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Spatial economic modelling of greenhouse gas abatement on agricultural land

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Abstract

This thesis investigates how agricultural land can be used for cost-effective abatement of greenhouse gas (GHG) emissions. The impacts of spatial relationships and characteristics of land on the costs of producing biofuel for the purpose of emissions abatement, and on global emissions caused by agricultural policies, are evaluated using spatial economic models.

The first paper in the thesis examines the cost-effective spatial configuration of production of second-generation biofuel on agricultural land in Sweden. To this end, a spatial economic biofuel localization model is developed. The results show that the localization of the few high-capacity biofuel production facilities is mainly determined by the opportunity costs for feedstock.

In the second paper, the focus is on the role of second-generation biofuel for cost-effective GHG emissions abatement in the transport sector. The biofuel model is expanded to incorporate transport fuel consumption and GHG emissions. Findings indicate that domestic biofuel is a cost-effective abatement measure, with high potential for reducing overall costs, particularly at low emissions targets.

The third paper investigates how the use of abandoned agricultural land affects the role of biofuel as a cost-effective GHG emissions abatement option in the transport sector. Abandoned agricultural land is found to reduce costs of emissions abatement substantially, primarily attributed to carbon sequestration and low feedstock costs.

In the fourth paper, the focus is shifted to the agricultural sector. This paper assesses the impact of coupled production subsidies under the EU Common Agricultural Policy on agricultural GHG emissions in the EU and globally, utilizing an agricultural sector model. The removal of the coupled subsidies is found to decrease EU GHG emissions, but there is a 75 per cent global emissions leakage.

Key words: Abandoned agricultural land, agricultural land, agricultural subsidies, biofuel, emissions leakage, greenhouse gas emissions, spatial models.

Spatial ekonomisk modellering av utsläppsminskningar på jordbruksmark

Sammanfattning

Den här avhandlingen utforskar hur jordbruksmark kan användas för att minska växthusgasutsläpp på ett kostnadseffektivt sätt. Spatiala ekonomiska modeller används för att analysera hur relationer mellan regioner och markens egenskaper påverkar kostnaden för att producera biobränslen som syftar till att reducera utsläpp, och hur de påverkar globala utsläpp som orsakats av jordbrukspolitiken.

Den första artikeln i avhandlingen undersöker hur produktion av andra generationens biobränslen på jordbruksmark i Sverige kan organiseras kostnadseffektivt. För att göra detta utvecklas en spatial ekonomisk modell för lokaliseringsbeslut av biobränsleproduktion. Resultaten indikerar att lokaliseringen av biobränslefabriker i huvudsak påverkas av råvarans alternativkostnader.

Den andra artikeln fokuserar på vilken roll andra generationens biobränslen kan ha för kostnadseffektiv utsläppsminskning inom transportsektorn. Biobränslemodellen utvecklas till att inkludera konsumtion av transportbränslen och växthusgasutsläpp. Resultaten tyder på att inhemsk biobränsleproduktion är en kostnadseffektiv åtgärd för utsläppsminskning, med störst potential vid låga utsläppsmål.

Den tredje artikeln analyserar hur användningen av övergiven jordbruksmark påverkar biobränslets roll för kostnadseffektiv minskning av utsläpp inom transportsektorn. Övergiven jordbruksmark reducerar kostnaderna avsevärt, främst på grund av ökad kolinlagring och lägre råvarukostnader.

I den fjärde artikeln skiftar fokus till jordbrukssektorn. Artikeln undersöker påverkan av ett kopplat produktionsstöd inom EU:s jordbrukspolitik på växthusgasutsläpp från jordbruket både i EU och globalt, med hjälp av en modell för jordbrukssektorn. Resultaten visar att om det kopplade stödet avlägsnas minskar utsläppen i EU men leder till en läckageeffekt där 75 procent av utsläppen överförs till resten av världen.

Nyckelord: biobränsle, jordbrukssubventioner, jordbruksmarkspatiala modeller, utsläppsläckage, växthusgasutsläpp, övergiven jordbruksmark.

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I. Ida Nordin, Katarina Elofsson and Torbjörn Jansson (2022). Optimal localisation of agricultural biofuel production facilities and feedstock: a Swedish case study. European Review of Agricultural Economics, vol 49 (4), pp. 910-941. doi:10.1093/erae/jbab043

II. Ida Nordin, Katarina Elofsson and Torbjörn Jansson. Cost-effective reductions in greenhouse gas emissions: reducing fuel consumption or replacing fossil fuels with biofuels (under review)

III. Ida Nordin. Cost-effective use of abandoned agricultural land for biofuel production (submitted)

IV. Torbjörn Jansson, Ida Nordin Fredrik Wilhelmsson, Peter Witzke, Gordana Manevska-Tasevska, Franz Weiss, and Alexander Gocht (2020). Coupled Agricultural Subsidies in the EU Undermine Climate Efforts. Applied Economic Perspectives and Policy, 43 (4), pp. 1503-1519. doi:10.1002/aepp.13092

Papers I and IV are reproduced with the permission of the publishers.

The contribution of Ida Nordin to the papers included in this thesis was as follows:

I. I formulated the research idea and formulated scenarios together with my co-authors. I formalized the model and coded it. I analysed the results and wrote a first draft of the paper. I finalized the manuscript together with my co-authors.

II. I formulated the research idea and formulated scenarios together with my co-authors. I formalized the model and coded it. I analysed the results and wrote a first draft of the paper. I finalized the manuscript together with my co-authors.

III. I formulated the research idea and formulated scenarios. I extended the model and coded it. I analysed the results and wrote the manuscript.

IV. I participated in the formulation of the research idea and development of scenarios together with my co-authors. I coded the scenarios with my co-authors. I contributed to the estimation of emissions intensities. I analysed the results together with my co-authors. Together with my co-authors, I wrote the first drafts of the manuscript, and rewrote the manuscript in the revision process.

List of figures

Abbreviations

ALA	Abandoned agricultural land
CAP	Common Agricultural Policy
CO_2	Carbon dioxide
EU	European Union
EU+	EU countries plus Norway, the United Kingdom, Turkey and the Balkan countries
GHG	Greenhouse gas
LUC	Land use change
SOC	Soil organic carbon

1. Introduction

Under the Paris Agreement most of the countries in the world have agreed to substantially reduce greenhouse gas emissions, as a means to reduce the negative impact climate change has on societies (UNFCCC, 2023). The transport sector covers 17 per cent of global greenhouse gas emissions, while the agricultural sector covers about 12 percent (Climate Watch, 2022). This implies that both sectors can be important for the overall reduction in greenhouse gas emissions. The Paris Agreement is implemented through policies in the signatory countries. The EU has an emissions reduction target of 55 per cent until 2030 relative to 1999 levels, and a target to be carbon neutral by 2050 (European Commission, 2021). Both the transport sector and the agricultural sector are expected to contribute to these targets. The emissions in these sectors are affected by sector specific climate policies that directly incentivize reductions in greenhouse gas emissions, or policies that have an indirect impact on greenhouse gas emissions.

Land use for the purpose of economics activity can both have negative and positive impact on climate change. The land use in a certain region is determined by regional natural and cultural characteristics. A limited share of the global land area is suitable for agricultural production. Climate, soil characteristics, demographics, technology, culture, etc., influence which crops are grown, which livestock is held and may graze the land, or if the land is not used for agricultural production at all. There is a large spatial variation in agricultural land use, as shown in Figure 1, which shows agricultural land use intensity in the EU. For example, the agricultural land use is intense in the north of France, but almost absent in the north of Finland. Food and fodder production have long been the main use of agricultural land, to secure food availability and provide income. However, the demand for agricultural land to produce feedstock for bioenergy can change this. The purpose of bioenergy production is to supply energy with lower greenhouse gas emissions than fossil fuels, and/or to secure domestic energy production. In this way, the energy sector and the agricultural sector become increasingly connected. Agricultural land used for the production of food and transport biofuel, the impact of land heterogeneity on these uses, the spatial configuration of these different land uses, and the resulting greenhouse gas emissions, are the focuses of this thesis.

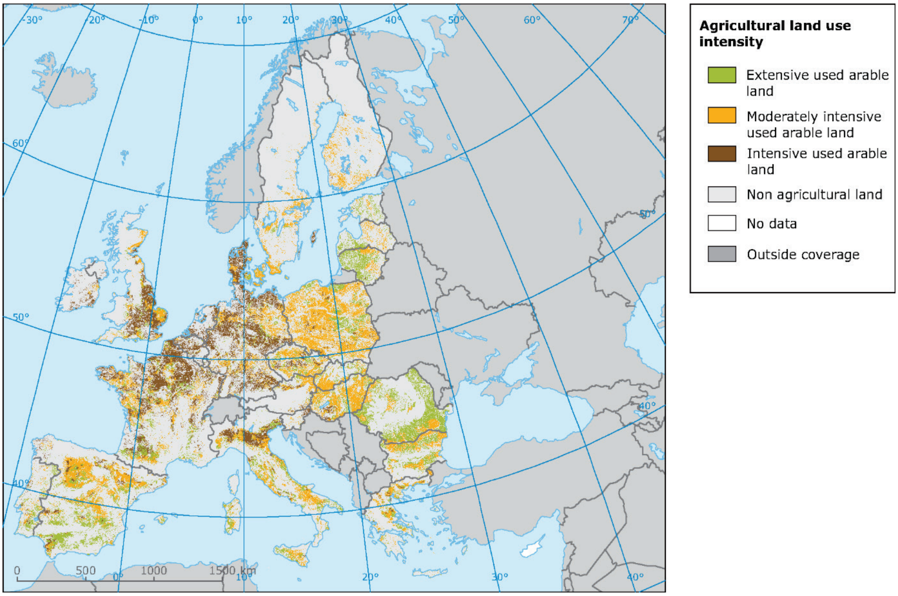

Figure 1 Agricultural land use intensity in the EU. Source: European Environment Agency (EEA).

1.1 Biofuel production for greenhouse gas emissions abatement

One of the main options for greenhouse gas emissions abatement in the transport sector is to use transport biofuels instead of fossil fuels, as biofuels generally have much lower emissions (Creutzig et al., 2015). Biofuels are convenient as in many cases they can be blended with fossil fuels for use in existing vehicles (Sims et al., 2014). For biofuel to significantly contribute to the ambitious emissions target in the Paris Agreement, more feedstock for biofuel production is needed. To increase feedstock production for this aim, the main land types to use are agricultural land and forests (Creutzig et al., 2015). Therefore, there is a need to assess the potential of biofuel production from agricultural land, which is the focus of this thesis.

First generation biofuel technologies use first generation bioenergy crops as feedstock to produce biofuels, mainly food crops such as corn, cereals and rapeseed. The process of converting these crops to biofuel is relatively simple and cheap. However, the first generation bioenergy crops compete directly with food production, and can pose a threat to food security (Jeswani et al., 2020). They give rise to the same level of greenhouse gas emissions as food crops during the cultivation, and can cause emissions from indirect land use changes if cropland is extended to compensate for land taken from food production (Berndes et al., 2011). This reduces the potential of emissions reduction by replacing fossil fuels with biofuels.

To lessen these problems, second generation, i.e. non-food, bioenergy feedstock can be used. By-products from agriculture form one stream of second generation feedstock that does not require additional land, but for larger increases in production, the production of dedicated bioenergy crops has to increase, by intensification, or by using more land (Prade et al., 2017). Perennial bioenergy crops are one type of second generation bioenergy crops and consist of lignocellulosic material that have a high energy yield and can grow with good results on low productive land. This implies that these crops compete less with food production than first generation bioenergy crops. They have relatively low production costs and little environmental impact in terms of, e.g., nutrient leakage, and the cultivation process give rise to less greenhouse gas emissions than food crops (Börjesson et al., 2013). The potential to produce the feedstock differs across space (Creutzig, et al., 2015).

The perennial bioenergy crops can be used as feedstock to process to biofuel by different so-called advanced biofuel technologies. However, the process of converting perennial bioenergy crops to biofuel is not as developed as that for first generation biofuel technologies. The production is still relatively expensive, but the cost is expected to decrease (Brown et al., 2020).

In the EU, biofuels are affected by the targets in the Renewable Energy Directive. These targets require 32 percent renewable energy as a share of final energy consumption by 2030, with national targets for each country (European Parliament, 2018). The specific target for transport fuels is that at least 14 per cent of consumed transport fuels in the EU should come from renewable sources by 2030 (European Parliament, 2018). Further, there are biofuel quotas, mandatory blending, and tax exemptions for biofuels in the EU (Banja et al., 2019). National climate polices affect biofuels, such as the greenhouse gas emissions reduction target for the transport sector in Sweden: a 70 per cent reduction by 2030 compared with 2010 (Government Offices of Sweden, 2017). The existence of these policies implies a need for more biofuel production in the EU, or imports, to fulfil specific biofuel targets. It also shows that there is a need to determine the role of biofuel for emissions abatement to be able to fulfil more general targets in a cost-effective manner.

1.2 Agriculture and greenhouse gas emissions leakage

As the agricultural greenhouse gas emissions constitute a large share of total global emissions there is a potential for the sector to contribute to emissions reductions (Allen and Maréchal, 2017; Grosjean et al., 2016). Livestock, and ruminants in particular, emits the largest share of agricultural greenhouse gas emissions. Meat and dairy that are produced from ruminants, have the highest greenhouse gas emission intensities per unit of product among agricultural products, but emission intensities differ across regions (Lesschen et al., 2011).

One of the key motivations for the EU's Common Agricultural Policy (CAP) is to provide income support to farmers to secure food production in the EU (Treaty Establishing the European Economic Community, 1957), while environmental sustainability has been added as an objective in later policy reforms (Brady et al., 2017). The current CAP has a large budget and consists mostly of direct per hectare payments for all qualifying agricultural land, other direct subsidies, and to a lesser extent specific subsidies to, e.g., rural development and environmental measures (European Commission, 2023a). Specific climate measures are rare in the CAP. However, the European Commission has stressed a need for more climate friendly agricultural practices (European Commission, 2017).

While carbon pricing is viewed as the most efficient way to incentivise emissions reductions, another option is to remove subsidies to polluting industries. Such subsidies increase production of a polluting good, where carbon pricing would have reduced it (van Beers and van den Bergh, 2001). In the agricultural sector, the CAP includes production subsidies mainly given to ruminant production. As ruminants have high greenhouse gas emissions, this can be seen as subsidies to a polluting industry, and it could be beneficial for the climate if it were removed.

A concern regarding implementation of unilateral climate polices is whether these policies cause emissions leakage. Emissions leakage is a phenomenon that can arise following a unilateral policy that decreases production and associated emissions in a country, or region such as the EU (Markusen, 1975; Zhang, 2012). The decrease in EU production leads to an increase in net imports to the EU and, hence, an increase in world market prices. This gives an incentive for non-EU producers to increase their production. Consequently, the EU production is "moved abroad", along with its emissions, and this is the so-called emissions leakage. The degree of leakage depends both on the magnitude of production changes made abroad, and on the difference in emissions per unit of product across countries. Characteristics of the agricultural sector, with considerable trade in agricultural products globally and differences in emission intensities across regions imply that there is a considerable risk for emissions leakage.

1.3 Aim and research questions

The aim of this thesis is to provide insights on the cost effectiveness of using agricultural land for abatement of greenhouse gas emissions. To this end, spatial economic models are used.

The aim is concretized in the following research questions:

- What is the cost and the spatial configuration of cost-effective production of second generation biofuel on agricultural land? (Paper I)
- What is the role of second generation biofuel for cost effective greenhouse gas emissions abatement in the transport sector? (Paper II)
- How can abandoned agricultural land contribute to, and alter, cost effective greenhouse gas emissions abatement in the transport sector? (Paper III)
- How does the coupled subsidies to animal production under the EU's Common Agricultural Policy affect agricultural greenhouse gas emissions in the EU and globally? (Paper IV)

1.4 Literature review

The relevant economics literature for the first three papers in the thesis includes literature on how biofuel production is organized spatially, literature on biofuels' role for emissions abatement, and literature investigating how marginal land and abandoned agricultural land contribute to biofuel production.

To develop strategies for biofuel production, it is important to take localization choices into account. The decisions for a single production facility can focus on the optimal land use choices around a facility, which is studied by, e.g., Lankoski and Ollikainen (2008). They use the von Thünen model and explore how crops should be produced around the facility. Rentizelas and Tatsiopoulos (2010) optimize the design and location of one facility within a specific region to minimize total costs. Rozakis et al. (2013) study the localization of one facility when feedstock supply is endogenous.

When the studied area is large enough, the localization of multiple production facilities and adjacent feedstock production become relevant. Leduc (2009) uses a localization model to minimise the total cost of the whole biofuel supply chain, with feedstock from the forestry sector. The optimal location and the optimal number of biofuel production facilities are decided simultaneously, to meet a national production target. De Jong et al. (2017), minimise the cost of locating biofuel production facilities using forest feedstock, and compare the role of different cost-reduction strategies to reduce costs for the whole forestry sector. Wetterlund et al. (2013) investigate the role of an explicit choice between different biofuel technologies for achieving production targets at least cost. Bai, Ouyang and Pang (2012) investigate the role of market power for the biofuel feedstock costs. They model this with a Stackelberg game where biofuel production facilities are modelled as leaders and the farmers as followers. The literature lacks spatial localization models for biofuel on agricultural land for many forest dominated regions. Further, modelling of the increasing opportunity costs that can occur due to competition over land is often missing. These gaps in the literature are addressed in the first paper of the thesis.

A range of studies focus on the role biofuel can have for greenhouse gas emissions abatement. For example, Mercure et al. (2018) show that biofuels can contribute the transport sector's contribution to reaching the Paris Agreement. They find that a mix of policies that lead to a decrease in travel distance, more fuel-efficient combustion engines, more electric vehicles, and the replacement of fossil fuel with biofuels could be sufficient to realize required emissions reductions in the EU. To this end they use a global dynamic least-cost simulation model for bioenergy and transportation. Chen et al. (2014) study the choice between biofuels and other transport related abatement measures with an integrated fuel and agricultural model. They find that the optimal response to a carbon tax would be a relatively low use of biofuel. Further, they find a biofuel blend-in requirement to be inefficient, because of a rebound effect on fossil fuel use arising due to decreases in world gasoline prices. Haasz et al. (2018) study the role of the transport sector for total greenhouse gas emissions reductions. They use an economy wide model and find that the transport sector's role should be rather small in the near future, due to high marginal abatement costs. In the long run, lowered costs for electric vehicles implies that the transport sector could contribute more. Millinger et al. (2018) show how different biofuel technologies should be mixed over time to achieve emission reductions at least cost by using biofuels.

While the role of biofuel for greenhouse gas emissions abatement have been investigated in various ways, the role of spatial configuration of biofuel production for emissions abatement have not been studied. The second paper of the thesis addresses this gap.

Land use for biofuel has been debated as it could cause competition with agricultural production (Jeswani et al., 2020). One alternative that has been proposed is the use of marginal land, which is low productivity land of little use for agricultural production. Abandoned agricultural land can be of particular interest as the competition over land with food production is small (Creutzig, et al., 2015). However, Bryngelsson and Lindgren (2013) find that to reach bioenergy production targets, production of feedstock for bioenergy would be more efficient on productive agricultural land than on low productivity land. Choi et al. (2019) model how different land types could be used to produce biofuels to reach greenhouse gas emissions abatement targets at least cost using an energy model linked to an agricultural land model, finding that some of the marginal land is used. Havlík et al. (2011) find a larger use of marginal land, modelled with a global economic partial equilibrium model for different land uses. Lee et al. (2023) model the possibility to use marginal land for biofuel feedstock production with a model that covers both agricultural production and the fuel market in the US, including impacts on the world market. They find that marginal land should be used for cost effective biofuel production, but that large areas of arable land also would be needed.

The literature mainly focuses on marginal land, while studies of the economic potential of abandoned agricultural land is scarce. The focus of similar studies has mainly been on how using abandoned land can reduce competition, but not addressing the potential for abandoned agricultural land to reduce costs for emissions abatement. Further, the impact of abandoned agricultural land on the spatial organization of biofuel production is missing. These issues are addressed in the third paper of the thesis.

For the fourth research question the relevant literature concerns the literature on environmentally harmful subsidies, and literature on the emissions leakage of regional policies in the agricultural sector.

van Beers and van den Bergh (2001) show that environmentally harmful subsidies can have a large impact on environmental outcomes. They argue that the large number of subsides in the agricultural sector can also cause a lock-in effect, where the subsidies are difficult to remove. Brady et al. (2017) simulate net global greenhouse emission reductions from removing the direct payments in the EU, which are environmentally harmful subsidies as agriculture causes greenhouse gas emissions. They find a net decrease in global emissions, but considerable emissions leakage. Moreover, the impact of unilateral policies on agricultural emissions are studied by Fellmann et al. (2012) and Fellmann et al. (2018), who simulate consequences of reducing greenhouse gas emissions in the agricultural sector in the EU, to comply with global climate agreements. Both studies find that trade in agricultural products lead to considerable increases in emissions in the rest of the world. Van Doorslaer et al. (2015) study different ways of implementing climate polices in the agricultural sector in the EU, and conclude that the leakage effect is dependent on policy design. Lee et al. (2007) use the greenhouse gas version of the US Agricultural Sector Model (ASMGHG) to study the impact on welfare and emissions leakage of implementing climate policies on different geographical scales.

