Strategies for Sustainable Livestock production in Brazil and the European Union

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DECLARATION

This thesis is the result of my own work and includes nothing which is the outcome of work done in collaboration except as declared in the Preface and specified in the text. I further declare that it is not substantially the same as any that I have submitted, or, is being concurrently submitted for a degree or diploma or other qualification at the University of Cambridge or any other University or similar institution except as declared in the Preface and specified in the text. I further state that no substantial part of my dissertation has already been submitted, or, is being concurrently submitted for any such degree, diploma or other qualification at the University of Cambridge or any other University of similar institution except as declared in the Preface and specified in the text

This thesis is does not exceed 60,000 words (excluding appendices and references).

Signed:

Erasmus Klaus Helge Justus zu Ermgassen

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SUMMARY

Livestock provide as much as one-third of all protein consumed by humans, but have a disproportionate and growing environmental impact. Livestock production occupies 50-75% of agricultural land, contributes 15% of anthropogenic greenhouse gas emissions, and drives agricultural expansion in the tropics through the global trade in animal feed. This thesis therefore evaluates two strategies for shrinking the environmental impact of the livestock sector.

First, I evaluate the potential for food losses (i.e. foods which were intended for human consumption, but which ultimately are not directly eaten by people) to replace grain- and soybean-based pig feeds in Europe. While food losses have been included in animal feed for millennia, the practice is all but banned in the European Union, because of disease control concerns. Several East Asian states have in the last 20 years, however, introduced regulated systems for safely recycling food losses into animal feed. I combine data from multiple sources (including government reports, the animal science literature, and factory-floor data from South Korean swill-feed factories), and find that the introduction of East Asian practices for recycling food losses as animal feed could reduce the land use of EU pork (20% of world production) by one fifth, potentially saving 1.8 million hectares of agricultural land. This would also reduce 12/14 other assessed environmental impacts and deliver economic savings for pig farmers, as swill (cooked food losses) costs 40-60% less than conventional grain-based feeds. In a survey of pig farmers (n=82) and other agricultural stakeholders (n=81) at a UK agricultural trade fair, we found high support (>75%) for the relegalisation of swill. Support for swill feeding arose in part because respondents thought that swill would lower costs, increase profitability, and be better for the environment. Our results also confirmed the critical importance of disease control and consumer communication when considering relegalisation, as respondents who thought that swill would increase disease risks and be unpalatable to consumers were less supportive of relegalisation. Any new system for the use of swill will require careful design of regulation to ensure that heat-treatment is sufficient, and to reduce to a negligible level the risk of uncooked animal by-products entering feed. Our results suggest, however, that if such a system can be established, there would be multiple benefits and widespread support for its relegalisation.

Second, I evaluated the potential to increase the productivity of cattle ranching in the Brazilian Amazon. While high hopes have been placed on the potential for intensification of low-productivity cattle ranching to spare land for other agricultural uses, cattle productivity in the Amazon biome (29% of the Brazilian cattle herd) remains stubbornly low, and it is not clear how to realize theoretical productivity gains in practice. I therefore (a) surveyed six on-the-ground initiatives which have been working with local farmers to improve cattle ranching, quantifying their farm practices, animal performance, and economic results; and (b) analysed the progress that has already been made in reconciling agriculture and forest conservation, by evaluating the impact of the flagship anti-deforestation policy, the priority list (Municípios Prioritários). The survey showed that cattle intensification initiatives operating in four states have used a wide range of technologies to improve productivity by 30-490%, while supporting compliance with the Brazilian Forest Code. Using two complementary difference-in-difference estimators, I then found no evidence for trade-offs between agriculture and forest conservation under the priority list; instead, reductions in deforestation in priority list municipalities were paired with increases in cattle production and productivity (cattle/hectare). The policy had no effect on dairy or crop production. Together, these results provide real-world evidence that increases in cattle production in Brazil do not need to come at the expense of the country's remaining native vegetation.

ACKNOWLEDGEMENTS

This thesis technically lasted three years, though it feels like the culmination of ten years of university study. During that time, I've called a few places home: I am grateful to the atmosphere found in Selwyn college, my home for three years; Grozier Road and Simmons Hall, my homes for one year in Massachusetts; 257 Campkin Road, the site of innumerable morale-boosting meals; plus the various haunts I shared with Rodrigo over five years. We survived radiator accidents, landlords who creep in the window at night, and many questionable (waste-minimizing) meals. You helped me learn that Kohlrabi *can* be purple, and that Chinese vinegar is a poor substitute for soy sauce. I also thank Andrew, Ellie, and Juan, who never hesitated to house me during the "wandering years".

During those ten years of study, I have been fortunate to have worked under several scientific role models. First, I thank my supervisor Andrew Balmford, for accepting and mentoring a veterinary student armed with an introductory course in ecology and no knowledge of linear models. I thank you both for your academic guidance and for supporting my independence, as I shuffled between Cambridge, Brazil, Laxenburg, London, and Belgium. Second, I thank Agnieszka Latawiec and Bernardo Strassburg for setting me up in Brazil and always making me feel welcome in Rio. Third, I thank Aline Mosnier and Alvaro Irribarem for hosting me in IIASA and I thank Georgina Mace and Patrick Meyfroidt for letting me drop in and out of the two UCLs. Lastly, I am grateful to Becki Lawson, Chinatsu Mukai, Alexander Travis, Line zu Ermgassen and the TBA team in Kibale (in particular Walter Hödl, Adolfo Amezquita, and Jette Knudsen) for giving up their time for the otherwise unrewarded task of supervising undergraduate projects. You could have done it faster alone. The exposure to research you gave me, however, led directly to this thesis.

I found that fieldwork can be a lonely and Portuguese-heavy place. I am grateful to Geraldo and his family for welcoming and guiding me with good humour and clear enunciation, to Bruno for opening the doors of his home to me, and to Renan and Bradley Davis for doing much of the work on-the-ground.

Unlike fieldwork, data analysis is never lonely. Whenever I ran into a statistics or programming problem, I turned to (Jon and) the online open source community. Their shared resources, built on the selfless contributions of innumerous people, make the world a smarter place.

The most enjoyable part of this thesis has been the collaborations and friendships that I have built along the way. I learnt huge amounts from Rami Salemdeeb, Mi Hyung Kim, Nicolas Koch, Johanna Wehkamp, Gregor Schwerhoff, Francisco Oliveira, Ellie Bladon, and Moira Kelly, and am proud to think that we completed projects together we wouldn't have been able to do alone.

I am similarly grateful to members of the Conservation Science Group (in particular Ben Phalan, Anthony Lamb, David Williams, and Rhys Green) for their time, shared insight, and collaboration.

This thesis would of course not have been possible without research funding – I am very appreciative of support from the BBSRC (my main source of funding), the Tim Whitmore Fund, the Cambridge Global Food Security initiative, and the German NMO.

On a more personal note, I cannot express enough gratitude and awe for my family, who have always supported my Peter Pan syndrome. I think I'm ready to graduate now. Tusind tak (in order of decreasing height – apologies if you're taller than I think you are) Far, Jon, Kelly, Rory, Sophus, Dave, Mark, Orla, Line, Chrissy, Rikke, Imme, Stine, Mor, Maggs, Adele, Jesper, Jonas, Elise, Magnus, Olaf, and Trine. You are the most incredible, thoughtful, and caring people I have ever met. Thank you for being my family.

Finally, to Moira, you endured the never ending blow-by-blow account of how my day, and thesis, has been going. By then end, I am sure you could have written most of this doctorate yourself. Thank you for always listening, thank you for giving me direction, thank you for being my wife.

A NOTE ON COLLABORATION

Each of the projects I undertook was collaborative in nature. In particular, chapters 3 and 6 were joint-first authored with Rami Salemdeeb and Nicolas Koch, respectively. The contributions of collaborators to each chapter are summarised below.

Chapter 2: [This chapter is published in Food Policy, https://doi.org/10.1016/j.foodpol.2015.11.001]. I am first author of this manuscript. I designed the study, collected the data, analysed the data, and led on the writing of the manuscript. The work was carried out in collaboration with three co-authors: Ben Phalan, Rhys Green, and Andrew Balmford, who provided comments on the manuscript.

Chapter 3: [This chapter is published in the Journal of Cleaner Production, https://doi.org/10.1016/j.jclepro.2016.05.049]. I am joint-first author of this manuscript. I co-designed the study, co-led the project management, provided data on European animal feed production, processed the data on Korean Ecofeed facilities, conducted the statistical analysis, produced all figures (except Figure 3.1), and led on the writing of the manuscript. The work was carried out in collaboration with four co-authors: Rami Salemdeeb (joint-first author), Mi Hyung Kim, Andrew Balmford, and Abir Al-Tabbaa. Rami Salemdeeb collated other life cycle assessment data and built and ran the hybrid life cycle assessment model; Rami's work on this chapter was also included in his PhD thesis, submitted to the Department of Engineering, University of Cambridge (May 2017). Mi Hyung Kim contributed data on Ecofeed facilities, and all co-authors provided comments on the manuscript.

Chapter 4: [This chapter is in review in PLOS One]. I am first author of this manuscript. I designed the study, wrote the survey, collected the data (together with other co-authors), analysed the data, and led on the writing of the manuscript. This work was carried out in collaboration with four co-authors: Moira Kelly, Eleanor Bladon, Rami Salemdeeb, and Andrew Balmford. Moira Kelly, Eleanor Bladon, and Rami Salemdeeb helped collect data and all co-authors provided comments on the manuscript.

Chapter 5: [This chapter is in review in Sustainability]. I am first author of this manuscript. I co-designed the study, wrote the survey, collected the data, analysed the data, and led on the writing of the manuscript. This work was carried out in collaboration with 15 co-authors: Melquesedek Pereira de Alcântara, Andrew Balmford, Luis Barioni, Francisco Beduschi Neto, Murilo M F Bettarello, Genivaldo de Brito, Gabriel C. Carrero, Eduardo de A.S. Florence, Edenise Garcia, Eduardo Trevisan Gonçalves, Casio Trajano da Luz, Giovanni M. Mallmann, Judson F. Valentim, Bernardo B.N. Strassburg, and Agnieszka Latawiec. The latter two authors (B.B.N.S. and A.L.) co-developed the idea of summarising results from intensification initiatives in the Amazon, M.P. de A., F.B.N., M.M.F.B., G. de B., G.C.C., E. de A.S.F., E.G., E.T.G., C.T. da L., G.M.M., and J.F.V., provided data, and all co-authors provided comments on the manuscript.

Chapter 6: [This chapter is in review in the American Journal of Agricultural Economics]. I am joint-first author of this manuscript. I co-designed the study, led the project management, collected and processed the input data, produced all plots, and co-led on the writing of the manuscript. This work was carried out in collaboration with four co-authors: Nicolas Koch (joint-first author), Johanna Wehkamp, Francisco Oliveira, and Gregor Schwerhoff. The concept and methods of the study were developed together with Johanna Wehkamp and Nicolas Koch. Nicolas Koch fit the difference-in-difference estimators and co-led the writing of the manuscript. The conceptual model was developed together with Johanna Wehkamp and Gregor Schwerhoff, and Francisco Oliveira provided additional data on the roll-out of the priority list and credit availability. All co-authors provided comments on the manuscript.

CONTENTS

List of Tables

List of Figures

List of Definitions and Acronyms

Anaerobic Digestion Fermentation of vegetable biomass (e.g. food waste) in the absence of oxygen, often with an aim to producing biogas for use as a fuel.

Animal by-products Materials of animal origin that people do not consume e.g. tendons, processed animal proteins.

BSE Bovine Spongiform Encephalopathy, i.e. mad cow disease.

DID Difference-in-difference.

EU European Union.

Food losses Foods which were intended for human consumption, but which ultimately are not directly eaten by people.

Food waste Food losses which are not legally permitted in livestock feed in the European Union, e.g. catering wastes.

Former foodstuffs Food losses which can legally be fed to livestock in the European union, e.g. biscuit crumbs from biscuit factories, or bread.

GAP Good Agricultural Practices (BPA, in Portuguese) a voluntary set of "gold-standard" guidelines for sustainable cattle production promoted by the Brazilian Agricultural Research Corporation, EMBRAPA.

GDP Gross Domestic Product.

GHG Greenhouse gases.

HP-AI Highly Pathogenic Avian Influenza.

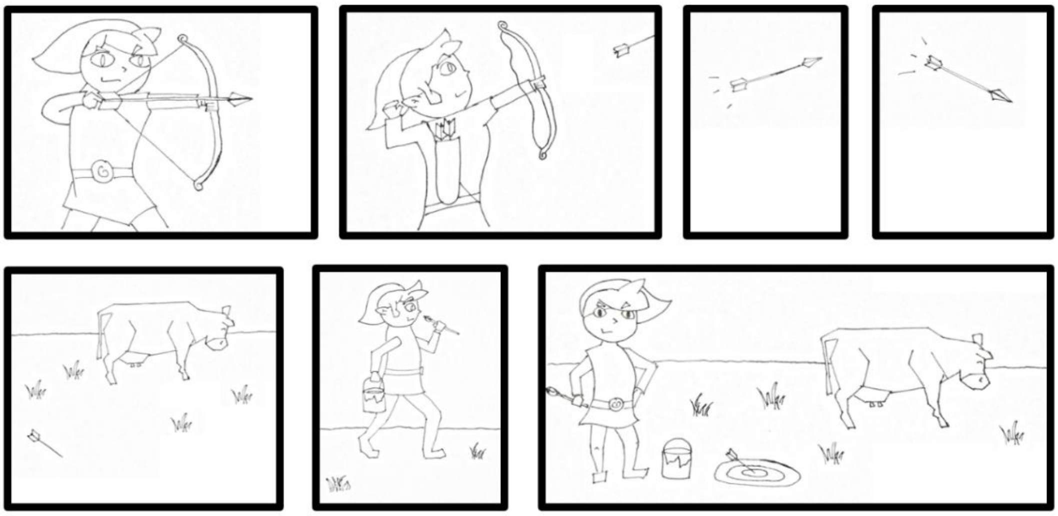
Mha Million hectares.

Monogastric Mammals, like pigs, which have a simple single-chambered stomach (unlike ruminants, which have a multi-chambered stomach and can readily digest forage).

PPM The annual municipal survey of Brazilian livestock (Pesquisa Pecuária Municipial)

TSE Transmissible Spongiform Encephalopathy, i.e. prion diseases, including BSE.

List of Appendices



The PhD process. Source: author's own calculations.

1. Introduction

1.1. Livestock, livelihoods, and the environment

Livestock play a key role in the food system and the global economy. Meat and dairy products make up 17-18% of kilocalorie and 25-33% of protein consumption globally (with large differences between rich and poor countries; Mottet et al., 2017; Rosegrant et al., 2009). The supply chains which connect livestock products to consumers employ around 1.3 billion people and directly support the livelihoods of 600 million smallholder farmers in the developing world (Herrero et al., 2009). Livestock are an important financial asset, with a global value exceeding $1.4 trillion (Herrero et al., 2009), act as a source of insurance for vulnerable communities (Anagol et al., 2013), and, of course, hold important cultural significance for many societies (Gandini and Villa, 2003; Thornton, 2010).

Livestock production, however, also has a disproportionate environmental impact, which makes future growth in the livestock sector problematic. While acknowledging that livestock can utilize resources, such as food losses and low-quality grazing land, which cannot otherwise be farmed (Fairlie, 2010; Godfray et al., 2010; H. H. E. van Zanten et al., 2015), there is an inherent inefficiency in consuming animal source foods at a higher trophic level than their plant-based alternatives (Alexander et al., 2017; Bonhommeau et al., 2013). Livestock production occupies 50-75% of agricultural land (Foley et al., 2011; Mottet et al., 2017), contributes 15% of anthropogenic greenhouse gas emissions (Gerber et al., 2013), is a leading source of nitrogen and phosphorus pollution (Bouwman et al., 2013; Liu et al., 2017), and drives agricultural expansion in the tropics through the global trade in animal feed (Karstensen et al., 2013; Nepstad et al., 2006).

These impacts are set to increase, if global demand for livestock products increases as projected. Increasing demand is driven by population growth – the human population is set to reach 9.6-12.3 billion by the end of the century (Gerland et al., 2014)

– and the trend for increased consumption of animal source foods in developing countries (Bodirsky et al., 2015; Tilman et al., 2011). Under business-as-usual scenarios, the global demand for meat and dairy products will increase 60-80% by 2050 (Alexandratos and Bruinsma, 2012; IAASTD, 2010). With most of this growth in demand occurring in megadiverse countries, where the expansion of agriculture will lead to disproportionately large losses of biodiversity (Machovina et al., 2015; Tilman et al., 2017), and recent recognition that it will not be possible to limit global warming to below two-degrees without changes in the livestock sector (Hedenus et al., 2014), there is a growing appreciation of the need to reduce the environmental impact of livestock production.

1.2. Options for reducing livestock's environmental impact

Strategies for reducing the environmental impact of livestock fall into three broad categories: (1) reducing demand (Bajželj et al., 2014; Erb et al., 2016; Fairlie, 2010; Garnett et al., 2017; Springmann et al., 2016; Tilman and Clark, 2014), principally in the developed world where meat and dairy consumption makes up a high proportion of calorific intake (Micha et al., 2015); (2) increasing production efficiency, i.e. reducing the quantity of feed required per kg of meat or dairy produced (Garnett, 2013; Havlík et al., 2014; Röös et al., 2017; Wirsenius et al., 2010) – while bearing in mind that intensification can lead to reductions in some environmental impacts while increasing others (Davis et al., 2015); and (3) changing livestock diets to lower-impact alternatives. Potential low-impact animal feeds include agricultural by-products (Schader et al., 2015), insects (Makkar et al., 2014), legumes (Jezierny et al., 2010), algae (Holman and Malau-Aduli, 2013), bacteria (Byrne, 2014), and food losses – foods which were intended for human consumption, but which ultimately are not directly eaten by people (Makkar, 2017; Makkar and Ankers, 2014).

In this thesis, I focus on the latter two of these strategies (Figure 1.1). First, I evaluate the potential for food losses to replace grain- and soybean-based pig feeds in Europe. I focus on pigs because they are an omnivorous species with a long history of food loss recycling, though food losses can also be included in the feed of other livestock, including poultry (Boushy et al., 2000; Ruttanavut et al., 2011) and fish (Mo et al., 2018). Second, I evaluate the potential to increase the productivity of cattle ranching in the Brazilian Amazon, a region characterised by low productivity and ongoing deforestation. Below, I provide some context to each of these study systems.

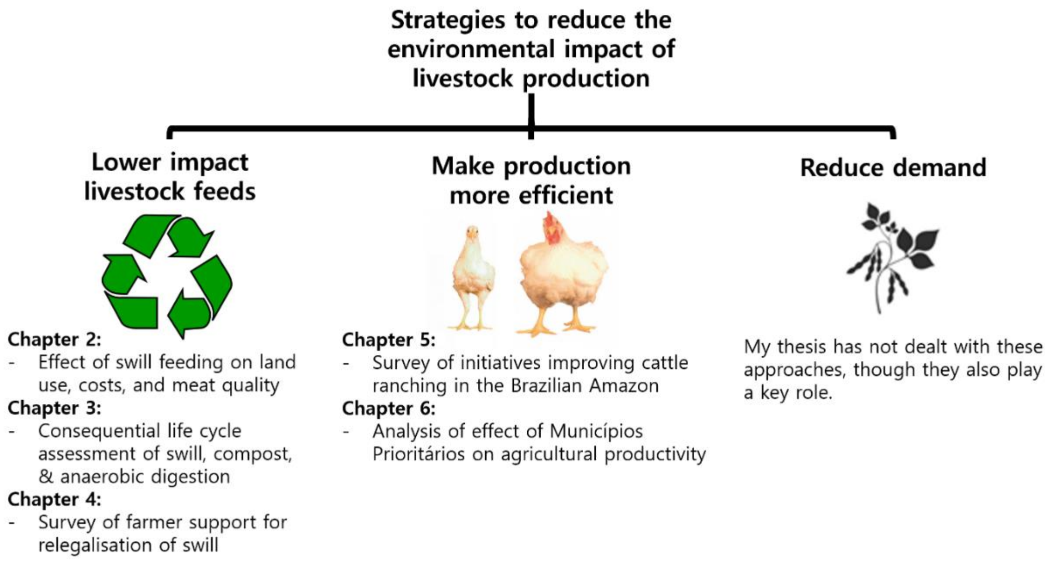


Figure 1.1 – Structure of this thesis. Chapters 2-6 of this thesis analyse two of three broad strategies for reducing the environmental impact of agriculture.

1.3. Pork production in Europe

While food losses have been included in animal feed for millennia (Fairlie, 2010), their use has been in-and-out of fashion in Europe. The use of swill (cooked food losses traditionally fed to pigs) began around nine thousand years ago in Anatolia and the Mekong Valley, modern-day Turkey and China, where wild pigs first raided the piles of food waste left by humans, thereby taking their first steps towards domestication (Pennisi, 2015). Food losses continued to be a mainstay in pig diets until the mid-20th century and were even promoted by national governments during the Second World War as a means of attaining food security (see, for example, the propaganda posters on page 31). The popularity of swill feeding decreased in the late 20th century, however, as the availability of abundant cheap grains led the pig industry to focus on increasing production efficiencies through grain- and soybean-based diets (Fairlie, 2010). By the millennium, only 1.4% of all pigs in the UK were reared on swill (Danby, 2015). The risks of uncooked swill were then demonstrated in 2001 when a UK farmer illegally fed uncooked food waste to pigs, precipitating the 2001 foot-and-mouth disease outbreak, which cost the UK economy approximately £8 billion (UK House of Commons report, 2002) and led to the slaughter of six million animals (four million for disease control purposes; and two million for welfare reasons). In response, swill feeding was banned in the UK in 2001, with the ban extended across the European Union (EU) the following year (EC, 2002).

Today, EU legislation permits the inclusion of only a small subset of food losses in animal feed. For example, all food losses containing animal by-products (i.e. materials of animal origin) are banned, except for those containing honey, eggs, pig or poultry gelatine, milk products, rendered fats, and collagen, where there is no risk of contamination with other sources of animal by-products (EC, 2013a). These legal food losses are known as former foodstuffs, of which 3-5 million tons (3-6% all food losses) are included in animal feed in the EU. All non-former foodstuffs are legally considered as "food wastes" (Trunk, 2016), and throughout this thesis I refer to non-legally permitted food losses as food waste. The legislation specifically bans catering wastes (i.e. food that has been through a home kitchen or restaurant, which make up 57% of food losses in the EU; Stenmarck et al., 2016) and feeds where there is the potential for intra-species recycling – i.e. pigs eating pork products, or chickens eating poultry products. It is notable that the ban on intra-species recycling is made on ethical, rather than scientific grounds. While concern about the inclusion of animal by-products in feed was initially motivated by the Bovine Spongiform Encephalopathy (BSE) crisis in the 1990's (Danby, 2015), there are no recorded cases of pigs, poultry, or fish ever naturally developing or transmitting prion diseases such as BSE (Andreoletti et al., 2007). The ban on intra-species recycling was instead justified by the EU's Economic and Social Committee "for ethical reasons and out of respect for the very nature of animals" (EESC, 2001). As has been noted elsewhere (Danby, 2015; p8), this position ignores that wild pigs and poultry "forage widely in the wild eating almost anything and everything, pigs are even known to eat other dead pigs. Pigs and poultry have evolved to eat animals and animal protein."

Under these regulations, EU pig farmers produce more than 20% of world pork, 34 kg of pork meat/EU citizen/year (FAO, 2014a). They rely, however, on grain- and soybean-based feeds, which have a sizeable environmental footprint. The negative environmental externalities of pork production exceed the other costs of production, with an estimated €1.9 of damage to the environment (from eutrophication, acidification, land use, and greenhouse gas emissions) per kg of pork produced, compared with the ca. €1.4/kg cost of production (Nguyen et al., 2012). Most (75.4%) of this environmental burden stems from feed production – in particular, the farming of soybean meal, the majority of which is imported from Latin America, where soy expansion is associated with large-scale habitat loss and greenhouse gas emissions (Graesser et al., 2015; Noojipady et al., 2017).

Amid recent calls to promote a circular economy (European Commission, 2015) and concern about the environmental impacts of livestock, there is growing recognition that the ban on the use of food waste was not the only possible policy response. While swill is banned in the EU, its regulated use is permitted in many countries, including Japan, South Korea, Taiwan, Thailand, the USA, and New Zealand (Danby, 2015; Leib et al., 2016; Lin et al., 2011; Menikpura et al., 2013). Some of these countries (Japan and South Korea) safely recycle close to 40% of their food losses as feed (notably, recent outbreaks of food-and-mouth in these countries have not been linked to swill feeding; they are thought to have been spread by farm workers returning from regions where the disease is endemic or else via wind-borne spread from North Korea; Muroga et al., 2012; Park et al., 2013; Yonhap News Agency, 2017). Disease outbreaks with swill feeding are not inevitable, if the correct procedures and monitoring are in place: a UK government risk assessment concluded that heat-treatment is sufficient to render food losses containing animal by-products safe for animal feed. The risk for the introduction of diseases principally comes from the contamination of feed with material that evades heat treatment (Adkin et al., 2014). The 2001 foot-and-mouth epidemic, for example, was arguably as much the result of poor enforcement as of farm malpractice. The farm where the outbreak originated was visited twice a year by a State Veterinary Service representative, who visited the farm less than a month before the disease broke out and renewed the farm's license to feed swill, despite suggestions that there was clear evidence of the use of unprocessed swill (Danby, 2015).

There have therefore been intermittent calls to relegalise the use of food wastes in feed (Fairlie, 2017, 2010; Stuart, 2009; UK parliamentary debate, 2004), including a campaign led by The Pig Idea in the UK (The Pig Idea, 2014). While the calls to lift the ban on swill gained media attention (e.g. Chynoweth, 2013; The Economist, 2013; Tyzack, 2013), there remained little peer-reviewed work on what the impacts of relegalisation would actually be – a topic that I address in this thesis. In chapter 2, I review the regulation that oversees the use of food losses as animal feed in Japan and South Korea, make the economic case for the use of swill, and evaluate the potential for swill feeding to reduce the land use of EU pork production. I then move beyond land use in chapter 3, using life cycle assessment to quantify 14 different measures of the environmental and human health impact of using food waste as swill, compared with other disposal options (composting or anaerobic digestion). Finally, the voice of those most affected by the ban on the use of food losses as feed – namely, pig farmers and workers in the agricultural sector – has been notably absent from the public debate. In chapter 4, I therefore report results from a survey investigating the attitudes within the UK farming community to the resumed use of food losses as feed.

1.4. Beef in Brazil

Cattle ranching in Brazil is big business, but is beset by sustainability challenges. Brazil has more than 200 million cattle (the second largest herd on the globe), producing 9.6 million tonnes (carcass weight equivalents) per year, of which 20% is exported (ABIEC, 2016). The sector employs around 7.5 million people (Hoffmann et al., 2014) and contributes 6.8% of Brazil's GDP (30% of agricultural GDP; ABIEC, 2016). Brazil's livestock (which is dominated by cattle ranching) are also responsible, however, for more than half of Brazil's greenhouse gas emissions and environmental footprint (Bustamante et al., 2012; Salvo et al., 2015) – in large part because of ongoing pasture expansion into native vegetation (Almeida et al., 2016; Beuchle et al., 2015; Oliveira et al., 2017).

Nowhere are questions about sustainable cattle ranching more relevant than in the Brazilian Amazon (Figure 1.2) – the focus of chapters 5 and 6 of this thesis. Cattle were introduced into the Amazon by early European settlers (Furtado, 1963), but the rapid expansion of cattle first began in the 1960s, following the government's provision of favourable tax and credit incentives for ranchers (Bowman et al., 2012). By 2015 the Amazonian cattle herd had grown to 61.5 million cattle (29% of the Brazilian national herd; IBGE, 2015), and much of this growth has come at the expense of forest: 60-80% of Amazonian deforestation has been for cattle ranching (Almeida et al., 2016; Nepstad et al., 2009), though there is ongoing debate about the relative importance of cattle production in deforestation, versus other drivers, notably land tenure insecurity and crop expansion (Barona et al., 2010; Bowman et al., 2012; Richards et al., 2014).

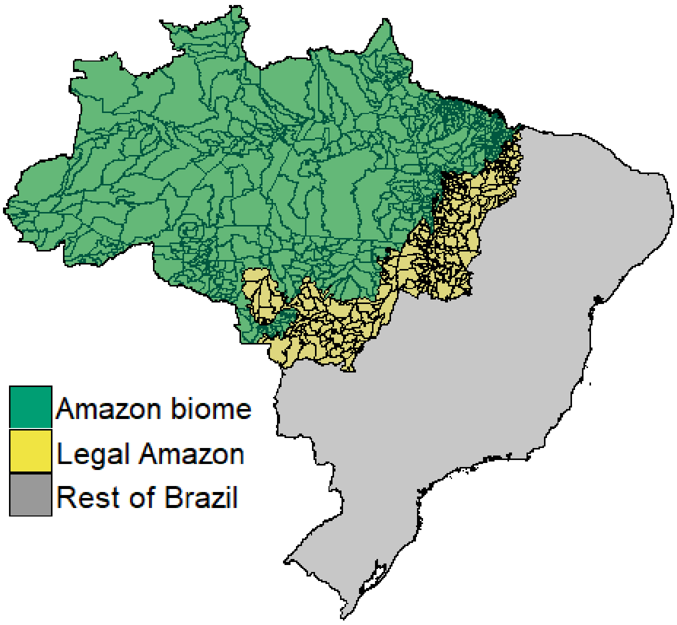


Figure 1.2 – Definitions of the Amazon. This thesis evaluates changes in productivity of cattle ranching in the Brazilian Amazon; in doing so, I make use of two different definitions of the Amazon. In Chapter 5, I report results from six cattle intensification initiatives located within the Amazon biome (green). In chapter 6, my analysis uses data at the municipal level, and I therefore report results for the Legal Amazon (green plus yellow), which is the political boundary of the Amazon, made up of the nine states (771 municipalities) of the Amazon basin.

Regardless of the underlying drivers of deforestation, it is promising that beef production and deforestation in the Amazon have recently uncoupled (Nepstad et al., 2014), as high hopes are placed on increases in cattle productivity (i.e. yields of beef per hectare) to spare land for other agricultural uses and contribute to reductions in Brazil's greenhouse gas emissions. From the mid-2000s onwards, deforestation fell 70% through a combination of increases in enforcement on private land (Assunção and Rocha, 2014; Börner et al., 2015), expansion of protected areas (Nolte et al., 2013; Pfaff et al., 2015;

Soares-Filho et al., 2010), market-initiatives (Gibbs et al., 2015), and an economic slowdown (Assunção et al., 2015), even as the cattle herd increased 14.7% (IBGE, 2015a). Still, the productivity of cattle ranching in the Amazon remains low (Dias et al., 2016; Nepstad et al., 2014) – only one third of its sustainable potential (Strassburg et al., 2014) – and increases in cattle ranching productivity are seen as key to many of Brazil's environmental targets. Theoretical modelling of pasture carrying capacities has, for example, suggested that Brazil can meet the projected demand for beef, crops, and timber until 2040 (including national consumption and exports) without further conversion of natural ecosystems – if the productivity of cattle ranching increases by 55% (Strassburg et al., 2014). It is similarly hoped that cattle intensification will reduce greenhouse gas emissions through land sparing (Cohn et al., 2014), increased soil carbon sequestration (De Oliveira Silva et al., 2017), and increased greenhouse gas intensity (i.e. lower emissions per unit product; Bogaerts et al., 2017).

Despite the inclusion of cattle intensification in both government and industry targets (ABIEC, 2016; Brazil, 2017; MAPA, 2014a), it is not yet clear how Brazil can realize these productivity gains in practice. What technologies are required to boost cattle production? Is intensification cost-effective? And how can adoption of higher-yielding production be fostered? It is these questions that I address in chapters 5 and 6 of this thesis.

In chapter 5, I report results from a survey of six on-the-ground initiatives which have been working with local farmers to turn theory into practice: increasing the productivity of cattle ranching in the Brazilian Amazon. I detail their farm practices, animal performance, and economic results, summarize common successes and challenges, and reflect on the risks and mechanisms for achieving wide-scale productivity gains in cattle ranching across the region.

In chapter 6, I focus on a different mechanism for increasing cattle productivity: reducing farmer's incentives for pasture expansion. While much work on agriculture-environment trade-offs has focused on how yield increases might spare forests from conversion (e.g. Burney et al., 2010; Cohn et al., 2014; Stevenson et al., 2013; Tilman et al., 2011), efforts to constrain agricultural expansion can also potentially induce intensification. While evidence for this mechanism is scarce, it has been proposed for cattle ranching in Latin America (Kaimowitz and Angelsen, 2008; Merry and Soares-Filho, 2017). I therefore analyse the effect of one of Brazil's flagship anti-deforestation policies, the priority list (Municípios Prioritários), on agricultural production and productivity, to determine how farmers responded to increased forest enforcement.

I conclude in Chapter 7 by summarising my findings, and reflecting on how my results contribute toward the broader literature on the environmental impact of livestock, and efforts to make livestock production more sustainable.

Propaganda posters produced by the UK government during the Second World War to promote the use of swill. Posters from: https://commons.wikimedia.org/ and the Imperial War Museum. Credit: © IWM (Art.IWM PST 14743).

2. Reducing the land use of EU pork production: where there's swill, there's a way[1]

2.1. Abstract

Livestock production occupies 50-75% of agricultural land, consumes 35% of the world's grain, and produces 15% of anthropogenic greenhouse gas emissions. With demand for meat and dairy products forecast to increase 60-80% by 2050, there is a pressing need to reduce the footprint of livestock farming. Food losses have a long history as a source of environmentally benign animal feed, but their inclusion in feed is all-but-banned in the EU because of disease control concerns. A number of East Asian states have in the last 20 years, however, introduced regulated, centralised systems for safely recycling food losses into animal feed. This chapter quantifies the land use savings that could be realised by changing EU legislation to promote the use of food losses as animal feed and reviews the policy, public, and industry barriers to the use of food losses as feed. Our results suggest that the application of existing technologies could reduce the land use of EU pork (20% of world production) by one fifth, potentially saving 1.8 million hectares of agricultural land. While swill presents a low-cost, low-impact animal feed, widespread adoption would require efforts to address consumer and farmer concerns over food safety and disease control

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2.2. Introduction

Livestock production has a large and growing environmental impact. While providing one-third of all protein consumed by mankind (Herrero et al., 2009), livestock production occupies 50-75% of agricultural land (Foley et al., 2011; Mottet et al., 2017), contributes 15% of anthropogenic greenhouse gas emissions (Gerber et al., 2013), and drives agricultural expansion in the tropics through the global trade in animal feed (Karstensen et al., 2013; Nepstad et al., 2006). With demand for meat and dairy products forecast to increase 60-80% by 2050 (Alexandratos and Bruinsma, 2012; IAASTD, 2010), there is growing recognition of the need to reduce the environmental impact of meat and dairy production.

Three principal strategies have been proposed to reduce the environmental impact of livestock: (1) reducing demand (Bajželj et al., 2014; Eisler et al., 2014; Fairlie, 2010; zu Ermgassen et al., 2014), principally in the developed world where meat and dairy consumption makes up a high proportion of food intake (Bonhommeau et al., 2013); (2) increasing efficiency, i.e. reducing the quantity of feed required per kg of meat or dairy produced (Garnett, 2013); and (3) changing animal diets to low-impact alternatives. Proposed novel, low-impact animal feeds include insects (Makkar et al., 2014), legumes (Jezierny et al., 2010), algae (Holman and Malau-Aduli, 2013), and bacteria (Byrne, 2014).

Low-impact animal feeds need not, however, be novel. Food losses have historically been recycled as livestock feed, particularly for pigs – cooked food losses fed to pigs is colloquially known as "swill". Pigs are a monogastric species whose digestive system is well adapted for the conversion of food losses into animal protein (Westendorf, 2000a); food losses produced in early human settlements is thought to have attracted wild pigs, leading to their domestication around 9,000 years ago (Fairlie, 2010). Swill can be a high-quality animal feed that requires no additional land to be brought into production, and hence has minimal or even positive environmental impact (food losses otherwise posing a disposal challenge). However, the use of swill is controversial in some countries and there is marked geographic variation in its acceptance and regulation. Though the recycling of food losses as swill is actively promoted in many nations, including South Korea, Japan, Taiwan, and Thailand (Menikpura et al., 2013), it was banned in the European Union (EU) in 2002 after the UK foot-and-mouth disease epidemic, which is thought to have been started by the illegal feeding of uncooked food waste to pigs. Proponents claim that swill is a cheap, environmentally benign animal feed (Fairlie, 2010; Stuart, 2009; Wadhwa and Bakshi, 2013), but critics claim that it is unsafe and produces pork of poor quality (García et al., 2005; House of Lords, 2014).

In this paper we address some of the controversies surrounding the recycling of food losses as animal feed and quantify the potential for food losses to replace conventional animal feed and reduce the environmental impact of meat production. First, we provide an overview of the history and regulation of swill feeding, focusing on the contrasting approaches taken by the EU and two East Asian states: Japan and South Korea. Second, we consider the role that swill can play in reducing the land required for meat production, through a quantitative case-study of pork production in the EU. We then discuss the impact of swill on other environmental impacts, including greenhouse gas emissions, before reviewing the barriers to swill feeding in Europe. We focus on the potential concerns of pig producers, the public, and policy makers. To finish, we briefly discuss the legal status of swill in other parts of the world, focussing on the world's two largest pork producers: the USA and China.

2.3. Swill in the EU, Japan, and South Korea

Although it is the archetypal pig feed, swill has been in and out of fashion in Europe. Swill was the prevalent pig feed in the early 20th century and was actively promoted by the UK government during the Second World War as a means of attaining food security (Fairlie, 2010). The popularity of swill feeding decreased in the late 20th century as the availability of abundant cheap grains led the pig industry to focus on increasing production efficiencies through grain- and soybean-based diets. The risks of uncooked swill were demonstrated in 2001 when a UK farmer illegally fed uncooked food losses to pigs, precipitating the 2001 foot-and-mouth disease outbreak, which cost the UK economy £8 billion (UK House of Commons report, 2002). In response, swill feeding was banned in the UK in 2001, with the ban extended across the EU the following year (EC, 2002). The ban still permits the feeding of food losses which have not been through a kitchen and where it can be demonstrated that there is no risk of contamination with meat products (these food losses are legally considered as "former foodstuffs"), but this represents only a small proportion (ca.3-6%) of all EU food losses (the other 94-97% are treated as a "waste" under EU legislation – see Appendix A for further details).

Today, the EU produces more than 20% of world pork, 34 kg of pork meat/person/year (FAO, 2014a), and relies on grain- and soybean-based feed, which has a sizeable environmental footprint. A life cycle assessment (LCA) of European pork production found that pork production causes €1.9 of damage to the environment (from eutrophication, acidification, land use, and greenhouse gas emissions) per kg of pork produced – in comparison, it costs the farmer on average €1.4 to produce each kg of pork (Nguyen et al., 2012). Most (75.4%) of this environmental burden stems from feed production – in particular, the farming of soybean meal. The expansion of soybean farming in South America to meet international demand for animal feed poses a significant threat to biodiversity and is a large source of carbon emissions from deforestation (Godar et al., 2015; Karstensen et al., 2013; Nepstad et al., 2006; Richards et al., 2014).

Not all modern pig production is reliant on grain and soybean feed. In the same year that the UK banned swill, the Japanese government introduced the opposite policy, promoting the inclusion of food losses in animal feed (Takata et al., 2012). South Korea and Taiwan have introduced similar food loss recycling systems (in 1997 and 2003, respectively). While the feeding of uncooked meat wastes to pigs can transmit diseases including foot-and-mouth and classical swine fever, appropriate heat treatment inactivates these viruses and renders food losses safe for animal feed (Edwards, 2000; OIE, 2009). In these countries, the industry is tightly regulated: the heat treatment of food losses is carried out by registered "Ecofeed" manufacturers (see Appendix B for details of food loss recycling practices in Japan and South Korea). Where Japan and South Korea formerly sent substantial quantities of food losses to landfill, in 2006-07 they respectively recycled 35.9% and 42.5% of food losses as animal feed (Figure 2.1; (Kim and Kim, 2010a; MAFF, 2012a, 2011).

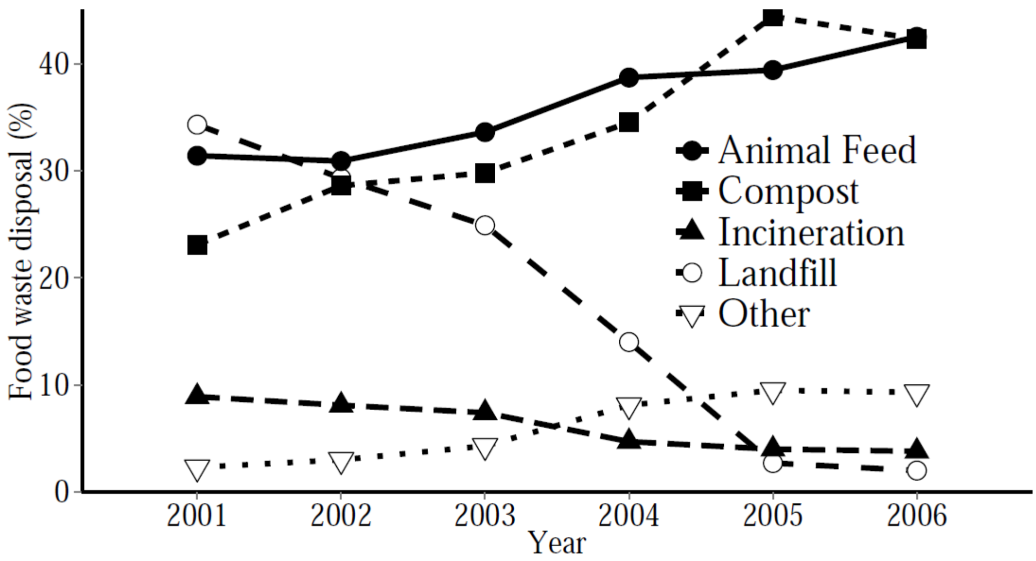


Figure 2.1 – The end-uses of food losses in South Korea 2001-06. After the introduction of food loss recycling legislation in 1997, South Korea achieved substantial increases in food loss recycling. The recycling of food losses for animal feed is shown as a solid line. These are the most recent available data (Kim and Kim, 2010).

2.4. The potential for swill to reduce the land use of EU pork

To estimate the potential land use saving of a change in EU regulation to promote the recycling of food losses as animal feed, we performed three complementary analyses. (a) We estimated the current land use of EU pork production; (b) we used data from feed trials comparing food losses and conventional diets to determine how the incorporation of food losses in pig diets affects the amount of feed and land required for pig production; and (c) we estimated the availability of food losses suitable for pig feed in the EU. We then combined these results to estimate the potential impact of promoting swill on the land use of EU pork production.

In this analysis we use land use as a footprint metric to assess the potential environmental benefits of the re-legalisation and promotion of swill in the EU. While measuring land use alone does not capture all of the environmental impacts of meat production, we consider land use an informative (though incomplete) metric for this analysis because (a) land use represents the majority (55%) of the environmental costs of European pork production (Nguyen et al., 2012); and (b) land use is a valuable indicator of the biodiversity impacts of products (Mattila et al., 2011). While other biodiversity metrics have been used in life cycle assessment (LCA), there remains no consensus on their relative validity (Souza et al., 2015).

2.4.1. The land use of EU pork production

To estimate the land use of EU pork, we calculated the land required across the entire lifecycle of pork production (breeding sows, piglets, and young and mature slaughter pigs) to grow the feed necessary to produce the 21.5 million tonnes of pork (live weight) which is produced in modern, large-scale pig production systems in the EU each year (for more details see Appendix C). The calculation was based upon weighted mean values of EU production statistics (e.g. the number of piglets weaned per sow per year, piglet mortality rates) and representative diets from the five leading producers of pork in the EU: Germany, Spain, Denmark, France, and Poland. These member states together represent >64% of EU pork production (Appendix C, Figure 9.3).

We found relatively little variation in the estimated land use across all five sets of diets (4.02m^2/kg pork; range: 3.6-4.3 m^2/kg) and determined that the land required to grow feed for EU pork was ca. 8.5 million ha (±0.7 Mha s.d.). Soybean production in 2010 represented ca. 15% of the total land area required for EU pig feed production, an area of 1.2 million ha (±0.2 Mha s.d.).

2.4.2. The effect of swill on land required for pig production

To determine how the inclusion of food losses in pig feed influences the land required for pork production, we conducted a comprehensive literature review (Appendix D) to identify 18 feed trials comparing the growth performance of pigs on 23 conventional and 55 food loss-based diets. For each diet, we recorded the proportion of the diet that was food losses (on a dry matter basis) and calculated the land use per kg of pork (Appendix D). We found a strong linear relationship between the land use per kg of pork and the proportion of the diet made up by food losses ($r = 0.97$, $n = 78$, $P < 0.0001$; Figure 2.2).

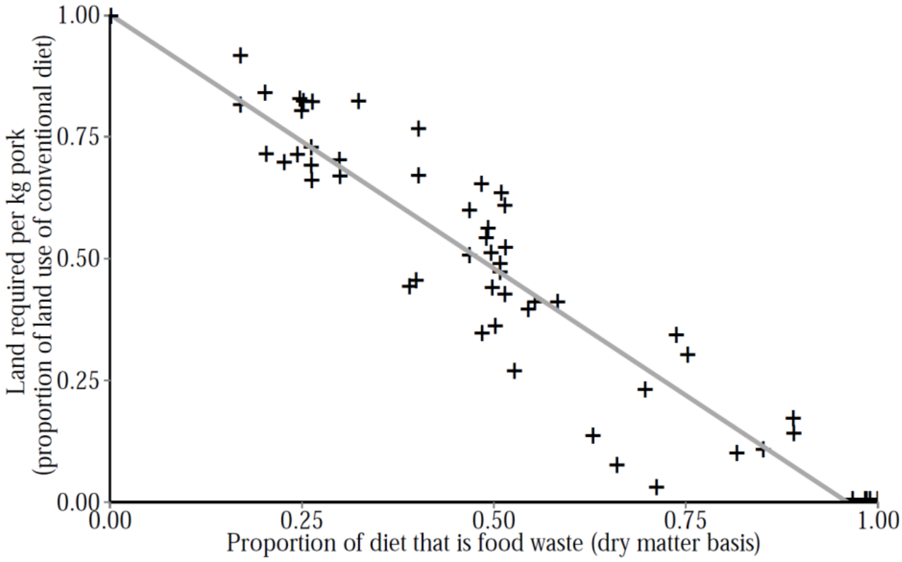


Figure 2.2 – The inclusion of food losses in pig diets linearly reduces the land required per kg of pork live weight; r = 0.97, n = 78, $p < 0.0001$. This linear relationship reflects that the inclusion of food losses in pig feed (a) has no measurable effect the feed conversion efficiency (it substitutes conventional feed almost 1:1 on a dry matter basis ($t = 1.15$, $p = 0.26$)), and (b) does not have a large effect on growth rates (for more details see Appendix D). Some diets have a land use of zero, without being 100% swill: they contain a small amount of other ingredients, such as vitamins and minerals, which also do not require agricultural land.

2.4.3. The availability of food losses in the EU

An estimated 102.5 million tonnes of food were wasted in the EU in 2015 (202 kg per person) (EC, 2010), from four principal waste streams: households (42%), manufacturing (39%), the food service/catering industry (14%), and retail (5% of food losses). These waste streams span the food supply chain post-harvest, and our definition of food losses excludes co-products (Appendix D) and agricultural wastes (i.e. pre-harvest wastes). The estimates of food losses are uncertain because of differing food loss definitions used by member states (e.g. classifications of green wastes), but are the best available data. We believe these figures are conservative estimates of EU food losses because they do not include agricultural wastes, which make up ca. 34% of all European food losses (Kummu et al., 2012), and we therefore used them as lower-bound estimates of the availability of food losses for use as pig feed in the EU.

Before estimating the quantities of food losses available for swill feeding, we made three adjustments. First, we subtracted the 3 million tonnes of manufacturing food losses (i.e. former foodstuffs) that are currently included in livestock feed in the EU (EFFPA, 2014). It is not clear whether these are excluded from the EU food loss data, so subtracting them makes our estimates of food losses available for pig feed conservative. Second, we allow for the fact that not all food losses defined by these statistics is available or suitable for pig feed. Only 35.9% and 42.5% of food losses are converted to animal feed in Japan and South Korea, respectively (Kim and Kim, 2010a; MAFF, 2012a, 2011). We assumed that a similar proportion can be used for the EU and took the mean of these two values (39.2%) to be the proportion of food losses available for recycling into animal feed, if swill feeding were legalised in the EU. Third, in the analyses above (Section 2.4.2) we calculated the proportion of animal feed that is food losses on a dry matter basis. To calculate the proportion of EU pig feed that could be replaced by swill we therefore converted our waste estimates into tonnes of dry matter (Appendix E).

Finally, for comparison with the proposed EU swill-feeding scenario, we also calculated the potential for increasing the use of legal food losses as animal feed under the current legislation. In this scenario, we estimated the land use savings of including in animal feed an estimated 2 million further tonnes of manufacturing food losses which are not currently used for animal feed but which could legally be fed to livestock (EFFPA, 2014).

2.4.4. The potential for swill in the EU

We then used the results from Sections 2.4.1-2.4.3 to estimate the potential for swill to reduce the land use of EU pork production (Appendix F). Our results indicate that if swill feeding were legalised and food losses recycled into animal feed at rates similar to those in Japan and South Korea, the land requirement of EU pork production could shrink by 1.8 million ha (1.7-2.0 Mha; 95% CI), from 8.5 to 6.7 million ha. This represents a 21.5% (19.6-23.5%; 95% CI) reduction in the current land use of industrial EU pork production. In doing so, swill would also replace 8.8 million tonnes of human-edible grains currently fed to pigs (Appendix F) – equivalent to the annual cereal consumption of 70 million EU citizens (FAO, 2014a).

Under the current EU legislation, only a small increase in the quantity of food losses used in animal feed is possible. These legal food losses could reduce land use by

1.2% (1.0-1.4% or 0.08-0.12 million ha; 95% CI). While this legislation stands, efforts to promote the inclusion of former foodstuffs in animal feed should be supported in order to realise these modest improvements in environmental impact; our results suggest, however, that far greater gains could be achieved by re-legalising and promoting the use of swill.

Use of swill might also help reduce the impact of EU pork production on global ecosystems. The inclusion of food losses in pig feed would reduce the area of soybean required by 268,000 ha (0.25 – 0.29 Mha; 95% CI) (Appendix F). In Brazil, the source of the majority (60%) of EU soybean (FAO, 2014a), soybean production is forecast to expand by 10.3 Mha by 2023 (MAPA, 2014b). While Brazil is not the sole source of EU soybean meal, the potential for EU swill-feeding to reduce demand for up to 268,000 hectares of soybean production could mitigate ca. 2.6% of the forecast expansion of soybean, reducing pressure on high-biodiversity tropical biomes accordingly.

2.5. Swill: beyond land use

The substitution of conventional feed with swill has the potential to reduce not only the land requirement for pork production, but also other environmental impacts associated with the production of animal feed, including greenhouse gas emissions and eutrophication. The impacts of swill feeding on these other environmental effects are more difficult to estimate. For greenhouse gas emissions, while eight LCA studies have compared the recycling of food losses into animal feed with other food loss disposal practices (including incineration, anaerobic digestion, and composting), the calculated emissions vary substantially and are sensitive to local conditions and study assumptions (Figure 2.3; Bernstad and la Cour Jansen, 2012). In particular, only one of these studies considers emissions associated with land use change, with the remaining studies therefore underestimating agricultural emissions of feed ingredients, such as soybean meal, by up to nine times (van Middelaar et al., 2013). Two multi-criterion LCAs have been conducted in the European context. Vandermeersch et al. (2014) compare two scenarios in Belgium: (1) sending retail food losses for anaerobic digestion and (2) recycling 10% as animal feed, with the rest sent for anaerobic digestion. This study found that the food loss feeding scenario scored better on 10 of 18 environmental criteria (including land use, marine eutrophication, and freshwater ecotoxicity), with anaerobic digestion scoring better on 8 criteria (including greenhouse gas emissions, ozone depletion, and freshwater eutrophication). Tufvesson et al. (2013) compare the use of manufacturing food losses (such as bread wastes and milk) for biofuel or animal feed in Sweden. They find that the use of these food losses as biofuel only results in environmental benefits (measured by greenhouse gas emissions, eutrophication, and acidification) if you do not take into account their potential use as animal feed. That is to say, they recommend the use of these losses as animal feeds, instead using dedicated biofuel crops for biofuel (though this study did not take into account greenhouse gas emissions from indirect land use change resulting from the expansion of crop-based biofuels, nor the potential use of those biofuel crops as animal feed). As evidenced by the caveats above and the variable results presented in Figure 2.3, the results of LCAs are often location, assumption, and study-dependent (Bernstad and la Cour Jansen, 2012). Future work should therefore analyse swill feeding and other uses of food losses in other EU member states, using alternative sources of food losses, and taking into account all agricultural emissions.

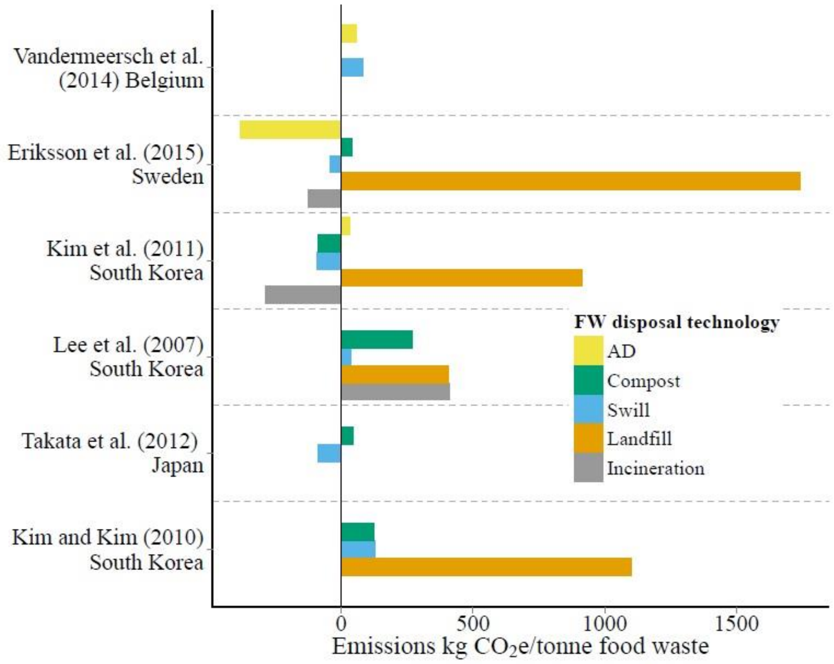


Figure 2.3 – Results of six LCA studies reporting the greenhouse gas emissions per tonne of food losses for different disposal options, including recycling food losses as swill. Negative emissions mean that the process has a net negative carbon balance, ie the emissions avoided are larger than emissions released. Swill, for example, avoids emissions associated with the production of conventional feed, and anaerobic digestion (AD) avoids emissions from the fossil fuels it replaces. Where a study reported emissions for multiple food loss types, the mean emissions are shown, and none of the studies shown include land use change, a major source of agricultural emissions, when calculating the emissions avoided from swill feeding. The swill data for Vandermeersch et al. (2014) are for a 10% swill, 90% anaerobic digestion scenario. Gaps are left where studies did not report particular food loss disposal options, and the country of study is listed under each reference. Two further LCA studies (Ogino et al., 2012; Tufvesson et al., 2013) use different units (reporting results per kg of animal feed and per MJ of fuel energy, rather than per tonne of food loss) and so cannot be displayed for comparison.

