Closing loops to rebalance the global carbon cycle: Biomass flows modelling of global agricultural carbon fluxes

Submitted by Thomas William Robert Powell, to the University of Exeter as a thesis for the degree of Doctor of Philosophy in Geography

In July 2015

I certify that all material in this thesis which is not my own work has been identified, and that no material has previously been submitted and approved for the award of a degree by this or any other University.

Signature: …….

“We are on the flight deck [of ‘Spaceship Earth’], and we are alone. We are at the controls, and we have no option but to use them. And we know where we want to go. The fact that we have only a dim idea of how to fly means we must act carefully and thoughtfully, not that we must not act.”

(Morton, 2009, pp. 292–3)

Abstract

Since the beginning of farming, and even before, humans have been actively modifying our environment in order to harvest biomass. With the 'Great Acceleration' of the industrial age, the global system of biomass harvest for food production has become a major driver of Earth system processes, and caused multi-dimensional sustainability issues which must be addressed in order to meet continued increases in demand for food and other biomass. In addition, bioenergy generation, with the subsequent storage of some or all of the carbon content of the feedstock (known as bioenergy with carbon storage or BECS), is now seen as an important tool for rebalancing the carbon cycle. This thesis has used a biomass flows modelling approach to examine possible trajectories for the socio-ecological metabolism of humanity, with a focus on fluxes of carbon contained in biomass. This approach connects social and economic drivers of biomass harvest with physical Earth systems processes such as the global carbon cycle. Meeting growing food demand in the years 2000-2050 is likely to be a significant challenge in its own right, necessitating the harvest of over 30% of terrestrial biomass. This can only be done without significant damage to natural ecosystems if large increases in efficiency and intensity of food production are achieved, or diets are altered. The production of livestock products is shown to be a major cause of inefficiency in biomass harvest, and changes to livestock demand or production are particularly powerful in ensuring a less damaging relationship with Earth system processes. If increases in efficiency are achieved, it may be possible to grow dedicated bioenergy crops, which, combined with the biomass available in waste and residue streams can be used to generate significant carbon dioxide removal (CDR) fluxes via BECS. Following this strategy it is possible to have a non-trivial effect on atmospheric CO_2 concentration by 2050. Increasing the intensity of biomass harvest, particularly when low intensity pasture is replaced with intense bioenergy cropping, also has significant implications for ecological energy flows, and the potential trade-off between protecting biodiversity and growing bioenergy crops to mitigate climate change is also discussed. This body of work presents several interesting areas of potential conflict in different drivers of biomass harvest, and suggestions are made for ways in which to develop the approach in order to explore them.

Acknowledgements

First, my sincere thanks to my supervisor, Tim Lenton, who has been immensely patient with me, providing reassurance, guidance, inspiration and encouragement in equal measure, while giving me a huge amount of space and freedom in which to explore ideas.

I am also extremely grateful for the support, encouragement and interesting discussion from friends and colleagues in the Earth System Science and Exeter Climate Systems group. My particular thanks to Steve and Chris for their forbearance with my frequent emergency requests for aid with mathematical and coding problems, of which I am far more ignorant than I should be. Thanks also to Luke, Claire, Siobhán, Alastair, Jim, Ben, Sarah, Nick and everybody else for both their distractions and their support.

Thanks also to those who made my year in Norwich such an enjoyable and stimulating one: Oli, Henry, Jeppe, Chan, Jaap, Delphine, and Martin provided the perfect community in which to move on from undergraduate life. I will remember trips to the broads with Professor Powly the punt with particular fondness.

I thank the University of Exeter for picking up my studentship and facilitating my move from UEA, and also Tim Quine, Tim Lenton and Hywel Williams for providing employment to keep the wolf from the door during my rather extended writing up phase. Their patience in allowing me the time to continue writing while engaged in other projects has been extremely kind.

My time thus far in Exeter has been an extremely happy one, and for that I owe immeasurable thanks to my housemates Hertha, Ian, Chris, Emilie and Rozzy (and Ferdia and Andy). We've achieved some amazing things together and will continue to do so, and they have been very tolerant of my absence and strange hours during the last couple of months of writing.

To my family I owe everything, especially my parents who have been unfailingly generous and supportive and without whom I wouldn't be who I am or where I am today. Special thanks also to my grandparents, Jean and Dick Grove, who created such a perfect atmosphere of warmth and openness as part of their research lives that I couldn't imagine wanting to be anything other than an academic. I also thank Grandpa for showing me my first article on Gaia at the age of 15, without which I would not have chosen to study Ecology.

Finally, Rozzy has provided more support than she knows in the last five years. For lending me confidence when mine ran out, for showing me what hard work really means, for our many shared adventures and for her love and tolerance I cannot thank her enough.

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Units, conversions and abbreviations

Units and unit conversions

A lack of standardization of units in the literature has caused me a great deal of frustration over the last few years, and on occasion I have found myself adding and removing strings of zeros over a series of calculations only to arrive back at the same number as I began with. With the aim of helping the reader I have tried to use units consistently throughout this thesis, and have expressed masses of carbon in petagrams rather than gigatonnes. The following is a brief list of the main units encountered in the following chapters:

t = tonne (metric). 1 t = 1,000 kg = 1,000,000 **g**

Mt = Megatonne. 1 Mt = 1,000,000 t = 10^{12} g

Pg = Petagram. 1 Pg = 1 billion t ($1x10^9$) = 10^{15} g

Gt = Gigatonne. 1 Gt = 1 Pg.

Kcal = Kilocalorie. 1Kcal = 1,000 calories = 4184 J.

J = Joule.

MJ = Megajoule. 1 MJ = 1,000,000 J

EJ = Exajoule. 1 EJ = 10^{18} J

ha = hectare. 1 ha = 10,000 m^2.

Mha = Million hectares. 1 Mha = 1,000,000 ha = 10,000 km^2.

Gha = Gigahectare. 1 Gha = 1,000,000,000 (10^9) ha = 10,000,000 km^2.

Abbreviations

BECS: Bioenergy with carbon storage (a broad term than includes BECCS, but also pyrolysis etc.).

BECCS: Bioenergy with carbon *capture* and storage, i.e. active capture of carbon from flue gasses after combustion.

CDR: Carbon dioxide removal.

NPP: Net primary productivity.

HANPP: Human appropriation of net primary productivity. For abbreviations of the constituent parts of HANPP, see Chapter 1 (p11).

LUC: Land-use change.

Chapter 1: Introduction

The nature and scale of the dilemma facing contemporary human societies is, for me, succinctly captured by the quotation on the opening page of this thesis; humanity has attained such an influence over the ‘controls’ of the Earth system that we must learn how they function, and teach ourselves to manipulate them responsibly. There is thus a pressing need to develop a broad and integrated understanding of the interactions between human activities and Earth system processes.

Our ascendance to such a major role in planetary scale processes has been quite rapid in terms of the evolution of the Earth system; but in human terms it has a long history, arguably with its origin in the Neolithic (Foley et al., 2013; Lewis and Maslin, 2015). Beginning around 11,500 years ago in Mesopotamia, China, Central America and several other sites across the world, the domestication of plants, particularly the grasses wheat, rice and maize, and of animals revolutionized human societies and simultaneously their relationship with the ecosystems in which they lived.

Agriculture, by definition, is the active alteration and management of ecosystems in order to harvest plant and animal products. As farming spread across the Neolithic world forests were cleared for crops and pasture, and with inventions such as irrigation, the use of draught animals, the plough and fertilization using manure or legumes humans began to alter soil and hydrological processes. It is also possible that the first tangible anthropogenic effects on the composition of the atmosphere occurred during this period; either through carbon dioxide emissions from deforestation (Kaplan et al., 2011; Ruddiman, 2013), or methane emissions from the expansion of rice production and the growing herds of domesticated ruminants, especially cattle (Ruddiman, 2013). These lines of evidence are somewhat controversial (Lewis and Maslin, 2015; Stocker et al., 2011), but may represent the emergence of humans as a species with global influence.

During the same period, the growing populations of early agrarian societies also began to experience degradation of their local environments caused by their farming practices, such as soil degradation and erosion, salination, and disruption of freshwater supplies (Lenton and Watson, 2011). Despite this, further innovations including using crop rotations to recycle nutrients and the use of horses as draught animals led to continued increases in production, hand

in hand with sustained population growth (Lenton and Watson, 2011). By the beginning of the 17th century, with a global population of around half a billion, humans leave an undeniable mark on global ecology, with European settlers in the Americas redistributing species across the globe at an unprecedented scale. Around this time maize, potatoes and several other staples of the modern diet were introduced to Europe, Africa and Asia from the Americas, while wheat, sugarcane, bananas and cattle were among the species exported in the opposite direction (Lewis and Maslin, 2015). At the same time, the violence and disease epidemics brought by the Europeans reduced the population of the Americas by about 90%. The resulting regrowth of forest on around 50 million hectares of abandoned farmland stored in the region of 5-40 Pg C in the space of a century, and has been implicated as a cause of the little ice age (Nevle and Bird, 2008).

The expansion of global trade, growing populations and, tragically, the export of slaves to the new world drove continued agricultural expansion into the 18th and 19th centuries. This, in turn, drove a positive feedback on population as the accumulating wealth from trade (based heavily on biomass products including cotton, sugar, spices and tea) drove increased investment in the growth and transport of agricultural products. This feedback is common throughout history, with improvements in agricultural productivity generally made by increasing energy inputs (e.g. the use of draught animals, building infrastructure), by improving recycling of nutrients (e.g. manure and crop rotations) or by efficiency increases (e.g. enclosures of common land in the UK) (Lenton and Watson, 2011). Rapidly growing populations in Europe, however, accompanied by food shortages and wars, prompted an awareness of the potential for exponential growth in human population and the difficulty in maintaining an equivalent growth in food supply, with Thomas Malthus publishing his ‘Essay on the Principle of Population’ in 1798.