The above-mentioned studies put focus on emissions leakage, which has been raised as a big concern in the agricultural sector. They do not, however, give an answer to how leakage arises due to changes in agricultural policies, and what the consequences are of specific polices under the EU's Common Agricultural Policy. Paper four of the thesis aim to fills this knowledge gap.

2. Method

To answer the research questions, two spatial models are used: a model of biofuel production on agricultural land and transport fuel consumption, which is successively developed in the first three papers of the thesis; and the agricultural sector model CAPRI which is used for the fourth paper.

2.1 Spatial optimization model for biofuel production localization (Paper I)

To be able to answer the research question in the first paper, a spatial cost minimization model for biofuel production is developed. Biofuel technologies are generally most efficient in large facilities and thus exhibits economies of scale (Leduc, 2009). Feedstock for biofuel, in particular second-generation bioenergy crops, has high transport costs as it is distributed over a large area and generally has low energy content per mass unit (Lundmark et al., 2018). This leads to a trade-off between agglomeration forces and dispersion forces that characterizes localization problems, that was first discussed using the von Thünen model approach (see e.g., Wood and Roberts, 2010, pp. 16-19). Due to these forces, and regionally heterogeneous conditions for feedstock production, it is not straightforward to see how cost-effective production of biofuel should be achieved. The model aims to capture these forces, and to locate feedstock production on arable land, biofuel production facilities, and deliveries of biofuel to end-users.

The model covers one country divided into heterogeneous regions. Distances between regions are included to be able to account for spatial relationships such as the trade-off between economies-of-scale of production facilities and transport costs. The model's decision problem is that of a social planner, whose objective is to meet a policy target to increase domestic biofuel production in a least cost way. This is achieved by choosing the optimal number and locations of biofuel production facilities, the quantities of biofuel production at each facility, the location and level of feedstock production, transport flows of feedstock to production facilities, and deliveries of biofuel from facilities to end users. Investment in a biofuel production facility of high or low capacity is a discrete choice, and the investment cost are characterized by economies of scale. Feedstock can be converted to biofuel with a linear technology in the production facility, and the operation costs are assumed to be linearly related to the production. The costs for feedstock are modelled with increasing opportunity costs that arise due to competition over land with other types of agricultural production. Further, there are restrictions on the areas of arable land available to produce feedstock on in each region. The cost for transport of feedstock and biofuel depends on quantities and distance. As the focus is on the potential of domestic biofuel production, imports of biofuel are not included in the model. Emissions are calculated as linearly related to feedstock cultivation, biofuel processing, and transport of feedstock and biofuel, respectively. Further, biofuel is assumed to replace the energy equivalent of gasoline consumption, which lead to a reduction in gasoline emissions. The model is parametrized with Swedish data.

2.2 Spatial optimization model for transport fuel consumption and biofuel production (Paper II)

For the second paper, the biofuel model from the first paper is included as a part in a spatial model for transport fuel consumption. The objective of this model is to minimize the cost of reaching a greenhouse gas emissions reduction target, where the reductions in greenhouse gas emissions can be achieved by decreasing fossil fuel use. To reduce fossil fuel use, one option is to reduce transport fuel consumption. Alternatively, biofuel can be blended into transport fuel and replace fossil fuel. Biofuel production is modelled as in the first paper, but the produced biofuel must now be blended into transport fuel in some region. There is a blend-in cap for biofuel in each region due to technological limits to blending. This restriction creates a spatial relationship between biofuel production and transport fuel consumption.

The cost for transport fuel reductions is measured in reduced consumer welfare. This implicitly includes changes in vehicle kilometres travelled, but also adjustments in the vehicle fleet and changes in transport mode. Empirically this is captured by using long run fuel demand functions that covers all these adjustments implicitly. For biofuel production, the same costs accrue as in the first paper, but the net cost of biofuel are lower due to savings from reduced fossil fuel purchases when fossil fuel is replaced.

2.3 Extension of the spatial optimization model to abandoned agricultural land (Paper III)

Abandoned agricultural land is modelled explicitly in the third paper. It extends the initial area for feedstock production with the area of abandoned agricultural land. Productivity on this land differs from arable land, but the quality of the feedstock is assumed equivalent. Feedstock costs on abandoned agricultural land increase the more abandoned land is taken into production due to increasing costs of converting the land to perennial bioenergy crops plantation. However, as there is no competition for abandoned land, the opportunity costs for arable land are excluded, and therefore total feedstock costs are on average lower than for arable land. Finally, use of abandoned agricultural land is assumed to lead to emissions reductions through carbon sequestration, implying that feedstock from abandoned agricultural land leads to larger net emissions reductions than feedstock from arable land.

2.4 Agricultural sector model CAPRI (Paper IV)

Another spatial model, the agricultural sector model CAPRI, is used for the fourth paper in the thesis. CAPRI is a partial equilibrium simulation model that covers the agricultural sector (Britz and Witzke, 2014). This is also a model with heterogeneous regions, with lower geographical resolution in Sweden than the model depicted above, but covering the whole world. CAPRI does not account for spatial relationships in production, but includes trade between regions. The model simulates land use and herd levels in detail for agricultural activities and at regional level for most European countries (EU+, EU countries plus Norway, the United Kingdom, Turkey and the Balkan countries). This gives results for production levels in EU+, while production outside EU+ is modelled in a simplified way. CAPRI covers trade with regions in the rest of the world, which are aggregated to about forty trade blocks. Trade flows are modelled based on the Armington assumption that products are treated as different based on origin. The demand for agricultural products is modelled with demand functions in each country or trade block. Prices are allowed to change in response to changes in production and demand, and are transmitted via trade.

Similar to the biofuel model developed in the first paper, CAPRI is a comparative static model, meaning that a policy scenario is compared to a baseline scenario. In the baseline scenario, the world is assumed to continue as business as usual, based on current trends. In contrast to the biofuels model, the CAPRI baseline is *calibrated* to follow the projected developments of agricultural production.

CAPRI covers the main agricultural greenhouse gas emissions of methane (CH_4) and nitrous oxide (N_2O). For the EU+ countries the emissions are computed bottom-up based on the production technologies. For the rest of the world, the emissions are modelled by multiplying output quantities with estimated emission intensities.

The CAPRI model is well suited to model consequences of agricultural policy changes as the Common Agricultural Policy in the EU is modelled in detail. In particular, the coupled subsidies to certain production activities (named the Voluntary Coupled Support under the budget periods up until 2023) is added for all member states. Thus, it can be used to simulate the impact of the coupled subsidies on global greenhouse gas emissions such as done in the fourth paper in this thesis. The modelling of trade makes it suitable to analyse leakage problems.

3. Summary of appended articles

3.1 Optimal Localization of Agricultural Biofuel Production Facilities and Feedstock: A Swedish Case Study

Climate and renewable energy targets in the EU could be reached, in part, by increasing the use of biofuels. For this, it would be necessary to increase biofuel production in the EU, or imports. A cost-effective development of biofuel production capacity at such a large scale needs to consider spatial characteristics of the area where the feedstock for biofuel comes from. In particular, the choice of localization of feedstock production and production facilities entails a trade-off between high transportation costs due to bulky feedstock biomass and a biofuel technology characterized by economies of scale.

In the first paper of the thesis, the cost-effective spatial configuration of biofuel production is examined for a range of biofuel production targets in Sweden. Biofuel is produced using perennial bioenergy crops grown on agricultural land. The spatially explicit cost minimization model for biofuel production, which was outlined in section 2.1, is used for this purpose. A national biofuel supply curve is obtained from the results. Marginal costs of biofuel production start at €1,030 per m^3 at the lowest target level, and initially increase slowly with the stringency of the production targets. For a range of low production targets, unit costs for feedstock increase, since larger total feedstock outtake implies larger competition for land. At the same time, unit costs for transport and investment decrease as the average distance from feedstock production to facilities falls, which allows for larger facilities, which are less expensive per unit of biofuel. At higher target levels, the marginal costs of biofuel production increases faster, mainly due to a larger increase in feedstock costs, and less possibilities for low transport and investment costs. For the highest target level, equal to using about 50 per cent of arable land currently used for ley production for the purpose of feedstock cultivation, the marginal cost is €1,420 per m³.

The feedstock uptake is centred to areas where the highest feedstock costs can be avoided, but where there at the same time is a high density of agricultural land (see Figure 2). This implies that production facilities and feedstock catchment areas are in the south at low target levels. The catchment areas increase with the target levels to cover the whole of Sweden, but initially only using a small share of the agricultural land available for feedstock production in each region, to avoid more expensive feedstock. At the highest target levels, a larger outtake of feedstock from most regions is needed despite high feedstock costs. The results show that the feedstock costs are of great importance for the location, and more important than transport costs. There are more high- than low-capacity facilities, which shows that the lower investment cost per unit at these facilities are important.

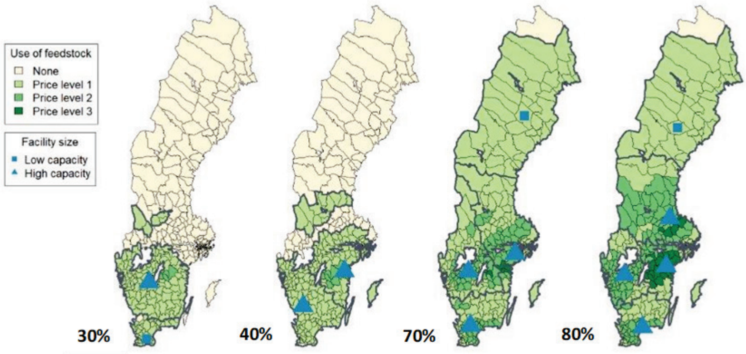

Figure 2 Location of biofuel production facilities and feedstock catchment areas. Triangles symbolise facilities of high capacity and squares low capacity. Shaded areas surrounded by borders denote feedstock catchment areas of each facility. Darker areas indicate larger uptake of feedstock. The percentages denote stringency of target levels for each scenario.

In the main scenarios, the biofuel is distributed to regions for consumption following road transport fuel consumption data. However, road transport might be electrified in the long term, and use of biofuel shifted to shipping and aviation. Two alternative scenarios show that if biofuel is distributed as airport fuel demand and as fuel demand in harbours, respectively, the feedstock uptake and facility localization are similar to those in the main scenario. There is an increase in the size of facilities close to large ports and airports, and feedstock catchment areas are shifted to the north. Total costs for these scenarios are slightly larger, due to increased biofuel delivery costs.

Another concern is that policy targets tend to be strengthened over time. In the case of biofuel production it could lead to sequential investments in production capacity, and could risk incurring additional costs. A scenario where capacity is first built for an intermediate target and then extended shows that sequential implementation of targets results in more and smaller facilities. Feedstock catchment areas remain quite similar, and costs are only about one per cent higher. The relatively small difference in costs can be explained by the feedstock costs being the main cost, and these remain almost the same.

The potential of the production to decrease fossil fuel is limited – six per cent of total liquid transport fuel use in 2018, and therefore there is limited potential to decrease fossil emissions in the transport sector. Implementing high production targets were found to imply large marginal abatement costs – up to €0.53 per kg CO_2, which can be compared to the Swedish tax on fossil fuel - €0.12 per kg CO_2. Another impact is that on fodder production. A high target was found to decrease the number of hectares of ley per grazing animal by as much as 30 per cent in the east of Sweden.

3.2 Cost-effective reductions in greenhouse gas emissions: reducing fuel consumption or replacing fossil fuels with biofuels

The second paper of the thesis focuses on cost-effective greenhouse gas emissions abatement choices in the transport sector, and in particular the role of second generation biofuel from agricultural land. Greenhouse gas emissions from the transport sector can be reduced be replacing fossil fuels by biofuels through blending in the current vehicle fleet, or by reducing fuel use. The latter can be achieved by reducing transports, shifting to more fuel-efficient vehicles, and by changing transport mode. The cost-effective combination of these measures is determined by spatially varying characteristics of fuel demand, feedstock production costs, greenhouse gas emissions from feedstock production, and possibilities to blend biofuel into fossil fuel. These choices are included in a spatial transport fuel consumption and biofuel localisation optimisation model (see section 2.2), where the model from the first paper is included to model biofuel production endogenously. The objective of the model is to reduce greenhouse gas emissions in the transport sector at least cost.

The model is used to analyse two main sets of scenarios with increasing stringency of emissions reduction targets. The first set has two abatement options: biofuel can be used for blending, or transport fuel consumption can be reduced. In the second set the only abatement option is to reduce transport fuel consumption. The results show that biofuel is cost-effective to use for emissions abatement. The marginal abatement costs for emissions reduction are up to 45 per cent lower per tonne CO_2 when biofuel replacement is an option (see Figure 3). A large share of the emissions reduction can be attributed to biofuel replacement for low and modestly stringent reduction targets. For more stringent target levels, the reduction of transport fuel consumption becomes increasingly important for emissions reduction. The blend-in restriction decreases the marginal gains of having biofuel as an abatement option at higher reduction targets, as there is too little gasoline to blend the biofuel into. Investments in vehicles using pure biofuels would be needed for larger employment of biofuel. The results imply that domestically produced biofuel can be important for climate policies for the transport sector. However, compared to greenhouse gas emissions abatement in other sectors, the marginal abatement costs are relatively high (High-Level Commission on Carbon Prices, 2017).

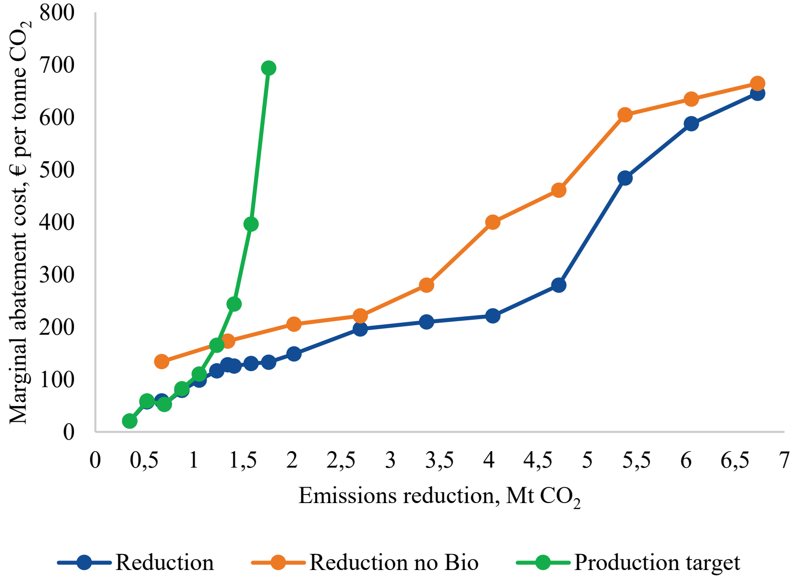

Figure 3 Marginal abatement cost in € per tonne CO_2 at different emissions reduction levels for scenarios with different abatement measures available. Blue line: Biofuel replacement and reduction in fuel consumption. Red line: Only reduction in fuel consumption. Yellow line: Only biofuel replacement, based on a production target.

An economic incentive structure could be constructed to realise the biofuel production suggested by the results. While a carbon price on all emissions, including biofuel emissions, could be efficient, this could be difficult to communicate to stakeholders. Instead, a policy with compensation to feedstock and biofuel producers could be set up, complemented with a carbon tax on end fuel use. Marginal costs of the optimal feedstock and production indicate the cost-efficient subsidies across space. The highest subsidies for feedstock producers would be around production facilities, and in the south. These areas should receive high subsidies as they are important to include in the production, despite high feedstock costs, to arrive at the optimal spatial configuration of biofuel production. Moreover, the subsidies for feedstock would be higher for higher targets. For production facilities, the subsidies should generally be lower at higher target levels, since there are more high capacity facilities which have lower investment costs at high target levels.

The targets for biofuel are often formulated in terms of biofuel quantities, even if emissions reductions is the intended aim. With a production target, only biofuel replacement contributes to emissions abatement. The results show that this makes the marginal abatement cost very high at large abatement levels, as the most expensive feedstock is used. For an emissions target, the less expensive choice of reducing fuel consumption is also used. However, results show that given that the optimal biofuel production level for a given emissions reduction level is known, a production based target can lead to a cost effective spatial organization of biofuel production. The reason is that the spatial differences in biofuel production related emissions have little impact on the location.

3.3 Cost-effective use of abandoned agricultural land for biofuel production

There is a risk for competition with food production when agricultural land is used to produce feedstock for biofuel production (Jeswani et al., 2020). As an alternative, using abandoned agricultural land has been proposed, since there is no current food production (Valentine et al., 2012). The third paper of the thesis examines whether using abandoned agricultural land for perennial bioenergy crop feedstock production also can decrease the costs for reducing greenhouse gas emissions in the transport sector. Further, it examines how different attributes of abandoned agricultural land contribute to the results. First, when abandoned agricultural land is included as a distinct type of agricultural land that can be used to grow feedstock on, the total available area for feedstock production increases. It also gives new possibilities to organize biofuel production, as the spatial configuration can be changed. Second, abandoned agricultural land has generally low productivity, but has low opportunity costs from competition over land, and therefore feedstock costs for perennial bioenergy crops on abandoned agricultural land can be lower than on arable land. Third, cultivation of perennial bioenergy crops leads to carbon sequestration on abandoned cropland, while the corresponding effect on arable land is smaller or negligible, due to risk of indirect land use change emissions.

In this study, the spatial optimization model from the two first studies is extended with explicit modelling of abandoned agricultural land (see section 2.3). The results of the paper show that abandoned agricultural land could reduce costs of greenhouse gas emissions abatement substantially. The differences in costs compared to the case without abandoned agricultural land increases with the stringency of the target level. This happens even though all abandoned agricultural land is used already at low target levels. The reason is that feedstock transport can be better organized spatially when abandoned agricultural land is considered. A larger share of emissions reductions is attributable to biofuel that replaces fossil fuels when abandoned agricultural land is included. However, the total quantity of biofuel could be larger or smaller with abandoned agricultural land included, depending on the target level and the trade-off between different underlying mechanisms.

Carbon sequestration on abandoned agricultural land is shown to be the main driver of the positive results. With carbon sequestration, a smaller volume of biofuel can result in the same, or even larger greenhouse gas emissions reduction than without abandoned agricultural land. Conversely, the expansion of land for feedstock production *per se*, and the lower feedstock costs are of less importance. However, low feedstock costs and expansion of land make the biofuel volume increase relative the case without abandoned agricultural land. The results show that there are mechanisms that can make costs for emissions reduction decrease both by reducing, and by increasing biofuel production. Thus, the results imply that abandoned agricultural land should be used for biofuel production, but it is uncertain how large the total biofuel production levels should be.

3.4 Coupled Agricultural Subsidies in the EU Undermine Climate Efforts

The fourth paper of the thesis investigates the impact of an agricultural policy on greenhouse gas emissions, including both local and global impacts. A decrease in production and related emission in one region, for example caused by a climate policy, could cause increases in production and emissions in another country, thereby diminishing the intended net decrease in emissions or even reverse it. This is called emissions leakage (Markusen, 1975; Zhang, 2012).

It is debated if the Common Agricultural Policy (CAP) in the EU is beneficial or damaging for the climate, as there is a risk for emissions leakage. The largest share of the CAP measures is the per-hectare based direct payments in the Basic Payment Scheme to agricultural land. Specific subsidies for environmental measures and rural development receives a smaller share of the total budget. In the fourth paper of the thesis, the impact on global greenhouse gas emissions of removing coupled subsidies to animal production under the CAP, is studied. During the budget period of the study the subsidies were named the Voluntary Coupled Support, which, with small amendments are kept during the current budget period, renamed to Coupled Income Support (European Commission, 2023b). The coupled subsidies could use 14 percent of the CAP budget at the time of the study, and currently 13 percent (European Commission, 2023b). Member states can chose to provide the subsidies to different agricultural activities. In practice, most member states use it to support ruminant production: beef and dairy cattle, sheep and goat (European Commission, 2022). This is of specific interest in a climate perspective as these activities are also the most emitting, and the coupled subsidies could therefore increase production and agricultural emissions in the EU. However, this might in turn lead to increased production and emissions outside the EU. As emissions intensities for agricultural products differ greatly across regions and products, with in general higher emission intensities outside the EU, the net impact on global emissions of the coupled subsidies is not obvious.

The agricultural sector model CAPRI is used for the paper (see section 2.4). CAPRI is well suited to model consequences of agricultural policy changes as the CAP in the EU is modelled in detail. In particular, the coupled subsidies are added for all member states. The modelling of trade in agricultural products makes the model particularly well suited to study emissions leakage. Greenhouse gas emissions in the EU countries plus Norway, the United Kingdom, Turkey and the Balkan countries are calculated bottom up based on production technologies. For the less detailed production in the rest of the world, new trends of emissions intensities are estimated for this paper, for all commodities covered in CAPRI. The trends are estimated to match total agricultural emissions reported in FAOSTAT greenhouse gas emissions inventories as closely as possible over time, given trends of agricultural production quantities. The trends are estimated in a Bayesian estimation framework, where prior distributions for the emission intensities are included, to improve the robustness of the results, given the many variables.