2.6. Barriers facing swill in the EU

While our EU-wide analysis is inevitably constrained by the available data, in particular by uncertainty about the quantity of food losses produced in the EU and their nutritional content, we are confident that our principal conclusion is robust: a policy promoting the recycling of food losses as pig feed has substantial potential to reduce the global land use of EU pork production. When selecting animal feeds, however, there are many more considerations than simply their environmental impact. The adoption of swill feeding in the EU would require backing from pig producers, the public, and policy makers. We next consider the potential barriers from each interest group in turn.

2.6.1. Support from pig producers

Pig producers want to produce pork of high quality, at affordable prices, with reliable profit margins, and the highest standards of food safety.

The 18 studies comparing food losses and conventional feed also reported a range of meat quality measures, allowing us to examine the effect of swill feeding on meat quality and palatability. We used linear mixed models to measure the effect of including food losses in animal feed on 18 different measures of meat quality, which were each reported by three or more studies. Since pig age and breed, both important determinants of meat quality, varied among studies, study was included as a random effect. Further details of the methods are listed in Appendix G.

While swill does have more variable nutrient composition than conventional feeds (Westendorf, 2000b), swill feeding had little effect on meat quality, with no effect detected for 16/18 measures (Table 1). The two detected effects were weak and did not detrimentally affect pork quality or value. Pigs fed a 50% swill diet had 1.4% higher monounsaturated fats percentages ($t=3.39$, data from 6 studies, $n = 23$, $p = 0.017$) and 13% greater meat marbling, the presence of streaks of fat within muscle tissue ($t=3.71$, data from 6 studies, $n = 22$, $p = 0.014$). Pork marbling is known to increase the flavour and tenderness of pork (Brewer et al., 2001). Indeed, three studies intentionally fed food loss diets with a low lysine content in order to increase meat marbling (Witte et al., 2000). Removing these three studies from the analysis abolished the effect ($t=-1.24$, data from 3 studies, $n = 10$, $p = 0.32$). These results suggest that the inclusion of food losses in animal diets can produce pork of similar quality to conventional diets, which may allay farmer concerns over product quality.

Table 2-1 – Relationships between the proportion of food losses in pig diets and measures of meat quality.

P-values also shown for quadratic relationships, where suggested in the literature.

Meat quality (range or measurement units)	Number of studies (points)	Coefficient (SE)	p-value	
			Linear model	Quadratic model
Juiciness (0-1)	4 (13)	0.08 (0.04)	0.173	-
Marbling (1-10)	6 (22)	1.30 (0.35)	0.014	-
Dressing percentage (%)	12 (38)	0.89 (0.76)	0.264	-
Meat colour (1-5)	5 (17)	0.21 (0.28)	0.490	-
Meat lightness (L* value)	9 (33)	1.42 (0.81)	0.116	-
Meat redness (a* value)	9 (33)	-0.01 (0.27)	0.983	-
Meat yellowness (b* value)	9 (33)	0.32 (0.28)	0.283	-
Fat lightness (L*value)	7 (29)	0.99 (1.21)	0.443	-
Fat redness (a* value)	7 (29)	-0.08 (0.50)	0.872	-
Fat yellowness (b*value)	7 (29)	-0.39 (0.33)	0.282	-
Fat free lean percentage (%)	4 (15)	1.14 (0.89)	0.280	-
Flavour (0-1)	3 (7)	0.03 (0.02)	0.319	-
Overall palatability (0-1)	3 (7)	0.03 (0.05)	0.584	-
Monounsaturated fats (%)	6 (23)	2.83 (0.83)	0.017	-
Saturated fats (%)	6 (23)	-1.30 (1.01)	0.243	-
Polyunsaturated fats (%)	6 (23)	-1.50 (1.02)	0.186	-
	6 (23)	-0.90 (0.55)	-	0.158
Backfat thickness (mm)	15 (53)	-0.58 (1.08)	0.599	-
	15 (53)	-0.32 (0.60)	-	0.600
Drip loss (%)	3 (11)	-0.65 (1.33)	0.673	-
	3 (11)	-0.32 (0.81)	-	0.729

Farmers are also acutely concerned about the profitability of pork production. Feed makes up 55-72% of the costs of EU pig production and is subject to significant price volatility, with prices of conventional feed rising 70% from 2005-2012 (from $267 to \$456/tonne) (AHDB Market Intelligence, 2013, 2006). Low-cost swill might therefore be a welcome alternative to conventional grain-based feed. Our results show that while swill feeding had no effect on feed conversion efficiencies ($t = 1.15$, $p = 0.26$), swill feeding did tend to slow pig growth rates ($t = -4.71$, $p < 0.0001$), which would necessarily increase labour and housing costs proportional to the number of extra days required to bring animals to slaughter. The relative merit of cheap, slower-growth swill and expensive, faster-growth conventional feed can be explored with a stylised example.

Assume an EU pig farmer is considering converting to a 50% swill-diet. For simplicity, their current cost of production is €1/kg pork (the EU mean is approximately €1.4/kg pork (Nguyen et al., 2012)), of which 60% are feed costs (EU range of 55-72%), i.e. €0.6/kg. Our results suggest that a diet containing 50% food losses produces 13% lower growth rates, and so the farmer's swill-fed pigs will need 13% longer to reach slaughter weight, making their conventional feed costs equal to €0.34/kg pork (1.13*0.3, where 0.3 = the 0.6 of costs due to conventional feed * 0.5, with the other half of the feed being swill). To conservatively estimate the cost savings of swill, we assume that all other costs also increase in proportion to the extra days required to reach slaughter weight (although fixed costs, such as depreciation and financial costs, make up 15-30% of the cost of production (AHDB Market Intelligence, 2013)). The farmer's non-feed costs would therefore be 1.13 * 40% = €0.45/kg pork. In this case, the farmer will have an overall lower cost of production if swill costs less than 70% the price of conventional feed (calculated as 1-the cost of swill production/the cost of the equivalent conventional feed*100: [€1 – €0.34 – €0.45] / €0.3 *100). In the centralised food loss recycling systems, swill typically costs only 40-60% of conventional feed (20 vs. 50¥/kg in (Takahashi et al., 2012) and 167 vs. 278₩/kg in (Nam et al., 2000) and Figure 2.4). For this farmer, swill feeding would therefore improve profitability. Swill has a more variable nutritional content than conventional feeds (Westendorf, 2000b) and will not suit the business models of all farms, but it could help many to improve profitability. This is especially the case if swill-fed pork is marketed as a premium, low environmental impact product, as it is in Japan ("Eco-pork", see Appendix B). There it receives an associated price-premium, which further boosts farm profits.

While swill feeding could benefit the bottom line of many individual farmers, there is concern that if the legalisation increased the risk of an outbreak of disease, such as foot and mouth or classical swine fever, the overall cost to the industry of such an outbreak could outweigh the financial gains (House of Lords, 2014). This concern is understandable given the £8 billion cost of the UK 2001 foot and mouth outbreak (UK House of Commons report, 2002). It is challenging to quantify the relative risk of a disease outbreak occurring under either of our two different policy scenarios: the status-quo ban on swill and the centralised, regulated use of swill, and it is not certain which policy is lower risk. While it may be argued that a total ban on swill feeding is safer than the regulated use of swill, this ignores the illegal feeding of food waste on smallholder farms which occurs under current, "low-risk" legislation. A survey of 313 smallholder farms in the UK, for example, found that 24% of smallholders fed uncooked household food waste to their pigs (Gillespie et al., 2015). It is worth noting that there have been no disease outbreaks linked to the use of swill in Japan and South Korea (Muroga et al., 2012; Park et al., 2013) and that the use of food losses as animal feed has consistently grown in both countries (by 125% in Japan from 2003-2013, Figure 9.1 in Appendix B, and by 35% in South Korea from 2001-06, Figure 2.1), suggesting strong farmer buy-in.

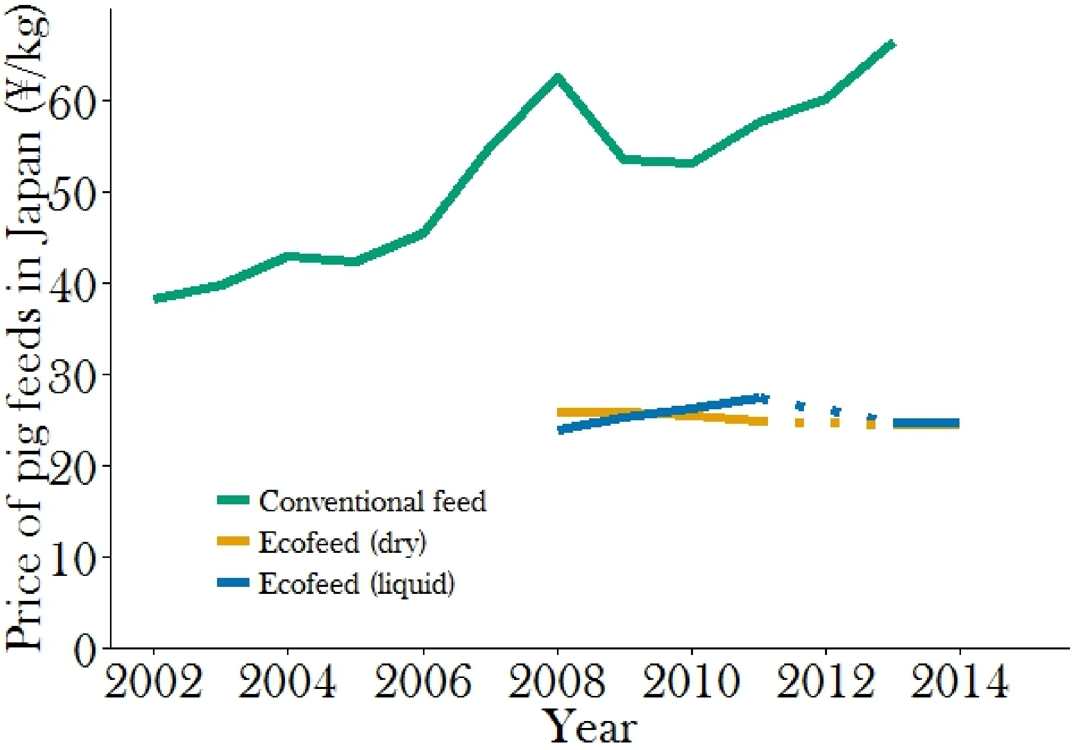


Figure 2.4 – Prices of conventional pig feed and swill (Ecofeed) in Japan. Dry Ecofeed is fed as a dehydrated pellet, liquid Ecofeed is fed as a wet feed. Dotted lines are an interpolation between the 2011 and 2013 values. Data from: (MAFF, 2014, 2013, 2012b, 2011, 2010, 2009).

Finally, food safety precautions should include not only heat treatment but also checks for potential contaminants in food losses. García et al. (2005) performed microbiological and chemical analysis of different Spanish sources of food losses and found high levels of heavy metals and dioxins in some household and restaurant wastes. All other food losses (e.g. retail meat, fruit, vegetable, and fish wastes) were deemed suitable for animal feed. The suspected sources of heavy metals were metal cans and piping. Contamination from these sources could be reduced through better collection, waste sorting, and storage procedures, as required by regulation in East Asian states (Appendix B).

2.6.2. Support from the public

Our results and the East Asian case studies demonstrate that food losses can be safely recycled as pig feed to produce pork of high quality and low environmental cost. Despite this, swill has previously faced resistance because of concerns over consumer acceptability. For example, the Co-operative, a UK food retailer, banned pork reared on swill from shops in 1996 citing it "was not a natural feeding practice" (Stuart, 2009). This is an issue of public awareness, however, not food safety. Pigs were domesticated on a diet of swill, and as such, it could be argued that swill is no less "natural" than the practice of feeding vegetarian diets to omnivorous pigs in modern, industrial systems. Our review included a number of blinded trials finding no difference between the flavour (n=4), colour (n=7 for fat; n=9 for meat), fat composition (n=6), or overall palatability (n=4) of conventional- vs. swill-fed pork (Table 1), suggesting that without labelling, consumers would not notice a difference. In fact, improving consumer awareness of swill has had positive effects in Japan, where certification has been introduced. A survey of consumers there found that those most knowledgeable about the pig industry showed the strongest approval of recycling food losses as feed (Sasaki et al., 2011). Public education may be beneficial in promoting the acceptance of swill in the EU.

2.6.3. Support for policy change

Although currently illegal, there is some precedent for reappraising the legal status of swill. First, there is a legal mandate for improved food loss recycling under the EU Waste Framework Directive 2008/98/EC (EC, 2008), and second, similar animal feed regulation is being reconsidered in light of the EU's deficit in protein sources for animal feed (EC, 2013b).

The EU Waste Framework Directive stipulates that EU member states apply a waste management hierarchy to select disposal options in order of their environmental impact (Figure 2.5). Under this legislation, the preferred options are to avoid food losses altogether or redistribute it to people. Next, the use of food losses as animal feed is preferable to composting, anaerobic digestion, or disposal in landfill (Papargyropoulou et al., 2014), though the legislation is notably not applied in this respect.

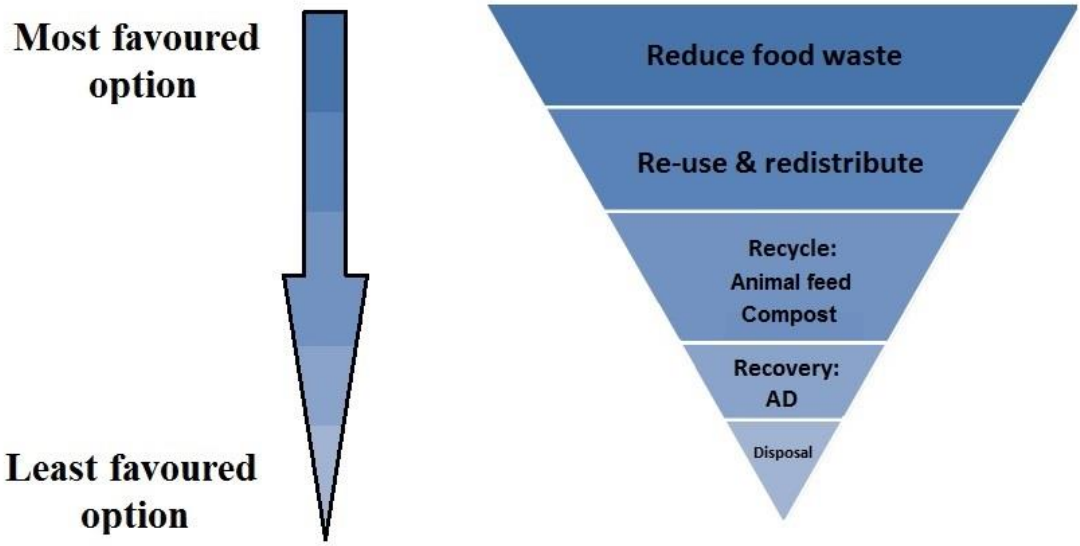


Figure 2.5 – EU food waste hierarchy showing the different levels of waste disposal established under the EU Waste Framework Directive (EC, 2008). Recycling food losses as animal feed is preferable to composting, anaerobic digestion (AD), or disposal in landfill, the latter of which is to be phased out by 2025 under new legislative proposals (EC, 2014). The diversion of food losses for animal feed would not necessarily reduce the availability of inputs for the AD or composting industries, because the inevitable end product of the use of food losses as pig feed – pig manure – is itself highly suitable for both composting and anaerobic digestion (Bernal et al., 2009; Fairlie, 2010; Holm-Nielsen et al., 2009; Stuart, 2009). Image adapted from (Papargyropoulou et al., 2014).

In 2001, the EU banned the use of all processed animal proteins (including pig by-products, such as tendons and trotters, which are fit for human consumption but not eaten by people for cultural or aesthetic reasons) in animal feed, in response to the Bovine Spongiform Encephalopathy crisis (EC, 2001). There are, however, no recorded cases of pigs, poultry, or fish ever naturally developing or transmitting diseases such as BSE (Andreoletti et al., 2007). After a scientific consultation (Andreoletti et al., 2007) and pressure from the animal feed industry (EFPRA, 2011; Searby, 2014), in 2013 the EU re-legalised the use of non-ruminant processed animal proteins in fish farming, and are currently considering its re-legalisation for use in pig and poultry feed (EC, 2013b). It is plausible that swill could undergo a similar process of re-legalisation. It is worth noting that the ban on processed animal proteins is still expected to prevent "intra-species recycling", i.e. the feeding of poultry waste to chickens, or pork waste to pigs. As swill can, and has always, contained pork wastes, swill-feeding legislation in the EU would have to permit this practice, as in the East Asian states described.

2.7. Food losses as animal feed: beyond pigs and beyond the EU.

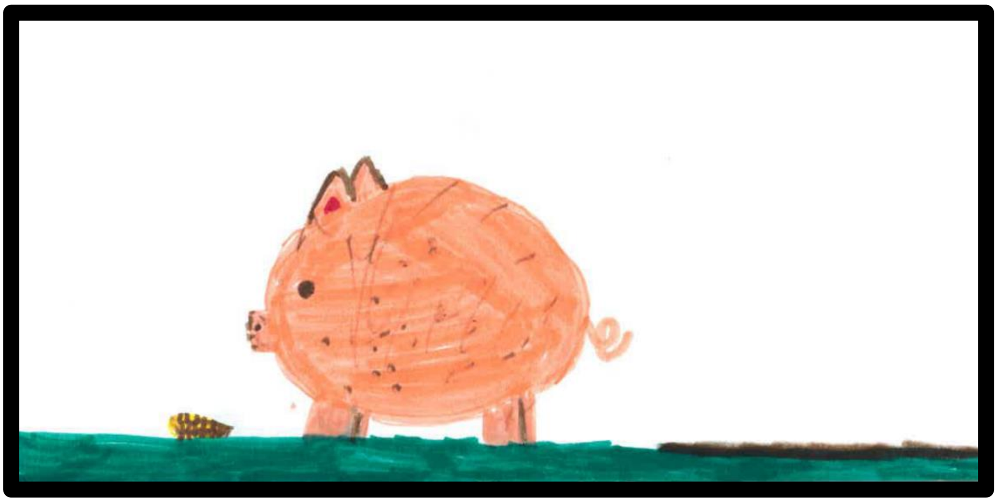
This chapter has focussed on the potential to reduce the land use of EU pork through recycling food losses as swill because of the current EU ban on swill, and because pigs are an omnivorous species with a long history of food loss recycling. Pigs are, however, not the only animal that can consume diets containing food losses. A number of studies have trialled food loss diets for poultry (Boushy et al., 2000; Ruttanavut et al., 2011), fish (Cheng et al., 2014), and ruminants (Angulo et al., 2012; Ishida et al., 2012; Summers et al., 1980), and the environmental gains of food loss feeding for these species represents an area for further work.

The results of this study are also relevant to other parts of the world. We consider briefly here the state of swill feeding in the two largest producers of pork: China and the United States of America (together 55.3% of world production (FAO, 2014a)). Swill feeding is banned in 15 US states (Leib et al., 2016), and across the USA swill-feeding has seen a similar historical trajectory as in the EU: the growth of modern industrialised production systems and availability of abundant grain feed led to a decline in the number of pigs fattened on swill from 130,000 in 1960 to less than 50,000 in 1994 (Westendorf, 2000b). In 2007, only 3% of pig farms fed swill and 95% of US food losses were disposed of in landfill (Leib et al., 2016; US Environmental Protection Agency, 2012). However, swill has recently received renewed interest in the USA. The US Food Waste Challenge, launched in 2013, aims to promote the recycling of food losses, including the use of food losses as animal feed (HLPE, 2014), and the Harvard Food Law & Policy Clinic recently published a guide to legally using food losses as feed (Leib et al., 2016).

In China the use of swill has remained common, and is one of the six highest-volume food loss disposal options nationally (Hu et al., 2012). Swill plays a particularly important role in backyard pig production (30-40% of pigs), where its low cost contributes to smallholder profitability (McOrist et al., 2011). As the Chinese pig industry becomes increasingly industrialised, however, there is a risk that the use of swill may decline (Fairlie, 2010), increasing the environmental impact of pork production, unless systems are put in place to produce swill for industrial pig producers. Centralised food loss recycling may be facilitated by the concentration of many industrial pork producers around densely populated urban areas (Gerber et al., 2005), thereby lowering transport costs and facilitating urban food loss recycling.

2.8. Conclusions

As the demand for livestock products grows over the next half-century, we must identify strategies to reduce the environmental footprint of current systems of meat production. One strategy is the promotion of low-impact animal diets. Food losses, when heat-treated appropriately, as in the centralised food loss recycling systems of Japan and South Korea, can be a safe, nutritious form of animal feed. In this chapter we quantified the potential for swill to reduce the land use of EU pork production. While swill feeding is not a substitute for efforts to reduce food losses, our results suggest that changing EU legislation to promote the use of food losses as swill could substantially reduce the land use impacts of EU pork production. These environmental benefits can be achieved while improving the profitability of many farming businesses and delivering high quality pork products. Similar benefits may be seen in other parts of the world, where swill feeding is currently uncommon or illegal.



Pig eating waste maize. Credit: Jesper Donaldson zu Ermgassen.

3. Environmental and health impacts of using food waste as animal feed: a comparative analysis of food waste management options[2]

3.1. Abstract

The disposal of food waste is a large environmental problem. In the United Kingdom (UK), approximately 15 million tonnes of food are wasted each year, mostly disposed of in landfill, via composting, or anaerobic digestion (AD). European Union (EU) guidelines state that food losses should preferentially be used as animal feed though for most food losses this practice is currently illegal, because of disease control concerns. Interest in the potential diversion of food waste for animal feed is however growing, with a number of East Asian states offering working examples of safe food waste recycling – based on tight regulation and obligatory heat treatment. This chapter investigates the potential benefits of diverting food waste for pig feed in the UK. A hybrid, consequential life cycle assessment (LCA) was conducted to compare the environmental and health impacts of four technologies for food waste processing: two technologies of South Korean style-animal feed production (as a wet pig feed and a dry pig feed) were compared with two widespread UK disposal technologies: AD and composting. Results of 14 mid-point impact categories show that the processing of food waste as a wet pig

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feed and a dry pig feed have the best and second-best scores, respectively, for 13/14 and 12/14 environmental and health impacts. The low impact of food waste feed stems in large part from its substitution of conventional feed, the production of which has substantial environmental and health impacts. While the re-legalisation of the use of food waste as pig feed could offer environmental and public health benefits, this will require support from policy makers, the public, and the pig industry, as well as investment in separated food waste collection which currently occurs in only a minority of regions.

3.2. Introduction

The disposal of food losses poses a large environmental problem. Food losses are abundant: in the UK, approximately 15 million tonnes are wasted annually (234kg/person/year or 50% of food) (WRAP, 2015) and the available disposal options each have substantial environmental impacts. Landfilling produces large quantities of greenhouse gases (GHG) and is therefore being phased out under new EU regulation (EC, 2014), but is still the destination of up to 48% of food losses in parts of the UK (House of Lords, 2014). Incineration and composting also produce greenhouse gases, and wastewater from anaerobic digestion causes eutrophication and acidification of local ecosystems. (Evangelisti et al., 2014; Salemdeeb and Al-Tabbaa, 2015; Whiting and Azapagic, 2014).

To aid the selection of food waste disposal technologies, the EU provides guidelines on which disposal technologies are preferable (EC, 2014). This so-called food waste hierarchy (Figure 2.5), stipulates that governments should prioritise efforts (in order of most to least preferable) to (i) reduce food losses, (ii) redistribute it (e.g. to the homeless), (iii) recycle it as animal feed and (iv) compost, (v) recover energy through anaerobic digestion, and finally, (vi) landfill the remainder. This legislation is, however, notably not applied with respect to the use of food losses as animal feed, because it is currently illegal to use most food losses as feed in the EU.

Though food losses are the archetypal pig feed, if they contain meat wastes and is not heat-treated it can transmit diseases, such as foot-and-mouth disease and African swine fever. In 2001, a UK farmer illegally fed uncooked food losses to pigs, precipitating the foot-and-mouth disease epidemic, which cost the UK economy £8 billion (UK House of Commons report, 2002). As a result, the recycling of most food losses as animal feed was banned across the EU (EC, 2002). The law still permits the feeding of some food losses where it can be demonstrated that there is no risk of contamination with animal products, but this represents only a small proportion of all EU food waste. Currently, of the 89-100 million tonnes of food losses produced in the EU per year (Monier et al., 2010), only around 3 million tonnes are recycled as animal feed (EFFPA, 2014).

In other parts of the world, however, food losses continues to be commonly used as animal feed, including in modern systems of pig production. Heat treatment renders food losses microbiologically safe for animal feed (Edwards, 2000; Garcia et al., 2005; OIE, 2009), and in nations such as Japan and South Korea 35.9% and 42.5%, respectively, of food losses are recycled as feed. There, the use of food waste is closely regulated: legislation governs the heat treatment, storage, and transport of food waste feed (Sugiura et al., 2009).

Amid increases and volatility in the price of conventional feed (AHDB Market Intelligence, 2013; AHDB Market Intelligence, 2006), and concerns about the environmental impact of grain- and soybean-based feeds (Nguyen et al., 2012), there is growing interest in the potential relegalisation and promotion of the use of food waste (i.e. non-legally permitted food losses) as pig feed (The Economist, 2013; The Pig Idea, 2014). A recent survey of 1195 animal feed practitioners (from industry, academia, and NGOs) identified the use of food waste as a priority research area for sustainable animal nutrition (Makkar and Ankers, 2014).

In this chapter we evaluate the environmental and health impacts of converting municipal food wastes into pig feed in the UK. We conducted a hybrid life cycle assessment (LCA) to compare the environmental impacts of two technologies for recycling municipal food waste as animal feed (as a dry or a wet pig feed), with two well established food waste management options: composting and anaerobic digestion (DEFRA, 2015a). In doing so, we address a gap in the literature. Few previous studies have evaluated the potential for recycling food losses as animal feed in the EU, even fewer consider environmental impacts other than greenhouse gas emissions or land use and, none, to the author's knowledge, have thus far specifically considered the use of municipal food wastes as animal feed. Results from Chapter 2 suggest that if the EU were to recycle food waste as pig feed at similar rates to nations such as Japan and South Korea, this would provide enough feed to support 20% of EU pork production, reducing the land use of EU pork by 1.8million hectares of farmland. Four European studies have evaluated environmental impacts beyond land use, though these considered only manufacturing or retail food losses or agricultural co-products (such as beet tails or soybean meal) (Eriksson et al., 2015; Tufvesson et al., 2013; van Zanten et al., 2014; Vandermeersch et al., 2014). These studies each adopted a bottom-up life cycle assessment approach and therefore have several inherent drawbacks that lead to system incompleteness and underestimate environmental impacts (Bernstad and la Cour Jansen, 2012; Laurent et al., 2014b, 2014a). We overcome these methodological limitations by taking a more holistic, hybrid LCA approach (described in more detail below). This chapter focuses on municipal food wastes because they make up 66% of EU food losses (Monier et al., 2010) and are suitable for animal feed – they are currently used in both South Korea and China (Chen et al., 2015; Stuart, 2009) and have historically been used in the EU (Fairlie, 2010).

3.3. Material and methods

We evaluated the environmental and health impacts of processing 1 tonne of municipal food waste in the UK using four different technologies: (a) conversion into dry pig feed, (b) conversion into wet pig feed, (c) anaerobic digestion, and (d) composting (Figure 3.1). We used a hybrid, consequential life cycle approach (developed by Rami Salemdeeb), expanding the system boundary of the analysis to take into consideration the substituted processes. Product substitution operates as follows: if food waste is processed to produce dry pig feed, for example, this will lead to avoided emissions from the substitution of conventional pig feed, but also knock-on emissions from the composting or anaerobic digestion that did not take place. Similarly, the total emissions from composting are the sum of the emissions released during composting, minus the emissions from the production of fertiliser which compost replaces, plus the additional emissions from the conventional pig feed and electricity production, which result from the food waste not being recycled as pig feed or anaerobically digested.

The hybrid LCA approach combines conventional process-based LCA and an input-output based LCA. This approach is used to counter the limitations of conventional LCAs, which face a truncation problem: system boundaries are set *a priori* and typically cut off part of the product life cycle for the sake of simplicity (Bullard et al., 1978; Lenzen, 2001). Input-output approaches use data on the total project cost to estimate upstream-processes that are not modelled using traditional LCA, such as the manufacture of electronic products or technical consulting services, and thereby mitigate truncation error. The input-output component of the hybrid model was a single region model with a domestic technology assumption (i.e. economic activities in the country of origin of imports are the same as in the importing country; Appendix H). The LCA component of the analysis was conducted in EASETECH, a LCA tool developed at the Technical University of Denmark (Clavreul et al., 2014).

We characterised and normalised results for 14 mid-point impact categories (detailed in Table 3-1) for each of our four food waste recycling technologies; these impact categories include a diverse set of environmental and human health indicators to give a multi-criteria assessment of the impacts of our four food waste disposal technologies. Characterisation involves the calculation of each impact (for example, global warming potential requires the weighting of impacts from emissions of carbon dioxide, nitrous oxides and methane). Normalisation then permits comparison of the relative importance of each impact category, by expressing the process' emissions as a proportion of the total emissions (per capita) in the EU-27 in 2010. The global warming potential and particulate matter emissions from recycling 1 tonne of food waste are, for example, scaled relative to the per capita greenhouse gas and particulate matter emissions in the year 2010 (and are reported in units of milli-Person equivalents, mPE). Characterisation and normalisation followed ILCD methods (Benini et al., 2014; JRC, 2010).

Table 3-1 – Environmental impact categories and normalisation references used in this chapter.

Impact category	Abbreviation	Method	Unit (characterised/ normalised)	Normalization factor per person (domestic)
Climate Change	GWP	IPCC	kg CO_2-eq./ mPE $year^{-1}$	9.22E+03
Stratospheric Ozone	ODP	WMO	kg CFC-11-eq./ mPE	2.16E-02
Human Toxicity, carcinogens	HT-C	USEtox	CTU_h / mPE $year^{-1}$	3.69E-05
Human Toxicity, non-carcinogens	HT-NC	USEtox	CTU_h/ mPE $year^{-1}$	5.33E-04
Ionizing Radiation, Human Health	IR	Dreicer	kBq U^{235} eq./ mPE $year^{-1}$	1.13E+03
Photochemical Ozone	POF	ReCiPe	kg-NMVOCeq/ mPE	3.17E+01
Freshwater Eutrophication	FEP	ReCiPe	kg P-eq./ mPE $year^{-1}$	1.48E+00
Marine Eutrophication	MEP	ReCiPe	kg N eq./ mPE $year^{-1}$	1.69E+01
Freshwater Ecotoxicity	ET	USEtox	CTU_e/ mPE $year^{-1}$	8.74E+03
Depletion of Abiotic Resources-Fossil	ADP-F	CML	MJ/ mPE $year^{-1}$	6.24E+04
Depletion of Abiotic Resources-Elements	ADP-E	CML	kg Sb-eq./ mPE $year^{-1}$	1.01E-01
Acidification	AP	Accumul	AE/ mPE $year^{-1}$	4.73E+01
Terrestrial Eutrophication	TEP	Accumul	AE/ mPE $year^{-1}$	1.76E+02
Particulate Matter	PM	Humbert	kg $PM_{2.5}$/ mPE $year^{-1}$	3.80E+00

[1] CTU_h comparative toxic unit for humans.

[2] CTU_e - comparative toxic unit for ecosystem.

[3] AE = Accumulated exceedance

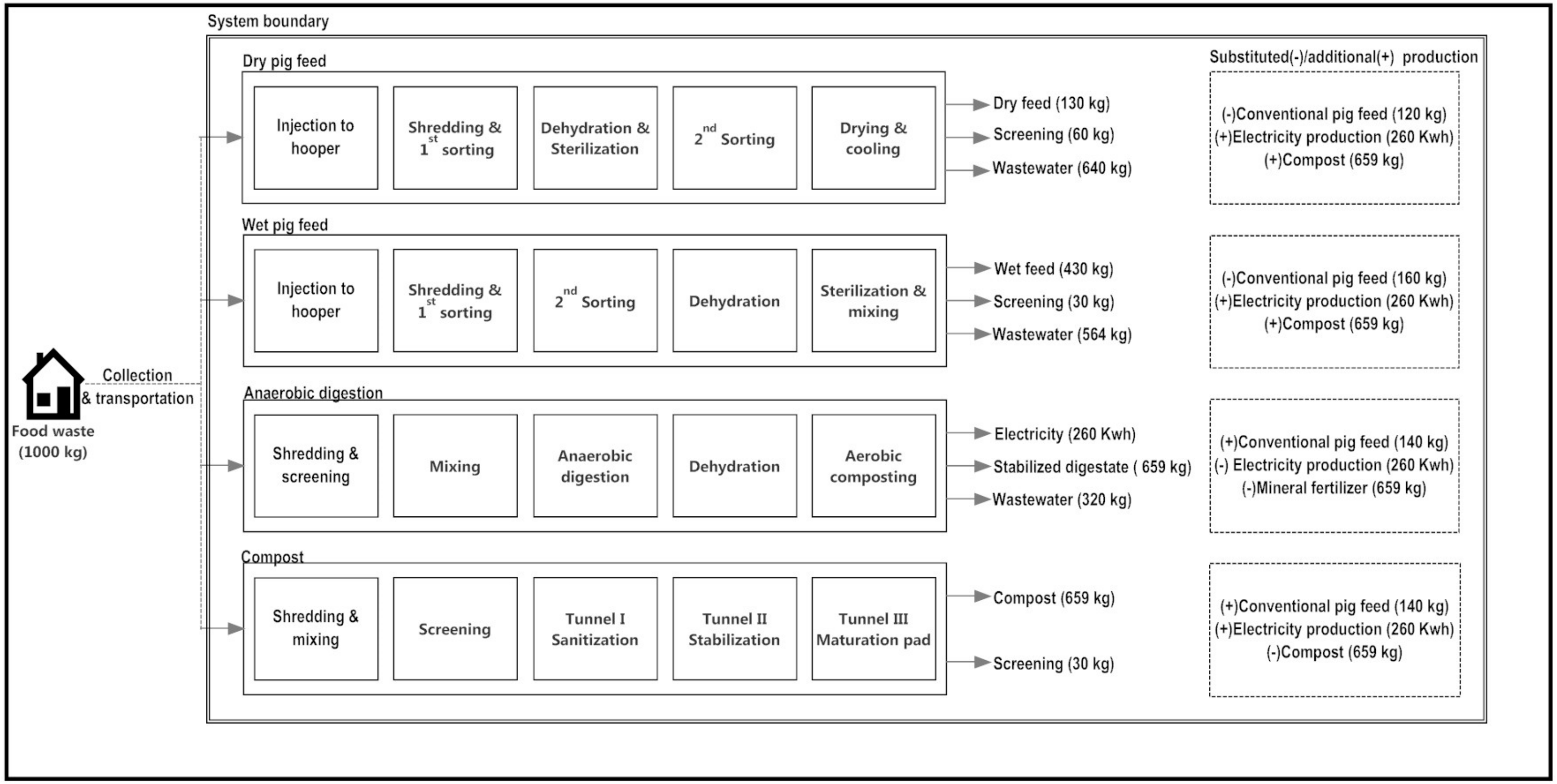

Figure 3.1 – Steps involved in the processing of food waste by the four food waste disposal technologies. Only major material flows are shown: minor inputs (e.g. water, corn in the case of wet feed) and evaporation are not included for the sake of clarity. Outputs are indicated by arrows and substituted products are shown in the boxes on the right-hand side. Mineral fertilizer substitution rates of digestate and compost are listed in section 2.1.3 and Appendix J, Table 9-8, respectively.

3.3.1. Food waste disposal technologies

The four food waste disposal technologies and substituted products are depicted in Figure 3.1. As all technologies require separate collection of food waste, food waste collection and transportation are excluded from this study. Food waste packaging is also excluded due to its insignificant impact (Lebersorger and Schneider, 2011).

3.3.2. Dry pig feed

As the use of municipal food waste as animal feed is illegal in the EU, we used process-specific data from factories producing food waste feed in South Korea (Kim and Kim, 2010b), where there were 259 registered feed manufacturers as of 2010 (Ministry of Environment, 2010a).

Food waste is loaded into a hopper, shredded and filtered for contaminants (Figure 3.1). It is then sterilised and dehydrated by air-drying at 390°C. Under South Korean law, food waste must be heat treated to a core temperature of >80°C for a minimum of 30 minutes (National Institute of Environmental Research, 2012); in comparison, before the ban on using food waste as animal feed, EU law used to mandate heating food waste to 100°C for 60 minutes (Stuart, 2009). The feed is sorted again before one more step of drying, producing 140kg of dry feed per tonne of food waste (with a moisture content of 21.8%, i.e. 109.5 kg of feed on a dry matter basis).

The food waste feed substitutes conventional feed 1:1 on a dry matter basis (Appendix D). The ingredients of the substituted conventional feed (Appendix I) are based on the weighted mean feed intake of all pigs in the pork production life cycle (sows, piglets, and slaughter pigs), taken from an LCA of UK pork production (Stephen, 2011). The impact of feed ingredients that are co-products was allocated according to their economic value. Soybeans, for example, are processed into both soybean meal, a common pig feed ingredient, and soy oil; soybean meal makes up 60% of soybean value, and soy oil the other 40% (USDA, 2012), and so 60% of the impact of soybean production was allocated to soybean meal. When calculating the environmental impact of soybean meal we use the most recent available inventory data on soybean production in Brazil (Nemecek et al., 2014); Brazil is the source of 88% of the UK soybean supply (FAO, 2014a).

In using data from South Korean processing plants, we assume that the municipal food waste used to generate animal feed in South Korea is comparable with municipal food waste in the UK. To check this assumption, we compared data for South Korean food waste with UK data, and found that the compositions are broadly similar (Table 3-2).

Table 3-2 – Municipal food waste composition data for the UK and South Korea. [1] ww= wet weigh

		United Kingdom			South Korea
		(Zhang et al., 2013)	(Banks et al., 2011)	(WRAP, 2010)	(Kim and Kim, 2010b)
PH		5.4	5	n.a	4.2
TS	%ww[1]	27.3	24.4	27.7	20
VS	%ww	25.4	22.3	23.35	14.7
Ash	%ww	1.8	2.1	2.0	5.3
CV	MJ/kg TS	21.1	21.2	26.53n	1.18-20.27
Elemental analysis					
N	%TS	2.9	3.2		3.6
C	%TS	49.7	50.3	49.32	51.0
H	%TS	6.4	6.3	6.5	6.0
S	%TS	n.a	0.2	0.4	0.2
O	%TS	34.7	31.7	37.1	39.2

3.3.3. Wet pig feed

When food waste is used as wet pig feed, it is injected into the hopper, shredded, and twice filtered for contaminants (Figure 3.1). It is then partially dehydrated and heat-treated to 100°C to sterilise it. It is mixed with 25kg of ground maize before storage, to produce 430kg of wet feed per tonne of food waste, with a mean dry matter content of 30.9%. The substitution of conventional feed is calculated as for dry feed.

3.3.4. Anaerobic digestion

In this process, food waste is shredded, sieved, and sent to a digestion tank. The digestate has a dry solids content between 25 and 40% and is digested at a temperature between 50 and 55°C (Hall et al., 2014). AD digestate utilization efficiencies are presented in Table 3-3. Biogas is then collected, purified and used to generate electricity (260 Kwh/tonne of processed food waste), which substitutes electricity produced from the UK energy mix (Table 4). Finally, the remaining digestate undergoes dewatering and refinement producing a high-quality AD cake, which substitutes nitrogen, phosphorous and potassium fertilisers with an efficiency of 34.5%, 46% and 60%, respectively. Benefits from the contribution made by sulphur, magnesium, and other organic compounds in compost are excluded (Wallace, 2011).

Table 3-3 – AD digestate utilization efficiencies. Data from: (Wallace, 2011). [1] 40% of the readily available content of nitrogen is lost during spreading.

	Unit	Value	Efficiency (%)
Readily Available N[1]	Kg/m^3	5.94	34.5
Total Phosphate (P_2O_5)	Kg/m^3	0.48	46
Total Potash (K_2O)	Kg/m^3	1.81	60

Table 3-4 – The 2010 UK electricity national grid. Data from: (DECC, 2014). [1]Total may not equal 1 kwh due to rounding.

Electricity sector	Amount (kwh)[1]	Percentage (%)
Hard coal	0.29	28
Hydropower	0.01	1
Natural gas	0.46	46
Nuclear	0.17	17
Industrial Oil	0.02	2
Wind power plant	0.03	3
Biomass	0.04	4

3.3.5. Composting

Incoming waste is shredded, mixed and aerated for 14-21 days at a minimum temperature of 60°C for 48 hrs (Hall et al., 2014). The compost is then stored in windrows for a 56-day maturation phase. The matured material is removed, screened, and packed as compost. The compost utilization efficiencies used are: 20% for N, 100% for P, and 100% for K (Andersen et al., 2010) and the compost is considered to be applied on loam soil (see Appendix J), where it substitutes synthetic fertilizers on a 1:1 basis. The composting process is assumed to be well managed, i.e. no failures occur that give rise to high emissions of methane and other products of anaerobic conditions. All leachate water is reused during the composting process for re-wetting in the reception area and the maturation pad.

3.4. Sensitivity analysis

A three-step sensitivity analysis approach based on Clavreul et al. (2012) was conducted to evaluate the level of uncertainty in our results. First, the stages with the highest environmental burdens were identified using a hotspot analysis. Then a perturbation analysis was conducted on stages identified in the previous step: we calculated the sensitivity ratio (Eq. 1) for all parameters, by varying each parameter by ±10%.

$$Sensitivity\ Ratio\ (SR) = \frac{\frac{\Delta result}{Initial\ result}}{\frac{\Delta parameter}{Initial\ parameter}} \qquad \text{[Eq. 1]}$$

For each of our four food waste disposal technologies, we selected the parameters which had the highest sensitivity ratios, assigned them probability distributions and performed a Monte Carlo analysis (1000 simulations) to generate confidence intervals for our results. The selected parameters and probability distributions are listed in Appendix K. For each metric, we tested for the significance of differences between technologies, also using Monte Carlo methods. We randomly sampled estimates of the mean for each technology, and calculated the difference between each technology, repeating this resampling 1000 times. We then tested to see if the difference between technologies overlapped with zero at the 99% confidence level.

Each technology was then ranked (1-4; 1=best, 4=worst) for each of our 14 mid-point metrics and the mean ranking for each technology was calculated.

3.5. Life cycle inventory data

The hybrid LCA analysis requires two datasets: process-based physical data, and input-output monetary data.

3.5.1. Physical data

Data, listed in Table 3-5, was either compiled or calculated based on information from project documents, literature, and the WRATE database (Hall et al., 2014). Upstream and downstream material flows and emissions were collected using existing databases, primarily the Swiss Eco invent database v2.2 (Ecoinvent, 2014). These processes include the acquisition of raw material and energy, production, on-site operation, and waste disposal (i.e. cradle to grave).

Table 3-5 – Life cycle inventory data of food waste management options. Sources: [1](Kim and Kim, 2010b) and [2](Hall et al., 2014).

	Materials	Unit	Animal dry feed[1]	Animal wet feed[1]	Anaerobic digestion[2]	Composting [2]
Input	Food waste	Kg	1000	1000	1000	1000
	Corn starch	Kg		250		
	Process water	Kg	2.53		236	110.8
	Woodchip	Kg				0.31
Energy	Gas		32.5			
	Electricity	kwh	24.6	3.86	65	5.78
	Diesel	Kg		2.47	0.081	3.29
Product		Kg	130	430	Digestate (659) Electricity (260 kwh)	659
Waste	wastewater	Kg	640	564	320	
	screening/rejected materials	Kg	60	30		30
Process air emissions						
	CO_2	Kg	8.7E+01	7.9E+00	2.6E-01	1.1E+01
	CH_4	Kg	1.6E-03	3.2E-04	3.4E-02	4.8E-03
	N_2O	Kg	1.6E-04	6.4E-05	1.9E-02	2.7E-02
	NO_x	Kg	2.3E-01	2.1E-02	4.4E-02	1.0E-01
	CO	Kg	3.1E-02	1.6E-03	1.5E-03	5.9E-02
	MVOC	Kg	7.8E-03	0.0E+00	2.4E-02	6.0E-03

3.5.2. Monetary data

Monetary data were obtained from two sources: data of animal feed technologies were obtained by direct communication with the Korean Ministry of Environment;

expenditure data for 2014, available in South Korean Won (₩), was converted into British pound using purchasing power parity coefficients of the year 2014 (OECD, 2015; see Appendix L). Data of both AD and composting were obtained from UK industrial partners (Appendix L). Appendix H lists sources of data and components for the IO-based element of the hybrid approach.

3.6. Results and discussion

The recycling of food waste as wet pig feed had the best score for 13 of 14 environmental and health impacts while dry feed had the second-best score for 12 of 14 impacts (Figure 3.2). The mean ranking of the four technologies (1=best, 4=worst) were wet feed: 1.1, dry feed: 2.2, AD: 3.3, and composting: 3.4. Composting had the worst score for seven environmental indicators, anaerobic digestion the worst score for eight (including two joint-worst scores shared with composting), and dry feeding the worst score for one indicator (depletion of fossil fuels).

After normalisation, composting and anaerobic digestion had disproportionate impacts through eutrophication (terrestrial, marine, and freshwater), environmental toxicity (including non-carcinogenic toxicity, HT-NC, and ecotoxicity, ET), and acidification (Figure 3.2). The superiority of wet and dry feed in these impact categories stems in large part from their substitution of conventional animal feed (Appendix M). All stages of conventional feed production, including the farming and transport of raw materials to the feed processing centre, the milling of the feed, and the storage of the feed mixes, contribute to the substantial emissions in these impact categories. For example, hotspot analysis shows that freshwater eutrophication impacts are principally caused by the use of phosphate-based fertilisers in the farming of feed crops, and marine eutrophication is principally caused by the energy consumption and fuel inputs involved in shipping feed ingredients (such as soybean meal).

For non-carcinogenic toxicity, we found that the concentration of zinc during the growth of rapeseed (an ingredient in conventional pig feed) accounts for nearly 35% of the impact. This result agrees with other studies highlighting concerns that rapeseed may contain high concentrations of heavy metals (such as zinc and copper) and allergens. Heavy metals from the soil are known to accumulate in the roots, plant, and seeds of rapeseed (van der Spiegel et al., 2013).

Our results support the diversion of food waste to animal feed, before composting or anaerobic digestion, as proposed under the food waste hierarchy. The difference between AD and composting is however less clear: composting rated better than AD for 7/14 indicators, including acidification, terrestrial eutrophication, and particulate matter; AD rated better for 5/14 indicators, including greenhouse gas emissions and ozone depletion, and there was no significant difference between them for 2/14 indicators (marine eutrophication and non-carcinogenic toxicity).

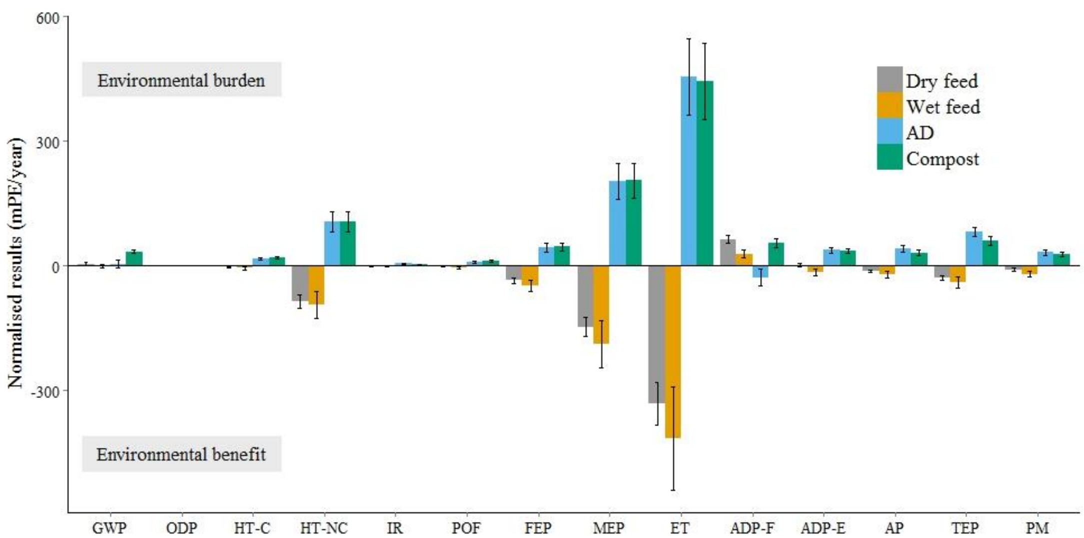

Figure 3.2 – Normalised environmental and health impacts of four recycling technologies for food waste: dry animal feed, wet animal feed, anaerobic digestion (AD), and composting. Units (mPE) relate a process' emissions to per capita emissions in the EU in 2010. GWP=global warming potential; ODP=ozone depletion; HT-C=emissions of carcinogens; HT-NC=emissions of non-carcinogenic toxins; IR=ionising radiation; POF=photochemical oxidant formation; FEP=freshwater eutrophication; MEP=marine eutrophication; ET=ecotoxicity; ADP-F=depletion of fossil fuels; ADP-E=depletion of non-fossil fuel

abiotic resources; AP=acidification; TEP=terrestrial eutrophication, PM=particulate matter emissions. Error bars show one standard deviation.

3.6.1. Comparison with previous literature

While these results suggest that the re-legalisation of the use of municipal food waste as animal feed has potential to reduce the impact of food waste disposal in the UK, LCA results are often location- and assumption-dependent (Bernstad and la Cour Jansen, 2012). We therefore compare our results with previous LCAs of food loss recycling. Previous studies did not evaluate the same portfolio of environmental indicators as this study, but greenhouse gas consequences have been calculated for nine studies, shown in Figure 3.3. Though the exact figures vary substantially between studies, some broad patterns emerge. Wet feed has lower emissions than AD (2/3 studies making this comparison); AD produces lower emissions than dry feed (4/5 studies); and wet and dry feed produce lower emissions than composting (4/4 and 4/6 studies, respectively). Some of the differences between studies may be due to particularities of the locations where the studies were performed and the waste stream analysed. Only two of these studies evaluated food loss recycling in Europe (Eriksson et al., 2015; Vandermeersch et al., 2014), and neither (as here) looked at municipal food waste (instead evaluating retail food losses).

Study assumptions also explain some of the differences: none of the studies in Figure 3.3 include land use change and they therefore underestimate the avoided emissions from animal feed substitution. This truncated-boundary problem underestimates the GHG emissions from animal feed ingredients by up to nine-times (van Middelaar et al., 2013). For example, Eriksson et al. use a GHG emission for soybean meal of 0.66kgCO_2e/kg, while our study uses the most recent figure of 4.4kgCO_2e/kg (Nemecek et al., 2014). Eriksson et al. also report large avoided greenhouse gas emissions when food waste is anaerobically digested compared with its use as dry animal feed (-381.4 vs -40.84kgCO_2e/kg; figures for this study: 3.80 vs 3.96kgCO_2e/kg). This difference stems from assumptions about the yields of biogas during anaerobic digestion and the energy mix substituted. Eriksson et al. assume that the entire theoretical yield of biogas was produced, while our work is based on actual AD plant figures; their study assumes biogas replaces diesel as a fuel for city buses, while this study assumes biogas substitutes UK electricity production (natural gas 61.46 % and coal 38.54%).

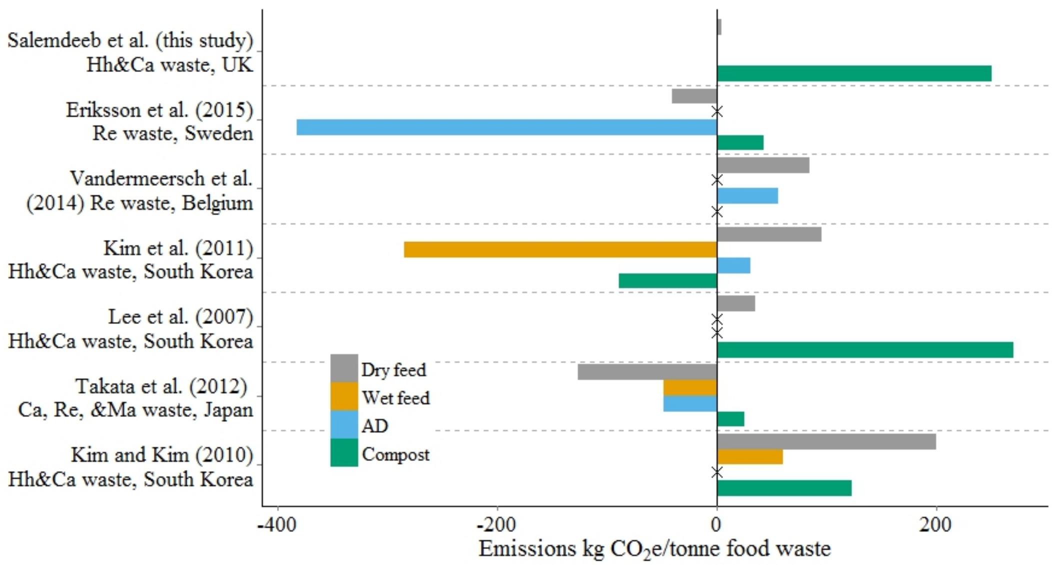

Figure 3.3 – Results of seven LCA studies reporting the greenhouse gas emissions per tonne of food waste. The location of and waste stream evaluated in each study (Hh=household, Ca=catering, Re=retail, and Ma=manufacturing food waste) are listed. Crosses are marked where a study did not include a technology in their analysis. Where a study reported emissions for multiple food waste types (e.g. meat or banana wastes from supermarkets), the mean emissions are shown. Two further LCA studies (Ogino et al., 2012; Tufvesson et al., 2013) use different functional units (reporting results per kg of animal feed and per MJ of fuel energy, rather than per tonne of food waste) and so cannot be displayed for comparison. AD=anaerobic digestion.

3.6.2. Robustness of results

To better understand the uncertainty in our results, we tested the sensitivity of our results to the parameter values chosen in the model. Despite the large variability in some parameters (Appendix K), the indicator values for all metrics are significantly different from one another ($p<0.01$), except for the effect of composting and anaerobic digestion on marine eutrophication and non-carcinogenic toxicity.

Land use change emissions are the largest source of greenhouse gas emissions associated with certain forms of animal feed, notably soybean meal (van Middelaar et al., 2013), yet are ignored in much of the literature on food waste disposal technologies (but see Tufvesson et al. (2013) and van Zanten et al. (2014)). In this study, we therefore used the most recent data available on land use change emissions for soybean meal (Nemecek et al., 2014), a major constituent of EU pig feed. This has a large effect on the modelled emissions from wet and dry feed (Figure 3.4). This shows the importance of using updated data inventories for agricultural products, whose emissions vary over time and whose measurement is rapidly improving.