For much of the 19th and 20th centuries, Malthus was apparently proven wrong in his fears, as the recognition of the vast energy reserves stored in fossil fuels and the invention of an efficient steam engine drove the industrial revolution. The subsequent enormous extra energy input into agriculture led to huge increases in production; first through mechanisation, then the use of fossil fuel energy to fix nitrogen from the atmosphere and mine phosphate rock, to

conduct intensive crop breeding programs and drive other important developments including refrigeration and the development of synthetic pesticides to reduce competition from other organisms (Figure 1.1). These innovations and the effort to spread them across the Western world from the 1930s-1960s, and subsequently to India, China and other developing countries, are collectively described as the 'green revolution'; a transition probably as significant as the industrial revolution itself. Of the 6 billion people alive at the turn of the millennium, 60-70% were fed by the extra production attributed to fossil fuels (Evans, 1998). In meeting that demand for food human agriculture covered 40% of the Earth's productive land surface (Ramankutty et al., 2008), and harvested around a quarter of its net primary productivity (Haberl et al., 2007).

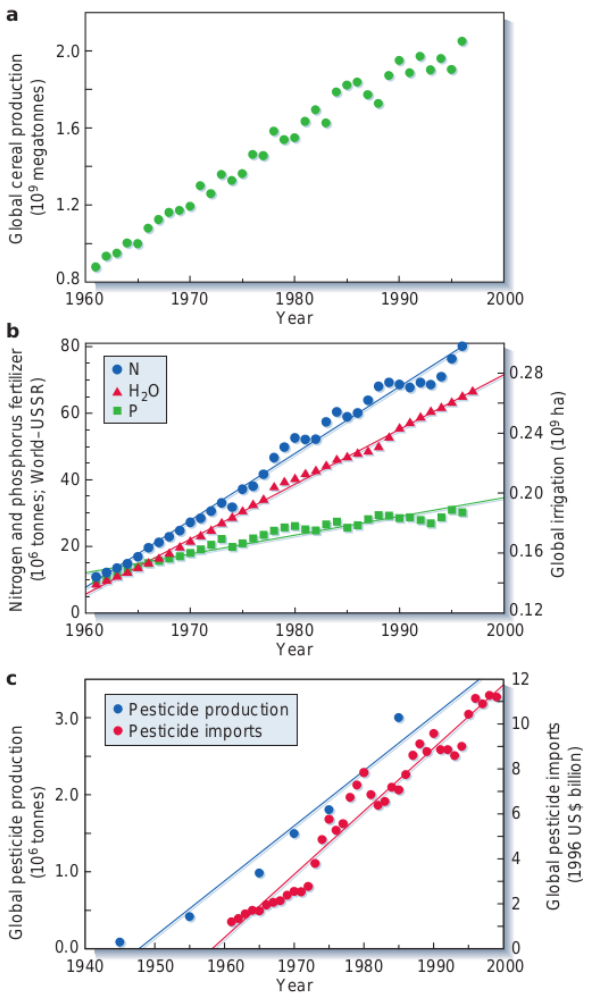

Figure 1.1: Global food production and inputs since the green revolution (reproduced from Tilman et al. (2002). These trends clearly show the critical role of increased nutrient and other chemical inputs in increasing food production in the latter half of the 20th century.

This exponential (or sometimes greater than exponential) growth in population and resource use has been the driving force behind the 'great acceleration' of the second half of the 20th century (Steffen et al., 2015), and has prompted new Malthusian concerns. With very high energy use now integral to human societies, but fossil fuel reserves becoming depleted, new energy sources are needed. In addition, the initial high rates of increase in crop yields and other measures of agricultural production, which were sustained through the 1970s and 80s by exporting industrial farming techniques to the rapidly growing populations of India and China, are now tailing off. Although population growth is slowing, the global population is expected to reach around 9.5 billion by 2050 and pass 10 billion by the end of the century; in combination with increasing wealth this is expected to drive almost a doubling in global food demand between 2000 and 2050, but stagnating crop yields make meeting that demand a difficult prospect. Populations interact with their environments not only by extracting resources, but also by releasing wastes, and in this respect, too, humans have grown to have a truly global influence; including through the emission of carbon dioxide, other greenhouse gasses, and CFCs, a doubling of the global flux of reactive nitrogen, and the release of plastics into the ocean.

Although the harvest of biomass is no longer the largest single primary energy supply for humanity, with food and biomass fuels providing around 50 exajoules (1EJ = 10^{18} J) of a total primary energy demand of around 500 EJ yr^{-1} (Smeets et al., 2007), it remains perhaps our most fundamental connection with our environment. Simply put, no human can survive without ingesting the energy fixed by plants in photosynthesis. In fact because of its enormous spatial extent and multidimensional interactions with land-surface processes (Foley et al., 2005), the management and harvest of biomass is arguably still the human activity with the largest impact on Earth system stability. Of nine defined 'planetary boundaries for a safe operating space for humanity' (Rockström et al., 2009b) , agriculture is a main driver of seven:

1) *Climate change*: Although a relatively low user of fossil fuels in comparison with other sectors (e.g. industry, transport), the carbon dioxide emissions associated with agricultural machinery, and with producing nitrogen fertilizers using the energy-hungry Haber-Bosch process, are non-trivial. Far larger, at around 1 Pg C yr^{-1} (Friedlingstein et

al., 2010), are CO_2 emissions from land-use change, which, in combination with the aforementioned fossil fuel emissions, methane (CH_4) emissions from livestock and rice production, and nitrous oxide (N_2O) emissions from fertilized soils constitute around a third of all anthropogenic greenhouse gas emissions (Metz, 2007).

2) *Stratospheric ozone depletion:* Since the regulation of halocarbons under the Montreal protocol, agricultural N_2O emissions are now the largest anthropogenic emission of ozone depleting compounds to the atmosphere, and are expected to remain as such for the rest of the century (Ravishankara et al., 2009; Portmann et al., 2012).

3) *Biogeochemical flow boundary (N and P cycles):* Agricultural fertilizer use has caused enormous disruption to nitrogen and phosphorous cycling across the globe, with a doubling of the amount of reactive N in the biosphere in the last century. Whether this is a truly global boundary is debated, but the scale of the anthropogenic influence is unequivocal (Rockström et al., 2009a).

4) *Global freshwater use:* Agricultural water use grew three-fold in the last 50 years, and represents 70% of global freshwater extraction. Water scarcity is already an acute problem in some areas, and is likely to increase in importance (McIntyre, 2009).

5) *Change in land-use:* Agriculture has been the main driver of land-use change throughout human history, and has resulted in enormous biodiversity loss as well as releasing over 150 Pg C to the atmosphere between 1850 and 2005 (Houghton, 2008). Cropland and pasture cover 40% of the Earth's productive land-surface (Ramankutty et al., 2008).

6) *Biodiversity loss:* Land-use change driven by agriculture is a key driver of habitat loss, especially in the tropics where much recent land-use change, along with two thirds of the world's species, are concentrated (Pimm and Raven, 2000). On land that is already used for agriculture, management practices such as intensive monoculture cropping and widespread use of agrochemicals also have a large impact on biodiversity (Haberl et al., 2004; Kleijn et al., 2009; Krupke et al., 2012).

7) *Chemical pollution:* Agricultural chemicals, including pesticides and herbicides, alongside pharmaceuticals and hormonal treatments given to livestock, are huge sources of chemical pollution. Agrochemicals, often complex organic molecules easily taken up by the cells of non-target organisms, have been implicated in disrupting pollinator networks (Krupke et al., 2012), endocrine functioning in aquatic species (Jobling and Tyler, 2003) and multiple other adverse effects.

The remaining two boundaries (*ocean acidification* and *atmospheric aerosol loading*) are also influenced by agriculture, though perhaps not to such a great extent. If we are at the controls of 'Spaceship Earth', then agriculture is one of the key means by which we are grasping them.

Faced with the prospect of meeting the demands of a growing population, as well as coping with these perturbations to Earth system stability, it seems clear that an enormous challenge lies ahead. We must continue to increase food production, but it is also vital that natural systems retain the capacity to provide crucial services such as maintaining freshwater supplies, a stable climate, biological pest control, and air quality. In fact, to return to Oliver Morton's 'Spaceship Earth' analogy, having reached the flight deck and found ourselves at the controls, our most pressing need is to work out how to keep the autopilot functioning. Whilst meeting food demand, mitigating the effects of climate change and other such targets are crucial goals in their own right, they must be seen as part of a multidimensional effort to integrate more sustainably with Earth system processes. Since the land-surface is where much of the interaction between human societies and ecological or Earth system processes takes place, it would be beneficial to see farmland as the interface between the human and natural components of a coupled system (Liu et al., 2007), with farmers and agricultural policy-makers as the stewards of those interactions.

With the certainty that humanity will demand a growing harvest of biomass for the remainder of this century, sustainable intensification is a key goal of contemporary agricultural science (Foley et al., 2005; Tilman et al., 2011; Foley et al., 2011). A key implication here is that the intensification of production in the 'green revolution' was *not* sustainable, and in fact a key element of sustainable intensification must be to consolidate the productivity increases of industrial farming while reducing its negative impacts. The technological advances of

‘precision agriculture’ and genetic modification, along with less hi-tech management strategies such as no-till farming, reducing nutrient over-use and new crop rotations are often invoked as having the potential to address this issue (West and Marland, 2002; Tilman et al., 2011; Mueller et al., 2014; Oldroyd and Dixon, 2014). In order to meet the increase in demand for food production above the current levels, an important strategy is closing the ‘yield gap’ between potential and achieved yields, mostly in developing countries and mostly through improved water and nutrient management (Mueller et al., 2012). Other possible strategies include better recycling of nutrients, from human waste for example, and increasing the efficiency with which the calorific content of crops is delivered to consumers, particularly by shifting diets away from livestock products (Cassidy et al., 2013).