The simulated emissions of two scenarios are compared: one baseline scenario where the CAP, including the coupled subsidies, is implemented, and one scenario where the coupled subsidies to ruminants are removed from the CAP. The budget for the coupled subsidies is reallocated to the Basic

Payment Scheme in the respective member states. The results show that removing the coupled subsidies reduce greenhouse gas emissions in the EU by 2,354 kt CO_2 equivalents annually, corresponding to −0.5 percent of total agricultural greenhouse gas emissions in the EU. However, emissions increase in the rest of the world (ROW) by 1,738 kt CO_2 equivalents, hence causing about 75 percent emissions leakage, as shown in the left bar of Figure 4. The net impact of removing the policy is a net decrease globally by 616 kt CO_2 equivalents.

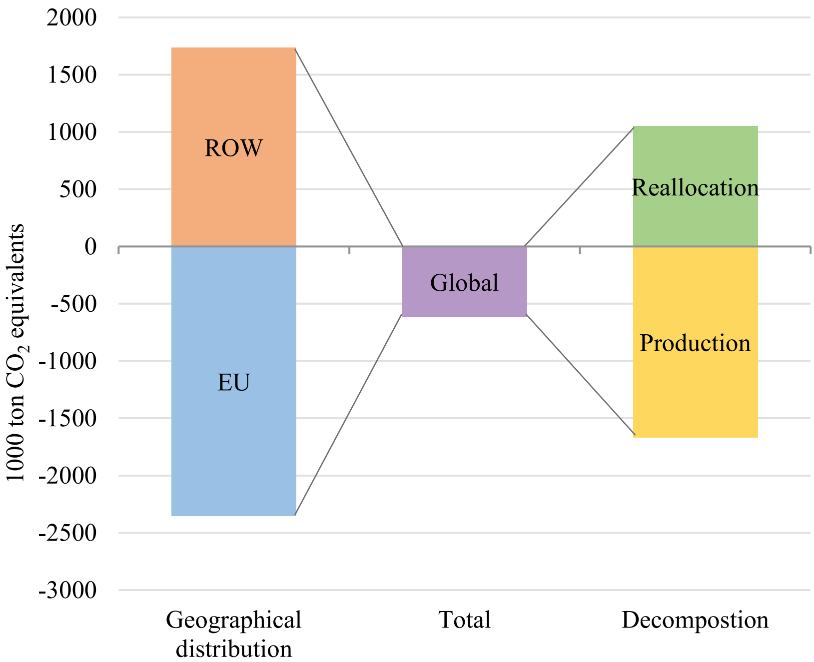

Figure 4 Global changes in greenhouse gas emissions in 1000 CO_2 equivalents annually. The geographical distribution indicates how the total global emissions changes are divided into changes in the EU, and in the rest of the world (ROW), respectively. The decomposition shows how the total global emissions changes are decomposed into those caused by production changes and those caused by differences in emission intensities in the producing countries.

The main part of the emissions reduction in the EU stems from a 1.1 percent decrease in beef production. For the dairy sector, the reduction in production, and thus emissions, is much smaller. The difference is explained by the importance of the coupled subsidies for the sectors: for beef producers the coupled subsidies constitute a much larger share of income than they do for dairy farmers. Some of the reduction in beef production is replaced by more pork and poultry production. This decreases net emissions, as pork and poultry have much lower emission intensities than beef. The decrease in production in regions with coupled subsidies partly shifts to EU countries that have not implemented the coupled subsidies, such as Germany, with a varying impact on emissions. There is also a large increase in beef production outside the EU, which compensates decreased exports and increased imports of beef to the EU. In addition, trade patterns outside the EU change, which for example increases beef production in Brazil. This leads to increases in emissions as beef production is relatively emission intensive in Brazil. Crop production changes too, but has little impact on emissions. The reduction in EU sheep and goat meat production results in a net increase in emissions globally, while the decrease in dairy production and emissions in the EU results in a decrease in emissions also outside the EU. This indicates that production subsidies can have very diverse impacts on emissions, depending on the trade relations and differences in emission intensities.

The changes in global emissions due to removing the coupled subsidies are also decomposed into changes caused by production changes, and changes caused by reallocation of production. The first category, called the production effect, includes the effect of decreasing global production, but assumes that there are no differences in emissions intensities at the new production locations. The production effect implies a decrease in global emissions by 1,666 kt CO_2 equivalents, which is shown in the right bar of Figure 4. The latter category is the additional change in emissions that arise as the production has be reallocated to regions with other emission intensities: an *increase* in global emissions by 1,050 kt CO_2 equivalents.

The overall welfare impact of removing the subsidies is positive, disregarding monetization of the value of emissions reduction. This means that this policy measure, removing the coupled subsidies, implies a net economic *gain* per tonne avoided CO_2 equivalents. However, other benefits of keeping the supported sectors are not included.

4. Concluding discussion

The contribution of this thesis is discussed in the next section, and policy recommendations are presented in the subsequent section. Limitations, and ideas for future research are discussed in the last section.

4.1 Contribution

The main methodological contribution of this thesis is the development a new model for cost-effective localization of biofuel production. The model locates multiple biofuel production facilities, and primarily distinguishes itself from other models by the regionally differentiated increasing opportunity costs of agricultural land, motivated by competition with alternative uses. This makes the modelling more realistic for large increases in biofuel production. The application to Sweden is new, and of interest as it represents a region that is to a varying degree dominated by forests, for example similar to Latvia and Finland, but different from the Netherlands and Italy. There are also large spatial variations in the agricultural landscape and climate across the country. The results from the first paper contributes with a quantification of a supply curve for biofuel based on perennial bioenergy crops from agricultural land in Sweden. The trade-off between agglomeration and dispersion forces are important for the results, while this is overlooked in many earlier studies applied in the agricultural context.

The next methodological contribution is the extension of the biofuel model in the second paper to incorporate biofuel production in a model over fuel consumption choice. Whereas other models include the choice between different greenhouse gas emissions abatement measures, this model considers the spatial configuration of biofuel production and fuel consumption as well as the restriction posed by the blending rate of biofuel into fossil fuel. The results show in detail how the technological restriction can influence the abatement potential in the transport sector, at stringent emissions reduction targets. The results also show that, given these characteristics of the transport sector, greenhouse gas emissions abatement costs can be significantly decreased with biofuel.

The third paper contributes by providing insights on how abandoned agricultural land can contribute to cost-effective emission abatement. It shows that carbon sequestration dominates the positive impact, while the lower feedstock costs have some impact, and abandoned agricultural land improves the conditions for an efficient spatial configuration of biofuel production, thereby reducing costs. The joint impact shows that abandoned agricultural land is valuable mainly due to the increased emissions abatement. These results provide another rationale for using abandoned agricultural land than to decrease competition for agricultural land.

The results from the three studies indicate that the possibility for carbon sequestration on abandoned agricultural land dominates the impact of biofuel on greenhouse gas emissions abatement. Spatially heterogeneous feedstock costs are second most important, while transport and investment costs are of less importance for total costs but have an impact on the spatial distribution of high and low-capacity production facilities.

The fourth paper contributes with a quantification of the greenhouse gas emissions leakage rate and the absolute global greenhouse gas emissions reduction from removing the coupled subsidies to ruminants in the EU's Common Agricultural Policy. In more general terms, it contributes by showing how emissions leakage arises from subsidies to a polluting industry. It shows the specific impact on emissions leakage from the high diversity of emission intensities that characterizes the agricultural sector. The contribution of the newly updated emissions intensities for agricultural products are highlighted by the decomposition of emissions. This shows that a reallocation of production to regions with higher emissions intensities have a large impact on global emissions.

4.2 Policy implications

The results from this thesis has several policy implications. For the first paper, the focus is on the costs of biofuel production. The results show that using feedstock with low feedstock costs is most important to minimize costs, which implies that areas with low competition for land with food production should be given most attention for feedstock production. For the localization of production facilities, it is also important to minimize transport costs, resulting in optimal locations centred in feedstock dense areas. In areas of low agricultural density, transport costs would be very high for high capacity facilities. However, if the feedstock costs are low enough the more expensive low capacity facilities could be considered in these areas, as these facilities would imply lower transport costs.

Even though stepwise increases in production targets are shown to lead to a suboptimal choice and localization of production facilities, the impact on total costs is relatively small. Therefore, a possibility that biofuel polices is implemented sequentially might not be worrisome. The reason for this is that the cost for feedstock would not change much, and is more important than transport and investment costs. Similarly, alternative spatial distributions of biofuel demand has little impact on total costs and facility localization. This implies that it could be sufficient to plan for road transport consumption, even if the biofuel consumption could eventually change to be used at airports for aviation or in harbours for shipping.

The estimated costs for biofuel production are currently not competitive with forecasted gasoline or ethanol prices. However, this could change when investment costs or operational costs for biofuel production facilities decrease in the future, and/or if climate policies become more stringent globally, which could cause higher world market prices for both fossil fuels and biofuels in the future.

The second paper shows that biofuel can be a cost-effective greenhouse gas emissions abatement measure in the transport sector, despite the relatively high costs shown in the first paper. However, several factors limit the potential of biofuel to reduce costs for emissions reductions at large scale emissions reduction. Limited areas of arable land for feedstock production and increasing feedstock costs implies large feedstock production becomes costly. In addition, the fact that there is a technical restriction on how much biofuel can be blended into fossil fuel, implies that the large reduction of fossil fuel at high reduction targets limits the amount of biofuel that can be used. Therefore, other abatement measures are needed at higher target levels.

The estimated marginal abatement costs associated with abatement in the transport sector for most of the studied emissions reduction targets are higher than, e.g., the carbon price suggested by The High-Level Commission on

Carbon Prices (2017) and suggests that the role of the transport sector should be limited. The marginal abatement costs in the studies in this thesis are higher than the current Swedish tax on greenhouse gas emissions from fossil fuels. However, they are lower than marginal abatement costs for some other Swedish policies for transport fuel substitution (NIER, 2017).

The results from the third paper show that abandoned agricultural land should be used for cost effective greenhouse gas abatement. A high carbon sequestration rate was the main driver behind the result. The importance of carbon sequestration also indicates that the use of detailed local information on sequestration rate would be valuable to choose the best areas of abandoned agricultural land. An obstacle for using abandoned agricultural land is that there are no premium entitlements for reintroduced abandoned agricultural land under the Common Agricultural Policy in the EU. The policy would need to be changed to ease the use of abandoned agricultural land for environmentally motivated purposes.

The second paper shows that a production-based policy and an emissions-based policy both imply a similar localization of production facilities and feedstock catchment areas. However, when abandoned agricultural land is included, with highly diverging implications for emissions, redirecting the localization of feedstock production from arable land to the abandoned agricultural land becomes very important.

A widely raised concern about biofuel is its impact on food production. The results shows that large scale biofuel production can have a large impact on livestock production, as it leads to reduction in fodder production, in particular in the eastern part of Sweden. The results in the third paper show that when abandoned agricultural land can be used for biofuel production, the use of arable land is reduced. However, the results also show that use of abandoned agricultural land implies that biofuel production for emissions abatement becomes more cost-effective. This can lead to an increase in biofuel production, and even in an increase in the use of arable land for biofuel production. Therefore, the impact on livestock production is not obvious when abandoned land is used.

The fourth paper shows that removing the coupled subsidies to ruminants under the EU's Common Agricultural Policy could be justified due to its effectiveness as a climate policy. Removing it would decrease global agricultural greenhouse gas emissions and increase welfare from the agricultural sector. However, the impact on total EU agricultural emissions is low, corresponding to 0.5 percent of total EU agricultural emissions. The removal of the coupled subsidies is therefore not a crucial policy, and if other benefits are large, they could outweigh the benefit for the climate. Removing the coupled subsidies results in a high rate of emissions leakage, which is mainly due to a reallocation of production to regions with high emissions intensities. This implies that a policy to decrease the leakage should aim at reducing these differences, or at avoiding products from high emitting regions.

4.3 Future research

This thesis covers some questions regarding biofuel production and its use as a greenhouse gas emissions abatement measure, and emissions leakage from agricultural policies. The papers come with some limitations and leave open questions to be handled in future research. One limitation of the studies concerning biofuels it the level of detail of the data. With multiple types of feedstock, production technologies, and feedstock transport modes, the quantified results could become more accurate. This would also give the possibility to investigate the importance of these choices, relative to opportunity costs for feedstock which were important for the results in the thesis. As these studies only cover domestically produced biofuel from agriculture, they miss possible competition or synergies from supply of biofuel from the forest sector. A spatial analysis where biofuel production from forest biomass is represented could show to what degree they affect each other.

The opportunity costs for feedstock production were modelled based on own price elasticities. As they are point estimates, this could be an issue for large scale use of agricultural land for feedstock production. In future studies, the competition for land could be modelled with explicit representation of other agricultural activities, for example with a sector model. This would also give the possibility to study the impact on agricultural land use and food production, which was calculated in a simple fashion in this thesis. The approach could also be used to investigate if opportunity costs would arise on abandoned agricultural land, which could happen if the demand for land becomes high enough. In the third paper, the opportunity cost for abandoned agricultural land was assumed to be zero, an assumption which might be too strong. Market power could pose a problem when there are a few large production facilities, in particular in light of the importance of feedstock prices. This is not considered in this thesis but could be modelled in a game theoretical framework for investors in biofuel production facilities or feedstock producers.

The modelling of emissions abatement through reduced transport fuel consumption is limited by the fact that it builds on demand elasticities for transport fuel consumption calculated based on historical data. The transport sector is facing many changes, e.g., rapid increases in electric vehicles. The costs of shifting to electric vehicles is to some extent reflected in the demand for fossil fuels, but to study long run changes in demand, technology shifts would be important to incorporate in a more explicit manner.

Possible biofuel imports could be included in the model to examine the potential of the domestic production in a global perspective. This should include the possibility for global biofuel prices to increase considering global changes in policies. Further, the omission of land use changes in other countries due to biofuel expansion was identified as an important cause of uncertainty in the third paper of the thesis. Including the impact on global changes in land use and agricultural production would make it possible to quantify the net increase or decrease in global carbon sequestration due land use changes, stemming from increased biofuel production in Sweden. This could also be used to study the trade-off of using land for biofuel or food production, which could be a serious concern (Searchinger et al., 2008).

The thesis focuses on greenhouse gas emissions, while there are other impacts of biofuel production and agricultural production. Future research could therefore examine the impact on, e.g., biodiversity caused by implementing the biofuel production quantities suggested in the thesis. Likewise, this could be analysed for removing the coupled subsidies to ruminants. Alternatively, the optimal localization of biofuel production given other environmental targets could be assessed.

The problem of emissions leakage in the fourth paper suggests that future research should examine polices that could decrease the degree of leakage. These could be policies that aim at avoiding production in regions with high emissions intensities, for example border carbon mechanisms. It could also be policies that try to shift emissions intensities endogenously, which would require modelling of non-EU emissions.

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Popular science summary

This thesis investigates how to reduce greenhouse gas emissions at least costs, by changing production on agricultural land. The main part of the thesis focuses on biofuels, such as ethanol, produced by biomass from perennial energy grasses grown on agricultural land. One advantage with energy grasses is that they compete less with food production over agricultural land than traditional energy crops. Advanced biofuel technologies are needed to process the energy grass to biofuels. Most of these technologies are more costly than traditional technologies. The biofuels can replace fossil transport fuels in cars, ships, and airplanes, and thereby reduce greenhouse gas emissions.

Regional variations in energy grass yield, costs for production, and costs for agricultural land, among other things, determine the suitability of regions for energy grass production. The costs for producing biofuels depend on costs for transporting the energy grass to biofuel production facilities. Large production facilities are more efficient, but require biomass from a large area. This implies large distances between the fields where the feedstock is produced and the production facilities, which increases transport costs. This regional perspective is crucial in understanding how biofuel can be used to reduce greenhouse gas emissions.

The first study examines how biofuel should be produced to meet national targets for biofuel production volumes at least costs. An economic model for biofuel production, divided into the 290 municipalities of Sweden, is developed. The model determines how biofuel production can be organized in Sweden at least cost. The results show how up to four biofuel production facilities with high production capacity, and one with low production capacity, are distributed in the country. The production facilities and the biomass production should be located where the costs for biomass are low, and where agricultural land is abundant. At low production volumes, production should be located in the south of Sweden, but at higher volumes parts of the agricultural land in the whole country should be used. The overall cost for producing the biofuel increases rapidly with the biofuel volume.

In the second study, the focus is on reducing greenhouse gas emissions in the transport sector. The biofuel model is extended to model two options to reduce emissions: reductions in transport fuel use, and replacement of fossil fuels by blending them with biofuel. Both lead to a reduction in greenhouse gas emissions from fossil fuels, but the costs differ. Reduced transport fuel use can for example be achieved by travelling less, and this is negative for the consumers, implying a cost to society. The cost for producing biofuel depend on biomass costs, operations costs, transport costs, investment costs, and costs for distributing the biofuel. The results show that both options should be used to minimize the costs of reducing emissions, and that applying biofuel blending reduces costs relative to only reducing fuel use.

Abandoned agricultural land has been proposed to be used for biofuel production, as using this land will not create competition with food production. The third study investigates the potential of abandoned agricultural land to also lower the costs for reducing greenhouse gas emissions. The reduction in costs is substantial, mainly due to large uptake of carbon dioxide, so called carbon sequestration, when perennial energy grasses are grown on abandoned agricultural land.

In the fourth study, the focus is shifted to the emissions in the agricultural sector. There are subsidies in the EU's Common Agricultural Policy is primarily directed to cattle, dairy, sheep and goat production. Since these farm animals are the largest sources of greenhouse gas emissions in the agricultural sector, the subsidies might increase the emissions. An agricultural sector model is used to examine what would happen if the subsidies were removed. Results show that removing the subsidies decreases EU production and greenhouse gas emissions. However, there is a so-called emissions leakage by 75 per cent - when one tonne of greenhouse gases is removed in the EU, emission increase by 0.75 tonne outside the EU. The reason is that when production decrease in the EU, production increase somewhat outside the EU to meet the demand for these products. This leads to an increase in emissions outside the EU. The increase in emissions is large as the new producing regions emits more greenhouse gases per kilo of product than in the EU.

Populärvetenskaplig sammanfattning

Den här avhandlingen granskar hur växthusgasutsläpp kan minskas till så låg kostnad som möjligt genom användning av jordbruksmark. Huvuddelen av avhandlingen handlar om biobränslen, såsom etanol, framställda av biomassa från perenna energigräs som odlas på jordbruksmark. En fördel med energigräsen är att de konkurrerar mindre med matproduktion om jordbruksmarken än traditionella bioenergigrödor. För att omvandla energigräs till biobränslen krävs så kallade avancerade biobränsleteknologier. De flesta av dessa teknologier är dyrare än traditionella teknologier. Biobränslen kan ersätta fossila bränslen i bilar, fartyg och flygplan, vilket minskar växthusgasutsläppen.

Regionala variationer i bland annat skördenivåer för energigräs, produktionskostnader och markkostnader påverkar var det är mest lämpligt att producera energigräs. Kostnaden för biobränsleproduktion beror av kostnader för att transportera biomassa till biobränslefabriker. Biobränslefabrikerna är mest effektiva om de är stora, men kräver stora markområden för att leverera de volymer av biomassa som behövs. Det leder till stora avstånd mellan biomassaproduktion och fabrik, med höga transportkostnader som följd. Detta regionala perspektiv är viktigt för att förstå hur biobränsle kan användas för att minska växthusgasutsläppen.

I den första studien undersöks hur biobränsle bör produceras för att nå nationella mål för biobränsleproduktion till lägsta kostnad. En ekonomisk modell för biobränsleproduktion i Sverige, indelad i 290 kommuner, utvecklas för att besvara frågeställningen. Modellen kan beräkna hur biobränsleproduktion ska organiseras i landet till lägsta möjliga kostnad. Resultaten visar hur upp till fyra stora biobränslefabriker, och en mindre fabrik, fördelas i landet. Biobränslefabrikerna och biomassaproduktionen bör placeras i regioner där kostnaderna för biomassa är låg, och där andelen

jordbruksmark är hög. Vid de lägre produktionsmålen betyder det att produktionen bör förläggas till södra Sverige, men för högre mål bör delar av jordbruksmarken i hela landet användas. Den totala kostnaden för att producera biobränsle ökar snabbt för ökande produktionsvolymer.
Den andra studien fokuserar på att minska växthusgasutsläpp inom transportsektorn. Biobränslemodellen utvidgas till att modellera två strategier att minska utsläppen: minska bränsleanvändning i transporter och ersätta fossila bränslen genom att blanda in biobränslen. Båda alternativen leder till minskade utsläpp från fossila bränslen, men med olika kostnader. Att minska bränsleanvändningen, till exempel genom att resa mindre, medför en samhällsekonomisk kostnad för konsumenten. Kostnaden för att producera biobränslen beror på kostnader för biomassa, investeringskostnader, driftskostnader, och distributionskostnader. Resultaten visar att båda strategierna bör användas för att minimera den totala kostnaden för att minska växthusgasutsläpp. Dessutom visar resultaten att inblandning av biobränslen minskar den totala kostnaden jämfört med att bara minska bränsleanvändningen.
Övergiven åkermark föreslås användas för biomassaproduktion till biobränslen för att undvika konkurrens med livsmedelsproduktion. Den tredje studien analyserar vilken potential den övergivna åkermarken har att även minska kostnaderna för att minska växthusgasutsläppen. Kostnadsminskningen är påtaglig, framförallt på grund av ökat koldioxidupptag och -lagring när perenna energigräs odlas på denna mark.

