on the consumption side (prices affected by the tax are only those produced by Home) while it is unchanged on the environmental side. It is thus optimal to raise the tax compared to under autarky. Trade creates a *pollution shifting* effect: while consumers benefit from the low prices allowed by pesticides used abroad, they want pesticides use restricted domestically to reduce pollution. Observe that the resulting situation is not Pareto optimal: indeed, if the two countries could agree on tax levels, each would have to account for the price effect of its tax on the other country's consumers. In our setup, the resulting Pareto optimal tax level is the autarky one.[21]

[21] Indeed, for any sharing $(q, 1-q)$ of the agricultural revenue $2w_A$, the Pareto optimal tax levels solve $\min_t 2(1-b)q\ln t - hN(2\ln t)^{1/2}$ and the equivalent program for Foreign.

Now suppose that governments are strategic in the sense that they take into account the effect of the tax on their shares of agricultural revenue. Using (III.12) and (III.27) we obtain that Home's best-response to t^* solves

$$\max_{t,q}\left\{\ln\left(1+\frac{2q(1-b)}{b}\right)-(1-b)\left[\int_0^q \ln \bar{p}_s(z)dz+\int_q^1 \ln \bar{p}_s^*(z)dz\right]-hZ : q=\xi^{-1}\left(\frac{t^*}{t}\right)\right\}.$$

It is implicitly defined by

$$\sqrt{2\ln t}\left\{q+\left[\frac{2}{b+2q(1-b)}\right]\frac{t^*}{t\xi'(z_s)}\right\}=\frac{Nh}{1-b}. \tag{III.30}$$

At a symmetric equilibrium, i.e. $q=1/2=z_s$, $t=t^*$, which implies that $A(1/2)=1$ and thus $\xi'(z_s)=A'(1/2)+4$, we get

$$\sqrt{2\ln t}=\frac{Nh}{1-b}\left[1+\frac{A'(1/2)}{A'(1/2)+8}\right].$$

The following proposition characterizes the optimal policy at equilibrium:

Proposition 3. *Suppose that there are no biodiversity effects. Then, at the symmetric strategic trade equilibrium, the environmental tax τ verifies $2\tau_A>\tau\geq\tau_A$, with $\tau=\tau_A$ in the limit case where $A'(1/2)=0$. Moreover, the steeper the comparative advantage function $A(z)$, the larger the environmental tax.*

Proof: see the appendix.

When comparative advantages are not too different, allowing farmers to use more pesticides could have a large impact on the country's market share of agricultural products. At the symmetric equilibrium, countries do not gain market share, but as this rivalry counteracts the *pollution-shifting* effect described above, this ineffective competition in terms of market share results in a situation which is a Pareto improvement compared to the non-strategic one.

III.2.2 Biodiversity effects

Biodiversity effects create two countervailing distortions in the governments' trade-off we have described above: On the one hand, as specialization increases the production externality, governments should be induced to lower the tax on pesticides. On the other hand, as the externality limits specialization, the effect of the tax on prices concerns a reduced set of crops, which should induce governments to increase the tax. To give a comprehensive appraisal of these countervailing effects, we consider that the relative potential yield function is given by

$$A(z)=\frac{1+m(2z-1)}{1-m(2z-1)} \tag{III.31}$$

where $0 < m < 1$. We have $0 < A(0) < 1$, $A(1) > 1$, $A(1/2) = 1$ and

$$A'(z) = \frac{4m}{[1 - m(2z-1)]^2}.$$

Hence, the larger m, the larger the discrepancy between the countries' relative potential yield away from $z = 1/2$ (graphically, the relative potential yield curve becomes steeper when m increases).

At a symmetric equilibrium, threshold crops given by (III.15) and (III.16) simplify to

$$\bar{z} = \frac{1}{2} + \frac{t\kappa}{2m(1+t\kappa)} \tag{III.32}$$

and

$$\underline{z} = \frac{1}{2} - \frac{t\kappa}{2m(1+t\kappa)}. \tag{III.33}$$

They are equally distant from the center of the range of crops (1/2), and the length of the subset of crops produced by both countries,

$$\bar{z} - \underline{z} = \frac{1}{m}\frac{t\kappa}{1+t\kappa}, \tag{III.34}$$

increases with κ and t and decreases with the relative potential yield parameter m.

In the non-strategic case, the condition that determines t at the symmetric equilibrium can be written as $(\partial V/\partial t)_{t^*=t} = 0$ where

$$\left.\frac{\partial V}{\partial t}\right|_{t^*=t} = -\frac{1-b}{t}\left[\frac{1+4t\kappa}{1+2t\kappa}\underline{z} + (\bar{z} - \underline{z})\frac{1+2t\kappa}{2(1+t\kappa)}\right] - h\frac{dZ}{dt}. \tag{III.35}$$

The last term corresponds to the environmental impact on consumers, which is positive since $dZ/dt < 0$. It leads the government to increase the environmental tax. The bracketed term is composed of two elements, the first one corresponding to the price effect on the goods produced locally and the second one to the price effect on the goods produced by both countries. In these terms, biodiversity effects are ambiguous. Indeed, using (III.33) and (III.34), the effect on goods produced locally can be rewritten as

$$\frac{1+4t\kappa}{1+2t\kappa}\underline{z} = \left(1 + \frac{2t\kappa}{1+2t\kappa}\right)\left(\frac{1}{2} - \frac{\bar{z} - \underline{z}}{2}\right)$$

which shows that compared to the case where $\kappa = 0$, intensification of farming has an ambiguous impact: in the first bracket, the fraction $2t\kappa/(1+2t\kappa)$ tends to reduce the tax on the crops produced locally while the second bracketed term highlights that the range of crops specific to Home is not half of the total but is reduced by $(\bar{z} - \underline{z})/2$, which tends to increase the tax. This effect due to the decrease in the range of specific crops exceeds the one concerning crops produced by both countries:

indeed, we have

$$\frac{\bar{z}-\underline{z}}{2}\left(1+\frac{2t\kappa}{1+2t\kappa}-\frac{1+2t\kappa}{1+t\kappa}\right)=\frac{\bar{z}-\underline{z}}{2}\left(\frac{2t\kappa}{1+2t\kappa}-\frac{t\kappa}{1+t\kappa}\right)>0.$$

The fact that both countries are producing crops belonging to $(\underline{z},\bar{z})$ thus tends to increase the tax level compared to the case where $\kappa = 0$. As a result, the environmental tax could be larger or lower than $2\tau_A$, depending on the relative potential yields of crops. More precisely, we have the following result:

Proposition 4. *Suppose that the relative potential yield function is given by (III.31). Then, at the symmetric non-strategic trade equilibrium, biodiversity effects result in a reduction of the environmental tax compared to the case where $\kappa = 0$ unless m is very small. Overall, the environmental tax is greater than under autarky.*

Proof: see the appendix.

When the discrepancy in relative potential yields is large between the two countries, specialization is important (the range of crops produced by both is relatively small), and the cross externality effect is optimally contained by an intensive use of pesticides.

In the strategic case, there is a marginal effect of the environmental tax on the share of the agricultural revenue that induces governments to reduce their environmental tax. Indeed, the marginal effect of the tax policy on welfare is given by

$$\frac{dV}{dt}=\frac{\partial V}{\partial t}+\frac{\partial V}{\partial q}\frac{dq}{dt}$$

which entails an additional term compared to the non-strategic case, where

$$\frac{\partial V}{\partial q}=\frac{2(1-b)}{b+2q(1-b)}-(1-b)\left[\frac{\underline{z}}{q+t\kappa}-\frac{1-\bar{z}}{1-q+t^*\kappa}+\frac{(\bar{z}-\underline{z})(t^*-t)}{q(t^*-t)+t(1+t^*\kappa)}\right]. \tag{III.36}$$

The first term corresponds to the direct effect on welfare due to the increase in revenue while the remaining terms concern the effects on the price of crops produced domestically, abroad and by both countries respectively. At a symmetric equilibrium, the price effects cancel out, leading to $(\partial V/\partial q)_{t^*=t,q=1/2} = 2(1-b)$. Using (III.31), we obtain that (III.28) is expressed as

$$\left.\frac{dq}{dt}\right|_{t^*=t,q=1/2}=-\frac{3+9t\kappa+4(t\kappa)^2}{12t(1+t\kappa)[m(1+t\kappa)+1]} \tag{III.37}$$

which decreases with κ: the greater the biodiversity effects, the stronger the negative impact of the environmental tax on the share of the agricultural revenue. However, we show in the appendix that

Proposition 5. *Suppose that the relative potential yield function is given by (III.31). Then, at the symmetric strategic trade equilibrium the environmental tax is generally greater than under autarky.*

Proof: see the appendix.

III.3 Trade and volatility

We compare the volatility of crop production through the variation coefficient (VC) which is defined for random variable $\tilde{X}$ as $v(\tilde{X}) \equiv \sigma(\tilde{X})/\mathrm{E}[\tilde{X}]$. Assuming that plots are independently affected by pests, the variance of crop z domestic production is given by $\mathrm{Var}(\tilde{y}(z)) = \bar{a}(z)^2 NB(z)\psi(z)(1-\psi(z))$. Denoting $\mu(z) \equiv \mu_0 \exp(\theta(z)^2/2)$ and using (III.6) and (III.8), we get for a crop produced by Home

$$v(\tilde{y}(z)) = \sqrt{\frac{t[1+t\kappa(z)B(z)] - \mu(z)}{\mu(z)NB(z)}}, \tag{III.38}$$

which increases with t and κ and decreases with N and B. Due to independence, both the variance and the mean increase linearly with N, which results in a negative "scale" effect on volatility. There is also a scale effect associated to intensification (an increase in B) that dominates biodiversity effects. The VCs of the worldwide production under autarky are given by

$$v(\tilde{y}_A^W(z)) = \frac{\sqrt{\mathrm{Var}(\tilde{y}_A(z)) + \mathrm{Var}(\tilde{y}_A^*(z))}}{\tilde{y}_A(z) + \tilde{y}_A^*(z)} = \frac{\sqrt{[\bar{a}(z)^2 + \bar{a}^*(z)^2]N\alpha(z)\psi(z)[1-\psi(z)]}}{[\bar{a}(z) + \bar{a}^*(z)]N\alpha(z)\psi(z)}.$$

Using (III.6), we get

$$v(\tilde{y}_A^W(z)) = \sqrt{\left\{1 - \frac{2A(z)}{[1+A(z)]^2}\right\}\left\{\frac{t_A[1+t_A\kappa(z)\alpha(z)] - \mu(z)}{\mu(z)N\alpha(z)}\right\}}. \tag{III.39}$$

While the second bracketed term in (III.39) is similar to (III.38), the first term reveals a yield effect on production volatility when the same crops are produced by both countries: as $A(z)/[1+A(z)]^2$ is cap-shaped with a maximum at $A(z) = 1$, this effect is decreasing for $z < 1/2$ and increasing for $z > 1/2$. Hence, the yield effect on volatility is higher the larger the difference between the crop yields of the two countries.[22] Assuming symmetry, $\alpha(z) = 1$ and $\theta(z) = \theta$ (which implies $\mu(z) = \mu$ and $\kappa(z) = \kappa$ for all z), the intensification effects are the same for all crops under autarky and for all exclusive crops under trade. This is not the case for crops produced by both countries under trade since farmland intensities vary: while the total share of land devoted to crops at the symmetric equilibrium is the same ($B(z) + B^*(z) = 2$ in any case), the relative importance of Home is decreasing with z (from $B(\underline{z}) = 2$ to $B(\bar{z}) = 0$), whereas it is constant under autarky. Using $\mathrm{Var}(\tilde{y}^W(z)) = \mathrm{Var}(\tilde{y}_T(z)) + \mathrm{Var}(\tilde{y}_T^*(z))$ we have

$$v(\tilde{y}^W(z))^2 = s(z)^2 v(\tilde{y}(z))^2 + s^*(z)^2 v(\tilde{y}^*(z))^2,$$

which gives

$$v(\tilde{y}^W(z)) = \sqrt{\frac{(1+t\kappa)(1+2t\kappa) - \mu\kappa}{2\mu N\kappa} - \frac{2(1+t\kappa)^2}{\mu N\kappa}\frac{A(z)}{[1+A(z)]^2}}. \tag{III.40}$$

[22] With identical yields, i.e. $A(z) = 1$, this term is equal to $\sqrt{2}/2$, the scale effect of a doubling of farmland.

As under autarky, there is a yield effect at work: the volatility index is decreasing over $[\underline{z}, 1/2)$, increasing over $(1/2, \bar{z}]$, and thus reaches a minimum at $z = 1/2$. Comparing the VCs under autarky and trade, we obtain the following result:

Proposition 6. *Without biodiversity effects, trade could potentially reduce the production volatility of all crops. However, because of a higher environmental tax than under autarky only the volatility of crops for which countries have large comparative advantages are reduced (if any). With biodiversity effects, trade increases the production volatility of crops produced by both countries and of the specialized crops with moderate competitive advantage. The volatility of large comparative advantage crops is reduced only if biodiversity effects are small and the environmental tax not too different from its autarky level.*

Proof: see the appendix.

Because the probability of the Bernoulli distribution $\psi(z)$ doesn't depend on N, the distribution of crop z production converges to the normal distribution $\mathcal{N}(y(z), \sigma(\tilde{y}(z)))$ when N is large. This allows us to derive some characteristics of the price distributions. In particular, the break-even price $\bar{p}(z)$ corresponds to crop z's median price: we have $\Pr[\tilde{p}(z) \leq \bar{p}(z)] = \Pr[\tilde{y}(z) \geq y(z)] = 1/2$ since the normal distribution is symmetric. Consequently, as $\bar{p}(z)$ is lower than the average market price $p(z)$ due to the correlation between prices and quantities, the price distribution is asymmetric.

This asymmetry is also revealed by the upper and lower bounds of confidence intervals implied by the distributions of crop production. Denoting by $y_d^\gamma(z)$ and $y_u^\gamma(z)$ the lower and upper quantity bounds of the confidence interval at confidence level $1-\gamma$, the corresponding price bounds are derived from $\Pr[y_d^\gamma(z) \leq \tilde{y}(z) \leq y_u^\gamma(z)] = \Pr[p_d^\gamma(z) \leq \tilde{p}(z) \leq p_u^\gamma(z)]$: we have $p_u^\gamma(z) \equiv \alpha(z)(1-b)LR/y_d^\gamma(z)$ and $p_d^\gamma(z) \equiv \alpha(z)(1-b)LR/y_u^\gamma(z)$. Because of the symmetry of the quantity distributions, $y_d^\gamma(z)$ and $y_u^\gamma(z)$ are equally distant from $y(z)$, but since prices and quantities are inversely related, this is not the case for $p_d^\gamma(z)$ and $p_u^\gamma(z)$. The following proposition completes these general features of the price distributions with some useful approximations:[23]

Proposition 7. *The expected value and the standard deviation of crop prices are approximated by*

$$p(z) \approx \bar{p}(z)[1 + v(\tilde{y}(z))^2] \tag{III.41}$$

and

$$\sigma(\tilde{p}(z)) \approx \bar{p}(z)v(\tilde{y}(z))\sqrt{1 - v(\tilde{y}(z))^2}.$$

Confidence intervals at confidence level $1-\gamma$ *are delimited by* $p_u^\gamma(z) = p(z) + s_u^\gamma \sigma(\tilde{p}(z))$ *and* $p_d^\gamma(z) = p(z) + s_d^\gamma \sigma(\tilde{p}(z))$ *with*

$$s_u^\gamma \approx \frac{v(\tilde{y}(z)) + s_\gamma}{\sqrt{1 - v(\tilde{y}(z))^2}[1 - s_\gamma v(\tilde{y}(z))]} \tag{III.42}$$

[23] Where Φ denotes the cumulative distribution function of the standard normal distribution.

and

$$s_d^\gamma \approx \frac{v(\tilde{y}(z)) - s_\gamma}{\sqrt{1 - v(\tilde{y}(z))^2}[1 + s_\gamma v(\tilde{y}(z))]} \tag{III.43}$$

where $s_\gamma \equiv \Phi^{-1}(1 - \gamma/2)$. *They are approximately equal to*

$$p_u^\gamma(z) \approx \bar{p}(z)\left[1 + v(\tilde{y}(z))^2 + v(\tilde{y}(z))\frac{v(\tilde{y}(z)) + s_\gamma}{1 - s_\gamma v(\tilde{y}(z))}\right] \tag{III.44}$$

and

$$p_d^\gamma(z) \approx \bar{p}(z)\left[1 + v(\tilde{y}(z))^2 + v(\tilde{y}(z))\frac{v(\tilde{y}(z)) - s_\gamma}{1 + s_\gamma v(\tilde{y}(z))}\right]. \tag{III.45}$$

Proof: see the appendix.

Because prices and quantities are inversely related, we have $s_d < s_u$, i.e the price distribution is skewed to the right: its right tail is longer and fatter than its left tail. The consequences on volatility are that the chances that a crop price is very low compared to the expected price, i.e., $\tilde{p}(z) \leq \bar{p}(z) < p(z)$, are larger than the chances of a high price, i.e. $\tilde{p}(z) > p(z)$, since $1/2 = \Pr[\tilde{p}(z) \geq \bar{p}(z)] > \Pr[\tilde{p}(z) > p(z)]$, but the possible range of high prices is wider than the range of low prices: $p_u(z) - p(z) > p(z) - p_d(z) > \bar{p}(z) - p_d(z)$.[24]

Figures III.1 and III.2 illustrate these findings. The solid curves with the marks depict the crop average price in each case as indicated (depicted are Home's autarky prices). The corresponding dashed curves depict the approximate values of $p_d(z)$ and $p_u(z)$. The vertical distance between these curves defines a confidence interval for a crop price at a confidence level equal to 95%. Compared to autarky, the non-strategic average prices are larger for almost all crops, and the confidence intervals are very large. This is due to the tightening of the pesticide regulations mentioned above. The strategic effects that loosen these regulations induce lower average prices and confidence intervals. Biodiversity effects are reflected in Fig. III.2 by strategic and non-strategic average price curves that encompass a flat portion around $z = 1/2$ which corresponds to the mix-production range. Confidence intervals over these ranges are smaller the closer the crop is to $z = 1/2$.

Tables III.1 and III.2 assess the way the biodiversity effects and potential yield differentials affect the trade pattern, the environmental tax policy, the food price average level and the crop price volatilities.[25]

Table III.1 summarizes the impacts due to the biodiversity effects: the larger κ is, the larger the negative impact of the cultivation intensity on the survival probability of plots. Biodiversity effects

[24] This is of course dependent on the convexity of the demand function. Indeed, the condition $p_u(z) - p(z) > p(z) - p_d(z)$, is equivalently written $p(z) < [p_u(z)+p_d(z)]/2 = [D^{-1}(y_d(z))+D^{-1}(y_u(z))]/2$. As $p(z) > \bar{p}(z) = D^{-1}(y(z))$ where $y(z) = [y_d(z) + y_u(z)]/2$, a necessary condition is $D^{-1}([y_d(z) + y_u(z)]/2) \leq [D^{-1}(y_d(z)) + D^{-1}(y_u(z))]/2$, hence $D^{-1}(y)$ must be convex from the Jensen's inequality.

[25] The tables report the tax as a percentage of production costs τ/c (c is the same whatever the case at hand). The food price index is defined as $\int_0^1 p(z)y(z)dz \Big/ \int_0^1 y(z)dz$ and approximated as detailed in the appendix. A food price index of 2 means that the average price of agricultural goods equals 2% of the industrial goods price. The VC of domestic crop prices is the same for all crops under autarky. Worldwide VCs indicated under autarky are derived from a fictitious price distribution corresponding to the sum of Home and Foreign productions. Worldwide VCs evolve between minimum value $v(\tilde{p}^W(1/2))$ and maximum values $v(\tilde{p}^W(0))$ (autarky) and $v(\tilde{p}^W(\underline{z}))$ (trade).