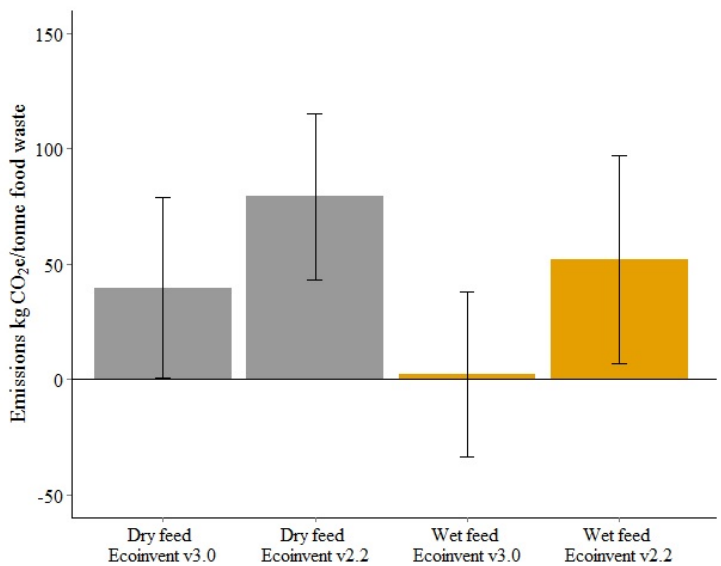

Figure 3.4 – Greenhouse gas emissions from using food waste as dry feed and wet feed, comparing the calculation using two different datasets for emissions from soybean production, Ecoinvent v3.0 or Ecoinvent v2.2. Ecoinvent v3.0 includes improved estimates of emissions from land use change (Nemecek et al., 2014), and therefore produces lower estimates of emissions from recycling food waste as feed (which leads to avoided emissions from the production of conventional feed). Error bars show one standard deviation.

3.6.3. Wet vs dry feed

We evaluated two different technologies for recycling food waste as animal feed. We find that the processing of food waste into a sterilized wet feed has lower environmental and health impacts for all indicators, compared with processing into a dry pig feed. The difference between wet and dry feed results in large part from the higher fossil fuel inputs required to dehydrate municipal food wastes (Figure 3.5). Municipal food wastes have a high water content (typically 65-80%), and their dehydration to make dry feed requires gas and electricity (Table 3-5). This result does not, however, suggest that food waste feed should always be fed as a wet feed, because these two technologies may be suitable for different pig production systems. In South Korea, for example, dry feed is often produced in centralised facilities before resale and transport to farmers, while wet feed has a much higher water content and is therefore more expensive to transport. It is typically produced on or near to pig farms in order to minimise post-processing transport costs. The suitability of dry or wet pig feed depends in part on the proximity of pig farms to sources of food waste. For this reason, wet food waste feed, or "swill", has long been a favoured pig feed for smallholder farmers (Westendorf, 2000b). Most industrial pig farms in the UK currently use dried feeds; wet feeding is more common in other EU nations, such as the Netherlands, where it is favoured because it permits the use of wet agricultural wastes, such as distillery wastes or beet tails (van Zanten et al., 2014), and because of reported nutritional benefits of wet feeding (Brooks et al., 2001; Missotten et al., 2015).

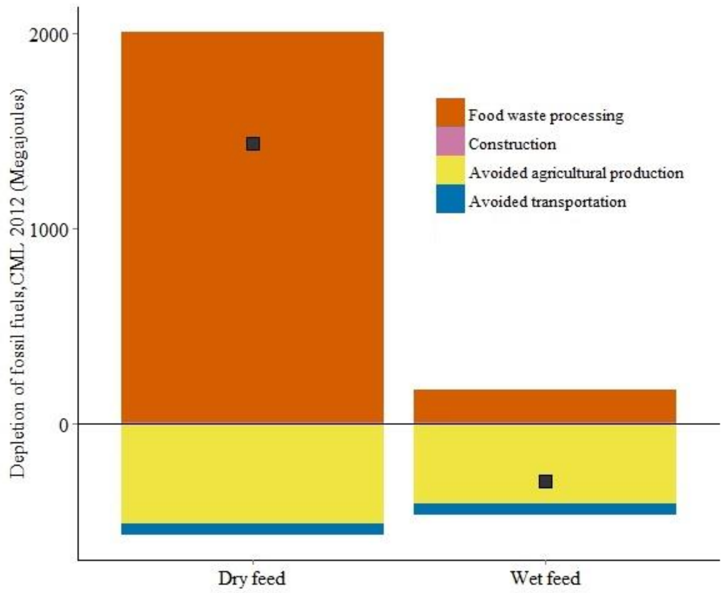

Figure 3.5 – Fossil fuel use (MJ) in the production of wet feed and dry feed. The processing stage for dry feed has much higher fossil fuel use than wet feed because of the additional dehydration of food waste during production. Both avoid fossil fuel use associated with the production of conventional pig feed. The black points mark the net fossil fuel use per technology.

3.6.4. Other species

Food waste can be fed to livestock other than pigs, including poultry, fish, and ruminants (Angulo et al., 2012; Boushy et al., 2000; Cheng et al., 2014). This study focussed on the use of municipal waste as pig feed because they have a long history of recycling waste into animal products (Fairlie, 2010), and because there are human health concerns with feeding food wastes which contain animal products to other livestock species, notably ruminants. The use of meat wastes in ruminant (cattle, goat and sheep) diets is banned in the EU because of concerns about Bovine Spongiform Encephalopathy (BSE), a disease that does not affect pigs, poultry, or fish (Andreoletti et al., 2007). Alternatively, food wastes can be fed to insects which may in turn be used as animal feed (van Zanten et al., 2015). This practice would be inherently less efficient than feeding food waste to pigs directly, and is also currently illegal, though there is an active campaign for the legalisation of the use of insects in animal feed (Searby, 2014).

3.6.5. Barriers to adoption

This study suggests that the use of municipal food waste as animal feed could reduce the impact of food waste disposal in the UK. This practice is currently illegal and there are a number of barriers to its adoption, both political and infrastructural.

Animal feeds are of course not only selected on their environmental merit. The re-legalisation of swill would require the confidence and support of the public, pig industry, and policy makers. Though heat treatment renders food waste safe for pig feed, there is some concern that the re-legalisation of heat-treated food waste feed might increase the risk of uncooked food waste entering the feed supply, potentially leading to disease outbreaks in livestock. If re-legalised, however, the potential benefits of using food waste as feed include reduced impacts on the environment, improved profitability for many farmers, and high meat quality and taste (for more detailed discussion see chapters 2 and 7).

Food waste can only be used in animal feed if it is collected separately from other wastes and is sufficiently fresh. While this is the case in countries like South Korea and China (Chen et al., 2015), food waste collection in the UK is currently more variable. In 2013, separate food waste collection occurred in 95% of Wales, but only 34% of Scotland, 26% of England, and 4% of Northern Ireland (House of Lords, 2014). The potential use of food waste as animal feed is therefore not only a function of the availability of food waste, but its accessibility and quality. Where food waste is of poor quality or not adequately separated, it can be diverted to composting or anaerobic digestion, in line with the food waste hierarchy. It is promising that separate food waste collection in the UK increased from 15,000 tonnes in 2006 to nearly 350,000 tonnes in 2012 (DEFRA, 2015b).

3.7. Conclusion

While feeding municipal food waste to livestock is currently illegal in the EU, it is a common practice in many parts of the world, and there is growing interest in its potential use as a replacement for high-impact, high-cost conventional pig feed. This is the first study to compare the environmental impacts of recycling municipal food waste as animal feed with alternative disposal options in the EU. We used a holistic, hybrid LCA approach to compare four food waste disposal technologies in terms of 14 different environmental and health impacts and found that converting municipal food wastes into pig feed would lead to lower environmental and health impacts than processing waste by composting or anaerobic digestion – the UK government's currently preferred disposal options (DEFRA, 2015a). The widespread use of food waste as animal feed in the EU will require consumer and industry support, policy change, and investment in food waste collection infrastructure. Our results suggest that if these barriers can be overcome, the re-legalisation of food waste in pig feed could lead to substantial environmental and health benefits.

Pigs being fed former foodstuffs (food losses from a supermarket) at the Spilvarken initiative in Gent, Belgium. Source: Ian Kelly.

4. Support amongst UK pig farmers and agricultural stakeholders for the use of food losses in animal feed[3]

4.1. Abstract

While food losses (foods which were intended for human consumption, but which ultimately are not directly eaten by people) have been included in animal feed for millennia, the practice is all but banned in the European Union. Amid recent calls to promote a circular economy, we conducted a survey of 163 pig farmers (n=82) and other agricultural stakeholders (n=81) at a UK agricultural trade fair on their attitudes toward the use of food losses in pig feed, and the potential relegalisation of swill (the use of cooked food losses as feed). While most respondents found the use of feeds containing animal by-products or with the potential for intra-species recycling (i.e. pigs eating pork products) to be less acceptable than feeds without, we found strong support (>75%) for the relegalisation of swill among both pig farmers and other stakeholders. We fit multi-hierarchical Bayesian models to understand people's position on the relegalisation of swill, finding that respondents who were concerned about disease control and the perception of the pork industry supported relegalisation less, while people who were concerned with farm financial performance and efficiency or who thought that swill would benefit the environment and reduce trade-deficits, were more supportive. Our results provide a baseline estimate of support amongst the large-scale pig industry for the relegalisation of swill, and suggest that proponents for its relegalisation must address concerns about disease control and the consumer acceptance of swill-fed pork.

[3] This chapter is in review at PLOS One. The text has been reformatted for inclusion in this thesis.

4.2. Introduction

Food losses, i.e. foods which were intended for human consumption, but which ultimately are not directly eaten by people (FAO, 2014b), have long been used as an animal feed – they have, for example, been fed to pigs since the very domestication of wild pigs, around 10,000 years ago (Fairlie, 2010). While food losses continue to be included in animal feed in many parts of the world, the use of food losses in animal feed was all but banned in the European Union (EU) in 2002, after the 2001 foot-and-mouth outbreak, which is thought to have been started by a farmer illegally feeding uncooked food waste to pigs in the UK (Scudamore, 2002).

Current EU legislation permits the inclusion of only a small subset of food losses in animal feed. For example, all food losses containing animal by-products (i.e. materials of animal origin that people do not consume; EC, 2017a) are banned, except for those containing honey, eggs, pig or poultry gelatine, milk products, rendered fats, and collagen, where there is no risk of contamination with other sources of animal by-products (EC, 2013a). These legal food losses are known as former foodstuffs. The legislation specifically bans catering wastes (i.e. food that has been through a home kitchen or restaurant, making up the 57% of food losses in the EU (Stenmarck et al., 2016)) and feeds where there is the potential for intra-species recycling – i.e. pigs eating pork products, or chickens eating poultry products.

These regulations deliver a safe food system to millions of Europeans, though they are not without their trade-offs. The current legislation limits the potential for nutrient recycling and a circular economy – food losses that are not used as feed are instead disposed of in less efficient ways, lower down the food waste hierarchy (Papargyropoulou et al., 2014). Previous chapters have shown that the relegalisation of food losses in animal feed could cut feed costs for pig producers, reduce the land use of EU pork production by 22% (1.8 Mha), and reduce a host of other environmental pressures. The ban on animal by-products in feed also treats all livestock in Europe as being essentially vegetarian, though, of course, pigs and poultry are omnivorous.

In light of these trade-offs and the existence of regulated systems for incorporating food losses in feed in other countries, there have therefore been intermittent calls to relegalise the use of food losses in feed (Fairlie, 2017, 2010; The Pig Idea, 2014; UK parliamentary debate, 2004). Japan and South Korea, for example, operate systems for safely recycling food losses as animal feed, based on the heat-treatment of food losses (heat-treated food losses are colloquially known as "swill", though they are marketed as "Ecofeed" in Japan. Heat-treatment inactivates pathogens (such as foot-and-mouth) in the food, renders it safe for use as animal feed, and facilitates these countries recycling ca. 40% of their food losses as animal feed, compared with the 3-6% achieved in the EU (FERA, 2012).

Still, the debate continues to be polarised, with some arguing that the use of swill is unsafe or unnatural – the UK retailer The Co-operative, for example, banned the use of swill in 1995 (Stuart, 2009) - while others argue that the ban was an exaggerated response to a manageable risk (Danby, 2015). Little work has been done, however, to determine the attitudes of the people most affected by the ban on the use of food losses as feed – namely, pig farmers and workers in the agricultural sector. We therefore conducted a survey to investigate the attitudes of the farming community to the use of food losses as feed.

4.3. Method

The survey was conducted at the British Pig & Poultry Fair on the 10-11th May 2016 at Stoneleigh, Warwickshire. This fair was selected because it is the largest industry fair in the country dedicated to the pig and poultry industries, with 10,000 attendees visiting 350 stands from businesses and organizations involved in the sector. Fifty-eight percent of visitors were pig or poultry producers, with the remaining 42% made up of traders, advisors, students, processors, veterinarians, retailers, and others (BPPF, 2017).

Visitors were invited to complete a survey at a University of Cambridge stand. Pairs of survey workers were also positioned at the entrance to the exhibit building to invite visitors to complete the survey as they entered. To incentivize completing the survey, visitors were offered food and drink at the stand, and people completing the survey were offered the chance to enter a raffle to win one of five £50 prizes. The survey was offered in both an electronic format (using the survey software Qualtrics, available on a tablet), or on paper.

Participants were assured that their responses were anonymous, and the study received ethical approval from the Ethics Committee for the School of the Humanities and Social Sciences, University of Cambridge, prior to be being conducted.

4.3.1. Survey structure

The survey consisted of 18 sets of questions on eight themes, described below. A copy of the survey is available in Appendix N.

(i) Respondents were asked about the acceptability of using ten different sources of food losses in pig feed (Table 4-1). These ten different sources of potential animal feed were selected for inclusion in the survey because they represent a range of combinations of legality, of whether or not they contain animal by-products, and of their potential for intra-species recycling. Respondents were asked: "How would you feel about the inclusion of the following in pig feed", and scored each feed on a 1-5 Likert scale, from "1=very uncomfortable" to "5=very comfortable". To check for the internal consistency of these constructs, the questions were repeated using two other Likert scales, "1=Very negative" to 5=Very positive" and "1=Very dissatisfied" to "5=Very satisfied". The order that each feed and each Likert scale was presented was randomized, and respondents were also given a "Don't know" option.

(ii) Respondents were asked how they thought heat-treated swill and conventional grain- and soybean-based feed compare in terms of eight different attributes, each scored on a five-point Likert scale (e.g. swill is: "1=Much less nutritious" to "5=Much more nutritious", "1=Much lower disease risk" to "5=Much higher disease risk", or "1=Much lower cost” to “5= Much higher cost”). The order in which each attribute was presented was randomized, and respondents were also given a “Don’t know” option.

(iii) Respondents were asked how they thought the performance of pigs reared on swill would compare with pigs reared on conventional diets. Four attributes (growth rates, feed conversion ratios, environmental impacts, and feed costs) were scored on a 5-point Likert scale (with a “Don’t know” option), and their order was randomized.

(iv) Respondents were asked how they believed pork from pigs reared on a diet containing swill would compare with pigs fed conventional diets. Six attributes (e.g. fattiness, tastiness, and marketability) were scored on a 5-point Likert scale, including a “Don’t know” option. The order of each attribute was randomized.

(v) Respondents were then asked to what degree they agreed with two statements: that swill is either a “unnatural practice” or a “traditional practice”. Both questions were scored on a 5-point Likert scale (“1=Definitely not” to “5=Definitely yes”), and the order of their presentation was randomised.

(vi) The next question was: “If the procedures were put in place to ensure the safety of swill (e.g. heat treatment was performed by regulated swill manufacturers), would you support the relegalisation of swill?”, and respondents reported their attitude on the same 5-point Likert scale.

(vii) Respondents were then asked to reflect on the values underlying their position on the relegalisation of swill, indicating how important twelve different issues were to them (e.g. food safety, perception of the pork industry, meat quality, environmental impacts etc.). The importance of these issues was scored on 5-point Likert scale (“1=Not at all important” to “5=Very important”), and the order of their presentation was randomized. Similarly, respondents were asked to agree/disagree with 12 statements about the impacts of using heat-treated swill (e.g. swill would “Lower dependence on foreign protein sources”, “Lower consumer acceptance of pork products”, or “Increase the risk of toxins entering the feed”. Their agreement was scored on a 5-point Likert scale, “1=Totally disagree” to “5=Totally Agree”, with a “Don’t know” option.

(viii) Finally, respondents were asked about their general characteristics (age, job, gender etc.), and pig farmers were asked about their farming practices. These included questions about the number of pigs, whether they use wet or dry feeds, whether they have previously used swill on their farm, whether their farm was affected by the 2001 foot-and-mouth outbreak, and whether they would consider using swill on their farm, if the use of swill were legalised.

The options included in each question on the comparative performance of swill and its perceived impacts were based on literature on the use of swill and the use of novel animal feeds (chapter 2 and Verbeke et al., 2015). Prior to the fair, the survey was piloted and refined, to ensure that questions were relevant and easily understood, and that the survey software worked smoothly.

Table 4-1 – Characteristics of different sources of food losses.

Food losses to be used as feed	Permitted under current legislation?	Potentially containing animal by-products?	Potentially entailing intra-species recycling?
Heat-treated restaurant leftovers	No	Yes	Yes
Biscuit crumbs from biscuit factories	Yes	No	No
Unsold confectionery containing porcine gelatine	Yes	Yes	Yes[a]
Unsold bread from supermarkets	Yes	No	No
Unsold egg sandwiches from supermarkets	Yes	Yes	No
Heat-treated leftovers from a college canteen	No	Yes	Yes
Misshapen chocolates from chocolate factories	Yes	No	No
Heat-treated, unsold bacon sandwiches from supermarkets	No	Yes	Yes
Heat-treated, unsold chicken sandwiches from supermarkets	No	Yes	No
Heat-treated household food leftovers	No	Yes	Yes

Food losses are listed by their legality, whether or not they contain animal by-products, and whether there is the potential for intra-species recycling. [a]Gelatine products are exempt from the ban on intra-species recycling.

4.3.2. Details of survey respondents

Across the two days of the fair, 163 people completed the survey, including 82 pig farmers (13 farmers both with pigs and poultry and 69 farmers who keep only pigs). The 81 non-pig farmers included a variety of professions associated with the livestock industry, including poultry farmers, agricultural advisors, traders, and veterinarians (Figure 9.8 in Appendix O). Since our sample included only six respondents who reported not being directly employed in the animal industry (of which one was a former pig farmer) and given that their attendance at an agricultural trade fair suggests a strong interest in farming, for the purposes of analysis, we grouped all respondents into one of two groups: pig farmers (including farmers both with pigs and poultry) and other agricultural stakeholders.

Amongst pig farmers, 73% (60/82) of our sample were farmers with more than 1,000 pigs – our sample therefore captures views within the mainstream pork production industry. Though there are many small farms (<100 pigs) in the UK (Figure 9.9 in Appendix O), these host only 2% of the national herd, and therefore represent only a small market share (Eurostat database, 2014), often for local consumption.

The median age group of respondents of respondents was 31-50 (Figure 9.8 in Appendix O), with 142 men and 21 women. Overall, 158 surveys were completed on tablets, and 5 completed on paper. Surveys took a median of 18 minutes to complete.

4.3.3. Statistical analysis

All statistical modelling was done in R version 3.1.1 (R Core Team, 2013). The internal reliability of the three constructs about the acceptability of different sources of food losses as pig feed (from (i), above) was tested by calculating their Cronbach alpha. Values exceeding 0.80 indicate a good degree of internal reliability. To understand the differences in acceptability of different sources of food losses as pig feed (i.e. why some are more acceptable than others), an ordered categorical model with a cumulative link was fit to the data. This modelled the acceptability of each feed as an ordered categorical variable (from "1=Very unacceptable" to "5=Very acceptable"), as a function of several predictors, including the respondent's characteristics (e.g. job, gender from (viii), above), and characteristics of the feed (e.g. its legal status, as listed in Table 4-1) with varying intercepts for each feedstuff and respondent. The models were fit with Bayesian methods using weakly informative priors and the "rethinking" package (McElreath, 2016a). We ran seven different models, with different predictors included in each (Table 4-2); these models were compared on the basis of their Widely Applicable Information Criterion (WAIC) score, an information criterion which makes no assumptions about the shape of the posterior distribution (McElreath, 2016b), and predictions were made using model averaging. Model fitting occurred in two stages. First, models were fit and de-bugged using three chains, each 4,000 iterations long, including a 2,000 iteration warm up. Once we were satisfied that each model had successfully converged (by checking chains, the effective number of samples, and ensuring the Gelman-Rubin convergence diagnostic, Rhat <1.01), a single longer chain (10,000 iterations with a 5,000 warm up iterations) was fit and used for parameter estimation, plotting, and prediction. The equations of these models are listed in Appendix O.

To understand each respondent's position on the relegalisation of swill, we also fit Bayesian ordered categorical models to predict both support for relegalisation (9 models fit to data from all respondents and 14 models fit to data from pig farmers only, respectively listed in Table 4-3 and Appendix O Table 9-14) and farmer willingness to use swill on their farm (19 models, Appendix O Table 9-15). Predictor variables included information about the respondent's characteristics (age, job, gender etc.), and the characteristics of the farm (number of pigs, whether or not the farm was affected by the 2001 foot-and-mouth outbreak etc.). We also used factor analysis to simplify the responses to the 12 questions about the importance of different issues (e.g. food safety) and the impact of swill (e.g. to what degree they (dis)agreed that swill would "lower dependence on foreign protein sources") into a smaller number of factors, which were also included as predictor variables. The equations of these models are listed in the Appendix O.

Factor analysis was done using the "psych" package (Revelle and Revelle, 2017) with polychoric correlation, as recommended for ordinal data (Holgado-Tello et al., 2010). We selected the number of factors on the basis of the Minimum Average Partial correlation (Velicer, 1976). We identified two factors which explained 45% of the variance in the response about the respondent's values (i.e. how important different issues were to them). The first factor combined concerns about disease control and the perception of the pork industry by consumers (i.e. factor 1 had factor loading >0.5 for communication with consumers, traceability, perception of the pork industry, labelling of the end product, food safety and disease control). The second factor identified concerns about financial performance and efficiency (factor loading >0.5 for feed prices, profitability, and efficient use of resources).

Similarly, we identified two factors which explained 43% of the variance in respondents' perception of the impact of swill. The first factor grouped perceptions that swill would benefit the environment, help farms financially, and reduce trade-deficits (factor loadings > 0.5 for reduce the environmental impact of food waste disposal, reduce the environmental impact of pork production, be an efficient way to use food waste, help farms reduce feed costs, help farmers improve profitability, and lower dependence on foreign protein sources). The second factor grouped perceptions that swill would increase disease risk and be unpalatable to consumers (factor loadings > 0.5 for increase the risk of prion diseases like bovine spongiform encephalopathy, reduce traceability, negatively affect the marketability of pork, increase the risk of toxins entering the feed, increase the risk of an outbreak of foot-and-mouth disease, and lower consumer acceptance of pork products). Missing values ("don't knows") were imputed as the median value (Revelle and Revelle, 2017).

Parameter estimates from the Bayesian models were converted into estimates of their effect size (e.g. how much support for the relegalisation of swill differed between farmers who used wet feeding systems versus those using dry feeding systems), by simulating the responses of 1,000 respondents in each group (e.g. wet feeders vs dry feeders), taking into account both parameter uncertainty and sampling uncertainty. Parameter uncertainty was included by sampling from the model averaged posterior distributions, and sampling uncertainty was accounted for by modelling responses using an ordered categorical probability density function. The data and code used for all analyses are available in the supplementary material (Appendix O).

4.4. Results

4.4.1. Acceptability of use of different sources of food losses in animal feed

Respondents thought that feeds containing animal by-products or which had the potential for intra-species recycling were less acceptable than feeds which did not (Figure 4.1) – equivalent to a 1.0-point and 0.7-point lower acceptability (scored from 1-5), respectively, than feeds without (Figure 9.11 in Appendix O). While the difference between the acceptability of legally permitted and non-legally permitted sources of feed was close to zero (Figure 4.1), a model including an interaction between job and legal status (model AC1) had similar Akaike weight to models not including this interaction (Table 4-2), so while it appears that non-pig farmers thought that legal feeds and the non-legal feeds were equally acceptable, we cannot rule out that pig farmers perceived feeds that are not legally permitted to be less acceptable than feeds that are currently permitted. Pig farmers, for example, were more accepting of using unsold bread from supermarkets as feed than other respondents were (mean acceptability of 4.26 vs 4.04, Figure 9.10 in Appendix O), while other respondents were more accepting of the use of heat-treated restaurant left-overs (mean acceptability of 3.17 vs 2.69). There was however, far greater variability between respondents than between the scores for different feeds (compare estimates of the variation between feeds and respondents, the data below the dashed line in Figure 4.1), indicating no consensus in the acceptability of different food losses as feed.

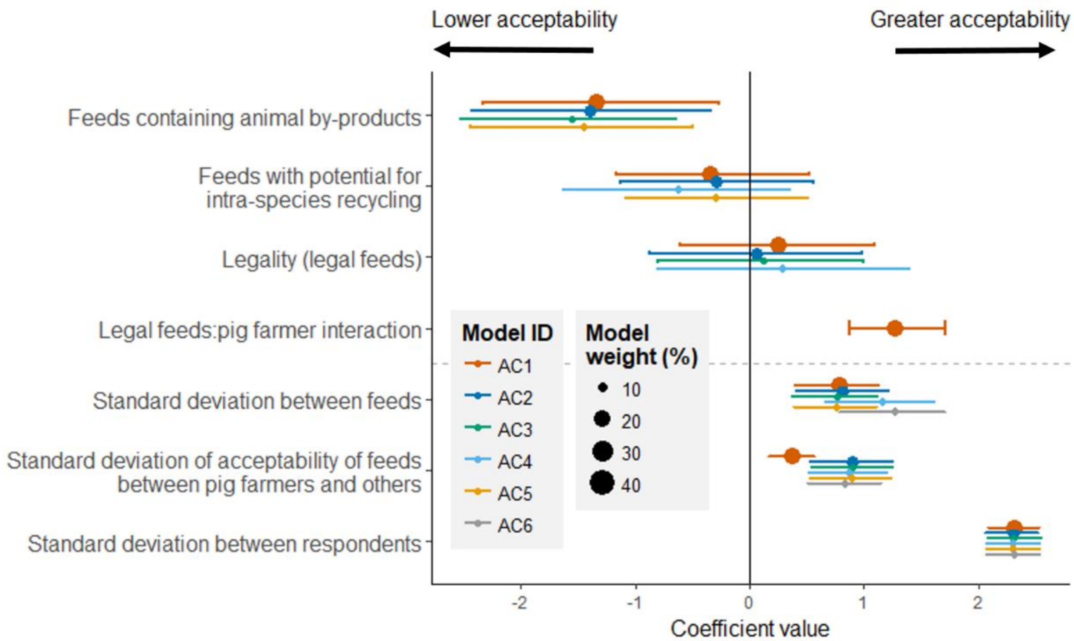

Figure 4.1 – Estimates from the six models with the greatest weighting (summing to 100% of model weight) of how the acceptability of different feedstuffs varies according to their characteristics (e.g. whether or not they contain animal by-products, or their legality). The variation between different feeds, respondents, and feed:job combinations is shown below the dashed line. Model weights are proportional to the size of the points. Error bars are 89% credible intervals.

Table 4-2 – Models explaining the acceptability of different feeds, listed in order of their Akaike weights. WAIC is the widely applicable information criterion score, pWAIC is the number of effective parameters, and model weights are the Akaike weights. Model equations are listed in Appendix O. Models AC2-AC6 have sufficiently similar WAIC scores that their relative weighting differs between model runs; the results reported here are representative of typical results.

	Predictors							Model output		
Model ID	Respondent intercepts	Feed intercepts	Feed:job interaction	Slope for feed legality	Slope for animal by-products	Slope for intra-species recycling	Job:legality interaction	WAIC	pWAIC	Model weight
AC1	Y	Y	Y	Y	Y	Y	Y	10337.5	182.7	0.28
AC2	Y	Y	Y	Y	Y	Y	-	10338.1	183.3	0.20
AC3	Y	Y	Y	Y	Y	-	-	10338.8	183.7	0.14
AC4	Y	Y	Y	Y	-	Y	-	10338.9	183.6	0.14
AC5	Y	Y	Y	-	Y	Y	-	10339.0	183.8	0.13
AC6	Y	Y	Y	-	-	-	-	10339.3	183.9	0.11
AC7	Y	Y	-	Y	Y	Y	-	10353.4	176.5	0.00

4.4.2. Comparison of swill and conventional grain- and soy-based feed

Most respondents thought that heat-treated swill is better for the environment, lower cost, and more ethical than conventional feeds, but more variable in nutritional content, and associated with a higher disease risk, lower microbiological safety, and lower chemical safety (Figure 4.2). There was split opinion about whether swill is more nutritious than conventional feeds, with 22% of respondents thinking that swill was less nutritious, 27% thinking it was more nutritious, and 37% responding "neither more nor less" (with 14% "don't know"). There was a similar distribution of opinions among both pig farmers and other agricultural stakeholders, except for the question about disease risks, where 58% of pig farmers thought heat-treated swill posed a higher risk ("much higher" or "higher disease risk"), compared with 36% of other respondents.

Figure 4.2 – Comparison of swill and conventional feed. Responses to the question: "Compared with feeding conventional grain- and soybean-based feed, heat-treated swill is:"

Respondents thought that using swill would have little effect on animal welfare (75% reported that pigs fed swill would have neither higher nor lower welfare) and would lower feed costs for farmers (84% reported swill would lead to "lower" or "much lower" feed costs; Figure 9.12 in Appendix O). Respondents were, however, unsure about the impact of swill-feeding on pig performance and meat quality (Figure 9.12 and Figure 9.13 in Appendix O). Twenty-five percent and 28%, respectively, of respondents replied "don't know" to questions about the effect of swill on pig growth rates and their feed conversion (i.e. how many kilograms of feed are required per kilogram of growth). Similarly, there was uncertainty about the effect of swill-feeding on pork colour (39% of respondents thought there would be no effect, 44% don't know), taste (46% no effect, 28% don't know), smell (55% no effect, 29% don't know), and fattiness (55% no effect, 29% don't know).

4.4.3. Opinion on the relegalisation of swill

If procedures were put into place to ensure swill was heat-treated, support for its relegalisation was high: 76% and 77%, respectively, of pig farmers and other respondents said they would probably or definitely support the relegalisation of swill (Figure 4.3); though some were strongly opposed: in total, nine percent of respondents would definitely not support its relegalisation (Figure 4.3). Most respondents (82%) considered using swill as a traditional farming practice, and 17% thought that it was unnatural (Appendix O, Figure 9.14).

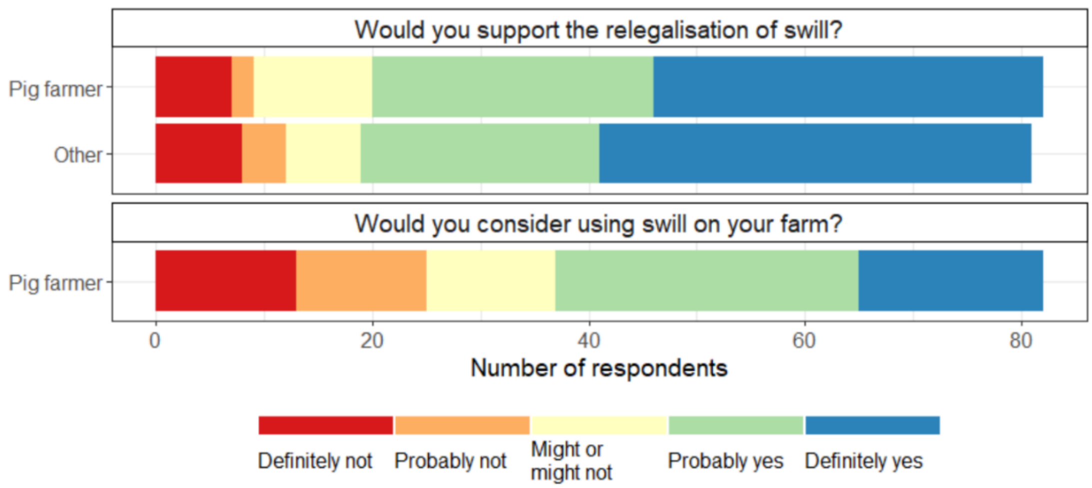

Figure 4.3 – Support for the relegalisation of swill amongst pig farmers and other agricultural stakeholders. Response to the question: "If the procedures were put in place to ensure the safety of swill (e.g. heat treatment was performed by regulated swill manufacturers), would you support the relegalisation of swill?"

Respondent's opinions on relegalisation were better predicted by their values and perceptions of swill than their characteristics (e.g. age, job, gender). Respondents for whom disease control and the perception of the pork industry by consumers were important (i.e. respondents who scored highly on the factor 1 from the factor analysis about farmer values), supported relegalisation less, while people who were more concerned with financial performance and farm efficiency (factor 2 about farmer values) were more supportive of relegalisation (Figure 4.4).

People who thought that swill would benefit the environment, help farms financially, and reduce trade-deficits (factor 1 from the factor analysis about the perceived impacts of swill) were more supportive, and people who thought that swill would increase disease risks and be unpalatable to consumers (factor 2 for the perceived impacts of swill) were less supportive (Figure 4.4).

There was little difference in support between age-groups (i.e. the difference between age groups was close to zero, Figure 9.17 in Appendix O). While the model including an interaction between gender and job had the lowest WAIC (Table 4-3), suggesting that female pig farmers were more supportive of relegalisation, the importance of gender and job in predicting support for the relegalisation of swill should be treated with caution. First, our data included only a small sample of female respondents (21/163 respondents). Second, we suspect that the gender difference in the highest weighted model may be partly explained by other, latent variables. In our sample, 1/7 female farmers were affected by the 2001 foot-and-mouth outbreak (14%), while 21/74 male farmers were affected (28%), which may in part explain why female farmers had higher acceptance for swill-based feeds. Farmers who were affected by the 2001 foot-and-mouth outbreak were less likely to support the relegalisation of swill (Figure 4.5). Finally, the measured difference between jobs was close to zero (Figure 4.4), suggesting similar levels of support for the relegalisation of swill among pig-farmers and other agricultural stakeholders (Figure 4.3).

Table 4-3 – Models predicting support for the relegalisation of swill, amongst all respondents (n=163).

	Predictors								Model output		
Model ID	Age group intercepts	Gender	Job	Job:gender interaction	Respondent's values: 1st factor loading	Respondent's values: 2nd factor loading	Respondent's perception of swill: 1st factor loading	Respondent's perception of swill: 2nd factor loading	WAIC	pWAIC	Model weight
AR1	Y	Y	Y	Y	Y	Y	Y	Y	342.9	13.4	0.12
AR2	Y	Y	Y	-	Y	Y	Y	Y	345.9	12.1	0.03
AR3	-	Y	Y	Y	Y	Y	Y	Y	340.6	11.8	0.39
AR4	-	Y	Y	-	Y	Y	Y	Y	344.2	10.7	0.06
AR5	Y	Y	-	-	Y	Y	Y	Y	344.0	11.1	0.07
AR6	Y	-	Y	-	Y	Y	Y	Y	347.1	10.6	0.02
AR7	-	-	Y	-	Y	Y	Y	Y	345.3	9.6	0.04
AR8	-	Y	-	-	Y	Y	Y	Y	342.2	9.7	0.17
AR9	-	-	-	-	Y	Y	Y	Y	343.1	8.4	0.10

Models are listed in order of their Akaike weights. WAIC is the widely applicable information criterion score, pWAIC is the number of effective parameters. Model equations are listed in Appendix O.

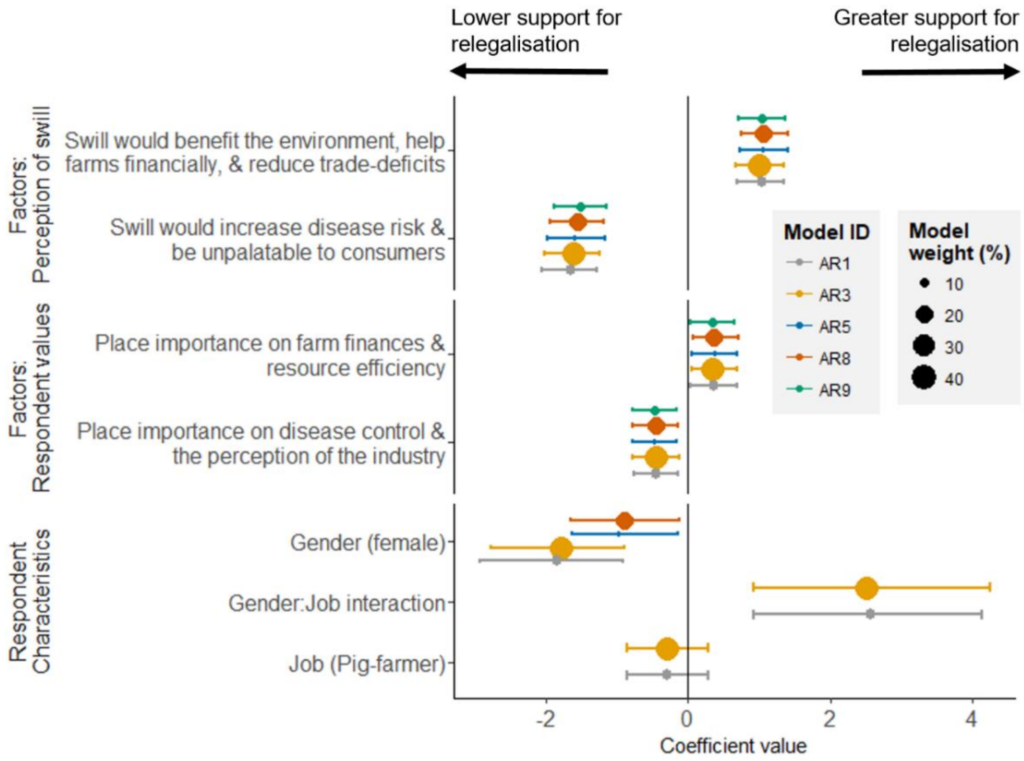

Figure 4.4 – Predictors of the support for the relegalisation of swill, among all respondents (n=163). The estimates plotted are from the five models with the greatest weighting (85% of model weight), where different colours are used for each model (listed in Table 4-3) and model weights are proportional to the size of the points. Error bars are 89% credible intervals. For clarity, the coefficients for age groups, which was included in two models, are not plotted here; these are shown in Appendix O, Figure 9.17.

Amongst pig farmers, there were no differences in support for relegalisation between farm sizes or age groups (models including these parameters had low model weights; Table 9-14 in Appendix O). Similar to the model fit to all respondents, farmer's values and perceived impacts of swill were also important predictors of support for relegalisation (Figure 4.5). Farmers who used wet feeding were less likely to support relegalisation (Figure 4.5, equivalent to a 0.37 lower score for support for relegalisation; Figure 9.18 in Appendix O). Models including whether farmers had previous experience using swill (which increased support for relegalisation), gender (female respondents showed greater support for relegalisation), and whether farms were directly affected by the 2001 foot-and-mouth outbreak (lower support if the farm was directly affected) were weighted almost equally, suggesting these may have played a role in farmer willingness to support swill, but their relative importance is uncertain.

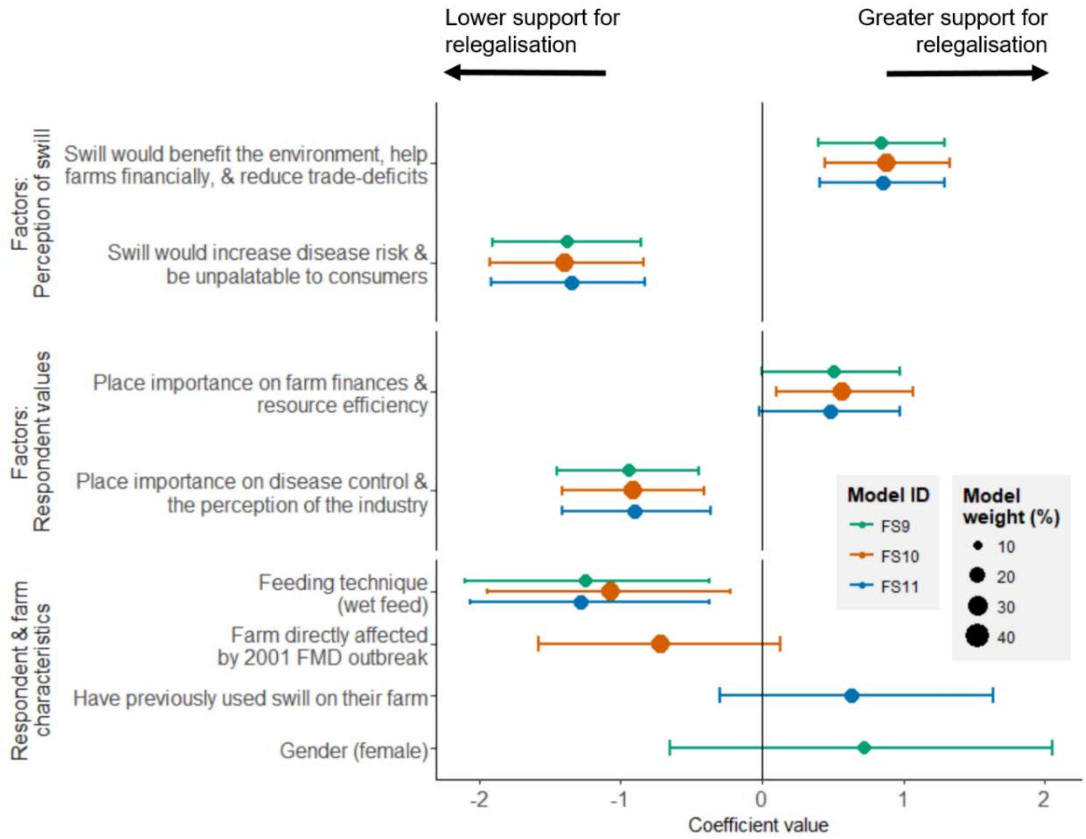

Figure 4.5 – Predictors of farmer support for the relegalisation of swill (n=82). The top three models shown had 55% of the model weight, and the structure of all models are listed in Appendix O Table 9-14. Error bars are 89% confidence intervals.

A lower proportion of respondents (55%) were willing to use swill on their farm if it were relegalised than supported relegalisation per se (Figure 4.3). In models exploring willingness to use swill on their farm, the perceived impact of swill was more important than the farmer's values (Figure 4.6 and Appendix O, Table 9-15). Unlike the model predicting support for relegalisation, the farmer's experience of foot-and-mouth disease, feeding technique, or gender did not affect their willingness to use swill. According to the model which had the greatest weight (Appendix O, Table 9-15), farmers who had previously used swill had a 0.6-point higher willingness to use swill, if it were relegalised (Appendix O, Figure 9.19).

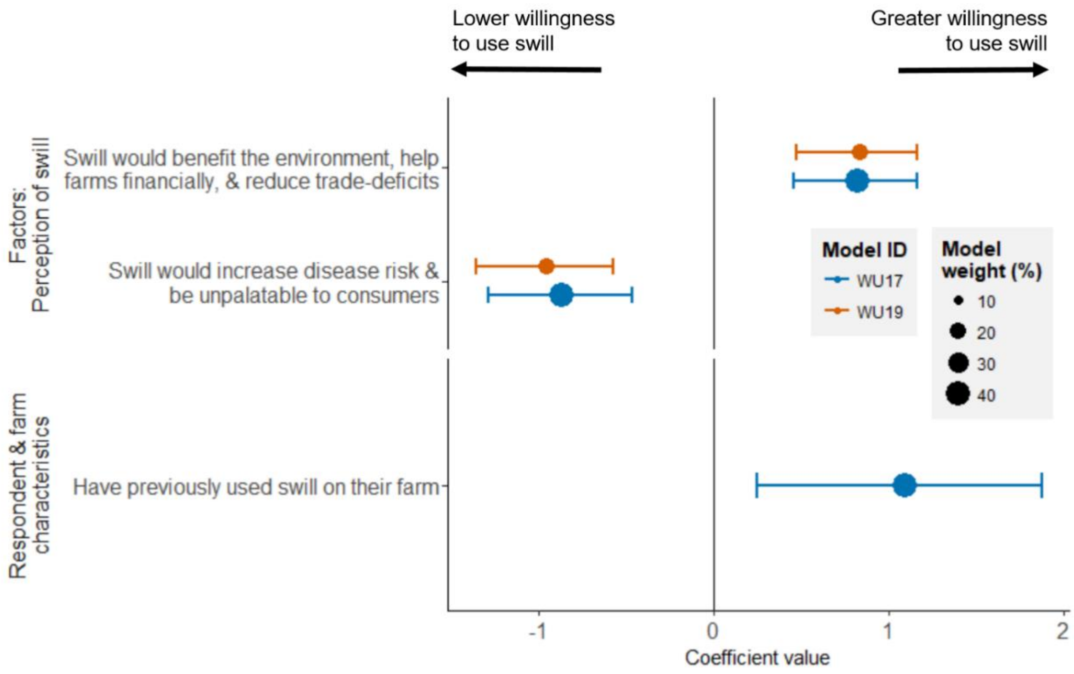

Figure 4.6 – Predictors of farmer willingness to use swill, if it were relegalised (n=82). The top two models shown had 60% of the model weight; all models are listed in Appendix O, Table 9-15.

4.5. Discussion

While respondents thought that feeds containing animal by-products and with a potential for intra-species recycling were less acceptable than those without, and pig farmers showed a preference for feeds that are currently legally permitted, this did not translate to support for the current ban. Overall, we found high support (>75%) for the relegalisation of swill, among both pig farmers and other agricultural stakeholders. Support for swill feeding arose in part because respondents thought that the relegalisation of swill would lower costs, increase profitability, and be better for the environment (Figure 4.2 and Figure 4.4), perspectives supported by previous literature on the economics and environmental impacts of using food losses as feed (chapter 2; chapter 3; Brancoli et al., 2017; Vandermeersch et al., 2014).

Though fewer farmers were willing to use swill on their farm than would support its relegalisation, more than half of all farmers reported that they would consider using swill, if it were relegalised (Figure 4.3). While farmer values predicted their support for relegalisation, they did not predict their willingness to use swill on farm (Figure 4.6), suggesting, perhaps unsurprisingly, that the business decision about which feeds to use are based more on the practicalities of using a particular feed than less tangible "values".

Our results also confirm the critical importance of disease control and consumer communication when considering the relegalisation of swill. Many respondents were concerned that using swill could increase the risk of a disease outbreak, and lower the chemical and microbiological safety of the feed (Figure 4.2) – and these concerns influenced their position on relegalisation. Respondents who thought that swill would increase disease risks and be unpalatable to consumers were less supportive of relegalisation (Figure 4.4). Perhaps reassuringly, a survey of 1500 consumers in Japan found that they did not perceive Ecofeed-fed pork (Ecofeed is the Japanese name for feed derived from food losses), differently from conventional pork (Sasaki et al., 2011), and we also found a strong consensus that respondent's thought of swill as a traditional, and not an unnatural, feed – contrary to the fears of some UK supermarkets (Stuart, 2009).

Farmers' experience was also important in determining their position on relegalisation. Farmers who were directly affected the foot-and-mouth outbreak in 2001 (caused by the illegal feeding of uncooked swill) were less likely to support relegalisation, and farmers who had experience using swill in the past were more supportive. While we found that women were, on average, more supportive of the use of swill, we caution against placing a lot of weight on this finding, given the small sample size.

Contrary to our expectation, farmers who used wet feeding were less supportive of relegalisation. As swill has traditionally been fed as a wet feed, these farmers would, in theory be better placed to use many sources of swill, if it were relegalised. Since swill is also fed as a dehydrated pellet in the modern systems of swill feeding in East Asia (Sugiura et al., 2009), and wet/dry feeding did not predict actual willingness to use swill, further research is needed to identify the underlying cause for the difference in support for relegalisation between these two groups.

4.5.1. Study limitations

Our sample size is not trivial – we sampled 82 pig farmers, including 60 who have farms larger than 1,000 animals, making up approximately 4% of the 1,410 large pig farms in the UK [7]. Our results therefore indicate that there is support for the relegalisation of swill among even large-scale producers – a group that previous work has suggested would be less supportive than smallholder pig farmers (Fairlie, 2010). As with all voluntary surveys, there is, however, the possibility of sample selection bias, where people with strong opinions on the topic chose to complete the survey, while people with less strong opinions are under-sampled. From our observation of the two days of the fair, we believe that this concern is not the case here. Many respondents completed the survey at the stand and appeared to be as much motivated by the opportunity to sit and sample complimentary food and drink, and enter a prize draw, as by the specific topic of the survey.

While our sample is representative of the attendees of the UK's largest pig and poultry trade fair, it is not clear how generalisable our results are to other countries. As the UK was the hardest hit by the 2001 foot-and-mouth outbreak, and other European countries had well developed swill feeding industries, prior to the ban on the use of swill (Fairlie, 2010), it is possible the UK agriculture sector may be more sceptical of the relegalisation of swill than in other countries. Future work should also evaluate support for the use of swill among other groups, such as the retail sector or the general public.

Our questionnaire asked whether respondents would support the relegalisation of swill, *if* practices for its safe inclusion in feed were introduced. Of course, questions remain about how the technologies and system for recycling food losses operating in East Asia could be best adapted to suit the UK or European context. A UK government risk assessment, for example, concluded that heat-treatment is sufficient to render food losses containing animal by-products safe for animal feed – the risk for the introduction of diseases comes, however, not from the failure of the heat treatment process itself, but from contamination of feed with material that evaded heat treatment (Adkin et al., 2014). Any new system for the use of swill will therefore require careful design of regulation and operating procedures to reduce the risk of uncooked animal by-products entering feed to a negligible level. Our results suggest, however, that if such a system for safe swill feeding can be established, there would be widespread support amongst UK pig farmers and other agricultural stakeholders for its relegalisation.

Alta Floresta (top left), a municipality in northern Mato Grosso, lying within the Amazon biome. Once a carpet of forest, today around 45% of land has been cleared for cattle ranching (bottom left). Colonised during the 1970s, the original plane (top right) that was used to bring in supplies before the development of the road network now stands in the town centre. Initiatives like the Novo Campo (bottom right) are making efforts to increase cattle productivity. Source: author.

5. Lessons from initiatives increasing cattle productivity in the Brazilian Amazon[4]

5.1. Abstract

Agriculture in Brazil is booming. Brazil has the world's second largest cattle herd and is the second largest producer of soybeans, with the production of beef, soybeans, and bioethanol forecast to increase further. Questions remain, however, about how Brazil can reconcile increases in agricultural production with protection of its remaining natural vegetation. While high hopes have been placed on the potential for intensification of low-productivity cattle ranching to spare land for other agricultural uses, cattle productivity in the Amazon biome (29% of the Brazilian cattle herd) remains stubbornly low, and it is not clear how to realize theoretical productivity gains in practice. We provide results from six initiatives in the Brazilian Amazon, which are successfully improving cattle productivity in beef and dairy production on more than 500,000 hectares of pastureland, while supporting compliance with the Brazilian Forest Code. Spread across diverse geographies, and using a wide range of technologies, participating farms have improved productivity by 30-490%. High-productivity cattle ranching requires some initial investment (R$1300-6900/hectare or US$410-2180), with average pay-back times of 2.5-8.5 years. We conclude by reflecting on the challenges that must be overcome to scale up these young initiatives, avoid rebound increases in deforestation, and mainstream sustainable cattle ranching in the Amazon.

[4] This chapter is in review at Agriculture, Ecosystems & Environment; a pre-print is available at: https://agrixiv.org/axyjk. The text has been reformatted for inclusion in this thesis.

5.2. Introduction

There is growing competition for land use in Brazil. Beef, soy, and bioethanol production are forecast to grow 24%, 39%, and 27%, respectively, in the next decade (FIESP, 2016), even as the government has committed to reforest 12 million hectares of land and reduce deforestation – with zero illegal deforestation by 2030 (Brazil, 2017). As pasture makes up the majority of agricultural land, high hopes are placed on the potential for increases in cattle productivity to spare land and accommodate the expansion of other land uses.

The productivity of Brazilian beef production is currently low; only one-third of its sustainable potential (Strassburg et al., 2014). Brazil could in theory meet demand for beef, crops, and timber until 2040 without further conversion of natural ecosystems, by increasing cattle productivity to half of that potential (Strassburg et al., 2014). Since livestock make up 37% of Brazil's greenhouse gas emissions (Barbosa et al., 2015) and extensive cattle ranching has historically been associated with deforestation, cattle productivity improvements are also key to Brazil's climate goals. It is hoped that cattle intensification will reduce greenhouse gas emissions through land sparing (Cohn et al., 2014), increased soil carbon sequestration (De Oliveira Silva et al., 2017), and increased greenhouse gas intensity (Bogaerts et al., 2017). The Brazilian contribution to the United Nations Framework Convention on Climate Change (UNFCC), includes commitments to reduce deforestation and increase cattle productivity through the restoration of 15 million hectares of degraded pasture (Brazil, 2017).

In this chapter, we report the results from six on-the-ground initiatives which have been working to turn theory into practice: increasing the productivity of cattle ranching in the Brazilian Amazon – a region with low productivity and high potential (Strassburg et al., 2014).

First, we describe the current state of beef and dairy production in the Brazilian Amazon, before we summarize the results from six initiatives which are raising cattle productivity in the region. We show that there are many ways for cattle ranching production to be increased on existing pastureland: these initiatives are diverse in geography and the technologies adopted, and we summarize common successes and challenges faced by all. We then finish by reflecting on the risks and mechanisms for achieving wide-scale higher-productivity cattle ranching in the region.

5.2.1. Beef production in the Brazilian Amazon

Nearly one third (29%) of the Brazilian cattle herd, the second largest in the world, is found in the Amazon biome (IBGE, 2015a). Beef production in the region is characterized by extensive, pasture-based systems. Farmers traditionally keep zebu cattle breeds – 80% of cattle are Nelore *Bos indicus* (Piccoli et al., 2014) – and use few chemical inputs (e.g. fertilizers) and little active pasture management, leading to gradual soil degradation and loss of productivity (Townsend et al., 2009; Valentim, 2016). By some estimates, 40% of pastures are in a moderate or advanced state of degradation (Dias-Filho and Andrade, 2006), and cattle stocking rates are well below their potential (Strassburg et al., 2014), with little increase seen since the early 2000s (Dias et al., 2016). These systems are typically only marginally profitable (Bowman et al., 2012).

The cycle of pasture degradation and low profitability has meant that cattle ranching has historically been associated with deforestation: pasture makes up 60% of deforested land in the Legal Amazon region (MAPA et al., 2014). Recently, beef production and deforestation have uncoupled (Figure 5.1) and there is growing acknowledgement of the complex mix of drivers underlying deforestation. From the mid-2000s onwards, deforestation fell 70% through a combination of improvements in enforcement on private land (Börner et al., 2015), expansion of protected areas (Soares-Filho et al., 2010), market-initiatives (Gibbs et al., 2015), and an economic slowdown (Assunção et al., 2015). As deforestation again creeps upwards (Tollefson, 2016), debate continues about the relative importance of beef production, land speculation, and the rapid expansion of cropland as underlying drivers of deforestation in the Amazon (Carrero and Fearnside, 2011; Datu research, 2014; Merry and Soares-Filho, 2017; Richards et al., 2014).