I den fjärde studien skiftar fokus till växthusgasutsläpp från jordbrukssektorn. I EU:s gemensamma jordbrukspolitik finns subventioner till produktion av nötkött, mjölk, får och get. Eftersom dessa husdjur utgör de största källorna till växthusgasutsläpp inom jordbrukssektorn så kan subventionen leda till att utsläppen ökar. En jordbruksmodell används för att undersöka vad som händer om subventionerna tas bort. Det visar sig att EU:s produktion och utsläpp minskar om subventionerna tas bort. Globalt blir det dock ett så kallat utsläppsläckage på 75 % - när ett ton växthusgaser tas bort i EU så ökar utsläppen med 0,75 ton utanför EU. Anledningen till detta är att när produktionen minskar i EU så ökar produktionen utanför EU något för att kunna möta efterfrågan på dessa produkter. Det gör att utsläppen ökar utanför EU. Ökningen i utsläpp är stor eftersom de nya producentregionerna släpper ut mer växthusgaser per kilo produkt än vad EU gör.

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I

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Optimal localisation of agricultural biofuel production facilities and feedstock: a Swedish case study

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Abstract

Policies for investment in biofuel production facilities and feedstock may be necessary in order to meet climate and renewable energy targets. These policies entail a trade-off between high transportation costs of biomass and economies of scale of production facilities. We develop a spatial optimisation model and investigate the cost-effective localization of production facilities for ethanol from agricultural land in Sweden. Feedstock costs are found to be most important in determining the location, although high feedstock density motivates locating large facilities in areas with high feedstock costs. At higher production, feedstock from the whole country is preferred despite high transport costs.

Keywords: Agricultural land use, biofuel, climate policy, cost-effectiveness, localisation, spatial optimisation

JEL classification: Q10, Q18, Q20, Q42, Q54

1. Introduction

The substitution of agricultural biofuels for fossil fuels has been suggested as a tool to reduce the impact of the transport sector on greenhouse gas emissions. To help achieve this, production of biofuels on agricultural land could be increased (Creutzig *et al.*, 2015). For increased production, investments in production facilities and increased feedstock uptake are needed. However, the introduction of biofuel production on agricultural land is associated with two

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major challenges relating to policy design: (i) the choice of location for production facilities and (ii) the competition between feedstock production and food production. Moreover, the biofuel supply chain is characterised by a technology which is generally most efficient when working at large quantities and thus exhibits economies of scale (Leduc, 2009). Costs could therefore be saved by concentrating production to a few facilities. The raw material, which mostly comes from forest or agricultural land, is distributed over a large area and is generally low in energy content, with potentially high transport costs as a result (Lundmark *et al.*, 2018). In particular, this is true if there are only a few, large production facilities in a given area. Therefore, the relation between transportation costs and facility investment costs is important (Yue, You and Snyder, 2014), and the choice of facility location and capacity becomes a first important step towards a cost-efficient production system. Large-scale biofuel production would compete with other types of production on agricultural land, although there is some room for using by-products from agriculture such as straw and food waste (Prade *et al.*, 2017).

The European Union (EU) has developed targets for renewable energy as a means to reach climate targets, set out in the Renewable Energy Directive II (RED II). The common EU-wide target is to reach 32 per cent renewable energy as a share of final energy consumption by 2030. However, targets differ between countries with different starting points and possibilities to increase renewable energy use. With transport accounting for a large share of fossil fuel use, a specific part of the target is that at least 14 per cent of transport fuels should come from renewable sources by 2030 (European Parliament, 2018). Due to concern over competition with food production, there are restrictions in RED II limiting the possibilities for crediting biofuel production against the targets for renewables. These limitations concern the use of agricultural land otherwise used for food and feed production and contexts where biofuel production negatively affects other sustainability factors such as biodiversity (European Parliament, 2018; European Commission, 2020). At the same time, biofuels for transport are supported by policies: in the EU mostly through biofuel quotas and mandatory blending but also with financial measures such as tax exemptions. Investment support directed towards production facilities is also provided in many cases but has been more common in the bioenergy heating sector (Banja *et al.*, 2019).

In line with the EU's aims, biofuel use in the transport sector increased from 1 per cent of total fuel consumption in 2005 to almost 6 per cent in 2018, but with a large regional variation (Eurostat, 2020a). Globally, the share of biofuels in the transport sector is relatively small and equalled 3 per cent of energy use in 2017 (World Bioenergy Association, 2020). To reach the current targets, biofuel production in the EU, or imports, must increase. There is therefore a need to understand how a policy for increased domestic biofuel production should be designed when both the scale of production and the location of production facilities and feedstock cultivation are still an open question, as well as the associated costs.

The purpose of this study is to identify a cost-effective supply chain of biofuel from agricultural land, given a policy target for biofuel production. In particular, we examine the trade-offs between economies-of-scale benefits of production facilities and feedstock production and transport costs. We do this using a spatial model that optimises the placement of investment in biofuel production facilities and the location of feedstock production in Sweden. The model is used to investigate how localisation choices are affected by the stringency of a hypothetical biofuel production target and how those choices are influenced by the geographical distribution of fuel demand. With this model setting, we derive the optimal organisation of supply for different production levels, thus obtaining a national supply curve. We compare estimated marginal costs of domestic production to projected world market ethanol prices and examine the cost-effectiveness of the resulting greenhouse gas emission reductions and consequences for the regional production of fodder and hence for livestock production.

The field of location studies is broad, where non-economic location studies often strive to distribute a given number of facilities in space by minimising the distance to demand sites, e.g. Comber *et al.* (2015). The best sites for location can also be selected by a detailed suitability analysis, as in Sharma, Birrell and Miguez (2017) that locates sites using a GIS suitability analysis.

Many economic studies on optimal localisation of biomass and biofuel facilities have been published in recent years, of which most focus on long-term strategic decisions (Zandi Atashbar, Labadie and Prins, 2018). Some studies analyse decisions concerning a single production facility. Lankoski and Ollikainen (2008) investigate the optimal use of land for feedstock around a given facility, using a von Thünen model. Rentizelas and Tatsiopoulos (2010) consider the best location and design of a single facility in an area, given available feedstock, while Rozakis *et al.* (2013) account for endogeneity of feedstock supply using a sector model.

Several economic studies have analysed policies over larger regions, which requires consideration of multiple production facilities. Such studies often minimise the cost of the biofuel supply chain, by identification of both the optimal location and the optimal number of facilities. The system boundary can be the biofuel system or, as in De Jong *et al.* (2017), the whole forestry sector. Most of the studies are static; however, there are some exceptions, such as Santibañez-Aguilar *et al.* (2015) who apply a dynamic setting for facility investment and production taking seasonality in feedstock supply into account. Some studies analyse biofuel produced with feedstock from the forest sector while others apply their analysis to the agricultural sector. The majority of studies assume restrictions on the availability of feedstock for biofuels and a regional but constant unit cost of feedstock. Bai, Ouyang and Pang (2012) model competition over feedstock as a Stackelberg game where biofuel production facilities are modelled as leaders and the farmers as followers. Britz and Delzeit (2013) couple a sector model with market feedback to their localisation model to get feedstock prices. The regional level is often at the country or state level including a few hundred regions or on e.g. the EU level but then with fewer regions within each country. Some studies are applied at the country or state level, with a lower spatial resolution, but allowing for other model extensions. For example, Wetterlund *et al.* (2013) model choices between different technologies for the conversion of feedstock into biofuels, and de Jong *et al.* (2017) study the impact on overall costs of different cost-reduction strategies. Furthermore, some studies allow for different transport modes: train, truck and ship. Meanwhile, others model the choice between alternative feedstock sources. In addition, some studies perform a suitability analysis that favours factors such as proximity to cities, roads or power lines to choose a subset of possible locations before the optimisation of the model (Wilson, 2009). An overview of some relevant studies referred to in this section can be found in Table A1.a in Appendix A1.

The focus on agricultural feedstock in this study is relevant in light of the more stringent climate policies required by the Paris Agreement, as more biomass must be used for biofuels if they are to be a significant part of the mitigation strategy. Further, the question has been raised whether forests should be used as carbon sinks rather than for bioenergy production (Hedenus and Azar, 2009; Vass and Elofsson, 2016). This suggests that it is important to assess the cost-effective potential of agriculture as an alternative source of biofuel feedstock.

We contribute to the literature on the localisation of biofuel production facilities by focusing on agricultural feedstock, using a model that takes into account regionally increasing opportunity costs that arise due to competition over land with other types of agricultural production. We highlight the trade-off between forces that work in opposite directions: feedstock transport costs motivate decentralised small-scale production facilities, while economies of scale at the facilities have the opposite effect.

The empirical application to Sweden is of particular interest in this context due to its large geographical heterogeneity. A higher feedstock production potential in the south suggests it could be beneficial to locate production facilities there. There could also be cost advantages associated with locating these facilities in the vicinity of major demand centres, typically located where population density is the highest. However, the potential for cheap feedstock production in the north, due to the low opportunity cost of land, is an argument in favour of locating the production facilities there.

The paper continues with a description of the case study in Section 2, the model in Section 3 and the data used in Section 4. Thereafter, results are presented in Section 5, and Section 6 identifies policy implications. Finally, Section 7 provides a discussion and policy recommendations.

2. Case study area

We apply the model to biofuel production with agricultural feedstock in Sweden. There are large geographical differences across the country. In the northern parts, yields are lower and agricultural land scattered. In the southern parts more fertile soils are found, mixed with areas of low-productive land.

Agriculture is concentrated where fertile soils are more abundant. A large share of the agricultural land is used for ley production (Statistics Sweden, 2019). Sweden has a low population density, with most people living in the south (Statistics Sweden, 2020a).

In Sweden, the share of renewables in total fuel consumption was 23 per cent in the year 2018 (Swedish Energy Agency, 2019), and hence the target for renewable transport fuels has already been accomplished. However, Sweden also has a more ambitious target to reduce greenhouse gas emissions in the transport sector by 70 per cent in 2030, compared to 2010 (Government Offices of Sweden, 2017), to which biofuels could contribute (Swedish Climate Policy Council, 2020). Biodiesel, made from forest products and imported oils, is the most produced biofuel in Sweden (Swedish Energy Agency, 2019). Cereals have hitherto been the most used agricultural crops for producing biofuel (ethanol), but the feedstock is mostly imported (Swedish Energy Agency, 2019).

There are several different biofuel technologies which are capable of producing a range of biofuels such as ethanol, biodiesel and biogas. Further, different types of biomass can be used as feedstock in production: for example, forest residues, oil crops, cereals and energy grasses. In this study, we consider one type of second-generation biofuel[1] technology: the production of bioethanol from lignocellulosic material, in this case reed canary grass grown on agricultural land. Although there is no consensus on which is the superior biofuel technology, Börjesson *et al.* (2013) argue that the largest quantitative potential lies in such lignocellulosic-based biofuels where ethanol is one option; this technology could also perform well in an environmental and cost perspective. Moreover, ethanol is easier to use (Prade *et al.*, 2017). So far, the production technology used to produce ethanol from lignocellulosic material has only been developed to industrial scale in a few places in the world, but its large potential motivates our choice to consider this technology. The choice of reed canary grass is also made because it is possible to grow in most parts of Sweden and on most types of arable lands (Börjesson, 2007).

3. Model description

We develop a static linear programming optimisation model to find the cost-effective number, capacity and location of biofuel facilities, as well as the associated spatial distribution of feedstock production and biofuel consumption, given a national biofuel production target. The model takes regional heterogeneity in feedstock production costs and biofuel demand into account. We assume that the social planner's objective is to minimise the total supply chain cost to meet a certain production target, and that farmers and production facilities are price takers and strive to maximise profits. In the following sections, the modelling of production, feedstock supply, biofuel demand

1 According to RED II (European Parliament, 2018), second-generation biofuel includes, for example, non-food crops.

and transport, investment and operational costs are described along with the optimisation problem.

3.1. Feedstock supply

Feedstock can be produced in all regions g, with $g = 1,\ldots,G$. Within each region, the opportunity cost for producing feedstock can vary. To represent increasing costs for the feedstock, the supply in each region is divided into three cost categories f, with $f = 1, 2, 3$, with different costs as described below. The quantity of feedstock supply for a particular category f is denoted $x_{f,g}$. The quantity available for each category f in each region g is limited to $\bar{x}_{f,g}$:

$$x_{f,g} \leq \bar{x}_{f,g} \tag{1}$$

The total sales of feedstock of a given category f from a given production region g, i.e. $x_{f,g}$, equals the sum of amounts supplied to production facilities in all regions i, with $x^{TR}_{f,i,g}$, i.e.:

$$x_{f,g} = \sum_i x^{TR}_{f,i,g}, \forall \boldsymbol{i} \in \boldsymbol{g} \tag{2}$$

where the superscript TR indicates ‘transportation’.

3.2. Biofuel production

Production of biofuel in region i is indicated by y_i. Similar to Wetterlund *et al.* (2012), Lin *et al.* (2013) and Bai, Ouyang and Pang (2012), we assume a constant linear conversion factor α, expressing the volume of biofuel obtained per ton of processed feedstock, equal for all facilities. This assumption is reasonable when we consider only a single type of technology and single feedstock. Other inputs such as labour and energy are assumed to follow the feedstock use and are not modelled. We assume that the total feedstock input to a facility in region i is the sum of purchased feedstock from all regions. Thus, the production of biofuel y_i in region i is obtained by

$$y_i = \alpha \sum_g \sum_f x^{TR}_{f,i,g} \tag{3}$$

For production to occur, investment in a production facility is necessary. The variable $I_{v,i} \in \{0, 1\}$ indicates investment in a production facility at location i, taking the value 1 in the case of an investment and 0 otherwise. Facilities can be of different capacities $v = \{L, H\}$, where L indicates low capacity and H indicates high capacity. The production at a facility is restricted by capacity constraints $\underline{y}_v$ and $\bar{y}_v$ specific for the capacity types:[2]

$$\underline{y}_v \cdot I_{v,i} \leq y_i \leq \bar{y}_v \cdot I_{v,i}, \text{ for all } v = \{L, H\}, \underline{y}_H = \bar{y}_L \tag{4}$$

2 A lower capacity constraint on low-capacity facilities is needed to rule out facilities with too small capacities, which would need even higher marginal investment costs. The restriction $\underline{y}_H = \bar{y}_L$ is used to have two distinct capacity size choices.

In addition, to facilitate computation, we add the restriction that only one production facility can be built in each region[3]:

$$\sum_{v} I_{v,i} \leq 1 \tag{5}$$

3.3. Biofuel demand

Total production of biofuel Y from all production facilities is

$$Y = \sum_{i} y_i \tag{6}$$

The produced biofuel is distributed from the production facilities to demand points. The variable $y_{i,h}^{DISTR}$ denotes the sales of biofuel from the facility in region i to meet demand in region h. All produced biofuel must be distributed to demand points h located in the regions of the model, i.e.

$$\sum_{h} y_{i,h}^{DISTR} = y_i, \ \forall h \in g. \tag{7}$$

We assume that biofuel prices are fixed and equal across the country and thus omit them in the model. This assumption is reasonable as the large fuel companies do not differentiate biofuel prices across space. We assume that in each region there is a maximum demand for biofuel based on possibilities to blend in biofuel in fossil fuel. The lower level is assumed to be zero.

$$0 \leq \sum_{i} y_{i,h}^{DISTR} \leq \ \bar{\beta}_h \tag{8}$$

3.4. Costs

The main costs associated with biofuel production are costs of investment in the production facility, operational costs, costs for purchase of feedstock and costs for transport of feedstock from the supplier to the facility and from the facility to the end user. These costs are assumed to be additive.

Investment in building a new facility comes at a fixed cost, increasing with higher capacity levels. Further, investment costs *per unit* of capacity can be assumed to be lower for larger facilities, see, for example, Akgul, Shah and Papageorgiou (2012). We model the annualised investment cost c_i^{INV} with a fixed part δ_v, depending on the size of the facility, and assuming that $\delta_H > \delta_L$. In addition, we follow Lin *et al.* (2013) by including a variable investment cost ρ_v per unit of increase in the production capacity level, with $\rho_L > \rho_H$:

$$c_{i,v}^{INV} = \rho_v \cdot y_i \cdot \ I_{v,i} \ + \delta_v \cdot I_{v,i} \tag{9}$$

The formulation in equation (9) implies that we have a linearised representation of a non-linear concave investment cost function, reflecting economies

3 A positioning of two facilities in the same region would be economically unreasonable in our case, given the small regions and hence small feedstock supply in each region. Further, we never find facilities located in neighbouring regions in the empirical simulations.

of scale. This is illustrated in Figure 1, where the solid and dashed black line depicts the cost of the low-capacity facility, which has a low intercept and a steep slope, compared to the grey line, which depicts a high-capacity facility. The solid parts of the lines show the capacity ranges where each facility type is superior to the other, and in the cost minimisation context they form the piecewise linear and (in this case almost) continuous investment cost function.

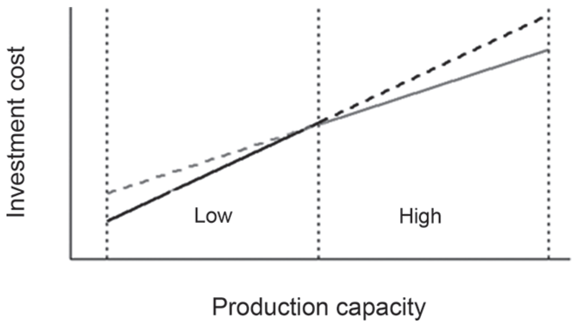

Fig. 1. Investment cost as a function of the production capacity level. The black solid and dashed line represents the cost for low-capacity facilities; the grey solid and dashed line represents the cost for high-capacity facilities. Solid lines indicate the minimum investment cost for a given capacity, and capacity spans are indicated by dotted lines.

The operational cost of the production process, c_i^{OP}, is assumed to depend only on the output level. Most earlier studies assume either a linear cost function which varies across facilities (e.g. Wetterlund *et al.*, 2012) or an identical linear cost function for all production facilities, where instead the economies of scale are modelled in the investment costs (Lin *et al.*, 2013). The latter approach is applied here, and σ, the cost per unit of produced biofuel, is assumed equal for all facilities:

$$c_i^{OP} = \sigma \cdot y_i \tag{10}$$

Earlier studies have made varying assumptions about feedstock costs. Sometimes the feedstock cost is assumed to be a fixed regional unit cost, as in Wetterlund *et al.* (2012). Alternatively, a supply function with increasing costs is included, as in Apland and Andersson (1996). Here, we take an intermediate approach, where we account for the fact that a larger feedstock production in a region will increase the opportunity cost of production in this region. The profit maximising farmer is assumed to only produce feedstock for biofuel production if the economic gain is equal to or exceeds the opportunity costs. We assume that the marginal cost of feedstock production is stepwise linear. Each category $f = 1{,}2{,}3$ represents taking more land into use for feedstock use. Feedstock of each category f is associated with a different unit cost $\vartheta_{f,g}$.

Consequently, in each region g, feedstock of the lowest cost level is chosen until its maximum capacity, given by the land available for the category, is exhausted. The total cost for feedstock of all categories f across all regions g for a given facility in region i is

$$c_i^{FEED} = \sum_g \sum_f \vartheta_{f,g} x_{f,i,g}^{TR} \tag{11}$$

As feedstock for producing biofuel can be bulky, transport brings important costs. We assume one transport mode (truck) while others, e.g. Akgul, Shah and Papageorgiou (2012), allow for different transport modes: train, truck and ship. The costs include loading and unloading as well as the time and fuel required for transportation. Therefore, transport costs c_i^{TR} increase with the feedstock used and the transport distance $d_{g,i}$ between the supplier at location g and facility at location i, given the unit feedstock transport cost φ^{FEED}. The cost function also has a fixed element ω^{FEED} per unit transported, reflecting the cost of loading and unloading, similar as in, e.g. Wetterlund *et al.* (2012) and Akgul, Shah and Papageorgiou (2012):

$$c_i^{TR} = \sum_g \sum_f \left(\omega^{FEED} + \varphi^{FEED} d_{g,i}\right) x_{f,i,g}^{TR} \tag{12}$$

Similarly, the cost c_i^{DISTR} for transporting biofuel from the facility at i to demand points h is determined by the unit transport cost φ^{FUEL} and the unit cost of loading and unloading, ω^{FUEL}:

$$c_i^{DISTR} = \sum_h \left(\omega^{FUEL} + \varphi^{FUEL} d_{h,i}\right) y_{i,h}^{DISTR} \tag{13}$$

3.5. Production target

Finally, we assume that there is a policy target for biofuel production at the national level that should be reached, defined as an annual production target Y^*:

$$Y \geq Y^* \tag{14}$$

3.6. Objective of the model

The objective of the model is to minimise the costs for meeting the production target Y^*, subject to the constraints in equations (1)–(8). The decision to invest in a production facility at a given location will depend on the spatial allocation of all facilities, feedstock production and biofuel demand. The problem includes agglomeration forces, due to the benefit of concentration that follow from economies of scale (equation 9), and dispersion forces from transport costs and increasing costs of feedstock production (equations 11–13). The problem can be described as follows:

$$\operatorname{argmin}_{x_{f,g}, x_{f,i,g}^{TR}, I_{v,i}, y_i, , y_{i,h}^{TR}} \sum_i \sum_v \left(c_{i,v}^{INV} + c_i^{OP} + c_i^{FEED}\right) + \sum_i c_i^{TR} + \sum_i c_i^{DISTR}$$

s.t.
Equations (1)–(14), and

$$x_{f,g}, x_{f,i,g}^{TR}, y_{i,}, y_{i,h}^{TR} \geq 0 \text{ and } I_{v,i} \in \{0,1\}$$

.