Improving the recycling of wastes and increasing the efficiency of resource use has been a key component of every major event in the evolution of the Earth system, with organisms evolving to recognize the potentially harmful wastes produced by other biological innovations as resource streams in their own right (Lenton and Watson, 2011). Indeed, before the industrial age of agriculture, which has focussed on increasing production by increasing energy inputs, these same factors were crucial to increases in agricultural productivity. Although much recycling is currently integral to even the most industrial farming systems, efficiencies of scale and the historic reliance on the low cost of fossil fuel energy and its products suggest that there may be considerable scope for increasing the output of the biomass harvest system without increasing inputs. The ‘closing of loops’ and utilization of waste streams as resources must therefore be a key part of a transition to a global agriculture that produces more, but which has a smaller footprint on natural systems. This has been one of the key themes of this work.

A key conceptual model throughout has been that of ‘socio-ecological metabolism’, which describes the relationships between human societies and their environments through the exchange of energy and materials (Fischer-Kowalski and Hüttler, 1998). This approach, coupled with the ‘methodological toolbox’ of material-flows analysis is extremely useful in establishing accounting structures for describing the physical interface between nature and culture, and also captures socially and economically driven processes like the intensification

of agriculture (Erb, 2012). In particular, I have focussed on the fluxes of carbon exchanged between human and natural systems. Carbon fluxes are a particularly powerful descriptor of the impacts of socio-economic metabolism because of carbon's dual role as the ecological currency of energy and as the thermostat of the Earth system via the biological and geological carbon cycles. The appropriation of biological carbon fluxes in the form of net primary productivity (NPP) is thus a good indicator of the ecological impacts of biomass harvest; while the release of waste carbon into the environment is an important and well understood driver of global climate change. The intensity and efficiency with which biomass carbon fluxes are harvested and processed is therefore a significant driver of both of these important impacts on the Earth system.

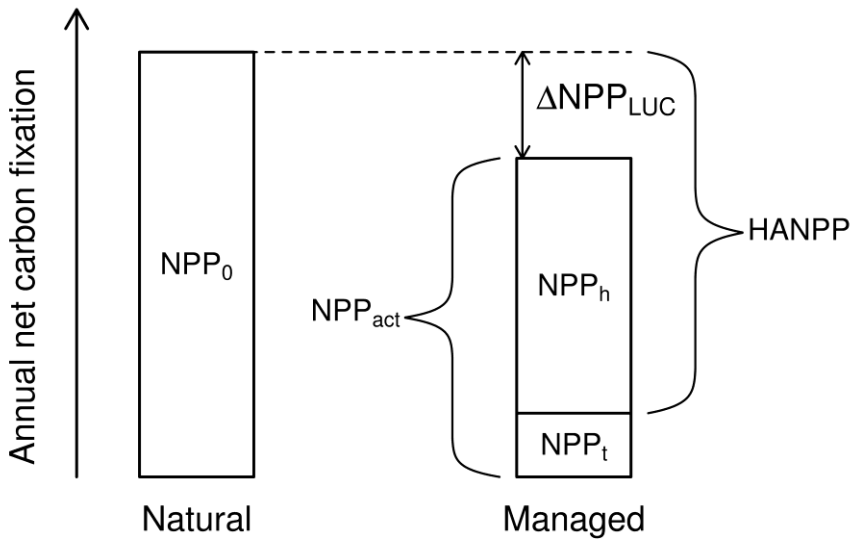

Figure 1.2: An illustration of human appropriation of net primary production (HANPP) (modified from Haberl et al. (2013)). NPP_0 is the natural potential NPP, in the absence of anthropogenic impacts. NPP_{act} refers to the NPP actually achieved under human management, with ΔNPP_{LUC} denoting the change in productivity associated with replacing a natural ecosystem with a managed one. NPP_{act} is divided into harvested biomass (NPP_h), and biomass remaining after harvest (NPP_t). NPP_h and ΔNPP_{LUC} together comprise the total anthropogenic biomass removal from the ecosystem, referred to as human appropriation of NPP (HANPP).

A key metric for the impact of harvesting biological carbon fluxes, which I have used extensively in subsequent chapters, is the human appropriation of net primary productivity (HANPP) (Haberl et al., 2007). HANPP attempts to capture the full range of impacts involved in the human management of biological carbon fixation, by dividing the natural potential NPP (NPP_0) into several

portions (Figure 1.2). The difference between the actual NPP of land under human management (NPP_{act}) and that of the original ecosystem (NPP_0), is referred to as ΔNPP_{LUC}. NPP_{act} is divided into that which is harvested for use by humans (NPP_h), and that which remains available to non-human elements of the ecosystem after harvest (NPP_t). Total HANPP comprises the sum of ΔNPP_{LUC} and NPP_h, and thus accounts for both the ecological impacts of converting natural ecosystems to managed land uses, and the impacts of biomass harvest itself.

The aims and structure of this thesis

The overall focus of the work presented in this thesis is on forecasting biomass harvest driven by population and dietary trends, with the aim of finding strategies to reduce the anthropogenic impacts on Earth system processes. In doing this I have concentrated on two major strategies: first, increasing the efficiency of food production through reducing wastes and changing diets, and; second, the potential for existing systems of human biomass harvest to fulfil a new objective alongside their present functions; the capture and long-term sequestration of CO_2 from the atmosphere. This is one of a suite of strategies referred to as carbon dioxide removal (CDR), and is likely to be a powerful tool in helping to rebalance the global carbon cycle (Lenton, 2010; Lenton and Vaughan, 2009; Royal Society, 2009; van Vuuren et al., 2011). The rational for this approach is described further in Chapter 2.

This project was prompted by initial assessments of the efficacy of various climate geoengineering options (Lenton and Vaughan 2009), and specifically the potential for land based biological CDR (Lenton, 2010). These studies concluded that capture and sequestration of biologically fixed carbon on a global scale is among the most potent available options for CDR, with the potential to fix around 10 Gt C yr^{-1} (Lenton, 2010), as well as being considered technologically feasible and relatively low cost. The principle pathway for biomass based CDR is the use of biomass as feedstock for energy generation, using processes which result in some or all of the carbon content of the original feedstock becoming suitable for long term storage. This is generally referred to as bioenergy with carbon storage (BECS), although a subcategory of systems which involve the combustion or fermentation of feedstocks and subsequent capture of CO_2 emissions are referred to as bioenergy with carbon *capture* and

storage (BECCS). Both acronyms are used throughout this work, though their similarity is frustrating. BECCS is considered integral to the IPCC's RCP 2.6 pathway (van Vuuren et al., 2011), as a viable option for large-scale energy generation with the potential for negative emissions.

In order to adopt a more nuanced approach, I constructed a simple, spreadsheet-based model of global anthropogenic biomass flows, which accounted for the biomass harvest required to meet food demand under changing population and dietary scenarios, and the associated biomass waste-streams. Also included was the land-use required to meet the required biomass harvest and the CO_2 emissions generated by land-use change. The model was then used to generate scenarios with varying diets and efficiency trends, which focussed on using biomass wastes, and bioenergy crops where possible, to generate CDR fluxes using BECS. This work was published in the journal Energy and Environmental Science as Powell and Lenton (2012), and is included here in its published form as Chapter 2, giving a comprehensive description of the groundwork and motivations with which this body of work began.

Following this the focus shifted slightly. Moving from concentrating on rebalancing the carbon cycle, I began to explore the potential conflicts or trade-offs involved in trying to reduce the footprint of global agriculture in multiple dimensions. Using an adapted version of the same model, I investigated the possible effects of the same CDR focussed scenarios on global biodiversity, using macro-ecological relationships associated with the extent and intensity of biomass harvest. This work, published in Environmental Research Letters (Powell and Lenton, 2013), indicated that while growing large areas of biomass crops for CDR could have a positive effect on global climate change, the same strategy could have a decidedly negative effect on biodiversity. The paper is included here as Chapter 3.

The work described in these two papers opened up interesting leads and pointed to some interesting potential trade-offs implicit in attempting to use agriculture to build a more sustainable relationship between humans and the Earth system. It also, however, highlighted some important inadequacies in the original biomass flows model, which made it difficult to take the analyses any further. Diets, in particular, were described by two simplistic 'black boxes',

representing plant products and animal products: The efficiency of production of these could be changed, but with no real relationship with the underlying trends in demand for particular products, or particular trends in livestock management. The second half of this thesis, therefore, describes a new model of global anthropogenic biomass harvest, with much greater disaggregation of important sectors including diet and livestock production, referred to as the **F**lux **A**ssessment of **L**inked **A**gricultural **F**ood production, **E**nergy potentials & **L**and-use change (FALAFEL) model. Chapter 4 describes the model inputs and structure, while Chapter 5 depicts a new set of scenarios which are compared with those in Chapter 2, and also the results of a full sensitivity analysis. The merits and failings of this model are discussed in Chapter 6, along with suggestions for its potential use in the further exploration of strategies to reduce the Earth system footprint of global agriculture.