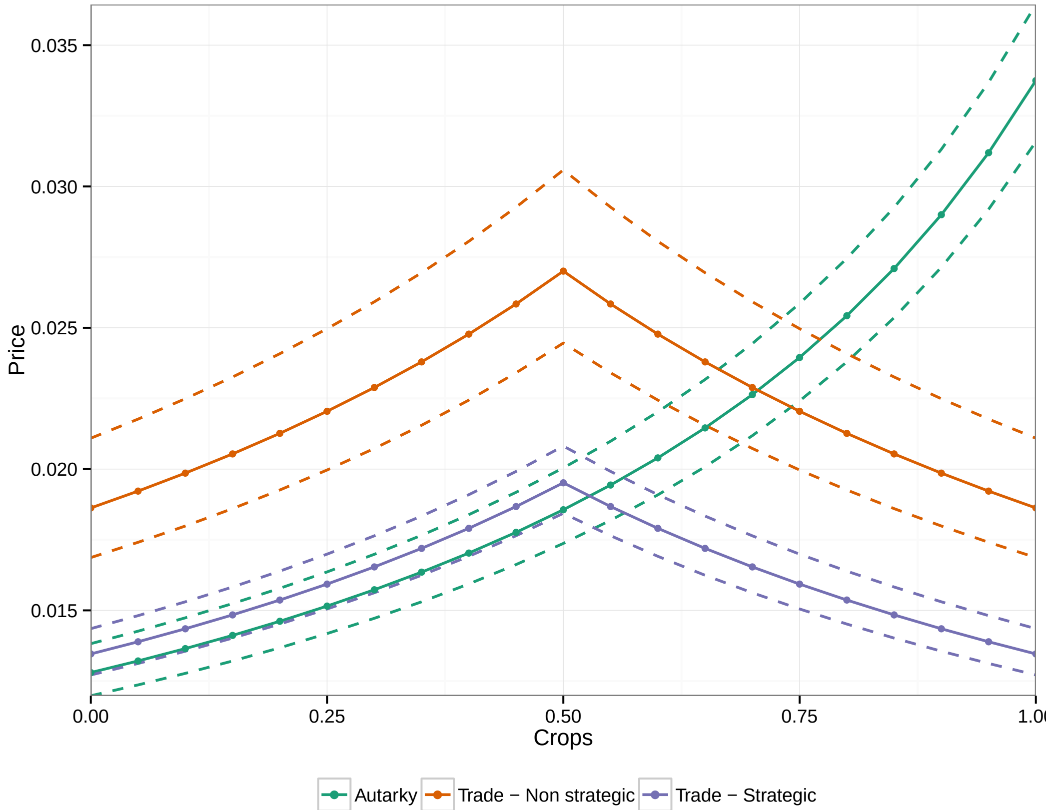


Figure III.1. Average prices and price volatility (confidence interval at 95% confidence level) without biodiversity effects. $\kappa = 0$, N=100, $h = 10^{-3}$, $b = 0.8$, $\ell = 20$, $m = 0.45$, $\mu = 1$, $a(1/2) = 29$.

play against the specialization induced by trade: the range of crops produced by both countries increases when κ rises. Since biodiversity effects impede production, prices increase with κ as shown by the trend in the food price index reported in table III.1. The larger κ is, the more effective pesticides are in reducing the negative externality due to the cultivation intensity and, therefore, the lower the environmental tax. Nevertheless, pesticides do not allow farmers to eradicate the biodiversity effect. Thus, even if more pesticides are applied when κ increases, the food price volatility increases. The tax on pesticides is more stringent under trade than under autarky, even when trade is strategic, for the reasons detailed above. In our simulations, non strategic taxes are more than 58% higher than strategic ones. Food price index levels reported in table III.1 show that the decrease in food prices due to trade is lower the larger the biodiversity effects. The ratio between the VC levels at $z = 1/2$ of the worldwide production and the domestic production under autarky gives the size of the scale effect on prices (e.g. for $\kappa = 0.1$, it corresponds to 5.46/8.62=63.3% which is lower than on the production side $\sqrt{2}/2 \approx 70.7\%$). The yield effect is maximum for crop $z = 0$: for $\kappa = 0.1$, it corresponds to an additional (6.24-5.46)/8.62=9%. Table III.1 also illustrates that the price volatility increases for crops produced by both countries even for small values of κ because

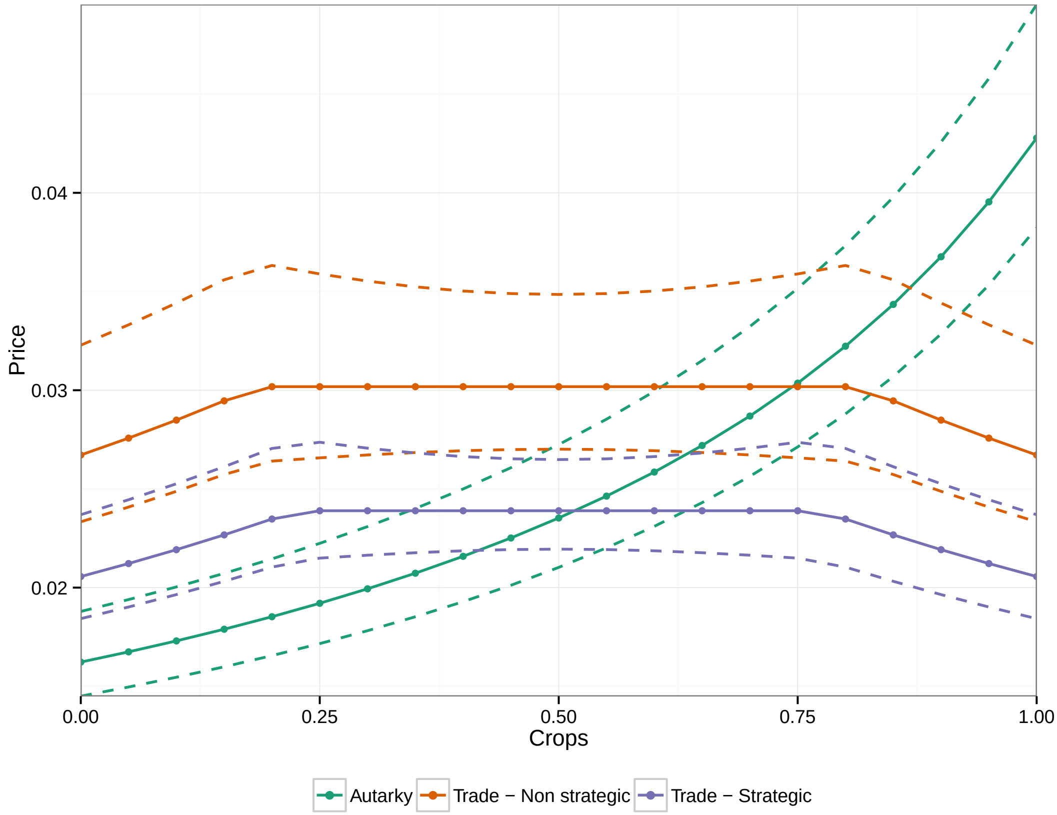


Figure III.2. Average prices and price volatility (confidence interval at 95% confidence level) with biodiversity effects. κ **= 0.3,** N**=100,** $h = 10^{-3}$**,** $b = 0.8$**,** $\ell = 20$**,** $m = 0.45$**,** $\mu = 1$**,** $a(1/2) = 29$**.**

of the significant increases in the environmental tax under strategic trade compared to autarky: even for $\kappa = 0.1$, the volatility of the specialized crops (7.07%) is larger than under autarky (6.24%).

Table III.2 describes the impact of the potential crop yield differentials: the larger m is, the larger the difference in the potential yield of each crop (away from $z = 1/2$), and the greater the difference in comparative advantages of the two countries. The range of crops that are produced by both countries under free trade $(\bar{z} - \underline{z})$ decreases with m. Effects of m on the environmental tax depend on whether trade is strategic or not. Under non strategic trade, the impact of the environmental policy on consumer welfare goes through the crop prices only. Therefore, the larger m is, the larger the specialization and the lower the tax on pesticides. Under strategic trade, the use of pesticides may have a significant impact on the market share of the country, particularly when comparative advantages are sufficiently close. This is reflected in table III.2: the smaller m is, the lower the environmental tax. The way the food price index varies with m also differs between the non strategic and the strategic case. Two effects play on the quantities produced: on the one hand, when m raises, the productivity increases and, on the other hand, when the environmental tax increases, the quantities produced decrease. In the non strategic case, when m increases, the

Table III.1. Sensitivity analysis on parameter κ

	Values of κ				
Variables (%)	0.1	0.3	0.5	0.7	0.9
Autarky					
τ/c	45.46	40.15	37.07	35.05	33.61
Food price index	2.03	2.38	2.73	3.09	3.45
$v(\tilde{p}(\cdot))$	8.62	10.10	11.39	12.53	13.57
$v(\tilde{p}^W(1/2))$	5.46	6.29	6.99	7.60	8.14
$v(\tilde{p}^W(0))$	6.24	7.16	7.94	8.61	9.20
Non strategic trade					
τ/c	83.29	73.23	69.03	66.49	64.68
Food price index	2.28	2.83	3.35	3.85	4.32
$v(\tilde{p}^W(1/2))$	8.00	8.84	9.76	10.62	11.42
$v(\tilde{p}^W(\underline{z}))$	8.82	10.61	12.19	13.55	14.75
$\bar{z}-\underline{z}$	20.66	46.96	64.70	77.69	87.65
Strategic trade					
τ/c	52.51	44.83	40.99	38.53	36.76
Food price index	1.77	2.25	2.68	3.08	3.46
$v(\tilde{p}^W(1/2))$	6.39	7.37	8.28	9.10	9.85
$v(\tilde{p}^W(\underline{z}))$	7.07	8.91	10.42	11.70	12.84
$\bar{z}-\underline{z}$	17.16	41.51	58.71	71.64	81.76

Note: Parameters' values are $m = 0.6$, N=100, $h = 10^{-3}$, $b = 0.8$, $\ell = 20$, $\mu = 0.7$, $a(1/2) = 290$.

tax decreases. Then, the two effects go in the same direction, the quantities produced increase and prices decline. Under strategic trade, m and the environmental tax both increase, their effects are countervailing on the quantities produced. The impact of the increase in the productivity prevails and the food price index decreases, even if the environmental tax is raised. Finally, in both strategic and non strategic cases, price volatility increases when the use of pesticides declines, i.e. when the environmental tax raises.

III.4 Fertilizers

Our focus being on biodiversity effects in agricultural production, we have so far not discussed the impact of trade on the use of fertilizers. However, because they have considerable effects on both crop yields and on the environment (and human health), changes in the openness of countries to trade are likely to impact the way their use is regulated.[26] We may thus expect that crop

[26] Commercial fertilizers are responsible for 30% to 50% of crop yields (Stewart et al., 2005). Sutton et al. (2011a,b) find that half of the nitrogen added to farm fields ends up polluting water or air. Excess of nitrogen and phosphorus in freshwater increases cancer risk and creates aquatic and marine dead zones through eutrophication. In the air, nitrates contribute to ozone generation which causes respiratory and cardiovascular diseases. Sutton et al. (2011a) estimates that in the European Union the benefits of nitrogen for agriculture through the increase in yields amount to 25 to 130 billion euros per year and that they cause between 70 and 320 billion euros per

Table III.2. Sensitivity analysis on parameter m

	Values of m			
Variables (%)	0.2	0.4	0.6	0.8
Autarky				
τ/c	45.46	45.46	45.46	45.46
Food price index	2.03	2.03	2.03	2.03
$v(\tilde{p}(\cdot))$	8.62	8.62	8.62	8.62
$v(\tilde{p}^W(1/2))$	5.46	5.46	5.46	5.46
$v(\tilde{p}^W(0))$	5.55	5.83	6.24	6.76
Non strategic trade				
τ/c	86.02	83.94	83.29	82.97
Food price index	2.70	2.47	2.28	2.12
$v(\tilde{p}^W(1/2))$	8.19	8.05	8.00	7.98
$v(\tilde{p}^W(\underline{z}))$	9.02	8.87	8.82	8.80
$\bar{z}-\underline{z}$	63.23	31.13	20.66	15.46
Strategic trade				
τ/c	46.82	49.83	52.51	54.81
Food price index	2.00	1.88	1.77	1.68
$v(\tilde{p}^W(1/2))$	6.18	6.29	6.39	6.49
$v(\tilde{p}^W(\underline{z}))$	6.84	6.96	7.07	7.17
$\bar{z}-\underline{z}$	50.19	25.43	17.16	13.01

Note: Parameters values are $\kappa = 0.1$, N=100, $h = 10^{-3}$, $b = 0.8$, $\ell = 20$, $\mu = 0.7$, $a(1/2) = 290$.

price volatility is also affected through this channel. It is possible to analyze these changes by considering that crop z's potential yield $\bar{a}(z)$ is the result of the intrinsic quality of land and the quantity of fertilizers spread on the field, $g(z)$. Denoting by $a_0(z)$ the potential crop z yield absent any treatment, we have $\bar{a}(z) = a_0(z)f(g(z))$ with $f(0) = 1, f'(g) > 0$ and $f''(g) < 0$. Total use of fertilizers, given by $G = N\int_0^1 B(z)g(z)dz$, has a negative impact on consumer welfare due to environmental damages. As pesticides, fertilizers have a direct positive impact on crop yields, but unlike pesticides, their productive impact is limited to the field they are spread on. Hence, the trade-off that defines the fertilizer policy is similar to the one of the pesticide regulation without biodiversity effects. While under autarky domestic consumers bear all the costs and reap all the benefits of the fertilizers used by their fellow farmers, this is no longer the case in free trade: they benefit from the crops produced abroad and share the advantages of a productive national sector with foreign consumers. As a result, restrictions on fertilizers are tighter under free trade than under autarky, with the same caveat as for pesticides: governments may use the fertilizer policy strategically. How lenient they are depends on the impact of fertilizers on relative yields: the more responsive is the relative yields function, i.e. the larger $f'(g)$, the lower the restrictions.

year in damage.

III.5 Conclusion

We have shown that biotic risk factors such as pests that affect the productivity of farming create biodiversity effects that modify standard results of trade models. An explicit account of their effects on production allows us to clarify the distribution of idiosyncratic risks affecting farming which depends on the countries' openness to trade. These production shocks translate into the price distribution and are impacted by environmental policies. Of course, these productive shocks are not the only ones affecting food prices, but they are an additional factor in their distributions that may explain their greater volatility compared to manufacture prices.

While these effects are analytically apparent within the standard two-country Ricardian model, an extension to a more encompassing setup involving a larger number of countries as permitted by Eaton and Kortum (2002) and applied to agricultural trade by Costinot and Donaldson (2012) and Costinot et al. (2012) is necessary to investigate their scope statistically. These studies incorporate a stochastic component to determine the pattern of trade but it is not related to the production process and somehow arbitrary. Our analysis offers an interesting route to ground these approaches at least in the case of agricultural products. Introducing these biodiversity effects should allow for a better assessment of the importance of trade costs in determining the pattern of trade.

CHAPTER IV

ESTIMATING THE IMPACT OF CROP DIVERSITY ON AGRICULTURAL PRODUCTIVITY IN SOUTH AFRICA

Diversity plays a key role in the resilience to external stresses of farm plants and animals. In particular, crop species diversity increases productivity and production stability (Tilman et al., 2005, Tilman and Downing, 1994, Tilman et al., 1996) in the sense that the probability to found at least one individual that resists to an adverse meteorological phenomenon (for example a drought or a heatwave), or pests and diseases, increases with the diversity within a population. Furthermore, the larger is an homogeneous population, the larger is the number of parasites that use this population as a host and therefore the larger is the probability of a lethal infection (Pianka, 2011). Finally, diversity also allows for species complementarities and therefore a more efficient use of natural resources (Loreau and Hector, 2001). For these reasons, the expansion of monocultures observed in many modern agricultural systems is presented as one of the causes of their increased vulnerability and, consequently, of the increased use of agrochemicals (Landis et al., 2008). In short, crop biodiversity has the potential to enhance resistance to strains due to biotic and abiotic factors and to improve crop production and farm revenues.

Our paper empirically investigates the role of crop biodiversity on agricultural production and on farmer's exposure to risk. We build a probabilistic model to describe crop survival and productivities according to diversity. From this analytic model, we derive a reduced form that is estimated using data on South African agriculture. Our results contribute to the existing literature in three main ways. First, we confirm that diversity[1] has a positive and significant impact on produced quantities, while previous studies sometimes find contrasting results.[2] An increase in biodiversity is equivalent to a third of the benefits of a comparable increase in irrigation, where

Chapter written with Jean-Marc Bourgeon, Eric Strobl and Élodie Blanc.

1 Note that we look at crop diversity, while previous studies mainly investigate the role of intraspecific genetic diversity. For a review of the main studies on the link between diversity and production see Di Falco (2012). In addition to the studies cited there, Carew et al. (2009) and Chavas and Di Falco (2012) also look at intraspecific diversity.

2 For example, Smale et al. (1998), examining the impact of wheat genetic diversity on wheat yield in Pakistan, find a positive impact of diversity in rain fed regions while in irrigated regions higher area concentration on few varieties is associated with higher yields.

irrigation is known to be an important impediment to crop productivity in South Africa, due to unreliable precipitation. Biodiversity also contributes to the reduction of production variance and, thus, to the stabilisation of agricultural revenues. Second, we use a very large dataset of highly disaggregated data at the field level derived from satellite imagery. Previous studies use either aggregate data (Smale et al., 1998) which do not allow for the control of farm characteristics, or farm-level data from household or farm surveys (for example Di Falco and Chavas, 2006, 2009), with fewer observations than those provided by satellite images. Thanks to our large dataset, we show that biodiversity has mainly a local impact. We quantify biodiversity using an index taken from the ecological literature, based on species richness (the total number of species) and their relative abundance, the Shannon index. This index captures the fact that biodiversity is high when the total number of species is large and the distribution of their relative abundances is homogeneous. We measure Shannon indexes on perimeters that differ by their size and find that biodiversity is a significant predictor of crop production on perimeters having a radius smaller than 1.75 kilometre. Third,our approach is built on the results of ecology literature. Unlike previous contributions, our probabilistic model makes explicit the relationship between biodiversity, biotic and abiotic factors that affect agricultural production. It represents how biodiversity impact agricultural production. Stochastic shocks affecting agricultural production are endogenous, in accordance with ecology findings, while they are exogenous in previous literature.

In the remainder of the chapter, the theoretical model that motivates our empirical investigations is developed in section IV.1. Then, the database on South African agriculture is presented in section IV.2. In section IV.3 , we use these data to investigate the impact of crop biodiversity on production levels and variability.