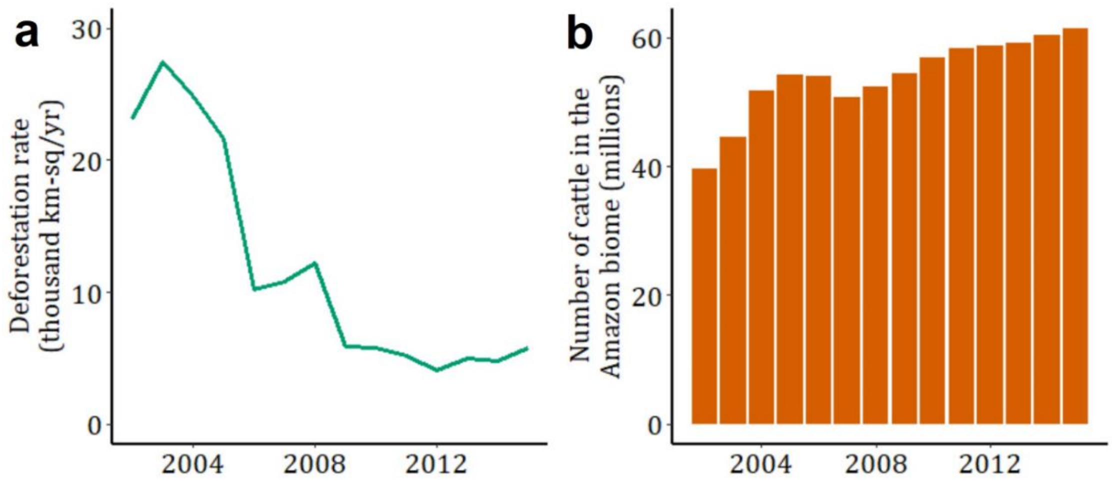

Figure 5.1 – Deforestation fell 70% from the mid-2000s onwards, (a), even as the cattle herd in the Amazon biome continued to grow (b). Data from: (IBGE, 2015a; PRODES, 2017).

Finally, Amazonian beef is becoming increasingly integrated into the global economy. Improved animal health control, such as expanding the zone of eradication of foot-and-mouth disease, has facilitated a growth in exports (Pacheco, 2012). While most beef from the Legal Amazon region is still consumed domestically, exports have more than doubled from <5% of production in early 2000s to 13.5-17.4% of production by 2011 (Appendix Q, Figure 9.20).

5.2.2. Dairy production in the Brazilian Amazon

Dairy production in the Amazon is a smaller scale operation than beef ranching. Dairy cattle make up only 3.9% of all cattle in the Amazon biome (IBGE, 2015a), which is responsible for 6.3-8.7% of Brazilian milk production (IBGE, 2015a). Dairy farming is dominated by family farms (Figure 5.2), producing milk for subsistence or the local market. These farms have up to 70 cattle per farm, with low use of chemical inputs and a strong reliance on family labour (Gomes and IMEA, 2012; Zoccal et al., 2011). Milk production is pasture-based, with some farms providing supplementary feed (e.g. sugar cane silage or concentrates) in the dry season or at the milking parlour.

Dairy productivity is therefore low and can be improved. Most dairy cattle are dual-purpose zebu breeds, though the use of dairy breeds and cross-breeds is increasing – for example, the number of registered Gir cattle (a specialized dairy breed) increased 70% (to more than 300,000 cattle) from 2007-2012 (Santana et al., 2014), though they still make up only a small proportion of the 22 million milked cows in the country

(IBGE, 2015a). Amazon municipalities have a median productivity of 689 L/cow/yr, which is lower than the median for the rest of Brazil (1,224 L/cow/yr), and lags behind other international milk producers, such as New Zealand and the European Union, which produce 3,500-4,200 and 4,000-8,000 L /cow/year, respectively (Eurostat database, 2014; IBGE, 2015a; LIC and DairyNZ, 2014).

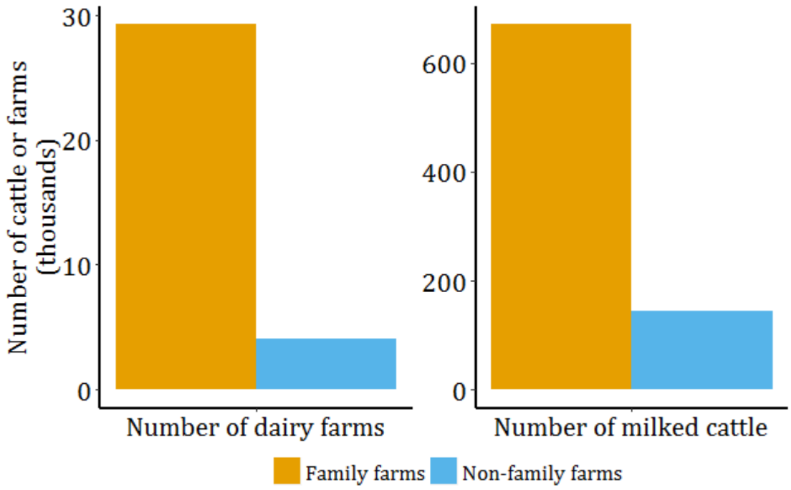

Figure 5.2 – Dairy farming is dominated by family farms, both in terms of the number of properties (left), and number of dairy cattle (right). Data from: (IBGE, 2006). Family farming in Brazil is legally defined by a maximum farm size (ranging from 20-440ha, dependent on the region), the number of permanent employees, and the proportion of non-agricultural income.

5.3. Method

A questionnaire about the financial and production performance of sustainable cattle ranching initiatives was shared with the representatives of each organization who attended a conference on sustainable cattle ranching initiatives in Rio de Janeiro in September 2015 – yielding three responses. To incorporate initiatives using a range of intensification technologies and include results for the dairy industry, three other cattle intensification initiatives in the Amazon region were then contacted through the Brazilian Roundtable for Sustainable Beef (Portuguese acronym, GTPS) to invite them to participate in the project.

Two versions of the survey were circulated, one for beef and one for dairy intensification initiatives (a copy of the beef survey is in Appendix P), structured as follows. Questions were grouped into eight sections on the (i) overview of the project (name of the initiative, and institutions involved); (ii) characteristics of the initiative (the number and types of farm participating, number of cattle and area of pasture intensified, and the year the initiative began); (iii) details of the package of technologies implemented on participating farms (farm and pasture management, forage species, use of supplementary feed etc); (iv) the costs involved in the implementation of improved farm management; (v) the costs involved in maintenance of improved pasture; (vi) the productivity achieved on the farm, in terms of stocking rates (animal units/ha, where one animal unit is equivalent to a 450kg cow), beef production (in arroba/hectare/yr, where one arroba, abbreviated as "@", is a common Brazilian livestock unit, equivalent to 15kg of carcass deadweight), or milk production (litres of milk per cow and per hectare per year); (vii) details of other measures of performance (e.g. environmental compliance, greenhouse gas emissions); and (viii) the respondent's reflections on the barriers and opportunities for improved cattle ranching.

The surveys were completed by project managers and field technicians for each initiative. Survey data was complemented with published results from initiatives where available (e.g. (Andrade et al., 2016; Carrero, 2016; Carrero et al., 2015a; Garcia et al., 2017; Marcuzzo and de Lima, 2015; Sá et al., 2016).

5.4. Results and discussion

We provide results from six sustainable cattle intensification initiatives in the Amazon biome, four working with beef producers and two with dairy producers (Table 5-1). While one of these initiatives was launched in 1976 and introduced legume pasture technologies which have since been adopted on more than 5,000 farms, the remaining initiatives are more recent (established post-2011). These latter initiatives operate on 63 farms raising 59,000 cattle on 35,000 hectares of pasture in three states (Figure 5.3). The technologies deployed are diverse, ranging from relatively low-input leguminous systems to more input-intensive rotational grazing systems.

Each initiative has shown significant increases in productivity – boosting meat production per hectare by 30-270% and dairy production per hectare up to 490% above the regional average (Table 5-2 and Table 5-3). While the use of higher-yielding technologies is profitable in most cases, it requires initial investment to improve farm productivity, with payback times ranging from 1.5-12 years. The specifics and results of each initiative are described in more detail below.

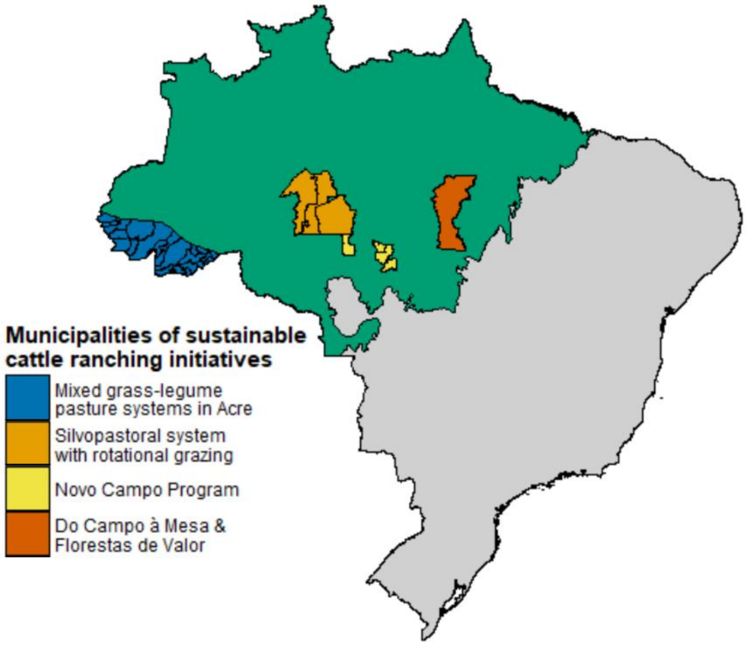

Figure 5.3 – Map of Brazil. The Amazon biome is coloured in green and the municipalities where the sustainable cattle initiatives reported in this chapter are present shown in other colours.

5.4.1. Case study #1 – Intensification of beef cattle production systems with the use of mixed grass-legume pastures in Acre

In 1976, the Brazilian Corporation for Agriculture Research (Embrapa) established the Program for Reclamation, Improvement and Management of Pastures in the Brazilian Amazon (PROPASTO) which included a series of on-farm experiments to promote the adoption of mixed legume pastures in the State of Acre (Valentim and Andrade, 2005a). A number of cultivars were launched, of which one legume, *Pueraria phaseoloides* (tropical kudzu), was the first to be adopted at scale. By 2004, tropical kudzu was present in over 30% (480,000 ha) of the total pasture area in Acre and has been successfully planted in combination with a variety of grass species (Appendix Q, Table 9-16) (Valentim and Andrade, 2005a).

Embrapa began by introducing mixed legume pastures on properties belonging to three farmers, who were identified as innovators (Valentim and Andrade, 2005a). Knowledge of these novel technologies then spread through word of mouth and trained agricultural extension officers. Legumes were promoted because of their ability to fix nitrogen, which reduces pasture maintenance costs and produces a protein-rich sward (Figure 5.4); a pasture sown with 20-45% Tropical kudzu produces nitrogen equivalent to approximately 60-120 kg of N/ha/year (Andrade, 2010). Grass-legume associations cost between R$1350-2000/hectare to implement, and around R$100/ha/year to maintain (Table 5-4), and are therefore a relatively low-cost intensification technology for pasture restoration and intensification. Tropical kudzu pastures produce modest productivity improvements, supporting 1.5 animal units/ha and producing 4.9-12.5 @/ha/yr (Table 5-2).

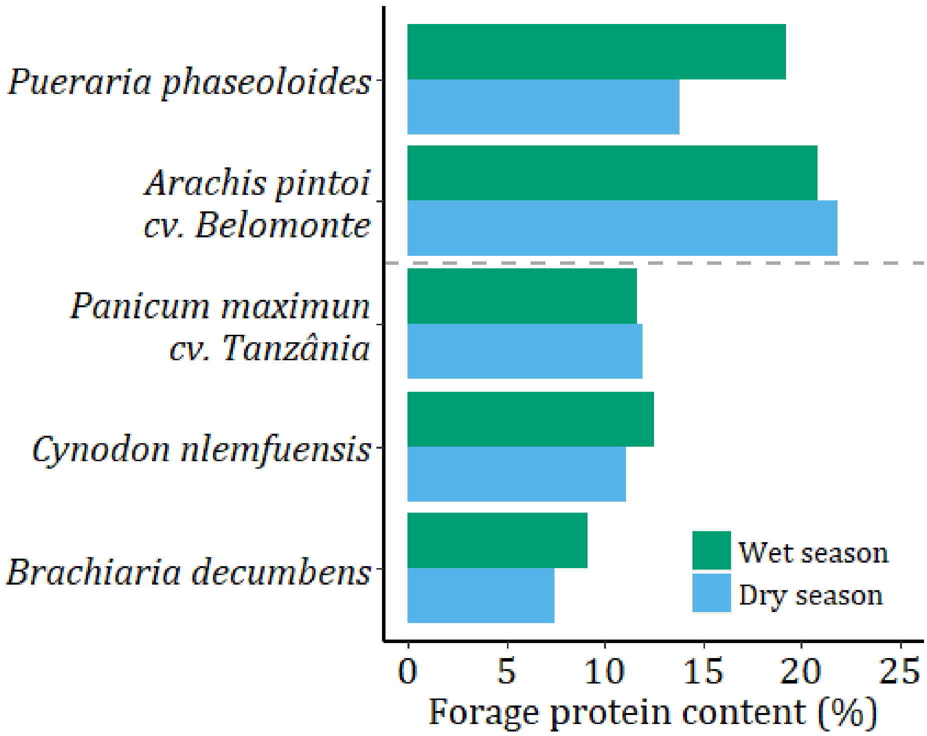

Figure 5.4 – Leguminous pasture plants (above the dashed line) have higher protein content than conventional grasses (below dashed line). Source: (Andrade, 2012).

Since a peak in the early 2000s, the popularity of Tropical kudzu has declined as it showed poor compatibility with some of the newer grass species being planted by farmers, such as African stargrass (*Cynodon nlemfuensis*), and also failed to persist when managed in mixed pastures with other grasses under rotational stocking at stocking rates above 1.5 animal units per hectare. For these situations, Embrapa promoted forage peanut cultivar Belomonte (*Arachis pintoi*) (Valentim and Andrade, 2005b).

This new cultivar was released in 1999 in Bahia, Brazil. First planted by a single farmer in Acre in 2000, in April 2001 20 farmers planted this legume together with a variety of grasses (Appendix Q, Figure 5.5 and Table 9-16). Adoption was rapid. By March 2004, close to 1000 small, medium and large farmers of Acre had already introduced forage peanut into 65,000 ha of pasture (Valentim and Andrade, 2005b), and by 2015 forage peanut was planted across 2000 farms and 137,000ha in Acre (Sá et al., 2016), approximately nine percent of the state's pasture area (INPE, 2014).

Forage peanut can be either planted along with other grasses during pasture restoration (i.e. replanting of a degraded pasture), or introduced onto existent pasture during the rainy season. Since forage peanut cultivar Belomonte does not produce seeds (it instead reproduces vegetatively), it must be planted using stolon cuttings. Farmers usually set aside an area (<1ha) where forage peanut grows in dense stands (i.e. without competing grasses), from which the vegetative stolons are then harvested for planting in pasture. Embrapa have successfully developed a number of techniques for establishing grass-legume pastures using vegetative propagation of forage peanut and stoloniferous grasses, depending on the farmer's technology level, ranging from semi- to fully-mechanized and either conventional or no-till agriculture (Andrade et al., 2016). African stargrass-forage peanut pastures managed under rotational grazing can support up to 3 animal units/ha (Table 5-2), producing Nelore × Angus crossbred steers ready for slaughter within 24 months (Appendix Q, Table 9-17), compared with the 36+ months typical of extensive systems (Valentim and Andrade, 2005b). These productivity improvements also improve the farm bottom line, increasing profitability from around R$41.10/ha/yr in traditional systems up to R$381.28/ha/yr (Table 5-2).

Figure 5.5 – Nelore cattle (*Bos indicus*) grazing mixed pasture. Forage peanut (*Arachis pintoi cv. Belomonte*) (the yellow flowering plants) together with *Brachiaria spp.*

Grass-legume pastures can mitigate greenhouse gas emissions by substituting for fossil-fuel dependent nitrogen fertilizers, by reducing cattle slaughter ages (Andrade, 2010; Valentim and Andrade, 2004), and by increasing soil carbon sequestration (Henderson et al., 2015). Additionally, Costa et al. (2016) reported that mixed pastures of *Brachiaria humidicola* and forage peanut cv. Mandobi in Acre had 24% lower N_2O emission (2.38 kg N $ha^{-1}yr^{-1}$) than pure pasture of the same grass (3.13 kg N ha-1 yr-1) and similar emissions to native forest (2.47 kg N ha^{-1} yr^{-1}).

5.4.2. Case study #2 –Novo Campo Program

The Novo Campo Program ("New Field" Program in English) has involved 23 farms in the Alta Floresta region of Mato Grosso since 2012. Led by the Instituto Centro de Vida (ICV), with collaboration with stakeholders from across the cattle supply chain (see Table 5-4 for a complete list of participating organizations), cattle productivity has been increased through a package of farm management changes. These include the introduction of pasture rotation, the adoption of so called "good agricultural practices" (GAP), correction of soil imbalances (e.g. by liming), pasture fertilization, and improved farm record keeping. The GAP is a voluntary set of "gold-standard" guidelines for sustainable production adopted across Brazil, which includes a check-list of 125 points of guidance across 11 areas of farm management, spanning farm economic management, social and environmental responsibilities, to pasture and herd management (Valle, 2013).

Together, these interventions have improved farm productivity and profitability, and reduced greenhouse gas emissions (Figure 5.6). Beef production per hectare has increased from ~4.7 to 7-24 @/ha/yr, which has reduced the cost of production per arroba on intensified farms by one third (R$66.33/@ vs R$95.80/@) (Marcuzzo and de Lima, 2015). Profit increased from less than R$100/ha/yr to more than R$600/ha/year of pasture (Table 5-2). These yield-raising technologies require an initial investment of R$1500-4000/ha, depending on the initial pasture condition, though these up-front costs are paid off after an average of 2.5 years (Table 5-2).

By improving animal growth rates and so achieving slaughter weight in fewer days, farmers reduce emissions from enteric fermentation across the animal's lifetime (enteric fermentation contributes 67-83% of emissions, excluding land use change – a topic we return to in Section 5.5 (Bogaerts et al., 2017; Cerri et al., 2016; Siqueira and Duru, 2016)). This is seen from the experience on Novo Campo Program farms. Emissions have been reduced by 36-59% (Figure 5.6), in large part through reductions in slaughter age, down to 20-24 months (Appendix Q, Table 6).

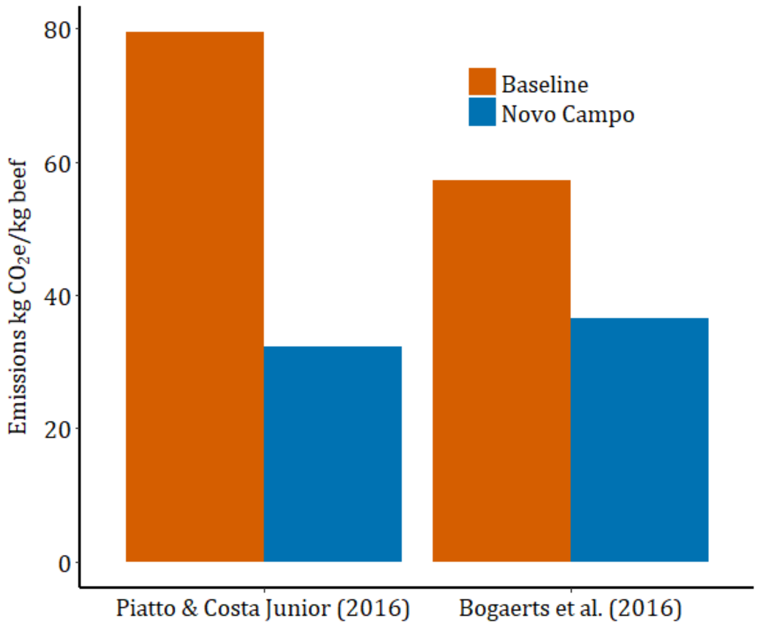

Figure 5.6 – Two estimates for the emissions per kilogram of beef of conventional ranches and Novo Campo Program farms. Piatto and Costa Junior (2016) compare emissions on pilot farms before pasture intensification (baseline), with emissions after two years of participating in the initiative. Bogaerts et al.

(2017)compare farms participating in the Novo Campo Program with neighbouring non-participating farms. While both studies include emissions from enteric fermentation, manure management, pasture fertilization, and fossil fuels required for pasture restoration, Bogaerts et al. also include emissions from concentrate feed production, and Piatto and Costa Junior (2016) include carbon sequestration in improved pasture and soil carbon emissions from degraded pasture. No emissions from land use change are included, because no recent deforestation occurred on sampled farms.

Key changes in farm management are improved book-keeping and the introduction of rotational pasture management. Adequate book-keeping is fundamental to understanding and improving farm management processes, and yet is not done by a majority of farm managers or owners (Barbosa et al., 2015). Farmers are therefore trained in the importance of recording the costs of all inputs and the quantity and value of beef produced, to allow the calculation of the income and profit per arroba of beef. Once the economic performance of the farm is established, rotational grazing is introduced.

Typically, 10-30% of the farms' pasture area is fenced off into ca. 5 hectare plots, which are targeted for pasture improvement. Pasture restoration begins with soil analysis to identify soil imbalances (e.g. pH). The pasture is then ploughed and limed (with typically 1500kg/ha lime; Appendix Q, Table 9-18), and the pasture is fertilized (400kg/ha) and replanted, with *Panicum maximum* cv. Mombaça *or Panicum maximum* cv. Tanzânia grasses (Figure 5.7).

These fertilized plots have much higher productivity than conventionally managed pasture – in the first two years of the project, they produced 20.75 @/ha/yr compared with 10.75 @/ha/year across the farm as a whole (Marcuzzo and de Lima, 2015). Cattle are moved through each fenced plot sequentially; the stocking rate and exact timing of the cattle rotation are based on the season, forage height, and planted species, manipulated to maximize cattle growth while maintaining pasture fertility. With *Panicum maximum* cv. Mombaça, cattle enter plots when the grass height is around 90 cm, and are moved when it has been grazed down to around 40 cm (approximately every five days in the wet season, and less frequently during the dry season).

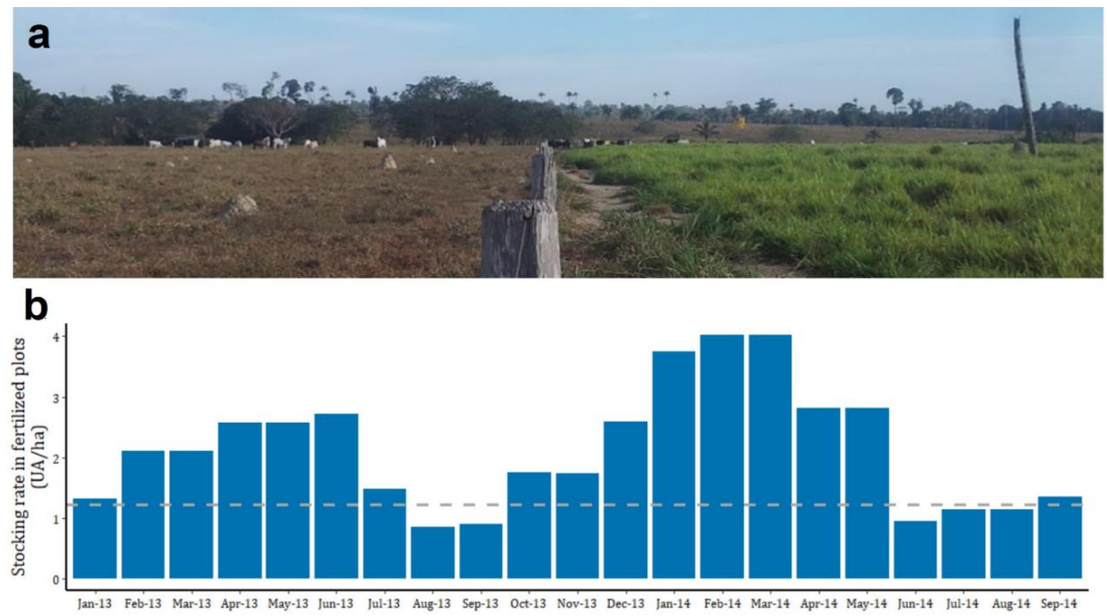

Figure 5.7 – Novo Campo pasture and forage seasonality. a) high-yielding cattle pasture (right) on a Novo Campo Program farm, one month after replanting, compared with conventional, unreformed pasture (left); b) Stocking rate in intensified pasture plots for the period January 2013 – September 2014. The grey dashed line represents the mean stocking rate for farms in the region. Adapted from: (Marcuzzo and de Lima, 2015).

While forage is in abundant supply in the rainy season (from approximately December - May), in the dry period, stocking rates in the intensified pasture areas are reduced (Figure 5.7), and supplementary feeding is necessary. Novo Campo Program farms have adopted a semi-feedlot feeding approach, where cattle are given supplementary concentrate feed in troughs in a confined area of pasture. One farm has also integrated soy and beef production to boost dry season feed availability. Soy is planted on 200ha, which is seeded with *Brachiaria* spp. after the soy harvest. This additional pasture area then serves as an additional forage source during the dry period.

All Novo Campo participating farmers qualify for the GAP certificate, developed by Embrapa. The adoption of GAP requires training of farm personnel, and ultimately, approximately a 50% increase in on-farm labour. To support the dissemination of knowledge to staff on Novo Campo farms and beyond, ICV therefore linked up with a local university, UNEMAT Alta Floresta, to train an additional 40 agricultural extension officers in GAP, environmental licensing, farm financial analysis, and the use of farm management software (Marcuzzo and de Lima, 2015).

Farms participating in the Novo Campo Program must also comply with Brazilian National Law No. 12.651, the so called 'Forest Code'. They must be registered in the rural environmental registry (Portuguese acronym: CAR), cannot be blacklisted by the environmental police (IBAMA), and must have had no illegal deforestation post-2008. On joining the Novo Campo Program, many farms had degraded riparian areas, which legally must be reforested within 20 years. Properties received support from ICV in restoring these areas, with the restoration actions depending on the degradation status and location of streams. Where streams had some secondary regrowth and/or nearby forest, this might include only fencing-off streams from cattle to foster natural regeneration; where riparian areas were more degraded or isolated, they may have required direct seeding of trees, removal of grasses and the control of pests. In both cases, restoration is not cheap – costs vary from R$2,360/ha for passive restoration, to R$9,654/ha for active replanting (IIS, 2015).

To scale-up the results achieved in the Novo Campo Program, a commercial spin-off, PECSA, was launched in 2015. While management of the Novo Campo continues under ICV, PECSA applies the same package of technologies, in some cases intensifying more than the 30% of pasture area used in the Novo Campo Program to increase farm productivity 3-5 times above the regional average (PECSA, 2017).

5.4.3. Case study #3 Do Campo à Mesa

Launched in 2013, as a collaboration between The Nature Conservancy and several partners along the beef supply chain (Table 5-4), the do Campo à Mesa initiative ("From Field to Table" in English) operates on 13 farms in São Félix do Xingu, Pará, to boost productivity through the establishment of rotational pasture and training in GAP. Results after one year of the project are promising – stocking rates increased 20% and beef productivity 30% (Table 5-2), with substantially greater gains expected; beef productivity is forecast to increase more than 3.5-times to 17@/ha (10-27@/ha) within 12 years of the start of the project (Garcia et al., 2017).

On joining the initiative, baseline data were collected by agricultural extension officers on the herd structure (e.g. number of animals, category, age, weight), farm operating costs, and soil condition. These are complemented with remote sensing analyses of the farm's land use. Many farms had considerable areas of degraded pasture: 23% and 18% of pasture was moderately or highly-degraded, respectively (Garcia et al., 2017). To combat this, management plans were drawn up to improve 20% of the farm's pasture area each year so that after five years all pasture would be in improved condition.

Goals for stocking rates were set: aiming to gradually increase stocking rates from 0.87AU/ha to 3.0 AU/ha. To achieve this, the initiative has used pasture improvement (weeding and liming, +/- resowing if degraded), education of farm workers in GAP, and the establishment of rotational grazing. Livestock are also given 1.5 kg/head/day of protein supplementary feed in the dry season, to overcome the seasonal deficit in feed availability.

The main costs of improving farming practices come in three forms: pasture improvement and maintenance, adoption of GAP, and costs of environmental compliance; these costs vary strongly with farm size. Pasture improvement requires an initial investment of between R$1300-1900/ha/yr (Table 5-4), and adoption of GAP requires not only worker education, but also improvements in infrastructure and more farm labour. After 12 years of the project, the requirement for labour is forecast to increase on average 54%, with larger increases on the biggest farms (Garcia et al., 2017). On small farms labour is family-based and will be kept constant, while large farms plan to increase their number of employees three-fold. Do Campo à Mesa farms must also be compliant with the Forest Code, which also incurs substantial costs. Compulsory restoration of deforested areas added an extra 30-250% to the cost of adopting of improved farm practices (Garcia et al., 2017). Taking all the costs of the transition to more sustainable farming practices together, pasture intensification and legal compliance generated better economic returns for large farms (>500 ha of pasture) (Garcia et al., 2017). Costs per hectare for the three smallest properties were on average 2.3 times higher than for other farms, and the two smallest farms, with 44 and 126 ha of pasture area, were not projected to make a profit and subsequently elected not to continue with cattle intensification. To help overcome the initial cost barrier and support the growth of the program, TNC helps farmers apply for loans from the Brazilian government's low carbon fund (the "Plano ABC", in Portuguese).

5.4.4. Case study #4 - Silvopastoral System with Rotational Grazing for Beef

In 2011, the Institute for Conservation and Sustainable Development of the Amazon (Portuguese acronym, Idesam) launched the Silvopastoral System with Rotational Grazing initiative ("Sistema Silvipastoril com Pastejo Rotacional", in Portuguese), on beef and dairy farms in Apuí, Amazonas. The initiative is working with 10 beef farms to boost productivity of smallholder beef production (results for dairy farms are listed in Section 5.4.5). While the planting of trees and shrubs, involves high up-front costs (Table 5-4), participating beef farms have improved productivity from 4-7@/ha/yr to 12-20@/ha/yr, and profitability from ~R$130/ha/yr to R$260/ha/yr (Table 5-2).

Farm improvement begins with a visit from an agricultural extension technician, collecting baseline farm information and drawing up management plans with the farmer. To introduce rotational grazing, an area of between 20-50 hectares is intensified on each farm by restoring pasture through the application of lime, where required. This area is then divided into six plots, sown with *Panicum maximum cv. Mombaça* or *Brachiaria brizantha*, fertilized with phosphorus, and managed in a rotational system: cattle are moved through each plot approximately every 6-7 days according to the pasture condition.

These plots are divided by double electric fences (1.5 to 2 meters of width) protecting a line of trees planted 3 meters apart. The trees are mostly native species, half of which are planted for their timber or other economic value and the other half are a mix of leguminous tree species (20-30 trees/ha), including Inga-de-metro (*Inga edulis Mart.*) Leucaena (*Leucaena leucocephala var. cunningham*) Paricá (*Schizolobium amazonicum*) Gliricídia (*Gliricidia sepium*), Jatobá (*Hymenaea courbaril*), and *Parkia* spp. Among the trees, fodder shrubs are also planted, including *Tithonia diversifolia* and Cratília (*Cratylia argentea*). The principal benefits of planting leguminous trees and fodder shrubs in pasture are that the leaves provide a high-protein feed (Cardona et al., 2014), increased shade which can reduce heat stress in cattle (Broom et al., 2013), and nitrogen-fixation which boosts grass growth and can improve soil condition (Burlamaqui Bendahan, 2015).

Silvopastoral systems do, however, require careful management and substantial initial investment. Trees need protection from heavy grazing for the first 12-24 months post-planting (Figure 5.8); thereafter they require occasional pruning if they get too broad – to avoid excessive shade hindering grass growth (Dalzell and Meat & Livestock Australia, 2006). Farmers must also closely monitor herd performance, including daily recording of stocking rates. These changes require on average a 20% increase in on-farm labour. Farmers are supported throughout the process by monthly visits from Idesam's agricultural extension staff. Costs of implementing silvopastoral systems are high, R$2400-3020/ha, though this is offset by low maintenance costs around R$220/ha/yr, in part because leguminous pastures do not require any nitrogen fertilizer application.

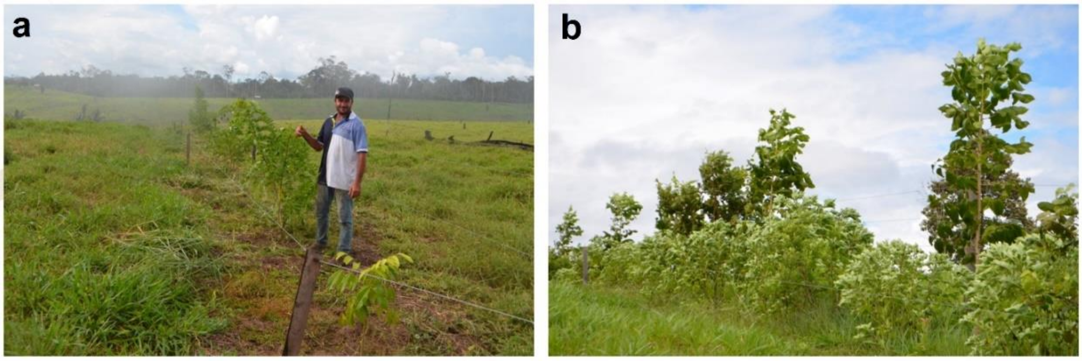

Figure 5.8 – Idesam silvopastoral systems. Left: Producer standing with a tree line of 4-month old leguminous trees. Right: silvopastoral system once the bushes and trees are 2 years old.

While it is hoped that productivity increases will reduce greenhouse gas emissions on participating farms, a recent analysis found that participating farms had higher greenhouse gas emissions than neighbouring farms (47 vs 40kg CO_2e/kg beef) (Bogaerts et al., 2017). These results should, however, be treated with caution as the analysis used an out-of-the-box greenhouse gas calculator which is not tailored for measuring emissions from integrated systems, and the input data were collected less than one year after the program's implementation. The environmental and economic impacts of integrated systems, such as silvopastoral systems, are difficult to model because the different parts of the management system interact (Marton et al., 2016) – in this case, leguminous trees fertilize the pasture, supporting grass growth. The Cool Farm Tool, used by Bogaerts et al. (2017), however, does not consider these interactions, simplifying the farm's environmental footprint and potentially over-estimating emissions. Similarly, while the Cool Farm Tool can calculate carbon sequestered in trees on-farm, this source of sequestration was not included in (Bogaerts et al., 2017). Additionally, the farm-level data used were collected shortly after the implementation of rotational grazing – emissions associated with pasture improvement were therefore counted before productivity gains had been realized. As it is expected to take five years for the systems to achieve full productivity (Table 5-2), using data from only the first year overestimates emissions from participating farms.

Finally, to participate in the initiative, farms must also be compliant with environmental legislation – they must be registered on the CAR, develop a PRAD (the "Projeto de Recomposição de Áreas Degradadas e/ou Alteradas", a plan for restoration if the property does not meet minimum legal requirements for forest cover), and agree to not clear any new areas of forest.

5.4.5. Dairy case study #1 - Silvopastoral System with Rotational Grazing for Dairy

Idesam also work with 11 smallholder pilot farms (ranging from 83-340 ha in size; Table 5-1) in the state of Amazonas, to increase dairy productivity through the rotational management of pasture lined with timber and leguminous trees, and shrubs. As for Idesam's beef intensification in the region, the dairy initiative has seen productivity improvements: a 1.26-fold increase in milk production per cow and 4.9-fold increase in milk production per hectare (Table 5-3).

Plots of intensively managed pasture are divided by doubled electric fences protecting a line of trees and shrubs. Compared with Idesam's beef system, the dairy farms use a greater number of plots (~40) and trees (50 to 110/ha). Around 6 hectares is targeted for intensive management on each farm, with 0.1 to 0.9 hectares per plot. Forty-four percent of the tree species were planted to provide shade for cattle and timber as a source of long-term income for farmers; 56% were leguminous (Carrero et al., 2015b). As often required in the region, the soil was supplemented with lime, before planting *Brachiaria brizantha*, *Panicum maximum cv. BRS zuri* or *cv. Massai* grasses, with phosphorus added as necessary. These grasses show high productivity in the shady conditions typical of silvopastoral systems (Alvim et al., 2005; Broom et al., 2013).

Laboratory experiments have shown, for example, that shade can even increase the protein content of *Panicum maximum* grasses (Silveira, 2012).

The system is managed in rotation, where cattle are moved through plots every 12h to two days, depending on grass height. In the drier months, the deep-rooted leguminous trees continue to provide a source of fodder, and some farmers supplement feed with maize silage or "cut-and-carry" feeding of *Tithonia diversifoli*a, *Inga edulis,* and *Cratylia argentea*. Lactating cattle on some farms also receive 1.5kg of maize-based concentrate feed at milking each day. Water availability is crucial for high-productivity dairy production, and so drinking water is pumped into elevated water boxes, which distribute it by gravity through a system of buried hoses to each pasture plot (Figure 5.9).

Systems with leguminous trees require approximately 15% more labour than conventional pasture-based systems. The trees require protection from grazing and insects for the first few months, as well as intermittent pruning during the first three years. The requirement for tree care reduces as the trees mature, and the rotational management of cattle in these systems requires no additional specific management, as cattle are moved twice a day for milking in any case.

The implementation of leguminous systems can be costly, ranging from R$4900-6900/ha to cover the costs of pasture reformation, tree planting, electric fencing (for managing rotational grazing), and construction of water sources in each plot (Appendix Q, Table 9-18). Though these initial costs are paid off within 2-7 years (Table 5-3), as improved management boosts profitability from R$1281.15/ha/year to around R$4425/ha/year, Idesam has provided financial support to the first farmers of the program. Farmers paid 20% of the cost of implementation, with Idesam covering the remaining 80%. To access this financial support and participate in the initiative, farmers must commit to legal compliance with the Forest Code. Farms must be registered in the CAR, commit to not deforest further, and restore non-forested areas and degraded riparian strips, in line with the PRAD.

Figure 5.9 – Silvopastoral System with Rotational Grazing for Dairy initiative. a) Installation of underground piping to deliver water to each plot; b) Dairy cattle drinking from one of water troughs, with electric fencing in the foreground.

5.4.6. Dairy case study #2 – Florestas de Valor

The dairy intensification project Florestas de Valor ("Forests of Value" in English) was launched by the Institute of Forestry and Agricultural Management and Certification (Portuguese acronym, IMAFLORA) in 2015, and operates on six farms in São Félix do Xingu in the state of Pará. By concentrating production on a small, intensively managed portion of pasture in each farm, they have increased stocking rates almost three-fold and improved productivity per cow 85%, from 1750L/cow/year to 3240L milk/cow/year (Table 5-3).

Florestas de Valor operates on small properties, ranging from 25-200 hectares in size (Table 5-1). These farms rely almost entirely on family labour, and so it is important that the intensification does not increase the overall requirement for labour – this is achieved by focusing production on a small area in each farm, where 3.5-11 hectares are selected for intensification and divided into 10-15 fenced plots (Figure 5.10). The soil in each plot is analysed before soil correction, and either direct resowing with *Brachiaria brizantha MG5*, *Panicum maximum cv. Mombaça*, *Brachiaria decumbens* or *Panicum maximum cv. Massai*, or pasture restoration through crop-livestock integration. On four properties, maize was planted on degraded pasture; once the maize was harvested, pasture grasses were then sown. The fences between pasture plots are planted with leguminous trees, including *Canavalia ensiformis*, *Inga edulis* and *Cajanus cajan*. Trees were planted three meters apart, with an average of 66 trees per hectare. Cattle remain approximately three days in each plot, thereby completing a cycle of each plot every 30-45 days. In the dry season, when grass growth is slower and over-grazing is more likely, less time is spent in the fenced plots, and cattle are instead put onto pasture that has been intentionally rested.

Overall, it costs around R$2500/ha to implement the rotational grazing and leguminous tree systems. These costs stem from costs of soil improvement, grass seeds, maize planting, fencing, solar panels, and in-pasture water sources (Appendix Q, Table 9-18). Of the 50 hectares intensified, Imaflora funded 36 hectares, with farmers covering the costs of the remaining 14 hectares. In either case, because of improvements in productivity, the total initial cost is expected to be paid off within 3-5 years.

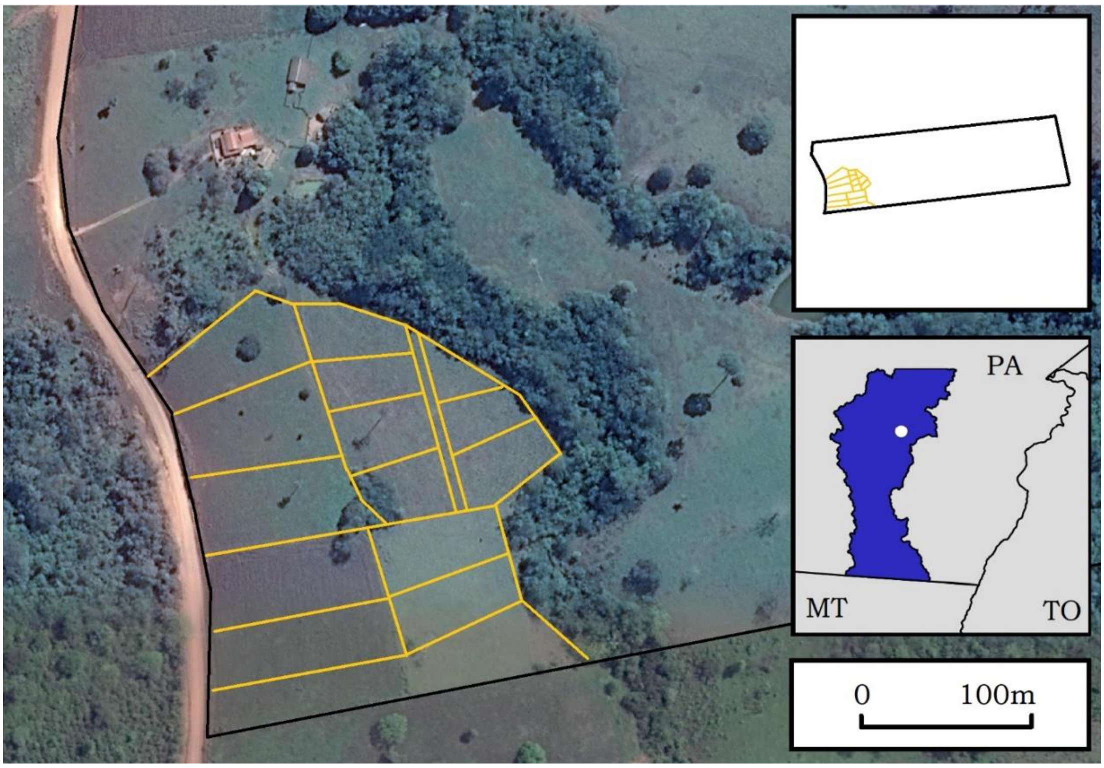

Figure 5.10 – Property map from the Florestas de Valor initiative. Around 10% of each property is divided into small plots using fences lined with leguminous trees (yellow lines in the main image). Top right: intensified pasture area shown within the total farm boundary (black line). Lower right: The farm location is shown as a point within Sao Felix do Xingu (municipality coloured in blue). State abbreviations: PA=Pará, MT = Mato Grosso, TO=Tocantins.

Table 5-1 – Characteristics of the cattle intensification initiatives surveyed. [a]Figures from 2004, the last year that production practices in the region were surveyed. [b]Estimate based on mean stocking rates and pasture area of farms. Ranges are listed in brackets, where provided. State abbreviations: MT = Mato Grosso, AM=Amazonas, PA= Pará.

Name of initiative	Lead organization	Location	Beef or dairy	Most important management features	Year project started	Number of farms	Hectares of land under intensification	Number of cattle	Mean farm size, hectares (range)	Forest Code compliance required?
Intensification of beef cattle production systems with the use of mixed grass-legume pastures in Acre	EMBRAPA	State of Acre	Beef	Vegetative planting of mixed legume-grass pastures, persistent legume supply of symbiotically fixed nitrogen	Puerária phaseoloides introduced in 1976 *Arachis pintoi* introduced in 1999	5400[a] 2000	480000[a] 137600	No data available	No data available	NA
Novo Campo Program	ICV	Alta Floresta, Nova Canaã do Norte, Paranaíta e Cotriguaçu (MT)	Beef	Pasture rotation, pasture fertilization, application of GAP	2012	23	14300	23800	200 (30-900)	Yes
Do Campo à Mesa	TNC	São Félix do Xingu (PA)	Beef	Pasture rotation, pasture fertilization, application of GAP	2013	13	20208	34043	3077 (100-6900)	Yes
Silvopastoral system with rotational grazing for beef	IDESAM	Apuí (AM)	Beef	Pasture rotation, agroforestry with timber and leguminous trees, improved book-keeping	2011	10	236	566[b]	570 (53-3020)	Yes
Silvopastoral system with rotational grazing for dairy	IDESAM	Apuí, Manicoré, Novo Aripuanã (AM)	Dairy	Pasture rotation, agroforestry with leguminous trees, improved book-keeping, and drinking water system	2014	11	95	332[b]	188 (83-340)	Yes
Florestas de Valor	IMAFLORA	São Félix do Xingu (PA)	Dairy	Rotational grazing, leguminous trees lining fenced plots	2015	6	50	145	83 (25-200)	Yes

Table 5-2 – Productivity and profitability of initiatives increasing productivity of beef production. Ranges listed in brackets, where provided. AU= animal stocking unit, equivalent to a 450kg cow; @=15kg of carcass (deadweight). Estimates of profitability do not include revenues from farm activities not directly related with cattle production (e.g. sale of timber trees or crops), and costs are representative of the interventions made on participating farms (they do not, for example, consider the cost of acquiring land or purchasing cattle, as participating farms used on-farm resources for intensification).

Name of initiative		Baseline stocking rate (AU/ha)	Stocking rate (AU/ha)	Increase in stocking rate over baseline	Baseline productivity (@/ha/yr)	Productivity (@/ha/yr)	Average increase in productivity	Years to break even on investment	Years to achieve max productivity	Typical profit/hectare/year (R$)	Additional references
Intensification of beef cattle production systems with the use of mixed grass-legume pastures in Acre	Mixed grass-Puerária phaseoloides pastures	1	1.5 (1-2)	1.5x	8 (4-10)	10 (4.9-12.5)	1.3x	3 (2-4)	2 (1.5-3)	149-271	(Andrade, 2010; Panerai, 2008; Sá et al., 2016, 2010)
	Mixed grass-Arachis pintoi pastures		2.2 (1.5-3.59)	2.2x		12 (13-35)	1.5x	4 (3-5)	3 (2-5)	296-381	
Novo Campo Program		1.22	2.8 (1.5-3.5)	2.3x	4.7	10.8 (7-27)	2.1x	2.5 (1.5-4)	Data not provided	602 (173-1140)	(Barbosa et al., 2015)
Do Campo à Mesa		0.87 (0-2.81)	1.06 (0.27-3.05)	1.2x	4.5 (0.9-10.5)	5.87 (1.42-19.2)	1.3x	8.5 (7-12)	~6	432 (-546 - 1103)	(Garcia et al., 2017)
Silvopastoral System with Rotational Grazing for Beef		0.60 (0.45-0.7)	2.4 (2.0-2.7)	4.0x	5.5 (4-7)	15 (12-20)	2.7x	5 (4-6)	5	~263	-

Table 5-3 – Productivity of dairy intensification initiatives surveyed. Ranges are listed in brackets, where provided.

Name of initiative	Baseline stocking rate (AU/ha)	Stocking rate (AU/ha)	Increase in stocking rate over baseline	Baseline productivity (L/ha/yr)	Productivity (L/ha/yr)	Increase in productivity over baseline	Baseline productivity (L/cow/yr)	Productivity (L/cow/yr)	Increase in productivity over baseline	Years to break even on investment	Years to achieve max productivity	Typical profit/ha/year (R$)
Silvopastoral System with Rotational Grazing for Dairy	0.75 (0.5-1.08)	3.5 (2.4-6.3)	4.7x	~1192	5794 (2969-9037)	4.9x	1551 (760-1825)	1954 (1642-2482)	1.26x	2.6 (1.8-6.8)	6 (5-7)	4425 (2176-8092)
Florestas de Valor	1.1 (0.9-1.2)	3.1 (2.5-3.7)	2.8x	Data not provided	~3190	Data not provided	~760	~1100	1.4x	4 (3-5)	3 (2-4)	Data not provided

Table 5-4 – Typical costs involved in each intensification initiative. Degraded pastures are pastures with declining pasture fertility; their restoration often requires soil correction, ploughing, and reseeding of grasses, whereas soil correction and ploughing may not be required for conventional pasture improvement. Ranges are listed in brackets, where provided.

Name of initiative	Organizations involved	Ranching systems	Cost of intensification (R$/ha)		Cost of pasture maintenance (R$/ha/year)	Cost of technical assistance (R$/property/year)
			Improvement of degraded pasture	Improvement of conventional pasture		
Intensification of beef cattle production systems with the use of mixed grass-legume pastures in Acre	Embrapa Acre, Federação de Agricultura do Estado do AcreFundo de Desenvolvimento da Pecuária do Estado do Acre, & Associação para o Fomento à Pesquisa de Melhoramento de Forrageiras	Cow-calf, calf raising & fattening, full cycle	Semi-mechanized conventional planting: 2011. Mechanized conventional planting: 1461-1920. Mechanized no-till planting: 1347-1806		~100	Data not collected
Novo Campo Program	ICV, International Institute for Sustainability (IIS), Embrapa, Solidariedad, Sindicatos Rurais de Alta Floresta e Cotriguaçu, JBS, McDonalds, Arcos Dourados, Imaflora, Althelia Ecosphere, Terras, GTPS, Fundo Vale, Norad, & the Moore Foundation	Calf raising & fattening	3500 (3000-4000)	2000 (1500-2000)	1800 (1500-2000)	8000 (6000-12000)
Do Campo à Mesa	TNC, Marfrig, Walmart, GTPS, & the Moore foundation	Calf raising & fattening	1890 (1750-1897)	1468 (1318-1571)	~680	Data not collected
Silvopastoral System with Rotational Grazing for Beef	Idesam, Centro para Investigación en Sistemas Sostenibles de Producción Agropecuária (CIPAV), Via Verde Consultoria Agropecuária, Fundo Vale, & Viveiro Santa Luzia	Cow-calf, calf raising & fattening	2666 (2412-3021)	All farms had degraded pasture	~216.25	~5480
Silvopastoral System with Rotational Grazing for Dairy	IDESAM, CIPAV, & Via Verde Consultoria Agropecuária, Fundo Vale, & Viveiro Santa Luzia	Cow-calf	5355 (4900-6866)	All farms had degraded pasture	~275	~5480
Florestas de Valor	IMAFLORA, CAMPPAX (Cooperativa Alternativa Mista do Alto Xingu), ADAFAX (Associação Desenvolvimento da Agricultura Familiar do Alto Xingu), CFA (Casa Familiar Rural de São Felix do Xingu), Petrobras, Fundo Vale, & Fundo Amazônia	Cow-calf	~2500	All farms had degraded pasture	2000 (1200-2560)	2500 (2000-3000)

5.5. Discussion

The results from these initiatives suggest that there are a variety of available technologies that can increase cattle ranching productivity and profitability in the Amazon. Though diverse in the details, these initiatives share many similarities, including their focus on farmer training, farm record-keeping, and improved pasture management – in particular, the adoption of rotational grazing and pasture fertilization using chemical inputs or leguminous plants. These management changes require some initial investment (R$1300-6900/hectare), which is paid off within 2.5-8.5 years.

5.5.1. Risks of cattle intensification

With the exception of the introduction of grass-legume pastures in Acre, the initiatives presented are young and further productivity gains are expected, with productivity expected to peak 1.5-7 years after implementation (Table 5-2 & Table 5-3). These promising results come with a number of caveats, however. First, our review is not exhaustive: we include results from six initiatives which we believe are representative of high-yielding cattle ranching in the Amazon, though we have not captured all the cattle intensification initiatives operating in the region. At least seven other sustainable cattle ranching initiatives exist in the Amazon biome (Appendix Q, Table 9-19).

Increasing cattle ranching yields is also not without risks (Latawiec et al., 2014). Improving productivity can reduce greenhouse gas emissions from beef production, as seen in results from the Novo Campo Program (Figure 5.6), other intensification initiatives in the region (Bogaerts et al., 2017), and life cycle assessments of beef production in other parts of Brazil (Cardoso et al., 2016; Palermo et al., 2014). However, by increasing the profitability of cattle ranching, there is a risk that cattle intensification may incentivize local deforestation– a rebound effect known as Jevon's paradox. While coupled economic-environmental modelling suggests that the adoption of GAP in Brazil could halve greenhouse gas emissions from deforestation and agriculture (Cohn et al., 2014), to achieve the government target of zero illegal deforestation or even deforestation-free production will require explicitly linking improvements in cattle ranching with habitat protection and efforts to reduce leakage in cattle supply chains (Alix-Garcia and Gibbs, 2017; Phalan et al., 2016). The recent initiatives described in this chapter therefore explicitly require participating farmers to comply with the Brazilian Forest Code, have no recent illegal deforestation, and develop land use plans for reforestation where required. Further discussion of potential risks is included in Appendix Q.

While the technologies discussed in this chapter can increase farm profitability, this is also not always the case – results from two of our initiatives show that pasture intensification may, in some cases, be more profitable on large, rather than small farms. In the do Campo à Mesa initiative, pasture intensification was not profitable within twelve years for the two smallest farms, suggesting that there is a tipping point in economic returns between 126 and 425 ha of pasture (Garcia et al., 2017). Similar economies of scale were found in a modelling study using data from the Novo Campo Program (which found that the introduction of GAP and rotational grazing intensification was only profitable on farms with >385 ha of pasture; IIS, 2015), and in other studies of cattle ranching economics (Barbosa et al., 2015; Harfuch et al., 2016b). On the other hand, these economies of scale appear to be technology and system dependent. Positive economic returns were seen for smallholder dairy producers, and silvopastoral beef systems in Apuí (Table 5-2 and Table 5-3), which can turn a profit with as little as 20 hectares of pasture. Similarly, grass-legume pastures have been adopted by small- and large-farms alike in Acre (Valentim and Andrade, 2005b; Valentim and Carneiro, 1998). Given that 78% of cattle-rearing farms (hosting 33% of cattle) in the Amazon biome have less than 200 hectares of pasture (Appendix Q, Figure 9.21), it is important that efforts to improve profitability and farmer livelihoods in the cattle sector include both large- and small- landholders.

5.5.2. Barriers to scaling up sustainable cattle ranching

Cattle production in Brazil is set to grow – the Brazilian government recently set ambitious targets for increasing beef and dairy production by 40% (MAPA, 2014a). The sustainable growth of the industry is not, however, guaranteed. To ensure that the cattle industry develops sustainably, improving farmer livelihoods while protecting the environment, will require a mix of the right financial incentives, efforts to support training of rural workers and agricultural extension services, and improved monitoring of cattle supply chains.

Most cattle ranchers adopt good agricultural practices because of the expected improvements in productivity and profitability (Latawiec et al., 2017). Implementation costs and difficulty in accessing credit are however barriers for many producers. Among producers surveyed in Mato Grosso regarding adoption of good agricultural practices, 18% cited financial constraints as a barrier to adoption (Latawiec et al., 2017), and respondents from four of the six cattle initiatives described in this chapter also perceived high implementation costs as an important barrier (Appendix Q, Figure 9.22).