Chapter 2: Carbon dioxide removal via biomass energy, constrained by agricultural efficiency and dietary trends

Abstract

We assess the quantitative potential for future land management to help rebalance the global carbon cycle by actively removing carbon dioxide (CO_2) from the atmosphere with simultaneous bio-energy offsets of CO_2 emissions, whilst meeting global food demand, preserving natural ecosystems and minimising CO_2 emissions from land-use change. Four alternative future scenarios are considered out to 2050 with different combinations of high or low technology food production and high or low meat diets. Natural ecosystems are protected except when additional land is necessary to fulfil the dietary demands of the global population. Dedicated bio-energy crops can only be grown on land that is already under management but is no longer needed for food production. We find that there is only room for dedicated bio-energy crops if there is a marked increase in the efficiency of food production (sustained annual yield growth of 1%, shifts towards more efficient animals like pigs and poultry, and increased recycling of wastes and residues). If there is also a return to lower meat diets, biomass energy with carbon storage (BECS) as CO_2 and biochar could remove up to 5.2 Pg C yr^{-1} in 2050 and lower atmospheric CO_2 in 2050 by 25 ppm. With the current trend to higher meat diets there is only room for limited expansion of bio-energy crops after 2035 and instead BECS must be based largely on biomass residues, removing up to 3.6 Pg C yr^{-1} in 2050 and lowering atmospheric CO_2 in 2050 by 13 ppm. A high-meat, low-efficiency future would be a catastrophe for natural ecosystems (and thus for the humans that depend on their services) with around 9.3 Gha under cultivation in 2050 and a net increase in atmospheric CO_2 in 2050 by 55 ppm due to land-use changes. We conclude that future improvements in agricultural efficiency, especially in the livestock sector, could make a decisive contribution to tackling climate change, but this would be maximised if the global trend towards more meat intensive diets can be reversed.

Declaration of authorship:

This chapter was originally published as:

Powell, T.W.R and Lenton, T.M. (2012) Future carbon dioxide removal *via* biomass energy constrained by agricultural efficiency and dietary trends. *Energy & Environmental Science*, **5**, 8116-8133.

DOI: 10.1039/c2ee21592f.

Biomass flows model and scenarios were constructed by TP, as was the bulk of the writing. TL contributed sections on pathways from biomass to carbon storage, and effects on atmospheric CO_2, alongside editing and comments.

2.1 Introduction

The impacts of humans on biogeochemical processes and natural ecosystems at the global scale are to a great extent defined by the appropriation of material and energetic resources, and the return to natural systems of high entropy 'wastes'. The scale of impact is thus driven by the magnitude of resource uptake, and the processing of resources within human systems, or 'social metabolism' (Fischer-Kowalski and Haberl, 2007). The industrial age has been defined by an enormous increase in social metabolism, such that primary energy use by humans has increased ten-fold over the last century to 510 EJ yr^{-1} (1EJ = $1x10^{18}$ J) (International Energy Agency, 2011; Smil, 2008). This has been fuelled largely by energy derived from fossil fuels, releasing C into atmospheric, terrestrial and marine pools, thus altering the balance of the matter and energy flows of the Earth system (Denman et al., 2011; Friedlingstein et al., 2010; Raupach and Canadell, 2010). Current concerns for the sustainability of human societies are due to our apparent proximity to physical constraints on social metabolism, in terms of the energy and resources available, and the impacts of wastes on the functioning and stability of Earth system processes.

The global carbon cycle is out of balance; inputs of CO_2 to the atmosphere from fossil fuel burning (~9.1 Pg C yr^{-1}) and land-use change (~1.1 Pg C yr^{-1}) have reached 10 Pg C yr^{-1} whilst natural 'sinks' only remove around 5 Pg C yr^{-1} from the atmosphere (Friedlingstein et al., 2010; Peters et al., 2012). That is why atmospheric CO_2 concentrations are rising, in turn contributing to climate change. The most obvious way to tackle this imbalance is to reduce CO_2 emissions, but they instead increased by about 25% over the past decade (Friedlingstein et al., 2010). This has provoked recent interest in methods of actively removing excess CO_2 from the atmosphere, then converting and storing the resulting carbon in long-lived forms; termed 'carbon dioxide removal' or CDR for short (Lenton and Vaughan 2009; Royal Society (Great Britain) 2009). Perhaps the most obvious route to CDR is to co-opt photosynthesis to remove CO_2 from the atmosphere for us (Lenton 2010). This has the advantage that when converting the resulting biomass into longer-lived forms of carbon, some energy can be derived, albeit with an energy penalty for the conversion and storage processes. Previous work has suggested that by mid-century a carbon

sink of around 5 Pg C yr^{-1} could be generated via land-based biological pathways of CDR (Lenton, 2010), which together with natural sinks and a reduction in CO_2 emissions could rebalance the carbon cycle and thus stabilise atmospheric CO_2. Such a "stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system" is the ultimate objective of the United Nations Framework Convention on Climate Change (UNFCCC, Article 2). This "*should be achieved within a time-frame sufficient to allow ecosystems to adapt naturally to climate change [and] to ensure that food production is not threatened...*" However, these underlying objectives of preserving natural ecosystems and food production are arguably more fundamental than solving the climate problem, which sets up a 'trilemma': We already co-opt plants to produce food for us on a global scale, and growing biomass for energy (with or without carbon capture and storage) can conflict with food production. Furthermore, replacing natural ecosystems with biomass plantations in order to help solve the climate problem and thus preserve natural ecosystems would be an ironic Catch 22. It is this 'trilemma' that we tackle quantitatively here.

Humans must harvest biomass for food and materials and the recent growth in global population, from 2 billion in 1927 to 7 billion in 2012, has seen a huge expansion of the management and appropriation of photosynthetic carbon capture. Human appropriation of net primary productivity (HANPP) now amounts to around a quarter of total NPP (Haberl et al., 2007; Krausmann et al., 2008; Smeets et al., 2007), occupying ~40% of the productive terrestrial surface of the globe (Foley et al., 2005) and contributing ~30-35% of anthropogenic greenhouse gas [GHG] emissions (Foley et al., 2011; Friedlingstein et al., 2010; Le Quéré et al., 2009; Smith, et al., 2007). As a consequence, the human manipulation of carbon fluxes through agriculture and forestry now constitutes a major perturbation of the global biogeochemical cycle of carbon and the interlinked cycles of nitrogen, phosphorous and water, with enormous impacts on ecosystem functioning and biodiversity (Foley et al., 2005; Haberl et al., 2007). Indeed land-use has direct and significant impacts on seven of the nine 'planetary boundaries' that have been suggested for the safe operation of humanity (Rockström et al., 2009b). The projected rise in human population to ~9.3 billion in 2050, and more importantly a projected 70% rise in global food demand, is likely to enlarge this footprint (FAO, Global Perspective Studies Unit,

2006). The strongest determinant of this future demand for food is a forecast increase in average per capita meat consumption from 16.6% to 18.8% of daily calorific intake (Smeets et al., 2007), because meat production is hugely inefficient with only ~3% of the feed energy consumed remaining in animal tissue (Wirsenius, 2003). An expected increase in per capita food demand from 2760 to 3302 kcal $cap^{-1}day^{-1}$ in 2050 adds to the challenge (Smeets et al., 2007). The increase in food demand could be met by a 76% expansion of vegetal crop production and a 110% increase in animal products, close to estimates given in the intergovernmental IAASTD report (McIntyre, 2009), the UK government Foresight report and other studies (FAO, Global Perspective Studies Unit, 2006; The Government Office for Science, 2011). This in turn will require growth in agricultural production of 1.42% yr^{-1}, higher than today's rate of ~1.25% yr^{-1}, necessitating the reversal of a 40 year downward trend in growth rate. This must come either from increasing yields, or through expansion of farmed land.

At the same time, diminishing fossil fuel reserves and a push towards 'carbon-neutral' energy sources are already driving increases in HANPP as feedstock for bio-energy generation. Biomass currently supplies about 10% of global primary energy (50-60 EJ yr^{-1} of a total supply of around 510 EJ yr^{-1}) (International Energy Agency, 2011), but about 80% of this is in the form of low-efficiency traditional biomass energy, overall demand for which is forecast (Chum et al., 2011; Smeets et al., 2007) to be approximately the same in 2050 as in 2000 (Chum et al., 2011; Smeets et al., 2007). Most projections of future energy supply and demand include a substantial (20-30%) contribution from modern bio-energy. Projections for human primary energy consumption in 2050 range from 600-1000 EJ yr^{-1}, the upper estimate of which is close to the current total above ground terrestrial NPP at around 1260 EJ yr^{-1}, indicating that for bio-energy to make a large contribution to future energy supply would require the appropriation of much of terrestrial biological carbon fixation. Previous estimates of the technical potential of bio-energy vary enormously, even reaching as high as 1,500 EJ yr^{-1} (Smeets et al., 2007). However, more realistic estimates are constrained by models of likely competition with food production, and in some cases with the preservation of biodiversity and carbon stocks in natural ecosystems (Beringer et al., 2011; Dornburg et al., 2010). These studies find a maximum potential of ~500 $EJyr^{-1}$ (often less), from a combination of

energy crops, primary residues from agriculture and forestry, and secondary and tertiary wastes from the processing and consumption of materials and food. While the bioenergy potential from waste streams is relatively certain (albeit subject to the influence of differing scenarios of food consumption and resource use), the calculation of the energy potential of feedstock crops is complex.

Opportunities to use primary productivity for carbon dioxide removal (CDR) come through both the management of NPP and natural carbon stocks towards net C fixation, and through the rebalancing and closing of loops in the biomass flows appropriated for food production and forestry. Here we focus on CDR methods of biomass energy with carbon storage (BECS), in particular biomass energy with CO_2 capture and storage (BECCS), and biochar production. BECCS refers to CO_2 capture and storage from gasification, combustion, or fermentation of biomass. Biochar production refers to pyrolysis of biomass to produce charcoal that is returned to the soil, potentially with gas as a co-product. The basic constraints on their CDR potential are the supply of feedstock, the efficiency of conversion to long-lived carbon, and the leakage rates back to atmospheric CO_2. Uncertainties around the quantitative potentials of these CDR methods are large, not least because many of their interactions with the natural carbon cycle are outside of human control and extremely heterogeneous according to local conditions (Lenton, 2010). They are therefore difficult to measure and to describe to a high degree of spatial resolution in models.