IV.1 A probabilistic model on the impacts of biodiversity on crop production

IV.1.1 The model

A very robust stylised fact in ecology describes the impact of biotic factors on agricultural production: the more area is dedicated to the same crop, the more pests specialise on this crop and the higher is the frequency of their attacks (Pianka, 2011). Relying on this stylised fact, we build a general probabilistic model of crop production where crops are affected by both abiotic (i.e. weather, water availability, soil properties. . .) and biotic (i.e. pests) factors causing pre-harvest losses.[3] More precisely, we consider that the total agricultural production depends on the survival probability

[3] Losses due to biotic factors can be significant. Oerke (2006) finds that, during the 2001-2003 period, without crop protection, losses due to pests for wheat, rice, maize, soybeans and cotton totalled respectively 49.8%, 77%, 68.5%, 60% and 80% of the potential yield, at the world level. Thanks to crop protection, actual losses were about 28.2%, 37.4%, 31.2%, 26.3% and 28.8% respectively. Fernandez-Cornejo et al. (1998) compile data on pesticide use in the US and find similar results: estimated yield losses from insects and diseases range between 2 to 26% and losses from weeds range between 0% and 53%.

of each crop, which is directly linked to the probability of a pest attack. The frequency at which pest attacks occur (and consequently the crop survival probability) is linked to the way crops are produced: the more diverse are crops, the lower is the probability of a pest attack, the higher is the survival probability and therefore the higher is the expected agricultural production.

To describe these impacts, we adopt a beta-binomial distribution, which is often used in ecology and plant physiology literature to describe spatial distributions that are not random but clustered, patchy or heterogeneous (Hughes and Madden, 1993, Shiyomi et al., 2000, Chen et al., 2008, Bastin et al., 2012, Irvine and Rodhouse, 2010). We assume that a region (or a country) produces Z different crops on I fields of the same size, each field i containing $n(z)$ patches of land. The size of these patches depending on the cultivated crop z, the number of patches on field varies (we thus have $n(z)$ patches on a field of crop z). On field i, only one crop is produced with a potential yield a_i for all its patches provided they are resilient enough. Indeed, patches are subject to potential lethal strains due, for example, to adverse meteorological conditions or pathogens. We suppose that a patch is destroyed with probability $1-\lambda$ from one (or several) adverse condition and that otherwise it produces the potential yield independently of the fate of the other patches.[4] With n patches devoted to the same crop within a field, the probability of having x patches unaffected (and thus $n-x$ destroyed) follows a binomial distribution

$$\Pr\{\widetilde{X}=x|n\}=\binom{n}{x}\lambda^x(1-\lambda)^{n-x}$$

where $\widetilde{X}$ is the random variable that corresponds to the number of patches that are indeed harvested among the n patches of a cultivated field. We consider that the survival probability of the patches of a field, λ, is identically and independently distributed. However, this probability may vary across fields of the same crop (we generally have $\lambda_i \neq \lambda_j$ for any couple of fields (i,j) of the same crop): it depends on natural conditions but also on the way crops are cultivated, and in particular, on the diversity of crops, measured by the Shannon index (for further details on this index, see section IV.2.4). More precisely, we assume that the survival probability of patches on a given field is a draw from a Beta distribution which is given by

$$\Pr\{\tilde{\lambda}=\lambda|S\}=\frac{\Gamma(\gamma+\beta-kS)}{\Gamma(\gamma)\Gamma(\beta-kS)}\lambda^{\gamma-1}(1-\lambda)^{\beta-kS-1}$$

where γ, β and k are positive parameters that depend on the crop considered (hence $\gamma(z)$, $\beta(z)$ and $k(z)$), $\Gamma(\cdot)$ the gamma function and S the Shannon index as defined in (IV.9). While γ and β determine the randomness of the survival probability of a crop cultivated intensively, the parameter k capture the effects of the distribution of crops in the perimeter ℓ.

[4] Obviously, this is a strong assumption. Pests and/or weather do not necessarily totally destroy a patch, but rather affect the quantity of biomass produced. But, in order to maintain tractability, we consider that a patch is either unaffected either totally destroyed, rather than partially affected, by adverse conditions. Thus, our random variable is the number of harvested patches rather than the share of biomass that is lost on each patch.

The expected number of patches among $n(z)$ that are harvested on field i is given by $\mathrm{E}(\widetilde{X}|z_i, S_i) = n(z)\psi(z_i, S_i)$ where

$$\psi(z, S) = \mathrm{E}[\tilde{\lambda}(z)|S] = \frac{\gamma(z)}{\gamma(z) + \beta(z)} \frac{1}{1 - \kappa(z)S} \tag{IV.1}$$

is the expected probability that a particular patch of the field i is harvested when the crop diversity is S_i and where $\kappa(z) = k(z)/(\gamma(z) + \beta(z))$. This survival probability is the product of two terms. The first one, $\gamma(z)/(\gamma(z) + \beta(z))$, corresponds to the expected resilience of a particular stand of crop when the intensity of production is maximal ($S = 0$). It increases with γ and decreases with β. The second term, $(1 - \kappa S)^{-1}$, is a scale factor which increases with the overall crop diversity S_i. The parameter $\kappa(z)$ determines the intensity of the impact of crop biodiversity. No production externality due to crop diversity would correspond to $\kappa(z) = 0$. We expect this parameter to be strictly positive, so that the crop survival probability, and thus crop production, increases with crop diversity.

The variance of the number of harvested patches also depends on the crop intensity. It is given by

$$\sigma^2_{\tilde{X}} = n(z)\sigma^2(z, S)$$

where

$$\sigma^2(z, S) = \psi(z, S)[1 - \psi(z, S)]\{1 + [n(z) - 1]\rho(z, S)\} \tag{IV.2}$$

with $\rho(z, S) = \{[\gamma(z) + \beta(z)][1 - \kappa(z)S] + 1\}^{-1}$ or, equivalently $\rho(z, S) = (\gamma(z) + \varphi(z, S) + 1)^{-1}$ where $\varphi(z, S) = \beta(z)[1 - \kappa(z)S] - \gamma(z)\kappa(z)S$. Equation (IV.2) corresponds to the variance of the survival probability of one patch when the overall crop diversity is S. Compared to the Bernoulli distribution, (IV.2) contains an additional term that accounts for the correlation between patches induced by the common distribution of the survival probability over fields, the correlation coefficient being given by $\rho(z, S)$.

The production on field i is given by $\widetilde{Y}_i = a_i \widetilde{X}_i$. It can be equivalently written as

$$\widetilde{Y}_i = \mathrm{E}(\widetilde{Y}_i)(1 + \tilde{\varepsilon}_i) \tag{IV.3}$$

where $\tilde{\varepsilon}_i = [\widetilde{X}_i - \mathrm{E}(\widetilde{X}_i)]/\mathrm{E}(\widetilde{X}_i)$ has a mean equal to 0 and a variance given by

$$\sigma^2_{\varepsilon_i} = \frac{1 - \psi_i}{\psi_i} \left(\frac{1}{n(z)} + \frac{n(z) - 1}{n(z)} \rho_i \right) \approx (\psi_i^{-1} - 1)\rho_i \tag{IV.4}$$

when $n(z)$ is large. This variance is mainly due to the correlation between the distributions of the survival probabilities across fields, captured by ρ. Indeed, $\tilde{\lambda}$ follows a beta distribution but the parameters of the distribution depend on the crop diversity surrounding each field, S_i, and are thus different across fields. In other words, with a sufficiently large number of patches on each field, the difference in the quantities produced is mainly driven by the crop biodiversity.

From this simple probabilistic model, we investigate the contribution of crop biodiversity, measured by the Shannon index, to the mean, the variance and the skewness of crop production, using data on South African agriculture.

IV.1.2 Empirical strategy

From our database, we derive Z_i, a measure of the agricultural production on field i, using a vegetation index (see section IV.2.2). Supposing that n is large so that (IV.4) is verified, we can estimate $\mathrm{E}(\widetilde{Y}_i) = a_i n \psi_i$ with

$$\mathrm{E}(\widetilde{Y}_i) = \frac{\prod_k \theta_k \exp(X_{ik})}{1 - \kappa_0 S_{i\ell}} \frac{n\gamma}{\gamma + \beta}$$

where X_{ik} is the vector of k control variables derived from the database and $S_{i\ell}$ is the Shannon index measured on each field i, for different perimeters ℓ. The term $\prod_k \theta_k \exp(X_{ik})$ captures all the effects that are specific to the field i and that determine the potential yield of this field, a_i.

From (IV.8), (IV.3) and (IV.1), we have

$$\ln Z_i = \frac{1}{c} \left[\sum_k \theta_k X_{ki} + \ln(1 - \kappa S_i) + \ln\left(\frac{n\gamma}{\gamma + \beta}\right) - \ln b + \ln(1 + \varepsilon_i) \right] \tag{IV.5}$$

The following regression specification

$$\ln Z_i = \hat{\theta}_0 + \sum_k \hat{\theta}_k X_k + \hat{\theta}_S S_{i\ell} + \epsilon_i \tag{IV.6}$$

leads to estimate κ by $\hat{\kappa} = \hat{\theta}_S / c$, assuming that κS is small so that we can approximate $\ln(1 - \kappa S)$ by κS. The error terms ϵ_i give an estimation of the series of $\varepsilon_i \approx c\epsilon_i$ (assuming that ε_i is small). Then, the regression of $\hat{\varepsilon}_i^2$ on S_i allows us to quantify the contribution of crop biodiversity to the variability of crop production.

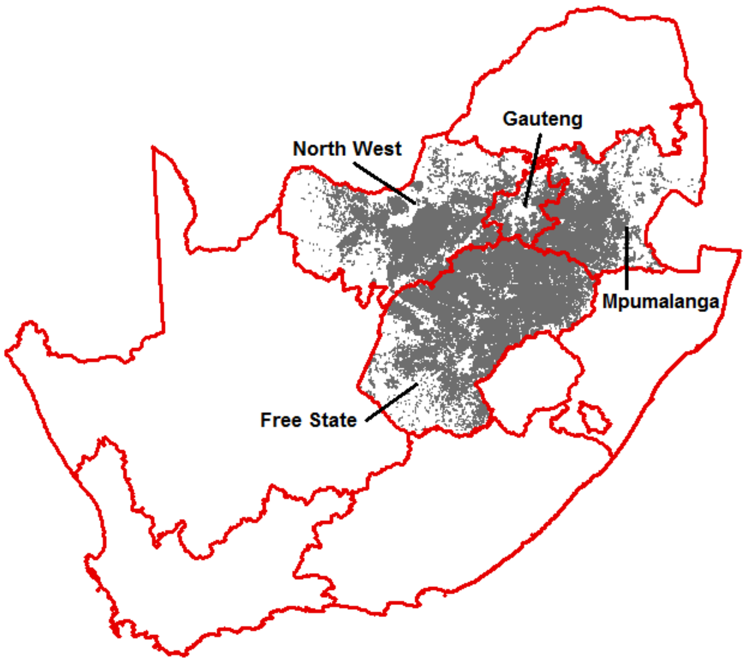
IV.2 Data

We combine different data sources to build a very detailed database on South African agriculture. First, field boundaries are identified, then agricultural production is characterised on each field by identifying the crops that are grown, measuring the biomass produced, determining the production system (rain fed or irrigated) and quantifying crop interspecific biodiversity.

IV.2.1 Crop fields

Field boundaries, available for the provinces of Free State, Gauteng, North West and Mpumalanga, are determined using the Producer Independent Crop Estimate System (PICES) which combines satellite imagery, Geographic Information System (GIS), point frame statistical platforms, and aerial observations (Ferreira et al., 2006). Satellite imagery of cultivated fields is obtained from

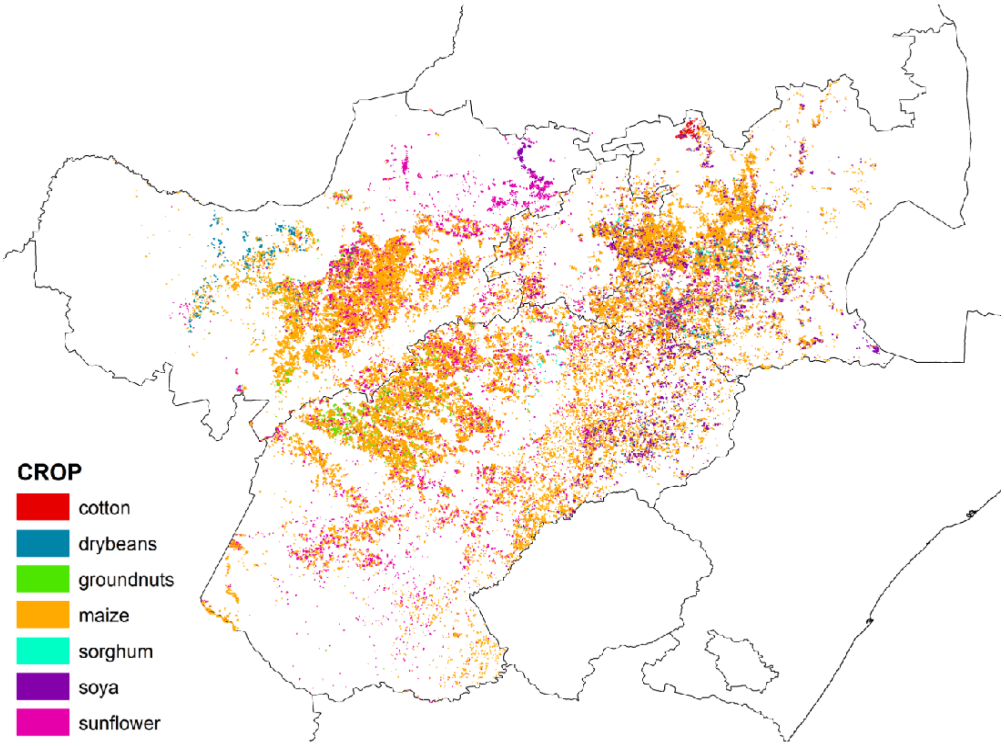
Figure IV.1. Localisation of the considered fields in South Africa



the SPOT 5 satellite at a 2.5-m resolution. Plot boundaries are then digitised using GIS and field cloud covered polygons are removed before processing. Over the four regions of interest, PICES distinguishes circa 280 000 fields covering an area of around 6.5 million hectares. To approximately match the resolution of the crop production indicator described below (see IV.2.2), which is only available at the 250-m resolution, the analysis is limited to fields larger than 6.25 ha. This restricts the sample to 213 110 fields. Figure IV.1 presents the location of the considered crop fields in South Africa. While the summary statistics in Table IV.1 show that fields are on average about 28.4 hectares, the large standard deviation (24.8) indicates that they vary substantially in size. For instance, the largest field is 720 hectares large, while the smallest is only 6.25.

Using the digitised satellite images described above, the Agricultural Geo-referenced Information System (AGIS) developed by the South African Department of Agriculture provides information on the crop cultivated on each field. To do so, sample points were selected randomly and surveyed by trained observers from a very light aircraft in order to determine crop type (Ferreira et al., 2006). Crop information collected during the aerial surveys on the sample points was subsequently used as a training set for crop type classification for each field and for accuracy assessment. These estimated crop classifications were then checked against a producer based survey for the Gauteng region. The Gauteng census survey showed that less than 1.8% of crop types had been misclassified. All in all, seven summer crops were distinguished for the provinces of Free State, Gauteng, North West and Mpumalanga for the summer season 2006/2007: cotton, dry beans, groundnuts, maize, sorghum, soybean and sunflower. An example of the distribution of crop types is provided in figure IV.2. The summary statistics for the entire sample in Table IV.2 show that maize was the dominant crop

Figure IV.2. Distribution of the studied crops in South Africa



cultivated in the three provinces: maize fields represent nearly 70 per cent of the total number of fields we consider. Other important crops were sunflower and soybean, standing at 15 and 11 per cent each. In contrast, all other crop types constituted less than 2 per cent individually. One should note that even if one were to adjust the crop type shares by their areas, a similar ranking remains, with a slight redistribution of shares towards the smaller crop types. For instance, the share of maize dropped to 62 per cent of the total crop area.

The AGIS crop boundaries dataset also provides information regarding whether each field is irrigated. As can be seen from Table IV.1, only about 5% of all fields were part of an irrigation system.

Finally, all fields can be linked to their respective farms with a unique farm identifier. In total the fields were owned by 12 462 different farms, where on average each farm was proprietor of 5 fields. However, ownership differed substantially, with the largest ownership gathering 193 fields, and 3 704 single field farms.

IV.2.2 Crop production measure

We estimate crop biomass production using the satellite derived normalised difference vegetation index (NDVI). Vegetation indices provide consistent spatial and temporal representations of vegetation conditions, when locally derived information is not available. As a matter of fact, numerous studies have demonstrated that NDVI values are significantly correlated with biomass production, and

Table IV.1. Plot summary statistics

Variable	Mean	Standard deviation	Variable	Mean	Standard deviation
All crops			Maize		
NDVI	0.61	0.13	NDVI	0.61	0.13
Water balance	−41.07	11.73	Water balance	−42.60	11.89
Season length (days)	129.64	35.48	Season length (days)	126.98	35.33
Plot area(ha)	28.36	23.47	Plot area(ha)	22.72	2.04
Farm area (ha)	315.2	500.00	Farm area (ha)	193.63	3.53
Irrigation (%)	5.05	–	Irrigation (%)	4.94	21.67
Longitude	27.7	1.48	Longitude	27.69	1.49
Latitude	−27.1	0.99	Latitude	−27.17	0.99
Cotton			Sorghum		
NDVI	0.73	0.05	NDVI	0.69	0.10
Water balance	−28.61	14.54	Water balance	−33.54	7.70
Season length (days)	130.94	52.00	Season length (days)	125.20	26.17
Plot area(ha)	15.63	1.66	Plot area(ha)	19.27	1.91
Farm area (ha)	184.41	2.32	Farm area (ha)	45.63	2.79
Irrigation (%)	56.60	49.80	Irrigation (%)	1.96	13.87
Longitude	29.23	0.06	Longitude	28.83	0.86
Latitude	−25.03	0.06	Latitude	−26.75	0.58
Dry bean			Soybean		
NDVI	0.62	0.14	NDVI	0.71	0.09
Water balance	−38.96	12.39	Water balance	−32.18	7.59
Season length (days)	145.58	38.66	Season length (days)	129.19	30.14
Plot area(ha)	21.43	2.03	Plot area(ha)	19.09	1.91
Farm area (ha)	68.50	3.10	Farm area (ha)	95.98	2.95
Irrigation (%)	3.75	19.00	Irrigation (%)	5.89	23.55
Longitude	27.83	2.15	Longitude	29.06	0.81
Latitude	−26.79	0.71	Latitude	−26.78	0.87
Ground nut			Sunflower		
NDVI	0.52	0.11	NDVI	0.59	0.13
Water balance	−55.56	6.34	Water balance	−44.20	12.10
Season length (days)	115.40	34.67	Season length (days)	131.27	38.01
Plot area(ha)	28.61	2.03	Plot area(ha)	20.54	2.01
Farm area (ha)	73.91	2.96	Farm area (ha)	80.59	3.14
Irrigation (%)	4.10	19.84	Irrigation (%)	7.80	26.81
Longitude	26.15	0.59	Longitude	27.16	1.14
Latitude	−27.67	0.59	Latitude	−27.19	1.11

Table IV.2. Distribution of the considered crops

Crop	Nb. of fields	Share of total fields (%)	Share of total area (%)
Dry beans	1 227	1.88	1.88
Ground nuts	1 292	2.5	2.50
Maize	45 256	69.77	72.27
Sorghum	715	1.10	0.93
Soybean	6 825	10.52	8.80
Sunflower	9 441	14.56	13.51
Cotton	106	0.16	0.10
Total	64 862	100.00	100.00

therefore yields, of various crops, including wheat (Das et al., 1993, Gupta et al., 1993, Doraiswamy and Cook, 1995, Hochheim and Barber, 1998, Labus et al., 2002), sorghum (Potdar, 1993), maize (Hayes and Decker, 1996, Prasad et al., 2006), rice (Nuarsa et al., 2011, Quarmby et al., 1993), soybean (Prasad et al., 2006), barley (Weissteiner and Kuhbauch, 2005), millet (Groten, 1993) and tomato (Koller and Upadhaya, 2005). Moreover, NDVI has also been shown to provide a very good indicator of crop phenological development (Benedetti and Rossini, 1993).