Sustainable growth in the cattle industry must also combine productivity improvements with the protection of native vegetation. In the Amazon biome, farmers are required to keep 80% of their land under forest (the threshold is set at 50% for small properties and properties in Ecological and Economic Zoning areas), and must reforest any land deforested above this threshold, or purchase certificates through the nascent forest trading scheme to compensate (Soares-Filho et al., 2016). As the forest certificate market is not yet operating at scale, farmers currently rely on on-site reforestation for legal compliance, and the costs can be substantial. The do Campo à Mesa and Novo Campo Program initiatives report reforestation costs of R$868-6068/ha and R$2360-9654/ha, respectively. These figures are roughly equivalent to the costs of pasture intensification and implementation of GAP. Unfortunately, even where farmers can access credit, no credit lines currently support costs of compliance with the Forest Code (Garcia et al., 2017).

The financial barriers to improved cattle production and compliance with the Forest Code can, however, be overcome by developing the right private and public incentives for producers (Appendix Q, Figure 9.22). Currently, farmers receive the same price for their product, regardless of their environmental management. This could be fixed by the development of sustainable beef price premiums and certification schemes, such as the "Standard for Sustainable Cattle Production Systems" developed by the Sustainable Agricultural Network, which delivers a financial reward to producers implementing good practices (Newton et al., 2015). Similarly, agricultural credit can be leveraged for sustainability, by making access to agricultural credit contingent on the adoption of sustainable ranching practices, and by supporting the costs of meeting the requirements of the Forest Code.

As an example of sustainable credit, in 2010 Brazil created the landmark ABC Program, one of the world's first credit lines for low carbon agriculture (Lopes and Lowery, 2017), which supports the costs of restoring degraded pasture and the implementation of integrated crop-livestock-forestry systems. The impact of the ABC Program has, however, been hampered by bureaucratic issues, unfavourable interest rates, and a lack of public awareness. Producers perceive the ABC program as being complex, slow, and overly bureaucratic (Latawiec et al., 2017) – posing a particular problem for small producers (Gurgel et al., 2017). The ABC Program is often out-competed by other credit lines: its interest rates (7.5-8% per year) are double that of loans available through the National Rural Credit System, Brazil's main source of agricultural credit (Lopes and Lowery, 2017). Public awareness is also a problem. Surveys in the Alta Floresta region of Mato Grosso show that most producers haven't heard of the ABC Program and are not familiar with the concept of sustainable credit lines (Gurgel et al., 2017). Only 10% of the ABC Program's budget is spent in the northern states, which make up the majority of the Amazon biome (Gurgel et al., 2014). Overall, sustainable credit lines make up only 1.9% of all agricultural credit in Brazil

(Lopes and Lowery, 2017), and there are currently no sustainable credit lines which are specifically aimed at smallholders, though these could be created within the existing National Program for Strengthening Family Agriculture (Portuguese acronym, PRONAF) (Lopes and Lowery, 2017).

The widespread adoption of sustainable cattle ranching will, of course, require more than just the correct mix of financial incentives. Barriers are also posed by a shortage of trained labour, farmer risk aversion, and the complexity of cattle supply chains. Improved farm performance cannot be achieved without the adequate training of farm staff. The lack of qualified labour is, however, acute in both beef and dairy production (Latawiec et al., 2017; Novo, 2012). Sixty-five percent of ranchers surveyed in Alta Floresta, Mato Grosso, cited a shortage of qualified labour as the main barrier to the adoption of good agricultural practices (Latawiec et al., 2017). Access to agricultural extension services is also limited (Valentim, 2016). Four-fifths of dairy farmers in Mato Grosso, for example, have never received technical assistance (Gomes and IMEA, 2012). Increasing access to technical assistance was identified as key by three of the six respondents from the cattle intensification initiatives in this chapter (Appendix Q, Figure 9.22).

Farmer psychology also plays an important role (Appendix Q, Figure 9.22). High-yielding cattle ranching costs more in the short term, though it generates positive returns in the longer-term (Table 5-2 and Table 5-3). Many cattle ranchers are, however, risk averse (Barbosa et al., 2015), and the transition from low-input, low-risk extensive systems to intensive pasture management requires a shift in mindset. Improved farm management begins with improved record-keeping, which is a foreign concept to most producers (Barbosa et al., 2015). As farmers don't keep financial records, they also often don't consider depreciation, which means that in the short-term at least, many don't recognize extensive production as a loss-making activity (IIS, 2015). Rotational grazing systems require that cattle are moved more frequently. Nelore cattle breeds have a reputation as being difficult to handle, though this is in large part because in extensive systems they are not used to contact with farm staff. While regular contact with farm laborers does improve their temperament (Ceballos et al., 2016), farmers can at first take some convincing about the feasibility of new management practices. The required shift in mindset is perhaps even greater for the adoption of silvopastoral systems and mixed grass-legume pastures. Farmers used to thinking of cattle as animals which graze grass may be initially reluctant to incorporate trees or herbaceous legumes (usually considered as undesirable species) into pasture as a source of forage and fertilizer.

These psychological barriers can perhaps be overcome by increasing familiarity with high-yielding systems, which remains low (Gurgel et al., 2017). Awareness can be raised by establishing demonstration units on real farms, as in the six initiatives described in this chapter, and open-farm field days so that local farmers can witness and learn about new management options (Appendix Q, Figure 9.22 and Figure 9.23). As Brazilian farmers' receive most of their farming advice from other farmers (Pinto et al., 2016), word-of-mouth dissemination of new technologies is critical, and can be effective – as seen in the experiences of legume pastures in Acre (Section 5.4.1). The existence of local champions, long-term commitment of key players, and strategic partnerships among local stakeholders are also key drivers in overcoming psychological barriers towards successful wide adoption of intensive cattle production systems (Valentim and Andrade, 2005b).

Finally, there are structural barriers to sustainable cattle ranching. Cattle supply chains are complex, which means that deforestation is difficult to eradicate. While market initiatives (such as the "Terms of Adjustment of Conduct" and "G4" agreements) require meatpacking companies to block sales from properties with illegal deforestation (the G4 prohibits new deforestation altogether), this applies only to properties which supply cattle directly to slaughterhouses. As cattle may be born on one ranch, reared on a second, and fattened on a third, leakage is widespread. Though these agreements have reduced deforestation among the direct suppliers of slaughterhouses, it has not led to overall reductions in deforestation (Alix-Garcia and Gibbs, 2017). To permit growth of the Brazilian beef industry while reducing deforestation will therefore require efforts to reduce leakage. This could be achieved either by monitoring the movements of individual cattle, for example, using unique ear tags, or by monitoring farm-to-farm movement of batches of cattle – information that is already collected as part of the Guide to Animal Transport (GTA) used to track animal sanitation and health, but which is not yet publicly available.

While the barriers to scaling-up high-yielding cattle ranching in the Amazon are numerous, there is cause for optimism: first, cattle productivity is already increasing in most regions of Brazil (Dias et al., 2016; Martha et al., 2012). Second, the example of leguminous pasture adoption in Acre shows that local demonstration farms can lead to technology diffusion at a regional scale in the Amazon. Third, though focused in southern Brazil, lessons can be learned from the dairy extension initiative, the Projeto Balde Cheio ("Full Bucket" project in English). The program began in 1999 in two municipalities in the states of São Paulo and Minas Gerais, where demonstration units were established on twelve farms. Operating on a budget of only R$5,000-45,000 (US$5,000-23,000) per year, agricultural extension officers from EMBRAPA worked with farmers to introduce a package of new practices, including improved farm book-keeping, soil conservation, pasture fertilization, and rotational management. On average, family farmers who joined the program increased milk production three-fold (Novo, 2012), with higher productivity arising from a combination of more lactating cows/area (31%), higher productivity/cow (24%), and better labour performance (37%), while using less land area (-7%). The initiative has since expanded, as the number of farmers assisted rose from 400 in 2010 to more than 3,000 in 2012, and is now present in 483 municipalities nationwide – including farms in Rondônia, Pará and Amazonas, within the Amazon biome (Appendix Q, Table 9-19).

5.6. Conclusion

Agricultural production in the Brazilian Amazon is growing, though questions remain about how production increases can be reconciled with Brazil's stringent restrictions on forest clearance. We present results from six initiatives which have increased the productivity and profitability of cattle ranching in the region, while also supporting compliance with the Brazilian Forest Code. Up-scaling these practices will require creating the correct mix of public and private incentives – supporting productivity increases for both large and small producers, alongside support for meeting the requirements of the Brazilian Forest Code – as well as improvements in farmer training and rural extension services, and increased traceability in cattle supply chains. If managed correctly, the Brazilian beef industry can profitably produce more on less land and thereby facilitate growth in the agricultural sector while protecting Brazil's remaining native vegetation.

Ibama law enforcement have been key in reducing deforestation in the Amazon biome. Source: Ibama.

6. Agricultural Productivity and Forest Conservation: Evidence from the Brazilian Amazon[5]

6.1. Abstract

A mix of public policy and market interventions in the mid-2000s led to historic reductions in deforestation in the Brazilian Amazon. The collateral impact of these forest conservation policies on agricultural production is still poorly understood, though evidence is sorely needed given the economic importance of agriculture in Brazil and many other forest-rich countries. We construct a ten-year panel dataset for agriculture and deforestation in the Brazilian Amazon (2004-2014), and use two complementary difference-in-difference strategies to estimate the causal effect of one of Brazil's flagship anti-deforestation strategies, the priority list (Municípios Prioritários), on agricultural production and productivity in three sectors: beef, dairy, and crop production. We find no evidence for trade-offs between agriculture and forest conservation. Rather, reductions in deforestation in priority municipalities were paired with increases in cattle production and productivity (cattle/hectare), consistent with a model where policy-induced increases in the cost of clearing new land cause farmers to shift investments from deforestation to capital investments in farming. The policy had no effect on dairy or crop production. Our results suggest that in regions with large yield gaps and where technologies for increasing yields are readily available, efforts to

[5] This chapter is in review at the American Journal of Agricultural Economics; a pre-print is available at: https://papers.ssrn.com/sol3/papers.cfm?abstract_id=3031416. The text has been reformatted for inclusion in this thesis.

constrain agricultural expansion through improved forest conservation policies may induce intensification.

6.2. Introduction

Tropical deforestation is a major cause of greenhouse gas emissions, biodiversity loss, and the erosion of ecosystem services (Baccini et al., 2012; Carrasco et al., 2017; Gibson et al., 2011). While forest conservation initiatives have had some success at reducing deforestation – notably in the Brazilian Amazon, where deforestation fell 71% from 2004-2016 (PRODES, 2017), a critical question is not only how effective forest conservation measures are at controlling deforestation (Börner et al., 2016), but what the collateral effects of forest conservation initiatives are on economic activity, and whether there are conflicts between forest conservation and economic development. Most deforestation in the tropics is for agriculture (Gibbs et al., 2010) and forest policies can displace agricultural production (Filho et al., 2015; le Polain de Waroux et al., 2016), which is a sector of high economic importance for many forest-rich countries (Barbier, 2004; Schwerhoff and Wehkamp, 2017). Questions therefore remain about potential trade-offs between forest conservation and agriculture.

In this chapter, we examine how agricultural production and productivity responded to a key forest conservation policy in the Brazilian Amazon – the government's priority municipality (*Municípios Prioritários*) list, which was introduced in 2008 (Decree no 6.321/07), as part of the action plan to curb deforestation (Portuguese acronym, PPCDAm). The priority list targets municipalities with high deforestation rates for a package of interventions, including increased field inspections and fines for deforestation (Assunção and Rocha, 2014). We exploit the policy-induced partitioning of municipalities into priority list and non-priority listed municipalities to estimate whether deforestation and agricultural productivity trends of priority municipalities shifted due to increased forest conservation. Our empirical analysis is guided by a simple model assuming a higher risk of punishments under the policy that makes illegal land expansion both more expensive and less valuable and, therefore, induces farmers in a growing agricultural market to reinvest into capital rather than land, leading to productivity gains (higher production per hectare).

Nowhere are questions about the linkages between forest and agricultural land uses more relevant than in the Brazilian Amazon, the world's largest tropical forest. The Brazilian Legal Amazon, made up of the nine states of the Amazon basin, is home to 40% of the Brazilian cattle herd and 36.5% of soy production (IBGE, 2015b, 2015a). Even as the cattle herd and soy production grew 14.7% and 94%, respectively, from 2004-2014, Brazil successfully reduced deforestation in the Amazon by 82% (PRODES, 2017). This reduction was achieved through a combination of command-and-control efforts on private land (Börner et al., 2015), expansion of protected areas (Soares-Filho et al., 2010) and the global economic slowdown (Assunção et al., 2015). The productivity of cattle ranching – the dominant land use in the region – remains low, however; stocking rates (the number of cattle per hectare – a key metric of cattle productivity) have changed little since the mid-2000s (Dias et al., 2016), in part because the expansion of cattle pasture is used as a means of land "speculation", i.e. in light of land right uncertainty farmers cut down forest for future use or sale even though they do not need additional land for production (Bowman et al., 2012).Yet, prior studies indicate that the country has a large productivity potential. Strassburg et al. (2014) show that Brazil's future demand for beef, soy, biofuels, and reforestation, can be accommodated on extant agricultural land, if beef productivity increases 50% from the current low baseline.

Our study contributes to the burgeoning policy evaluation literature on forest conservation initiatives in the Brazilian Amazon. The priority list, for example, is estimated to have reduced deforestation by 600-11,396 km^2 in its first 3-4 years of implementation (Arima et al., 2014; Assunção and Rocha, 2014; Cisneros et al., 2015), a substantial reduction, given mean annual deforestation of 6,300km^2 across the Amazon post-2009 (PRODES, 2017). We instead focus on the impact of the priority list on agriculture, which is largely underexplored in the literature, despite the stated policy objective of promoting sustainable land management (MMA, 2013). A notable exception are the incidental findings by Assunção and Rocha (Assunção and Rocha, 2014) indicating the absence of policy effects on agricultural GDP and crop production.

In a study with a similar focus to ours, le Polain de Waroux et al. (2017) examine the impact of forest conservation policies in five South American countries on soy and cattle expansion. Using panel regression for municipal data from 2001-2012, they find that only within the Amazon biome have forest conservation policies decreased pasture expansion, coincident with pasture intensification. We seek to take a step forward by (i) looking at a specific policy (rather than using a scaled policy stringency metric that could be subject to measurement error), (ii) exploiting the variation in municipality-level regulation created by the Priority List in order to provide causal effect estimates, and (iii) also discussing the mechanisms through which intensification may be realized.

More specifically, we combine satellite-based land cover data with agricultural and livestock surveys to provide a first comprehensive investigation of policy-induced intensification effects in the beef, dairy, and crop sectors. We use two different empirical strategies to deal with the non-randomness of the assignment to the priority list that makes the identification of causal policy effects difficult. First, we adopt a matching difference-in-differences (DID) estimator with two distinct features: (i) it is non-parametric and makes no functional form assumptions; (ii) it is augmented with a regression-based bias adjustment in order to mitigate any bias introduced by poor matching quality (Abadie and Imbens 2006, 2011). Second, we combine a DID approach with a synthetic control method based on entropy balancing (Hainmueller 2012). This technique addresses the difficulty of finding good matches when sample sizes are relatively small, like in the case of municipality level data.

Our findings confirm previous literature showing that the priority list reduced deforestation. We then find no evidence that the priority list negatively affects agricultural production, though we do find that the effect of the policy on agricultural productivity differs in the three examined agricultural sectors. While the productivity of cattle ranching increased in priority municipalities, so that the total cattle numbers and stocking rates (i.e. the number of cattle heads per hectare) were higher than in the absence of the policy, we find no evidence of a policy effect on absolute or relative production in the dairy and crop sectors. Overall, our findings suggest that there was no trade-off between agricultural production and forest conservation in priority municipalities, with the most land-demanding industry, beef production, experiencing large land sparing increases in productivity. This is in line with a simple economic mechanism whereby a decrease in the benefits of clearing land causes credit constrained farmers to substitute pasture expansion with capital investments in farming, which causes productivity gains and aggregate output increases, in particular when capital investment is initially low. Our findings and evidence from prior research (le Polain de Waroux et al. 2017) suggest that in regions with large yield gaps and where technologies for increasing yields are readily available, such as in the Amazon, improved forest conservation policies may induce intensification. In the discussion, we reflect on the potential mechanisms through which these productivity gains were realized.

6.2.1. The priority list

The priority list involved multiple interventions targeted at municipalities with high deforestation rates. Of the 771 municipalities making up the Legal Amazon region, 36 municipalities were included in the initial list. Formally, the criteria for being included are: (i) high total deforestation; (ii) high deforestation in the preceding three years; and (iii) deforestation increasing in at least three out of the previous five years. Listed municipalities are legally subject to (i) increased enforcement by the Brazilian environmental law enforcement, IBAMA, (ii) improved monitoring – landholdings were required to obtain a georeferenced certification (through the rural cadastre, CAR) as a precondition for authorized forest clearings; (iii) access to agricultural credit was, in theory, restricted, subject to proof of compliance with the Forest Code. Moreover, these municipalities have (iv) reputational damage – the list was colloquially known as a “blacklist” (“lista negra”, in Portuguese); and (v) positive incentives, notably financial and logistic support from international NGOs and public administrations (Cisneros et al., 2015). The external support included efforts to increase capacity in monitoring and enforcement of deforestation, support in registering properties within the CAR, and efforts to promote sustainable agricultural practices (MMA, 2015; Viana et al., 2016). To be removed from the list, municipalities must register 80% of their area in the CAR and reduce deforestation below 40 km^2/year. The priority list is updated annually and since 2008 eleven municipalities have been removed and sixteen added (Cisneros et al., 2015).

The mechanisms through which the priority list has been effective in reducing deforestation are debated. Most likely, its main effect is related to punitive measures (fines and sanctions) associated with increased controls in listed municipalities, precisely because inspection teams were primarily allocated to districts on the list (Assunção and Rocha, 2014). Others have also highlighted the importance of reputational risks (Cisneros et al., 2015). Though the priority list was further meant to tighten credit restrictions for deforesting properties, agricultural credit appears to have either been unaffected or even increased under the policy (Assunção and Rocha, 2014; Cisneros et al., 2015)

6.3. Methods

6.3.1. Conceptual framework

In this section, we discuss a simple conceptual framework inspired by some salient features of the agriculture sector in the Brazilian Amazon in order to analyse the effect of the priority municipality list on key economic variables. We consider the maximization problem of a farmer producing with capital K, already cleared land L and newly cleared land L^n. Clearing land has specific costs, so that we model it as a distinct decision variable. Capital includes machines, building material, all kinds of other physical assets and fertilizer. Total output is given by $Y = f(K, L + L^n)$. We assume that f is twice differentiable, concave and that cross-derivatives are positive (this allows for a broad range of production functions, including Cobb-Douglas functions).

Capital rental cost is given by the interest rate r. We assume that farmers utilize their already acquired land L themselves by default, because (i) land can be assumed to be a short-term fixed input factor in agricultural production (Gameiro et al., 2016) and (ii) land rental markets in Brazil are underdeveloped (Assunção and Chiavari, 2014). For this reason, farmers do not optimize over how much of the already owned land to utilize and do not take the opportunity cost into account. We normalize the price of the output to 1.

Since land tenure is not clearly defined in many parts of the Amazon (Araujo et al., 2009), clearing land can be used both to gain land as a production input, and as a means of "reserving" land for future use or sale. Cattle ranching, in particular, is the lowest-cost method of making land claims on deforested land, and land speculation is key to understanding the extensive production practices characteristic of the region (Bustamante et al., 2012; Strassburg et al., 2014). Bowman et al. (2012) show that from a simple profit maximizing logic, land acquisition for beef production is not profitable across large parts of the Amazon, though farmers do deforest to make new pasture – even though most land clearance is illegal (deforestation can be illegal in three ways: farmers (i) do not have the rights on the land; (ii) go beyond the legal limits they can use in their lands; (iii) have the rights on the land and are within the legal limits, but they do not have the authorization) – precisely because they take the future benefits of land speculation into account. In the Amazon state of Rondônia, for example, farmers expect a 10% average yearly increase in the value of deforested land (Vale, 2015).

We therefore explicitly model the possibility of reserving land. More specifically, clearing new land has a net benefit, v, which is determined by the value of attaining de facto property rights to the land, minus clearance costs. Clearance costs come in two parts: first, there are the costs of physically clearing the land, by removing natural vegetation; this cost is paid ex-ante and is thus included in the budget constraint.

Second, there are also expected punishments for illegal deforestation. The cost of the expected punishment is given by the probability of being caught multiplied by the value of fines (R$5,000/ha deforested) and/or property seized or destroyed by IBAMA, and lost revenues from having the property "blacklisted", preventing (legal) sales of livestock or crops from the property (Börner et al., 2014).

Concerning the farmer's budget, we also account for evidence that farmers in the Amazon face a binding credit constraint (Assunção et al., 2017b, 2013). Thus, we assume that the farmer must allocate his available resources M to either physically clearing new land, with a cost of cL^n or investments into physical capital K.

Taking all these features into account the optimization problem of the farmer can be written as:

$$\max_{K,L^n} f(K, L + L^n) + vL^n - rK + \lambda(cL^n + K - M),$$

where λ is the Lagrange multiplier associated with the budget constraint. The farmer chooses investments into capital and deforestation. First order conditions are $f_K(K, L + L^n) - r + \lambda = 0$ and $f_L(K, L + L^n) + v + \lambda c = 0$.

Prior to the introduction of the PPCDAm, the policy environment in the Amazon was characterized by a low expectation of punishments for deforestation and high benefits of obtaining property rights (i.e. v was high). This encouraged clearing land beyond the level that would be optimal for agricultural production in the current period. The introduction of the priority municipality list then has two potential effects. First, higher enforcement means that the expected cost of getting punished for illegal deforestation increases. Second, regularizing the use of deforested land in the listed municipalities shrinks the benefits of reserving land through illegal land clearance. Improved land regularization is indirectly achieved through increased enforcement and the CAR. While registering a property in the CAR does not confer the land a formal land title, it does reveal the level of compliance with the Brazilian Forest Code, identifying what must be regularized (i.e. reforested) – at the land owner's (sometimes considerable) expense (Azevedo et al., 2017; Garcia et al., 2017). The priority list thus had the effect of decreasing the value v of clearing land in affected municipalities. Next, we study the implications of this decrease in v.

Proposition. A decrease in the benefits of clearing land v causes (i) a reduction in the amount of newly cleared land $\mathrm{L^n}$, (ii) an increase in capital investments K, (iii) an increase in total output, Y, and (iv) an increase in output per unit of land, $\frac{\mathrm{Y}}{\mathrm{L+L^n}}$.

Proof. (i) We rewrite the first FOC to $\lambda = r - f_K(K, L + L^n)$ and the credit constraint to $K = M - cL^n$. Inserting both into the second FOC we obtain $f_L(M - cL^n, L + L^n) + v + (r - f_K(M - cL^n, L + L^n))c = 0$, which defines the optimal level of L^n.

Using the optimality condition we define an auxiliary function $H = f_L(M - cL^n, L + L^n) + v + (r - f_K(M - cL^n, L + L^n))c$. We have $\frac{\partial H}{\partial L^n} = -cf_{KL} + f_{LL} + c^2 f_{KK} - cf_{KL} < 0$ and $\frac{\partial H}{\partial v} = 1$. Applying the implicit function theorem we have $\frac{dL^n}{dv} = -\frac{\frac{\partial H}{\partial v}}{\frac{\partial H}{\partial L^n}} > 0$.

(ii) By the credit constraint we have $\frac{dK}{dv} = -\frac{dL^n}{dv} < 0$.

(iii) We have $\frac{dY}{dv} = \frac{df(M - cL^n, L + L^n)}{dv} = -cf_K\left(-\frac{dL^n}{dv}\right) + f_L \frac{dL^n}{dv} = (f_L - cf_K)\frac{dL^n}{dv}$. We have $\frac{dL^n}{dv} > 0$. Combining FOCs we have $f_L - cf_K = -cr - v$. This expression could be positive in principle, because clearing land could be beneficial even for $v < 0$ due to the positive contribution to production. Based on the above discussion, we consider $v > 0$ the more likely case, however. In addition, if $-cr - v > 0$, the credit constraint is not binding, meaning that farmers would prefer not to invest their entire available resources into deforestation or into purchasing inputs to their farm. Based on the empirical evidence we assumed that the credit constraint is binding so that we have $-cr - v < 0$. As a result, we have $\frac{dY}{dv} < 0$.

(iv) Follows immediately from (iii) and (i). □

Prior to the introduction of the priority list, clearing land for future use is even more profitable than investments in already cleared land. This causes an efficiently low level of investment. When the priority list is introduced, clearing land becomes less attractive. As a consequence, farmers shift their investment from clearing land to investing in capital, leading to productivity gains and aggregate output increases. The data requirements to fully test this structural model prediction are demanding, in particular due to lack of capital data on a locally disaggregated level. We therefore resort to a reduced-form test focusing on the policy effect on directly observed production and productivity measures in different agricultural activities at the municipality level.

We expect that the policy will affect our three agricultural sectors differently. Beef production is the dominant land use in the region – cattle pasture makes up 65% of deforested land (Almeida et al., 2016) – and is the main sector associated with land speculation; beef production is therefore the most likely to be affected by a policy-induced change in the costs and benefits of deforestation. Dairy production, in contrast, is a much smaller operation: dairy cattle make up only 5% of cattle in the Legal Amazon (IBGE, 2015a), and dairy farming is primarily practiced by family farms (Appendix R, Figure 9.24). Smallholder deforestation is less controlled than deforestation on larger properties (Assunção et al., 2017a; Godar et al., 2014), because the Brazilian satellite monitoring system cannot detect deforested patches less than 25 hectares in size, and because it is less cost-effective to patrol small clearings than large ones (Godar et al., 2014). Both these features of enforcement mean that dairy producers may not have perceived a change in the attractiveness of forest clearing. Finally, crop agriculture is also less likely to have been affected by the priority list than beef production. Agriculture in the Brazilian Amazon is dominated by soy, which made up 73% of cropland in 2014 (IBGE, 2015b). Deforestation for soy farming was, however, already constrained in the Amazon biome by the Soy Moratorium. Introduced in 2006, the Soy Moratorium prohibited the purchase of soy produced on recently deforested land; as deforestation for soy declined by 97% across the Amazon (Gibbs et al., 2015), the priority list is unlikely to have created any additional land scarcity, beyond the effect of the Moratorium.

6.3.2. Data

We combined multiple data sources for the Brazilian Legal Amazon to make our panel dataset, from 2004-2014. The Legal Amazon consists of 771 municipalities (we used administrative boundaries from 2007 to ensure consistency over the time period of analysis), but we restricted our analysis to 492 municipalities with >10% forest cover in 2002 for which all data were available, as in (Cisneros et al., 2015). Data sources for covariates are listed in Table 6-1, and data on our agricultural outcome variables are described in more detail below.

Table 6-1 – Data sources for covariate data.

Covariate	Data source
Total deforested area in 2007	PRODES
Deforestation 2005	PRODES
Deforestation 2006	PRODES
Deforestation 2007	PRODES
Forest area	PRODES
Mean rainfall	INPE
Mean temperature	INPE
Mean slope	INPE
Municipality size	Cisneros et al. (2015) PLoS ONE
Accessibility (average distance to municipality center)	Cisneros et al. (2015) PLoS ONE
Population density	IBGE Demographic Census
Municipal GDP	IBGE
Soy price	IBGE-PAM
Timber price	IBGE-PEVS
Farms density	IBGE 2006 Agricultural census
Share of small farms	IBGE 2006 Agricultural census
Percentage of land owners	IBGE 2006 Agricultural census
Tractors per farm	IBGE 2006 Agricultural census
Share of strictly protected reserves	IBAMA
Share of indigenous territory	IBAMA
Share of settlement projects	INCRA
Field-based enforcement inspections	Cisneros et al. (2015) PLos ONE
Value of subsidized agricultural credits	Brazilian Central Bank
Federal party affiliation	TSE

Beef productivity was measured as the stocking rate, the number of cattle head per hectare of pasture. No more direct measure of beef productivity (e.g. meat production per hectare) is available at the municipal level, and stocking rates have been used in previous studies of agricultural productivity in Brazil (Dias et al., 2016; Strassburg et al., 2014). The number of cattle per municipality was taken from annual municipal livestock surveys (Portuguese acronym, PPM). The primary source of information is the sales of foot-and-mouth disease vaccines, which are obligatory for all cattle under the Brazilian foot-and-mouth eradication program. Vaccine sales figures are complemented with farm surveys and are ultimately summed to form the basis of the Brazilian national contribution to FAO livestock statistics. Pasture area was calculated as the sum of four pasture classifications (Pasto.com.Solo.Exposto, Pasto.Limpo, Pasto.Sujo and Regeneracao.com.Pasto) in the high-resolution (ca. 30m^2) Terraclass dataset for 2004, 2008, and 2014. We exclude 32 (non-listed) municipalities from the analysis of beef productivity, because they have outlier stocking rates (>15 head/ha) in at least one year. In most of these cases, they are recorded as having very little pasture in at least one of the measured years – a measurement error caused by cloud cover (i.e. they also have concomitant high estimates of unobserved land).

Milk productivity was measured as the milk production per milking cow per year. This was calculated from PPM data: the total milk production per municipality was divided by the number of milking cows. Milk production figures are based on quantities of milk marketed at dairy processing plants and cooperatives, plus milk used for self-consumption or sold directly to consumers.

Our analysis of crop production focused on six major crops, which made up 87.3% of the total agricultural value of the legal Amazon from 2002-2014: soy (48.3%), maize (11.1%), cotton (10.1%), cassava (8.8%), rice (5.0%), and sugar cane (4.0%). Data on crop production, crop area planted, and crop yields came from the municipal agricultural survey (Portuguese acronym, PAM), an annual dataset based on monthly crop production statistics collected through a network of agricultural technicians and producers in each municipality. To allow aggregation of data from different crops, our main measure for crop productivity is production per hectare (i.e. aggregate gross production value divided by aggregate cropland) and not yield, though we also report results for changes in yield for each individual crop. When calculating the changes in cropland area, we corrected for the portion of land that is double-cropped with soy-maize.

In Table 6-2 we report the mean and standard deviation of our agricultural outcome variables for three non-overlapping periods: period 1 covers the years prior to the introduction of the priority list (2002-07); period 2 is the introduction year (2008);

period 3 encompasses the years following the policy enforcement (2009-14). Average productivity among priority list municipalities between period 1 and 3 increased approximately by 5% and 15% for beef and milk but it decreased by 15% for crops. Among non-listed municipalities, beef productivity decreased by 4%, while milk and crop productivity increased by 15% and 7%, respectively. Table 6-2, however, also illustrates that productivities in period 1, i.e. prior to the policy enforcement, are very different between priority list and non-priority list municipalities, highlighting that the simple comparisons are only explorative.

Table 6-2 – Summary statistics for outcome variables. Standard deviation in parentheses.

	Cattle head/hectare of pasture		Thousand liters of milk/ milked cow		Gross crop production value/ hectare of cropland	
	Non-priority list	Priority list	Non-priority list	Priority list	Non-priority list	Priority list
Period 1 (2002-2007)	1.62 (1.58)	1.27 (0.57)	0.62 (0.30)	0.88 (0.27)	2.54 (4.05)	2.96 (6.23)
Period 2 (2008)	1.55 (1.33)	1.23 (0.54)	0.63 (0.30)	0.92 (0.29)	2.09 (2.19)	2.09 (1.37)
Period 3 (2009-2014)	1.56 (1.35)	1.33 (0.47)	0.71 (0.41)	1.01 (0.34)	2.73 (7.90)	2.51 (4.22)

6.3.3. Empirical strategy

Our "fundamental problem of causal inferences" (Holland, 1986) is to construct a tenable counterfactual for deforestation and productivity trends of priority municipalities in the absence of their inclusion on the priority list. While a simple but naïve counterfactual is provided by the selection criteria of the priority list, which leaves us with two natural comparison groups of municipalities, these two sets of municipalities differ both in their pre-2008 productivity outcomes (Table 6-2) and district characteristics also measured prior to the policy (Table 6-3). We therefore need more elaborate methods to make causal inferences about the effect of the priority list.

The basic idea of these adjustment methods is to resort to a restricted comparison of ex-ante observationally equivalent districts. Below, we discuss our two different empirical strategies to conduct this comparison (these two sets of models were fit by Nicolas Koch).

Table 6-3 – Covariate balance in full and pre-processed sample. N_T and N_C are the number of treated and control districts.

Covariate	Full sample ($N_T = 36$; $N_C = 440$)			Preprocessed sample ($N_T = 36$; $N_C = 36$)	Preprocessed sample with matching ($N_T = 36$; $N_C = 18$)
	Mean non-priority list	Mean priority list	Normalized difference	Normalized difference	Normalized difference
Total deforested area in 2007	994.77	4,539.22	1.87	0.96	0.98
Deforestation 2005	24.78	301.75	1.64	1.16	1.13
Deforestation 2006	10.37	134.08	1.19	0.96	0.95
Deforestation 2007	11.58	146.76	1.14	0.87	0.83
Forest area	0.45	0.63	0.71	0.89	0.73
Mean rainfall	2,043.81	1,975.58	-0.21	0.12	-0.06
Mean temperature	259.99	254.08	-0.64	-0.01	0.26
Mean slope	2.52	2.7	0.14	-0.09	-0.16
Municipality size	7,565.84	21,696.50	0.62	0.6	0.54
Accessibility (average distance to municipality center)	702.98	906.09	0.29	0.68	0.49
Population density	23.59	2.49	-0.29	-0.56	-0.23
Municipal GDP	5.31	7.33	0.38	0.04	0.11
Soy price	0.11	0.31	0.81	0.3	0.27
Timber price	71.25	120.56	0.88	0.11	0.14
Farms density	0.65	0.14	-0.73	-0.75	-0.67
Share of small farms	0.72	0.64	-0.47	-0.58	-0.73
Percentage of land owners	0.73	0.8	0.36	-0.23	-0.18
Tractors per farm	0.14	0.25	0.28	0.24	0.40
Value of subsidized agricultural credits	5,080,000	12,700,000	0.6	0.04	0.57
Share of strictly protected reserves	0.03	0.02	-0.11	-0.03	-0.02
Share of indigenous territory	0.06	0.13	0.45	0.46	0.28
Share of settlement projects	0.13	0.08	-0.29	-0.34	-0.21
Field-based enforcement inspections	14.83	53.38	0.82	0.61	0.81
Federal party affiliation	0.11	0.12	0.08	-0.14	0.17

6.3.4. Difference-in-differences with bias-adjusted covariate matching

Our first empirical strategy builds on the two-stage approach proposed by Imbens and Rubin (2015).

We begin with "pre-processing" our data in order to filter out inappropriate control districts from our pool of 440 non-priority list municipalities. More specifically, we use propensity score matching (without replacement)[6] to match each treated district to the closest control district in terms of their probability of being listed given observable pre-treatment characteristics. Details on the data-driven algorithm for the specification of the propensity score function (i.e. the choice among the possible correlated covariates) and the parameter estimates are provided in Appendix R. The pre-processing leaves us with sample of 36 treated and 36 control districts. Table 6-3 (columns 3 and 4) compares the normalized differences[7] in our various districts characteristics both for the original sample and the pre-processed subsample. The results show that the pre-processing substantially improves the covariate balance as most normalized differences are significantly smaller in the pre-processed sample than in the full sample. Yet, many variables still exhibit a considerable degree of imbalance.

With a more appropriate subsample of priority list and non-priority list municipalities in hand, in the second stage we estimate the average treatment effect of the priority list policy. The very magnitude of remaining imbalances, however, questions the applicability of linear regression methods. We prefer a non-parametric

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[6] Because we do not seek to create matches for specific municipalities but to create a sample with substantial overlap propensity score matching is preferable over Mahalanobis matching (Imbens and Rubin 2015).

[7] It is defined as $\frac{\overline{X_t}-\overline{X_c}}{\sqrt{s_c^2/N_c+s_t^2/N_t}}$, where $\overline{X_c}$ and $\overline{X_t}$ denote the sample averages of covariate values by treatment group, N_c and N_t are the number of control and treated units, and s_c^2 and s_t^2 are the within-group sample variances of the covariate. We focus on the normalized difference, rather than on the t-statistic, because the former provides a scale and sample size free way of assessing overlap.

estimation technique over previous conventional regression approaches (see Cisneros et al., 2015; Hargrave and Kis-Katos, 2013) because it puts no functional form assumptions on the distributions of variables. Parametric assumptions may lead to severe biases in the estimated treatment effect if substantial differences between treatment and control groups exist (Imbens, 2014). In the spirit of (Heckman et al., 1998, 1997), we implement the following non-parametric matched DID approach that compares the before-after productivity of priority list municipalities with a weighted average of before-after changes in the non-listed sample. The average policy impact is given by

$$\hat{\tau}_{DID} = \frac{1}{N_1} \sum_{j \in \mathcal{J}_1} \left\{ \left(Y_{jt}(1) - Y_{jt'}(0) \right) - \sum_{k \in \mathcal{J}_0} w_{jk} \left(Y_{kt}(0) - Y_{kt'}(0) \right) \right\} \qquad [3]$$

where $\mathcal{J}_1$ denotes the set of priority list municipalities, $\mathcal{J}_0$ denotes the set of non-listed municipalities, and N_1 is the number of municipalities in the treatment group. The priority list municipalities are indexed by j; the non-listed municipalities are indexed by k. $Y_{jt}(1) - Y_{jt'}(0)$ is the change in the outcome variable for treatment observation j between the period t' and t, which denote the time period before and after the policy. w_{jk} is the weight attached to control municipalities k when constructing the counterfactual estimate for the treated municipalities j.

The weighting in our analysis is based on a nearest neighbour covariate matching technique (with replacement). It helps us to find a control municipality in the pre-processed data that is most similar to each municipality on the priority list in terms of our full covariate space presented in Table 3. We additionally require that treated and control districts are in the same federal state to capture the effects of state regulation as well as state specific shocks. Given that we match each treated municipality with a single control municipality (using only one control district is more likely to yield an unbiased estimate of the treatment effect, albeit at the cost of sacrificing some precision; Imbens and Wooldridge, 2009), w_{jk} is set to 1 for the selected neighbor and zero for all other members of the non-listed control group. Panel A of Figure 6.1 illustrates the selected neighbours.

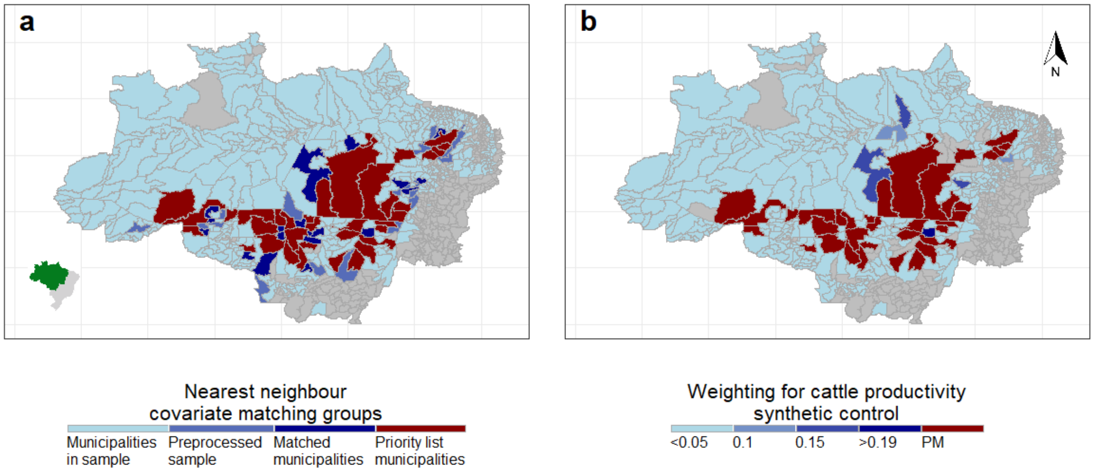

Figure 6.1 – Maps of the Brazilian Legal Amazon. (Inset: Brazil). Panel a) groups of municipalities used in the difference-in-differences with bias-adjusted covariate matching analysis. Panel b) weightings assigned to municipalities included in the difference-in-differences with entropy balancing analysis, shown for the cattle productivity outcome. Weightings sum to 1; PM = priority list municipalities. Municipalities in grey were excluded from the analysis because they have low forest cover or incomplete data sets.

The DID extension of matching is particularly effective at reducing bias since any time-invariant unobservable differences between priority list and non-priority list municipalities are eliminated by differencing post-policy outcomes with respect to pre-policy outcomes. A further important feature of our covariate matching procedure is that it is augmented with a regression-based bias adjustment in order to mitigate any bias introduced by poor matching quality. More specifically, after matching each treated district with its nearest neighbour, within-pair differences are adjusted using a linear regression of the control outcome on the covariate space as suggested in (Abadie and Imbens, 2011, 2006). Given the persistent traces of imbalance in our data even after two-stage matching (see the last column of Table 6-3), this bias-correction appears an unavoidable choice for our application. Abadie and Imbens show that the regression adjustment eliminates a large part of the bias that remains after simple matching.

6.3.5. Difference-in-differences with entropy balancing

Our second empirical strategy differs with respect to the construction of the control group. It does not take the detour via pre-processing and covariate matching, but instead constructs the control group by means of a synthetic control method. More specifically, we apply a recent reweighting technique – entropy balancing (Hainmueller, 2012) – that focuses directly on the balancing of covariates. It assigns a weight to each of the 440 non-listed control districts such that the moments of the covariates of the reweighted control group are (almost) equal to the moments of the treated group. The main advantage of using entropy balancing rather than matching techniques is an increase in balance quality.

In this chapter, the first moment of the covariates, namely the mean, of the treatment and the control group is balanced. Given our limited sample size, we also have to restrict the balancing to a limited number of covariates. We select pre-treatment trends of the outcome variables, which technically ensures that the common trend assumption is valid. Additionally, we ensure balance in deforestation trends and those variables that were shown to predict treatment: the accessibility, the share of settlement projects and the share of strictly protected reserves. We then use the weights from entropy balancing in our DID estimator given in equation (1). Panel B of Figure 6.1 illustrates the weighting of the non-priority list control districts. While the environmental and agricultural economics literature has largely overlooked synthetic counterfactual approaches (with the notable exception of Sills et al. 2015), DID with entropy balancing is emerging as an important tool in health (Markus 2013; Markus and Siedler 2015) and labour economics (Freier et al. 2015).

6.4. Results & Discussion

In this section, we present our treatment effect estimates obtained from the DID matching and DID entropy balancing estimators. Stability of estimates across the two different procedures can be interpreted as an indication that the methodologies adequately control for unobserved differences (i.e. potential biases), while instability indicates that the effect estimate is subject to caution.

We begin by re-examining the effects of the priority list on municipality deforestation trends. We then present our primary results, the agricultural productivity response in priority list municipalities, and assess the plausibility of the underlying identifying assumptions. We then reflect on the mechanisms driving the measured productivity increases, and the limitations of our study. Figure 6.2 summarizes our main findings.

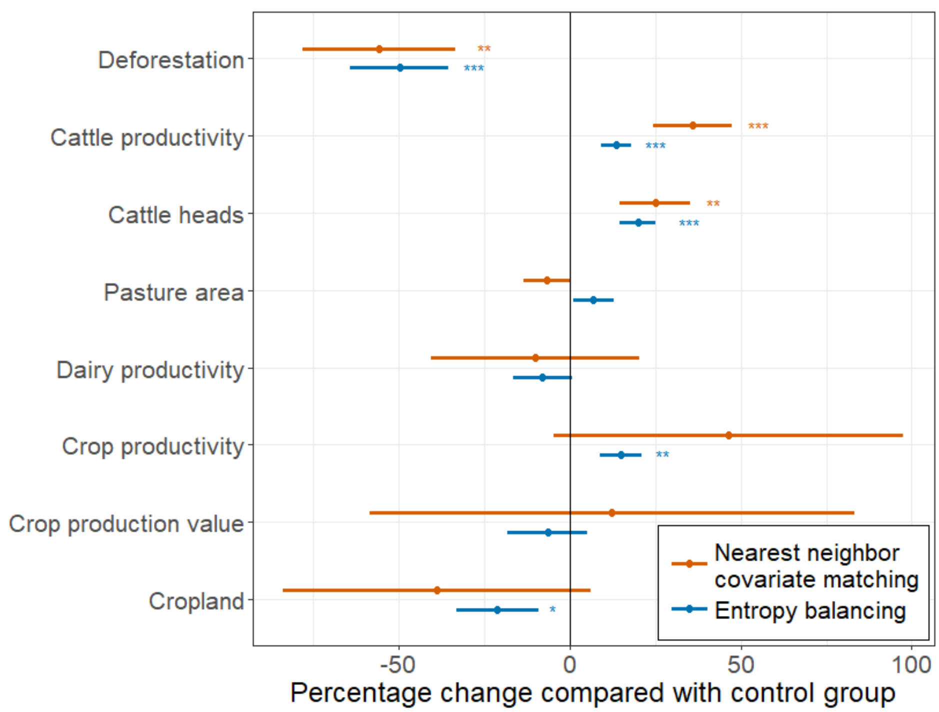

Figure 6.2 – Impact of priority listing on deforestation and agriculture. Error bars are one standard error. *p <0.1, **p <0.05, ***p<0.01.

6.4.1. Effects on deforestation

Previous literature has shown that the introduction of the priority list reduced deforestation (Arima et al. 2014; Assunção and Rocha 2014; Cisneros et al. 2015; Sills et al. 2015). However, these estimates are based on earlier time periods and/or different samples of municipalities. Most estimates also rely on strong functional form assumptions; results may therefore be sensitive to minor changes in the specification. We seek to replicate these prior findings by using flexible nonparametric DID estimators that may produce more accurate and less model-dependent causal inferences.

Our results (Table 6-4) corroborate the finding that inclusion in the priority list significantly reduced deforestation. The reported results in row (1) are based on log transformed data and can thus be interpreted as the average treatment effect in percentage terms. Our bias-adjusted matching specification shows that the treatment effect amounts to -55.8% and is statistically significant at the 5% level. When we use entropy balance to construct the control group, the treatment estimate falls to -49.9%. The effect is statistically significant at the 1% significance level. Taken together, both estimates indicate a substantively important percentage decrease in deforestation in priority municipalities over the 2008-2014 period as a result of the policy. When we translate the estimated relative treatment effect into absolute reductions in deforestation, our estimates suggest average post-policy reductions between 128.24 and 143.41 km² per municipality. At the basin scale this corresponds to avoided deforestation from 2008 to 2014 in the range of 4,617-5,163 km².

Our range of the percentage treatment effect is well in line with estimates between -55.8% and -44.7% in Assunção and Rocha (2014) for the period 2008-2011 (50 priority municipalities). Our estimates are, however, more pronounced than those found in Cisneros et al. (2015), who estimated a reduction between -29.7% and -27.6% for 2008-2012 (50 priority municipalities). Even so, their corresponding central estimate at the basin scale of 4,022 km² is close to our evaluation. Compared to Arima et al. (2014), who report absolute reductions per municipality in the interval [-247.75, -53.59], our absolute estimates are at the lower end. The differences in the magnitudes of the priority list's impact estimated in different studies could arise because of a number of methodological choices. First, studies differ in the deforestation data used – while we used PRODES data which includes estimates of how much deforestation occurs under cloud cover (but is not directly observed), Cisneros et al. (2015), for example, use raw deforestation estimates and correct for cloud cover in their regression analysis. Second, given our non-panel study design, we limit our analysis to the first municipalities placed on the list in 2008. Third, we have a longer post-policy time window than previous studies, spanning the period 2008 to 2014. Finally, we use a non-parametric estimator that either controls for bias due to insufficiently comparable treatment groups or builds on a synthetic control group.

Table 6-4 – Estimation results.

	Nearest neighbor matching estimates	Entropy balancing estimates
(1) Deforestation	-0.558**	-0.499***
	(0.223)	(0.143)
(2) Cattle productivity	0.360***	0.135***
	(0.115)	(0.044)
Cattle heads	0.249**	0.198***
	(0.102)	(0.053)
Pasture area	-0.0673	0.0689
	(0.069)	(0.058)
(3) Dairy productivity	-0.102	-0.0802
	(0.306)	(0.087)
(4) Crop productivity	0.464	0.148**
	(0.512)	(0.061)
Production value	0.122	-0.0659
	(0.709)	(0.118)
Cropland	-0.389	-0.213*
	(0.449)	(0.12)

Standard errors in parentheses.

* $p<0.10$, ** $p<0.05$, *** $p<0.01$

6.4.2. Effects on agricultural productivity

Cattle grazing productivity. We find no evidence that that priority list has displaced agricultural production, instead finding a significant positive effect of the priority list on the productivity of cattle ranching. The magnitude of our estimates, however, varies considerably (Figure 6.2 and Table 6-4). While the matching estimator suggests that priority municipalities have increased their cattle stocking rates by 36%, the estimated policy effect is 13.5% using entropy balancing. This range in the relative treatment effects corresponds to an economically meaningful absolute increase of 0.17 to 0.48 head/hectare.

A closer inspection reveals that the policy-induced productivity gain is driven by an expansion of cattle herds on the given pastureland, as witnessed by significant positive treatment effects on the number of cattle heads but statistically insignificant effects on the pasture area.

Dairy productivity. Next, we study the policy effect on dairy production. Row (3) of Table 6-4 presents results when the outcome variable is dairy productivity as measured by thousand litres of milk per milked cow. The treatment effect for this outcome levels out at -8% to -10%, but it is statistically insignificant. Put differently, we find no empirical evidence for policy-induced productivity changes.

Crop farming productivity. Finally, we investigate the productivity response in crop farming of priority list municipalities. Our main crop productivity measure is the gross crop production value per hectare of cropland of the six main crops (soy, maize, sugar, cotton, rice and cassava) in the legal Amazon. While we find no significant effect using the matching estimator, the entropy balancing estimator yields a statistically significant point estimate (at the 5% level) (Table 4). The measured effect suggests a 14.8% increase in the production value per hectare in municipalities on the priority list. The marginally significant negative policy impact on the denominator of our productivity measure (i.e. cropland) suggests that the increase in productivity could be associated with a reduction in the area of cultivated land.

We caution, however, against placing too much significance on these cropping results. First, we find inconsistent results between our two estimation methods. Second, when we break down the aggregate productivity measure and estimate matched treatment effects for each individual crop, none of the estimated effects is statistically significant (Table 9-20 in Appendix R).

6.4.3. Assessment of identification assumptions

Our main identification assumption that gives the above estimates a causal interpretation is that no unobserved variables exist that simultaneously influence changes in productivity and the probability of being inserted on the priority list. This is called the unconfoundedness assumption. In our specific DID framework it requires that in the absence of treatment the productivity of treated municipalities and matched control municipalities must follow the same trend. We subsequently investigate the plausibility of this assumption. While it is not testable, we can still conduct supporting analyses focusing on estimating "pseudo"-causal effects with a priori known values, under more restrictive assumptions.

Our approach to assess the tenability of the unconfoundedness assumption relies on the use of outcome data for (i) the year 2007, (ii) the years 2007 and 2006, or (iii) the last three pre-treatment years as alternative pseudo-outcomes. These pseudo-outcomes are known a priori not to be affected by the priority list policy precisely because their values are determined prior to the policy introduction. Finding a significant treatment effect in this setting would cast serious doubt on the credibility of inferences. Table 6-5 presents the results. In all three cases, the treatment effect on all pseudo-outcome variables is statistically insignificant. This evidence suggests that the identified treatment effects are not the results of pre-existing differences between priority and non-priority municipalities, i.e. the matched data at hand are supportive of the unconfoundedness assumption.

In addition to unconfoundedness, we must rule out the possibility of spillover effects from regulated to unregulated municipalities, e.g. through leakage or deterrence. This assumption is referred to as the stable unit treatment value assumption (SUTVA). Following Cisneros et al. (2015), this can be tested by using the non-priority list neighbours of priority municipalities as if treated. Indeed, 98 of the 440 non-priority municipalities in our sample share at least one border with a priority list municipality. Finding an insignificant treatment effect for these direct neighbours would make it more plausible that the no-interference assumption holds. The final row in Table 6-5 shows that the pseudo-treatment effects for all outcome variables are clearly statistically insignificant. Thus, it is unlikely that our treatment effect estimates are biased by the presence of spatial spillovers.

Table 6-5 – Assessment the plausibility of identification assumptions

	Deforestation	Cattle productivity	Dairy productivity	Crop productivity
Unconfoundedness				
Y_{2007}	0.0849	0.226	-0.0138	0.124
	(0.30)	(1.63)	(-0.12)	(0.68)
$\frac{Y_{2007}+Y_{2006}}{2}$	0.350	0.103	0.102	-0.197
	(1.14)	(0.91)	(0.82)	(-1.09)
$\frac{Y_{2007}+Y_{2006}+Y_{2005}}{3}$	0.0212	-0.0226	0.0668	-0.103
	(0.13)	(-0.17)	(0.96)	(-0.73)
Spatial spillover				
Non-listed neighbors	-0.170	0.00478	0.00521	-0.0106
	(-0.99)	(0.07)	(0.07)	(-0.12)

t statistics in parentheses

* p<0.10, ** p<0.05, *** p<0.01

6.4.4. What drove the observed changes in cattle productivity?

Before discussing the potential mechanisms that drive the observed productivity gains in the beef sector, we use a simple back-of-the-envelope calculation based on our two estimated cattle productivity responses (in Table 6-4) to translate the treatment effect estimates into tangible land sparing effects. The measured beef productivity gains are equivalent to 0.17-0.48 greater head/hectare in priority municipalities, which would have required an additional 14,542-41,060 km^2 of pasture at 2008 stocking rates. These figures are substantially larger than our estimate of the avoided deforestation, 4,617-5,163 km^2. Though we cannot with our dataset definitively unpick the role of different mechanisms in driving the observed increase of cattle productivity, the magnitude of both the estimated stocking rate and land sparing effect suggest that a mix of factors could have contributed to this policy outcome.

As discussed before, the priority list has the effect of making illegal deforestation less attractive through two channels: (i) the policy increases the costs of clearing land, and (ii) it reduces the benefits from clearing land. This reduced attractiveness of expanding land use can be expected to reduce deforestation and cause a substitution from land to capital, as discussed in our conceptual framework. These effects are in line with other empirical evidence from Brazil.

Substituting capital for land has, for example, been intensively discussed in the Brazilian context. Using municipal data from agricultural censuses from 1960-2006, Barretto et al. (2013) find that intensification in both cattle and crop production occurred more in consolidated regions, where land was more scarce, than in frontier areas. Similarly, using a micro-level analysis of farm production in Rondônia state, Fontes and Palmer (2017) find that cattle stocking rates were higher and deforestation rates lower in farms that were closer to market, where opportunity costs were higher and less forest was available for expansion. These findings lend support to recent calls for prioritizing forest conservation in efforts to promote sustainable land use in the Amazon (Merry and Soares-Filho, 2017). In the broader context of the literature on agricultural intensification, the findings are supported by Villoria et al. (2014) who similarly show that intensification (i.e. factor substitution of capital for land) only occurs where land is a scarce production factor and land opportunity costs are sufficiently high.

Of course, scarcity-induced substitution requires that yield-raising technologies are available and affordable. Cattle ranching in the Amazon remains, for the most part, a low-input production system and there are many opportunities for increasing productivity through improved farm and pasture management, or the introduction of cattle feedlots (Barbosa et al., 2015; GTPS, 2016). Soy production in the Amazon, is however, a technologically mature industry (more than 70% of Brazilian soybean production is genetically-modified (Garrett et al., 2013b)), where yields are already comparatively high – which may make changes in yields in the soy sector less responsive to new efforts to constrain expansion.