Here we examine the potential for future land-use to rebalance the carbon cycle, without compromising food production or natural ecosystems, by directing HANPP into carbon dioxide removal with some attendant bioenergy displacement of fossil fuel use, but some CO_2 emissions from land-use change. Our assumed 'hierarchy of needs' is food production, ahead of preserving natural ecosystems, ahead of CDR or mitigating CO_2 emissions, ahead of bio-energy generation. We prioritise the use of biomass to rebalance the carbon cycle ahead of maximising energy generation from it, because bio-energy is a very inefficient way of capturing solar energy (compared to solar photovoltaic or solar thermal power) (Pickard, 2010). We do not draw a value distinction between reducing CO_2 sources and creating CO_2 sinks, because the atmosphere cannot tell the difference. We focus instead on which strategies

maximise the contribution of HANPP to rebalancing the carbon cycle, within the broader constraints set. To examine the issue quantitatively, we generate simple scenarios of future land-use based on underlying scenarios of population, dietary preference, and efficiency gains in agricultural production out to 2050. We consider four alternative futures with different combinations of high or low meat diets and high or low efficiency food production. These scenarios determine the supply of biomass as residues and waste products, and if there is any room for dedicated biomass energy crops. In most scenarios, food demand trumps the preservation of natural ecosystems and total human land-use has to expand beyond the current ~5.2 Gha. If there is a subsequent contraction of agricultural land then we permit bio-energy crops to be planted on the land liberated (as the natural ecosystem has already been replaced). Our results should be viewed as 'first-order' estimates as we do not take a spatial modelling approach, however we do review and compare our results to literature estimates from more detailed spatial models.

The remainder of the paper is organised as follows: In Section 2 we assess the current biomass carbon fluxes appropriated by humans. Section 3 assesses the major pathways available to appropriate or redirect biomass to rebalance the carbon cycle. Section 4 develops our four future scenarios. Section 5 converts these scenarios into potential CDR fluxes, offsets of fossil fuel CO_2 emissions, and land-use change CO_2 emissions, estimating their effects on future atmospheric CO_2 concentration. Section 6 explores the key results with a sensitivity analysis and makes some suggestions for further research.

2.2 Current biomass carbon fluxes appropriated by humans

To assess the potential for future land-use to help rebalance the global carbon cycle, we start by reviewing current biomass carbon fluxes appropriated by humans. A central concept here is that of human appropriation of net primary productivity (HANPP), which is already of the same order of magnitude as total global net primary production (NPP). This suggests useful carbon dioxide removal (CDR) fluxes are as likely to come from redirecting biomass flows within the current system as from the harvest of extra biomass. With this in mind we try to identify inefficiencies in the current system of biomass appropriation.

2.2.1 The scale of HANPP

The scale of biomass harvest by humans is huge by any measure (Table 2.1). A review of biomass flows based on FAO statistics from the year 2000 calculates the total mass of biological material appropriated by humans as approximately 18.7 Pg yr^{-1} (1 Pg = 1 Gt, or a billion tonnes), or about 16% of global NPP (Krausmann et al., 2008). This includes all biomass in agricultural systems (accounting for grazed land by using grazing conversion factors and cattle stock data), as well as vegetation destroyed in fires (2.5 Pg yr^{-1}). Herein we use Pg C yr^{-1} as a common currency and assume an approximate average carbon content of 0.5 kgC per kg dry material (unless otherwise stated) (Krausmann et al., 2008; Royal Society, 2009). Hence current human biomass harvest represents a carbon flux of ~9.3 Pg C yr^{-1}.

To properly represent HANPP in a global context we must in addition account for potential changes in NPP driven by land-use change and management practices (ΔNPP_{LUC}) (DeFries et al., 1999; Haberl et al., 2002). Often this constitutes a reduction in NPP due to the clearance of productive vegetation such as forest, although in some environments the use of irrigation and chemical fertilizers help to boost NPP well above natural levels (Haberl et al., 2007, 2002). Total HANPP is thus described as the difference between the NPP of *potential* vegetation (NPP_0) in the absence of human activity and the portion of NPP remaining after harvest (NPP_t). A spatially explicit approach using the Lund-Potsdam-Jena [LPJ] dynamic global vegetation model (DGVM) in conjunction with agricultural data from FAO harvest indices finds that ΔNPP_{LUC} amounts to an anthropogenic reduction in global NPP of 9.6%, or 6.3 Pg C yr^{-1} (Haberl et al., 2007). When added to biomass harvest this puts HANPP at 15.6 Pg C yr^{-1}, or 23.8% of the estimated NPP_0 of 65.5 Pg C yr^{-1} (26.3% of *actual* NPP (NPP_{act})), with biomass harvest contributing 53%, ΔNPP_{LUC} 40% and the remaining 7% destroyed in anthropogenic fires.

This DGVM estimate of HANPP falls between low and high estimates of 11.5 Pg C yr^{-1} and 20.8 Pg C yr^{-1} (20.2 – 36.6% NPP_{act}) given in a different study which uses consumption data rather than harvest indices and derives NPP from a model based on satellite remote sensing data rather than the process-based DGVM (Imhoff et al., 2004). Here the lower estimate excludes “components of NPP that are lost to land transformation (for example, ‘shifting cultivation’ and

'land clearing')", i.e. some or all of ΔNPP_{LUC}, and also the below ground biomass of grazed land, whilst the high estimate includes measures of both of these.

Table 2.1: Summary of estimates of HANPP.

HANPP	HANPP (% NPP_{act})	Year	Key features and assumptions	Ref.
3.2 – 26.4 Pg C yr^{-1}*	3 – 39%	1986	A range of estimates based on different definitions of HANPP, with higher estimates coming from more inclusive definitions. Biomass appropriation data from 1970's-80's making it somewhat out of date, but broadly in agreement with later studies.	Vitousek et al. (1986)
20 ± 14 Pg C yr^{-1}	32 ± 22 %	2001	Statistical assessment of HANPP estimates derived from the available literature using stochastic variation within error margins and 1m permutations, reveals very high uncertainty.	Rojstaczer et al. (2001)
11.5 Pg C yr^{-1} 20.8 Pg C yr^{-1}	20.2% 36.6%	2004	CASA model based on satellite data, HANPP calculations from biomass consumption indices by country.	Imhoff et al. (2004)
15.6 Pg C yr^{-1}	26.3%	2007	LPJ-DVGM, clearly defined definition of HANPP. Reworking along definition from Vitousek et al. (1986) shows agreement with other studies.	Haberl et al. (2007)
273 $EJyr^{-1}$	12%	2007	Uncertainty in conversion factors from biomass to energy value greater than for biomass to carbon content. Does not account for ΔNPP_{LUC} or fires.	Smeets et al. (2007)
9.3 Pg C yr^{-1}	16.0%	2008	Unpacks biomass flows	Krausmann et al. (2008)

* Obtained from values for biomass given in the paper, using the conversion rate given by the authors of 2.2 kg biomass per kg C.

Given that the DGVM study does not include below ground biomass on grazing lands, their mid-range estimate implies good agreement with the satellite-based approach. Indeed much of the variation in estimates of HANPP (Table 2.1) arises due to the different definitions used by different authors (Haberl et al., 2007; Vitousek et al., 1986). Thus we consider that these studies represent a

robust approach, making the best use of available spatially explicit global databases in conjunction with a process-based model (Bondeau et al., 2007; Haberl et al., 2007), and we use them as our starting point.

2.2 Biomass flows

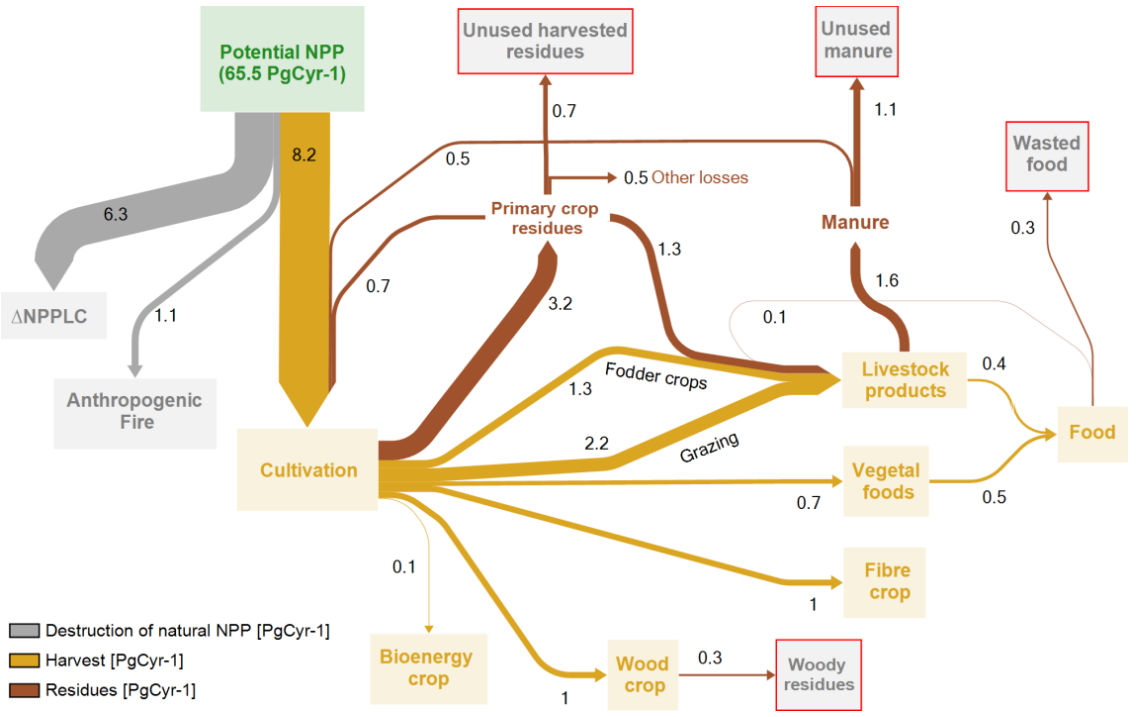

Figure 2.1: A schematic representation of the major biomass flows associated with HANPP in the year 2000, based on data from FAOSTAT, Haberl et al. (2007), Krausmann et al. (2008) and own calculations. Boxes outlined in red indicate unused biomass flows with the potential to generate CDR fluxes.