More precisely, the NDVI can be related to produced quantities by the following relationship, extrapolated by Ma et al. (2001) for soybean:

$$Y = d + bNDVI^c \tag{IV.7}$$

where Y represents the quantities produced (or the yield), d, b and c are three parameters, with c being in [2;35] and b in $[3.10^3; 72.10^3]$ (Ma et al., 2001). Denoting by

$$Z_i = NDVI_i - NDVI_0 \tag{IV.8}$$

with $NDVI_0 = |d/b|^{1/c}$, bZ_i^c gives an estimation of the quantities produced on field i, $Y(i)$.[5]

The NDVI index is calculated using ratios of vegetation spectral reflectance over incoming radiation in each spectral band. The NDVI data are extracted from the MOD13Q1 dataset,[6] which gathers together reflectance information collected by the MODerate-resolution Imaging Spectroradiometer (MODIS) instrument operating on NASA's Terra satellite (Huete et al., 2002). From these data, NDVI can be formulated as:

$$NDVI = \frac{NIR - VIS}{NIR + VIS}$$

where the difference between near-infrared reflectance (NIR) and visible reflectance (VIS) values is normalised by the total reflectance and varies between - 1 and 1 (Eidenshink, 1992).[7] Negative and very low values corresponding to water and barren areas were excluded from the analysis by design. Due to the nature of the source data NDVI derived estimates were available as 16-day composite indices at a resolution of 250 m. Area-weighted averages of the 16-day NDVI values for each crop field are calculated using the 'zonal statistics' tool in ArcGIS.

Crop growing seasons are characterised by the planting date and the phenology cycle, which determines the length of the season. In South Africa, planting generally occur somewhere between October and December in order to reduce the vulnerability to erratic precipitation (Ferreira et al., 2006). However, phenology cycles, and hence growing seasons, can differ substantially among crop

[5] For value smaller than $NDVI_0$, the produced quantities are equal to 0, the NDVI capturing the light reflected by the bare soil.

[6] Available online from https://lpdaac.usgs.gov/lpdaac/content/view/full/6652

[7] Plant leaves strongly absorb visible light (VIS), while they reflect near-infrared light (NIR). Then, the more leaves a plant has, and therefore the more biomass it produces, the more near-infrared and visible reflectance will be affected. The more biomass is produced, the more the NDVI is close to 1.

types and even for fields of the same crop type. In order to take account of this, we used the TIMESAT program[8] (Jönsson and Eklundh, 2002, 2004) to determine crop and field specific growing seasons. We are then able to approximate the start and end of growing seasons based on distribution properties of the NDVI. Summary statistics in Table 1 show that growing seasons are on average 130 days, with a standard deviation of 35 days.

Finally, as is standard in the literature of satellite derived plant growth measures, we use the maximum NDVI over the growing season as an indicator of crop production (Zhang et al., 2006). It takes on an average value of 0.61 with a standard deviation of 0.13 (see Table IV.1).

IV.2.3 Crop water balance

An important determinant of crop growth is water availability. A common simple proxy for it is the difference between rainfall and the evaporative demand of the air, i.e, evapotranspiration. To calculate this, we use gridded daily precipitation and reference evapotranspiration data taken from the USGS Early Warning Famine climatic database.[9] More specifically, daily rainfall data, given at the 0.1 degree resolution (ca. 10 km), are generated with the rainfall estimation algorithm RFE (version 2.0) dataset implemented by the National Oceanic and Atmospheric Administration (NOAA) - Climate Prediction Center (CPC) using a combination of rain gauges and satellite observations. Daily reference evapotranspiration data, available at a 1 degree resolution, were calculated using a 6-hourly assimilation of conventional and satellite observational data of air temperature, atmospheric pressure, wind speed, relative humidity and solar radiation extracted from the National Oceanic and Atmospheric Administration's Global Data Assimilation System. Using these gridded data each field was then assigned a daily precipitation and potential evapotranspiration value over its growing season to then calculate out its average daily water balance. The mean and standard deviation of this measure are given in Table IV.1.

IV.2.4 Biodiversity index

We approximate crop biodiversity at the field level using the following specification of the Shannon index (Shannon, 1948):

$$S_{i\ell} = -\sum_{z} B_l(z) \ln B_l(z) \tag{IV.9}$$

where the subscript i refers to a specific field, ℓ defines the size of the perimeter considered as relevant, and $B_\ell(z)$ is the proportion of area within perimeter ℓ that is of crop z type. S_{il} is then calculated

[8] The algorithm within the TIMESAT software is commonly used to extract seasonality information from satellite time-series data.

[9] http://earlywarning.usgs.gov/fews

Table IV.3. Summary statistics for the Shannon index

	All crops		Dry bean		Ground nut		Maize		Sorghum		Soybean		Sunflower	
ℓ	$\overline{S}$	σ_S	$\overline{S}$	σ_S	$\overline{S}$	σ_S	$\overline{S}$	σ_S	$\overline{S}$	σ_S	$\overline{S}$	σ_S	$\overline{S}$	σ_S
750 m	0.03	0.14	0.17	0.31	0.21	0.31	0.08	0.21	0.11	0.25	0.17	0.29	0.14	0.26
1000 m	0.06	0.18	0.27	0.37	0.35	0.35	0.14	0.26	0.20	0.31	0.28	0.33	0.23	0.31
1250 m	0.07	0.21	0.36	0.39	0.43	0.35	0.19	0.29	0.28	0.35	0.37	0.35	0.30	0.33
1500 m	0.09	0.23	0.45	0.41	0.48	0.34	0.23	0.30	0.35	0.36	0.44	0.36	0.35	0.34
1750 m	0.10	0.24	0.50	0.42	0.51	0.34	0.27	0.31	0.40	0.37	0.49	0.35	0.40	0.34
2000 m	0.12	0.25	0.55	0.43	0.53	0.32	0.31	0.32	0.45	0.37	0.54	0.35	0.43	0.33
2250 m	0.13	0.26	0.60	0.43	0.55	0.31	0.33	0.32	0.49	0.37	0.58	0.34	0.46	0.33
2500 m	0.13	0.27	0.63	0.43	0.56	0.30	0.36	0.32	0.52	0.37	0.61	0.33	0.48	0.32
2750 m	0.14	0.28	0.66	0.43	0.57	0.30	0.38	0.31	0.56	0.36	0.63	0.32	0.50	0.32
3000 m	0.15	0.28	0.68	0.43	0.58	0.29	0.40	0.31	0.59	0.36	0.65	0.32	0.52	0.31

Note: The table reports the mean ($\overline{S}$) and the standard deviation (σ_S) of the distribution of the Shannon index, measured for the different crops we considered, on different perimeters, characterised by their radius, ℓ.

for a given perimeter ℓ, defined by its radius, applied to the centroid of the field i considered.[10] The distance threshold for the radius is 0.75 kilometre; the distance is then increased 250 metres by 250 metres to reach 3 kilometres, the maximum distance considered. We provide summary statistics for the Shannon index in Table IV.3. Widening the perimeter under consideration increases the value of the Shannon index substantially. For example, the 3 km index is nearly 5 times larger than that of 0.75 km. This strongly suggests that crop types are strongly spatially agglomerated, and thus locally less diverse.

IV.3 Empirical analysis

IV.3.1 Impact on the production level

IV.3.1.1 Regression specification

Our first empirical task is to investigate whether biodiversity affects crop field health. To this end we use the following regression specification:

$$\ln(NDVI_i) = \theta_{0\ell} + \theta_{S\ell} S_{i\ell} + \sum_k \theta_{k\ell} X_{ik\ell} + \mu_m + \epsilon_{i\ell} \tag{IV.10}$$

where $NDVI_i$ is the maximum NDVI over the crop's growing season on field i, $S_{i\ell}$ is the Shannon index for perimeter ℓ as defined above, $X_{i\ell}$ is a vector of k control variables, and μ_m is a farm fixed effect. These effects depend on natural conditions (weather, season length. . .), field attributes (irrigation, area. . .) and farm management attributes (pesticides, mechanisation, economies of scale. . .). Therefore, the vector of control variables X includes crop fixed effects, crop water

[10] The Shannon index is widely used in ecology literature to quantify specie diversity or entropy of an ecosystem. In our case, the more diverse crops are and the more equal their abundances, the larger is the Shannon index. When all crops are equally common, all $B(z)$ values will equal $1/Z$ (Z being the total number of crops) and S will be equal to lnZ. On the contrary, the more unequal the abundances of the crops are, the smaller is the index, approaching 0 (and being equal to 0 if $Z = 1$).

balance (WB) and its squared value (WB^2), an irrigation dummy indicator, the season length, the logarithm of the field area in hectares ($\ln(AREA)$), the latitude ($LATITUDE$) and longitude ($LONGITUDE$) of the centroid of the field, the percentage of cropland within a defined perimeter that is irrigated (PC_AREA_IRR), and the percentage of land devoted to the same crop that belongs to the same farm, within a defined perimeter (PC_AREA_FARM). One should note that by including farm fixed effects, we are able to capture crop management techniques that are common within farms, such as, for example, pesticide use, as well as farm wide economies of scale. Crop specific dummies allow us to control for the fact that different crops will have different vegetation growth intensity as captured by satellite reflectance data.

One concern in estimating (IV.10) with standard linear estimators is that our dependent variable is theoretically bounded between 0 and 1. However, in our data the maximum NDVI of fields never reached either of the boundaries – the minimum and maximum values were 0.07 and 0.93, respectively. We can thus proceed with estimating (IV.10) with a standard linear model. In order to make our coefficients more readable, we multiply NDVI by 10 000. Standard errors are clustered at the farm level.

IV.3.1.2 Results

Our estimation results are presented in Table IV.4. In the first column, we simply include our field specific control variables (vector X). As can be seen, crop water balance has a significant positive and exponentially increasing impact on crop growth. However, while having an irrigation system also acts to increase the productivity of fields, it also makes crops less reliant on water balance (in a linear fashion) as would be expected. The coefficient on season length suggests that the longer the season lasts, the lower crop health is. In other words, the longer the season is, the higher are the probabilities that an adverse event affects crops. Larger fields are less productive per hectare than smaller ones. Finally, being located more in the east results in greater crop health, while being further south or north is inconsequential for field productivity within our sample.

In the second column, we add the Shannon index using a perimeter of 0.75 kilometre to define the neighbourhood ℓ. A greater value of S (i.e., a greater surrounding biodiversity) increases field productivity. Arguably, however, our diversity index may just be capturing the fact that neighbouring areas are different in ways that are correlated with the diversity of crops. To take account of these factors, we thus next include the percentage of the surrounding area that is irrigated and the percentage of the surrounding area of fields of the same crop type that belongs to the same farm. Accordingly, controlling for these factors does indeed reduce the coefficient of S by about 17 per cent, and thus suggests that it is important to control for surrounding area specific characteristics.

We subsequently increased the defined perimeter to calculate the Shannon index to 1 km, as shown in the forth column, adjusting the variables PC_AREA_IRR and PC_AREA_FARM in an analogous fashion. The S coefficient remain statistically significant, but decreases by 30%. As far as control variables are concerned, we find that the share of area irrigated unequivocally increases the biomass production. The share of area belonging to the same farm within the perimeter we consider seems to have no significant impact on the biomass production. Further increasing the perimeter similarly continues to produce a significant positive impact of biodiversity, the coefficient increasing by 48% per cent. However, when further expanding the threshold of our definition of the relevant neighbourhood to more than 1.5 kilometres, biodiversity no longer acts as a significant predictor of crop production, as shown in the last columns of the table.[11] This suggests that biodiversity is relatively locally defined, i.e., within less than 1.75 kilometres, but likely close to 1.25 kilometres. One may also want to note that among our neighbourhood specific controls, only the percentage of area irrigated consistently has a significant (positive) effect on crop field health.

We then apply the same regression specification by crop, i.e. only to the fields dedicated to one of the six crops for which data are available (cotton is not considered in the regressions by crop since the available sample – 106 fields, 0.16% of the total available fields and 0.1 of the total cropland considered – is too small). The results show that, on the one hand, biodiversity has a significant impact on the biomass production of maize, soybean and sunflower, and that the relevant perimeter size of the biodiversity index depends on the crop. Table IV.5 also reveals that biodiversity has no significant impact on dry bean, ground nuts and sorghum. This can be explained by the fact that each of the latter crops represents less than 2% of the total number of fields. In other words, the area dedicated to these crops is small, the fields are probably sufficiently scattered to not suffer from the proliferation of their pests. Therefore, the biodiversity variation on the perimeter that we consider has a negligible marginal effect on the biomass production. When looking at the crops which biomass production is affected by crop biodiversity, we see that the relevant perimeter for biodiversity varies: biodiversity has a positive and significant impact on the production of maize only for perimeters smaller than 1.5 kilometres whereas the relevant perimeter for soybean is between 1.25 and 2.75 kilometres. Surprisingly, biodiversity has a negative and significant impact on sunflower biomass production on perimeters with a radius larger than 2.25 kilometres. A positive significant impact on sunflower is found only for ℓ equal to one kilometre. This result is probably linked to the fact that the pests that affect the biomass production are not the same for the three crops we consider.[12]

The impact of irrigation also varies and depends on crop characteristics, as shown by tables IV.6, IV.7 and IV.8. Maize is one of the most efficient cultivated plants in South Africa as far as water use is concerned (DAFF, 2014a), hence a positive and significant impact of irrigation. On the contrary, sunflower is highly inefficient in water-use and, as well as soybean, is mostly rain-fed grown.[13] This

[11] We also experimented with increasing the perimeter up to 10 kilometres, but the coefficient on S remains insignificant in all cases.

[12] The main potential crop losses are caused by weeds, Nevertheless, thanks to the improvement in weed control techniques, the main actual losses come from animals (mainly insects) and pathogens (Oerke, 2007). Maize is mainly attacked by insects (DAFF, 2014a), while sunflower and soybean are mainly attacked by diseases caused by fungi and viruses (DAFF, 2009, 2014b).

[13] Soybean is mostly rain-fed grown because of low profitability and difficult water management. Indeed, water

Table IV.4. Regression results, all crops

Variables	(1)	(2) $\ell = 0.75$km	(3) $\ell = 1$km	(4) $\ell = 1.25$km	(5) $\ell = 1.5$km	(6) $\ell = 1.75$km	(7) $\ell = 2$km	(8) $\ell = 2.25$ km	(9) $\ell = 2.5$ km	(10) $\ell = 2.75$km	(11) $\ell = 3$km
Shannon index (S_l)		62.53***	44.02**	64.95***	28.87**	25.33	20.83	−1.78	1.87	−9.95	−21.97
		(22.81)	(19.94)	(20.79)	(14.05)	(16.03)	(15.01)	(17.60)	(15.96)	(18.47)	(17.17)
Water balance (WB)	11.66***	11.20***	10.98***	10.94***	10.09***	11.09***	11.27***	11.46***	11.44***	11.47***	11.58***
	(2.52)	(2.47)	(2.50)	(2.47)	(2.46)	(2.48)	(2.50)	(2.49)	(2.51)	(2.49)	(2.50)
WB^2	0.17***	0.17***	0.17***	0.17***	0.16***	0.17***	0.17***	0.17***	0.17***	0.17***	0.17***
	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)
Irrigation (IR)	539.13***	507.62***	495.86***	486.96***	495.91***	510.37***	521.80***	528.30***	530.05***	530.39***	533.25***
	(89.97)	(88.46)	(89.43)	(89.81)	(90.10)	(90.10)	(90.10)	(89.66)	(89.12)	(89.62)	(89.66)
WB × IR	−18.28***	−17.84***	−18.32***	−18.47***	−18.59***	−18.46***	−18.26***	−18.27***	−18.28***	−18.37***	−18.37***
	(5.27)	(5.27)	(5.27)	(5.18)	(5.11)	(5.15)	(5.20)	(5.15)	(5.17)	(5.21)	(5.24)
WB^2 × IR	−0.002	0.008	0.003	0.004	0.001	0.00	0.001	0.00	0.00	−0.00	−0.001
	(0.07)	(0.07)	(0.07)	(0.07)	(0.07)	(0.07)	(0.07)	(0.07)	(0.07)	(0.07)	(0.07)
Season length	−5.25***	−5.23***	−5.24***	−5.25***	−5.27***	−5.27***	−5.27***	−5.27***	−5.26***	−5.26***	−5.26***
	(0.35)	(0.35)	(0.35)	(0.35)	(0.35)	(0.35)	(0.35)	(0.35)	(0.35)	(0.35)	(0.35)
ln($AREA$)	−73.0***	−68.51***	−70.62***	−71.09***	−72.29***	−72.30***	−73.06***	−72.88***	−73.30***	−73.51***	−73.71***
	(8.07)	(8.18)	(8.18)	(8.11)	(7.99)	(8.04)	(7.99)	(8.09)	(8.04)	(8.06)	(8.14)
Longitude	1229.69***	1254.00***	1258.71***	1279.73***	1273.08***	1268.87***	1253.39***	1241.49***	1246.15***	1245.12***	1239.04***
	(268.90)	(269.06)	(268.37)	(267.75)	(269.40)	(269.82)	(269.73)	(269.87)	(268.38)	(268.04)	(269.11)
Latitude	−357.88	−344.10	−349.28	−353.55	−347.52	−345.18	−347.73	−350.00	−341.21	−334.49	−337.23
	(521.67)	(485.94)	(488.78)	(489.05)	(510.83)	(515.18)	(522.57)	(522.06)	(534.80)	(540.31)	(539.05)
PC_NEAR_IRR		225.33***	265.33***	329.28***	345.78***	322.21***	305.35***	256.87***	254.75***	249.47***	194.05***
		(32.16)	(33.36)	(27.98)	(34.66)	(40.74)	(38.31)	(44.09)	(41.74)	(49.48)	(48.81)
PC_NEAR_FARM		19.43	11.57	5.29	−9.99	−5.38	−13.33	−6.31	−24.06	−32.40	−36.14
		(11.92)	(11.82)	(14.53)	(17.0)	(16.3)	(17.72)	(21.90)	(25.09)	(27.75)	(31.95)
Number of fields	64 862	64 862	64 862	64 862	64 862	64 862	64 862	64 862	64 862	64 862	64 862
Number of farms	12 462	12 462	12 462	12 462	12 462	12 462	12 462	12 462	12 462	12 462	12 462

Note: ***, ** and * indicate 1, 5 and 10 per cent significance levels, respectively. Robust standard errors are in parentheses, standard errors are clustered at the field level. Farm and crop fixed effects are included but not reported. Treatment (1) includes field specific control variables only, while treatments from (2) to (10) include field specific variables and the Shannon index, calculated for ℓ between 0.75 and 3 km.

could explain the absence of a significant impact of irrigation on biomass production for these crops. Finally, unlike maize and sunflower, soybean biomass production is positively affected by the size of the field. This effect could be related to the physiology of the plant or to the higher mechanisation that is allowed by larger fields and that can have a positive impact on the final yield.