There is also evidence in the literature for the relevance of the second component of the combined effect, the policy-induced changes in incentives for speculative land clearing. A range of studies support the argument that cattle ranching in the Brazilian Amazon is used as a land speculation strategy (Hecht, 1993; Richards et al., 2014; Walker et al., 2009). Similarly, land speculation has been suggested as an explanation to the puzzling low rates of capitalization of the cattle sector (Bowman et al., 2012; Hecht, 1993). There is also evidence that insecure land tenure encourages clearing more land than is strictly required for agricultural production (Brown et al., 2016). In this respect, the increased regularization of land in priority municipalities, through increased enforcement and the adoption of the CAR – one of the requirements to be removed from the Priority List – may have further reduced incentives to clear new land. Farmers interviewed in Pará reported, for example, that they have reduced their use of land as a result of registering their properties in the CAR (Jung et al., 2017), and Alix-Garcia et al. (2017) find that properties registered in the CAR in Mato Grosso and Pará saw a 62.5% reduction in deforestation. Similarly, (Azevedo et al., 2017) find that the CAR has reduced deforestation, though the effect varied between states, property-sizes, and years.

While the mechanisms highlighted by our model have strong support in the related literature, we cannot with our dataset definitively rule out a number of alternative channels that could also play some role in explaining the observed productivity effect.

First, the support provided by municipal governments and civil society organizations may have helped farmers to increase their productivity. Being included in the priority list led to the "crowding in" of international and NGO support for the transition to sustainable land management (Cisneros et al., 2015). Priority list municipalities signed agreements with national and international donors (including the European Commission, the Fundo Vale, and the Amazon fund), and NGOs (including The Nature Conservancy, Instituto Centro da Vida, and Imazon) (MMA, 2015; Sassaki, 2014; Sills et al., 2015). While in some cases these agreements included commitments to foster sustainable agricultural practices, in practice these efforts were small-scale (Chapter 5; Piketty et al., 2015), and municipalities had a far stronger emphasis on measures directly related to the conditions for being removed from the priority list – reducing deforestation and supporting the municipalities in regularizing land use (Thaler, 2017).

A second factor that may have played a role are reputational damages caused by civil society to "blacklisted" municipalities and farmers (Cisneros et al., 2015). Such reputational risk may have added additional costs to those directly imposed by the policy (higher risk of fines and seizures). Finally, changes in the availability of agricultural credit have also received attention in the literature. While credit restrictions in frontier regions are thought to reduce deforestation rates (Assunção et al., 2013), and the priority list was meant to involve tighter restrictions on credit for properties with illegal deforestation, when measured at municipal-level at least, there appears to have been little effect on credit availability (Assunção and Rocha, 2014; Cisneros et al., 2015)(Assunção and Rocha, 2014; Cisneros et al., 2015). The relevance of credit restrictions therefore seems limited.

Future research to untangle the mechanisms through which forest conservation policies impact agricultural productivity could try to directly measure input substitution, by for example monitoring fertilizer sales or the adoption of different livestock production systems, though these data are currently not available at the municipal level in Brazil. First studies in that direction indeed highlight recent increases in feedlot systems (Macedo et al., 2012). Future work would ideally ask similar questions using farm-level datasets, potentially looking at the responses of landholdings of different sizes (e.g. are effects different for small and large producers; see (Assunção et al., 2017b; Godar et al., 2014)) and with different land tenure regimes (e.g. properties with/without formal title or CAR/no CAR) on agricultural productivity. A recent systematic review found only three studies that have evaluated links between land tenure security and agricultural productivity in Latin America (Lawry et al., 2014); though all three found positive effects, recent research on soy production in Brazil found no effect of land tenure on soybean yields (Garrett et al., 2013a).

6.4.5. Study Limitations

To ensure the robustness of our results, we report two different empirical strategies to develop our counter-factuals for deforestation, agricultural production, and productivity in municipalities assigned to the priority list. Our methods and data inevitably have a number of limitations, however. First, we measure cattle productivity through two indirect measures, the stocking rate (head/hectare) and milk production per cow per year. While beef and milk production per hectare per year would be more direct measures of productivity, the results are unlikely to differ from our indirect measures. In Brazil's low-input beef and dairy systems, improvements in stocking rates and productivity per animal usually go hand-in-hand with increases in production per hectare (Chapter 5; Novo, 2012). It is also possible that the measured increase in stocking rates may not be sustainable – inappropriate increases in cattle stocking rates can lead to pasture degradation in the long term. Though longer-term studies will be required to definitively rule this out, we are again hopeful that this is not the case, as stocking rates in the Amazon are currently well below their sustainable potential (Strassburg et al., 2014).

There are also uncertainties in the underlying data. Pasture is notoriously difficult to classify from satellite imagery, though we use the highest resolution data currently available for the Amazon region (Almeida et al., 2016). For our crop data, we rely on annual farm survey data, which, because it does not sample all farms in the municipality each year, also has some associated uncertainty. There are, for example, discrepancies between the area of cropland measured from survey data and satellite imaging (Figure 9.25 in Appendix R). We use the survey data because it distinguishes between different crops (Terraclass satellite data group all agriculture into "annual agriculture") allowing calculation of crop production ($/ha) and yield (kg/ha). Reassuringly, our results match those from (le Polain de Waroux et al., 2017), who used pasture and cropland area datasets based on MODIS satellite data (Graesser et al., 2015), and also found that cattle, but not crop, intensification had occurred in the Amazon region.

6.5. Conclusion

In this chapter, we find no evidence of a trade-off between agricultural production and one of Brazil's flagship forest conservation policies, the municipal priority list. These results corroborate the importance of strong environmental governance in guiding intensification without deforestation (Ceddia et al., 2014). While much work on agriculture-environment trade-offs has focused on how yield increases might spare forests from conversion (e.g. Burney et al., 2010; Cohn et al., 2014; Stevenson et al., 2013; Tilman et al., 2011), our results suggest that the causality can run both ways. At least in areas with large yield gaps and where high-yielding technologies are readily available (as in Brazilian cattle ranching), policies that induce land scarcity may spur intensification (Kaimowitz and Angelsen, 2008; Merry and Soares-Filho, 2017). We caution, however, that efforts to pair forest conservation and the implementation of high-yielding farming practices are likely to be more successful than any one intervention alone (Phalan et al., 2016).

The priority list was a hybrid governance initiative working at sub-national scale to control deforestation while maintaining agricultural production (Viana et al., 2016). As such, it represents an interesting test of new models of supply chain management, so called "jurisdictional approaches" (Nepstad, 2017), which aim to improve land management at a sub-national scale by aligning incentives for sustainable commodity production for multiple stakeholders, including local government, producers, NGOs, companies, and financial institutions. In the priority list, deforestation was enforced mostly by the Federal Government, land use was regularized and good agricultural practices promoted by local government and NGO efforts, together with (in some cases at least) commitments from agricultural interest groups and civil society. We add to a young literature (e.g. Nolte et al., 2017) suggesting that sub-national efforts can be effective in reconciling deforestation and agriculture.

"Meat is an extravagance. However, to conclude that veganism is the 'only ethical response' is to take a big leap into a very muddy pond… Livestock farming is a subject, I have discovered, where every answer uncovers two questions, and every statistic cloaks an ideological assumption."

- *From "Meat: a benign extravagance" by Simon Fairlie, Permanent Publications (2010).*

7. Discussion

In this thesis, I have presented analyses of the potential to (i) increase the use of food losses as feed in Europe and (ii) increase the productivity of cattle ranching in Brazil. Below, I summarise my findings and reflect on where they fit in the wider literature on the sustainability of livestock.

7.1. Food losses as feed

While the use of uncooked food losses in animal feed can pose a disease risk, the EU-wide bans on the use of catering wastes and animal by-products were not the only possible policy response. Nations such as Japan and South Korea operate large-scale regulated systems for safely recycling food losses as feed. I have shown that introducing similar systems in Europe could reduce demand for agricultural land by 1.8Mha (1.7–2.0 Mha; 95% CI), and free up enough cereals to meet the annual consumption of ~70 million EU citizens. Swill reduces feed costs for farmers, without necessarily reducing product quality – I found little effect of swill feeding on 18 different measures of meat quality. Swill-feeding similarly reduces the environmental burden of food waste disposal, compared with composting or anaerobic digestion. I have also presented results from a survey of a UK agricultural trade fair, which revealed widespread support for the relegalization of swill, and highlighted ongoing concerns about disease risks. These findings deepen our understanding of the environmental impact of food waste disposal, and the role of omnivorous livestock as nutrient recyclers, in three main ways.

Firstly, my results contribute to a young literature evaluating the impacts of different technologies for food waste disposal (Figure 3.3). My results confirm the order of the food waste hierarchy (Figure 2.5) – i.e. that it is preferable to use food losses as animal feed, rather than for biogas or compost production (Papargyropoulou et al., 2014). In line with the food waste hierarchy, it is also important that increases in the use of swill should not come at the expense of reductions in food waste. Again, lessons can be learned from East Asia, as South Korea successfully reduced household and restaurant food waste by 30-40%, even while increasing their recycling rate to 85% (E. K. H. J. zu Ermgassen et al., 2016).

Secondly, the survey results are relevant to the debate in Europe around the inclusion of animal by-products in animal feed. While animal by-products are all-but-banned in monogastric and ruminant feed, the European Commission recently permitted the use of insect meal in fish feed (EC, 2017b), and there is pressure from the feed industry to also permit the use of insects and processed animal proteins for monogastrics (Searby, 2014). (Processed animal proteins are animal by-products, such as tendons and trotters from monogastrics, which are fit for human consumption but rarely eaten by people, because of cultural or aesthetic reasons). This move would recognize that monogastrics are omnivores, though it would not lift the ban on intra-species recycling. In chapter 4, however, I showed that while farmers have a preference for using vegetarian pig feeds, and feeds which avoid intra-species recycling, that does not mean that farmers are against the inclusion of animal by-products or intra-species recycling *per se*. Three-quarters of farmers would support the relegalization of swill, and more than half said they would consider using swill on their farms. Similarly, previous work has shown that the legalisation of the inclusion of insects in monogastric feed would also be well-received by the farming industry (Verbeke et al., 2015).

Thirdly, my results provide a real-world policy example of "livestock on leftovers" approaches to livestock sustainability (Garnett et al., 2017). While 32-36% of global grain production is currently fed to livestock (Cassidy et al., 2013; Mottet et al., 2017), for many groups, the use of human-edible food as livestock feed is anathema. A growing research effort has therefore tried to evaluate what a livestock system without grain would look like, and how much food it would produce. Variously called a "livestock on leftovers", "default livestock", "ecological leftovers", or "consistency strategy" (Fairlie, 2010; Garnett et al., 2017; Röös et al., 2017; Schader et al., 2015; H. H. E. van Zanten et al., 2015), these approaches propose limiting livestock production to their traditional role as nutrient recyclers, converting resources that humans do not eat – food losses, agricultural by-products, and forages – into produce that we can, namely meat and milk. This work remains, for the most part, entirely theoretical, and it is not clear how to deliver a "livestock on leftovers" system in practice. The relegalization of swill, however, is one example of a policy which could help livestock on the path toward being net contributors to the human food supply.

7.1.1. Prospects for legal change

So, what are the prospects for the large-scale use of swill in Europe? While I report an encouraging level of support amongst UK pig farmers for the relegalization of swill, public support in other countries remains unquantified, and even in the UK many important stakeholders remain to be convinced. The UK National Pig Association, for example, supports the ban, arguing that (i) the cost of regulating swill would not make it a competitive option, (ii) food waste would produce meat of poor quality, and (iii) the risk of a disease outbreak is too high (Wilson, 2016).

While the meat quality issue is addressable (chapter 2), more work is warranted on the economic costs of implementing swill feeding on a large scale. In chapter 2, I showed that swill feeding can reduce costs for farmers, even when taking into account negative effects on growth rates. This analysis was, however, simplistic. The full costs of implementing swill feeding will include investing in improved food waste disposal – a task to which the UK has, in any case, committed itself under the EU waste directive (EC, 2015) – and re-establishing the infrastructure for a swill feeding industry. Japan and South Korea provided some initial government support to kick-start the Ecofeed industry (Appendix B), and it is noteworthy that the UK government provided similar support for the nascent anaerobic digestion industry, with £10 million in loans (DEFRA and DECC, 2014).

A responsible effort to promote the use of food losses in feed must also address persistent concerns about disease control. A recently released risk assessment from the

UK government provides an important step in that direction (Adkin et al., 2014), though its results are highly uncertain, and follow up work should be done to address some of these uncertainties.

Adkin et al., (2014) built a stochastic model of the probability of an animal getting infected in the UK with any of 16 different pathogens, including bacteria, prions and viruses, under two scenarios: (a) the relegalisation of the use of heat-treated catering waste in feed; and (b) relegalization of the use of (heat-treated) commercial catering wastes (i.e. from restaurants), but with the intra-species recycling ban still enforced, and household food wastes not permitted. While scenario (b) showed even lower disease risks than scenario (a), I focus on the first scenario in the discussion below, because the intra-species recycling ban would likely hinder large volumes of food waste being included in feed.

For many pathogens the risk of an outbreak was negligible (Table 7-1), though for three diseases, African swine fever, Highly Pathogenic Avian Influenza (HP-AI), and Newcastle disease, the model suggested a medium risk of infection (i.e. one outbreak every 2-6 years), and for foot-and-mouth disease an outbreak was predicted every 74 years. HP-AI and Newcastle disease are poultry diseases (the model assumes that 7% of swill, i.e. 0.35% of UK food losses, might be used for poultry feed), and the model appears to substantially overestimate the risk of disease introduction from swill feeding. From 1938-2001 (i.e. before the ban on swill), the UK saw only four outbreaks of HP-AI (Peiso et al., 2011), and Newcastle disease has been rare since vaccination was introduced in the 1970s, with only three outbreaks from 1978-2001 (Peiso et al., 2011). In both cases, these diseases are much more commonly linked to spread from wild birds, live poultry markets, or farm workers travelling from regions where the disease is endemic, than from spread via swill (Alexander et al., 1985; Gao, 2014; The Global Consortium for H5N8 and Related Influenza Viruses, 2016).

Table 7-1 – Modelled mean risk of infection per year for 16 diseases, if 5% of food wastes in the UK were fed as swill. Data are from a scenario where food waste is heat-treated at 100°C for 1 hour; data for heat-treatment at 70°C at 30 minutes are also included in the report, though their scenarios do not include the effect of fermentation (i.e. low pH) on pathogen inactivation, and so are not discussed further here. Adapted from: (Adkin et al., 2014).

Disease risk	Estimated frequency of disease outbreaks	Pathogens
Negligible	Once every > 1000 years	Bacillus anthracis, Brucella, Salmonella gallinarum, chronic wasting disease, highly pathogenic porcine reproductive and respiratory syndrome, Aujeszky's disease, Enzootic bovine leucosis, sheep pox and goat pox, Swine Vesicular Disease.
Very low	Once every 700 years	Highly pathogenic porcine epidemic diarrhoea
Low	Once every 70-74 years	Foot and mouth, classical swine fever, infectious pancreatic necrosis
Medium	Once every 2-6 years	African swine fever, HP-AI, Newcastle Disease

The model's estimates are also highly uncertain – as demonstrated by the figures for African Swine Fever. While the mean risk was to have an outbreak of African Swine Fever every two years, the 90% confidence intervals ranged from the risk being "negligible" to "very high" – i.e. from an outbreak every >1000 years, to one every year. The authors identified uncertainties in (i) the mass of infected products that are imported into the UK. This was assumed to be the same as estimates for classical swine fever in illegally imported meat products (Hartnett et al., 2004), which is an overestimate for most pathogens – notably so for African Swine Fever, where it is 5,700x times higher than the most recently published estimate, 263kg/yr vs 0.046kg/yr (Hartnett et al., 2004), though these figures are out-of-date, given the recent expansion of African Swine fever to Eastern Europe (Sánchez-Vizcaíno et al., 2013); (ii) the infectivity titre in meat products (ID50/kg); (iii) the oral dose (Oral ID50/ID50), the probability of pathogen survival post-handling in food, and – most importantly – (iv) the assumed rate of cross-contamination of swill with uncooked food wastes. Since heat treatment is sufficient to inactivate pathogens such as African swine fever and foot-and-mouth disease (Adkin et al., 2014; OIE, 2009; Scudamore, 2002), the risk of these diseases comes because of the assumed rate at which non-cooked food wastes enter animal feed. i.e. if the risk of cross-contamination can be reduced to a negligible level, then swill feeding can be practised safely. Adkin's et al.'s assumptions about the rate of cross-contamination are therefore worth considering in more detail.

In the absence of data from swill operators, the authors based their estimates of the risk of cross-contamination on a similar challenge presented by the EU's TSE legislation. Specifically, fishmeal is banned in ruminant feed, though it is permitted for monogastrics. The authors used data from the UK's National Feed Audit (the sampling program used to support controls in the EU TSE regulations), finding that out of 54,272 inspections of feed mills between 2005-2012, there were 25 where fishmeal was found in ruminant feed (i.e. a probability of 4.6×10^{-4} per sample), and of 1,352 farm visits, there were 4 breaches of the legislation (3.2×10^{-3} breaches per inspection). They therefore assumed that raw food waste would contaminate swill at the mean rate of these two segregation failures (5.36×10^{-4} per sample/inspection). Given the lack of data on the number of breaches in the systems operating in South Korea and Japan, these assumptions are sensible – and they highlight that cross-contamination would have to be reduced to very low levels to minimize disease risks.

Can the risk of cross-contamination be lowered to an acceptable level? There are a few reasons to be optimistic; first, that in Japan and South Korea, to the best of my knowledge, no disease outbreaks have been linked to swill feeding (Muroga et al., 2012;

Park et al., 2013; Yonhap News Agency, 2017) – though more knowledge is required about their operating procedures and the rate of segregation failures. Second, we can learn from other disciplines when designing best-practices for minimizing cross-contamination. This is, for example, a problem routinely dealt with in the biohazardous waste sector. Third, simple solutions may help reduce the risks dramatically. The figures from the National Feed Audit suggest, for example, that segregation failures are more common at the farm-level than at the factory; under the previous swill legislation, swill manufacturing plants could, however, be located on the same site where livestock were reared (though they had to be technically managed as separate premises). Safety gains may be made by spatially separating swill production and livestock rearing.

Finally, when evaluating the risk associated with the relegalization of heat-treated swill, we also need an estimate of the background risk of an outbreak under the current legislation, in order to provide a fair comparison. There may be a false sense of security about the ban on the use of catering waste – a "ban" which is routinely broken by smallholder farmers, who do not heat-treat their kitchen wastes (Gillespie et al., 2015). It is therefore not yet certain whether the risks of a disease outbreak would be lower under the current ban, or whether the introduction of a large-scale, legal system for collecting and processing swill would reduce the feeding of uncooked food waste, while delivering a host of other benefits to farmers and the environment.

7.2. Beef in Brazil

Brazil has vast areas of pasture (170 Mha; compared with 60 million hectares of cropland; Strassburg et al., 2014), which are – for the most part – used for extensive cattle production. Given government and industry plans for large-scale forest restoration and crop expansion, several parties have made calls for the intensification of cattle ranching to make space for other expanding land uses and help Brazil deliver on its environmental and agricultural goals (e.g. Harfuch et al., 2016; Latawiec et al., 2015; le Polain de Waroux et al., 2017; Nepstad et al., 2014; Strassburg et al., 2014). In this thesis, I have presented two analyses of how higher-yielding, sustainable cattle ranching can be achieved in the Amazon – a region with persistently low stocking rates, ongoing deforestation, and home to 29% of the Brazilian cattle herd.

First, I summarized the results of six on-the-ground initiatives which are supporting farmers to adopt new technologies, and help them meet the requirements of the Brazilian Forest Code. Though these initiatives are spread across four states and use a wide range of technologies, they share many similarities; they all have a strong focus on farmer training, farm record-keeping, and improved pasture management – in particular, the adoption of rotational grazing and pasture fertilization using chemical inputs or leguminous plants. These management changes require initial investment (R$1300-6900/hectare), which is paid off within an average of 2.5-8.5 years.

Second, I analysed how agricultural production fared under the anti-deforestation policy, the Municípios Prioritários. I found that the priority list reduced deforestation by 50-56% from 2008-2014, and that cattle producers responded by intensifying production: increasing the number of cattle by 20-25% and stocking rates by 13.5-36%. I observed no change in the dairy and cropping sectors. Together, these results provide real-world evidence that it is possible to increase beef production in Brazil without increasing deforestation.

Having analysed two contrasting approaches for increasing the productivity of cattle ranching, it is reasonable to ask which of these approaches is likely to be most effective – the carrot (technological support), or the stick (increased enforcement)? Though economic modelling suggests that the adoption of GAP could halve greenhouse gas emissions from deforestation and agriculture in Brazil (Cohn et al., 2014), it is a risky strategy to rely on intensification alone to deliver sustainable land use practices. A narrow focus on the intensification of cattle ranching is unlikely to deliver environmental gains, at least locally, because of the threat of rebound increases in deforestation, driven by increases in profitability (Byerlee et al., 2014; Ewers et al., 2009; Merry and Soares-Filho, 2017; Phalan et al., 2016). It is for this reason that the organisations leading the six cattle intensification initiatives described in chapter 5 supported both farm improvements and compliance with the Forest Code. Similarly, though local forest protection is undoubtedly important – not least because a large portion of the greenhouse gas mitigation potential in tropical agriculture is from reduced deforestation – it is also no guarantor of knock-on improvements in farming. I found, for example, no increase in the productivity of dairy or crop production, and while I argue that this is in large part because these sectors were not targeted for increased enforcement, more research is needed to understand under what conditions forest conservation initiatives are undermined by leakage (i.e. displaced agricultural production). The Soy Moratorium, for example, is notorious for having displaced soy production to the Cerrado (Gibbs et al., 2015; Harris et al., 2017; Noojipady et al., 2017), and cattle farming is also not immune to leakage: Jadin et al. (2016) found that while the forest transition in Costa Rica was associated with the intensification of cattle production, it also coincided with a reduction in the country's beef exports.

Given the context-specific feedbacks between agricultural intensification and forest protection, efforts to reconcile agriculture and forests should look to pair efforts to support farmers with efforts to simultaneously protect forests (Phalan et al., 2016). Though examples of these joined-up initiatives remain rare, it is promising that a handful of sub-national approaches for sustainable land management are under development (Nepstad, 2017). The Brazilian state of Mato Grosso, for example, launched their "Produce, Conserve, Include" initiative at COP-21. This multi-stakeholder initiative sets specific targets for 2030 to reduce deforestation and increase reforestation, to increase production of agriculture and livestock on already cleared lands, and incorporate smallholders and indigenous people in low-emission rural development (Miller et al., 2017).

7.2.1. Limits to intensification

My analyses have focused more on how the productivity of cattle ranching in the Brazilian Amazon could be increased, rather than on the environmental case for intensification, a topic which I return to now. Intensification can reduce greenhouse gas emissions by shortening the production cycle, and so reduce emissions from manure and enteric fermentation across the animal's lifetime (Figure 5.6). Moreover, because the vast majority (~90%) of cattle emissions in the Amazon region stem from deforestation (Bustamante et al., 2012), by reducing the area required for cattle ranching, it is hoped that intensification will reduce deforestation through land sparing. Environmental impacts, of course, extend beyond greenhouse gases. Land sparing reductions in deforestation would also deliver biodiversity benefits, since most species in the region are forest-adapted, and higher-yielding cattle systems tend to cause less eutrophication and acidification than more extensive systems (Röös et al., 2013).

Intensification can, of course, go too far. While feedlot systems are highly efficient from a greenhouse gas perspective, they generate large volumes of manure that can pose disposal problems and leach into the environment if not properly handled. There are also welfare concerns about their proliferation (Grandin, 2016). That intensification does not necessarily generate win-wins across all metrics is easily seen at the global level, where livestock intensification from 1961-2010 caused reductions in greenhouse gas emissions (-46%) and land use (-62%) per animal calorie, but large increases in nitrogen emissions (+188%) (Davis et al., 2015). The intensification described in this thesis, however, is still at the low end of the productivity spectrum. The vast majority of cattle in Brazil are reared in extensive pasture-based systems (Strassburg et al., 2014); intensification within pasture-based systems can attain significant (>50%) greenhouse gas emission reductions (Gerssen-Gondelach et al., 2017), and is unlikely to meet many environmental trade-offs – especially if growth in the sector includes efforts to reduce leakage in cattle supply chains, and protect forest fragments and water bodies from cattle intrusion, as described in chapter 5 and Appendix Q.

7.3. Concluding remarks

Despite recent interest in plant-based diets (e.g. Harwatt et al., 2017), for the foreseeable future, livestock will likely continue to play a key role in global agriculture.

The livestock sector employs millions of people, produces food that many people want to eat, and can contribute positively to food security. Livestock can convert resources that humans don't eat, including food losses and low-quality grazing land, into products that we can. Traditionally, livestock have also be used to even-out fluctuations in grain production – mopping up excess grain when harvests are bountiful (Fairlie, 2010).

It is undeniable, however, that the livestock sector, as it operates today has a huge environmental impact. I have presented studies of two opportunities to reduce that impact. While these results are promising, it is worth reflecting that these initiatives, namely promoting the use of low-impact food losses as feed and increasing the efficiency of extensive cattle production, only deliver partial improvements in the sustainability of the livestock sector; reductions in the consumption of animal source foods in high-consuming regions are also required.

Even if "livestock on leftovers" approaches were applied in a consistent manner at a global scale, they would not produce enough to meet the intakes of animal source foods currently found in the developed world. I estimated that East Asian-style recycling of food losses as feed could replace 20.3% of EU pig feed (Appendix F); to eliminate grain use from pork production would therefore require reducing EU pork production (or finding new sources of human-inedible by-products to include in diets). While livestock on leftovers might produce between 11-32g of animal protein/capita/day (with large uncertainty around these estimates; Garnett et al., 2017), current consumption in OECD nations is 60g/capita/day (the global average is 26g/capita/day). The gap between these estimates suggests that large reductions, or at least wide-scale redistribution, of the consumption of livestock products would be required for the "livestock on leftovers" vision of livestock sustainability to be feasible.

Similarly, while intensification of livestock production can reduce greenhouse gas emissions per unit product, since it often goes hand-in-hand with local increases in production (e.g. Bogaerts et al., 2017), intensification will only reduce farm-level emissions under specific conditions. This occurs where intensification leads to reduced deforestation, where improved management fosters long-term soil carbon sequestration (e.g. de Oliveira Silva et al., 2017; Doran-Browne et al., 2017), or where total production is constrained. For many production systems, however, increases in production mean that the net emissions of livestock continue to increase, rather than fall (Dangal et al., 2017). This thesis therefore concludes like many others (Bajželj et al., 2014; Bryngelsson et al., 2016; Erb et al., 2016; Garnett et al., 2017; Lamb et al., 2016; Röös et al., 2017), that while there are opportunities to reduce the impacts of livestock, a sustainable livestock sector must not only improve production practices, but also shrink in size.

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9. Appendices

APPENDIX A – CHAPTER 2, EU FOOD WASTE LEGISLATION

EC regulation 1774/2002 regulation was introduced across the EU in 2002, after an initial ban on swill in the UK in 2001. It bans the use of catering wastes, whether from household, restaurant, or central kitchens for use as animal feed, effectively ending the 9,000 year-old practice of recycling food wastes as animal feed.

The continued use of food wastes is permitted only where it can be demonstrated that there is no risk of contamination with meat, fish, or other animal products. This requires either that a facility handle no animal products or they establish completely separate handling streams for animal and non-animal products, along with Hazard Analysis and Critical Control Point (HACCP) procedures. This prevents all but the largest businesses from recycling food waste as feed.

The proportion of food waste that is recycled as animal feed is therefore small. Of the 102.5 million tonnes of food waste produced in the EU per year (EC, 2010), it is estimated by the European Former Foodstuff Processors Association (EFFPA, the body which represents European processors of food wastes) only 3 million tonnes of manufacturing food wastes are currently recycled as animal feed, and that there are a further 2 million tonnes which could be legally recycled, under existing legislation (EFFPA, 2014). Food wastes recycled as animal feed are known in the processing industry as "former foodstuffs".

The current method of disposal of most EU food wastes is not well known, because of limited data collection. The EU waste directive sets a target for 2025 that no biodegradable waste (including food wastes) be landfilled by 2025 (EC, 2014). Progress is slow, however, and large amounts of food waste are still disposed of in landfill. In parts of the UK, for example, food waste makes up to 48% of landfilled waste (House of Lords, 2014). Improved food waste recycling requires separate food waste collection, as occurred in 95% of Wales, 34% of Scotland, 26% of England, and 4% of Northern Ireland in 2013 (House of Lords, 2014).

Appendix B – Chapter 2, Japanese & South Korean food waste legislation

Japanese food waste legislation:

In 2001 Japan introduced the Promotion of Utilization of Recyclable Food Waste Act (Food Waste Recycling Law) which has seen large increases in food waste recycling, including the recycling of food wastes into animal feed (Table 9-1). Animal feed from recycled food waste is known as "Ecofeed".

The Food Waste Recycling Law regulates the collection, transport and storage of food wastes and Ecofeed products. In 2007 the law was amended to make animal feed the priority use of food wastes, in preference to composting or incineration, and to create "recycling loops" by requiring companies which produce food waste to preferentially purchase Ecofeed-reared pork (Takata et al., 2012). In 2006, Japan successfully recycled 52.5% of its manufacturing, retail, and catering food waste as animal feed (MAFF, 2011) – the remaining portion being composted, incinerated, or landfilled on the grounds of being inedible, like orange peels or rotten food, or being produced in locations without the necessary recycling infrastructure. Recycling rates differ between industries: less food is currently recycled from catering outlets, which are diffusely distributed and individually have small waste streams (16% of food waste recycled in 2009, up from 9% in 2001) than from food manufacturing plants, which are more concentrated and produce larger waste streams (93%, up from 50% in 2001).

Table 9-1 - Food waste recycling in Japan, from 2001 to 2009. (Reported as the percentage of food waste recycled for all purposes, including the production of Ecofeed, compost, and anaerobic digestion). Retail figures are a mean of wholesaler and retailer food waste recycling rates. Household food waste is not recycled in Japan, but is in South Korea (Stuart, 2009). Modified from (MAFF, 2012b; Takata et al., 2012).

Food waste source	2001	2002	2003	2004	2005	2006	2007	2008	2009
Manufacturing (%)	50	60	65	65	76	76	77	93	93
Retail (%)	23.5	26.5	30	29	42	44	45	48	47
Catering and food service (%)	9	8	11	12	14	16	16	13	16

Ecofeed manufacturers (see http://ecofeed.lin.gr.jp/map.cgi) operate under Japanese food safety law which requires that food waste containing meats must be heated for a minimum of 30 minutes at 70°C or 3 minutes at 80°C (Sugiura et al., 2009). Household wastes (31.6% of all food waste) are not currently recycled into animal feed in Japan because they are vulnerable to contamination by foreign objects (e.g. cutlery (Sugiura et al., 2009)), although household wastes are recycled in South Korea (Stuart, 2009), where food waste is screened for potential contaminants before use. The use of meat wastes in ruminant (cattle, goat and sheep) diets is banned because of concerns about Bovine Spongiform Encephalopathy (BSE), a disease that does not affect pigs or poultry (Andreoletti et al., 2007).

Since its introduction, the Ecofeed market has grown year-on-year (Figure 9.1), and food wastes made up 5.8% of all concentrate animal feed (for pigs, poultry, and ruminants) in 2013. To promote Ecofeed further, the government has provided financial support and introduced Ecofeed certification. Ecofeed receives support under the ¥23 billion ($194m) "Grant to Create a Strong Agricultural Industry" and the ¥89 million ($750,000) "Urgent Plan to Increase Ecofeed Production" (MAFF, 2014). Certification was introduced in March 2009. To be certified, animal feeds must contain more than 20% food waste (with at least 5% of the entire feed made up by "promoted food wastes", which include noodle debris, plate scraps, waste oil, and coffee grounds; see Table 9-2). Forty-nine feeds were Ecofeed certified as of September 2013. Similarly, certification of products from livestock reared on Ecofeed was introduced in 2011, with 8 brands certified by September 2013).

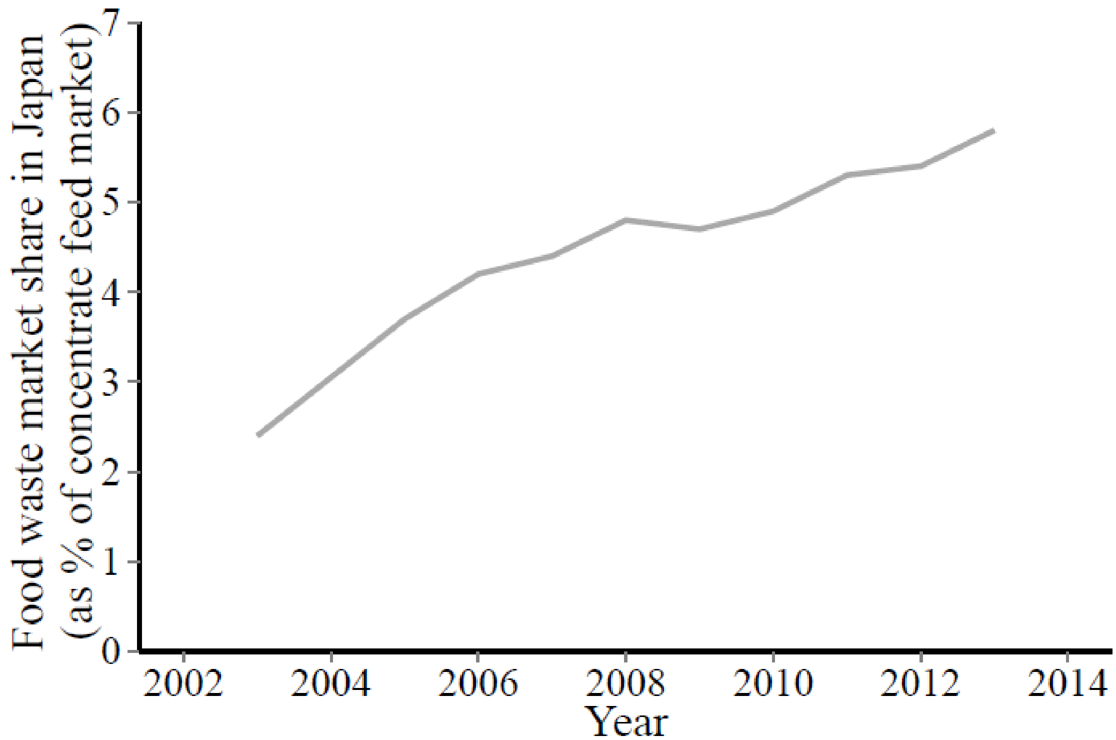

Figure 9.1 – Food waste makes up an increasing proportion of animal feed in Japan. Source:(MAFF, 2014).

Table 9-2 – Classification of food wastes under the Ecofeed certification scheme. To be certified, at least 5% of the entire feed must be "promoted" food wastes, with at least 20% of the total feed made up by a combination of both types of food waste. Adapted from Sugiura et al. (2009).

Type of food waste	Example
Ordinary	Distillery waste, beet pulp, rice bran, wheat bran, soybean dregs (excluding imported soybean dregs)

Promoted	Plate scraps, noodle debris, bread crumbs, cake crumbs, gluten debris, bean curd, mushroom-growing bed waste, sake lees, rice vinegar lees, tea dregs, squeezed fruit waste, coffee waste, cacao grounds, dairy plant wastes, frozen food plant wastes, cooking waste, waste oil, waste boxed lunches

South Korean food waste regulation:

The recycling of food waste in South Korea is regulated under both the Wastes Control Act (Ministry of Environment, 2010b) and the by the Control of Livestock and Fish Feed Act (Ministry of Agriculture, Food, and Rural Affairs, 2010). In 2006, 42.5% of all food waste was recycled as animal feed (the most recent data available; Kim and Kim, 2010).

Under the Control of Livestock and Fish Feed Act Article 8, food waste can only be included in animal feed if it has been treated at registered feed production facilities – of which there were 259 facilities in 2010 (Ministry of Environment, 2010a). Facilities which produce wet feed from food waste are often located on-farm to minimise transport costs, while facilities which produce dry feed are often near urban centres and can be operated by either local government or private firms (see Figure A2 for information on the possible management structures; Ministry of Environment, 2012).

The process of swill production is standardised under Article 11 of the Control of Livestock and Fish Feed Act. In all cases, food waste must be heat treated for 30 minutes to a core temperature of at least 80°C in order to be included in animal feed; the exact process differs between dry and wet feed. For the production of dry feed, food waste is typically dehydrated by mixing with air heated to 390°C. This method sterilises the feed, increases the feed shelf life, and avoids destroying nutrients (National Institute of Environmental Research, 2012). Wet feed production typically involves two steps. First, the feed is sterilized by heating to more than 80°C. Second, the moisture content of the feed is standardised to 70-80% by mixing with corn or rice husks. Both these production processes must also meet the conditions of the Article 14 of the Control of Livestock and Fish Feed Act, which sets limits on the acceptable standard of hazardous materials in animal feed, such as heavy metals and fungal toxins.

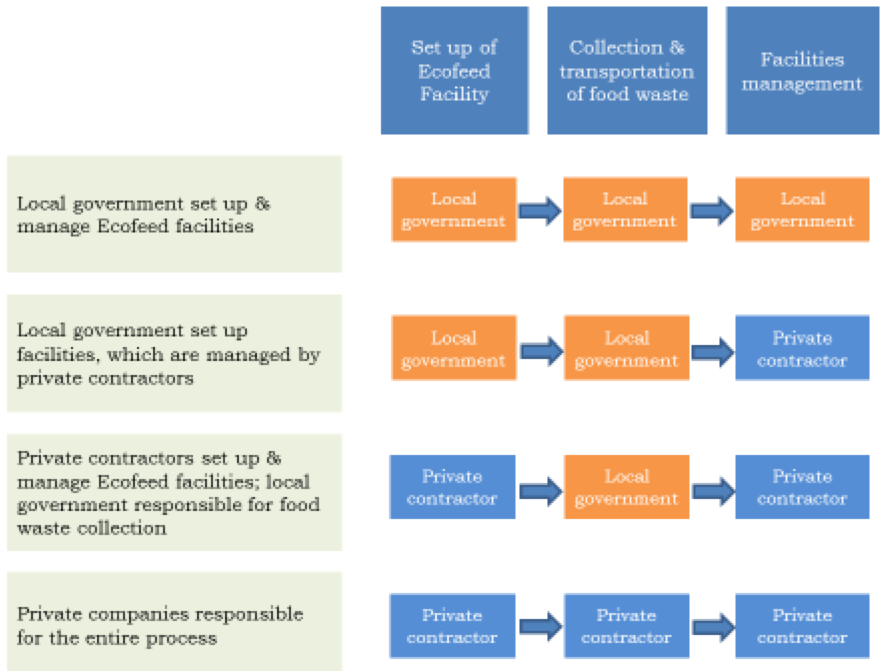

Figure 9.2 – Ecofeed facilities in South Korea are managed through a combination of public and private partnerships.

It is worth noting that foot-and-mouth disease outbreaks in Japan (2010) and South Korea (2010-11) were not linked to swill feeding practices (Muroga et al., 2012; Park et al., 2013).

APPENDIX C – CHAPTER 2, THE LAND USE OF EU PORK PRODUCTION

The great majority of EU pork production occurs in industrial, indoor systems, with 95% of all pork in 2010 coming from farms with more than 50 slaughter pigs (pigs >20kg, reared for slaughter; Figure 9.2). Pork from farms holding more than 50 slaughter pigs is hereafter named "industrial" production. While the diets of pigs in smallholder systems (<50 pigs per farm) may be more variable, industrial pork production is characterised by animals fed grain- and soybean-based diets, maximising feed efficiency, with animal feed sourced off farm, thus decoupling traditional livestock and crop nutrient cycling (Naylor, 2005). As this analysis is concerned with the potential for food waste to replace grain-based feed, we limited our calculation of the land use of EU pork to the 21.5 million tonnes of pork produced in EU industrial systems annually (Eurostat database, 2014). For reference, we list characteristic statistics for EU industrial pork production in Table 9-3.

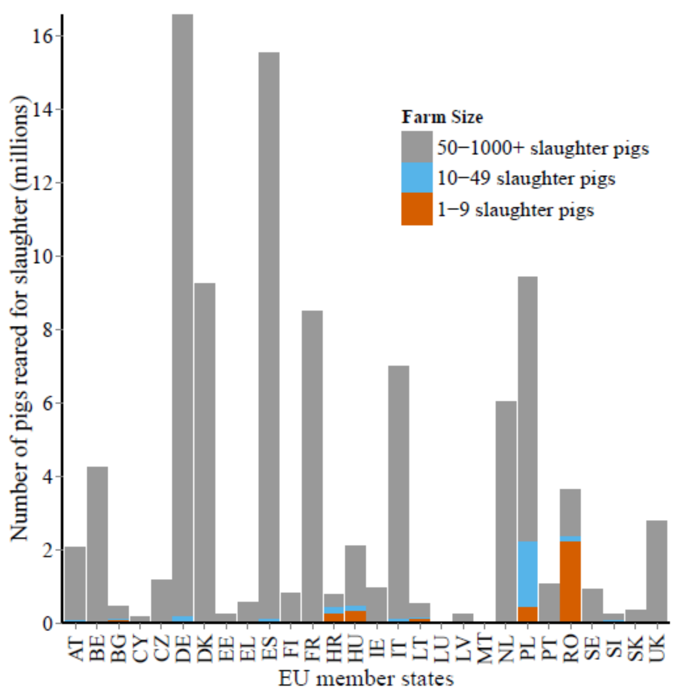

Figure 9.3 – The number of slaughter pigs (all pigs reared for slaughter, excluding breeding animals and piglets <20kg) reared on farms with different herd sizes in the EU in 2010. DE = Germany, ES = Spain,

DK = Denmark, FR = France, PL = Poland, other country codes listed in electronic supplementary material, Table A3. Source: (Eurostat database, 2014).

The land required per kg of EU pork production, $\mathrm{LU}_{\mathrm{kg\ pork\ EU}}$, was calculated as the sum, across all dietary ingredients, *i*, and pig types, *t*, (sows, piglets, and young and mature slaughter pigs) of the land area required to grow the feed necessary to produce 1 kg of pork (live weight) (equation S1).

$$LU_{kg\ pork\ EU} = \sum_{t,i}\left(FCE_t * Prop_{t,i} * \frac{1}{yield_i} * EA_i\right) \quad \text{[Equation S1]}$$

FCE (Feed Conversion Efficiency) is the kg of feed required per kg of pork live weight gain; Prop_i is the proportion of the diet that is ingredient, *i,* on a dry matter basis (Appendix E); $\frac{1}{\mathrm{yield}_i}$ is the area required to produce 1kg of raw product (ha/kg raw product, e.g. soybeans). Finally, EA_i is an economic allocation factor for the proportion of the land required to produce ingredient *i* which is allocated to ingredient *i*, rather than to other co-products. Allocation is used to divide land use between different co-products of a crop, and was weighted according to the economic value of co-products, as in previous life cycle analyses of livestock products (de Vries and de Boer, 2010; Guinée et al., 2004). Soybeans, for example, are processed into both soybean meal, a common pig feed ingredient, and soy oil; soybean meal makes 60% of soybean value, and soy oil the other 40%, and so soybean meal has an EA of 0.6 (see electronic supplementary material, Table A4).

FCE was calculated according to Nguyen et al. (2011) (detailed in electronic supplementary material, Table A5), using weighted mean values of national pig production statistics. Production statistics were weighted according to the proportion of EU production that occurred in each state in 2010, calculated using data available from 12 EU member states (Austria, Belgium, Czech Republic, Denmark, France, Germany, Ireland, Italy, the Netherlands, Spain, Sweden and the UK) (AHDB Market Intelligence, 2013). Member states for which production data were not available were assigned production statistics from states with similar pig production (e.g. the piglet mortality rate in Poland was estimated from Czech production figures; see electronic supplementary material, Table A3).

Table 9-3 – EU pork production statistics.

Summary of data listed in Tables A3, A5, and A7 in the electronic supplementary material (https://doi.org/10.1016/j.foodpol.2015.11.001). These statistics are based on a weighted mean of production statistics from 12 EU member states (representing > 92.9% of EU industrial pork production) and representative diets from 5 EU member states (>64% of EU industrial pork production). Dietary ingredients do not sum to 100% because feed ingredients can belong to multiple groups (e.g. wheat bran is both a cereal and a co-product).

Summary statistics from Tables A3, A5, and A7	Value
EU industrial pork production statistics	
Weight at slaughter (kg)	114.8
Feed conversion ratio (kg feed/kg pig produced)	2.83
Piglets weaned/sow/year	25.1
Ratio sows:slaughter pigs	1:23.6
EU industrial pork diet ingredients in percent (standard deviation)	
Cereals (e.g. oats, corn, wheat)	71.4 (±8.5)
Soybean meal	9.5 (±1.6)
Agricultural co-products (excluding soybean meal, e.g. rapeseed meal, wheat bran, molasses)	14.8 (±5.3)
Vitamin and mineral supplements	3.0 (±1.0)
Other (e.g. peas, cassava, amino acid supplements)	5.0 (±6.0)

$\mathrm{Prop_i}$ was calculated using representative diets from the five leading producers of industrial pork in the EU: Germany, Spain, and Denmark, France, and Poland, which together represent >64% of EU industrial pork production (diets listed in electronic supplementary material, Table A7). Dry matter contents were based on values from (FAO, 2014c) and $\mathrm{Prop_i}$ was calculated using the EU weighted mean, as above.

Yields were calculated from a five-year mean (2008-12) of production from the EU's largest national producer in 2012. For example, France produced 40.3 million tonnes of wheat in 2012 (30% of EU production), with a five year mean yield of 7.1 tonnes/ha (FAO, 2014a). The yields of crops not grown in large quantities in the EU (soybeans, palm oil, tapioca, and safflower oil) were based on a five-year mean of the nation that exported the greatest quantity of that crop into the EU in 2010. To ensure land use was estimated conservatively, we assumed 25% of the Brazilian soybean harvest was double cropped, as in an LCA of Argentinian soybean production (Dalgaard et al., 2008). Our study is concerned with the land required to produce pig feed, and so the land occupied by pig farms themselves (a very small proportion of the area required for feed production) is not considered, as in previous LCAs of pig production (Dalgaard et al., 2008; Reckmann et al., 2013).

Finally, to calculate the total area required for EU pork production, $LU_{kg\ pork\ EU}$ was multiplied by the total output of EU industrial pork production, 21.5 million tonnes (live weight) (Eurostat database, 2014).

Appendix D – Chapter 2, modelling the effect of food waste on land use of pork production

To determine the effect of food waste on the land required for pork production, we searched for relevant literature published between January 1900 and September 2014. To maximise the likelihood of finding relevant studies, we used multiple paired search terms, including ANIMAL FEED, FEED, LIVESTOCK, PIG, or PORK and WASTE, FOOD WASTE, FORMER FOODSTUFF, ECOFEED, SWILL, and RECYCLING. We read the references of identified studies and followed up any which appeared relevant. Studies were translated from the original Japanese or South Korean, where required. We applied the following inclusion criteria for our study: we included only studies which recorded the feed intake and the weight gain of pigs fed conventional and food waste diets, as well as the proportion of the diet that was made up of food wastes.

We identified 18 feed trials comparing the growth performance of pigs on 23 conventional and 55 food waste-based diets (Chae et al., 2003, 2000; Cho et al., 2004; Kjos et al., 1999; Kumar et al., 2014; Kwak and Kang, 2006; Maeda et al., 2014; Márquez and Ramos, 2007; Mitsumoto et al., 2006; Moon et al., 2004; Myer et al., 1999; Nam et al., 2000; Ohmori et al., 2007; Sirtori et al., 2010; Takahashi et al., 2013a, 2013b, 2012; Westendorf et al., 1998). Each feed trial evaluated how pig growth and meat quality were affected by the inclusion of food waste in pig feed; these feed trials mimicked conventional production systems where possible, using, for example, pig breeds common in modern production systems.

For each diet (listed in electronic supplementary material, Table A8), we recorded the proportion of each diet that was food waste (on a dry matter basis) and calculated the land requirement per kg of pork according to equation S1, assigning food waste a land use of zero. The distinction between co-products and food waste can be a grey area. Potato peels or brewing wastes, for example, may be considered a food waste or co-product, depending on whether or not they are a traded commodity. In order to conservatively estimate the land use savings of swill feeding, we classify potato peels, brewing wastes, beet pulp, and dairy wastes (e.g. whey), which are not infrequently used for animal feed in the EU, as co-products and assign them a land requirement accordingly. Previous studies have shown that, compared with grain-based feed, the inclusion of co-products in animal feed can lower the environmental impact of meat production (Elferink et al., 2008), though soybean meal is a notable exception.

Having calculated the land use of each diet, we fitted a linear model to determine the effect of the inclusion of food waste on the land required per kg pork (Fig. 2.2). To allow comparison across different studies, which used different conventional diets (and therefore the land use of conventional diets differed between studies), we fit the land use of each diet as a proportion of the land use of the conventional pig feed in that study. We used untransformed proportion data in our model because errors were approximately normally distributed and applying the logit transformation (Warton and Hui, 2011) reduced model fit ($r = 0.97$ vs 0.94). We tested for differences between four sources of food waste (household wastes, retail [e.g. supermarket] wastes, food service industry [catering or restaurant] wastes, or manufacturing [e.g. sandwich factory] wastes), because food waste composition can vary according to source (Esteban et al., 2007; Zhang et al., 2007), but found no difference between a model pooling food wastes and one differentiating them according to source ($F_{3,76} = 1.78$; $p = 0.157$). The linear model for pooled food wastes was therefore used for subsequent steps in the analysis. All statistical modelling was performed in R version 3.0.1 (R Core Team, 2013).

We find that the inclusion of food waste in pig diets linearly reduces the land required per kg of pork live weight ($r = 0.97$, $n = 78$, $p < 0.0001$). This linear relationship reflects that the inclusion of food waste in pig feed (a) has no effect the feed conversion efficiency (it substitutes conventional feed 1:1 on a dry matter basis), and (b) does not have a large effect on growth rates. While food waste diets do produce slower growth than conventional diets ($t = -4.71$, $p < 0.0001$), in part because their nutrient content is more variable, this effect is relatively small (see Section 2.6.1). If food waste diets did slow growth rates substantially, then the data would be poorly approximated by a linear relationship, and we would see many points in the upper right quadrant of Fig. 2.2 (i.e. above the linear model fitted). To use an example, if a pig fed a 50% food waste diet grows much slower than a pig fed a conventional grain diet, then the animal's total grain use, and the land required for that diet, would decrease by less than 50%, because the animal would be alive for longer before reaching slaughter weight, and would be eating some grain on each of those additional days. We find instead that the relationship is well described by a linear model ($r=0.97$) and has a slope steeper than 1 ($t=-59.2$, $p<0.0001$).

Appendix E – Chapter 2, EU food waste on a dry matter basis

Food waste diets differ greatly in their moisture content, depending on whether fed as a pellet or liquid. We therefore modelled the land required per kg of pork as a function of the proportion of animal feed that is food waste on a dry matter basis.

To determine the proportion of conventional feed that may be replaced by swill we therefore needed to first estimate the dry matter content of EU food wastes. We searched for studies reporting the dry matter content of food wastes by conducting a literature search for studies published between January 1900 and August 2014 using Thomson Reuter's Web of Science® and Google Scholar in August 2014. To maximise the likelihood of capturing relevant studies, we used multiple paired search terms, including the same search terms as in Appendix D. We also searched using the search terms: FOOD WASTE and BIOGAS or ANAEROBIC DIGESTION because the dry matter of food waste is often reported in studies evaluating the potential use of food waste as a biofuel feedstock. We read the references of identified studies and followed up those which appeared relevant.

This literature review identified 220 estimates of the dry matter percentage of food wastes from all four food waste sources (a minimum of 50 estimates for each source, listed in electronic supplementary material, Table A9). We recorded the food waste source, region of origin (EU or non-EU), and dry matter percentages for each estimate. Studies of mixed municipal wastes were not included because of potential contamination with non-food items (e.g. paper and garden wastes) and when a range of dry matters for a particular food waste was quoted, the mean was used. The data were logit transformed (Warton and Hui, 2011) and explored using ANOVAs. There was no difference between the dry matter of food wastes sampled in EU and non-EU countries ($F_{1,215} = 1.42$, $p = 0.235$), so the dry matter estimates of food wastes from all regions were pooled. There was a significant difference between the mean dry matter contents of different food waste sources (Table 9-4; $F_{3,216} = 2.90$, $p = 0.036$) and bootstrapped

95% confidence intervals for the mean dry matter content of each food waste source were computed by resampling 10,000 times with replacement (Table 9-4).

Table 9-4 – Fresh weight and dry matter content of EU food wastes in 2015. The EU food waste figures (EC, 2010) assume that food waste was produced in the same proportions in 2015 as in 2006 (i.e. households and retail wastes, for example, made up 42% and 5% of food wastes in 2015, as in 2006). The 39.2% figures (second row) represent the proportion of food wastes potentially recyclable as animal feed.

	Manufacturing food waste	Retail food waste	Catering food waste	Household food waste
EU food waste (tonnes)	38,786,404	5,122,616	14,343,324	43,029,974
39.2% of food waste recycled as feed (tonnes)[a]	15,204,271	2,008,065	5,622,583	16,867,750
Dry matter content (%) of food waste (95% CI)	29.8 (24.7-36.8)	23.3 (18.0-30.4)	21.5 (20.1-23.0)	26.0 (24.3-27.6)
Food waste recyclable as feed (DM tonnes)	4,530,873	467,879	1,208,855	4,385,615

[a] The figure for the percentage of manufacturing food wastes available for recycling excludes the 3 million tonnes of former foodstuffs which are already used for animal feed in the EU.

APPENDIX F – CHAPTER 2, LAND USE SAVING OF SWILL FEEDING IN THE EU

The potential land use saving of EU swill feeding was calculated according to equation S2, where $LU_{kg\ pork\ EU}$ is the total land area required to produce pork in the EU (main text, Section 2.4.1, Appendix C, and Table 9-5), $coef_{FW}$ is the slope of the relationship between land use and the proportion of pig feed from food waste (Figure 2.2 and Table 9-5), and $FWprop_y$ is the proportion of pig feed in the EU that could be replaced by different food waste sources, y (main text, Section 2.4.3 and Table 9-5). Confidence intervals (95%) for the land use savings were computed using the bootstrapped values of the dry matter content of EU food wastes (Table A10).

$$Reduction\ in\ area\ required\ (ha) = LU_{kg\ pork\ EU} * \sum_y \left(-coef_{FW} * FWprop_y \right) \quad \text{[Equation S2]}$$

This calculation is run twice, first to estimate the land use savings possible if EU legislation were changed and 39.2% of EU food waste were included in pig feed, and second to estimate the land use savings possible under the current legislation. In the latter case, we measure the land use saving possible if two million tons of legal food wastes (known as former foodstuffs), which are not currently used in animal feed, were included in pig feed (see main text, Section 2.4.3 and Table 9-5).

Table 9-5 – Parameters used in land use calculations (Equation S2).