In the following, the model is simulated numerically using data relevant for our case study.

4. Data

This section gives a brief overview of the data; more details are given in Appendix A2.

4.1. Spatial structure

In the calculations below, the regional unit used for feedstock supply, location of production facilities and demand for biofuel is the municipality. The regional units are indicated by *g, i* and *h*, when related to feedstock production, biofuel facility location and location of final demand, respectively. We include all municipalities in Sweden, 290 in total. For these, the median total land area is 670 km².

To calculate the transport distances $d_{g,i}$ and $d_{h,i}$ between municipalities, we follow Leduc (2009) and Akgul, Shah and Papageorgiou (2012) by multiplying Euclidean distances between regions with a tortuosity factor (Zamboni, Shah and Bezzo, 2009) that accounts for the irregularities of the road network (Zamboni, Shah and Bezzo, 2009).

4.2. Feedstock

We assume that up to 50 per cent of the total agricultural land area in Sweden currently used for ley, and land classified as fallow or other unused land, can be used for the purpose of feedstock (reed canary grass) production. Of this, we assume half can be classified as cost category 1 and half as cost category 2. Moreover, we assume that up to 10 per cent of arable land used for crop production can be used for the same purpose and can be classified into cost category 3. The yield of reed canary grass is assumed to be 5.85 tons dry weight per hectare in Umeå municipality in the northeast of Sweden and differentiated across Sweden in proportion to ley yields, thereby reflecting spatial variations in soil quality and climate. This gives maximum regional supply quantities $\bar{x}_{f,g}$ with a total potential production of reed canary grass of 5.8 million tons per year in Sweden. The density of the potential feedstock area in relation to the total land area (also including non-agricultural land) is illustrated in panel A of Figure 2. The conversion rate of biomass to ethanol is set to 0.3 m³ of ethanol per ton of feedstock, which is the same as used by Lin *et al.* (2013) for another type of bioenergy grass, *Miscanthus*, for production facilities for ethanol from lignocellulosic material.

4.3. Biofuel demand

Currently, blending of ethanol into gasoline, or E85, is done at fuel distribution terminals, which are spread out at harbours in Sweden, in particular in the southwest. It is also possible to blend in the ethanol in the pump at the retail station. In our

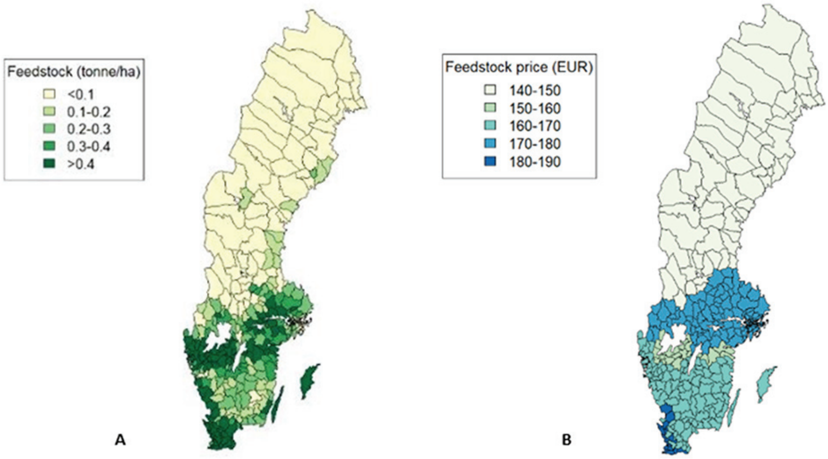

Fig. 2. Panel A: Density of feedstock, ton per hectare total land area. Panel B: feedstock costs, EUR per ton dry weight for cost category 1.

baseline scenario, we assume that the produced biofuel is both mixed with gasoline and used as bioethanol for ethanol vehicles, with blending at retail stations. The upper limit $\bar{\beta}_h$ is assumed to be 38 per cent of current gasoline consumption per region, in ethanol equivalents, and, when also considering ethanol as fuel, corresponds well to a proposed blending target for 2030 (Swedish Energy Agency) of about 28 per cent. For a scenario where all biofuel is used for shipping the spatial distribution of demand is calculated based on passenger and freight traffic at the largest harbours. In a third scenario, it is assumed that all biofuel is used for aviation and biofuel consumption distributed in proportion to the number of passengers per airport. The lower bound on demand is zero. Figure A2.a in Appendix A2 shows the density of fuel use for each end use.

4.4. Costs

All costs are given in EUR 2019. Investment and production costs for biofuels depend on the scale of the production facility, and the amount of biofuel produced. We use cost estimates from Lin *et al.* (2013), who model production of ethanol from Miscanthus grass in the USA. For our high- and low-capacity spans we use their two low and medium capacity segments (15,000–180,000 and 180,000–360,000 m^3 ethanol[4]), with variable and fixed investment costs ρ_v and δ_v listed in Table A2.a in Appendix A2.

The current scale of production of reed canary grass is small in Sweden, and production-cost data are not available. Therefore, the production cost $\vartheta_{1,g}$ for reed canary grass for cost category 1 is calculated as the opportunity cost for silage production. This is motivated because both crops are cultivated on similar types of

4 We can note that, thereby, the maximum capacity of the larger facilities exceeds the capacity of the single existing Swedish ethanol production facility, which has a capacity of about 230,000 m^3 of ethanol per year (Agroetanol, 2020).

land, using similar production processes. The costs for the second and third cost categories are calculated using the price elasticities for forage products in Sweden for the eight NUTS2 regions. This approach implies that the cost estimates take into account adjustments in other parts of the agricultural sector. The resulting feedstock opportunity costs for cost category 1 are shown in panel B of Figure 2.

Transport costs c_i^{TR} and c_i^{DISTR} for feedstock and biofuel, respectively, are based on the amount transported as well as the distance covered. These costs are based on transport costs for wood chips and ethanol in Sweden from de Jong *et al.* (2017) and are given in Table A1.b.

4.5. Production target levels

We base the levels of ethanol in the hypothetical production targets in our scenarios on the available feedstock. With a total of about 5.8 million tons of available feedstock, which could produce 1.7 million m³ ethanol in Sweden, the highest target Y^* is set a little lower, to 1.5 million m³ ethanol. This gives about 5.8 TWh, corresponding to 21 per cent of gasoline use in Sweden in the year 2018. This is in the same range as was found by Prade *et al.* (2017) to be a sufficient contribution of the agricultural sector to Sweden's emission targets for the transport sector.

5. Results

The results obtained from the model consist of locations for production facilities; production capacity levels of these facilities; flows of feedstock supply from regions to facilities and the distribution of biofuel to end users. Further, associated costs and transport distances can be studied. We analyse the effects of increased biofuel production in different scenarios, which are detailed in the next section, and describe the results of these in the subsequent sections and in the policy implications section. Thereafter we perform a sensitivity analysis. The results are calculated using a Mixed Integer Linear Programming model, programmed in the GAMS software (see Appendix A3).

5.1. Scenarios

We examine three sets of scenarios: *Target levels, Sequential* and *Demand distribution*.

The first set of scenarios (*Target levels*) are used to investigate how the location and production of production facilities and the total costs change with the stringency of the production target Y^*. The target levels in the scenarios range from 10 to 100 per cent of the maximum target of 1.5 million m³ ethanol. For these scenarios, the regional biofuel demand is allowed to be within a span, as outlined above.

For the remaining two sets of scenarios (*Sequential* and *Demand distribution)*, the *Target level 70 per cent* (1.05 million m³ ethanol) is used as a main reference scenario to compare with. This level equals 60 per cent of potentially available feedstock.

In the three *Sequential* scenarios, we investigate the effects of a sequential increase in the production target Y^*. They demonstrate a policy that is gradually introduced, for example, because there is uncertainty about the scale of final biofuel production that would be cost-efficient, given the overall target for renewables, or because the policy maker is budget constrained. Such a sequential implementation could potentially lead to a suboptimal distribution

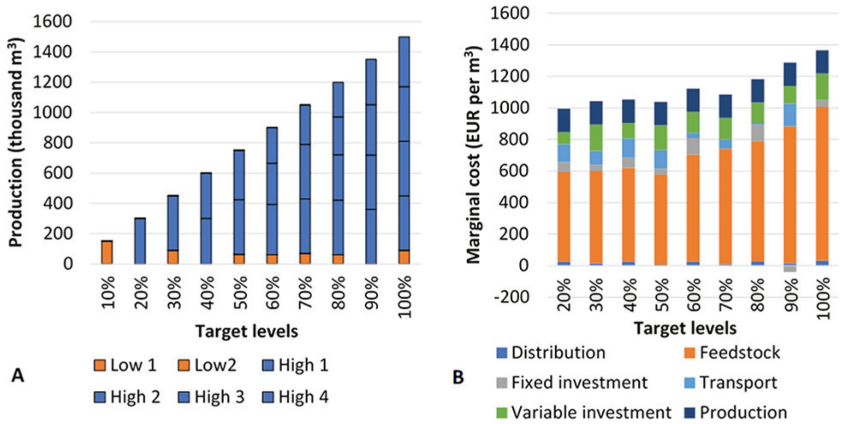

Fig. 3. Panel A: Production of ethanol at each facility for Target-level scenarios. Blocks representing low- and high-capacity facilities respectively, with production at each site shown by the size of the block. Panel B: Marginal costs in EUR per m³ for different production targets, divided into cost categories.

of production facilities. In *Sequential A*, we use the result from *Target level 30 per cent* as the first step in the sequential policy. Holding facility capacities from *Target level 30 per cent* fixed, the 70 per cent target is subsequently optimised. For *Sequential B*, there is an additional step in the optimisation: First the 50 per cent target is optimised holding facility capacities from *Target level 30 per cent* fixed, and then the 70 per cent target is optimised holding facility capacities from the previous step fixed. For *Sequential C*, we take the capacity of the single existing facility in Sweden (with a capacity of 230,000 m³ ethanol, located in Norrköping in the southeast) as the first (fixed) step in the policy, subsequently optimising the system to achieve the 70 per cent target.

In the third set of scenarios (*Demand distribution*), we compare the role of the geographical distribution of different types of end use of biofuel. There are two scenarios: in *Aviation demand,* demand is distributed as airport fuel demand, and in *Shipping demand,* demand is distributed as fuel demand per harbour.

5.2. Stringency of target levels

We illustrate the results for the *Target-level* scenarios in two figures. Figure 3 summarises biofuel production and costs for the whole range of target levels (Panel A) and shows the associated marginal costs (Panel B). Figure 4 shows the geographical location of facilities and feedstock supply. The blocks in Figure 3, panel A, show the number of facilities of low and high capacity, respectively, as well as the produced quantities at each of these facilities. At the lowest target level, there is one low-capacity facility, while at the highest target levels there are four high-capacity facilities and one of low capacity. The capacity for each facility generally increases with target levels until one additional facility is needed for a higher production level, only then with an increasing number of facilities as a result (see e.g. between 70 and 80 per cent). There are more high- than low-capacity facilities, illustrating the importance of the lower investment cost per unit at these facilities. Thus, investment costs are more important than transport costs. Many low-capacity

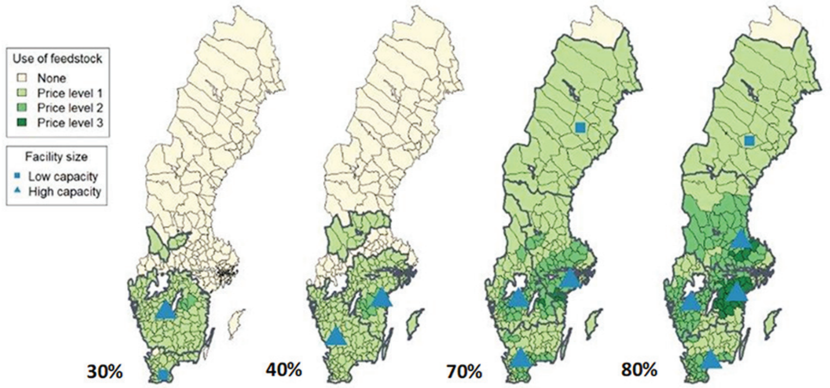

Fig. 4. Location of production facilities. Triangles symbolise facilities of high capacity and squares low capacity. Shaded areas surrounded by borders denote areas with supply to a facility. Darker areas indicate larger uptake of feedstock.

facilities would instead have been optimal if the transport distances to the facilities were most important.

Panel B shows the marginal cost at different production target levels, in EUR per m³ ethanol, decomposed into cost items (see Section A2.4.1 in Appendix A2 for the calculations). Marginal costs are increasing, which is mainly due to high and increasing feedstock costs, while there is a tendency towards a decreasing share of transport costs of the total marginal costs when the number of facilities is high. Due to the presence of investment costs, marginal costs are at some instances decreasing locally when production increases.

The maps in Figure 4 show the results for the selected target levels 30, 40, 70 and 80 per cent, with the locations of high- and low-capacity facilities (blue triangles and squares), their respective feedstock catchments areas (black lines) and the highest cost category for feedstock supplied from a region (light to dark green, with a darker shade indicating a higher-cost category). At lower target levels, the facilities are located in the south using feedstock of cost category 1 in the surrounding area. With increasing target levels, the feedstock catchment area increases. Feedstock of cost category 1 is almost solely used across the whole country before using some feedstock in higher-cost categories. This pattern of feedstock use shows that the feedstock costs are of great importance for the location and more important than transport costs. A consequence of the spread of the feedstock catchment area is that transport distances initially increase but then decrease when feedstock of a higher-cost category begin to be used.

The location differs between target levels, but persistently the low-capacity facilities are located in the north and the high-capacity facilities in the south. Some locations are quite stable, such as one high-capacity facility in the southwest and one high-capacity in the east. The southwest region is a suitable location as it is characterised by a relatively low feedstock price and high density of land available for feedstock production (see Figure 2), which decreases the cost of transported feedstock. Low-capacity facilities play a minor role in overall production, but they make up the least-cost method of utilising the more spread out, but relatively cheap, feedstock in the northern part of the country. These differences affect the unit cost structure for the different facilities, as shown in the 70 per cent scenario in Figure A4.a in Appendix A4.

5.3. Sequential scenarios

In the *Sequential* scenarios, the facilities from a previous target (*Target 30 per cent*) remain fixed. These are one low-capacity facility in the south and one high-capacity facility in the southwest (see Figure 4). Figure A4.b in Appendix A4 shows the production for the different scenarios. An increase to 70 per cent (*Sequential A*) results in a situation with more and smaller facilities than in the direct optimisation at this level (*Target 70 per cent*). For *Sequential B*, there are even more and smaller facilities. Consequently, the average distances from supply regions to facilities are lower. In addition, a larger share of the total production is in low-capacity facilities. The new locations are in both cases quite close to those in *Target 70 per cent* but shift in *Sequential A* and *Sequential B* to use the feedstock in the north rather than the south. For *Sequential C*, the initial facility is one high-capacity facility at a site close to one in *Target 70 per cent*, and thus the difference between the scenarios is not that large. The total costs are higher in all the sequential scenarios than in the *Target 70 per cent* scenario, but the difference is less than 1 per cent. One explanation for the small difference is that the important feedstock costs are hardly affected by fixed locations and capacities.

5.4. Spatial distribution of demand

In the *Demand distribution* scenarios, demand is concentrated to a few geographical points (see Figure A2.a in Appendix A2). However, the results show that the location of facilities is very similar to the main scenario (*Target 70 per cent*), as transport costs for ethanol are lower than those for feedstock. Nevertheless, the capacity level at each facility is different; in *Aviation demand*, the facilities close to the large airports, Arlanda in the east and Landvetter in the southwest, are at the highest allowed capacity, and in *Shipping demand*, the capacities are highest to the south and east where there are a lot of passenger ferries. This shows that the transport of fuel has some impact on the facilities. The less spread-out demand in these scenarios increases the costs relative to the main scenario: for *Aviation demand*, it is 1.3 per cent higher, while for *Shipping demand* it is 0.5 per cent higher.

5.5. Sensitivity analyses

Finally, we carry out sensitivity analyses with respect to key parameters. All the sensitivity analyses use the scenario *Target 70 per cent* and biofuel demand based on current fuel consumption as a base. There are four sets of scenarios: first, variable transport costs change by ±20 per cent; second, feedstock prices change by ±20 per cent; third, fixed investment costs change by ±10 per cent[5] and fourth,

5 To change the investment costs, the fixed cost component of each of the two capacity levels is changed so that the total investment cost at the intersection of the low- and high-capacity increases (decreases) 10 per cent, while variable costs are left unchanged. Thus, the relative change in fixed investment cost actually differs for high and low capacities; the resulting change

available feedstock changes by ±20 per cent, by changing feedstock of all cost categories.

A reduction in the fixed investment cost has a large effect as it implies more low-capacity facilities—three large and three small, which can be seen in Figure A4.c in Appendix A4. Also, with higher transport costs or a higher feedstock price, there are two rather than one small facility, meaning that these costs are quite important when it comes to localising facilities in areas with lower feedstock availability. For the other sensitivity analysis scenarios, the production per facility and locations are quite stable and similar to the main scenario. However, with decreased transport costs, the facilities are somewhat more spread out. At lower feedstock prices, the catchment area is more to the south, requiring lower transport costs. The total cost difference relative to the main scenario is largest for feedstock cost change as this cost constitutes the largest share of total costs (12 per cent cost increase and decrease, respectively).

6. Policy implications

As investments in biofuel production facilities can receive governmental support (such as is the case for biogas facilities in Sweden (Government Offices of Sweden, 2020)), their capacity and localisation is of relevance for such policy schemes. In addition to this, the results are relevant for the agricultural sector and for the development of policies for more renewables in the transport sector and for mitigation of greenhouse gases. These policy implications are further discussed in the following.

6.1. Animal fodder availability

The land assumed to be available for biofuel feedstock is currently mostly used for ley production. Therefore, fodder production would be affected by increased biofuel production. To measure the impact, we compare hectares of ley per grazing animals (cattle, sheep and goat) measured in livestock units (LSUs), with and without biofuel production The largest relative and absolute change in hectares per LSU occurs in east Sweden with more than 30 per cent decrease in area per LSU (see Figure A4.d in Appendix A4). In the north, the change is smaller in relative terms but larger in absolute terms due to high initial levels. The decrease in the south is smaller in absolute terms, but the effect is large in relative terms.

6.2. Policies for renewables

A first question is how the modelled biofuel production could contribute to existing targets for renewables. Our maximum target corresponds to 6 per cent of total current (2018) liquid fuel and 21 per cent of gasoline use, which together with the present use of renewables from other sources would be sufficient to comply with the EU target for 2030, that requires 14 per cent renewable fuel for both total liquid fuels and gasoline.

A second question is whether domestic production of biofuels or imports is a cheaper way to meet blending targets. Our results show that the marginal cost of domestic production ranges between 1030 and 1420 EUR per m³, see Figure 2. This can be compared to the projected world ethanol prices of 290 EUR per m³

is greater for low-capacity facilities. The 10 per cent level is chosen as we would otherwise get negative investment costs for the very small facilities.

in 2019 by OECD/FAO (2020) to increase to 360 EUR per m³ in 2029.[6] This suggests that domestic production is not competitive. However, the policy-driven global demand for biofuels could increase beyond the OECD/FAO projections, for example, if further policies put restrictions on the use of first-generation ethanol. Moreover, technology development could lead to decreases in production and investment costs by more than 50 per cent (Brown *et al.*, 2020).

6.3. Climate emissions and policy

The reductions in GHG emissions for reaching the 30 per cent and 70 per cent and production targets in the *Target levels* scenarios are 0.5 and 1.1 Mt CO_2 eq, respectively. These levels can be put in relation to the Swedish national target to decrease GHG emissions from the transport sector by 70 per cent until 2030, corresponding to a reduction by 11.2 Mt CO_2 eq. At the 30 and 70 per cent production targets in our analysis, the marginal cost per kg CO_2 reduction are EUR 0.23 and EUR 0.26, respectively (see Table A4.a in Appendix A4). This takes into account emissions from the ethanol production process, feedstock production, transports and negative emissions from the replacement of gasoline. Thus, the abatement cost is higher than the Swedish tax on GHG emissions for energy and transport, EUR 0.12 per kg CO_2 in 2021 (Government Offices of Sweden, 2021). However, Sweden also implements several abatement policies for which the marginal costs are higher. For example, the Swedish government applies a combined subsidy-tax scheme for new cars, which is judged to be highly cost inefficient (Brännlund, 2018; NAO (The National Audit Office), 2020). The marginal cost is for a similar subsidy scheme for electric cars in Norway is estimated to EUR 11.5 per kg of reduced CO_2 emissions (Holtsmark and Skonhoft, 2014), where the Norwegian scheme is slightly more ambitious. Moreover, NIER (2017) finds that current Swedish policies for transport fuel substitution imply a cost of EUR 0.5–0.8 per kg CO_2 avoided. See Appendix A2.5.2 for the calculations of the marginal abatement costs.