Table IV.5. Impact of biodiversity on crop production

	Crops					
Distance	Dry bean	Ground nut	Maize	Sorghum	Soybean	Sunflower
750 m	−43.31	91.37	85.52**	−2.70	−0.70	36.63
	(94.80)	(160.8)	(33.44)	(142.3)	(31.89)	(61.43)
1000 m	4.94	−86.55	47.01**	202.44	16.03	80.66*
	(90.90)	(182.6)	(23.4)	(128.6)	(29.81)	(47.78)
1250 m	82.51	−37.26	54.20**	143.75	62.71**	43.97
	(74.94)	(203.2)	(23.18)	(145.2)	(31.19)	(39.21)
1500 m	12.70	−305.00	10.78	−26.17	79.49**	−24.11
	(89.57)	(214.0)	(18.30)	(133.4)	(32.88)	(55.66)
1750 m	105.38	−95.21	19.52	6.34	125.99***	−16.10
	(91.12)	(228.5)	(19.82)	(115.5)	(31.35)	(55.89)
2000 m	−30.91	118.9	15.58	42.48	134.98***	−47.67
	(87.49)	(228.4)	(17.17)	(113.8)	(38.30)	(60.05)
2250 m	13.50	−29.43	−1.15	55.41	94.60**	−96.54
	(96.94)	(271.2)	(20.42)	(124.6)	(37.36)	(52.49)
2500 m	70.90	−50.72	16.01	129.71	85.19**	−159.23***
	(117.1)	(345.4)	(16.21)	(184.8)	(32.95)	(54.44)
2750 m	89.75	−42.84	17.17	10.37	66.00*	−199.04**
	(119.2)	(361.1)	(19.06)	(161.8)	(36.32)	(78.24)
3000 m	−25.43	−35.19	11.07	69.33	61.93	−268.42***
	(115.2)	(366.1)	(22.09)	(201.8)	(39.05)	(77.38)

Note: Robust standard errors in parentheses, $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$. The table shows the coefficients for the variable S_l for each of the six crops considered, multiplied by 10 000 to facilitate the reading. Complete regression results are reported in tables IV.6 to IV.8.

These results confirm the positive impact of crop biodiversity on agricultural production, with the exception of sunflower. But what about farmer's exposure to risks? In other words, does biodiversity also participate in the reduction of production variability, as ecology studies suggest? Previous papers (Di Falco and Chavas, 2006, 2009) find that crop diversity reduces the risk of crop failure. In particular, in Di Falco and Chavas (2009), genetic diversity increases the variance of barley production in Ethiopia as well as its skewness. The skewness effect dominates the variance effect and finally results in the reduction of the exposure of downside risk. Like Di Falco and Chavas (2009), we examine the contribution of diversity to production variance using a second regression,

shortage is critical during the pod set stage while excessive water supply prior or after the flowering may jeopardise the final yield.

Table IV.6. Regression results, maize

Variables	$\ell = 0.75$ km	$\ell = 1$ km	$\ell = 1.25$ km	$\ell = 1.5$ km	$\ell = 1.75$ km	$\ell = 2$ km	$\ell = 2.25$ km	$\ell = 2.5$ km	$\ell = 2.75$ km	$\ell = 3$ km
Shannon index (S_ℓ)	85.52**	47.01**	54.19**	10.78	19.52	15.58	−1.15	16.01	17.17	11.07
	(33.44)	(23.40)	(23.18)	(18.30)	(19.82)	(17.17)	(20.43)	(16.21)	(19.06)	(22.09)
Water balance (WB)	5.34	5.00	4.94	5.00	5.42	5.55	5.64	5.56	5.64	5.76
	(3.56)	(3.68)	(3.70)	(3.68)	(3.68)	(3.66)	(3.65)	(3.68)	(3.67)	(3.65)
WB^2	0.12**	0.11**	0.11**	0.11**	0.12**	0.12**	0.12**	0.12**	0.12**	0.12**
	(0.05)	(0.05)	(0.05)	(0.05)	(0.05)	(0.05)	(0.05)	(0.05)	(0.05)	(0.05)
Irrigation (IR)	731.07***	731.27***	723.09***	727.73***	743.11***	752.58***	756.74***	758.90***	759.19***	759.81***
	(114.79)	(116.41)	(117.38)	(116.10)	(114.64)	(113.94)	(113.97)	(115.11)	(114.89)	(114.64)
WB × IR	−9.01	−9.04	−9.59	−9.86	−9.88	−9.66	−9.61	−9.51	−9.63	−9.65
	(6.53)	(6.54)	(6.59)	(6.55)	(6.51)	(6.50)	(6.49)	(6.55)	(6.53)	(6.53)
WB^2 × IR	0.06	0.06	0.06	0.05	0.05	0.05	0.05	0.05	0.05	0.05
	(0.09)	(0.08)	(0.09)	(0.08)	(0.09)	(0.09)	(0.09)	(0.09)	(0.09)	(0.09)
Season length	−5.70***	−5.70***	−5.69***	−5.72***	−5.72***	−5.72***	−5.72***	−5.71***	−5.71***	−5.70***
	(0.40)	(0.39)	(0.40)	(0.40)	(0.40)	(0.40)	(0.40)	(0.40)	(0.40)	(0.40)
ln($AREA$)	−63.89***	−66.32***	−66.99***	−67.88***	−68.23***	−69.17***	−68.93***	−69.53***	−69.95***	−70.36***
	(8.88)	(8.95)	(8.91)	(8.86)	(8.81)	(8.73)	(8.77)	(8.82)	(8.89)	(8.98)
Longitude	962.56***	972.54***	981.79***	980.43***	968.68***	955.94***	948.98***	956.40***	953.72***	951.89***
	(292.06)	(292.32)	(294.53)	(296.82)	(297.18)	(297.05)	(296.18)	(295.65)	(294.77)	(294.44)
Latitude	175.12	170.91	175.41	178.10	179.65	178.65	178.88	186.16	178.53	181.66
	(540.15)	(541.23)	(543.88)	(559.37)	(567.85)	(580.03)	(581.60)	(590.99)	(595.01)	(599.89)
PC_NEAR_IRR	283.56***	346.00***	405.00***	417.23***	309.50***	285.39***	294.35***	287.82***	226.33***	192.80***
	(37.99)	(44.14)	(48.21)	(58.94)	(48.03)	(55.80)	(49.70)	(54.84)	(61.47)	(70.50)
PC_NEAR_FARM	20.26*	10.77	4.25	−6.86	−5.45	−21.88	−19.42	−39.83	−50.13**	−66.95**
	(11.09)	(12.85)	(16.10)	(15.96)	(15.50)	(17.95)	(21.95)	(25.67)	(24.99)	(27.44)
Number of farms	10 551	10 551	10 551	10 551	10 551	10 551	10 551	10 551	10 551	10 551
Number of fields	45 256	45 256	45 256	45 256	45 256	45 256	45 256	45 256	45 256	45 256

Note: ***, ** and * indicate 1, 5 and 10 per cent significance levels, respectively. Robust standard errors are in parentheses, standard errors are clustered at the field level. Farm and crop fixed effects are included but not reported. Treatments include field specific variables and the Shannon index, calculated for ℓ between 0.75 and 3 km.

Table IV.7. Regression results, soybean

Variables	$\ell = 0.75$ km	$\ell = 1$ km	$\ell = 1.25$ km	$\ell = 1.5$ km	$\ell = 1.75$ km	$\ell = 2$ km	$\ell = 2.25$ km	$\ell = 2.5$ km	$\ell = 2.75$ km	$\ell = 3$ km
Shannon index (S_l)	−0.70	16.03	62.71**	79.49**	125.99**	134.98***	94.60**	85.19**	66.00*	61.93
	(31.89)	(29.81)	(31.19)	(32.88)	(31.35)	(38.30)	(37.36)	(32.95)	(36.32)	(39.05)
Water balance (WB)	10.00	9.92	9.99	10.46*	10.45*	10.48*	10.42*	10.32	10.25	10.14
	(6.24)	(6.15)	(6.21)	(6.27)	(6.25)	(6.28)	(6.27)	(6.25)	(6.18)	(6.23)
WB^2	.29	0.29**	0.29**	0.30**	0.30***	0.30**	0.30**	0.30**	0.30**	0.30**
	(0.13)	(0.13)	(0.13)	(0.13)	(0.13)	(0.13)	(0.13)	(0.13)	(0.13)	(0.13)
Irrigation (IR)	151.49	150.57	136.72	158.20	141.36	152.55	150.47	148.87	145.32	150.21
	(140.09)	(137.92)	(139.14)	(140.83)	(142.59)	(142.08)	(142.21)	(142.27)	(141.43)	(141.63)
WB × IR	−12.47**	−12.57	−12.87	−12.28	−13.01	−12.45	−12.85	−13.25	−13.43	−13.12
	(8.92)	(9.01)	(8.84)	(8.91)	(8.93)	(8.85)	(8.87)	(8.99)	(8.96)	(8.88)
WB^2 × IR	−0.11	−0.11	−0.12	−0.11	−0.12	−0.11	−0.12	−0.13	−0.13	−0.13
	(0.20)	(0.20)	(0.19)	(0.19)	(0.20)	(0.20)	(0.19)	(0.20)	(0.20)	(0.20)
Season length	−4.65***	−4.68***	−4.68***	−4.68***	−4.69***	−4.68***	−4.69***	−4.71***	−4.70***	−4.72***
	(0.45)	(0.46)	(0.47)	(0.47)	(0.47)	(0.46)	(0.46)	(0.46)	(0.46)	(0.46)
ln($AREA$)	45.15***	42.68***	42.28***	40.76***	41.91***	40.87***	40.24**	40.32**	39.96**	39.69**
	(16.18)	(15.99)	(15.94)	(15.71)	(15.55)	(15.59)	(15.61)	(15.79)	(15.84)	(16.01)
Longitude	2138.97***	2156.27***	2171.44***	2161.12***	2151.44***	2195.33***	2173.92***	2168.77***	2156.57***	2162.75***
	(681.75)	(677.05)	(677.36)	(677.17)	(673.02)	(672.67)	(685.60)	(682.83)	(679.82)	(681.58)
Latitude	−1304.35	−1361.76*	−1389.11*	−1392.56*	−1420.42*	−1411.74*	−1392.86*	−1364.11*	−1319.19	−1318.97
	(797.04)	(799.14)	(790.734)	(787.69)	(777.57)	(790.14)	(810.05)	(817.98)	(841.64)	(834.58)
PC_NEAR_IRR	46.76	36.31	82.94	−9.08	140.24*	125.91	69.44	79.74**	229.69**	175.23
	(74.00)	(65.64)	(67.14)	(79.89)	(82.74)	(86.57)	(92.96)	(89.39)	(102.08)	(116.12)
PC_NEAR_FARM	66.73***	66.17***	48.85**	42.61	53.38	29.22	18.62	27.83	−2.71	−8.67
	(22.42)	(18.25)	(24.46)	(32.70)	(34.27)	(40.97)	(47.60)	(45.45)	(46.83)	(52.83)
Number of farms	2 248	2 248	2 248	2 248	2 248	2 248	2 248	2 248	2 248	2 248
Number of fields	6 825	6 825	6 825	6 825	6 825	6 825	6 825	6 825	6 825	6 825

Note: ***, ** and * indicate 1, 5 and 10 per cent significance levels, respectively. Robust standard errors are in parentheses, standard errors are clustered at the field level. Farm and crop fixed effects are included but not reported. Treatments include field specific variables and the Shannon index, calculated for ℓ between 0.75 and 3 km.

Table IV.8. Regression results, sunflower

Variables	$\ell = 0.75$ km	$\ell = 1$ km	$\ell = 1.25$ km	$\ell = 1.5$ km	$\ell = 1.75$ km	$\ell = 2$ km	$\ell = 2.25$ km	$\ell = 2.5$ km	$\ell = 2.75$ km	$\ell = 3$ km
Shannon index (S_l)	36.63	80.66*	43.97*	−24.11	−16.10	−47.67	−96.54*	−159.23***	−199.04**	−268.42***
	(61.43)	(47.77)	(39.21)	(55.66)	(55.89)	(60.05)	(52.49)	(54.45)	(78.24)	(77.38)
Water balance (WB)	17.49***	17.18***	17.06***	17.10***	16.97***	16.94***	17.14**	17.28***	17.38**	17.38***
	(5.15)	(5.12)	(5.14)	(5.16)	(5.18)	(5.19)	(5.20)	(5.17)	(5.15)	(5.15)
WB^2	0.18**	0.17**	0.17**	0.17**	0.17***	0.17**	0.17**	0.17**	0.18**	0.18**
	(0.08)	(0.08)	(0.08)	(0.08)	(0.08)	(0.08)	(0.08)	(0.08)	(0.08)	(0.08)
Irrigation (IR)	418.04	387.46	345.82	357.07	351.61	351.84	356.13	346.43	339.11	351.20
	(283.66)	(286.04)	(288.54)	(294.03)	(292.16)	(293.16)	(288.87)	(281.65)	(281.27)	(277.03)
WB × IR	−44.13***	−44.43***	−44.43***	−44.72***	−44.45***	−44.43***	−44.66***	−45.33***	−45.39***	−45.59***
	(15.11)	(15.48)	(15.51)	(15.66)	(15.66)	(15.86)	(15.83)	(15.58)	(15.51)	(15.32)
WB^2 × IR	−0.47**	−0.47**	−0.46**	−0.47**	−0.46**	−0.46**	−0.47**	−0.48**	−0.47**	−0.48**
	(0.20)	(0.21)	(0.21)	(0.21)	(0.21)	(0.21)	(0.21)	(0.21)	(0.21)	(0.21)
Season length	−2.27**	−2.29**	−2.34**	−2.35**	−2.36***	−2.37***	−2.37***	−2.36***	−2.35**	−2.36***
	(0.89)	(0.90)	(0.89)	(0.90)	(0.90)	(0.90)	(0.90)	(0.90)	(0.90)	(0.89)
ln(*AREA*)	−176.03***	−178.38***	−177.74***	−179.14***	−177.89***	−176.27***	−175.74***	−176.58***	−174.99***	−176.80***
	(22.77)	(23.59)	(22.81)	(22.91)	(22.90)	(22.60)	(22.70)	(22.58)	(22.74)	(22.65)
Longitude	2652.50***	2642.40***	2639.27***	2580.98***	2584.23***	2528.84***	2485.28***	2460.29***	2437.61***	2447.18***
	(711.86)	(710.87)	(707.95)	(718.08)	(704.95)	(717.52)	(717.78)	(712.56)	(691.20)	(684.86)
Latitude	−3750.76***	−3740.78***	−3745.43***	−3704.66***	−3737.15***	−3769.75***	−3699.76***	−3628.35***	−3600.40***	−3503.20***
	(1127.16)	(1129.86)	(1131.27)	(1145.88)	(1144.54)	(1129.58)	(1128.09)	(1125.61)	(1123.87)	(1132.23)
PC_NEAR_IRR	−263.41***	−78.04	117.72	77.98	155.50	213.33	157.13	180.39	314.26*	121.37
	(93.61)	(89.82)	(119.24)	(119.55)	(145.13)	(140.64)	(154.78)	(158.57)	(168.79)	(171.09)
PC_NEAR_FARM	50.74	2.79	−4.08	−13.08	4.65	54.37	76.09	78.60	103.05	126.00
	(36.94)	(32.15)	(49.64)	(61.62)	(64.84)	(76.64)	(82.19)	(90.14)	(94.93)	(99.30)
Number of farms	3 994	3 994	3 994	3 994	3 994	3 994	3 994	3 994	3 994	3 994
Number of fields	9 441	9 441	9 441	9 441	9 441	9 441	9 441	9 441	9 441	9 441

Note: ***, ** and * indicate 1, 5 and 10 per cent significance levels, respectively. Robust standard errors are in parentheses, standard errors are clustered at the field level. Farm and crop fixed effects are included but not reported. Treatments include field specific variables and the Shannon index, calculated for ℓ between 0.75 and 3 km.

on the error terms of the first regression, in accordance with the theoretical model that supports our investigations.

IV.3.2 Impact on the variability of production

IV.3.2.1 Regression specification

Following the empirical strategy developed in section IV.1.2, we now use the error terms of the regression presented in (IV.10) to estimate the contribution of crop biodiversity to production variance. Our theoretical model is built to illustrate the way crop biodiversity impacts agricultural production level and variance. It therefore assumes that the production variance is mainly driven by biodiversity: it shows how to capture the effects of explanatory variables other than S_i on the average production but not on its variance. Nevertheless, in the following regression, we also include the vector X to investigate the main drivers of agricultural production variance and their relative importance. Then, the regression specification is given by

$$\epsilon_i^2 = \delta_{0\ell} + \sum_k \delta_{k\ell} X_{ik\ell} + \delta_{S\ell} S_{i\ell} + \epsilon'_{i\ell} \tag{IV.11}$$

where the vector $X_{i\ell}$ is the same vector of k control variables as the one described in section IV.3.1.1.

Table IV.9. Contribution to production variability – Regression results, all crops

Variables	$\ell = 0.75$km	$\ell = 1$km	$\ell = 1.25$km	$\ell = 1.5$km	$\ell = 1.75$km	$\ell = 2$km
Shannon index (S_ℓ)	-20.19^{**}	-12.04^{***}	-24.04^{***}	-13.31^{**}	-19.06^{***}	-17.72^{***}
	(7.80)	(4.18)	(4.27)	(5.88)	(5.53)	(5.43)
Water balance (WB)	6.05^{***}	6.07^{***}	6.12^{***}	6.02^{***}	5.99^{***}	5.93^{***}
	(1.79)	(1.80)	(1.78)	(1.78)	(1.77)	(1.78)
WB^2	0.06^{***}	0.06^{***}	0.06^{***}	0.06^{***}	0.06^{***}	0.07^{***}
	(0.02)	(0.02)	(0.02)	(0.02)	(0.02)	(0.02)
Irrigation (IR)	-41.78	-41.13	-36.87	-43.84	-44.71	-48.74
	(35.63)	(35.93)	(36.75)	(37.47)	(37.15)	(36.62)
WB × IR	-8.30^{***}	-8.15^{***}	-8.19^{***}	-8.26^{***}	-8.21^{***}	-8.05^{***}
	(1.72)	(1.74)	(1.77)	(1.77)	(1.76)	(1.72)
WB^2 × IR	-0.15^{***}	-0.15^{***}	-0.15^{***}	0.15^{***}	0.15^{***}	-0.14^{***}
	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)
Season length	-0.16	-0.16	-0.17	-0.17	-0.17	-0.17
	(0.17)	(0.17)	(0.17)	(0.17)	(0.17)	(0.17)
ln($AREA$)	21.02^{***}	20.81^{***}	19.92^{***}	20.27^{***}	20.26^{***}	20.79^{***}
	(3.71)	(3.74)	(3.72)	(3.70)	(3.69)	(3.68)
Longitude	95.89	91.88	84.08	86.08	87.28	94.57
	(80.03)	(81.26)	(82.57)	(83.31)	(83.13)	(81.93)
Latitude	27.88	28.54	34.80	39.56	32.19	29.75
	(129.13)	(129.19)	(135.95)	(133.71)	(137.06)	(137.47)
PC_NEAR_IRR	-62.70^{***}	-55.32^{***}	-61.44^{***}	-48.31^{***}	-46.57^{**}	-32.64
	(13.62)	(17.14)	(17.13)	(17.95)	(22.05)	(21.40)
PC_NEAR_FARM	11.81^{***}	9.20^{***}	5.42	4.98	3.90	3.72
	(3.26)	(3.06)	(3.42)	(11.09)	(4.32)	(3.78)

Note: ***, ** and * indicate 1, 5 and 10 per cent significance levels, respectively. Robust standard errors are in parentheses, standard errors are clustered at the field level. Farm and crop fixed effects are included but not reported. Treatments include field specific variables and the Shannon index, calculated for ℓ between 0.75 and 3 km. Treatments in which the Shannon index has not a significant contribution are not reported.