A basic schematic of the carbon fluxes associated with HANPP is given in Figure 2.1. Of the 9.3 Pg C yr^{-1} appropriated as biomass approximately a third (3.3-3.6 Pg C yr^{-1}) is not further imported into any human process. Some of this biomass is destroyed in anthropogenic fires (1.14 Pg C yr^{-1}), and the remainder is largely made up of the unused roots and residues of agricultural and forestry crops and animal manure (Haberl et al., 2007; Krausmann et al., 2008). These latter are sometimes described as backflows to natural cycles, although in fact a significant fraction of the residues remain within intensely managed systems, and make an important contribution to soil fertility (Lal, 2004), and could thus be counted in HANPP. This highlights some of the blurring of boundaries involved in defining HANPP; systems managed by humans interact with, and rely upon, natural processes such as the cycling of carbon in soils which are difficult to quantify and highly variable. Despite this, some unused wastes and residues of

biomass appropriation represent the potential for efficiency to be gained within the current harvesting process, providing more useable fixed C from the present system (see Section 2.3.2).

Of the ~6 Pg C yr^{-1} of biologically fixed carbon ultimately used by humans around 20% is used as raw material. A further 10% provides fuel; both the 'traditional' burning of wood which supplies 37-43 EJ yr^{-1} and 'modern' bioenergy feedstocks which currently supply 11.3 EJ yr^{-1} (Chum et al., 2011). Approximately 12% (0.73 Pg C yr^{-1}) is used directly in the production of vegetal foods, while a further 58% (3.5 Pg C yr^{-1}, including both fodder crops and grazed biomass) is consumed by livestock to produce meat, eggs and milk (Krausmann et al., 2008). These figures highlight the inherent inefficiency of food energy derived from livestock products; in 2000 meat made up 16.06% of a global food energy consumption of 25.63 EJ yr^{-1} (Smeets et al., 2007), giving average realized food energy yield per unit biomass turnover of 29.36 EJ per Pg C for vegetal products and only 1.21 EJ per Pg C for livestock products. The enormous inefficiency of meat production stems from the energy losses involved in turning primary phytomass into animal matter; of all the feed energy consumed by cattle ~46% is lost in manure, ~43% in respiratory heat, ~6% as methane, and 1% in the by-products of slaughter; leaving only 4% in food products (Wirsenius, 2003).

Ratios of energy yield to biomass turnover also reveal a further inefficiency; global food energy consumption of 25.63 EJ yr^{-1} in fact represents only 1.1% of terrestrial NPP (2280 EJ yr^{-1}), but to provide this energy to people as food involved the harvest of 4.25 Pg C yr^{-1}, or 7.2% of terrestrial NPP. Based on others' figures and FAOSTAT we calculate the total efficiency of the food system (i.e. total energy value of food related biomass appropriation divided by the food energy actually consumed) as around 10.3%. A more detailed analysis, including the losses of energy at the levels of processing and consumption, puts it at 8% (Wirsenius, 2003). These low figures indicate significant potential for reducing the footprint of agriculture through increased efficiencies at every level from management of inputs to harvesting, livestock rearing and post-harvest processing, as well as management of wastes.

2.3 Biomass pathways to rebalancing the carbon cycle

Biological carbon fixation could play an important role in rebalancing the global carbon cycle, as a potentially cost-effective mechanism for generating a flux of carbon from the atmosphere to long-term storage, whilst also providing replacements for fossil fuels with lower net CO_2 emissions. Biomass used in this way must either come from the redirection of extant flows, e.g. wastes, residues and more efficient processing, or new biomass flows must be generated. Here we introduce the different pathways to increasing HANPP, to increasing the efficiency of use of HANPP, and to converting biomass into long term carbon stores, considering the broader environmental consequences of these activities. We have chosen to focus on capturing and storing carbon rather than offsets through carbon neutral energy generation, since photosynthesis is in fact somewhat inefficient at converting sunlight into useable energy (at best about 0.5%, compared to around 20% for solar photovoltaics (MacKay, 2010; Pickard, 2010)), but is nonetheless effective at capturing carbon from the atmosphere.

2.3.1 Increasing HANPP

The human appropriation of net primary productivity can be increased either through increasing the NPP_{act} of extant managed systems, or by replacing natural ecosystems with crops (which we only consider justified in pursuit of food production, not bio-energy).

Historically the area of land under management has increased in response to increasing food demands, but in the last 50 years increasing yields on existing lands met the great majority of increased demand; there was only a 12% expansion of managed land between 1960 and 2000 (Foley et al., 2005). Conversion of natural vegetation to croplands is currently highest in the tropics, with much recent growth having occurred in South America. However, the clearance of tropical vegetation incurs very high costs in terms of carbon emissions, for relatively low yields due to poor soils, suggesting that intensification of farming on current land is desirable over expansion (West et al., 2010). In fact, meeting food demand through yield increases rather than expansion has saved emissions of 161 Pg C since 1961 (Burney et al., 2010). There are also advantages in terms of HANPP; an increase in HANPP through intensification represents a smaller proportion of global NPP than the same

increase through expansion, because the latter implies a simultaneous reduction in natural NPP.

Increasing the productivity of managed land, effectively reducing ΔNPP_{LUC} to bring NPP_{act} closer to or even above NPP_0, in theory has the potential to provide extra biomass flows without further appropriating biomass from natural ecosystems. This can be done without creating a significant new CO_2 source: CO_2 fluxes from established agricultural land are large, but approximately balanced by uptake leaving very small net emissions of ~0.02 Pg C yr^{-1} (Bondeau et al., 2007; Smith, et al., 2007). However, intensification can increase the emissions of more potent greenhouse gases. Currently, methane (CH_4) and nitrous oxide (N_2O) emissions from cropland and livestock amount to the equivalent of 1.4-1.7 Pg C yr^{-1} of CO_2 emissions. Still, there is significant potential for changing land-use and management strategies to mitigate combined greenhouse gas emissions (CO_2, CH_4, N_2O) by the equivalent of up to 1.6 Pg C yr^{-1} by 2030 (Foley et al., 2011; Roy et al., 2002; Smith, et al., 2007).

Continued increases in yield are implicit in most discussions of future food supply, and certainly those that consider the environmental costs of the growing food system (Nakićenović and Swart, 2000; van Vuuren et al., 2011). However, whilst yields continue to increase, the rate at which they do so has been falling; overall crop yields increased 20% between 1985 and 2005, compared to 56% between 1965 and 1985 (Foley et al., 2011). This suggests that a renewed drive to increase yields must come by different means to those employed in recent decades, although of course the breeding techniques developed during the green revolution are likely to remain integral. Some developments in farming methods do appear promising in this sense; the widespread shift to no-till farming in the US, for example, has seen reduced soil erosion and lowered inputs and GHG emissions, although evidence that the practice actively sequesters carbon is contested (Marland, et al., 2008; West and Marland, 2002).

2.3.2 Increasing efficiency

The recycling of wastes and residues can reduce the impact humans have on the Earth system by effectively increasing the efficiency of our social

metabolism; reducing both the inputs and wastes required for a given level of 'metabolic' activity (Fischer-Kowalski and Haberl, 2007). The considerable streams of wasted biomass generated by agriculture and the food industry represented a flux of ~2.7 Pg C yr^{-1} in 2000 (Krausmann et al., 2008); around a quarter of total anthropogenic carbon emissions in the same year (Le Quéré et al., 2009). Permanently sequestering this existing carbon flux could substantially reduce the current imbalance in the global carbon cycle, with potentially much lower environmental impact than expanding managed land or intensifying production. It would also make more efficient use of the enormous inputs of energy and chemicals required to grow crops in the first place. Indeed waste biomass is expected to make up a large portion of bioenergy feedstock in the future (Calvin et al., 2009; Chum et al., 2011).

The recycling of surplus biomass from our managed land poses an interesting question, however: Since appropriated biomass that is wasted represents not just a waste of carbon or stored energy, but also a wasted investment of costly nutrients which themselves are non-renewable or energy intensive to produce, is there greater net benefit in sequestering the carbon in waste biomass and making use of some of the energy it contains, or in recycling the nutrients it contains? In natural ecosystems, dead plant matter and animal faeces are recycled by decomposers in the soil, allowing nutrients to be reused and making organic molecules available to other organisms, forming a vital part of the ecosystem. In traditional agriculture, non-edible plant matter is composted and returned to the system or burned *in situ*, in both cases allowing some recycling of nutrients, and food waste is either fed to livestock or composted. Manure has historically been the best available fertilizer, containing 55-95% of the plant nitrogen consumed by livestock (Oenema and Tamminga, 2005).