Parameter	Value (95% confidence intervals)
$LU_{kg\ pork\ EU}$	8.5 million ha
$coef_{FW}$	-1.06
$FWprop_{household}$	0.084 (0.079-0.089)
$FWprop_{manufacturing}$	0.087 (0.070-0.104)
$FWprop_{retail}$	0.009 (0.007-0.011)

$FWprop_{catering}$	0.023 (0.022-0.025)
$FWprop_{current\ legislation}$	0.011 (0.009-0.014)

As well as calculating the total land use savings of swill feeding, we also report our results in terms of how the use of swill could reduce demand for both cereals (in tonnes) and soybean production (in hectares). Our previous calculations (Appendix D and E) show that swill can replace 20.3% of EU pig feed (on a dry matter basis). As 71.4% (±8.5% s.d.) of EU conventional pig feed (totalling 60.8 million tonnes) is comprised of cereals, including wheat, barley, oats, triticale, and corn, this suggests swill can replace 8.8 million tonnes of cereals currently used for pig feed. This quantity is equivalent to the annual cereal consumption of 70.3 million EU citizens (124.9 kg cereals/yr/capita of wheat, barley, corn, rye, and oats) (FAO, 2014a). Similarly, when calculating the area of soybean production potentially spared by swill feeding, the area calculated includes only the 9.5% (±1.6% s.d.) of our EU pig feed diets which is comprised of soybean. i.e. we do not double count the savings made from swill replacing soybean and cereals in conventional pig feed.

Equation S2 assumes that the food wastes used in the 38 identified food waste diets are similar in nutrient composition to EU food waste. We believe this to be a valid assumption because: (1) these diets include a representative range of food waste sources, from bakery wastes to household wastes, to supermarket leftovers; (2) we found no difference between the dry matter content of food wastes in EU and non-EU countries suggesting that food waste compositions, though variable between samples, do not differ systematically between locations; and (3) the high rates of food waste recycling as swill in countries such as Japan (35.9%) and South Korea (42.5% of food waste) suggest that many food wastes are suitable as pig feed, if the correct infrastructure is in place to treat them.

APPENDIX G – CHAPTER 2, THE MEAT QUALITY OF PIGS REARED ON FOOD WASTE DIETS

We fitted linear mixed models for 18 meat quality measures which were reported by three or more of the identified studies (Appendix D and Table 2-1). Since pig age and breed, both important determinants of meat quality, varied between studies, study was included as a random effect. Where studies in the literature have postulated a quadratic relationship between the proportion of food waste in diets and meat quality measures (Kjos et al., 1999), quadratic models were also tested. All mixed modelling used the "lme4" package in R and p-values for fixed effects were calculated using Kenward-Roger approximations generated using the "pbkrtest" package (Halekoh and Højsgaard, 2014), and the assumptions of statistical models were tested using a full residual analysis.

When comparing the flavour, juiciness, and overall palatability of pork reared on different diets, because different scales were used in different studies, scores were standardised as a proportion of the maximum potential score. Marbling scores were standardised to a 1-10 scale, in accordance with the National Pork Producers Council scoring system (Takahashi et al., 2012). Colour data were similarly standardised to a 1-5 scale for inter-study comparison. Where drip loss was recorded after multiple time points, the latest recording was used to maximise the likelihood of detecting a difference between the pork reared on conventional and food waste diets.

APPENDIX H – CHAPTER 3, INPUT OUTPUT DATASETS

Table 9-6 – Input-output datasets

Dataset	Description
The UK Input-Output Analytical Tables (IOATs)	The 2010 UK IOATs form the bedrock of the input-output element of the hybrid method. Published in February 2014 (ONS, 2014), the 2010 IOATs are consistent with the Eurostat's Standard Industrial Classification and the classification of products by activity 2008 (EC, 2008; ONS, 2007).
Environmental accounts	The UK environmental accounts were obtained from the latest available version of the UK environmental accounts (ONS, 2013).
Economic model	The economic model was developed using a three-step process: firstly, a project cost model was constructed and used to adjust the input requirements of each option. For any waste management infrastructure, there is usually a detailed cost breakdown which can be used to create the model. Secondly, data was adjusted to the corresponding hybrid LCA-model year to compile with the IOTAs using the consumer price index published by the Office of National Statistics. Finally, expenditure, reported in purchase price, was converted into basic price using a conversion ratio calculated using the UK supply and use tables (Reynolds et al., 2015). In order to avoid double counting, the costs for the materials that are included in the process-based LCA are deducted from the corresponding industrial sectors. Costs of process input were based on unit prices obtained from PRODOCOM data and Spon's architect' and builders' price book (Langdon, 2009).

Sources:

EC (2008) Establishing a New Statistical Classification of Products by Activity (CPA). Brussels, European Enion: Official Journal of the European Union.

Langdon D (2009) Spon's Architects' and Builders' Price Book. Abingdon: Taylor & Francis.

ONS (2013) UK Environmental Accounts -2013. Newport: Office for National Statistics.

ONS (2007) UK Standard Industrial Classification of Economic Activities 2007 (SIC 2007) Structure and Explanatory Notes. L. Prosser ed. Newport: Office for National Statistics.

ONS (2014) United Kingdom Input-Output Analytical Tables 2010. Newport: Office for National Statistics.

APPENDIX I – CHAPTER 3, DATA ON PIG DIETS

Table 9-7 – Conventional pig diets used in the analysis. The feed composition is calculated from a weighted mean of the feed intake of all pigs in the pork production life cycle (sows, piglets, and slaughter pigs). Data from: (Dalgaard et al., 2008; Stephen, 2011)

Animal feed	Composition (kg/kg feed)	Source	ship	truck	ship	Sources
Barley grains	0.22	UK	0	50	0	Assumption: domestic production
Wheat grains	0.29	UK	0	50	0	Assumption: domestic production
Soybeans, at farm	0.11	Brazil	12082	850	1381	Dalgaard et al 2008
Rape seed	0.1	Germany	0	850	0	Dalgaard et al 2008
Wheat grains[1]	0.2	UK	0	50	0	Assumption: domestic production
Molasses, from sugar beet, at sugar refinery	0.04	UK	0	50	0	Assumption: domestic production
Others[2]	0.03	UK	0	50	50	Assumption: domestic production

[1]comprises of Wheat Bran, Endosperm and other starch screenings.

[2]This category consists of molasses, fats, vitamins and minerals.

APPENDIX J – CHAPTER 3, LIFE CYCLE INVENTORY DATA FOR COMPOST SUBSTITUTION

Table 9-8 – Life cycle inventory of compost substitution and use-on-land of a functional unit (i.e. 1 tonne of FW) Sources: Andersen et al., 2010; Hall et al., 2014).

Name	Amount	Unit
Ammonium nitrate phosphate, as N, at regional storehouse, RER	0.1821	kg
Potassium sulphate, as K2O, at regional storehouse, RER	3.126	kg
Single superphosphate, as P2O5, at regional storehouse, RER	2.473	kg
Horn meal, at regional storehouse, CH	2.018	kg
Park chips, softwood, u=140%, at forest road, RER[1]	1.466	m3
Peat, at mine, NORDEL	100.3	kg
Truck, 14t-20t, Euro6, highway	0.00002738	tkm

APPENDIX K – CHAPTER 3, SENSITIVITY ANALYSES

Table 9-9 – Sensitivity analysis parameters for all scenarios. [Table on next page].

Stage	Process	Unit	Distribution type	Average	Deviation
			Dry feed		
Construction	Excluded due to its insignificant contribution to the overall environmental burden.				
Operation	Electricity consumption[2]	Kwh	Log-normal	2.5E-02	2.0E+00
Output utilization	Ecofeed substitution	kg	Normal	1.2E+02	1.7E+01
	Animal feed (conventional production)	kg	Normal	See Table 9-10	
System balance	Electricity conventional production[3]	kwh	Normal	2.6E-01	1.0E-01
	Mineral fertiliser (conventional production)	kg	Normal	6.0E-01	6.0E-02
			Wet feed		
Construction	Excluded due to its insignificant contribution to the overall environmental burden.				
Operation	Electricity consumption[2]	Kwh	Log-normal	3.9E-03	2.0E+00
Output utilization	Ecofeed substitution	kg	Normal	1.6E+02	3.9E+01
	Animal feed (conventional production)	kg	Normal	See Table 9-10	
System balance	Electricity conventional production[3]	kwh	Normal	2.6E-01	1.0E-01
	Mineral fertiliser (conventional production)	kg	Normal	6.0E-01	6.0E-02
			Anaerobic digestion		
Construction	Steel	kg	Uniform	3.6E-04	2.7E-04
	Concrete	m3	Uniform	2.6E-06	2.7E-06
	Bitumen	kg	Uniform	8.2E-05	1.0E-04
	Polyethylene	kg	Uniform	7.3E-04	7.4E-04
Operation	Electricity consumption[2]	Kwh	Log-Normal	4.6E-02	2.1E+00
	Lime[4]	kg	Normal	1.3E-03	1.3E-04
	Inorganic chemicals[4]	kg	Normal	4.9E-03	4.9E-04
Energy Recovery	Electricty recovery	Kwh	Normal	2.6E-01	1.0E-01
System balance	Animal feed (conventional production)	kg	Normal	See Table 9-10	
			Compost		
Construction	Steel	kg	Uniform	1.2.E-03	4.8.E-04
	Concrete	m3	Uniform	4.1.E-06	3.7.E-06
	Aluminum	kg	Uniform	1.0.E-04	2.1.E-04
	Polyethylene	kg	Uniform	2.2.E-05	2.5.E-05
Operation	Electricity consumption[2]	Kwh	Log-normal	1.4.E-03	2.0.E+00
	Diesel	kg	Log-normal	8.7.E-03	2.0.E+00
Output utilization	N fertilizer substitution	kg	Uniform	4.0.E-01	2.0.E-01

System balance	Electricity conventional production[3]	Kwh	Normal	2.6E-01	5.2E-02
	Animal feed (conventional production)	kg	Normal	See Table 9-10	

Table 9-10 – Sensitivity analysis parameters of conventional animal feed [1]Coefficient of variation is assumed to be 10% of the average value.

Name	Unit	Distribution type	Average	Standard deviation[1]
Barley IP, at feed mill, CH	kg	Normal	2.2E-01	2.2E-02
Wheat IP, at feed mill, CH	kg	Normal	2.9E-01	2.9E-02
Wheat IP, at feed mill, CH	kg	Normal	2.0E-01	2.0E-02
Rape seed IP, at regional storehouse, CH	kg	Normal	1.0E-01	1.0E-02
Soybean meal, at oil mill, BR	kg	Normal	1.1E-01	1.1E-02

APPENDIX L – CHAPTER 3, FINANCIAL DATA

Table 9-11 – Financial breakdown of food waste to animal feed technology (animal dry feed). Treatment capacity of this facility is 100 metric ton/day. Costs were converted to GB prices using purchasing power party (year 2014): 852.505 for South Korea and 0.70054 for the UK. Lifespan of the facility is 20 years.

Activity		Cost (GB£)	Note
Public work (civil engineering work)		844,565	Source: Ministry of Environment, Korea (2014)
Building construction and equipment		1,147,720	
Machinery	pre-treatment	887,476	
	main-treatment	2,133,664	
	post-treatment	375,254	
	others	430,097	
	wastewater treatment	1,136,922	
	odor treatment	760,448	
Electric equipment		596,992	
Investment		8,313,138	
Investment per ton	GB£/ton	11	

Table 9-12 – Financial breakdown of food waste to animal feed technology (animal wet feed)

Activity		Cost (GB£)	Note
Public work (civil engineering work)		844,565	Source: Ministry of Environment, Korea (2014)
Building construction and equipment		1,147,720	
Machinery	pre-treatment	887,476	
	main-treatment	648,350	
	post-treatment	210,036	
	others	347,923	
	wastewater treatment	385,970	
	odor treatment	539,059	
Electric equipment		350,060	
Investment		5,361,159	
Investment per ton	GB£/ton	7	

Table 9-13 – Financial breakdown of AD and composting facilities per functional unit. [Table on next page].

SIC-07 code	Industry	Anerobic digestion			Compsoting		
		Construction	Operation	Maintenance	Construction	Operation	Maintenance
14	Wearing apparel	0.0E+00	2.2E-07	0.0E+00	0.0E+00	0.0E+00	0.0E+00
20B	Petrochemicals - 20.14/16/17/60	0.0E+00	0.0E+00	0.0E+00	3.3E-08	2.0E-06	0.0E+00
20.4	Soap and detergents, cleaning and polishing preparations, perfumes and toilet preparations	0.0E+00	2.2E-08	0.0E+00	0.0E+00	2.0E-08	0.0E+00
26	Computer, electronic and optical products	0.0E+00	4.4E-07	0.0E+00	1.7E-08	5.0E-08	0.0E+00
27	Electrical equipment	0.0E+00	3.3E-08	0.0E+00	3.3E-09	5.0E-09	0.0E+00
28	Machinery and equipment n.e.c.	6.3E-06	1.0E-05	0.0E+00	1.7E-06	1.3E-06	0.0E+00
31	Furniture	0.0E+00	6.7E-07	0.0E+00	3.3E-09	2.5E-08	0.0E+00
35.1	Electricity, transmission and distribution	1.1E-07	0.0E+00	0.0E+00	1.7E-08	5.0E-07	0.0E+00
36	Natural water; water treatment and supply services	1.8E-09	0.0E+00	0.0E+00	0.0E+00	5.0E-07	0.0E+00
37	Sewerage services; sewage sludge	1.1E-06	0.0E+00	0.0E+00	0.0E+00	1.0E-07	0.0E+00
38.1	Waste collection	0.0E+00	0.0E+00	0.0E+00	0.0E+00	1.2E-06	0.0E+00
38.2	Treatment and disposal services	0.0E+00	8.4E-07	0.0E+00	0.0E+00	1.2E-06	0.0E+00
39	Remediation services and other waste management services	4.0E-07	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
41-43	Construction	0.0E+00	0.0E+00	0.0E+00	3.3E-06	0.0E+00	0.0E+00
46	Wholesale trade services, except of motor vehicles and motorcycles	0.0E+00	0.0E+00	0.0E+00	0.0E+00	5.0E-07	0.0E+00
49.3-5	Land transport services and transport services via pipelines, excluding rail transport	0.0E+00	0.0E+00	0.0E+00	0.0E+00	2.5E-06	0.0E+00
50	Water transport services	1.1E-08	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
61	Telecommunications services	3.3E-08	0.0E+00	0.0E+00	0.0E+00	1.0E-07	0.0E+00
62	Computer programming, consultancy and related services	1.7E-08	0.0E+00	0.0E+00	0.0E+00	1.0E-07	0.0E+00
63	Information services	5.6E-09	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
64	Financial services, except insurance and pension funding	1.1E-08	0.0E+00	0.0E+00	3.3E-08	0.0E+00	0.0E+00
65.1-3	Insurance, reinsurance and pension funding services, except compulsory social security & Pensions	5.3E-08	7.8E-07	0.0E+00	6.7E-08	1.0E-06	0.0E+00

69.1	Legal services	5.6E-08	2.2E-07	0.0E+00	1.7E-07	5.0E-07	0.0E+00
69.2	Accounting, bookkeeping and auditing services; tax consulting services	0.0E+00	8.9E-07	0.0E+00	0.0E+00	2.0E-06	0.0E+00
70	Services of head offices; management consulting services	0.0E+00	3.0E-06	0.0E+00	0.0E+00	6.7E-06	0.0E+00
71	Architectural and engineering services; technical testing and analysis services	1.1E-06	0.0E+00	0.0E+00	3.3E-08	0.0E+00	0.0E+00
72	Scientific research and development services	0.0E+00	2.2E-07	0.0E+00	0.0E+00	0.0E+00	0.0E+00
74	Other professional, scientific and technical services	0.0E+00	2.2E-06	2.2E-06	0.0E+00	2.5E-07	2.5E-07
77	Rental and leasing services	0.0E+00	3.3E-06	0.0E+00	0.0E+00	6.3E-06	0.0E+00
78	Employment services	0.0E+00	4.4E-07	0.0E+00	0.0E+00	2.5E-07	0.0E+00
80	Security and investigation services	2.8E-08	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
81	Services to buildings and landscape	6.7E-09	2.2E-07	0.0E+00	0.0E+00	0.0E+00	0.0E+00
85	Education services	0.0E+00	2.2E-07	0.0E+00	0.0E+00	0.0E+00	0.0E+00

APPENDIX M – CHAPTER 3, HOTSPOT ANALYSES

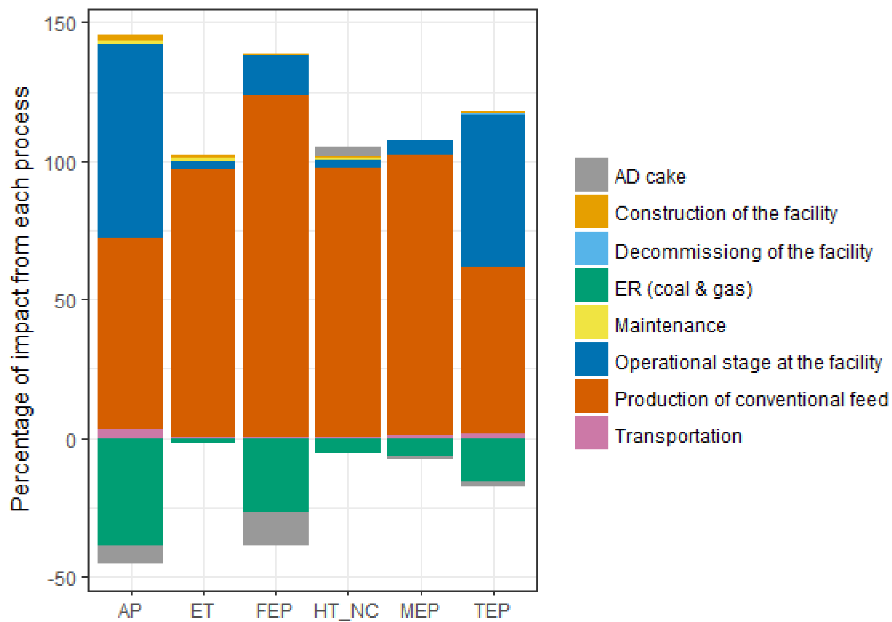

Figure 9.4 – Hotspot analysis of anaerobic digestion of 1 ton of food waste. HT-NC=emissions of non-carcinogenic toxins; FEP=freshwater eutrophication; MEP=marine eutrophication; ET=ecotoxicity; AP=acidification; TEP=terrestrial eutrophication.

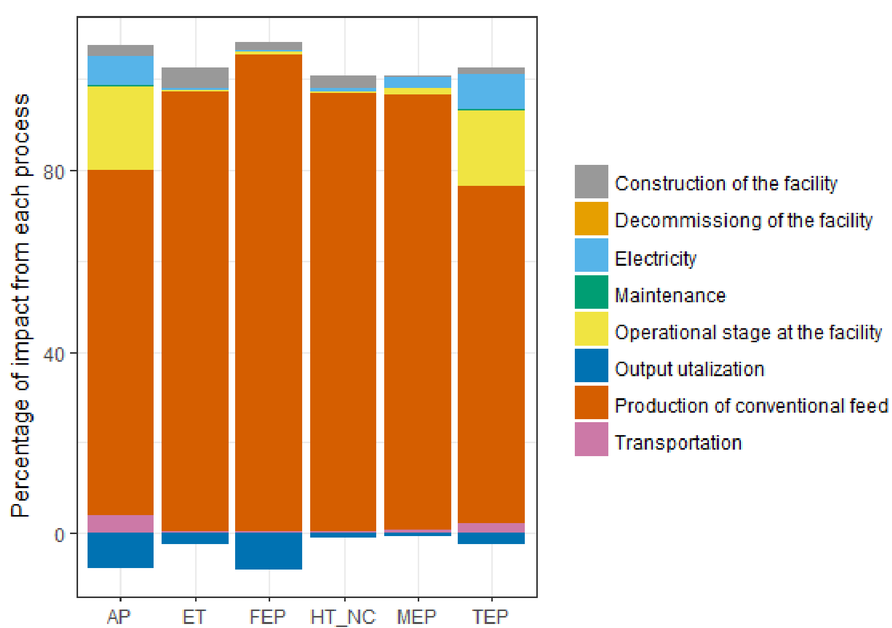

Figure 9.5 – Hotspot analysis of composting 1 ton of food waste. HT-NC=emissions of non-carcinogenic toxins; FEP=freshwater eutrophication; MEP=marine eutrophication; ET=ecotoxicity; AP=acidification; TEP=terrestrial eutrophication.

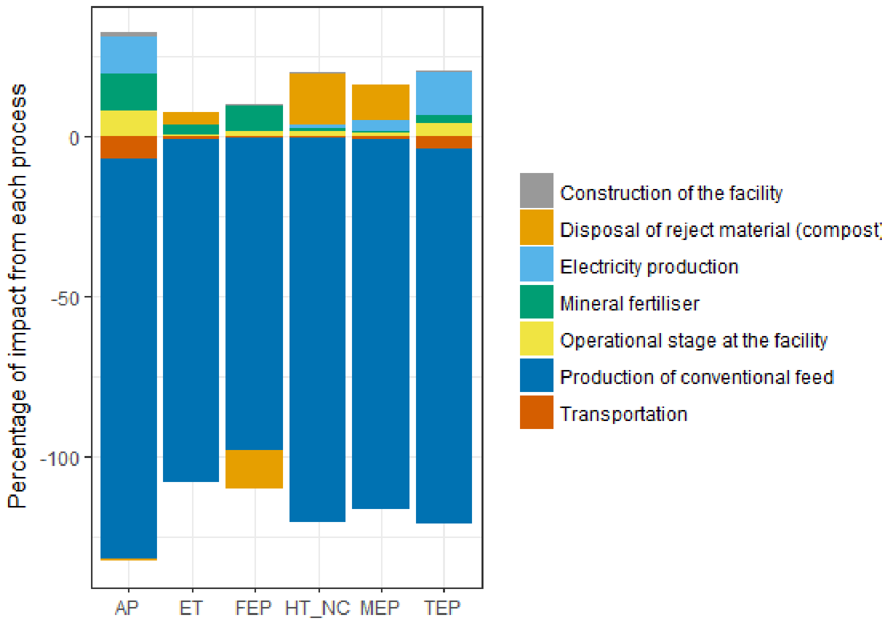

Figure 9.6 – Hotspot analysis of recycling 1 ton of food waste as a dry pig feed. HT-NC=emissions of non-carcinogenic toxins; FEP=freshwater eutrophication; MEP=marine eutrophication; ET=ecotoxicity; AP=acidification; TEP=terrestrial eutrophication.

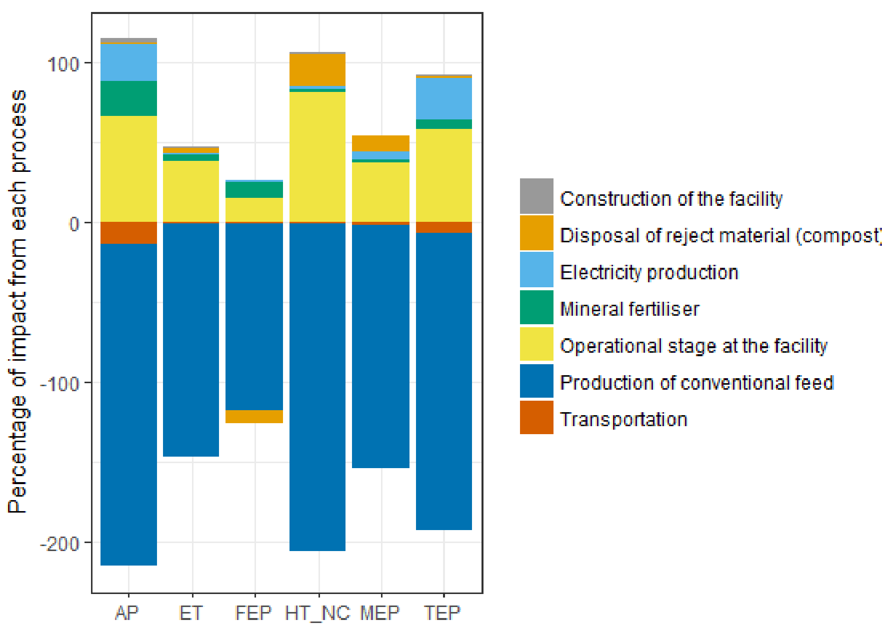

Figure 9.7 – Hotspot analysis of recycling 1 ton of food waste as a wet pig feed. HT-NC=emissions of non-carcinogenic toxins; FEP=freshwater eutrophication; MEP=marine eutrophication; ET=ecotoxicity; AP=acidification; TEP=terrestrial eutrophication.

APPENDIX N – CHAPTER 4, COPY OF SURVEY DONE AT UK PIG & POULTRY FAIR, MAY 2016.

Please complete this survey about the use of swill as pig feed, contribute to University research, and don't miss out on the opportunity to win one of FIVE £50 cash prizes.

You do not need to be a pig farmer to participate.

The survey should take less than 15 minutes to complete.

Please return your completed survey to stall 359A or leave it with one of our team of researchers (wearing maroon t-shirts) who will be collecting them at the exit to the Blackdown buildings.

Thank you for your contribution!

The reason for the survey:

While the use of swill (food leftovers) as animal feed is currently banned in the EU, there are some calls for its re-introduction, following the example of countries like Japan and South Korea which operate regulated systems for heat-treating their food leftovers and recycle 40% of food leftovers as feed.

Our survey wants to hear your opinion on the use of swill as pig feed.

1. Compared with feeding conventional grain- and soybean-based feed, heat-treated swill is:

Much less damaging to the environment	Less damaging to the environment	Neither more nor less damaging to the environment	More damaging to the environment	Much more damaging to the environment	Don't know
☐	☐	☐	☐	☐	☐

Much less nutritious	Less nutritious	Neither more nor less nutritious	More nutritious	Much more nutritious	Don't know
☐	☐	☐	☐	☐	☐

Much less variable in nutritional content	Less variable in nutritional content	Neither more nor less variable in nutritional content	More variable in nutritional content	Much more variable in nutritional content	Don't know
☐	☐	☐	☐	☐	☐

Much lower cost	Lower cost	Costs the same	Higher cost	Much higher cost	Don't know
☐	☐	☐	☐	☐	☐

A much lower disease risk	A lower disease risk	Neither higher nor lower disease risk	A higher disease risk	A much higher disease risk	Don't know
☐	☐	☐	☐	☐	☐

Has much lower microbiological safety	Has lower microbiological safety	Has neither higher nor lower microbiological safety	Has higher microbiological safety	Has much higher microbiological safety	Don't know
☐	☐	☐	☐	☐	☐

Has much lower chemical safety	Has lower chemical safety	Has neither higher nor lower chemical safety	Has higher chemical safety	Has higher chemical safety	Don't know
☐	☐	☐	☐	☐	☐

Much less ethical	Less ethical	Neither more nor less ethical	More ethical	Much more ethical	Don't know
☐	☐	☐	☐	☐	☐

2. How would you feel about the inclusion of the following in pig feed:

Please tick which box applies to each row.

	Very negative	Negative	Neither positive nor negative	Positive	Very positive
Heat-treated household food leftovers	☐	☐	☐	☐	☐
Misshapen chocolates from chocolate factories	☐	☐	☐	☐	☐
Heat-treated, unsold chicken sandwiches from supermarkets	☐	☐	☐	☐	☐
Heat-treated leftovers from a college canteen	☐	☐	☐	☐	☐
Unsold bread from supermarkets	☐	☐	☐	☐	☐
Unsold egg sandwiches from supermarkets	☐	☐	☐	☐	☐
Unsold confectionary containing porcine gelatine	☐	☐	☐	☐	☐
Biscuit crumbs from biscuit factories	☐	☐	☐	☐	☐
Heat-treated, unsold bacon sandwiches from supermarkets	☐	☐	☐	☐	☐
Heat-treated restaurant leftovers	☐	☐	☐	☐	☐

3. How would you feel about the inclusion of the following in pig feed:

Please tick which box applies to each row.

	Very uncomfortable	Uncomfortable	Neither comfortable nor uncomfortable	Comfortable	Very Comfortable
Heat-treated, unsold chicken sandwiches from supermarkets	☐	☐	☐	☐	☐
Unsold confectionary containing porcine gelatine	☐	☐	☐	☐	☐
Heat-treated, unsold bacon sandwiches from supermarkets	☐	☐	☐	☐	☐
Biscuit crumbs from biscuit factories	☐	☐	☐	☐	☐
Heat-treated household food leftovers	☐	☐	☐	☐	☐
Heat-treated restaurant leftovers	☐	☐	☐	☐	☐
Misshapen chocolates from chocolate factories	☐	☐	☐	☐	☐
Unsold bread from supermarkets	☐	☐	☐	☐	☐
Unsold egg sandwiches from supermarkets	☐	☐	☐	☐	☐
Heat-treated leftovers from a college canteen	☐	☐	☐	☐	☐

4. How would you feel about the inclusion of the following in pig feed:

Please tick which box applies to each row.

	Very dissatisfied	dissatisfied	Neither satisfied nor dissatisfied	Satisfied	Very satisfied
Heat-treated restaurant leftovers	☐	☐	☐	☐	☐
Biscuit crumbs from biscuit factories	☐	☐	☐	☐	☐
Unsold confectionary containing porcine gelatine	☐	☐	☐	☐	☐
Unsold bread from supermarkets	☐	☐	☐	☐	☐
Unsold egg sandwiches from supermarkets	☐	☐	☐	☐	☐
Heat-treated leftovers from a college canteen	☐	☐	☐	☐	☐
Misshapen chocolates from chocolate factories	☐	☐	☐	☐	☐
Heat-treated, unsold bacon sandwiches from supermarkets	☐	☐	☐	☐	☐
Heat-treated, unsold chicken sandwiches from supermarkets	☐	☐	☐	☐	☐
Heat-treated household food leftovers	☐	☐	☐	☐	☐

5. To what extent do you agree with the following statements?

Using heat-treated swill would…

	Totally disagree	Disagree	Neither agree nor disagree	Agree	Totally agree	Don't know
Lower dependence on foreign protein sources	☐	☐	☐	☐	☐	☐
Reduce the environmental impact of food waste disposal	☐	☐	☐	☐	☐	☐
Reduce the environmental impact of pork production	☐	☐	☐	☐	☐	☐
Help farms reduce feed costs	☐	☐	☐	☐	☐	☐
Help farmers improve profitability	☐	☐	☐	☐	☐	☐
Lower consumer acceptance of pork products	☐	☐	☐	☐	☐	☐
Increase the risk of an outbreak of foot-and-mouth disease	☐	☐	☐	☐	☐	☐
Increase the risk of prion diseases like BSE (mad cow disease) or vCJD (Creutzfeldt-Jacob disease)	☐	☐	☐	☐	☐	☐
Increase the risk of toxins entering the feed	☐	☐	☐	☐	☐	☐
Reduce the traceability of feed production	☐	☐	☐	☐	☐	☐
Be an efficient way to use food waste	☐	☐	☐	☐	☐	☐
Negatively affect the marketability of pork	☐	☐	☐	☐	☐	☐

6. Compared with pigs fed conventional diets, pigs fed heat-treated swill have:

Much slower growth rates	Slower growth rates	Neither faster nor slower growth rates	Faster growth rates	Much faster growth rates	Don't know
☐	☐	☐	☐	☐	☐

Much higher feed conversion ratios (less efficient)	Higher feed conversion ratios (less efficient)	Has neither higher nor lower feed conversion ratios	Lower feed conversion ratios (more efficient)	Much lower feed conversion ratios (more efficient)	Don't know
☐	☐	☐	☐	☐	☐

Much lower welfare	Lower welfare	Neither higher nor lower welfare	Higher welfare	Much higher welfare	Don't know
☐	☐	☐	☐	☐	☐

Much lower feed costs	Lower feed costs	Neither higher nor lower feed costs	Higher feed costs	Much higher feed costs	Don't know
☐	☐	☐	☐	☐	☐

SURVEY CONTINUED ON NEXT PAGE…

7. Compared with PORK from pigs fed conventional diets, PORK from pigs fed diets containing heat-treated swill is:

Much worse for the environment	Worse for the environment	Neither better nor worse for the environment	Better for the environment	Much better for the environment	Don't know
▢	▢	▢	▢	▢	▢

Much less fatty	Less fatty	Neither more nor less fatty	More fatty	Much more fatty	Don't know
▢	▢	▢	▢	▢	▢

Much lighter in colour	Lighter in colour	Neither lighter nor darker in colour	Darker in colour	Much darker in colour	Don't know
▢	▢	▢	▢	▢	▢

Much less tasty	Less tasty	Neither more nor less tasty	More tasty	Much more tasty	Don't know
▢	▢	▢	▢	▢	▢

Much worse smelling	Worse smelling	Neither better nor worse smelling	Better smelling	Much better smelling	Don't know
▢	▢	▢	▢	▢	▢

Much less marketable	Less marketable	Neither more nor less marketable	More marketable	Much more marketable	Don't know
☐	☐	☐	☐	☐	☐

Much less profitable	Less profitable	Neither more nor less profitable	More profitable	Much more profitable	Don't know
☐	☐	☐	☐	☐	☐

8. To what extent do you agree with the following statements?

	Definitely not	No	Not sure	Yes	Absolutely yes
Feeding swill is a traditional farming practice	☐	☐	☐	☐	☐
Using swill is an unnatural feeding practice.	☐	☐	☐	☐	☐

9. If the procedures were put in place to ensure the safety of swill (e.g. heat treatment was performed by regulated swill manufacturers), would you support the re-legalisation of swill?

Definitely not	No	Not sure	Yes	Definitely yes
☐	☐	☐	☐	☐

10. When considering the re-legalisation of swill, how much importance do you place on the following considerations?

	Not at all important	Not important	Neither important nor unimportant	Important	Very important
Food safety	☐	☐	☐	☐	☐
Traceability	☐	☐	☐	☐	☐
Profitability	☐	☐	☐	☐	☐
Meat quality	☐	☐	☐	☐	☐
Communication with consumers	☐	☐	☐	☐	☐
Environmental impacts	☐	☐	☐	☐	☐
Labelling of the end product	☐	☐	☐	☐	☐
Consumer acceptance	☐	☐	☐	☐	☐
Disease control	☐	☐	☐	☐	☐
Feed prices	☐	☐	☐	☐	☐
Efficient use of resources	☐	☐	☐	☐	☐
Perception of the pork industry	☐	☐	☐	☐	☐

11. What is your gender?

Male	☐
Female	☐

12. What is your age bracket?

0-18	☐
19-30	☐
31-50	☐
51+	☐

13. Please select the profession which best describes your job.

Pig farmer/pig farm manager	☐	Retailer	☐	
Poultry farmer/poultry farm manager	☐	Student	☐	
Farmer/farm manager of both a pig and poultry farm	☐	Veterinarian	☐	
Trader	☐	Food service industry	☐	
Feed processor	☐	Other: involved in the animal industry	☐	Description:
Agricultural advisor	☐	Other: not involved in the animal industry	☐	Description:

If you selected that you are a pig farmer, there are 5 more quick questions, below.

If you are not a pig farmer – thank you for completing the survey! Please hand it in to one of our research team (wearing maroon t-shirts) at stand 359A or the exits to the Blackdown buildings.

To be in with a chance of winning one of our FIVE £50 cash prizes, please list your email address here ________________________________

14. How many pigs do you have at any one time?

1-9	☐
10-99	☐
100-199	☐
200-399	☐
400-999	☐
1000-4999	☐
5000+	☐

15. Do you use wet or dry feeding?

Wet	☐
Dry	☐

16. Have you ever used swill on your farm before?

No	Yes	Not sure
☐	☐	☐

17. If the use of swill were legalised, and procedures were put in place to ensure its feed safety, would you consider using swill on your farm?

Definitely not	No	Might or might not	Yes	Definitely yes
☐	☐	☐	☐	☐

18. Was your farm directly affected by the 2001 Foot and Mouth disease outbreak?

No	Yes	Not sure
☐	☐	☐

Thank you very much for completing our survey and sharing your opinions with us! Please hand it in to one of our research team (wearing maroon t-shirts) at stand 359A or the exits to the Blackdown buildings.

To be in with a chance of winning one of our FIVE £50 cash prizes, please list your email address here ________________________________

If you have any questions about our research, please don't hesitate to come to our stand, or send us a message at CambridgeSwillSurvey@gmail.com

APPENDIX O – CHAPTER 4, ADDITIONAL ANALYSES

Data and code

The data and code used for all analyses can be found at https://github.com/ErasmuszuE/zu_Ermgassen_PLOS_ONE.

Characteristics of survey respondents

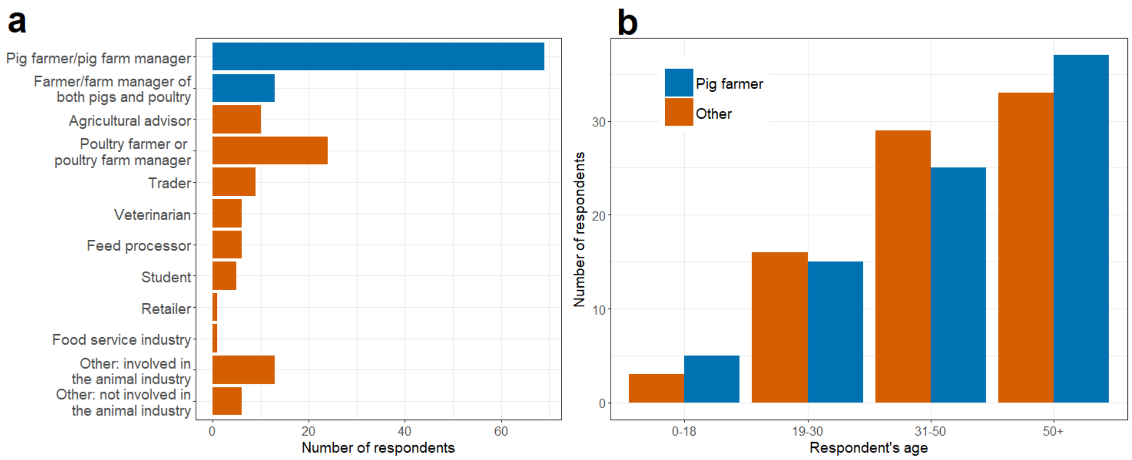

Figure 9.8 – Respondent jobs (a) and ages (b).

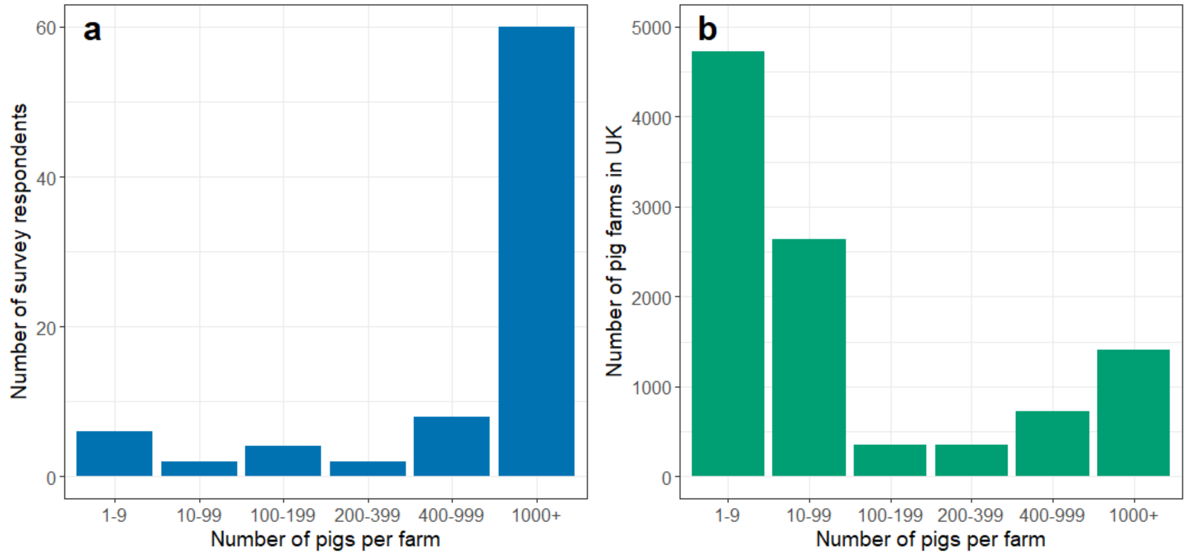

Figure 9.9 – (a) Farm size distribution for the 82 pig farmers who completed the survey; (b) farm size distribution for the 10,190 pig farms in the UK. Source: (Eurostat database, 2014).

Data on the acceptability of different food losses as feed

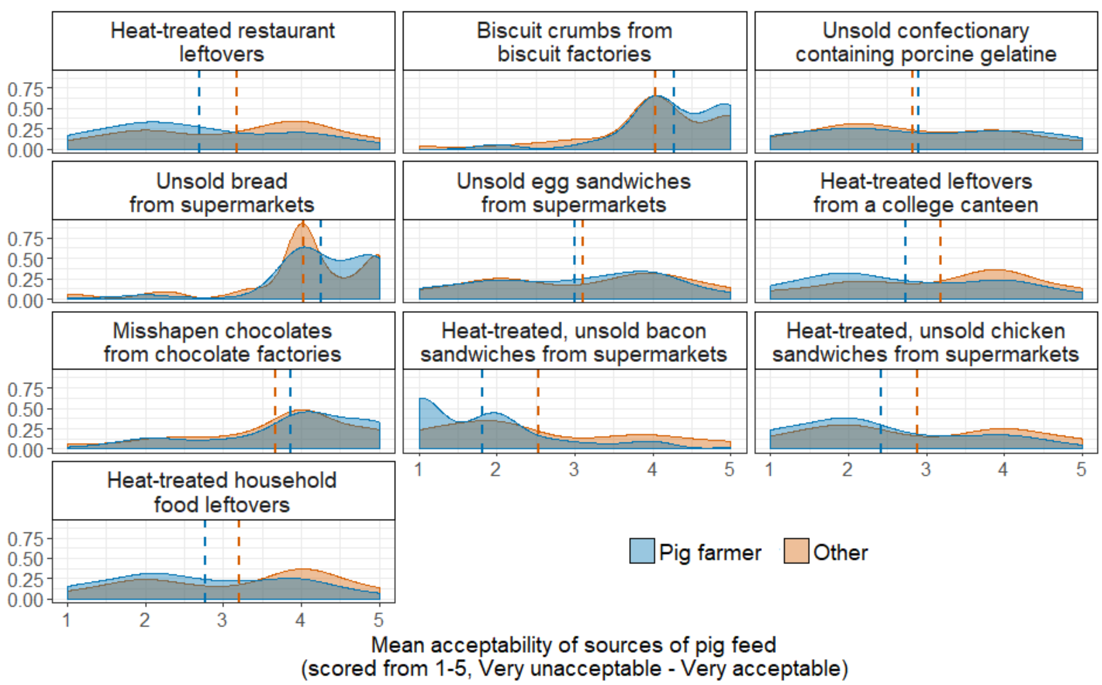

Figure 9.10 – Density plot (i.e. a smoothed histogram) of the acceptability of different feeds. Results are the answer to the question: "How would you feel about the inclusion of the following in pig feed?"

Respondents were asked to score feeds from 1-5, using three different scales: very uncomfortable – very comfortable, very dissatisfied – very satisfied, very negative – very positive. These different constructs had high internal reliability (alpha = 0.95), and so the mean of these scores per respondent is plotted here. The vertical lines designate the overall mean for each feed, per job group.

Models of acceptability of different food losses as feed

The structure and priors used for the maximal model, (model AC1) are described in detail below. The predictors included in subsequent models are described in Table 2; these models were fit using the same priors.

Model AC1:

Model structure

$LikertScore \sim Ordered(\varphi)$ [likelihood]

$logit(\varphi_k) = \alpha_k + \alpha_{RESPONDENT[i]} + \alpha_{FEED[f]} + \beta_{FEED[f]} * J +$ [cumulative link & linear model]

$\beta_{LEGAL} + \beta_{ABP} + \beta_{INTRA\_SPP} + \beta_{JOB,LEGAL}$ […continued]

$\begin{bmatrix} \alpha_{FEED} \\ \beta_{FEED} \end{bmatrix} \sim MVN\left(\begin{bmatrix} 0 \\ \beta \end{bmatrix}, S\right)$ [joint distribution for varying effects]

$S = \begin{pmatrix} \sigma_\alpha & 0 \\ 0 & \sigma_\beta \end{pmatrix} R \begin{pmatrix} \sigma_\alpha & 0 \\ 0 & \sigma_\beta \end{pmatrix}$ [covariance matrix]

Priors

$\alpha_k = Normal(0,10)$ [common prior for each intercept]

$\alpha_{RESPONDENT} = Normal(0, \sigma_R)$ [prior for respondent intercept]

$(\beta, \beta_{LEGAL}, \beta_{ABP}, \beta_{INTRA_{SPP}}) = Normal(0,1)$ [prior for each slope]

$(\sigma_\alpha, \sigma_\beta, \sigma_R) = HalfCauchy(0,2)$ [prior for each σ]

$R = LKJcorr(4)$ [prior for correlation matrix]

Where,

$LikertScore$ is the score for the acceptability of each feed (1-5, Very unacceptable – Very acceptable, Very negative - Very positive, or Very dissatisfied - Very satisfied).

$Ordered$ is an ordered categorical log-odds probability density function.

φ_k is the probability of responding in each category k (below the maximum category *k+1*).

α_k are estimated intercepts for each response category k.

$\alpha_{RESPONDENT[i]}$ are estimated intercepts for different respondents.

$\alpha_{FEED[f]}$ are estimated intercepts for different feed types.

$\beta_{FEED[f]}$ are estimated slopes of the interaction between each feed and job.

J is the value for job (1=pig farmer, 0=other).

β_{LEGAL}= slope for the legal status of the feed (0=illegal, 1=legal).

β_{ABP} = the slope for whether the feed potentially includes animal by-products (0=no ABPs, 1=may contain ABPs).

β_{INTRA_SPP} = slope for whether or not the feed potentially allows intra-species recycling (0=no intra-species recycling, 1=potential for intra-species recycling).

$\beta_{JOB,LEGAL}$ is the slope for the interaction between job (pig farmer=1, other=0), and legal status (0=illegal, 1=legal).

β is the slope common to all feeds.

S is the covariance matrix for the two-dimensional Gaussian distribution linking the intercepts (α_{FEED}) and slopes (β_{FEED}) of each feed.

R is the correlation matrix.

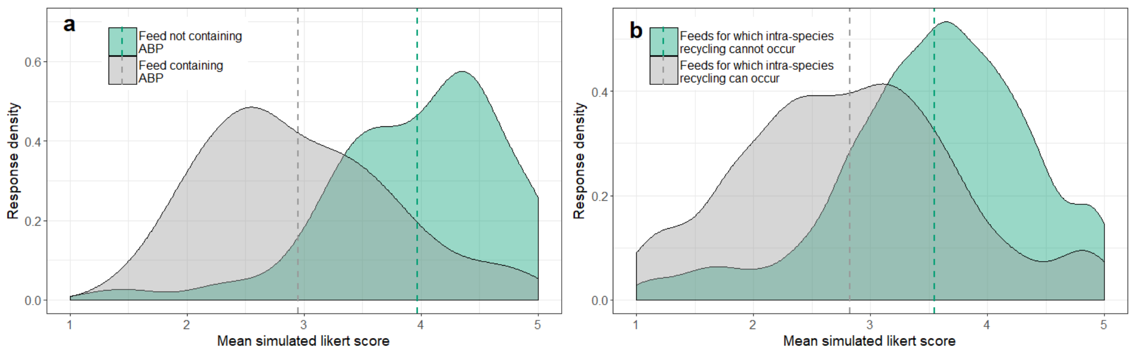

Figure 9.11 – Density plot of the acceptability of (a) feeds with and without animal by-products (ABPs), and (b) feeds for which intra-species recycling can/cannot occur. The data are the mean simulated

responses from 1000 respondents, based on model averaged output (Table 2). The vertical lines designate the overall mean for each category of feeds.

Farmer perceptions of swill

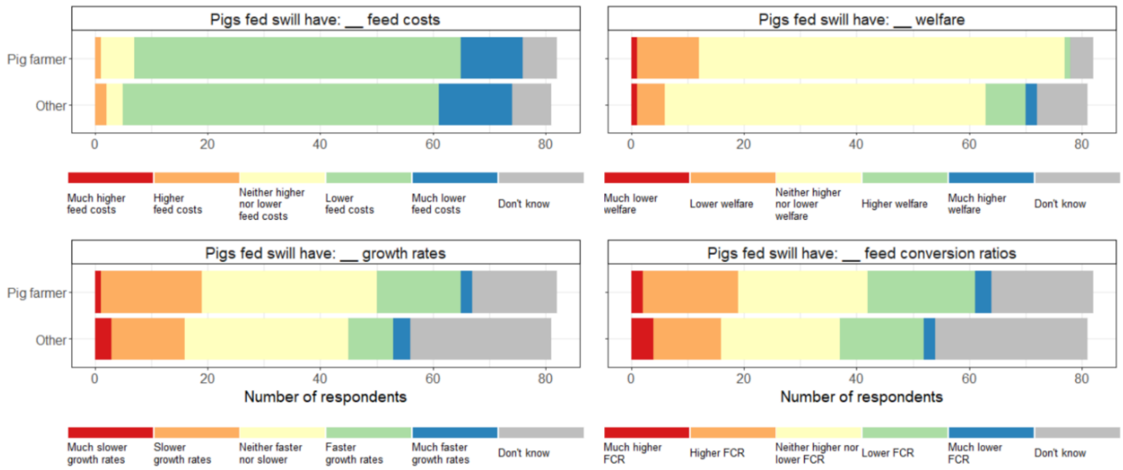

Figure 9.12 – Comparison of pig performance when fed swill or conventional diets. FCR = feed conversion ratio (i.e. how many kilograms of feed are required per kilogram of growth). Responses to the question: "Compared with pigs fed conventional diets, pigs fed heat-treated swill have:".

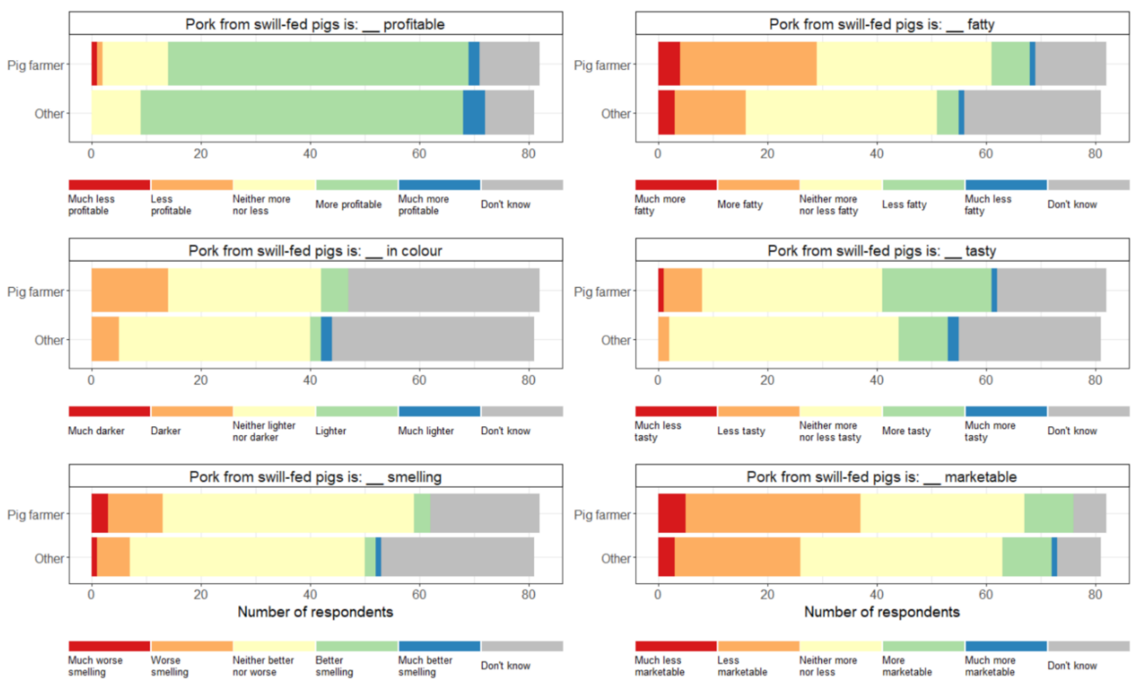

Figure 9.13 – Comparison of the attributes of pork from pigs fed swill or conventional feed. Responses to the question: "Compared with pork from pigs fed conventional diets, pork from pigs fed diets containing heat-treated swill is".

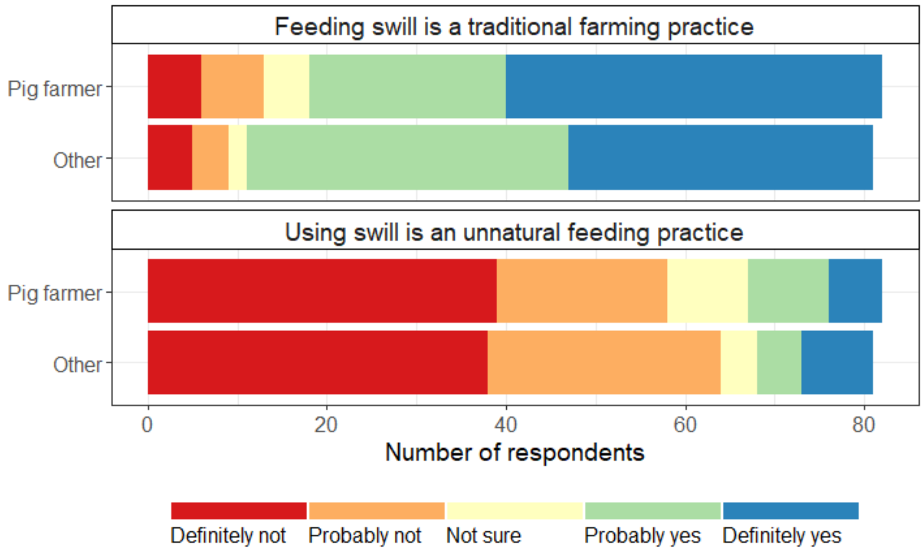

Figure 9.14 – The perception of swill as a traditional or an unnatural farming practice. Response to the question: "To what extent do you agree with the following statements?".

Data used in factor analysis

Farmer values:

The most important issues for respondents, when considering the relegalisation of swill, were food safety (67% thought it was "very important", and 30% "important") and disease control (73% thought it was "very important", 23% "important"), though all of the identified issues were considered to be important by the majority of respondents (Appendix O, Figure 9.15).

These scores for farmer values were simplified using factor analysis, as described in the methods section, and included in models of respondent support for the relegalisation of swill, and farmer willingness to use swill.

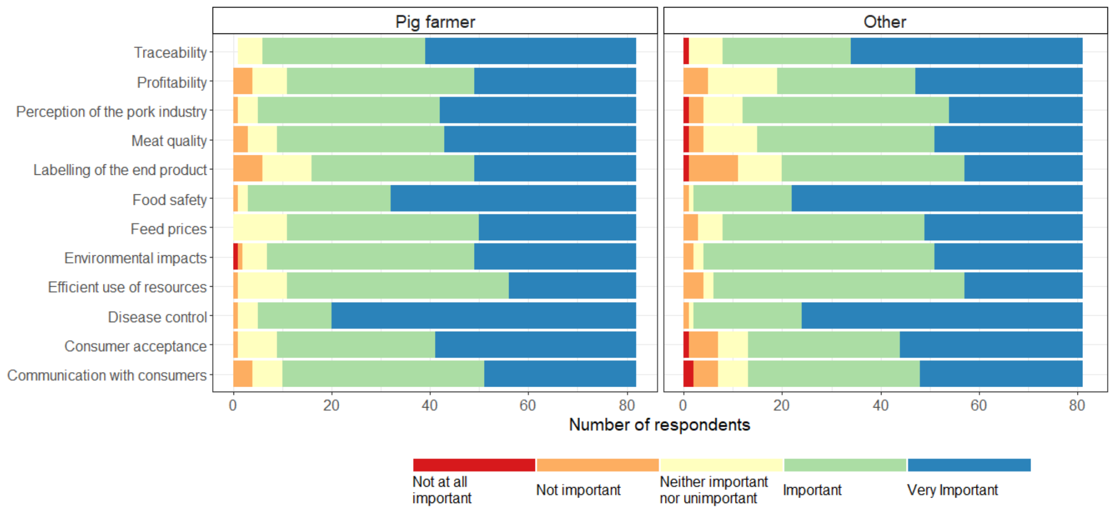

Figure 9.15 – The importance of 12 different issues to respondents. Answer to the question: "When considering the relegalisation of swill, how much importance do you place on the following considerations?". These data were simplified using factor analysis, and included in modelling of the support for relegalisation, and willingness to use swill.