IV.3.2.2 Results

Regression results show that crop biodiversity has a significant and negative impact on agricultural production variance both when we consider all crops together (table IV.9) and maize (table IV.10) alone. This impact is verified only for perimeters having a radius equal or smaller to 2 kilometres.

More generally, variance of agricultural production (both for maize and for all crops) appears to be driven by two main effects: cultivation intensity and water management. A low cultivation intensity has a negative impact on variance (leading to its decrease), as it is shown by the contribution of the Shannon index and of the area of the field, measured by $\ln(AREA)$. An increase in the surrounding biodiversity, i.e. a decrease in cultivation intensity, measured by the Shannon index, leads to a decrease in production variance, all else being equal. In parallel, when the area of the considered field increases, variance increases. Indeed, a larger field forms a bigger island for parasitic species. Their attacks being more frequent, the variance of the production increases. Water management also has a significant impact. Water balance has a positive impact on variance, as expected. Indeed, water balance is measured by evapotranspiration. The larger is evapotranspiration, i.e. the higher is the water stress of plants, the greater is the production impediment and the higher is the variance in the production. Irrigation makes crops less reliant on water balance, in a non linear fashion this time, but does not have a significant impact on variance by itself. Furthermore, irrigated surrounding fields have a negative significant impact on variance. They probably allow some indirect increase in water availability. This effect confirms the role that water management can have in the reduction of production variance.

IV.4 Conclusion

Using a new large database built from satellite imagery, we confirm that crop biodiversity has a positive impact on agricultural production, with the exception of sunflower. Maintaining a large diversity of crops in the landscape increases agricultural production level while decreasing its variance. These impacts, that were previously described at regional scales, are robust when we consider a whole country. Furthermore, we show the consistency of these results with the underlying ecologic and agricultural mechanisms. For this purpose, we build a probabilistic model in which stochastic factors linked to biodiversity, namely pests, are endogenous, as it is shown in the ecology literature.

Nevertheless, the impacts of biodiversity on the skewness of production, i.e. on the probability of extreme events, is not tested at this stage, while Di Falco and Chavas (2009) found that they are significant. Our reduced form approach could be developed with a third step to test for these effects. Alternatively, a structural approach could be built on the theoretical model we use to describe the relationships between crop production and biodiversity. The parameters of the beta-binomial distribution could be estimated to fully describe the impacts of biodiversity on the production distribution.

However, from the results we already have, crop diversification can be seen as a possible strategy to increase agricultural productivity or to maintain its level while decreasing the use of pesticides.

Table IV.10. Contribution to production variability – Regression results, maize

Variables	$\ell = 0.75$km	$\ell = 1$km	$\ell = 1.25$km	$\ell = 1.75$km	$\ell = 2$km
Shannon index (S_ℓ)	-30.75^{***}	-16.10^{***}	-26.96^{***}	-18.54^{**}	-14.73^{**}
	(11.25)	(7.71)	(4.94)	(7.25)	(7.23)
Water balance (WB)	8.67^{***}	8.62^{***}	8.58^{***}	8.63^{***}	0.08^{***}
	(3.32)	(3.30)	(3.28)	(3.33)	(3.34)
WB^2	0.08^{***}	0.08^{***}	0.08^{***}	0.08^{***}	0.08^{***}
	(0.03)	(0.03)	(0.03)	(0.03)	(0.03)
Irrigation (IR)	-113.87	-116.72	-110.52	-117.98	-119.53
	(76.28)	(75.18)	(75.37)	(77.50)	(77.29)
WB $\times$ IR	-14.01^{***}	-14.10^{***}	-13.77^{***}	-13.86^{***}	-13.64^{***}
	(3.23)	(3.19)	(3.20)	(3.27)	(3.27)
WB^2 $\times$ IR	-0.25^{***}	-0.25^{***}	-0.24^{***}	-0.24^{***}	-0.24^{***}
	(0.04)	(0.04)	(0.04)	(0.04)	(0.04)
Season length	-0.24	-0.25	-0.26	-0.25	-0.25
	(0.17)	(0.17)	(0.17)	(0.17)	(0.18)
ln($AREA$)	29.12^{***}	28.75^{***}	28.00^{***}	28.39^{***}	28.78^{***}
	(3.69)	(3.74)	(3.71)	(3.72)	(3.73)
Longitude	93.10	84.20	83.79	91.38	97.90
	(81.91)	(82.91)	(84.46)	(85.28)	(84.83)
Latitude	24.26	36.82	45.38	36.88	33.14
	(107.98)	(106.66)	(110.04)	(111.12)	(110.44)
PC_NEAR_IRR	-42.91^{***}	-32.33^{***}	-25.37^{**}	-1.59	18.44
	(15.35)	(11.81)	(11.82)	(15.78)	(18.78)
PC_NEAR_FARM	13.45	10.52^{***}	1.01	-2.12	-1.73
	(4.63)	(3.74)	(4.01)	(4.76)	(5.56)

Note: ***, ** and * indicate 1, 5 and 10 per cent significance levels, respectively. Robust standard errors are in parentheses, standard errors are clustered at the field level. Farm and crop fixed effects are included but not reported. Treatments include field specific variables and the Shannon index, calculated for ℓ between 0.75 and 3 km. Treatments in which the Shannon index has not a significant contribution are not reported.

CHAPTER V

CONCLUSION

V.1 General conclusion

This thesis has tried to introduce new elements to questions related to building or implementing agricultural environmental policies in a globalized world. Policies related to reducing the use of polluting intermediary consumptions, in particular pesticides and fertilisers, were analysed. We considered reducing use of pesticides and fertilizers by promoting organic agriculture. This specific policy is used to illustrate and quantify the global indirect impacts driven by market and price effects which a policy, known for its local benefits, could have if implemented unilaterally. We also noted the distinctive role played by crop biodiversity in the use of pesticides. We integrated this feature in a theoretical trade model to analyse the effects at work if we consider environmental policy (a tax on pesticides), biodiversity effects and free trade simultaneously.

Countries faced with long term environmental damage from intensive agriculture and the difficulties related to international coordination, may opt for pro-environment unilateral agricultural policies which can generate some leakages. We quantify the resulting indirect environmental impacts, driven by market mediated responses, of such an environmental policy implemented unilaterally, taking the example of a significant shift to organic agriculture in the European Union. Using a computable general equilibrium model, we add an economic component to standard environmental analysis and show that a significant shift to organic farming in the European Union would imply sufficiently large price changes to be an incentive to intensify agricultural production in other world regions, and to bring new land into cultivation. Overall, few of these effects are large enough to totally offset the local benefits of organic farming, however, they should be included in life cycle analysis and sustainability impact assessments. In particular, it is likely that indirect emissions caused by land use changes offset the emission savings made possible by organic agriculture. Results concerning fertilizer use are positive, while those concerning biodiversity are more difficult to assess. Of course, as discussed in Chapter II, these results will depend on the environmental regimes put in place by other countries to limit intensification and deforestation, and there is no reason to single out organic farming for its land use change effects; other policies could have much larger effects. However, our results raise several issues relevant for policy making. On the one hand, they show the need for integrated impact analysis of all policies, including those that favour the environment. On the other hand, they underline the key role played by the gap between organic and conventional yields as a driver of indirect impacts. Of course, after decades of under-investment in research and development for organic farming, it is reasonable to expect rapid improvements in organic yields. Nevertheless, since one of the basic features of organic farming is to replace intermediary consumptions by labour, and given that labour is relatively expensive in the European Union, even when technical progress reduces some of the obstacles, economic factors may not warrant increased yields unless public policies modify the relative prices of production factors.

The second part of the thesis dealt with crop biodiversity. Chapter III showed that in a trade context, biodiversity effects can be complex. Indeed, trade specialization induced by comparative advantage works to discourage crop biodiversity. In this case, environmental policy has a strategic aspect and governments face a trade-off between protecting human health and the environment and gaining market share in agricultural markets. We show that if crop biodiversity effects are taken into account, environmental policies are more lenient under strategic trade than under non strategic trade, but that they nevertheless remain more stringent than under autarky. Indeed, strategic effects are weaker than the *not in my backyard* effect which leads domestic consumers to benefit from low international prices (enabled by the use of pesticides in foreign countries) while demanding stricter environmental policy to protect their health and their environment. Since environmental taxes on pesticides are generally higher under trade than under autarky, production (and price) volatility is higher under trade than under autarky apart from some specific cases (small biodiversity effects, taxes under autarky and under trade are very close…). Since this volatility is linked to agricultural specificities, it can explain the "background noise" on agricultural markets, which are known to be more volatile than manufacturing markets. Of course, these effects are minor compared to the effects of supply (weather events, level of stocks…) and demand (e.g. new demand from energy markets) shocks which are the main causes of agricultural price volatility or spikes such as occurred in 2007-2008.

Chapter IV returned to the mechanisms linking crop diversity and agricultural production, using a new and very large dataset of data obtained from satellite imagery. It showed that crop biodiversity has a significant and positive impact on agricultural productivity. First, we built a probabilistic model to detail the relationship between biodiversity and productivity and the resulting effects on the level and variability of agricultural production. Then, we explored these effects quantitatively using a reduced form approach to analyse the role of biodiversity in farmers' exposure to production risks.

V.2 Perspectives

This thesis has developed two models of international agricultural trade incorporating environmental policies. The first is empirical - a global multi-sector computable general equilibrium model, which we used to quantify market driven environmental impacts. Results for land use responses are subject to significant uncertainty in which the values of some critical parameters play key roles (Plevin et al., 2010, Laborde and Valin, 2012, De Cara et al., 2012, Valin, 2014). Econometric estimations are available for only a few of these parameters; the other values are arbitrarily (but at least commonsensically) set and then tested in sensitivity analyses. Laborde and Valin (2012)'s analysis of uncertainty linked to key behavioural parameters in MIRAGE-BioF is completed in the annex to Chapter II for the new key parameters proposed to deal with organic farming. However, lack of data limits the modelling possibilities. Our model represents organic production technologies for only some crops, namely maize, rapeseed, sunflower and wheat, while, in the European Union, only 15% of the total organic area is devoted to cereals, and 1.5% to oilseeds. In addition to not being representative of the whole organic sector, the development of these sectors implies the mobilization of large quantities of natural fertilizers, in particular animal or green manure. To overcome this limiting choice, detailed data - presently missing - on animal production technologies are needed. Not taking account of organic feedstock production means ignoring potential constraints on fertilizing capacities and, therefore, higher assumed yields than expected to organic farming in areas such as the United States where the organic to conventional yield ratio is larger than in the European Union, could lower the indirect negative impacts. It is hoped that monitoring activities related to the recent trade agreement, which is due for review in 2015, will be an opportunity to obtain detailed data on at least a part of trade in organic products.

The second model developed in the thesis is a theoretical, Ricardian trade model with two countries and many goods, which is useful to detail and understand the mechanisms at stake, by nature, cannot quantify their impact. One of the questions that arise from a study of the interaction between crop biodiversity effects, trade and an environmental policy on pesticides, is exactly quantification of the effects of this interaction on production and price volatility. To investigate this question, we need a more complex framework, such as the one developed by Eaton and Kortum (2002). Extending the current model to incorporate several countries would allow us to estimate and test the scale of biodiversity effects. Note that applying Ricardian trade models to agriculture enables full testing of Ricardian theory since we know the productivities for crops in which countries do not specialize (and which are no longer produced) thanks to agronomic knowledge of potential yields depending on soil and environment qualities (Costinot and Donaldson, 2012).

APPENDIX A

APPENDIX TO CHAPTER II

A.1 Sector and region aggregations

Sector and region aggregations are presented in tables A.1 and A.2.

A.2 Model description

In this section, standard features of the MIRAGE-BioF are presented. Detailed description of these features are given by Al-Riffai et al. (2010), Bouët et al. (2010), Laborde (2011), Laborde and Valin (2012).

A.2.1 Yield response to fertilizers

Details on fertilizers modelling are mainly given by Al-Riffai et al. (2010). In sum, a simple constant elasticity of substitution form can not properly represent the impact of fertilizers on agricultural production. Indeed, an increase in the use of fertilizers allows an increase in agricultural production, but only within a maximum level. The easiest way to represent this saturation effect is through a logistic function. Nevertheless, this kind of function can not directly be introduced in a CGE framework, mainly because of convexity and calibration difficulties. Then, a modified logistic function is introduced:

$$\frac{TEFS_{j,r,t}}{TE_{j,r,t}} = \frac{\left[\exp(a\_TE_{j,r}\frac{FERT_{j,r,t}}{TE_{j,r,t}}) - 1\right] yield\_max_{j,r} + 2yield\_min_{j,r}}{\left[\exp(a\_TE_{j,r}\frac{FERT_{j,r,t}}{TE_{j,r,t}}) + 1\right]}$$

where

- $TEFS_{j,r,t}$ is a CES composite of land and fertilizers used to produce crop j, in region r, at time t;
- $TE_{j,r,t}$ is the land used to produce crop j, in region r, at time t;

Table A.1. Sectoral nomenclature

Sector	Description	Sector	Description
AirSeaTran	Private services	OilPalm	Palm oil
Biodiesel	Biodiesel	OilRape	Rapeseed oil
Cattle	Cattle meat milk fibers	OilSoyb	Soybean oil
Coal	Coal	OilSunf	Sun oil
Construction	Construction	OthAnim	Other animals (hog, chicken. . .)
DDGSBeet	Beet pulp	OthCrop	Other crops
DDGSCane	Cane Coproducts	OthFood	Other food and beverages
DDGSMaize	DDGS Maize	OthMin	Minerals and other extraction
DDGSWheat	DDSG Wheat	OthOilSds	Other oilseeds
ElecGas	Electricity and gas	PalmFruit	PalmFruit
Ethanol	Ethanol	PetrNoFuel	Petroleum products, excl. fuel
EthanolBeet	Ethanol process - sugar beet	PrivServ	Road transportation
EthanolCane	Ethanol process - sugar cane	PubServ	Public services
EthanolMaize	Ethanol process - maize	Rapeseed	Rapeseed
EthanolWheat	Ethanol process - wheat	Rapeseed_norg	Conventional rapeseed
Fertiliz	Fertilizer and agricultural chemicals	Rapeseed_org	Organic rapeseed
Fishing	Fishing	Rice	Rice
Forestry	Forestry	RoadTrans	Air and water transportation
Fuel	Refined fuel	Soybeans	Soybeans
Gas	Gas	Sugar	Sugar
Maize	Maize	Sugar_cb	Sugar cane and sugar beet
Maize_norg	Conventional maize	Sunflower	Sunflower
Maize_org	Organic maize	Sunflower_norg	Conventional sunflower
Manuf	Other manufactured products	Sunflower_org	Organic sunflower
MealPalm	Palm oil coproducts	VegFruits	Vegetable and fruits
MealRape	Rapeseed meal	Wheat	Wheat
MealSoyb	Soybean meal	Wheat_norg	Conventional wheat
MealSunf	Sunflower meal	Wheat_org	Organic wheat
MeatDairy	Meat and dairy products	WoodPaper	Wood and paper industry
Oil	Oil		

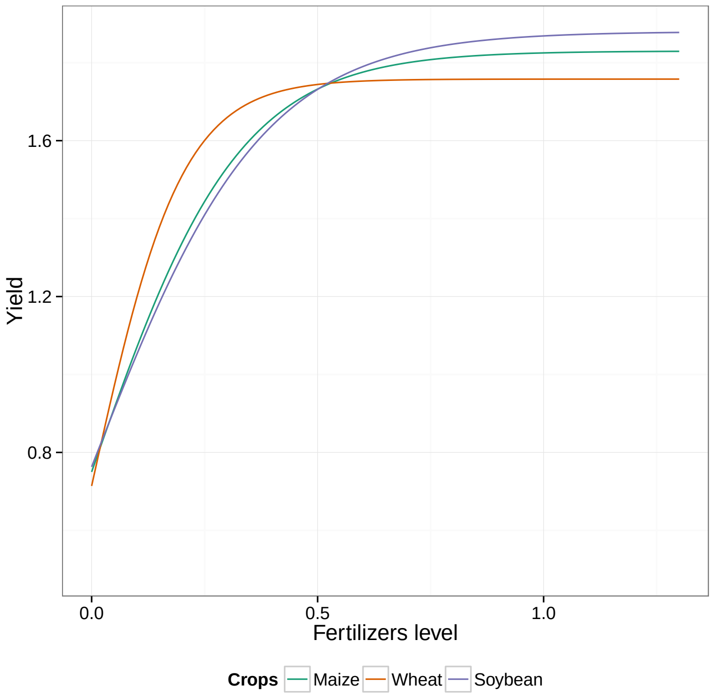
Table A.2. Regional nomenclature

Region	Description
Brazil	Brazil
EU27	European Union (27 member states)
IndoMalay	Indonesia and Malaysia
RoW	Rest of the World
USA	United States of America

- $FERT_{j,r,t}$ is the use of fertilizers for the production of crop j, in region r, at time t ;
- $yield\_max_{j,r}$ is the maximum yield attainable by crop j in region r;
- $yield\_min_{j,r}$ is the minimum yield for crop j in region r;
- $a\_TE_{j,r}$ is a calibration coefficient, crop and region specific.

$TEFS_{j,r,t}/TE_{j,r,t}$ determines the yield of crop j, while $FERT_{j,r,t}/TE_{j,r,t}$ represents the intensity in the use of fertilizers and $a\_TE_{j,r}$ determines the way the maximum yield is atteined. Figure A.1 shows some example of endogeneous yield function for Brazil.

Figure A.1. Endogeneous yield functions for selected crops in Brazil



A.2.2 Land subsitution and land expansion

Details of the representation of land substitution are given in the section II.2 and are available in Al-Riffai et al. (2010). Substitution is possible between the different uses of managed land (i.e. with an economic return, figure A.2 gives the list of the considered categories of managed land), following a constant elasticity of substitution specification:

$$\frac{L_1}{L_2} = a\left(\frac{P_1}{P_2}\right)^{\sigma}$$

where L_1 and L_2 are per hectare productivity of two different land uses and P_1 and P_2 are the prices associated to these uses. a is a calibration coefficient (share parameter) and σ is the transformation elasticity. CET functions are nested in several levels in order to have a flexible enough production function, as detailed in figure A.2. They are implemented in each agro-ecological zone of each region in order to describe substitution patterns of crops that thrive in the same agro-climatic conditions. For details on the calibration of transformation elasticities, refer to Laborde and Valin (2012). Transformation elasticity between the same conventional and organic crop is arbitrarily set at a high value, since econometric estimations are not available. Sensitivity analysis are carried on this value.