In modern industrial agriculture, while 20-40% of manure and perhaps 20% of food waste is still recycled globally (Oenema et al., 2007; Wirsenius, 2003), the availability of mineral fertilizers has reduced emphasis on the importance of recycling of organic waste in providing nutrients and maintaining the physical properties of the soil. In fact, removing even 40% of crop residues can lead to soil erosion above acceptable levels, although no-till approaches allow a larger portion to be removed (Sheehan et al., 2003). The long term benefits of leaving crop residues on the field, including yield gains and soil carbon sequestration,

may well outweigh any short term advantages of removing them to produce biofuels (Lal, 2004). While animal manure and urine that is excreted while grazing is left on the ground, rapidly decomposing and releasing its nutrient content, around 50% of excreta are released by animals housed in barns and must be collected (Oenema and Tamminga, 2005). This nutrient rich manure is subsequently stored before being applied to farmland, and as a consequence about 30% of the nutrient content is lost as NH_3, N_2O, NO and N_2 before application, with a further 19% lost as NH_3 from the land (Oenema and Tamminga, 2005).

2.3.3 Converting biomass to longer-lived carbon stores

There are several pathways to convert different forms of biomass to longer-lived stores of carbon (Figure 2.2). From the perspective of rebalancing the global carbon cycle, a 'long term' storage of carbon means for millennial timescales. Most dead biomass, in contrast, decays over days to decades. Our main focus here is on methods of converting dead biomass to either stored CO_2 or charcoal (termed 'biochar' when added to soil), which also produce energy that can be used to offset fossil fuel CO_2 emissions. These methods are collectively termed 'biomass energy with carbon storage' (BECS), whereas the very similar 'biomass energy with carbon capture and storage' (BECCS) refers just to those methods that capture CO_2.

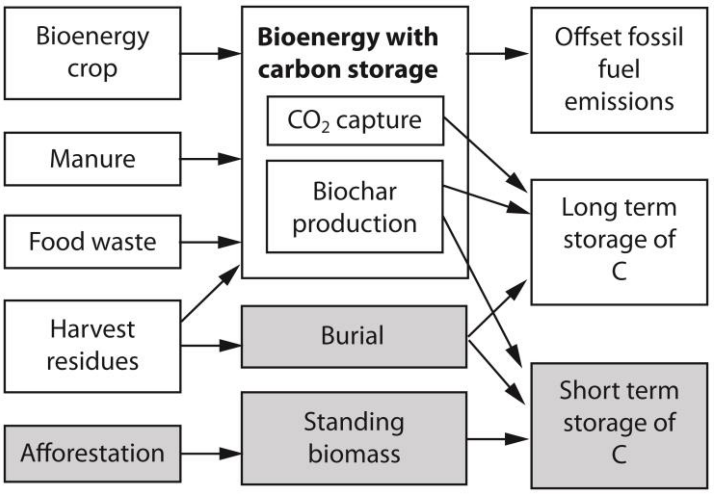

Figure 2.2: Pathways for converting biomass carbon to captured carbon. We focus on the options in white in this Perspective.

Biomass energy with carbon capture and storage (BECCS)

There are several pathways (Möllersten et al., 2003) for converting biomass carbon to captured CO_2 including; (1) biomass combustion with flue gas CO_2 capture, (2) biomass gasification then CO_2 capture (with an optional CO shift) before combustion or conversion to fuel, (3) air separation of pure O_2 for biomass combustion with CO_2 capture, (4) biomass fermentation to biofuel (sometimes preceded by saccharification) with CO_2 capture, (5) biomass conversion to biofuel via the Fischer-Tropsch process with CO_2 capture. For long-term storage, captured CO_2 can be compressed and thus liquefied before injecting into geological reserves, or it can be reacted with basic minerals such as lime or calcium carbonate to produce a charge neutral solution that can be added to seawater. These processes of capture and storage carry energy penalties, with the penalty being greater when the CO_2 stream is less pure.

Carbon capture potential varies considerably across these technologies, as do the potential offsets of fossil fuel burning. Carbon capture yields of 90% (and possibly higher) with a corresponding ~30% energy yield as electricity, are claimed for (2) a biomass integrated gasification combined cycle (BIGCC) with CCS (Azar et al., 2006; Klein et al., 2011). However, other authors estimate only a 55% carbon yield with 25% energy yield as electricity for the same type of system (Rhodes and Keith, 2005). Higher energy yields are estimated in the form of hydrogen production (55%) or heat production (80%) (Azar et al., 2006). Lowest carbon capture yields are for converting sugar cane to ethanol (4), which leaves two-thirds of the carbon in the ethanol and releases one-third as CO_2. However, future biofuel is likely to be dominated by lignocellulosic crops, for which processing by saccharification and fermentation (4) or Fischer-Tropsch (5) leaves around half of the carbon in the fuel, with carbon yields of (4) ~13% or (5) ~41% as high purity CO_2 (Luckow et al., 2010).

BECCS technologies generally lend themselves to relatively uniform feedstock such as dedicated bioenergy crops. However, CO_2 flue gas capture from combustion (1) can use mixed feedstock and indeed co-firing of coal and biomass is already occurring. Captured CO_2 from BECCS could ultimately compete for geological storage capacity with conventional CCS from fossil fuel burning with current estimates of storage capacity ranging widely over 500-3000 Pg C (Lenton, 2010).

Biochar production

Charcoal is typically produced by biomass pyrolysis although thermo-catalytic depolymerisation has also been demonstrated. When returned to soil as biochar a significant (but debated) fraction, e.g. 85%, is long-term resistant to biological decay (Woolf et al., 2010). The carbon and energy yields of biochar production vary greatly with the temperature of pyrolysis. In systems optimised for biochar yield, up to 63% carbon capture is possible via pressurised flash-pyrolysis, with an energy yield of around 35% in gas (59% of the energy is left in the char and 6% lost) (Shackley et al., 2012). A more conservative figure is ~50% carbon capture with a similar energy yield (Woolf et al., 2010).

Returning biochar to the soil can have further benefits, helping retain water and mineral nutrients and thus boosting productivity on poor soils, or reducing the need for application of fertilizers (Lehmann et al., 2006). Emissions of N_2O can also be reduced (via ammonium absorption) in some agricultural systems (Singh et al., 2010; Spokas et al., 2009; Zhang et al., 2012) but not others (Clough et al., 2010), whereas methane emissions can be increased leading to little net effect on global warming potential (Zhang et al., 2012). Biochar production lends itself to biomass residues from agriculture and forestry and other mixed feedstocks like food waste and manure, providing a convenient recycling mechanism for organic wastes. Widespread biochar application has not yet been trialled however, and the specific mechanisms of its effects on soil properties are not well understood; a more recent and comprehensive review of the biochar literature has suggested that the evidence does not exist to support the claims made about its potential (Gurwick et al., 2013). Global storage capacity for biochar in cropland, grassland and abandoned land soils has been estimated at ~500 Pg C (Lenton, 2010).

Afforestation and biomass burial

Afforestation, if it replaces an ecosystem which had less carbon storage, is a CDR mechanism, but the forest must be permanently maintained. The potential carbon sink that could be generated by afforestation has been estimated at ≤1.5 Pg C yr^{-1} in 2050 (and ≤3.3 Pg C yr^{-1} in 2100) but these estimates depend crucially on the assumed supply of land (Lenton, 2010). Furthermore, the yield of carbon per unit area can be an order of magnitude less for permanent afforestation (~1 Mg C ha^{-1} yr^{-1}) than for regularly harvested dedicated woody

biomass energy crops (~10 Mg C ha^{-1} yr^{-1}). Hence, with a restricted supply of land (due to constraints from food production and preservation of natural ecosystems) if the objective is to maximise carbon dioxide removal, then other pathways of biomass energy with carbon storage should take priority. Of course, there will remain a global demand for wood products which is considered in our scenarios (below).

Alternatively, burial of either forestry residues or agricultural crop residues have been proposed on a global scale (Metzger et al., 2002; Strand and Benford, 2009; Zeng, 2008). Whilst this might create a significant CDR flux of up to ~0.5 Pg C yr^{-1} based on year 2000 biomass flows, it would provide no energy benefits or corresponding offsets of CO_2 emissions. It also raises questions about the ecological and erosion impacts of removing carbon from ecosystems, and whether anaerobic decomposition of buried biomass could create a methane flux that would counter the CDR flux (Lenton, 2010). Hence we do not consider biomass burial in our scenarios here.

2.4 Future scenarios

In this section we develop our future scenarios out to 2050. First we estimate the future demand for food and introduce the four scenarios for meeting that demand. Then we consider key controls on the efficiency of agricultural systems, which are increased in our high-efficiency scenarios. Finally we estimate the future HANPP and land-use under the four scenarios.

2.4.1 Food demand

Increasing global food demand drives our future scenarios. Projections for food consumption into the future are based on a combination of population size and demography (including diet). In all our scenarios we assume a population rise from 6.1 billion in 2000 to a little over 9.3 billion in 2050 following the UN world population prospects (2010 revision) medium scenario. We also assume a demographically-led increase in per capita food demand from 2760 to 3302 kcal $cap^{-1}day^{-1}$ in 2050 (FAO, Global Perspective Studies Unit, 2006; Smeets et al., 2007). This represents an increase of 82% in demand for food energy, from approximately 26 $EJyr^{-1}$ to 47 $EJyr^{-1}$.

To assess the impact of this increased food demand in terms of carbon fluxes, we constructed four simple scenarios based around differing food production systems, for the period 2000 to 2050, using data from the FAOSTAT database. We chose 2000 as a start year because a fairly complete dataset is available then, acknowledging that at the time of writing we are already 12 years into the scenarios. All four scenarios assume sustained increases in yield of vegetal products (i.e. vegetal foods and animal fodder) of 1% yr^{-1}, similar to the current rate of around 0.95% yr^{-1} (Foley et al., 2011)(as commonly assumed in other studies), which could be viewed as optimistic as the rate of yield increase is currently falling. Where the scenarios differ is in their assumptions about the dietary demand for meat and the efficiency of the agricultural system used to meet food demand.