Impacts of swill:

There was very high agreement (>80% "agree" or "totally agree") that using heat-treated swill would be an efficient way to use food waste, would reduce the environmental impact of pork production, and would lower dependence on foreign protein sources (Appendix O, Figure 9.16). Opinions were more evenly split on whether swill would negatively affect the marketability of pork (44% of respondents "agree" or "totally agree", vs 39% who "disagree" or "totally disagree"), increase the risk of an outbreak of foot-and-mouth disease (35% vs 39%), or lower consumer acceptance of pork (40% vs 33%). It is interesting to note that 40% of respondents thought that using swill would increase the risk of prion diseases (such as BSE, or mad cow disease), though there is no evidence of pigs ever naturally contracting prion diseases (Andreoletti et al., 2007).

The scores for respondents' perceptions of the impact of swill were simplified using factor analysis, as described in the methods section, and included in models of respondent support for the relegalisation of swill, and farmer willingness to use swill.

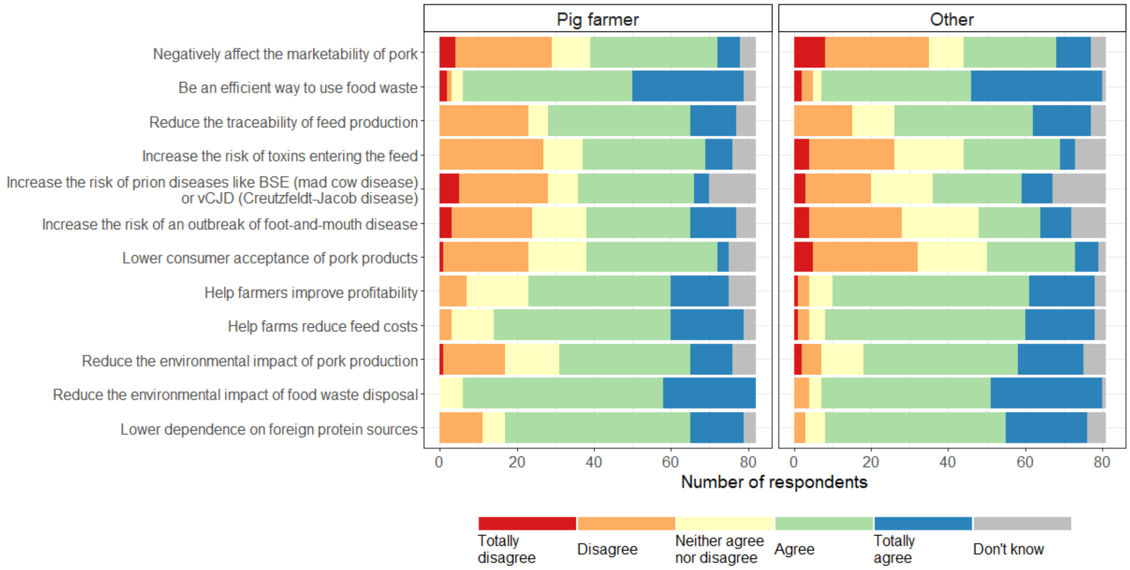

Figure 9.16 – Farmer perceptions of the impacts of swill. Response to the question: "Using heat-treated swill would…". These data were simplified using factor analysis, and included in modelling of the support for relegalisation, and willingness to use swill.

Models of respondents' support for relegalisation of swill

The structure and priors used for the maximal model (model AR1) are described below. The predictors included in subsequent models are described in Table 3; these models were fit using the same priors.

Model AR1:

Model structure

$LikertScore \sim Ordered(\varphi)$ [likelihood]

$logit(\varphi_k) = \alpha_k + \alpha_{AGE_{GROUP[a]}} + \beta_{GENDER} + \beta_{JOB} + \beta_{JOB,GENDER} +$ [cumulative link & linear model]

$\beta_{VAL\_F1} + \beta_{VAL\_F2} + \beta_{IMP\_F1} + \beta_{IMP\_F2}$ […continued]

Priors

$\alpha_k = Normal(0,10)$ [common prior for each intercept]

$\alpha_{AGE\_GROUP[a]} = Normal(0, \sigma_A)$ [prior for age group intercepts]

$(\beta_{GENDER}, \beta_{JOB}, \beta_{JOB,GENDER}, \beta_{VAL_{F1}}, \beta_{VAL_{F2}}, \beta_{IMP_{F1}}, \beta_{IMP_{F2}}) = Normal(0,10)$
[Priors for slopes]

$\sigma_A = HalfCauchy(0,1)$ [prior for σ_A]

Where,

$LikertScore$ is the score for the support for relegalisation of swill, (1-5, Definitely not – Definitely yes), amongst all respondents (n=163).

$Ordered$ is an ordered categorical log-odds probability density function.

φ_k is the probability of responding in each category k (below the maximum category $k+1$).

α_k are estimated intercepts for each response category k.

$\alpha_{AGE\_GROUP[a]}$ is the intercept for different age groups (shown in Appendix O, Figure 9.8).

$\beta_{GENDER}$is the slope for the respondent's gender (1=female, 0=male).

β_{JOB} is the slope for the respondent's job (1=pig farmer, 0=other).

$\beta_{JOB,GENDER}$ is the slope for the interaction between job (1=pig farmer, 0=other) and gender (1=female, 0=male).

$\beta_{VAL_{F1}}$ & $\beta_{VAL_{F2}}$ are the slope for the first and second factor loadings for respondent's values.

β_{IMP_F1}& β_{IMP_F2} are the slope for the first and second factor loadings for respondent's perception of the impacts of swill.

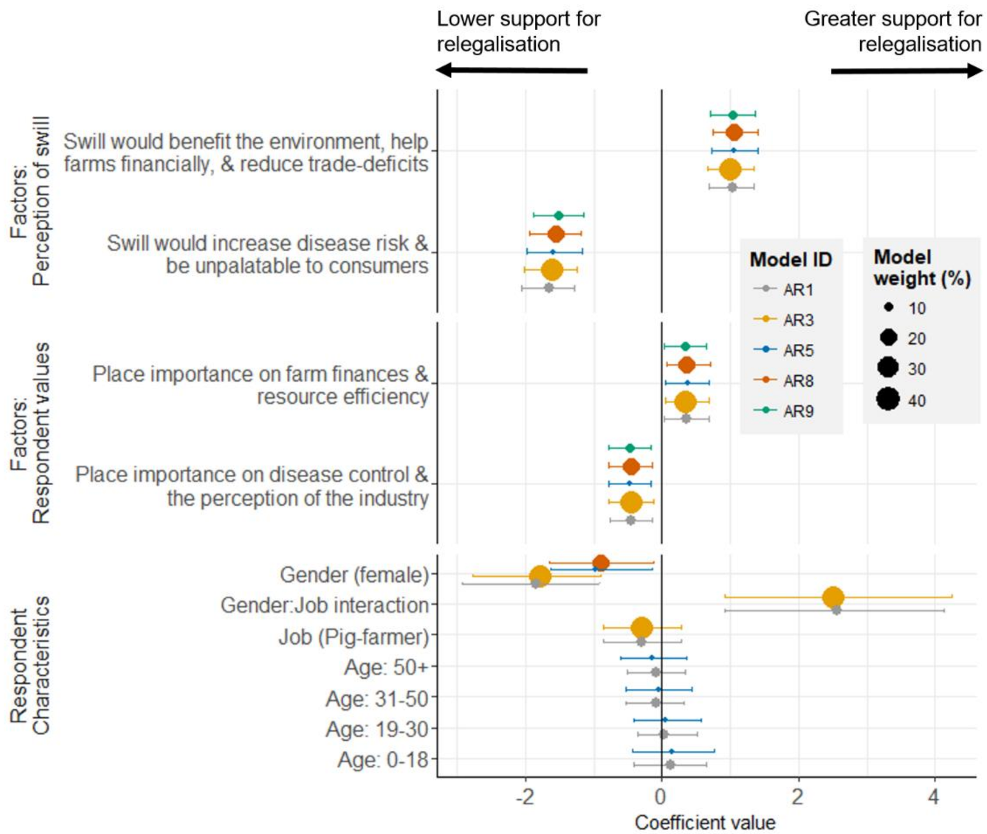

Figure 9.17 – Predictors of the support for the relegalisation of swill, among all respondents (n=163). The estimates plotted are from the five models with the greatest weighting (85% of model weight), where different colours are used for each model and model weights are proportional to the size of the points. Error bars are 89% credible intervals.

Models of farmer support for the relegalisation of swill

The structure and priors used for the maximal model (model FS1) are described below. The predictors included in subsequent models are described in Appendix O, Table 9-14; these models were fit using the same priors.

Model FS1:

Model structure

$LikertScore \sim Ordered(\varphi)$ [likelihood]

$logit(\varphi_k) = \alpha_k + \alpha_{AGE_{GROUP[i]}} + \alpha_{FARM_{SIZE[i]}} + \beta_{GENDER} +$ [cumulative link & linear model]

$\beta_{FMD_{EXP}} + \beta_{SWILL_{EXP}} + \beta_{FEED_{TECH}} +$ [...continued]

$\beta_{VAL\_F1} + \beta_{VAL\_F2} + \beta_{IMP\_F1} + \beta_{IMP\_F2}$ [...continued]

Priors

$\alpha_k = Normal(0{,}10)$ [common prior for each intercept]

$\alpha_{AGE\_GROUP[a]} = Normal(0, \sigma_A)$ [prior for age group intercepts]

$\alpha_{FARM\_SIZE[F]} = Normal(0, \sigma_F)$ [prior for farm size intercepts]

$(\beta_{GENDER}, \beta_{FMD_{EXP}}, \beta_{SWILL_{EXP}}, \beta_{FEED_{TECH}}, \beta_{VAL_{F1}}, \beta_{VAL_{F2}}, \beta_{IMP_{F1}}, \beta_{IMP\_F2}) =$
$Normal(0{,}10)$ [Priors for slopes]

$(\sigma_A, \sigma_F) = HalfCauchy(0{,}1)$ [priors for σ_A, σ_F]

Where,

$LikertScore$ is the score for the support for relegalisation of swill, (1-5, Definitely not – Definitely yes), amongst farmers (n=82).

$\alpha_{AGE\_GROUP[a]}$ is the intercept for different age groups (shown in Appendix O, Figure 9.8).

$\alpha_{FARM\_SIZE[i]}$ is the intercept for different farm sizes (shown in Appendix O, Figure 9.9).

$\beta_{GENDER}$is the slope for the respondent's gender (1=female, 0=male).

$\beta_{FMD_{EXP}}$ is the slope for whether or not the farm was directly affected the 2001 foot-and-mouth outbreak (1=affected, 0=not affected).

β_{SWILL_EXP} is the slope for whether or not the farm has previously used will (1=yes, 0=no).

β_{FEED_TECH} is the slope for whether the farmer uses wet or dry feed (1=wet, 0=dry).

$\beta_{VAL_{F1}}$ & $\beta_{VAL_{F2}}$ are the slope for the first and second factor loadings for respondent's values.

β_{IMP_F1}& β_{IMP_F2} are the slope for the first and second factor loadings for respondent's perception of the impacts of swill.

Table 9-14 - Models predicting support for the relegalisation of swill, amongst pig farmers (n=82).

	Predictors										Model output		
Model	Age group intercepts	Farm size intercepts	Gender	FMD experience	Experience using swill	Feed technology	Values: 1st FL	Values: 2nd FL	Perception of swill: 1st FL	Perception of swill: 2nd FL	pWAIC	WAIC	Model weight
FS1	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	17.9	187.1	0.01
FS2	Y	-	Y	Y	Y	Y	Y	Y	Y	Y	15.7	185.9	0.01
FS3	-	Y	Y	Y	Y	Y	Y	Y	Y	Y	16.4	186.4	0.01
FS4	-	-	Y	Y	Y	Y	Y	Y	Y	Y	13.6	183.9	0.03
FS5	-	-	-	Y	Y	Y	Y	Y	Y	Y	12.5	182.1	0.08
FS6	-	-	Y	Y	-	Y	Y	Y	Y	Y	12.3	182.2	0.08
FS7	-	-	Y	Y	Y	-	Y	Y	Y	Y	12.3	184.8	0.02
FS8	-	-	Y	-	Y	Y	Y	Y	Y	Y	12.1	182.1	0.08
FS9	-	-	Y	-	-	Y	Y	Y	Y	Y	10.9	180.6	0.16
FS10	-	-	-	Y	-	Y	Y	Y	Y	Y	11.1	180.1	0.21
FS11	-	-	-	-	Y	Y	Y	Y	Y	Y	11.0	180.4	0.18
FS12	-	-	-	Y	Y	-	Y	Y	Y	Y	11.1	183.0	0.05
FS13	-	-	Y	Y	-	-	Y	Y	Y	Y	10.9	182.7	0.06
FS14	-	-	Y	-	Y	-	Y	Y	Y	Y	10.9	184.8	0.02

"FL" is factor loading; WAIC is the widely applicable information criterion score; pWAIC is the number of effective parameters.

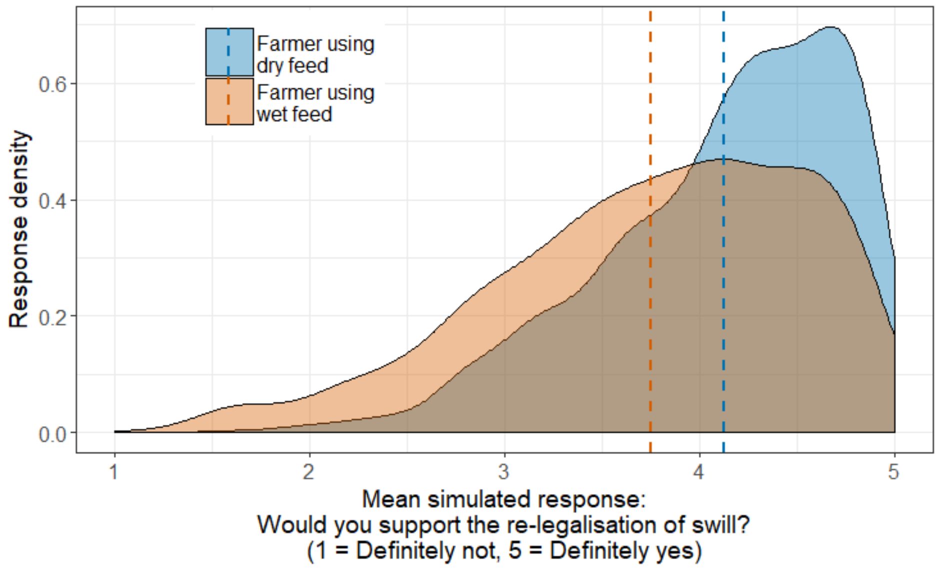

Figure 9.18 – Density plot of farmer support for the relegalisation of swill, comparing farmers who use wet or dry feed. The data are the mean simulated responses from 1000 respondents, based on model output from the three models with greatest weight (55% of model weight; Appendix O Table 9-14). The vertical lines designate the overall mean for each category of feeds.

Models of farmer willingness to use swill, if it were relegalised

The structure and priors used for the maximal model (model WU1) are described below. The predictors included in subsequent models are described in Appendix O Table 9-15; these models were fit using the same priors.

Model WU1:

Model structure

$WillLikert \sim Ordered(\varphi)$ [likelihood]

$logit(\varphi_k) = \alpha_k + \alpha_{AGE_{GROUP[i]}} + \alpha_{FARM_{SIZE[i]}} + \beta_{GENDER} +$ [cumulative link & linear model]

$\beta_{FMD_{EXP}} + \beta_{SWILL_{EXP}} + \beta_{FEED_{TECH}} +$ [...continued]

$\beta_{VAL\_F1} + \beta_{VAL\_F2} + \beta_{IMP\_F1} + \beta_{IMP\_F2}$ [...continued]

Priors

$\alpha_k = Normal(0,10)$ [common prior for each intercept]

$\alpha_{AGE\_GROUP[i]} = Normal(0, \sigma_A)$ [prior for age group intercepts]

$\alpha_{FARM\_SIZE[i]} = Normal(0, \sigma_F)$ [prior for farm size intercepts]

$(\beta_{GENDER}, \beta_{FMD_{EXP}}, \beta_{SWILL_{EXP}}, \beta_{FEED_{TECH}}, \beta_{VAL_{F1}}, \beta_{VAL_{F2}}, \beta_{IMP_{F1}}, \beta_{IMP\_F2}) = Normal(0,10)$ [Priors for slopes]

$(\sigma_A, \sigma_F) = HalfCauchy(0,1)$ [priors for σ_A, σ_F]

Where,

$WillLikert$ is the score for the willingness to use swill, if it were relegalised, (1-5, Definitely not – Definitely yes), amongst farmers (n=82).

$\alpha_{AGE\_GROUP[a]}$ is the intercept for different age groups (shown in Appendix O, Figure 9.8).

$\alpha_{FARM\_SIZE[i]}$ is the intercept for different farm sizes (shown in Appendix O, Figure 9.9).

$\beta_{GENDER}$is the slope for the respondent's gender (1=female, 0=male).

$\beta_{FMD_{EXP}}$ is the slope for whether or not the farm was directly affected the 2001 foot-and-mouth outbreak (1=affected, 0=not affected).

β_{SWILL_EXP} is the slope for whether or not the farm has previously used will (1=yes, 0=no).

β_{FEED_TECH} is the slope for whether the farmer uses wet or dry feed (1=wet, 0=dry).

$\beta_{VAL_{F1}}$ & $\beta_{VAL_{F2}}$ are the slope for the first and second factor loadings for respondent's values.

β_{IMP_F1}& β_{IMP_F2} are the slope for the first and second factor loadings for respondent's perception of the impacts of swill.

Table 9-15 - Models predicting farmer willingness to use swill, if it were relegalised (n=82).

Model ID	Age group intercepts	Farm size intercepts	Gender	FMD experience	Experience using swill	Feed technology	Values: 1st FL	Values: 2nd FL	Perception of swill: 1st FL	Perception of swill: 2nd FL	pWAIC	WAIC	Model weight
	Predictors										Model output		
WU1	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	17.1	248.5	0.00
WU2	Y	-	Y	Y	Y	Y	Y	Y	Y	Y	14.9	245.4	0.00
WU3	-	Y	Y	Y	Y	Y	Y	Y	Y	Y	15.0	246.2	0.00
WU4	-	-	Y	Y	Y	Y	Y	Y	Y	Y	12.8	243.9	0.00
WU5	-	-	-	Y	Y	Y	Y	Y	Y	Y	12.0	242.0	0.01
WU6	-	-	Y	Y	-	Y	Y	Y	Y	Y	12.0	245.2	0.00
WU7	-	-	Y	Y	Y	-	Y	Y	Y	Y	11.6	241.6	0.01
WU8	-	-	Y	-	Y	Y	Y	Y	Y	Y	11.5	241.5	0.02
WU9	-	-	Y	-	-	Y	Y	Y	Y	Y	10.7	243.5	0.01
WU10	-	-	-	Y	-	Y	Y	Y	Y	Y	11.3	243.7	0.01
WU11	-	-	-	-	Y	Y	Y	Y	Y	Y	10.5	239.2	0.05
WU12	-	-	-	Y	Y	-	Y	Y	Y	Y	10.7	239.7	0.04
WU13	-	-	Y	Y	-	-	Y	Y	Y	Y	10.7	243.1	0.01
WU14	-	-	Y	-	Y	-	Y	Y	Y	Y	10.3	239.4	0.05
WU15	-	-	Y	-	-	-	-	-	Y	Y	6.8	238.7	0.06
WU16	-	-	-	Y	-	-	-	-	Y	Y	6.9	238.7	0.06
WU17	-	-	-	-	Y	-	-	-	Y	Y	6.9	234.9	0.42
WU18	-	-	-	-	-	Y	-	-	Y	Y	6.8	238.5	0.07
WU19	-	-	-	-	-	-	-	-	Y	Y	5.7	236.6	0.18

"FL" is factor loading; WAIC is the widely applicable information criterion score; pWAIC is the number of effective parameters.

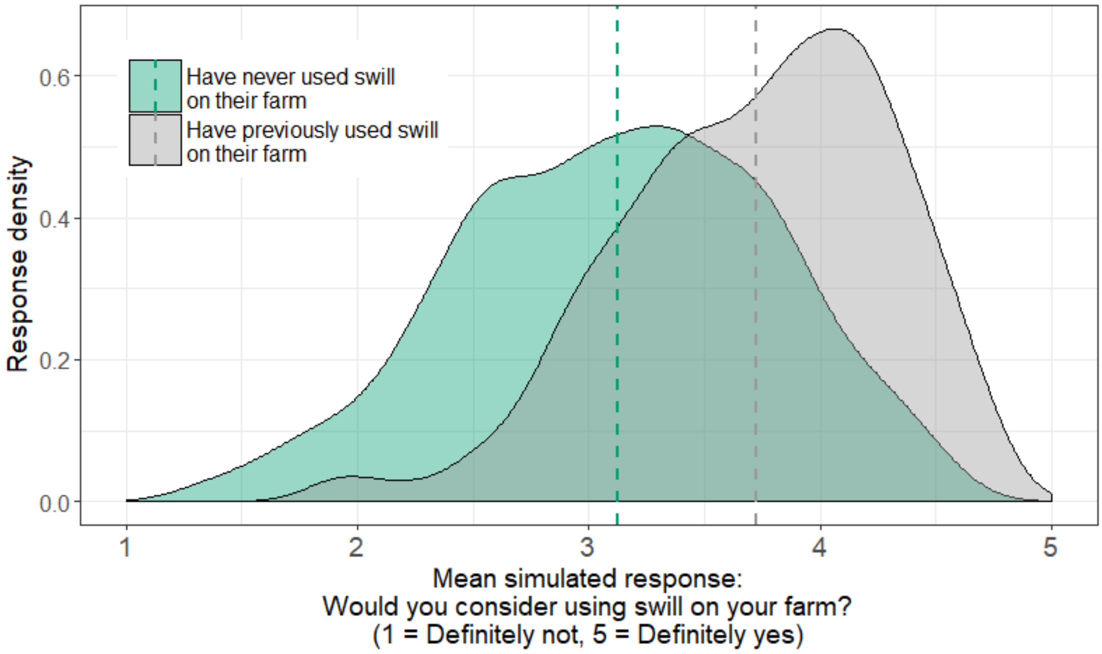

Figure 9.19 – Density plot comparing the effect of a respondent having previous experience of using swill on their willingness to use swill, if it were relegalised. The data are the mean simulated responses from 1000 respondents (controlling for other factors), based on model output from the model with highest weight (model WU17; Appendix O, Table 9-15).

APPENDIX P – CHAPTER 5, COPY OF BEEF SURVEY

Seção 1 - Perguntas gerais:
1. Nome do projeto de intensificação:
2. Instituição(ões) envolvida(s) no projeto:
3. Nome da pessoa completando este formulário:

Seção 2 - Detalhes do projeto e da tecnologia		
4. Qual tipo de pecuária está intensificada no projeto? *Assinale as caixas que se aplicam:*	Pecuária de corte	
	Pecuária de leite	

Se você marcou "pecuária de leite", tem um outro questionário para completar, com questões de leite. Se você não o tem, por favor contate *ekhjz2@cam.ac.uk*.

5. Qual tipo de pecuária está intensificado no projeto? **Marque mais que um, se aplicam:**	**Cria**	
	Recria/engorda	
	Ciclo completo	

6. Municípios (e Estado) do projeto:
Caso que o projeto está em mais de um município, liste todos os municípios por favor. Ex: São Félix do Xingu (PA), Santarém (PA).

7. Você tem os endereços ou pontos de GPS das fazendas? Coloque abaixo.	
Estes dados seriam usados somente para gerar mapas, para o artigo sobre a intensificação. Os dados são anônimos e não são vinculados a nenhuma fazenda em particular. Não precisa os compartilhar, se não quiser.	

8. Ano do início do projeto/introdução da tecnologia:	

9. Quantos propriedades rurais são atendidas pelo projeto?	

10. Tem dados sobre o número de propriedades adotando a tecnologia de intensificação, por ano? Se não, vá à pergunta onze. Se sim: Complete, por favor a tabela abaixo. Um exemplo está completado:	Sim	
	Não	

EXEMPLO	Ano _1__	Ano _2__	Ano _3__	Ano _5__	Ano _10__
Número de propriedades	5	10	45	~100	~120
RESPOSTA	Ano ___	Ano ___	Ano ___	Ano ___	Ano ___
Número de propriedades					

11. As fazendas intensificadas tem quantos hectares de pasto no total?	ha	Não sei

Se não sabe, marque a opção correspondente, por favor.		

12. As fazendas intensificadas tem quanto gado no total? Se não sabe, marque a opção correspondente, por favor.	cabeças	Não sei

13. Quais tamanhos da fazenda estão no projeto/tem adotado a tecnologia:			
	Mínimo	Máximo	Média
EXEMPLO Número de hectares	100	10000	500
RESPOSTA Número de hectares			

14. Como propagar conhecimento do projeto ou da tecnologia de intensificação? Marque todas as células que se aplicam, e por favor classifique-as em ordem de importância. 1=mais importante, 7=menos importante; NA = não se aplica. Abaixo há um exemplo preenchido, em que a mídia local é o mais importante(1), os dias de campo são o menos importante (5), as consultorias são mais ou menos importantes (3), e o pessoal não tem treinamento da assistência técnica (NA).			
EXEMPLO			
A mídia local___1___	Boca-a-boca ___4___	Treinamento da assistência técnica ___NA___	Outros: ___NA___ ________
As consultarias agrícolas ___3___	Dias de campo ___5__	Relatórios disponível no internet ___2___	

RESPOSTA:			
A mídia local_______	Boca-a-boca _______	Treinamento da assistência técnica _______	Outros: _______ _______
As consultarias agrícolas _______	Dias de campo _______	Relatórios disponível no internet _______	

15. **Tem condições para participar do projeto ou adotar a tecnologia?**
Ex: Fazendas no projeto precisam de adequação ambiental?

Seção 3 - Detalhes da intensificação
16. Quais tipos de capim são utilizados?
Liste as espécies de capim, por favor.

17. **Tem consórcio com leguminosa?**	**Sim**	
	Não	
Caso que respondeu “Sim” acima - quais espécies?		

18. **Quais são as estratégias adotadas?** Se você marca uma opção, dê uma descrição, por favor. Um exemplo já está completado.
EXEMPLO:

Marque as opções que se aplicam	Maneej	Descrição do programa
X	Pastejo rotacionado	Uma área de pasto de 100ha está dividido em *piquetes de ca. 10 ha. No periodo de gorda (quase 6 meses) os bois entram nos piquetes e estão movidos quase cada 5 dias de um piquete ao outra - depende da chuva e a altura do capim, quem está manejado para manter uma altura de 50cm. As vacas não usam as áreas intensificadas, usando só o pasto conventional. Etc...*
	Manejo reprodutivo	
X	Manejo no período da seca	Nós damos 1kg/cabeça/dia de silagem de cana... Etc...

RESPOSTAS:		
Marque as opções que se aplicam	Manejo	Descrição
	Pastejo rotacionado	

<table>
<tr><td></td><td>Manejo reprodutivo</td><td colspan="3"></td></tr>
<tr><td></td><td>Bem-estar animal e manejo pré-abate</td><td colspan="3"></td></tr>
<tr><td></td><td>Controle de informações e gerenciamento</td><td colspan="3"></td></tr>
<tr><td></td><td>Treinamento de mão-de-obra</td><td colspan="3"></td></tr>
<tr><td></td><td>Capacitação de técnicos</td><td colspan="3">Caso se aplique, quantos técnicos capacitados ou em capacitação?
______________________________</td></tr>
<tr><td></td><td>Creep-feeding</td><td colspan="3"></td></tr>
<tr><td rowspan="5"></td><td rowspan="5">Suplementação alimentar</td><td rowspan="5">Quando está a ração suplementar usado?
Marque mais que um, se aplicam:</td><td>Não usa</td><td></td></tr>
<tr><td>O ano inteiro</td><td></td></tr>
<tr><td>Na seca</td><td></td></tr>
<tr><td>No período de terminação</td><td></td></tr>
<tr><td>Outra época: __________</td><td></td></tr>
</table>

		Os animais que recebem ração suplementar tem quantos kg de ração/cabeça/dia?	Kg/cabeça/dia	
		Tem confinamento ou semi-confinamento?	Sim	
			Não	
		Caso se aplique: quantos meses de confinamento/semi-confinamento em média antes de abate?	Meses	
		Usa sal-mineral suplementar?	Sim	
			Não	
		Outros detalhes da suplementação:		
	O programa de intensificação inclui acesso ao crédito?			
	O produto das fazendas intensificadas é diferenciado no mercado?	Ex: sistema de certificação		
	Outros estratégias adotadas - quais? ____________	Ex: integração lavoura-pecuária		

Seção 4 - Implementação da intensificação

19. Qual é o custo de intensificação do pasto?						
	Partindo do pasto degradado			Partindo do pasto normal		
	Mín R$ /ha	Máx R$ /ha	Média R$ /ha	Mín R$ /ha	Máx R$ /ha	Média R$ /ha
Reforma de pasto						
Recuperação de pasto						

20. Descriminação dos insumos e custos médios da implementação:		
	Partindo do pasto normal	
	Quantidade (Kg/ha)	Custo (R$/ha)
Calcário		
Nitrogênio		
Outros insumos: Quais? __________________ __________________ Ex: arame, sementes, uso de equipamento		

Seção 5 - Manutenção da área intensificada			
21. Qual é o custo anual de manutenção da pastagem intensificada?			
	Mínimo	Máximo	Média
Custo anual de manutenção (R$/ha/ano)			

22. Descriminação do custo médio anual de manutenção da pastagem intensificada:			
	A frequência com que é aplicada? (ex: anual/cada 5 anos)	Quantidade (Kg/ha)	Custo (R$/ha/vez)
Calcário			
Nitrogênio			
Outros custos: Quais? ________________ ________________ Ex: Mão de obra			

23. Qual é o custo da Assistência Técnica por propriedade/ano?

	Mínimo	Máximo	Média
Custo anual da assistência técnica (R$/ha/ano)			

24. O projeto aumentou a demanda por mão-de-obra na fazenda?	**Sim**	
	Não	
Caso que respondeu "Sim", *quanto maior é* a demanda por mão de obra? Por exemplo: quase 20% maior. __________________ % maior demanda para mão de obra		

Seção 6 - Produtividade e outros parâmetros			
25. Qual é a produtividade alcançada?			
	Mínimo @/ha/ano	Máximo @/ha/ano	Média @/ha/ano
Ano 0 (ano base)			
Ano 1 a 2			
Ano 3+			

26. Qual é a taxa de lotação? **(UA/há, na fazenda inteira)**			
	Mínimo (UA/ha)	Máximo (UA/ha)	Média (UA/ha)
Ano 0 (ano base)			
Ano 1 a 2			

Ano 3+			

27. **Qual é a idade de abate?**			
	Mínimo	Máximo	Média
A idade de abate	Meses	Meses	Meses

28. **Qual é o peso dos animais abatidos?**			
	Mínimo	Máximo	Média
O peso dos animais abatidos? <u>(Peso vivo)</u>	Kg	Kg	Kg

29. **Qual é o tempo médio para atingir a produtividade ótima?**			
	Mínimo	Máximo	Média
Anos para atingir a produtividade ótima?	Anos	Anos	Anos

30. **Qual é o tempo médio para atingir o retorno econômico do investimento com intensificação?**			
	Mínimo	Máximo	Média
Anos para atingir um retorno econômico	Anos	Anos	Anos

31. Qual é a rentabilidade média por hectare por ano (R$/ha/ano), alcançada?			
	Mínimo	Máximo	Média
Rentabilidade média (R$/ha/ano)			

Seção 7 - Outros indicadores		
32. O projeto realiza o monitoramento do:		
a) Balanço de carbono	Sim	
	Não	
Caso se aplique, pode descrever:		
b) Quantidade e qualidade da água	Sim	
	Não	
Caso se aplique, pode descrever:		
c) Adequação ambiental	Sim	
	Não	
Caso se aplique, pode descrever:		
a) Manejo integrado de pragas	Sim	
	Não	
Caso se aplique, pode descrever:		

Seção 8 - Reflexão sobre a sistema de intensificação
33. Favor, descreva a(s) principal(is) barreira(s) para a implementação da iniciativa

34. Quais são as soluções para vencer estas barreiras?
35. Observações complementares que possa julgar interessante relatar:

36. Se você tem fotos dos sistemas intensificados, por favor compartilhe-as, para acompanhar a descrição do sistema de intensificação no artigo.

Obrigado para completar o questionário!

Estamos muito animados para escrever esse artigo juntos.

APPENDIX Q – CHAPTER 5, ADDITIONAL DISCUSSION.

As well as impacts on productivity, greenhouse gases, and deforestation, cattle ranching intensification also has repercussions for animal welfare, nutrient cycling, and farm labour conditions. For a more detailed description of the risks and potential benefits of cattle intensification, readers are directed toward (Latawiec et al., 2014).

Though high-productivity livestock production can compromise animal welfare, there is plenty of opportunity for Brazilian cattle production to simultaneously improve productivity and animal welfare. The productivity increases achieved in the initiatives described in this review rose in large part because of improved nutrition and animal performance. Rainfall is strongly seasonal in the Amazon, and in the dry season grass production is greatly reduced. Without supplementary feeding or active pasture management, cattle gain weight in the wet season, only to lose much of it in the dry season because of nutritional deficiencies (Silva et al., 2009). As good welfare requires that nutritional and health needs are met (Mellor and Stafford, 2001), addressing these nutritional deficiencies through improved pasture management delivers coupled welfare and productivity gains.

Not all management changes have the same welfare consequences, however, and improved nutrition is not sufficient for good welfare. Cattle in agroforestry systems show more cohesive social behaviour and benefit from reduced heat stress as well as improved nutrition (Broom et al., 2013). For feedlots, the picture is however, more mixed. Feedlots in Brazil are becoming more common - Mato Grosso's feedlot capacity grew 48% from 2009-2016 (IMEA, 2016) – and while feedlots provide high-energy nutrition that maximize animal growth rates, careful management is required to ensure adequate welfare. In feedlot systems heat stress, mud, and welfare during dehorning, castration, and branding are key concerns (Grandin, 2016), which can be mitigated through training in good agricultural practices, e.g. training staff to provide analgesia prior to dehorning (Stock et al., 2013). In all systems, welfare continues beyond the farm-gate, with welfare in transport and slaughter also critical. While it is therefore encouraging that Embrapa's good agricultural practices and the Brazilian Roundtable for Sustainable Beef (GTPS) include detailed recommendations on cattle management and welfare both on and off farm (GTPS, 2016; Valle, 2006), animal welfare remains an evolving science. As the study of animal welfare increasingly looks beyond the traditional "five freedoms" – freedom from i) thirst, hunger and malnutrition; ii) discomfort and exposure; iii) pain, injury, and disease; iv) fear and distress and v) the freedom to express normal behaviour – to look at new measures of welfare, such as "a life worth living" (Mellor, 2016), cattle production systems must ensure that increases in productivity do not come at the expense of welfare in order to remain acceptable to society today and in future (Broom, 2010).

Cattle intensification also faces concerns of increased nutrient run off and water pollution (Latawiec et al., 2014). This challenge is greatest for feedlot systems, which produce large volumes of waste in a concentrated area. Most Brazilian production, like the initiatives described in this review is, however, pasture-based (Strassburg et al., 2014), where urine and manure are deposited directly onto pasture, rather than stored before disposal. In pasture-based systems, the effect of this diffuse nutrient pollution can be mitigated by restricting the access of cattle to streams – riparian areas are in any case protected under the Brazilian forest code, which requires that landowners reforest 5-100m either side of streams (Soares-Filho et al., 2014). The do Campo à Mesa, Novo Campo, and Silvopastoral system initiatives therefore all explicitly require fencing off degraded riparian areas and the installation of pumps to provide cattle with alternative water sources in pasture areas.

This review and many sustainable cattle ranching initiatives have a stronger focus on agronomic changes than social impacts of intensification (Alice Ferris et al., 2016), though this does not mean that these initiatives do not consider labour conditions. The do Campo à Mesa and Novo Campo initiatives, for example, both focus on the implementation of Embrapa's good agricultural practices (GAP) which includes consideration of the farmer's social responsibilities and the social function of farming businesses (Valle, 2006). Other initiatives also have a strong focus on working conditions, as seen in the Pecuária Verde program in Paragominas (SPRP, 2014). There, workers reported 15% higher wages and higher work satisfaction than on neighbouring farms (da Silva and Barreto, 2014). Though cattle productivity gains are often delivered through training of farmer workers and increases in demand for on-farm labour, the implications of different methods of intensification for labour-markets and rural communities remains understudied.

Additional figures and tables:

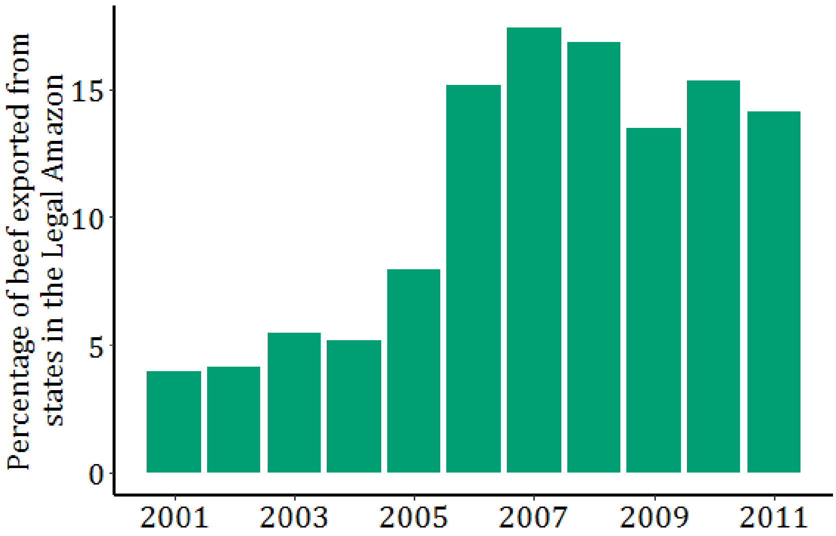

Figure 9.20 - Exports of beef from the Brazilian Legal Amazon have increased since the early 2000s.

Data from: (TRASE, 2017).

Table 9-16 – Grass species successfully planted in Acre in mixed pastures with the legumes Tropical kudzu Pueraria phaseoloides and Forage peanut Arachis pintoi.

Legume	Complementary grass species
Tropical kudzu Pueraria phaseoloides	Brachiaria brizantha cultivares (cv.) Marandu, Xaraés
	Brachiaria humidicola cv. comum
	Brachiaria decunbens cv. *Basilisk*
	Panicum maximum cv. Tanzânia, Mombaça
Forage peanut *Arachis pintoi*	Brachiaria brizantha cv. Marandu, Xaraés
	Brachiaria humidicola cv. comum
	Brachiaria decunbens cv. *Basilisk*
	Panicum maximum cv. Tanzânia, Mombaça
	Cynodon nlemfuensis cv. Lua
	Brachiaria arrecta x Brachiaria mutica cv. Laguna

Table 9-17 – Slaughter ages and weights achieved on intensified farms. No data were provided from the Florestas de Valor initiative.

Name of initiative	Lead organization	Age at slaughter (months)	Weight at slaughter? (1 liveweight @ = 30kg)
Novo Campo Program	ICV	Steers: 24 (20-40) Cows: 20 (18-36)	Steers: 21 (18-23) CowsL 13.5 (12-17)
Silvopastoral System with Rotational Grazing for Beef	Idesam	24 (22-34)	15 (14-20)
Intensification of beef cattle production systems with the use of mixed grass-legume pastures in Acre	Embrapa	Nelore: 36 (30-42) Crossbreed Nelore x Aberdeen Angus 27 (24-30)	17 (16-20)
Do Campo à Mesa	TNC	~28	16-18

Table 9-18 – Example breakdown of costs in each initiative.

Name of initiative	Breakdown of typical inputs and costs of intensification					
	Pasture liming		Fertilizers		Other	
	Quantity (kg/ha)	Cost (R$/ha)	Quantity (Kg/ha)	Cost (R$/ha)	Examples:	Cost (R$/ha)
Novo Campo Program	1500	350	400	850	Wire, wood (for fencing), machine rental, seeds, plumbing and operational costs.	1800
Silvopastoral System with Rotational Grazing for Beef	2000-2500	300-750	120-150	360-600	Machine rental for ploughing and application of inputs (e.g. fertilizer), electric fencing, Infrastructure (water pump and in-pasture drinkers), planting of leguminous trees.	4600
Intensification of beef cattle production systems with the use of mixed grass-legume pastures in Acre	<600 kg/ha.	180	300	600	Herbicides, machine rental for ploughing and planting of legumes.	450
Do Campo à Mesa	1500	345	-	-	Seeds, fencing, machine rental for pasture restoration.	1783
Florestas de Valor	1000	367	500	500	Wire, wood (for fencing), insulation for electric fence, grass seed, electrified appliance, solar panel, drinking fountains, machine rental for pasture restoration and maize planting.	1650
Silvopastoral System with Rotational Grazing for Dairy	2000-2500	300-750	120-200	360-600	Machine rental for ploughing, application of inputs, electric fencing, installation of water system, planting of leguminous tress (seeds and seedlings).	4500

Table 9-19 – Seven other sustainable cattle ranching initiatives in the Amazon biome.

Name of initiative	Number of farms/farm area	Region	Reference
Pecuária Sustentável na Prática	4,547 ha	Rondônia	(GTPS, 2017)
Projeto Balde Cheio	41 farms in Rondônia, unknown number of farms in Pará and Amazonas	Rondônia, Pará and Amazonas	(Novo, pers. Comm)
Intensificação na Produção e Proteção a pequenos proprietários e reservas indígenas na Amazônia	4,000 ha	Novo Santo Antônio (Piauí)	(GTPS, 2017)
Piloto de Pecuária Sustentável no Vale do Araguaia	140,000 ha	Vale do Araguaia (Mato Grosso)	(GTPS, 2017)
Sustainable Agriculture Network	3 farm units	Juruena (Mato Grosso)	(Newton et al., 2015)
Terracerta	**2,323,583 ha**	**Redençao, Paragominas (Pará)**	**(GTPS, 2017)**
Pecuária Verde	5,207 ha on 6 farms	Paragominas (Parà)	(SPRP, 2014; D. Silva pers. Comm)

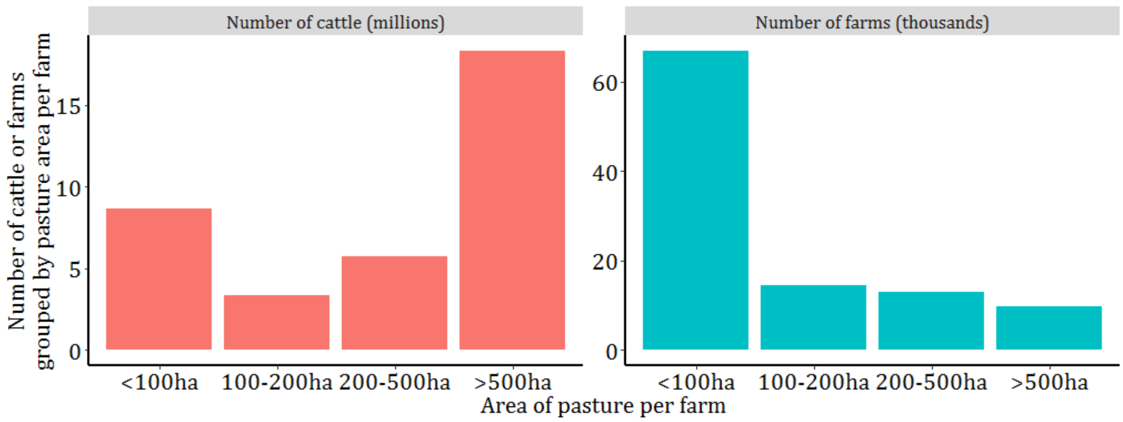

Figure 9.21 – Farm size distribution in the Amazon. While half of all cattle (51%) are found on farms of with pasture areas > 500ha (left), these make up only 9.4% of properties (right). Most cattle ranches (78%

of properties, rearing 33% of cattle) have pasture areas less than 200 hectares - a size below which some pasture intensification technologies may not be financially viable. Farm size data from: (IBGE, 2006). These data do not include farms with fewer than 50 cattle head, and so probably underestimate the number of small farms.

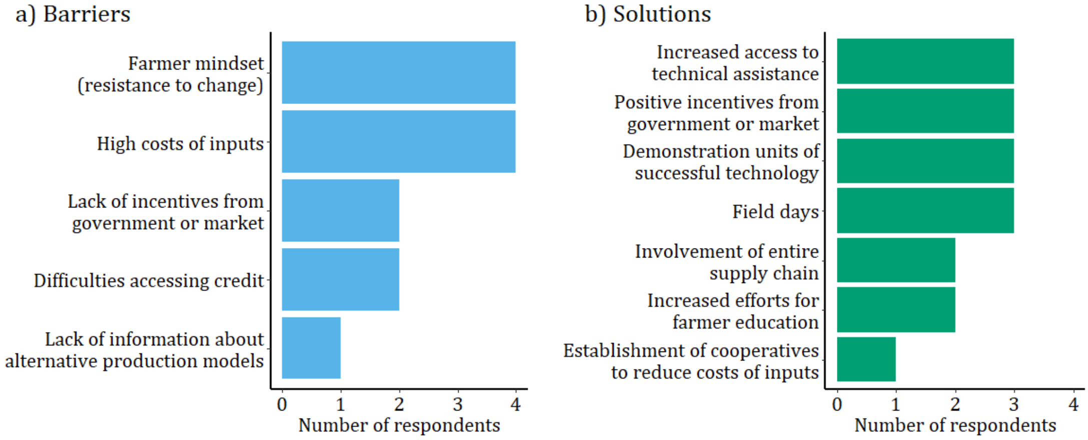

Figure 9.22 – a) Barriers to adoption of sustainable cattle initiatives, and b) solutions to these barriers, as perceived by staff at the six surveyed sustainable cattle ranching initiatives. Respondents from each initiative (6) were asked a) to describe the principal barriers to the implementation of the initiative, and b) what the solutions are for overcoming these barriers.

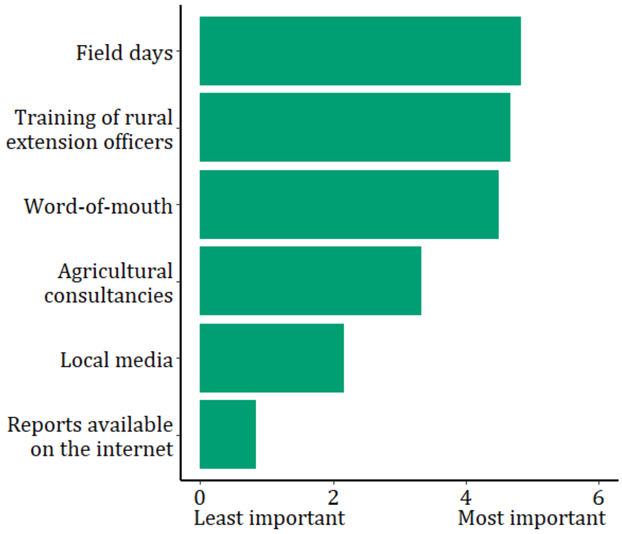

Figure 9.23 – Mean ranking (1=least important, 6 = most important) of different channels for diffusion of high-yielding technologies from cattle ranching initiatives, as suggested by respondents from the six intensification initiatives in this chapter. Two respondents also suggested field materials (e.g. training manuals/leaflets) and online social networks as important for the spread of new technologies (ranked as 4 and 1, respectively).

APPENDIX R – CHAPTER 6, ADDITIONAL DISCUSSION.

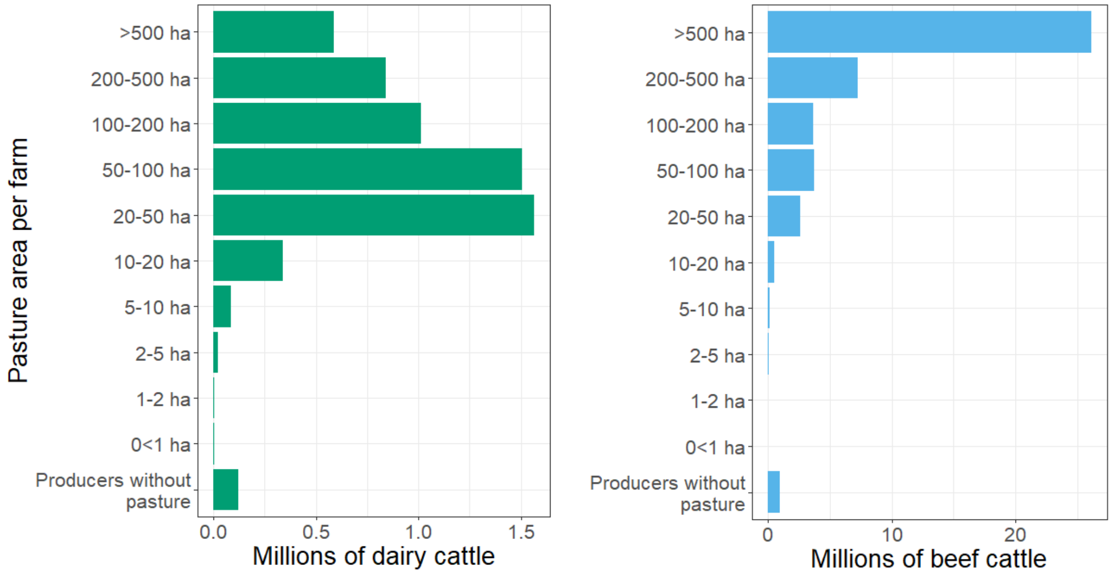

Figure 9.24 – Dairy and beef farm size distributions. Dairy farming in the Brazilian Legal Amazon (left) is dominated by smallholder production, with most cattle kept on farms with less than 500 hectares of pasture. Most beef production (right), in contrast, occurs on larger farms with more than 500 hectares of pasture. Note that the x-axes differ between the two plots. Data from (IBGE, 2006).

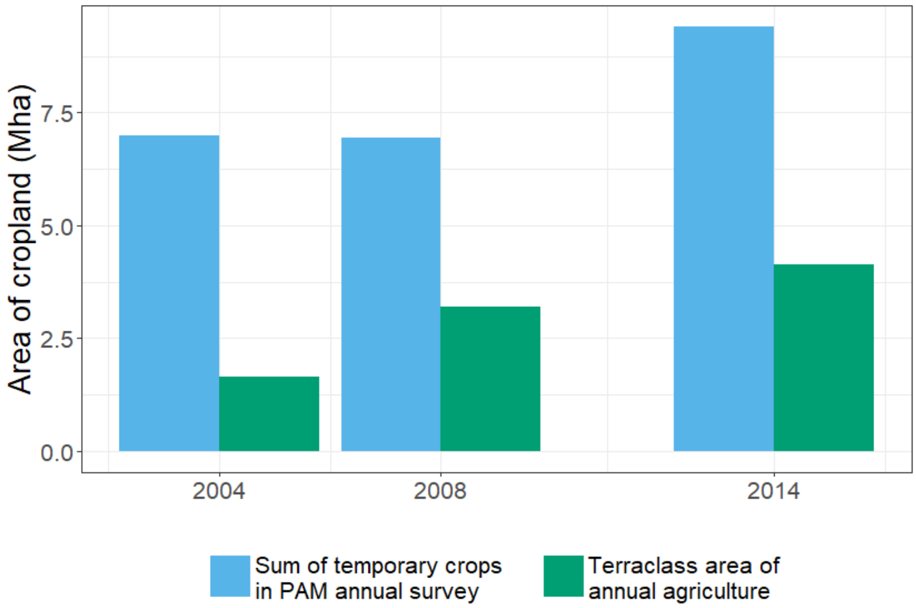

Figure 9.25 - Comparison of survey estimates of cropland area from the Pesquisa Agricola Municipal (PAM), and annual agriculture as reported in the Terraclass satellite-imaging dataset. The PAM figures are corrected for the area of maize double cropping.

Table 9-20 – Non-significant nearest neighbour matching estimates of the effect of inclusion on the priority list on the yield (kg/ha) of the six dominant crops in the Amazon.

Yield (kg/ha)	Nearest neighbor matching estimates
Soy	-0.242
	(-0.16)
Maize	-0.691
	(-0.84)
Sugar	0.626
	(0.15)
Cotton	-0.547
	(-1.11)
Rice	0.167
	(0.57)
Cassava	0.0118
	(0.04)

t statistics in parentheses

* $p<0.10$, ** $p<0.05$, *** $p<0.01$

Propensity score specification algorithm and parameter estimates:

Here we describe the data-driven, stepwise procedure for the specification of the propensity score proposed in Imbens and Rubin (2015).

Consider a covariate vector X that consists of K components. Our specification problem is to choose among a potential set of K. This is done in two stages. In the first stage, K_B basic covariates are selected to be included in the propensity score that are a priori expected to be substantially correlated with the outcome of interest or the assignment process. In the second step, we decide which of the remaining $K - K_B$ covariates are additionally included in the propensity score function. The decision is based on a likelihood ratio test statistic for the null hypothesis that an additional covariate has a zero coefficient. More specifically, each remaining covariate is separately added to the basic specification with K_B covariates and its likelihood ratio statistic is calculated. If at least one of the likelihood ratio test statistic is greater than some pre-determined threshold C_L, we add the covariate with the largest likelihood ratio statistic. We then check whether any of the now remaining covariates should be included based on the same procedure. We continue this process until none of the remaining likelihood ratio statistics exceeds C_L. At the end of this stage, we have a total of K_L covariates (including the pre-selected covariates K_B) entering linearly in the propensity score.

We follow Imbens and Rubin (2015) and use a threshold value for the likelihood ratio statistic of $C_L = 1$, which correspond implicitly to a z-statistics of 1. Total deforested area in 2007 and deforestation in 2005, 2006 and 2007 for automatic inclusion in the propensity score, based on our knowledge of the priority list inclusion criteria. In addition to this pre-selected covariate (i.e. K_B), the algorithm leads to the inclusion of four further covariates. Thus, we have $K_L = 8$ selected covariates. Given this vector of nine terms (including intercept), we estimate the propensity score based on a logistic regression model by maximum likelihood. Table A1 reports the parameter estimates with the variables in the order in which they were selected for inclusion in the specification of the propensity score.

Table 7 – Propensity score estimation.

	Est.	t-stat
Constant	-134.95	-2.54
Total deforested area in 2007	12.33	2.31
Deforestation 2005	12.87	2.53
Deforestation 2006	0.59	0.45
Deforestation 2007	1.21	0.49
Accessibility (average distance to center)	-6.22	-2.01
Share of settlement projects	-8.86	-1.15
Share strictly protected reserves	-36.48	-1.35
Forest area	18.77	1.31
N	476	
Pseudo R^2	0.94	