When land prices (and therefore crop prices) are sufficiently high and substitution possibilities sufficiently small, some land expansion will occur. This expansion is modelled through the following equation:

$$LANDEXT_{z,r,t} + MANAGED\_LAND_{z,r,t=0} = MANAGED\_LAND^{Exo}_{z,r,t} \left[\left(\frac{P^{\text{Managed land}}_{z,r,t}}{P_{r,t}} \right)^{\sigma} \right]$$

$$\text{with } \sigma = \sigma^L_{z,r} \left(\frac{Land\_avail_{z,r} - LANDEXT_{z,r,t}}{Land\_avail_{z,r}} \right)$$

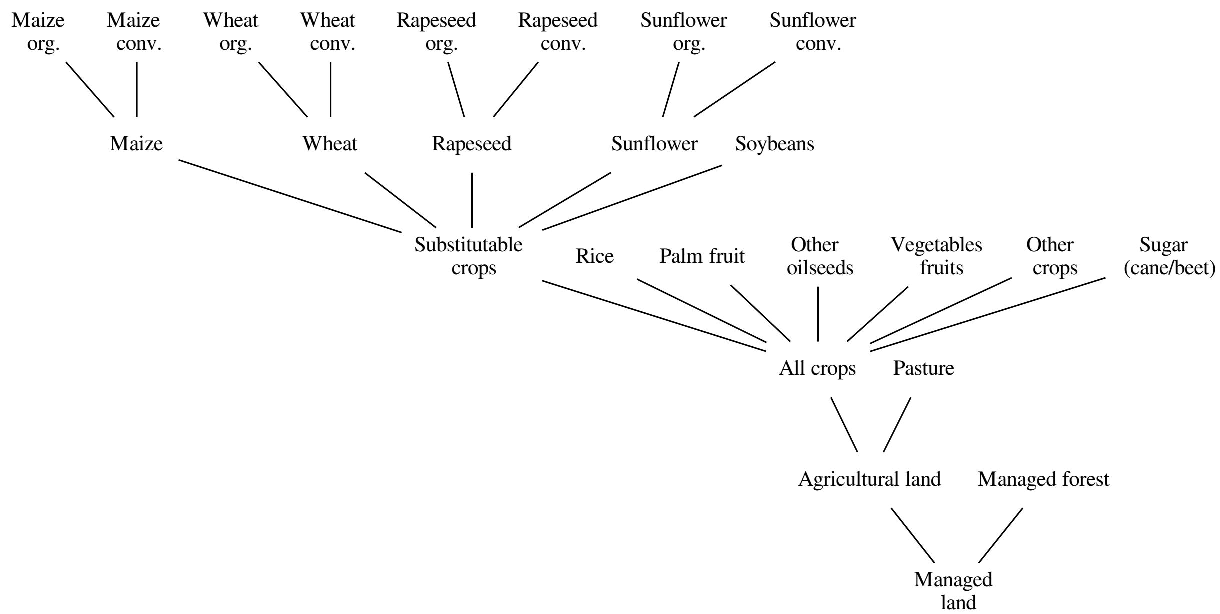
where

- $LANDEXT_{z,r,t}$ is the expansion of managed land (more precisely of cropland) into unmanaged land, in agro-ecological zone z of region r;
- $MANAGED\_LAND_{z,r,t=0}$ is the area of managed land in agro-ecological zone z of region r at the beginning of the simulation;
- $MANAGED\_LAND^{Exo}_{z,r,t}$ is an exogeneous trend of the area of managed land, based on historical data;
- $P^{\text{Managed land}}_{z,r,t}$ is the average price of managed land in agro-ecological zone z of region r;
- $P_{r,t}$ is a reference price, namely the shadow price of utility, in region r;
- $\sigma^L_{z,r}$ is the initial elasticity of managed land supply;
- $Land\_avail_{z,r}$ is the total area of land available for managed land expansion (and not already in use).

At the beginning of the simulation, $MANAGED\_LAND_{z,r,t=0} = MANAGED\_LAND^{Exo}_{z,r,t}$ and the land expansion is zero. Then, when cropland price increases, land expansion occurs, according to the variation of the elasticity of expansion. At the initial point, since there is no land expansion, the value of this elasticity is $\sigma^L_{z,r}$. Then, the bigger is the land expansion, the smaller is the elasticity of expansion. In other words, the more land is already in use, the harder is the expansion.

$MANAGED\_LAND^{Exo}_{z,r,t}$ allows to take into account an exogenous historical trend in land expansion and therefore to take into account non agricultural drive land use changes such as those linked to urbanization. The value of the initial elasticity of land supply vary across agro-ecological zones and regions, the elasticity being higher in southern countries than in northern countries and its values being between 2.10^{-4} and 10^{-2}. Because of the lack of econometric foundations, sensitivity analysis are critical to account for the uncertainties on these values.

Figure A.2. Nesting of CET functions for the representation of land substitution in MIRAGE-BioF



A.2.3 Computation of greenhouse gas emission

To calculate emissions from land use change, we rely on the guidelines for National Greenhouse Gas Inventories of the International Panel on Climate Change (IPCC, 2006), according to Bouët et al. (2010). We use the Tier 1 methodology, which provides generic estimates of the carbon stocks in different climate zones (these climate zones are matched with the agro-ecological zones used in the Mirage-BioF model following Bouët et al. (2010)). We consider the emissions due to (a) the conversion of forest to other types of land (deforestation), (b) the cultivation of the land that was previously uncultivated. To determine emissions from deforestation, we take into account the stock of carbon both above and below ground for managed and primary forests. We compute emissions induced by the cultivation of new land through the variation of the content of soil in mineral carbon. The IPCC Tier 1 method gives indicative release of carbon for different management practices. In order to simplify the computation, the different practices we consider are non cultivation of land, cultivation with full tillage, rice cultivation under irrigation and land set aside. We consider a medium level of input in each case. Finally, we compare the carbon stocks in forest biomass and in soils (forest + cultivated) in 2020 in our scenarios to the carbon stocks in 2020 in the baseline in order to estimate the carbon emissions due to land use change in our scenarios.

At this stage, we consider only emissions of carbon, although nitrous dioxide releases are recognized to play a significant role. We do not take into account (a) the emissions of N_2O due to the increase of fertilizers on the land where there is an intensification in the production and (b) the decrease in the N_2O emissions on the land cultivated under organic farming [1].

A.3 Computation of the MSA value of our scenarios

Alkemade et al. (2009) report relative share of intensive and low-input agriculture for different world regions. We use their share of low-intensity agriculture for the regions other than Europe. For Europe, Alkemade et al. (2009) consider that all area is intensively farmed while we use the share of organic farming we found in our simulations. They attribute the MSA value to a land use using a map of potential vegetation. Because they assume that the potential vegetation in Europe is forest, they consider all grassland in Europe as man-made pasture, with a MSA value of 0.1. We followed a different assumption for two reasons. First, data limitations prevent us to combine the Agro-Ecological Zones (AEZ) available in our dataset with the map of potential vegetation, due to inconsistent spatial definition. Second, we consider that a MSA of 0.1 poorly reflects the richness in biodiversity of some natural pastures in Europe (Bignal and McCracken, 1996). We therefore consider that in our land data, the *"Savannah grassland"* category refers to natural land, to which we assign a MSA value of 1, independently from its potential vegetation, and the *Pasture* category refers to land under economic use and has a MSA value of 0.3. Simulations suggest that changes in this last value only slighlty impact our results. The land use categories in our model also comprise a mixed category, called *Other*, for which we can not compute a MSA. But, since the change in this category is very small between the ORG and the BASE scenario, its MSA value does not significantly affect the result: we find very close results whether the MSA of the *"Other"* category is 1 or 0.05.

A.4 Sensitivity analysis

The values of some key parameters of the model are highly uncertain. Many of them rely on assumptions since econometric estimations are not available. Previous studies have carried out sensitivity analysis on these parameters of the MIRAGE-Biof model. In particular, Laborde and

[1] Tuomisto et al. (2012) show that median nitrous oxide emissions per unit of field area were 31% lower in organic systems than in conventional systems. This lowering is mainly due to the lower use of nitrogen inputs in organic farming than in conventional farming.

Valin (2012) investigate the impact of changing the values of six parameters[2] on the NDF. They find large intervals for the possible values of the estimates of iLUC factors. We rather focus on the uncertainty that surrounds parameters specific to the representation of organic sectors that we have implemented. In particular, we change the values of the parameter governing the growth of the total factor productivity of organic sectors.

A.4.1 Increased productivity in organic sectors

One of the key drivers of our results is the difference between organic and conventional yields. Even if the calibration of the production technologies relies on micro-level data, a large uncertainty prevails on relative yields and on their evolution in the next years. Indeed, high organic yields are reached under experimental conditions, at least for some crops, and the potential for yield increase appears to be much larger in organic than in conventional farming, especially in the EU. Our *ORG* scenario is quite conservative concerning this last point: we assume that the growths in total factor productivity (TFP) of organic and conventional crops are equal. To test the sensitivity of our results to this assumption, we have first set the growth in TFP in organic sectors to be twice the growth in the corresponding conventional sectors. [3] Then, in a second step, we look for the necessary TFP growth rate to reach 20% of organic land in the EU without any land use change (i.e. a NDF equal to zero).

Doubling the TFP growth rate of organic sectors reduces the impact on world markets of the European policy that we simulate (20% of cultivated area under organic farming by 2020). The supply shocks for maize, rapeseed, sunflower and wheat are respectively reduced by 264 000, 14 000, 234 000 and 551 000 tons with respect to the *ORG* scenario presented in the paper. The impacts on world prices are therefore smaller and the resulting supply and demand displacements are limited. Increase in global cropland is reduced by 7.5% and the increase in deforested area by 8.1%, 12 400 hectares of forest are spared if the TFP growth doubles in organic sectors. GHG emissions are reduced by 8.9%. The resulting NDF is 7.2% lower than in the *ORG* scenario. Nevertheless, changes in MSA are marginal and the increase in the global MSA remains equal to 0.016%. Results concerning the use of fertilizers are also quite unchanged. Because of reduced supply displacement, the use of fertilizers increases slightly less in the regions other than Europe but this does not significantly affect the global result which still shows a decrease in fertilizer use by 0.35%, as in the *ORG* scenario.

We then look at the growth in the TFP of organic sectors that minimizes the global indirect environmental impacts of having 20% of the European cultivated area under organic farming by

[2] They consider five elasticities, those of endogeneous yield response, land substitution between highly subsitutable crops, land substituion between other crops, land expansion and the Armington elasticity. They also change the value of the coefficient of marginal yield return of newly cultivated land.

[3] Note that the consumption subsidy on organic products is the main driver of organic demand, and therefore of the share of agricultural land devoted to organic farming. Hence, after having changed the TFP growth rate of organic sectors, we adjust the rate of subsidy to organic products in order to keep the share of organic land in 2020 equal to 20%. When doubling the TFP growth rate, the subsidy rate decrases from 53.8% to 52.9%

2020. Annual growths between 2.5 and 2.7% for maize, sunflower and wheat and of 3.5% for rapeseed result in a NDF equal to 0.0057, 12 times lower than in the *ORG* scenario.[4] The increase in factor productivity reduces, and even cancels (in some sectors), the negative supply shock induced by the shift to organic farming. The supply of maize, sunflower and wheat by the EU increases by 182 000, 92 000 and 176 000 tons respectively compared to the baseline,[5] while the supply of rapeseed still decreases by 416 000 tons (instead of 874 000 tons in the *ORG* scenario). As a consequence, changes in world prices are almost non-existent: the price of rapeseed shows the highest increase, around 0.3%. Hence, there are almost no production or demand displacements. For example, the demand for vegetable oils other than those of rapeseed or sunflower, such as palm oil, does not significantly increase. At the world level, the total cropland area increases by 43 000 hectares with respect to the baseline, 93% less than in the *ORG* scenario. The newly cultivated land mainly comes fron the conversion of managed forests (-21 000 hectares) and pasture (-15 000 hectares). It is interesting to note that in this sensitivity analysis, the EU is among the regions most affected by land use changes: 62% of the converted managed forests are located in Europe. Greenhouse gas emissions due to land use changes increase by 10 million tons of CO_2 equivalent with respect to the baseline, while they increase by 95 million tons in the *ORG* scenario. Of these 10 million tons, 4 million tons come from the conversion of forests in Europe, 2.5 million tons from the conversion of peatlands in Indonesia and Malaysia and 2.1 million tons from the conversion of savannah and grassland in Brazil. Since primary forests are almost unaffected by these land use changes, biodiversity of primary habitats is not disturbed by the European organic policy. The rate of decrease in the use of fertilizer in the EU is unchanged between the sensitivity and the *ORG* scenarios, but the increase in the other regions lessens since production displacements fall.

[4] Following these annual increases, the total factor productivity of maize, sunflower and wheat is multiplied by 1.5 between 2004 and 2020, and the total factor productivity of rapeseed is multiplied by 1.7. The productvities of conventional maize, rapeseed, sunflower and wheat are in the meanwhile multiplied by 1.03, 1.07, 1.15 and 1.01. We obtain the annual rates of growth trying first to apply the same rate to all the crops and then adjusting the rate for rapeseed, since this parameter was critical to reach a very low NDF.

[5] In the *ORG* scenario, European supplies of maize, sunflower and wheat decrease by 3.5 million tons, 294 000 tons and 9.7 million tons respectively (see table II.2).

APPENDIX B

APPENDIX TO CHAPTER III

B.1 Proof of Proposition 1

For given t, t^*, c, c^*, (III.13) and (III.14) define a system of two linear equations with two unknowns. Solving this system gives (III.17). By definition of threshold crops $\underline{z}$ and $\bar{z}$, we must have $B(z) = 0$ for all $z \geq \bar{z}$ and $B^*(z) = 0$ for all $z \leq \underline{z}$. This implies that we must have $\phi(z) \geq 1/q$ or all $z \geq \bar{z}$ and $\phi^*(z) \geq 1/q^*$ for all $z \leq \underline{z}$. Differentiating (III.18) and its counterpart for Foreign, we get

$$\dot{\phi}(z) \equiv \frac{\phi'(z)}{\phi(z)} = \frac{A'(z)}{t^*/t + A(z)} - t^*\kappa_0 e^{-\theta(z)^2/2}\frac{\alpha'(z) - \theta'(z)\alpha(z)}{1 + t^*\alpha(z)\kappa(z)}$$

and

$$\dot{\phi}^*(z) = -\dot{A}(z) + \frac{A'(z)}{t^*/t + A(z)} - t\kappa_0 e^{-\theta(z)^2/2}\frac{\alpha'(z) - \theta'(z)\alpha(z)}{1 + t^*\alpha(z)\kappa(z)}.$$

Suppose $\kappa_0 = 0$: As $\dot{A}(z) > 0$, we have $\dot{\phi}(z) = A'(z)/[t^*/t + A(z)] > 0$ and $\dot{\phi}^*(z) = -\dot{A}(z)(t^*/t)/[t^*/t + A(z)] < 0$. Both conditions are thus satisfied if κ_0 is sufficiently small. They are also satisfied whatever the value of κ_0 if $\alpha(z)$ and $\theta(z)$ are constant as supposed in the symmetric case considered in section 3. Eq. (III.15) and (III.16) are derived from $\phi(\bar{z}) = 1/q$ and $\phi^*(\underline{z}) = 1/q^*$ respectively. Using these equations, we obtain $A(\bar{z})/A(\underline{z}) = [q^* + t^*\kappa(\bar{z})\alpha(\bar{z})][q + t\kappa(\underline{z})\alpha(\underline{z})]/(qq^*) > 1$. As $A(z)$ is increasing, we thus have $\bar{z} > \underline{z}$.

B.2 Proof of Lemma 1

Consider the crops produced more intensively by Home (symmetric results hold for Foreign). For all $z < \underline{z}$, it is obvious by setting $t = t_A$ and $q = 1/2$ in (III.22) that $\bar{p}(z) > \bar{p}_A(z)$ given by (III.7) where $c = (\ell - 1)(1 - b)/b$ and $B(z) = \alpha(z)$. For Foreign, we have $\bar{p}(z) < \bar{p}^*_A(z)$ iff

$$\frac{1 + t_A\kappa(z)\alpha(z)}{1 + 2t_A\kappa(z)\alpha(z)} > A(z).$$

As $\phi^*(z) > 1/q^*$ for all $z < \underline{z}$, we obtain by setting $t = t_A$ and $q^* = 1/2$ that

$$\frac{1 + 1/A(z)}{1 + t_A\kappa(z)\alpha(z)} > 2$$

for all $z < \underline{z}$, which gives, re-arranging terms

$$A(z) < \frac{1}{1 + 2t_A\kappa(z)\alpha(z)} < \frac{1 + t_A\kappa(z)\alpha(z)}{1 + 2t_A\kappa(z)\alpha(z)}$$

hence $\bar{p}(z) < \bar{p}_A^*(z)$ for all $z < \underline{z}$. For crops z in $[\underline{z}, \bar{z}]$, we obtain using (III.21) that $\bar{p}(z) < \bar{p}_A(z)$ iff $2/[1 + A(z)] < 1$, hence the result.

B.3 Proof of Proposition 3

Using (III.22) which simplifies to

$$\bar{p}_s(z) = \frac{2(\ell - 1)(1 - b)tq}{ba(z)}$$

and denoting by ϖ the Lagrange multiplier associated with the constraint, the first-order condition with respect to t gives

$$-(1 - b)q + \frac{Nh}{\sqrt{2\ln t}} - \varpi\frac{1}{\xi'(z_s)}\frac{t^*}{t} = 0$$

and the one with respect to q

$$\varpi = \frac{2(1 - b)}{b + 2q(1 - b)}.$$

Plugging the latter expression into the former and rearranging terms gives (III.30). At a symmetric equilibrium, using $\xi'(z_s) = A'(1/2) + 4$ and $\tau_A/c_A = Nh/(1 - b)$, we get

$$\frac{\tau}{c} = \frac{\tau_A}{c_A}\left[1 + \frac{A'(1/2)}{A'(1/2) + 8}\right]$$

where $c = c_A$. We thus have

$$\tau = \tau_A\left[1 + \frac{A'(1/2)}{A'(1/2) + 8}\right].$$

Denoting $M \equiv A'(1/2)$, we get $\lim_{M\to 0}\tau = \tau_A$, $\lim_{M\to +\infty}\tau = 2\tau_A$ and

$$\frac{d\tau}{dM} = \frac{8\tau_A}{(M + 8)^2} > 0.$$

B.4 Proof of Proposition 4

Differentiating (III.27) with respect to t, we obtain the following FOC

$$\frac{\partial V}{\partial t} = -(1-b)\left[\underline{z}\frac{q+2t\kappa}{t(q+t\kappa)} + \frac{(\bar{z}-\underline{z})(q^*+t^*\kappa)}{q(t^*-t)+t(1+t^*\kappa)} - \int_{\underline{z}}^{\bar{z}} \frac{A(z)/t^* - t^*/t^2}{A(z)t/t^* + t^*/t}dz\right] - h\frac{dZ}{dt}.$$

At a symmetric equilibrium, using $(A(z)-1)/(A(z)+1) = m(2z-1)$ and integrating gives (III.35). Using (III.34) in (III.35) and collecting terms, we arrive at

$$\frac{\partial V}{\partial t} = \frac{Nh}{t\sqrt{2\ln t}} - (1-b)\left[\frac{1}{2t} + \kappa\frac{2m(1+t\kappa)^2 - t\kappa}{2m(1+2t\kappa)(1+t\kappa)^2}\right].$$

Denote by t_0 the optimal tax when there is no cross-externality effects, i.e. $\kappa = 0$. It verifies (III.29) where $q = 1/2$. We have

$$\left.\frac{\partial V}{\partial t}\right|_{t=t_0} = -(1-b)\kappa\frac{2m(1+t_0\kappa)^2 - t_0\kappa}{2m(1+2t_0\kappa)(1+t_0\kappa)^2}$$

which is positive if

$$m \leq \frac{t_0\kappa}{2(1+t_0\kappa)^2} \leq 1/8.$$

We also have

$$\begin{aligned}\frac{1}{1-b}\left.\frac{\partial V}{\partial t}\right|_{t=t_A} &= \frac{1}{2t_A} + \frac{\kappa}{1+t_A\kappa} - \kappa\frac{2m(1+t_A\kappa)^2 - t_A\kappa}{2m(1+2t_A\kappa)(1+t_A\kappa)^2} \\ &= \frac{1}{2t_A} + \frac{t_A\kappa^2}{1+t_A\kappa}\frac{2m(1+t_A\kappa)+1}{2m(1+2t_A\kappa)(1+t_A\kappa)} > 0 \qquad \text{(B.1)}\end{aligned}$$

hence $t > t_A$ at the optimum.