7. Discussion

This study provides insights regarding the optimal localisation of both facilities and feedstock for biofuel production on agricultural land. We developed a cost-effectiveness model for biofuel production, applied to Swedish data, with which we computed feedstock use, facility location, end use, costs, and greenhouse gas emissions. This enabled us to analyse trade-offs between scale benefits and transport costs, taking into account regional supply and demand differences and derive a national supply curve for biofuel production.

We found that the spatially differentiated opportunity costs of feedstock are the most important for the choice of location. High-capacity production facilities are located in areas combining relatively low feedstock cost and high feedstock density. In Sweden, this was found in the southwest, where most of the facilities would optimally be located. We saw that the significance of feedstock costs made it important to also utilise the cheaper feedstock in the north despite the higher transport costs when higher production targets need to be fulfilled. In that case, some low-capacity facilities were located there. To some degree contrasting with our results, Wetterlund *et al.* (2013) found that transportation costs can be of great importance for forest feedstock, especially when biomass availability is

6 World ethanol production is to the largest part first-generation ethanol.

restricted, implying high transport distances. In our case, feedstock availability played a smaller role, whereas transportation costs played a larger role at medium production target levels and a smaller role at higher production targets, which might be explained by a higher importance of increasing feedstock costs. Similar to Natarajan *et al.* (2014), who focus on the forest sector, we found that the heterogeneous spatial distribution of feedstock played a large role in facility distribution; but contrary to that study, we found that the demand distribution was of little importance. De Jong *et al.* (2017), also focusing on the forestry sector in Sweden, found that higher production implies a need for feedstock from further afield, which is more costly; this was also true of our results, up to the point when feedstock that was geographically closer but of a higher-cost category was chosen. The producers, as modelled by De Jong *et al.* (2017), would take all available feedstock from a region at once, as their marginal feedstock costs are constant. This shows the importance of considering regionally increasing costs of feedstock at high production targets, which arise due to competition over land. De Jong *et al.* (2017) found, similarly to us, that with larger-scale production the facilities were more decentralised. Some general conclusions we draw are that feedstock cost will be of largest importance, and that for high production targets the facilities would be spread out geographically, to compete over land at a low level everywhere rather than be clustered in some regions.

Our study has implications for policies for renewables and climate policies, how to organise production and the impact on agricultural production. The marginal costs we found were not competitive with current gasoline or ethanol prices but provide an indication of the future potential in Sweden. Further, the technology is developing and thus production costs could decrease. The marginal costs for GHG removal is higher than the Swedish CO_2 tax but lower than some other for other policies included in Swedish climate policy.

Energy policies are often developed in several steps. Production targets could, for example, be implemented via stepwise decisions, with targets increasing over time, rather than setting a high target all at once. This could risk incurring additional costs if the outcome deviates substantially from the optimal. We find that such a development would indeed result in suboptimal locations but that the additional cost is small. The presence of policy uncertainty could also imply higher feedstock costs if farmers require a risk premium, and the sensitivity analysis shows that this leads to a higher portion of production in small facilities and a substantial increase in costs.

Our model has limitations related to the scope and the availability of data. The trade-off between detail and a technically well-working model is important, and we use a static model to permit more detail and a clear interpretation of the results. More technologies and feedstock choices could be modelled than in our analysis, such as, e.g. gasification of energy forest into biodiesel or biogas production from agricultural crops and food waste (Börjesson *et al.*, 2016), where biogas would be the most qualitatively different. The use of another technology could imply lower investment costs, and our sensitivity analysis shows that this would in turn result in more and smaller facilities. The use of a single technology and feedstock type allows us to focus on the spatial distribution in the whole country and the effects of different forces on this distribution. Moreover, the yield of reed canary grass that can be obtained under commercial large-scale production, and thus feedstock availability, is uncertain. The sensitivity analysis shows that a lower availability of feedstock implies increased total costs but a similar number of facilities.

Our focus is on the least-cost solution and the calculations assume a competitive market, hence disregarding potential market failures associated with market power, which could be an issue when there are few large production facilities (Bai, Ouyang and Pang, 2012). However, market power can have a significant impact on feedstock prices, and our results showed that feedstock costs are important, suggesting that further analysis of market power, applied to the studied sector, could be relevant. In addition, we abstract from the possibility of policy failure, which could, for example, occur if governments are unable to implement cost-efficient policy instruments.

The trade-off with food production is a potentially serious concern regarding biofuel production. For example, Searchinger *et al.* (2008) find that large-scale bioenergy production could have a significant negative effect on food production, while on the other hand, Lotze-Campen *et al.* (2014) do not find any significant effects on food prices. In our scenarios, food production is not directly affected, as land for crop production is almost not used at all. However, we find that the impact of increased biofuel production on animal production can be large, due to a loss of forage area. In addition, ley production could increase on land for crop production, an effect not accounted for in the above analysis.

Potential extensions of the model could be to improve the assessment of the sustainability impact of biofuel production on, e.g. biodiversity, GHG emissions from land use changes and nutrient leakage and comparisons of the cost-effectiveness of agricultural biofuel production with other measures relevant for meeting targets for renewables and climate. Moreover, feedstock and biofuel can be traded across borders, and an extension of the spatial coverage of the model could aid understanding of how increased feedstock use on an international scale might affect the location of production and biofuel prices.

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Appendix A1: Literature overview

Table A1.a. Categorised literature

Study	Static/dynamic	Single/multiple facility	Facility regions	Feedstock	Feedstock costs	Feedstock choice	Technology choice
Lankoski and Ollikainen (2008)	Static	Single	One	Agricultural	Based on transport	Yes	No
Akgul, Shah and Papageorgiou (2012)	Static	Multiple	17	Forest	Fixed	Yes	Yes
Bai, Ouyang and Pang (2012)	Static	Multiple	20	Agricultural	Stackelberg game	No	No
Britz and Delzeit (2013)	Static	Multiple	401	Agricultural	Market model	Yes	No
Comber et al. (2015)	Static	Multiple	524	Agricultural	No	No	No
De Jong *et al.* (2017)	Static	Multiple	366	Forest	Fixed	Yes	Yes
Leduc (2009)	Static	Multiple	>400	Forest	Fixed	Yes	Yes
Lin *et al.* (2013)	Static	Multiple	102	Agriculture	Fixed	No	Yes
Natarajan *et al.* (2014)	Static	Multiple	120	Forest	Fixed	Yes	No
Rentizelas and Tatsiopoulos (2010)	Static	Single	Fine grid	Agricultural	Fixed	Yes	No
Rozakis *et al.* (2013)	Static	Single	>300 farm in sector model	Agricultural	Sector model	Yes	No
Santibañez-Aguilar *et al.* (2015)	Dynamic	Multiple	6	Mixed	Fixed	Yes	Yes
Sharma, Birrell and Miguez (2017)	Static	Multiple	Fine grid	Agricultural	Fixed	Yes	No
Wetterlund *et al.* (2012)	Static	Multiple	8 EU regions	Mixed	Fixed	Yes	Yes
Wetterlund *et al.* (2013)	Static	Multiple	>50	Forest	Fixed	Yes	Yes
Wilson (2009)	Static	Multiple	Fine grid	Agricultural	Fixed	No	No

Appendix A2: Detailed data

A2.1. Spatial structure

To calculate the transport distances $d_{g,i}$ and $d_{h,i}$ between municipalities, we first measure the Euclidean distances between regions. We identify the centre points of all municipalities and then measure the distance in kilometres between these centre points. The distance within a region is assumed to be zero. Geospatial data on the geographical extent of each municipality are taken from Esri's processing of data from Statistics Sweden (Esri, 2012). We construct tortuosity factors at the county level. For this, we use Google Maps (www.google.se/maps) to pick two arbitrary points in opposing ends of each county and then measure both the shortest route via the road network and the Euclidean straight distance and divide the former by the latter. To reduce the risk of possible measurement errors, the average between the county's calculated tortuosity factor and the average in Sweden, 1.3, is used as tortuosity factor for each county. The result is tortuosity factors in a range between 1.3 and 1.5, with higher levels more common in the north. The average is similar to that for road transport, 1.4, used by Leduc (2009) and Akgul, Shah and Papageorgiou (2012).

A2.2. Feedstock

The available land for feedstock is based on statistics on agricultural land in different municipalities, obtained from the Swedish Board of Agriculture (2020a). We assume that up to 50 per cent of the total land area currently used for ley, and land classified as fallow or other unused land, can be used for the purpose of reed canary grass production. Moreover, we assume that in addition up to 10 per cent of the arable land used for crop production can be used. The land areas in question are calculated as the average area over the years 2015–2019, which yields a total of about 750,000 hectares available for canary grass production. The yield of reed canary grass is assumed to be 5.85 tons dry weight per hectare in Umeå municipality in the north east of Sweden. This equals current yield as observed in field trials plus an expected future yield increase of 30 per cent as suggested by Börjesson (2007). The yield is assumed to be differentiated across Sweden proportionally to standard yield levels for ley in the corresponding agricultural production areas over the years 2014–2018 (Statistics Sweden, 2020b). This gives the maximum regional supply quantities $\bar{x}_{f,g}$ and a total potential production of reed canary grass of 5.8 million tons per year.

A2.3. Biofuel demand

For our baseline scenario, it is assumed that the produced biofuel is both mixed with gasoline and used as bioethanol for ethanol vehicles. Currently most blending is done at fuel distribution terminals. We assume blending is done in pumps at retail stations due to the assumed increase in biofuel use. The upper limit $\bar{\beta}_h$ is assumed to be 38 per cent of current gasoline consumption per region, in ethanol equivalents. This is 10 per cent above the maximum assumed supply target in the model, to make sure that there is enough demand. Further it corresponds well to a proposed blending target for 2030 of 28 per cent (Swedish Energy Agency). We assume the lower limit is zero in each region. To determine fuel use in different municipalities, we calculate the average energy equivalents of liquid fuel used for transport for the years 2014–2018 and multiply it with the average national share of gasoline to calculate gasoline use in each region. These data were obtained from the

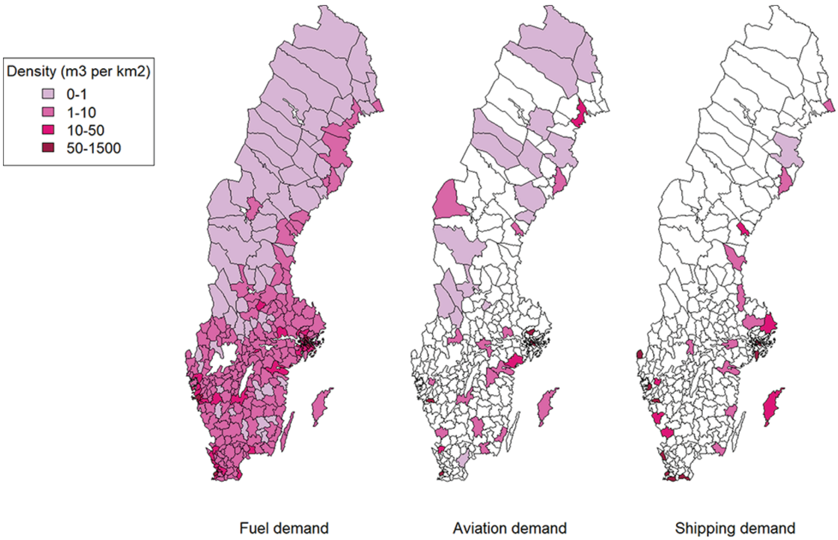

Figure A2.a. Spatial demand distribution. From left: fuel, aviation and shipping. Density fuel use per 1000 km².

Swedish Energy Agency (2020). The gasoline use was converted into the equivalent volume of ethanol. For the aviation scenario, the same approach was used, but biofuel consumption is distributed in proportion to the number of passengers per airport, using the average of 2018 and 2019 from the Swedish Transport Agency (2020). For the scenario where all biofuels are used for shipping, the spatial distribution of demand is calculated based on passenger and freight traffic at the largest harbours. The amount of fuel to passenger shipping and freight, respectively, is given by the average of fuel use for these purposes for domestic and international shipping from Sweden from 2003 to 2011 (Swedish Energy Agency, 2013). Of this fuel, the share to each harbour from passenger traffic is calculated based on the average number of passengers per year for the 10 largest harbours. For fuel for freight, consumption is calculated based on tons of goods per year per regional division, with the regional divisions allocated to municipalities with harbours (Transport Analysis, 2019). Together, this gives us an approximation of fuel per municipality with a harbour, from which the shares of total demand can be derived. Figure A2.a shows the density in demand for each end-use type, in total use[7] per 1000 km².

A2.4. Costs

All costs are given in EUR 2019. For the conversion of currencies, we use exchange rates from The Swedish Riksbank (2021) and the consumer price index from Statistics Sweden (2021). We use cost estimates from Lin *et al.* (2013), who model the production of ethanol from Miscanthus grass in the USA. They base investment cost on observed costs at a single facility and scale these costs to different capacity levels, assuming decreasing marginal costs. They then perform a linear approximation of the costs, in three segments of capacity

7 Of the biofuel produced in this scenario.

Table A2.a. Investment and operational costs, technology specification, per year

		High capacity	Low capacity
Capacity, $\underline{y}_v - \bar{y}_v$	1000 m^3 ethanol	180–360	15–180
Operational cost, σ	EUR/m^3 ethanol	145	145
Variable investment cost, ρ_v	EUR/m^3 ethanol	129.7	185.7
Fixed investment cost, δ_v	Million EUR	15.4	6.0
Conversion, $\boldsymbol{\alpha}$	m^3 ethanol/ton feedstock	0.3	0.3

Source: Lin *et al.* (2013).

Table A2.b. Transportation costs

	Distance-based EUR per km	Loading and unloading EUR
Feedstock	0.153 per ton	4.81 per ton
Biofuel	0.120 per m^3	9.71 per m^3

levels. The investment costs are annualised assuming a 15-year lifespan. We sum up the costs for pre-processing plants and biofuel production facilities, assuming that the both activities take place in a single type of facility. For our high- and low-capacity spans, we use their two lower-capacity segments, with costs listed in Table A2.a. These approximately give an equal cost at the break between high and low facility. The variable operations costs σ are assumed to be equal for all facilities.

The production of reed canary grass is low in Sweden, and production-cost data are not available. Therefore, the production cost $p_{f,g}$ for reed canary grass is assumed to equal the opportunity cost for silage production, on the agricultural production region level, from the Agriwise business-calculation database (Agriwise, 2019). This is motivated by the fact of the two crops being cultivated on similar types of lands, using similar production processes. The opportunity cost is based on the costs for silage production and foregone profits for spring barley, when the land is used for biofuel feedstock. Moreover, we assume three cost categories f in the stepwise linear supply function. For the land currently used for ley production or not being used, which we assume is available for feedstock production, we assume that half can be used to produce feedstock of cost category 1 and half for cost category 2. When land currently used for crop production is used for feedstock production, we assume this is associated with cost category 3. The resulting feedstock opportunity costs are shown in Figure 4. The cost for the second cost category is calculated using the price elasticities for forage products in Sweden at the NUTS2 level from the CAPRI database (CAPRI Modelling System, 2020) (see table PELAGRP), ranging from 0.08 to 1.5. Thus, a 25 per cent reduction in the area of land for forage production would increase feedstock production costs differently for each NUTS2 region. For the third cost category, we use the same price elasticity, calculating the effect of a 50 per cent reduction in the area of forage crops (corresponding to a situation where more than 50 per cent of the area ley and fallow land is used for feedstock production).

For each unit transported, there is both a fixed cost for loading and unloading and a distance-based cost for transportation. These costs are based on transport costs for wood chips and ethanol in Sweden from de Jong *et al.* (2017) and given in Table A2.b.

A2.4.1. Marginal cost

To approximate marginal cost we take the average total cost increase per extra unit of biofuel between the applied production level and the closest lower production level. The total cost c_y^{TOT} for a production level y for a specific scenario in the country is given by

$$c_y^{TOT} = \left[\sum_i \sum_v \left(c_{i,v}^{INV} + c_i^{OP} + c_i^{FEED} \right) + \sum_i c_i^{TR} + \sum_i c_i^{DISTR} \right]_y . \tag{A2.1}$$

The marginal cost, MC_y^{TOT}, is given by

$$MC_y^{TOT} = \frac{c_y^{TOT} - c_{y-\Delta y}^{TOT}}{\Delta y} \tag{A2.2}$$

where Δy is the difference in biofuel production between two production targets in the scenarios, i.e. 150,000 m³ ethanol.

A2.5. Emissions

A2.5.1. Emission calculation

Emissions from biofuel production are assumed to be linearly related to the different steps in the production process. We assume emissions from feedstock, $e_{i,g}^{FEED}$, to be differing regionally based on geographic characteristics with emission intensity ε_i^{FEED} for each unit of feedstock, while other emissions are equal across the country: e_i^{OP}, for production, $e_{i,g}^{TR}$, for transport of feedstock $e_{i,h}^{DISTR}$, and for transport of biofuel with emission intensities, ε^{OP}, ε^{TR}, and ε^{DISTR}. The emission from transport depends linearly on distances in addition to volume.

$$e_i^{OP} = \varepsilon^{OP} y_i \tag{A2.3}$$

$$e_{i,g}^{FEED} = \varepsilon_i^{FEED} x_{i,g} \tag{A2.4}$$

$$e_{i,g}^{TR} = d_{i,g} \varepsilon^{TR} x_{i,g} \tag{A2.5}$$

$$e_{i,h}^{DISTR} = \varepsilon^{DISTR} d_{i,h} y_{i,h} \tag{A2.6}$$

The total reduction r in emissions is calculated based on the volume of gasoline that the ethanol, $y_{i,h}^{DISTR}$, could substitute, and the gasoline emission factor ε^{GAS}, less the emission of ethanol:

$$r = \sum_i \left(\sum_h y_{i,h}^{DISTR} \varepsilon^{GAS} - e_i^{OP} - \sum_g \left(e_{i,g}^{FEED} + e_{i,g}^{TR} \right) - \sum_h e_{i,h}^{DISTR} \right) \tag{A2.7}$$

A2.5.2. Marginal abatement cost

Marginal abatement cost is given by the marginal cost per marginal abatement at a production level y (r_y). First, marginal abatement is given by

$$MA_y^{TOT} = \frac{r_y - r_{y-\Delta y}}{\Delta y} \tag{A2.8}$$

Marginal abatement cost MAC_y^{TOT} also deducts the forgone marginal costs for gasoline per m³ biofuel, $p_0^{GASOLINE}$. The forgone marginal costs for gasoline, $p_0^{GASOLINE}$, is based on the Swedish gasoline price, EUR 1500 per m³ (Swedish Energy Agency, 2021), less the carbon tax, EUR 120 per ton CO_2:

$$MAC_y^{TOT} = \frac{MC_y^{TOT}}{MA_y^{TOT}} - p_0^{GASOLINE}. \tag{A2.9}$$

A2.5.3. Emission data

Data on emissions from feedstock production, ε_i^{FEED}, was taken from Ahlgren *et al.* (2011) who give emission intensities in g CO_2 eq/kg dry matter crop, for reed canary grass at the county level in Sweden. Data on emission from the biofuel production, ε^{OP}, were taken from Bonomi *et al.* (2019), reporting numbers from the New EC Bioenergy model, with the case of ethanol made from wheat straw in Europe. Emissions for transport of feedstock was taken from Leduc (2009), and $\varepsilon^{TR} = 48$ g CO_2/km/ton. For similarity with ε^{TR}, emissions for transport of biofuel $\varepsilon^{DISTR} = \varepsilon^{TR} \frac{\varphi^{FEED}}{\varphi^{FUEL}}$ CO_2 km/m³. Data on the emissions from the combustion of gasoline and diesel were taken from Swedish Environmental Protection Agency (2019), given in kg CO_2/l ethanol with ε^{GAS}=1.5.

A2.6. Impact on animal production

Data on ley production and the number of LSUs for grazing animals are used to calculate the potential impact on animal production. To calculate the change in land for ley production, we take the amount of land used for biofuel feedstock production in a region and subtract the initial level of fallow land and land of unknown use, as these are assumed to be used first, and not counted as used for ley production. The number of grazing animals (cattle and sheep) per county were obtained from the Swedish Board of Agriculture (2020b) and transformed into LSUs (Eurostat, 2020b). Dividing change in hectares by the LSU numbers we get the direct change in LSU per hectare.

Appendix A3: Software

Results are calculated using the GAMS software, using version 30.3 for calculating starting values in first simulation that optimises the model without considering demand distribution (GAMS Development Corporation, 2020) and 24.7 when running the full model using the starting values (GAMS Development Corporation, 2016). The optimisation is solved with the OSICPLEX solver. The solutions are calculated with a minimum gap tolerance from optimality equal to 1 per cent for starting values and 2.5 per cent for the full model.