B.5 Proof of Proposition 5

At a symmetric equilibrium (III.36) simplifies to

$$\frac{\partial V}{\partial q} = 2(1-b) + 2(1-b)\frac{1-(\bar{z}+\underline{z})}{1+2t\kappa}$$

where $\bar{z} + \underline{z} = 1$, hence $\partial V/\partial q = 2(1-b)$. We also obtain using (III.28) that at a symmetric equilibrium

$$\frac{dq}{dt} = -\frac{\frac{2(1+t\kappa)}{t}\int_{\underline{z}}^{\overline{z}} \frac{A(z)}{[A(z)+1]^2}dz - \frac{1}{2t}\int_{\underline{z}}^{\overline{z}} \frac{A(z)}{A(z)+1}dz}{t\kappa + \overline{z} - \underline{z}}.$$

Denoting $m_0 = (m+1)/2$, using

$$\begin{aligned}\int_{\underline{z}}^{\overline{z}} \frac{A(z)}{[A(z)+1]^2} dz &= \int_{\underline{z}}^{\overline{z}} [m_0 - mz - (m_0 - mz)^2] dz \\ &= (\overline{z} - \underline{z})(m_0 - m/2) + \frac{(m_0 - m\overline{z})^3 - (m_0 - m\underline{z})^3}{3m} \\ &= \frac{t\kappa}{2m(1+t\kappa)} \frac{2(1+t\kappa)^2 + 1 + 2t\kappa}{6(1+t\kappa)^2}\end{aligned}$$

and

$$\int_{\underline{z}}^{\overline{z}} \frac{A(z)}{A(z)+1} dz = \int_{\underline{z}}^{\overline{z}} (1 - m_0 + mz) dz = (\overline{z} - \underline{z})(1 - m_0 + m/2) = \frac{t\kappa}{2m(1+t\kappa)}$$

yields (III.37). Differentiating, we get

$$\frac{dq^2}{dt d\kappa} = -\frac{(4-m)(1+t\kappa)^2 + 4m(1+t\kappa) + 2}{12(1+t\kappa)^2[m(1+t\kappa)+1]^2} < 0.$$

Using (B.1) we obtain

$$\left.\frac{1}{1-b} \frac{dV}{dt}\right|_{t=t_A} = \frac{1}{2t_A} + \frac{t_A\kappa^2[2m(1+t_A\kappa)+1]}{2m(1+2t_A\kappa)(1+t_A\kappa)^2} - \frac{3 + 9t_A\kappa + 4(t_A\kappa)^2}{6t_A(1+t_A\kappa)[m(1+t_A\kappa)+1]} \tag{B.2}$$

where the second term diverges when m tends to 0 and decreases with m while the last term increases with m (decreases in absolute value). Assuming that m is large enough and $\kappa \approx 0$, we have

$$\begin{aligned}\left.\frac{1}{1-b} \frac{dV}{dt}\right|_{t=t_A} &= \frac{1}{2t_A} - \frac{1 + 3t_A\kappa}{2t_A[m(1+2t_A\kappa) + 1 + t_A\kappa]} + o(\kappa) \\ &= \frac{1}{2t_A}\left[\frac{m(1+2t_A\kappa) - 2t_A\kappa}{m(1+2t_A\kappa) + (1+t_A\kappa)}\right] + o(\kappa)\end{aligned}$$

where the bracketed term is negative if

$$m < \bar{m} \equiv \frac{2t_A\kappa}{1+2t_A\kappa}.$$

However, as the second term of (B.2) diverges when m approaches 0, we cannot neglect this term unless m is above some threshold $\underline{m} > t_A\kappa/(1+t_A\kappa)$. Indeed, in the case where $m = t_A\kappa/(1+t_A\kappa)$, we have

$$\begin{aligned}\left.\frac{1}{1-b} \frac{dV}{dt}\right|_{t=t_A} &= \frac{1}{2t_A} + \frac{\kappa}{2(1+t_A\kappa)} - \frac{3 + 9t_A\kappa + 4(t_A\kappa)^2}{6t_A(1+t_A\kappa)^2} \\ &= \frac{(t_A\kappa)^2}{3t_A(1+t_A\kappa)^2} > 0.\end{aligned}$$

Hence, assuming κ is small enough, cases where the environmental tax at equilibrium is lower than under autarky correspond to $m \in [\underline{m}, \bar{m}]$ where

$$\bar{m} - \underline{m} < \frac{2t_A\kappa}{1+2t_A\kappa} - \frac{t_A\kappa}{1+t_A\kappa} = \frac{t_A\kappa}{(1+t_A\kappa)(1+2t_A\kappa)} \leq \frac{\sqrt{2}}{4+3\sqrt{2}} \approx 0.17.$$

B.6 Proof of Proposition 6

Without biodiversity effects, using (III.38) and (III.39) with $\kappa = 0$, we obtain that $v(\tilde{y}_A^W(z)) \geq v(\tilde{y}^W(z))$ iff

$$1 - \frac{4A(z)}{[1+A(z)]^2} \geq \frac{t - t_A}{t_A - \mu}.$$

As $A(z)/[1+A(z)]^2$ is cap-shaped with a maximum equal to $1/4$ at $z = 1/2$, this condition is satisfied for all z only if $t = t_A$. With $t > t_A$, it could be satisfied for z belonging only to one of the extremes of the crops' range, i.e. for z either close to 0 or close to 1, if $t - t_A$ is small enough. With biodiversity effects, for $z \in [\underline{z}, \bar{z}]$, using (III.40) and assuming that $t \geq t_A$, we obtain that $v(\tilde{y}_A^W(z)) \geq v(\tilde{y}^W(z))$ iff

$$\frac{4A(z)}{[1+A(z)]^2} - 1 \geq \frac{t\kappa(1+t\kappa) - \kappa t_A(1+t_A\kappa)}{(1+t\kappa)^2 - \kappa t_A(1+t_A\kappa) + \mu\kappa}$$

which is impossible unless $t = t_A$ and $z = 1/2$ since the last term is positive. For all $z \in [0, \underline{z}] \cup [\bar{z}, 1]$, using (III.38), we have $v(\tilde{y}_A^W(z)) \geq v(\tilde{y}^W(z))$ iff

$$2 - \frac{4A(z)}{[1+A(z)]^2} \geq \frac{t(1+2t\kappa) - \mu}{t_A(1+t_A\kappa) - \mu}.$$

A necessary condition is given by $2 > [t(1+2t\kappa) - \mu]/[t_A(1+t_A\kappa) - \mu]$, or re-arranging terms $t_A - \mu > (t - t_A)[1 + 2\kappa(t + t_A)]$ which is satisfied only if $t - t_A$ is not too large and κ sufficiently small.

B.7 Proof of Proposition 7

A second-order approximation gives

$$\begin{aligned}\tilde{p}(z) &= \frac{\alpha(z)(1-b)LR}{\tilde{y}(z)} \approx \frac{\alpha(z)(1-b)LR}{y(z)}\left[1 - \frac{\tilde{y}(z) - y(z)}{y(z)} + \left(\frac{\tilde{y}(z) - y(z)}{y(z)}\right)^2\right] \\ &= \bar{p}(z)\left[2 - \frac{\tilde{y}(z)}{y(z)} + \left(\frac{\tilde{y}(z) - y(z)}{y(z)}\right)^2\right]\end{aligned}$$

and thus

$$p(z) \approx \bar{p}(z)\left[1 + E\left(\frac{\tilde{y}(z) - y(z)}{y(z)}\right)^2\right] = \bar{p}(z)[1 + v(\tilde{y}(z))^2].$$

A first order approximation yields

$$\frac{E[(\tilde{p}(z)-\bar{p}(z))^2]^{1/2}}{\bar{p}(z)} \approx E\left[\left(1-\frac{\tilde{y}(z)}{y(z)}\right)^2\right]^{1/2} = v(\tilde{y}(z)),$$

which gives

$$\begin{aligned}\sigma(\tilde{p}(z)) &\approx E\left[\tilde{p}(z)-\bar{p}(z)-\bar{p}(z)v(\tilde{y}(z))^2\right]^{1/2} = \left(E\left[(\tilde{p}(z)-\bar{p}(z))^2\right]-\bar{p}(z)^2 v(\tilde{y}(z))^4\right)^{1/2} \\ &= \bar{p}(z)v(\tilde{y}(z))(1-v(\tilde{y}(z))^2)^{1/2}.\end{aligned}$$

From $p_u^\gamma(z) = p(z) + s_u^\gamma \sigma(\tilde{p}(z))$ we get

$$\begin{aligned} s_u^\gamma &\approx \frac{1}{\sigma(\tilde{p}(z))}\left(\frac{\alpha(z)(1-b)LR}{y(z)-s_\gamma\sigma(\tilde{y}(z))} - \bar{p}(z)(1+v(\tilde{y}(z))^2)\right) \\ &= \frac{\bar{p}(z)}{\sigma(\tilde{p}(z))}\frac{y(z)(1+v(\tilde{y}(z))^2)-y(z)+s_\gamma\sigma(\tilde{y}(z))}{y(z)-s_\gamma\sigma(\tilde{y}(z))} \\ &\approx \frac{y(z)v(\tilde{y}(z))^2+s_\gamma\sigma(\tilde{y}(z))}{v(\tilde{y}(z))(1-v(\tilde{y}(z))^2)^{1/2}(y(z)-s_\gamma\sigma(\tilde{y}(z)))} \\ &= \frac{v(\tilde{y}(z))+s_\gamma}{(1-v(\tilde{y}(z))^2)^{1/2}(1-s_\gamma v(\tilde{y}(z)))}\end{aligned}$$

which gives (III.44). Similar derivations for $p_d^\gamma(z) = E[\tilde{p}(z)] - s_d^\gamma \sigma(\tilde{p}(z))$ yield (III.43) and (III.45).

B.8 Food Price Index

The food price index is defined as

$$F = \int_0^1 y(z)p(z)dz \Big/ \int_0^1 y(z)dz$$

where $p(z)$ is approximated by (III.41). As we also have $\bar{p}(z)y(z) = cNB(z)$, we get $p(z)y(z) \approx cNB(z)[1+v(\tilde{y}(z))^2]$. In autarky, $c = (\ell-1)(1-b)/b$. Using $B(z)=1$, $\theta(z)=\theta$, which implies $\mu(z)=\mu$, and $y(z) = a(z)N/[t(1+t\kappa)]$ gives

$$\int_0^1 p(z)y(z)dz = \frac{N(\ell-1)(1-b)}{b}\left[1+\frac{t(1+t\kappa)-\mu}{N\mu}\right].$$

Using $a(z) = \mu_0 e^{\theta^2/2}[1-m(2z-1)]$ yields

$$\int_0^1 y(z)dz = \frac{N}{t(1+t\kappa)}\int_0^1 \mu_0 e^{\theta^2/2}[1-m(2z-1)]dz = \frac{N\mu_0 e^{\theta^2/2}}{t(1+t\kappa)}$$

and thus

$$F_A = \frac{(\ell-1)(1-b)t(1+t\kappa)}{b\mu_0 e^{\theta^2/2}}\left[1+\frac{t(1+t\kappa)-\mu}{N\mu}\right].$$

Under free trade, we have $\bar{p}(z)y(z) = 2N(l-1)(1-b)/b$. $B(z)=2$ for crops produced by only one country and we get, using $\underline{z}+\bar{z}=1$,

$$\int_0^{\underline{z}} p_s(z)y(z)dz + \int_{\bar{z}}^1 p_s(z)y^*(z)dz = \frac{4\underline{z}N(l-1)(1-b)}{b}\left[1+\frac{t(1+2t\kappa)-\mu}{2\mu N}\right].$$

For crops produced by both countries we obtain

$$\begin{aligned}\int_{\underline{z}}^{\bar{z}} p_m(z)y_W(z)dz &= \int_{\underline{z}}^{\bar{z}} \frac{2N(\ell-1)(1-b)}{b}\left[1+\frac{(1+t\kappa)(1+2t\kappa)-\mu\kappa}{2\mu N\kappa} - \frac{2(1+t\kappa)^2}{\mu N\kappa}\frac{A(z)}{[1+A(z)]^2}\right]dz \\ &= \frac{2N(\ell-1)(1-b)}{b}\left\{\left[1+\frac{(1+t\kappa)(1+2t\kappa)-\mu\kappa}{2\mu N\kappa}\right](\bar{z}-\underline{z}) - \frac{t[2(1+t\kappa)^2+1+2t\kappa]}{\mu N m 6(1+t\kappa)}\right\}.\end{aligned}$$

The corresponding quantities are given by

$$\begin{aligned}\int_0^{\underline{z}} y(z)dz + \int_{\bar{z}}^1 y(z)dz &= \frac{2N}{t(1+2t\kappa)}\left\{\int_0^{\underline{z}} a(z)dz + \int_{\bar{z}}^1 a^*(z)dz\right\} \\ &= \frac{2N\mu_0 e^{\theta^2/2}}{t(1+2t\kappa)}\left[\underline{z}(m+1-m\underline{z}) + 1 - \bar{z}(1+m\bar{z}-m)\right]\end{aligned}$$

and

$$\int_{\underline{z}}^{\bar{z}} y_W(z)dz = \frac{N}{t(1+t\kappa)}\int_{\underline{z}}^{\bar{z}}[a^*(z)+a(z)]dz = \frac{2N\mu_0 e^{\theta^2/2}(\bar{z}-\underline{z})}{t(1+t\kappa)} = \frac{2N\mu_0 e^{\theta^2/2}\kappa}{m(1+t\kappa)^2}.$$

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Abstract This thesis analyses both theoretically and empirically some of the issues that emerge when applying environmental policies to the agricultural sector in a trade context. In a first part, Chapter II illustrates the possible leakage effects of environmental policies implemented unilaterally. A computable general equilibrium model is used to quantify the indirect global impacts of a greening of European agriculture through a large shift to organic farming. Organic farming is known for its local environmental benefits, especially on water and soil quality, biodiversity and greenhouse gas emissions. However, organic yields are on average 25% lower than those of conventional farming. We calibrate organic production technologies using micro-level data and find that using organic production techniques on 20% of the European area cultivated with maize, rapeseed, sunflower and wheat results in a large negative productivity shock. This shock affects global markets and induces production and demand displacements, unless the yield gap is reduced. The resulting land use changes are assessed, as well as the corresponding changes in greenhouse gas emissions, chemical inputs use and biodiversity. The negative indirect effects on the environment appear limited compared to the local benefits of adopting greener forms of agriculture in the EU. However, in the case of greenhouse gases, the indirect emissions more than offset the local benefits of organic agriculture. In the case of chemical pollution and biodiversity, results show that indirect effects deserve to be accounted for in life cycle analyses. These findings should not be used to point a finger on organic farming, a large variety of policies and consumption patterns have greater land use change impacts. Nevertheless, they rise some issues, especially on the need for more systematic sustainability assessments, even for environmental polices, the importance of research and development in organic farming to reduce yield gaps and of public policies to help to remove economic factors that could limit the increase of organic yields, such as the relative cost of production factors. In a second part, focus is on crop biodiversity, which is known to maintain agricultural productivity under a large range of environmental conditions. Interactions between crop biodiversity effects, environmental policies and trade are complex. Specialisation induced by trade, following comparative advantages, tends to reduce the number of crops cultivated in a given country and then reduces crop biodiversity. A decrease in crop biodiversity results in lower resilience to pest attacks. To face higher pest attacks, farmers use pesticides. But since pesticides harm environment and human health, governments regulate their use. In a free trade context, an environmental policy on pesticides can thus have a strategic aspect: allowing the use of more pesticides can lead to gain larger agricultural market shares. Chapter III represents these interactions in a Ricardian trade model. It shows that, because *not in my backyard* effects are larger than strategic impacts, the optimal environmental policy is more stringent under trade than under autarky. Furthermore, because of this stringency, production volatility is generally higher under trade. This could explain part of the background volatility observed on agricultural markets, which have been historically more volatile than those of manufactured products. Chapter IV empirically confirms the positive impact of crop biodiversity on agricultural production using a large dataset on South African agriculture. Developing a structural approach, it also analyses the role played by biodiversity on the exposure of farmers to production risks and downside risks.

Résumé Cette thèse analyse à la fois théoriquement et empiriquement certaines des questions qui se posent lors de l'utilisation de politiques environnementales dans le secteur agricole, en situation de commerce. Dans une première partie, le chapitre II illustre les effets de fuite que peuvent engendrer des politiques environnementales mises en œuvre unilatéralement. Un modèle d'équilibre général calculable est utilisé pour quantifier les impacts indirects sur l'environnement à l'échelle mondiale d'un accroissement des surfaces dédiées à l'agriculture biologique en Europe. L'agriculture biologique est connue pour ses bénéfices locaux sur l'environnement mais ses rendements sont inférieurs de 25% en moyenne à ceux de l'agriculture conventionnelle. Nous calibrons les technologies de production de l'agriculture biologique avec des données micro-fondées et trouvons qu'utiliser ces techniques sur 20% des surfaces européennes consacrées au maïs, colza, tournesol et blé conduit à un choc de productivité négatif. Ce choc a des conséquences sur les marchés mondiaux et induit des déplacements d'offre et de demande. Les changements d'utilisation des sols résultants sont évalués, ainsi que les changements en termes d'émissions de gaz à effet de serre, d'utilisation d'intrants et de biodiversité. Les effets indirects négatifs sur l'environnement semblent limités, sauf en ce qui concerne les émissions de gaz à effet de serre. Nous montrons également que les effets indirects concernant l'utilisation d'intrants et la biodiversité méritent d'être pris en compte dans les analyses de cycle de vie. Ces résultats ne doivent pas être utilisés pour pointer du doigt l'agriculture biologique, mais ils soulèvent quelques questions, en particulier sur la nécessité d'effectuer des analyses d'impact de façon plus systématique, y compris pour les politiques environnementales, et l'importance de la recherche et développement mais également des politiques publiques pour lever les obstacles techniques et économiques à l'augmentation des rendements en agriculture biologique. Dans une deuxième partie de la thèse, l'attention est portée sur la biodiversité des cultures, reconnue pour stabiliser la productivité agricole sous différentes conditions environnementales. Les interactions entre les effets de cette biodiversité, les politiques environnementales et le commerce sont complexes. La spécialisation induite par le commerce réduit la biodiversité en diminuant le nombre d'espèces cultivées. La biodiversité influe positivement sur les niveaux de production, entre autre, en améliorant la résistance aux ravageurs. Pour faire face à des attaques plus fréquentes, les agriculteurs utilisent des pesticides. Mais ces derniers ont des impacts négatifs sur l'environnement et la santé humaine, leur utilisation est donc réglementée. Une politique environnementale concernant les pesticides peut ainsi avoir un aspect stratégique: autoriser l'utilisation de plus de pesticides peut permettre de gagner en compétitivité. Le chapitre IV représente ces interactions dans un modèle ricardien de commerce. Il montre que, parce que les effets $NIMBY$ sont plus importants que les impacts stratégiques, la politique environnementale est plus stricte en situation de commerce qu'en autarcie. De ce fait, la volatilité de la production agricole est généralement plus élevée en commerce. Cela pourrait en partie expliquer la volatilité de fond observée sur les marchés agricoles, historiquement plus volatiles que ceux des produits manufacturés. Le chapitre IV confirme empiriquement l'impact positif de la biodiversité sur la production agricole en utilisant une large base de données sur l'agriculture sud-africaine. En utilisant une approche structurelle, il analyse également les liens entre biodiversité et exposition des agriculteurs aux risques de production, en particulier ceux à la baisse.