Appendix A4: Results

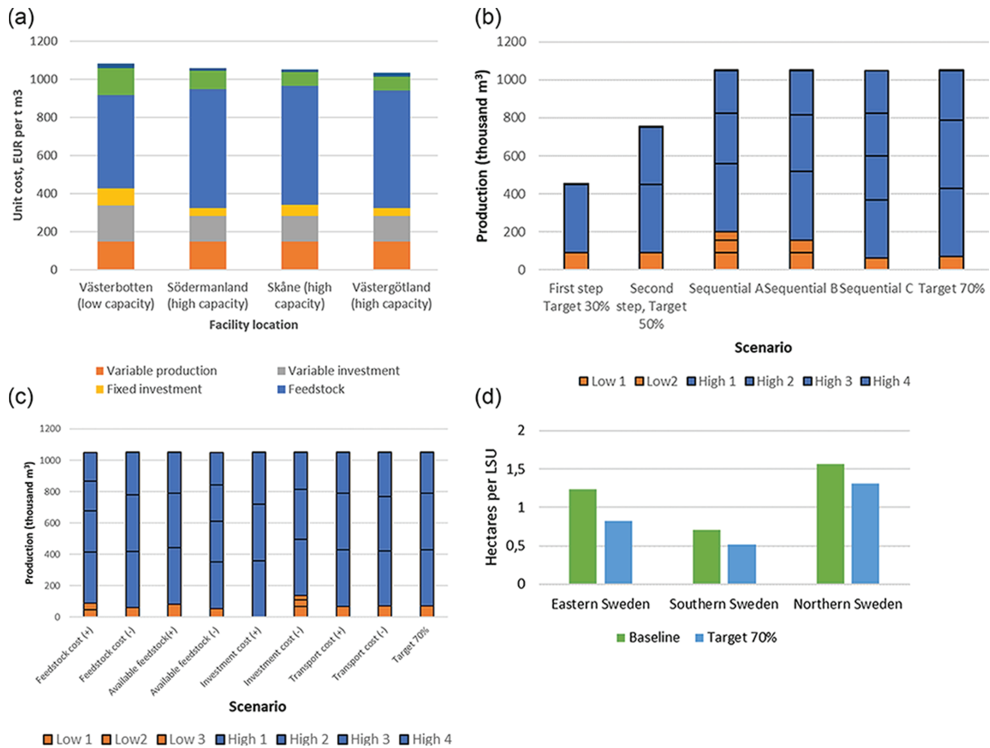

Figure A4. (a) Unit costs for facilities in the Target levels 70 per cent scenario. (b) Sequential scenarios—production of ethanol at each facility. Blocks representing low- and high-capacity facilities respectively, with production at each site shown by the size of the block. Shaded block represents intermediate steps. (c) Production of ethanol at each facility—sequential scenarios. Blocks representing low- and high-capacity facilities respectively, with production at each site shown by the size of the block. (d) Hectares of land for forage production per LSU, with and without biofuel production in Target 70 per cent.

Table A4.a. Marginal abatement costs for different Target-level scenarios. EUR per kg CO_2

Scenario	EUR per kg CO_2
10%	0.17
20%	0.22
30%	0.23
40%	0.21
50%	0.30
60%	0.29
70%	0.26
80%	0.35
90%	0.42
100%	0.53

IV

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Featured Article

Coupled Agricultural Subsidies in the EU Undermine Climate Efforts

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Abstract *Subsidizing polluting industries generally leads to increased pollution locally. However, given the diversity of production technologies across countries and international trade, the global impact of unilateral policies is not* a priori *clear. We use the agricultural sector model CAPRI to simulate the impact of removing the voluntary coupled support for ruminants, presently permitted under the EU Common Agricultural Policy. We find that this reduces greenhouse gas emissions in the EU. However, emissions leakage significantly diminishes the global mitigation effect since about 3/4 of the reduction in the EU is offset by increased emissions in the rest of the world.*

Key words: Agricultural Policy, Climate Change, Coupled Support, Emissions Leakage, EU.

JEL codes: Q18, Q54, Q17.

Introduction

A significant proportion of all greenhouse gas (GHG) emissions in the EU come from the agricultural sector,[i] which has a largely untapped potential to reduce these emissions (Allen and Maréchal 2017; Grosjean et al. 2016).

[i] *About 11% of net GHG emissions in the EU in 2017 according to the EEA. That number excludes land use and land use change and energy use in agriculture.*

Therefore, emission reductions in agriculture can be vital in helping the EU achieve its 40% target for reduction in domestic GHG emissions by 2030 (European Environment Agency 2015). Indeed, the European Commission emphasizes the need for the future Common Agricultural Policy (CAP) support to farmers to be conditioned on adoption of climate-friendly practices (European Commission 2017a).

Despite the potential to reduce emissions, the agricultural sector is exempt from the EU emissions trading system (EU-ETS)—the cornerstone of EU efforts to limit global warming. The sector is exempt from the EU-ETS due to concerns about emissions leakage, i.e., reallocation of production to other countries, and due to difficulties monitoring emissions in the sector (European Commission 2016). Even though the livestock sector (ruminants in particular) has the highest GHG emission intensity and highest total emissions within agriculture (*e.g*. Lesschen et al. 2011; Golub et al. 2013), the current CAP allows countries to subsidize ruminant production using Voluntary Coupled Support (VCS, described below in more detail).

Removing production subsidies for polluting production, such as VCS to ruminants, is a potentially cost-effective climate policy. Removing VCS would mean fewer ruminants in the EU and consequently less GHG emissions there. However, it may result in increased production and thus higher emissions in other countries, both within and outside the EU, particularly if the emissions per unit of product (emission intensities) are relatively higher in these countries. This *emissions leakage* (Markusen 1975; Zhang 2012) could limit or even reverse the positive impact on global warming that could come from removing VCS in the EU. Does the risk of emissions leakage justify the existence of VCS if GHG emission intensities are lower in the EU than in other countries? In other words, does more agricultural production in the EU reduce production abroad and thereby reduce the global emissions of GHG?

We analyze the likely impact on global GHG emissions resulting from removal of the current VCS in the EU. Our analysis is carried out with the CAPRI model (Britz and Witzke 2014), which is an agricultural sectoral simulation model. The model is extended with the inclusion of VCS for each of the EU member states (MS) to facilitate the analysis. The overall emission change is decomposed into production-level effects and reallocation effects in order to identify the causes and size of emissions leakage. An extensive and systematic sensitivity analysis with respect to key model parameters confirms the robustness of the main results.

A deeper understanding of the global effect on emissions and emissions leakage of unilateral removal of production subsidies harmful to the environment can facilitate better-designed agricultural policies. That is, policies that align with the climate policy objectives and effectively reduce global GHG emissions, not just domestic emissions. Thus, this article contributes by: (i) quantifying and assessing the climate impact of production subsidies for ruminants in EU MS and the emission leakage resulting from removing VCS; (ii) extending the CAPRI model with the inclusion of all VCS for all EU MS, which will enable further analysis of the increased use of coupled support and *de facto* nationalization of the agricultural policies; and (iii) developing a systematic sensitivity analysis for model parameters in the CAPRI model so that the robustness of the results and importance of key model parameters can be assessed in simulations with the model.

The remaining part of the paper is structured as follows. The next section reviews other studies of emissions leakage in agriculture. Then, there is a section on data and methods where we describe relevant parts of the CAPRI model, the estimation of GHG emissions, the European agricultural policy context and the scenarios applied. The results are presented in the fourth section and discussed in the fifth.

Previous Simulations of Climate Policy and Emissions Leakage in Agriculture

A few previous studies have considered emissions leakage within the agricultural sector, but to the best of our knowledge, the impact on global GHG emissions of EU production subsidies within the CAP has not previously been analyzed. Fellmann et al. (2012) and Fellmann et al. (2018) used CAPRI to simulate EU-wide reductions in GHG emissions of 20% and 28% by 2020 and 2030, respectively, relative to 2005, in response to global climate agreements. Specific policy changes were not investigated, however. One of the findings in these studies was that the reductions in GHG emissions in the EU were accompanied by significant emission leakage. Lee et al. (2007) used the GHG version of the US Agricultural Sector Model (ASMGHG) to simulate the welfare impact and emission leakage from unilateral, partial global, and full global implementation of mitigation policies related to emissions reduction actions on agricultural production and international trade. They found that under a unilateral policy, total GHG emissions decline, but substantial emission leakage occurs. Van Doorslaer et al. (2015) found that emission leakage can significantly reduce the benefits of emission reductions in the EU, depending on how climate policies are implemented in the EU. This implies that a policy effective at reaching regional climate objectives (*e.g.*, reducing GHG in the EU) may not be the best way to reduce global emissions. Reviewing the literature on carbon leakage, Zhang (2012) found that most models predict significant leakage effects, though mostly well short of 100%. When comparing *ex-ante* to *ex-post* results, they found that the predicted leakage was difficult to verify empirically, suggesting that models tend to overestimate leakage. However, none of the studies surveyed looked specifically at agricultural markets, and the models used were mostly computable general equilibrium models, and hence Zhang's observations, albeit interesting, are not directly transferable to our case.

Theory and Method

Based on economic theory we expect that removing production subsidies, in our case VCS in the EU, will reduce domestic production. The decline in domestic production causes an increase in import demand in the EU, a reduction in export supply from the EU, and a consequent rise in prices on the world market. This in turn provides incentives to increase production outside the EU. In other words, part of the EU's ruminant production and associated emissions would reallocate abroad, causing emission leakage, as discussed by Markusen (1975) and Zhang (2012). This emissions leakage might be expected to offset emissions reductions obtained in the EU, or even lead to an increase in total global emissions. Therefore, the effect of policy changes—specifically the effect of removing VCS—on global GHG emissions is not a priori clear, but needs to be quantified.

The CAPRI Modeling System

The present analysis was based on CAPRI Stable Release 1.3 (STAR 1.3, publicly available from www.capri-model.org), but with updated data in the area of GHG emission estimates. The CAPRI model is a partial equilibrium simulation model covering the agricultural sector (Britz and Witzke 2014). The model simulations provide results for the global impact on production and trade in the agricultural sector, aggregated to about forty trade blocks, and detailed results for NUTS2 (Nomenclature of Territorial Units for Statistics) regions within the EU. Countries outside the EU are represented in a more simplified fashion than EU countries (EU+), and therefore less detailed information on production and emissions is available for these. Trade flows between the forty regions are modeled based on the Armington assumption of product differentiation by origin. With regard to global trade, the model includes policy data on tariffs, tariff rate quotas, and the trigger price system of the EU. For EU countries, the model also contains a detailed representation of the CAP's policy measures, thus making it suitable for analyzing the impacts of agricultural policy reform scenarios. In addition, we have added VCS measures for all EU MS to the model in order to better represent the production coupling of the CAP and simulate the impact of VCS on GHG emissions.

CAPRI is a comparative static model, meaning the policy impact is inferred from a comparison of a baseline and a policy scenario at a specific point in time. In the present study, this point in time was set as 2030, after the end of the next multiannual financial framework.[ii] The CAPRI model is frequently used to assess the impact of changes in the CAP on aspects such as production, trade, and selected environmental indicators. Recent examples include: simulations of the impact of currently proposed EU free trade agreements and carbon taxes on GHG emissions (Himics et al. 2018); simulations of the impact of the so-called "greening" measures in the 2013 CAP reform (Gocht et al. 2017); and, used together with other models, simulations of the impact of climate change on agriculture (Blanco et al. 2017).

GHG Emissions in CAPRI

CAPRI's coverage of GHG emissions is global, but the method used to calculate emissions varies depending on the availability of detailed production data from the simulations. For EU+ countries,[iii] more details on production are available than for other regions, allowing a bottom-up computation of emissions based on production technology. For all regions, the main direct and indirect emissions of methane (CH_4) and nitrous oxide (N_2O) from agriculture are covered[iv] (representing agricultural emissions according to the UNFCCC classification). The CO_2 emissions from land use, land use change, fertilizer production, and energy use on farms are omitted from our analysis, as they are not yet covered globally in the CAPRI model. Gerber et al. (2013) estimate that about 75% of emissions from beef production are in the form of N_2O and CH_4, and about 25% are CO_2 emissions from land use and land use

[ii] *The duration of the multiannual financial framework has not yet been decided, but could be 5–10 years after 2020 (European Commission 2017c).*

[iii] *The twenty-eight countries of the EU before Brexit plus the Western Balkans, Turkey, and Norway.*

[iv] *The following emissions categories are included in our study: Methane: Enteric fermentation, Manure management (housing and storage), Manure application on soils except pastures, and Rice cultivation. Di-nitrous oxide: Manure deposition on pastures, Inorganic fertilizer application, Crop residues, Indirect from ammonia volatilization, Indirect from leaching and runoff, and Cultivation of organic soils.*

change, but with large uncertainties. The effect on our results of omitting emissions from land use and land use change are unclear, as the importance of omitted emissions and production methods varies across regions.

To compare emissions of different gases, Global Warming Potential (GWP) was used to convert all gases into carbon dioxide equivalents[v] (CO_2-eq.). The climate change induced by the change in emissions would also have an impact on agricultural systems. That feedback is not modeled in CAPRI.

For EU+ regions, emissions are computed endogenously in the CAPRI model based on detailed input and output data. This means, for example, that changes in the feed mix for animals due to a policy change can be captured and thus result in changes in emissions. For the main emission sources, the calculation is performed using a more detailed method (Tier 2 in the 2006 IPCC 2006 guidelines), while for some sources with lower total contributions to emissions, a simplified method (Tier 1) is used. Emissions are calculated per hectare of land or per animal production activity, and then allocated to commodities associated with those agricultural activities. A more detailed description of the method is available in Leip et al. (2010), Pérez Domínguez (2005), and Pérez Domínguez et al. (2012).

The high level of detail on production technology used to compute emissions in the EU+ is not available for other regions. For these regions, computations of GHG emissions are based on estimated emission intensities (EI) per tonne (metric ton) of product, without capturing endogenous changes in the composition of inputs that may take place in simulation (Pérez Domínguez et al. 2012). This means production technology outside the EU+ is assumed not to be affected by policy changes in the EU. To calculate total emissions in each scenario, the emissions coefficients are multiplied by production level.

EI for non-EU regions are estimated to follow the overall agricultural emissions reported in FAOSTAT GHG inventories as closely as possible over time. The estimation, carried out for each non-EU region and emission category individually, is based on time series data of regional GHG inventories and production of agricultural commodities. Data on production quantities come from the CAPRI database, and the GHG inventories come from FAOSTAT (FAO 2010–2018). In most cases the data cover the period 1990–2009, while in some cases fewer years are available. In many cases, we have many commodities compared to the number of years of GHG inventory and production data, and thus the degrees of freedom might end up being small or even negative. In order to improve the robustness of the estimates, we include prior distributions for the emission intensities in a Bayesian estimation framework (*e.g.* Koop 2003, p. 15). To capture the possible change in emission intensities over time, the estimations also contain a trend component.

Bayesian prior distributions for the EI are derived from various sources, such as the expert estimates in Leip et al. (2010). Additionally, we construct priors for many commodities and emission categories with data on activity levels and production levels from the 2014 version of the AGLINK-COSIMO model (OECD 2015). Emissions per activity are computed following the Tier 1 methodology in the IPCC Guidelines (IPCC 1997; IPCC 2006), and then converted to emissions per product. Also, average EU emission coefficients computed in the CAPRI model are used as priors when the previous sources are not available.

[v]*The GWP conversion factor used is 28 for methane and 265 for nitrous oxide, from the latest IPCC report (AR5) with a 100-year time-horizon, without inclusion of climate–carbon feedbacks (IPCC 2014).*

Decomposition of Emission Changes

Emissions leakage is influenced by changes in the level of production, but also by its reallocation to regions with different emission intensities. When production is reallocated to regions with higher emission intensities, the total emissions will increase for a given level of production and *vice versa*. In order to disentangle the impacts of production changes and changes in average EIs, we made an additional computation of emission changes: First, we set all the EIs equal to the global average in the reference scenario for all countries, and thereafter we calculated the emissions using the production changes in the policy scenario. This computation captures *only* the effect of changing global production levels. Those calculated changes (*i.e*., changes due to changed production levels) were subtracted from the global changes in GHG emissions computed using regionally specific emission factors, giving the emissions changes caused by reallocation of production to regions with different EIs as a residual.

Baseline for Agriculture and Policy in the EU

The CAPRI baseline projects agricultural production and emissions to the year 2030 under a business-as-usual scenario. Trends for factors exogenous to the model such as population growth and consumer preferences are set based on external projections. The development of agriculture in the EU is based on the Agricultural Outlook published by the European Commission. The CAP is assumed to be fully implemented up to 2021 and then unchanged.

Within the CAP, the largest part is Pillar I measures, which mainly involve support and some market intervention schemes. Pillar II covers support to certain agricultural production, environmental measures, and rural development. Within Pillar I, most support (75%) consists of direct payments to farmers on a per-hectare basis for all qualifying agricultural land. The largest proportion of these payments is the Basic Payment Scheme (or the Single Area Payment Scheme in some regions), with support allocated to all agricultural land with entitlements. This support is considered to be decoupled from production, and member states are obliged to harmonize per-hectare rates across regions (European Union 2013). The greening payment is another large part, and it comes with associated constraints on crop diversification, grassland maintenance, and keeping ecological focus areas. A smaller part of Pillar I is dedicated to payments to young farmers and smaller farms, and areas with natural constraints. In addition, there is crop-specific coupled support for cotton in some countries, and complementary National Direct Payments in some countries.

VCS, the focus of the present study, permits MS to use up to 13%[vi] of the Pillar I payments for coupled support to sectors undergoing economic, social, or environmental difficulties in maintaining/increasing production (European Commission 2017b). The measure is used by most MS and mainly targets cattle[vii] and other ruminants[viii] (European Commission 2019). In total,

[vi] *The exact maximum depends on the circumstances (European Commission 2017b).*

[vii] *VCS to cattle is applied in: Belgium, Bulgaria, Czechia, Estonia, Spain, France, Croatia, Italy, Cyprus, Latvia, Lithuania, Hungary, Malta, Poland, Portugal, Romania, Slovakia, Slovenia, Finland, Sweden and in the UK (Scotland).*

[viii] *VCS to the sheep and goat sector is applied in: Belgium, Bulgaria, Czechia, Greece, Spain, France, Croatia, Italy, Cyprus, Latvia, Lithuania, Hungary, Malta, The Netherlands, Austria, Poland, Portugal, Romania, Slovakia, Finland and in the UK (Scotland).*

we modeled 278 different VCS measures across the EU member states. Thirteen percent of the total budget might be considered a minor share of the budget, but it has a potentially strong impact on emissions: Most VCS, about 43% of the total in our data set, is linked to the production of beef and veal, and another 12% to sheep and goat production. These sectors together cause large emissions of the GHG methane and N_2O, either directly or via fertilizer used for producing fodder. The dairy sector also receives much VCS, about 20% of total VCS payments, but in dairy it generally constitutes a smaller proportion of the revenues than in beef production.[ix] Among the crop sectors, the production of protein receives notable amounts (8.5% of the total) of VCS in many member states, followed by fruit and vegetables (at 5%), but these sectors are less interesting from a GHG emissions perspective.

Simulated Scenarios

Two policy scenarios were considered:

- A reference scenario, abbreviated "Ref."
- A policy scenario, abbreviated "No VCS."

In the reference scenario, the current CAP was assumed to continue until 2030, thus including VCS as described above.

The policy scenario was identical to the reference scenario, except that VCS for ruminants was removed. In the CAPRI model, these subsidies are implemented as a direct subsidy per head, with budgetary ceilings as reported by EU countries. The budget that was released when VCS was removed was allocated to the other farm payments (the Basic Payment Scheme) in each MS, so that the total budget for farm payments in each MS remained unchanged in the reference and policy scenarios. The redistribution of support in the policy scenario resulted in an average increase in per-hectare payments for agricultural land of 6.5% in the EU, while support linked to beef cattle decreased by 69% per head, support for dairy cows by 41% per head and for sheep and goats by 36% per head. The remaining coupled support consisted of payments that are not part of VCS: national payments such as Nordic Aid and environmental and rural development support. The impact of a policy change in 2030 was derived by comparing the two scenarios.

Sensitivity Analyses

The CAPRI model results depend on a large number of parameters, some of which are more uncertain than others. In order to analyze how the results obtained in this paper depend on uncertain parameters, a set of sensitivity analyses were carried out. We selected four types of parameters that were assumed to be most critical to emissions leakage, and varied those in three levels: "low" (lo), "high" (hi) and "most likely" (ML). ML is the value used for the main results in this study. The groups of parameters subjected to the sensitivity analyses are as follows:

[ix] *In the CAPRI baseline, about 4% of the revenues of beef and ruminants in the EU are VCS, whereas only 0.8% of the revenues in dairy are VCS. Regionally and locally the shares can be much larger, since some regions like Germany apply no VCS at all.*

- The elasticities of supply (SupElas) of ruminants in the EU are influenced by the slope of the marginal cost function.[x] Higher slope means lower supply elasticity and *vice versa*. The slope was varied +/− 50% to create the lo and hi scenario variants.
- The elasticities of demand (DemElas) for meat and dairy products. We recalibrated the demand systems for all countries so that the own-price demand elasticities would be as close as possible to +/− 50% of the standard value, while observing relevant regularity conditions for demand systems.
- Substitution elasticities (CES) between imports and domestic products and between different import sources were also set to +/− 50% of the standard values. The standard values differ per product, ranging from 2 to 10.
- GHG emission factors (EF) per commodity outside of the EU. Emissions leakage depends more on the relationship between EF in the EU to those outside the EU than on the absolute level. Therefore, we chose to vary only the factors outside of the EU. Since, in general, N_2O factors are considered less certain than emissions of CH_4, which in turn are less certain than CO_2, we chose to apply the uncertainty ranges indicated in a recent IPCC report (Blanco et al. 2014, p 363) to construct the hi and lo scenarios. These ranges were +/− 60% for N_2O and +/− 20% for CH_4.

We do not know the covariance of the uncertain parameters across regions and products. In order to avoid running a very large number of simulation experiments, we chose to vary the parameters for all products and regions in concert by setting all parameters of the same type to lo/ML/hi simultaneously. For instance, we set the demand elasticities of products in all countries simultaneously to hi, ML or lo, giving just 3 demand settings instead of thousands, and similar for the other parameters in the sensitivity analysis. We thus obtain $3 \times 3 \times 3 \times 3 = 81$ result sets; this should span the extremes of the result space.

Results

Global Changes in Emissions of GHG from Agriculture

When VCS for ruminants was removed, emissions in the EU, but also outside the EU, were affected. Figure 1 shows differences in agricultural GHG emissions in thousand ton (kt) CO_2-eq. between the policy scenario and the reference scenario (*i.e.*, the simulated impacts of removing VCS) for 2030. The asterisks (*) in the top panel show the results with standard (ML) parameter settings. With the policy change, the GHG emissions in the EU decreased by 2,354 kt. However, there was an emissions leakage effect, as emissions in the rest of the world increased by 1,738 kt. This resulted in a net decrease on a global basis of 616 kt, or approximately 26% of the emissions decrease in the EU.

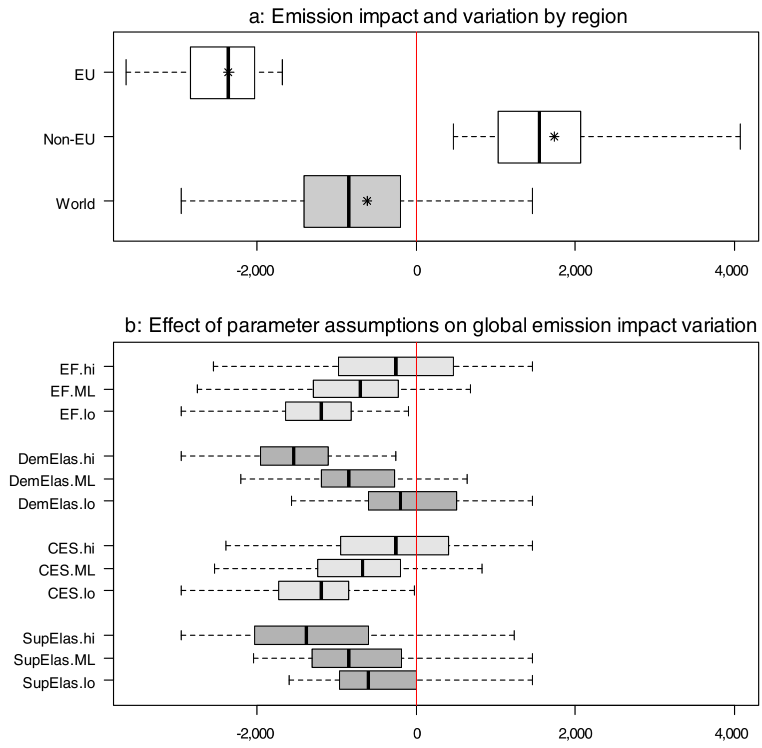
The boxes in figure 1 indicate sensitivity with respect to the four groups of parameters: supply elasticities (SupElas), demand elasticities (DemElas), import substitution elasticities (CES), and emission factors of non-EU regions (EF). The sensitivity analyses in Panel A show that the emissions in the major regions analyzed (EU, non-EU, World) depend strongly on parameters of the model, so that our results on global emissions change could be larger or

[x]CAPRI contains quadratic cost functions in the tradition of Positive Mathematical Programming (PMP). In the sensitivity analyses, we varied the coefficient of the quadratic term.

smaller, with the extreme outcomes for the "World" region ranging from −2,956 kt to +1,465 kt. There are more outcomes in the lower range than in the higher range, as indicated by the median line being to the left of the asterisk.

The results seem about equally sensitive to variations in the four parameters, yet the disaggregation in Panel B allows some general conclusions. Each box in Panel B shows the variation of global emissions (*i.e.* the shaded box in Panel A) if each group of parameters in turn is fixed at one of the three levels: If the emission factors of the Non-EU regions are at EF.lo (20–60% lower than standard), or demand elasticities are at DemElas.hi (50% higher than standard), or the import substitution parameters are at CES.lo (50% less than

Figure 1 Impacts on agricultural GHG emissions, with sensitivity analyses (difference to reference scenario kt CO_2-eq. per year). **Panel A**: Impacts on emissions in the EU, outside of the EU, and in total for the World (vertical axis). The main scenario outcomes, when all parameters set to "most likely" (ML), are indicated with asterisks (*). Each box with whiskers shows the variation in outcomes in 81 sensitivity experiments. The central box covers the two central quartiles, the whiskers indicate extreme values, and the heavy vertical lines in boxes indicate median results. **Panel B**: Each box with whiskers shows the variation of global emissions (the box "World" in panel A) when one group of parameters is fixed at a particular level, indicated at the vertical axis. "EF" = Emission intensities, "DemElas" = Demand elasticities, "CES" = Armington substitution elasticities, "SupElas" = Supply elasticities. "hi", "ML" and "lo" denote each of the three levels (high, most likely, and low) of the parameters that were analyzed in the sensitivity experiments. Each box thus summarizes the result of 27 sensitivity experiments, with box and whiskers defined as in A. [Color figure can be viewed at wileyonlinelibrary.com]



standard), the global emissions change is negative, regardless of how the other parameters are set within the ranges analyzed. The bottom three boxes, showing dependence on supply elasticities within the EU, illustrate how these parameters merely scale the total results, and thus are of importance to the absolute size of the impact, but not to the qualitative results.

Studying the main results in more detail, we find that about 90% of the emissions reduction in the EU derived from production of beef, with an absolute decrease in emissions of 2,088 kt CO_2-eq (Table 1). This was a result of less production, as production in relative terms decreased by 1.1% (see Table 2). As can be seen in Table 1, milk was the largest source of emissions in the EU, but the change in emissions for milk—where VCS is less important—was much smaller than for beef. Emissions from pork and poultry increase

Table 1 Emission Impacts in Major Regions of the World Attributable to Changes in Production of Various Commodities (kt CO_2 eq. per year)

	EU		Non-EU		World	
	Ref	**No VCS**	**Ref**	**No VCS**	**Ref**	**No VCS**
Cereals	35,763	8	261,089	−22	296,853	−14
Oilseeds	8,377	12	58,685	−24	67,062	−13
Other arable field crops	1,312	2	14,784	−2	16,096	−1
Vegetables and Permanent crops	3,312	−1	42,922	0	46,234	−1
All other crops	1,286	1	4,694	0	5,979	1
Beef	129,281	−2,088	2,742,253	1,606	2,871,535	−482
Pork meat	45,295	69	178,796	0	224,091	68
Sheep and goat meat	19,864	−75	652,177	195	672,041	120
Poultry meat	7,612	12	97,375	5	104,986	17
Raw milk	175,299	−305	1,008,638	−8	1,183,938	−313
Eggs	2,751	2	30,310	−1	33,060	1
Secondary products	5,066	9	966,617	−10	971,683	−1

Note: For each region EU, Non-EU and World, the two columns indicate in turn (Ref) the amount of emissions attributable to the commodity groups indicated in the table rows in the reference scenario, and (No VCS) the impact of the policy scenario expressed as difference to reference scenario.

Table 2 Impact of Removal of Voluntary Coupled Support (VCS) for Ruminants on the Beef Market in the European Union

		No VCS	
	Ref	**Difference to Ref**	**% change to Ref**
Production (kt)	7,900	−89	−1.1%
Consumption (kt)	7,955	−50	−0.6%
Import (kt)	781	17	2.2%
Export (kt)	726	−22	−3.1%
Producer price (€ per tonne)	4,367	105	2.4%
Consumer price (€ per tonne)	9,146	105	1.1%

Note: The column Ref shows the situation in the reference scenario. Production, consumption, import and export quantities are given in thousands of tonnes (kt), whereas prices are given in EUR per tonne. The impact in No VCS is given both as difference and as percentage change to Ref.

due to consumers replacing some of the more expensive beef with relatively less expensive pork or poultry. Since emission intensities for poultry and pork are significantly lower than for beef, the emission increase associated with pork and poultry production was small. For crop products, emissions barely changed. The slight increases in emissions associated with arable crops were caused by a larger crop area combined with lower average yields. Feed demand went down and exports increased, leading to the small net increases in emissions in the EU shown in Table 1.

Different products have different sensitivities to emissions leakage. For beef, much of the reduction in the EU was canceled out by increased emissions outside the EU. For sheep and goat meat, there was even an increase in emissions globally, despite the 75kt CO_2-eq. reduction in the EU in the policy scenario. In contrast, the reduction in emissions from milk production in the EU was accompanied by an additional small emissions reduction outside of the EU, caused mostly by a reallocation of production among world regions. For crops, increased exports from the EU replaced production abroad, leading to reduced emissions there and a small net reduction associated with crops globally.

Beef markets merit extra attention, because beef meat was the largest contributor to the change in GHG emissions following the removal of VCS. Table 2 shows changes in the EU beef market. In the policy scenario, beef production in the EU decreased, leading to higher producer and consumer prices for beef meat in the EU. The higher prices dampened the negative impact on production. Production decreased by 89kt, while consumption was rather inelastic and decreased by only 50kt. The balance between decreased production and consumption of beef was maintained by a reduction in exports (−22kt) from the EU, and by increased imports to the EU (+17 kt). This caused production changes in countries outside the EU, driving the results on emissions leakage.

Table 3 shows impacts on production in and trade with the non-EU regions of CAPRI that are most strongly affected. Imports of beef to the EU increased most from the US, while exports from the EU decreased, in particular for

Table 3 Impacts on Production and Trade of Removing Voluntary Coupled Support (VCS) for Ruminants in the European Union (EU) for Selected non-EU Countries and Regions with Large Impacts

Country or region	Ref(kt)			No VCS(difference to Ref, kt)		
	Production	Import	Export	Production	Import	Export
USA	11,627	29	0	5	7	0
Brazil	10,818	75	5	8	1	−1
Russia	1,784	0	46	6	0	−8
Mediterranean[a]	1,028	1	44	2	0	−8
Kazakhstan	449	0	13	1	0	−2
Western Balkans[b]	196	7	25	01	1	−1

[a]*Tunisia, Algeria, Egypt, and Israel.*
[b]*Albania, Macedonia, Serbia, Montenegro, Bosnia and Herzegovina, and Kosovo.*
Note: The reference scenario (Ref) values are in thousand tonnes (kt). For No VCS, the values are differences to Ref (kt).

Table 4 Impacts on Beef Production, the Suckler Cow Herd, Methane Emissions and Total non-CO_2 Emissions After Removing Voluntary Coupled Support (VCS) for Ruminants in the European Union (EU) Including the UK, for Selected Countries with Large Impacts

	Suckler cows[a]		**Beef production**[b]		**Methane emissions**[c]		**Non-CO2 emissions**[c]	
Germany	**23**	**4.7%**	**−1**	**−0.1%**	**21**	**0.1%**	**14**	**0.0%**
Spain	−153	−7.1%	−10	−1.5%	−404	−1.9%	−563	−1.6%
France	−339	−7.1%	−35	−2.0%	−686	−1.6%	−924	−1.2%
Ireland	41	4.2%	2	0.3%	76	0.4%	104	0.4%
Italy	−16	−5.8%	−6	−0.9%	−75	−0.4%	−101	−0.3%
Portugal	−53	−9.1%	−2	−1.3%	−152	−3.0%	−213	−2.8%
United Kingdom	73	5.5%	4	0.5%	122	0.4%	156	0.3%
Poland	−14	−4.4%	−6	−1.3%	−118	−0.9%	−167	−0.5%

[a]*Thousand animals and percentage change* vs *baseline.*
[b]*Thousand tonnes and percentage change* vs *baseline.*
[c]*Thousand tonnes CO_2 eq. and percentage change* vs *baseline.*

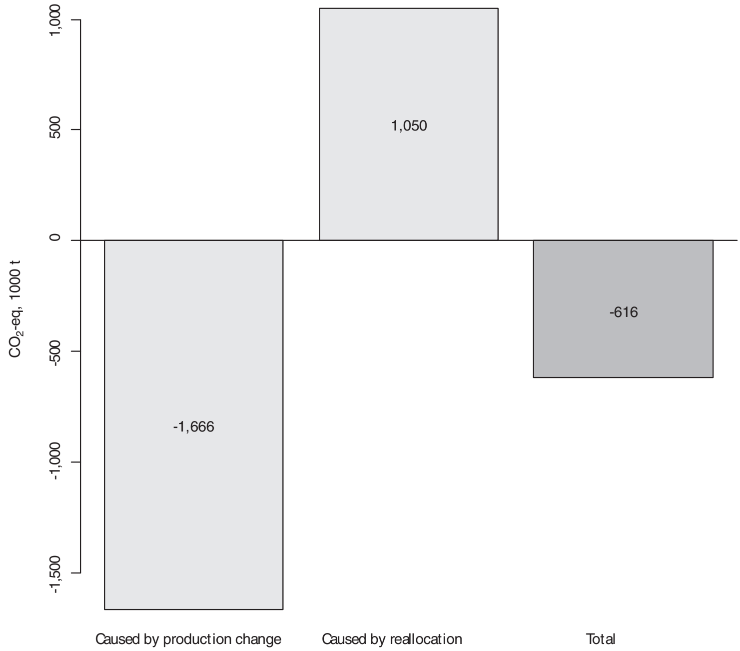
Mediterranean countries and Russia. The latter was met by a production increase in Russia. Argentina and Brazil remained the main trading partners, but their exports to the EU did not change greatly. Instead, changing world market prices affected their trade with other parts of the world, resulting in large production increases in Brazil. Other regions outside the EU also changed their production and trade relations. India's production and exports increased slightly, which had a large effect on global emissions, since Indian production is relatively emissions intensive.

Within the EU, the largest decreases in GHG emissions in absolute terms were found in France, Spain, and Poland. In contrast, in the United Kingdom, Ireland, and Germany, where ruminant production receives little or no VCS, GHG emissions increased, since in these regions ruminant herds slightly increased in response to the higher prices. Table 4 shows the changes in the suckler cow[xi] herd, beef production, methane, and total non-CO_2 GHG emissions in the EU countries with the largest absolute changes in the latter. The increase in emissions in the countries with expanding ruminant sectors might be considered a case of "intracommunity leakage" where the GHG-saving effects in the majority of countries are counteracted by emission increases in others.

In our computations, removing VCS to ruminants increases agricultural incomes in the EU by about €1,400 million annually.[xii] The income increase is due to two things: Firstly, the VCS funds are transferred to the basic farm payment, where it tops up income without requiring additional variable costs, *i.e.*, animals that were unprofitable without subsidies are no longer produced, while the subsidy is still obtained. Secondly, the prices of some animal products rise and thus raise farm incomes. There is reduction in consumer welfare of €868 million annually due to the higher prices. The impact on tax

[xi]*Cows rearing calves for beef production*
[xii]*In CAPRI this is computed as gross value added plus subsidies,* i.e. *the total amount available for remuneration of capital and labor.*

Figure 2 Global changes in greenhouse gas emissions in 1000 tonnes annually, decomposed into those caused by production and those caused by differences in emission intensity in producing countries



payers is negligible, €40+ million spending, since the total CAP budget is unchanged. Thus, we might expect a gain in welfare in the EU across these three groups totaling €492 million euro annually.

Decomposition of Emissions Leakage

The results show that abolishing VCS to ruminants would reduce global agricultural GHG emissions due to the reallocation of production. To gain insights into this process, we decomposed the changes in emissions. The obvious reason for increases in emissions outside the EU is increased production of beef in countries outside the EU. Another reason is that production is more or less intense in terms of GHG emissions in different locations, which means that reallocation of production has an impact on emissions. In addition, changing conditions may alter production technology, which could affect the emission intensity of a product. In our simulations, these technological changes were only modeled endogenously for EU+ countries.

The disaggregation of emissions changes for beef resulting from production volume and reallocation effects are presented in figure 2. The bar to the left shows the emissions changes that would have occurred if the average emission intensity in the world (from the reference scenario) applied to all regions, while the production changes remained the same. This emissions change can be attributed to the change in global production volume. The reduction in production would thus have reduced global emissions by 1,666 kt CO_2-eq. However, the actual emissions reduction globally was 616 kt

CO_2-eq., which is 1,050 kt less than the emissions reduction brought about by production level changes. This discrepancy is explained by the reallocation of production to locations with higher emission intensity than the EU.

Summary and Conclusions

This study used the simulation model CAPRI to analyze impacts of the current voluntary coupled support for the ruminant sectors in the EU on GHG emissions in the EU and globally. Our results show that removing VCS of ruminants in the EU may lead to an emissions reduction of −2,354kt CO_2 eq. annually, corresponding to −0.5% of total agricultural GHG emissions in the EU. However, about three-quarters of this reduction would be canceled out by emissions leakage (*i.e.*, increased emissions outside the EU).

Inelastic demand and opportunities to trade would cause a shift in production from the EU to other countries, and hence the higher emissions outside the EU. In addition to the impact on emissions caused by higher production volumes outside the EU, emissions leakage is further magnified by the emissions-intensive production methods used in countries where production might expand (*e.g.*, Brazil and India). This illustrates one of the problems with a unilateral policy and policies mainly affecting EU production volumes rather than production technologies and consumption. Emissions leakage means that in order to attain a specific global reduction in emissions, unilateral local policies would have to reduce local emissions to a much larger extent than indicated by the global reduction target.

Furthermore, the emissions leakage would vary across product categories. For example, the global emissions for goat and sheep meat would increase even though EU emissions declined. For beef meat, the global emissions reduction would be about 23% of the emissions reduction in the EU, while for milk the global emissions reduction would be even slightly larger than in the EU. This indicates that production subsidies for some products may cause more harm to climate efforts than subsidies to others depending on trade relations and relative emission intensities, but further research on specific products is required to form a solid base for policy decisions.

Our analysis also entailed a sensitivity analysis of how key results depend on selected model parameters. Demand elasticities, emission intensities, and the preferences for domestic as opposed to imported food all influence the results strongly, although our main results are stable for the bulk of the sensitivity analysis outcomes. Despite uncertainties when pushing critical parameters far, our results clearly stress the importance of keeping emissions leakage in mind when designing policies. They also show that subsidies to the emissions-intensive ruminant segment of agriculture can exacerbate climate change. Compared with other studies on EU agriculture, the leakage effect in our analysis was quite modest, which might be a particularity of the VCS instrument. For example, Fellmann et al. (2018) found that emissions leakage effects reduced the impact of more general policies to reduce EU agricultural emissions by as much as 91%, of which about 90% was attributable to cattle. Van Doorslaer et al. (2015), also using CAPRI, found that unilateral policies aimed at reducing emission intensities via improved production technologies generally led to less leakage than policies setting reduction targets achieved mainly by reduced production. They also found that for more ambitious mitigation targets the leakage is generally larger, and thus the cost of achieving a global emissions reduction target using unilateral policies would increase with the level of ambition in emissions reduction targets.

A reduction in global emissions, albeit small and despite leakage effects, achieved by *not* subsidizing a polluting industry might be an efficient contribution to climate policy, since shifting coupled subsidies to decoupled subsidies may be expected to improve efficiency in the economy, and thus improve overall welfare. If the combined welfare[xiii] change for agricultural producers, consumers, and tax payers (€494+ million annually) is divided by the reduction in emissions in the EU (2,354kt annually), we find that each tonne of emission reduction is associated with a social *benefit* of €209 per tonne on average. However, the reduction in emissions achieved should also be viewed in the context of conflicting policy objectives. The stated policy objective for VCS is to maintain important and vulnerable agricultural subsectors (European Commission 2017b). The scheme can be perceived as successful in this regard, as our results clearly showed that removal of the subsidy would cause a decline in production. Whether the potential benefits of VCS for ruminants in terms of maintaining production in the EU justify the negative impact on the climate is a political question that should be a key element in evaluation of the policy.

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[xiii]Producer welfare of farmers is measured in CAPRI by the increase in gross value added, while consumer welfare derives from the money metric related to the demand system in CAPRI. Welfare for non-EU regions is measured differently in some details in CAPRI such that global welfare is not reported to avoid problematic aggregations

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Acta Universitatis Agriculturae Sueciae

Doctoral Thesis No. 2024:8

This thesis investigates how agricultural land can be used for cost-effective abatement of greenhouse gas emissions. The impact of spatial relationships and characteristics of land on the costs of producing biofuel for the purpose of emissions abatement in the transport sector, and on global emissions caused by agricultural policies, are assessed using spatial economic models.

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