An approximate 'hierarchy' of foods are preferred as populations become wealthier; from a diet based on maize and coarse grains societies undergoing urbanization trend towards increasing use of wheat and rice, while wealthy populations consume far more meat, dairy fruit and vegetables. Developing countries with rapidly growing populations and economies accounted for over 50% of milk and meat production in 2005, and this figure is likely to continue rising (The Government Office for Science, 2011). Increases in cereal and feed production (usually maize and soybean) are required to support the meat and dairy consumption of a wealthier global population. Average per capita annual meat consumption is expected to rise from 16.6% to 18.8% of daily calorific intake by 2050 and we use this in our 'high-meat' variants. In contrast, in our 'low meat' variants we assume that per capita consumption of animal products declines, contributing 15% of daily energy intake by 2050. This is comparable to the global average consumption of animal products in the 1960s, and combined with the increasing average calorific consumption is very similar to the average diet of East Asia today (Smeets et al., 2007). It would, however, represent a considerable increase in meat eating in Africa or Southern Asia, and a halving of the meat intake of the average Western diet.

In our 'low efficiency' variants, the methods used to meet rising food demand (including the balance of livestock products and the balance of grazing and fodder feeding of livestock) remain the same as in 2000. Hence the residues from food related biomass harvest remain proportional to those in 2000. In

contrast, in our ‘high efficiency’ variants there are shifts in the types of livestock used to supply meat demand, in the balance of grazing and fodder feeding of livestock, and in the amount of recycling of biomass within agricultural systems.

The four scenario combinations are thus ‘high-meat, low-efficiency’, ‘low-meat, low-efficiency’, ‘high-meat, high-efficiency’ and ‘low-meat, high-efficiency’. The ‘high-meat, low-efficiency’ scenario essentially asks ‘what would happen if we tried to meet expected food demand in 2050 using current methods?’ (albeit with a 1% yr^{-1} sustained increase in yield of vegetal products). The ‘low-meat, low-efficiency’ scenario asks how the results would be changed by a dietary shift to less meat. The ‘high-meat, high-efficiency’ scenario asks how the results would be changed by increases in the efficiency of agricultural systems. The ‘low-meat, high-efficiency’ scenario looks at the combined effects of dietary change and increased efficiency.

2.4.2 Controls on efficiency

Key controls on overall agricultural efficiency are the types of livestock used to supply meat demand and the balance of grazing and fodder feeding of livestock (Figure 2.3). Yields in the livestock production system have increased significantly between 1985 and 2005, probably through a combination of intensification of industrial livestock production methods, and the use largely of pigs and poultry rather than cattle to meet increasing demand for animal products in the developing world (FAOSTAT, 2014; Foley et al., 2011; Wirsenius, 2003). The feed conversion efficiencies of pig and poultry farming may be up to a factor of 10 higher than those of cattle farming (Figure 2.3 a) due to the diets and life-histories of domestic birds and pigs, especially in systems in which food-waste contributes a significant portion of their diet (Wirsenius, 2003). In all our scenarios, the biomass harvest required by the livestock sector is calculated using overall efficiency values (total livestock products divided by total biomass harvest of the livestock sector) of dairy 3.1%, bovine meat 0.505%, eggs and poultry meat 3.6%, pig meat 6% (Wirsenius, 2003). While fodder crops are not explicitly represented in FAO data, we assume that their yields increase at the same rate as other vegetal crops.

In our high-efficiency scenarios, overall growth in the livestock sector to meet rising demand for livestock products occurs in conjunction with a continued shift

towards pig and poultry products, which each triple as a proportion of total livestock relative to the 2000 baseline (calculations from FAOSTAT data). Poultry are in fact expected to meet a significant part of the growth in demand for meat, increasing by 83% from 2002 to 2020 (Roy et al., 2002).

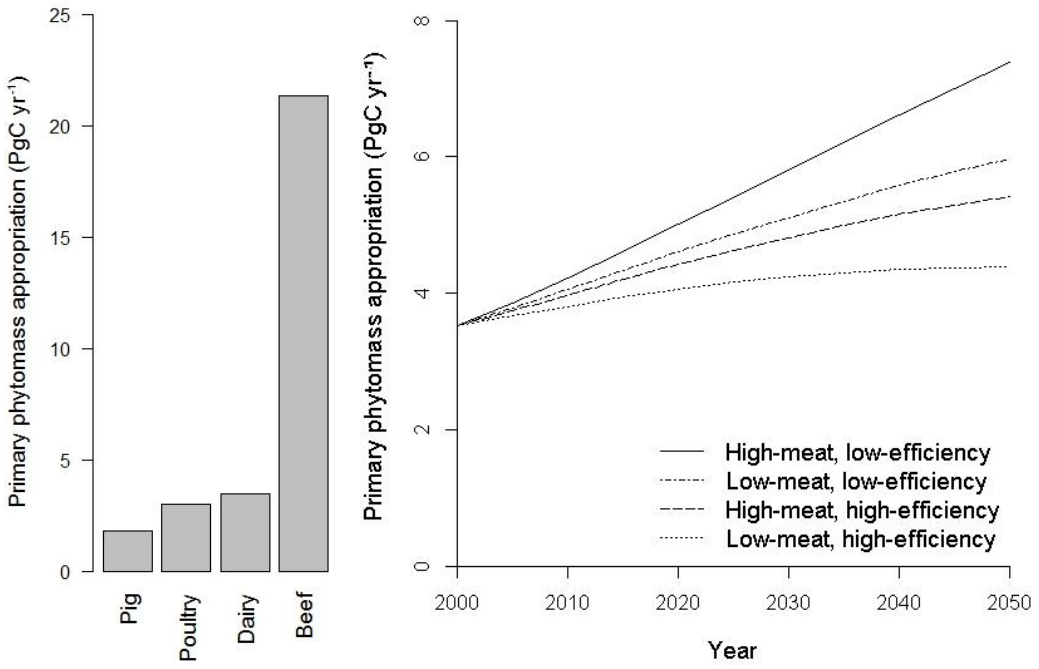

Figure 2.3: The (in)efficiency of global livestock production: **a)** shows the primary phytomass necessary to feed the worlds' demand for livestock products in the year 2000 at the conversion efficiencies of four main livestock categories (following Wirsenius (2003)); **b)** shows the trajectories of phytomass demand followed in our four scenarios, as a result of the changing mix of animal species in the high-efficiency scenarios, and the decline in demand for livestock products in the low-meat scenarios.

The overall efficiency from biomass harvest to final consumption of food products is also increased in both livestock and vegetal sectors through an assumed 20% reduction in the proportion of food wasted in distribution and post-purchase, resulting in an increase of efficiency from 29.8% to 32.7% for vegetal crops between 2000 and 2050. This is further enhanced in the livestock sector by increases in the fractions of agricultural residues and food wastes fed to livestock by 15% and 20% respectively; artificially high efficiencies could be obtained by increasing this proportion still further, but there is evidence that much of the more nutritious residues are already used in feed, but currently little in the way of food waste is recycled in the developed world, and so some potential remains (Wirsenius, 2003).

Residue fractions (Table 2.2) are calculated from the residues in 2000 as a fraction of the total biomass harvest of the relevant sectors; i.e. vegetal food and fodder crops for primary crop residues; livestock biomass harvest for manure; forestry for woody residues. Since the four scenarios vary in their efficiency of conversion of biomass to food energy, an initial value for the residue fraction for food waste was used to calculate wasted food in proportion to food energy consumption; this then declined by 20% between 2000 and 2050 in the high-efficiency scenarios. Unused primary crop residues are calculated as a fraction of combined vegetal food crops and fodder crops, and manure is calculated as a fraction of the total biomass harvest of the livestock sector.

Table 2.2: Residue fractions for key residues.

Residue	Residue fraction*	Reference
Unused primary crop residues	0.436	Krausmann et al. (2008)
Harvested primary residues	0.724	Krausmann et al. (2008)
Manure	0.462	Wirsenius (2003)
Forestry residues	0.337	Krausmann et al. (2008)
Food waste	0.062	Wirsenius (2003)

*Residue fraction is kg residue per kg primary biomass harvested

The combined effects of these changes in the high-efficiency scenarios increase the overall efficiency of the livestock system from 3.06% in 2000 to 4.28% in 2050 (FAOSTAT, 2014; Krausmann et al., 2008; Wirsenius, 2003). The overall efficiencies we find in the livestock and vegetal sectors in 2000 are comparable to the averages given elsewhere; 21.2% for vegetal products and 1.6% for all animal products; with differences arising through the different time periods and biomass harvest datasets used in the studies.

2.4.3 HANPP

Primary phytomass appropriation of the livestock sector increases in all our scenarios to 2050 (Figure 2.3 b), only stabilising in the high efficiency-low meat scenario. Increases in efficiency reduce phytomass demand slightly more than the assumed lower meat diet, and their effects are nearly additive. The livestock sector makes the dominant contribution to HANPP in 2050, which differs considerably between scenarios (Figure 2.4): With high-meat, low-efficiency,

the HANPP of the food system alone increases 104% from 5.0 Pg C yr^{-1} in 2000 to reach 10.2 Pg C yr^{-1} in 2050, or 17.2% of NPP_{act} (NPP_{act} in 2000), at 9.3% overall efficiency. With recycling efforts and a shift toward the higher efficiencies associated with pig and poultry farming, the high-meat, high-efficiency scenario still sees growth of 55%, reaching 7.75 Pg C yr^{-1} in 2050, with an overall efficiency of 10.9%. The low-meat, low-efficiency scenario gives larger growth of 72%, reaching 8.6 Pg C yr^{-1} in 2050 at 10.8% efficiency. Whereas in the low-meat, high-efficiency scenario biomass harvest increases just 32% by 2050, to 6.59 Pg C yr^{-1}, alongside an increase in overall efficiency to 12.8%.

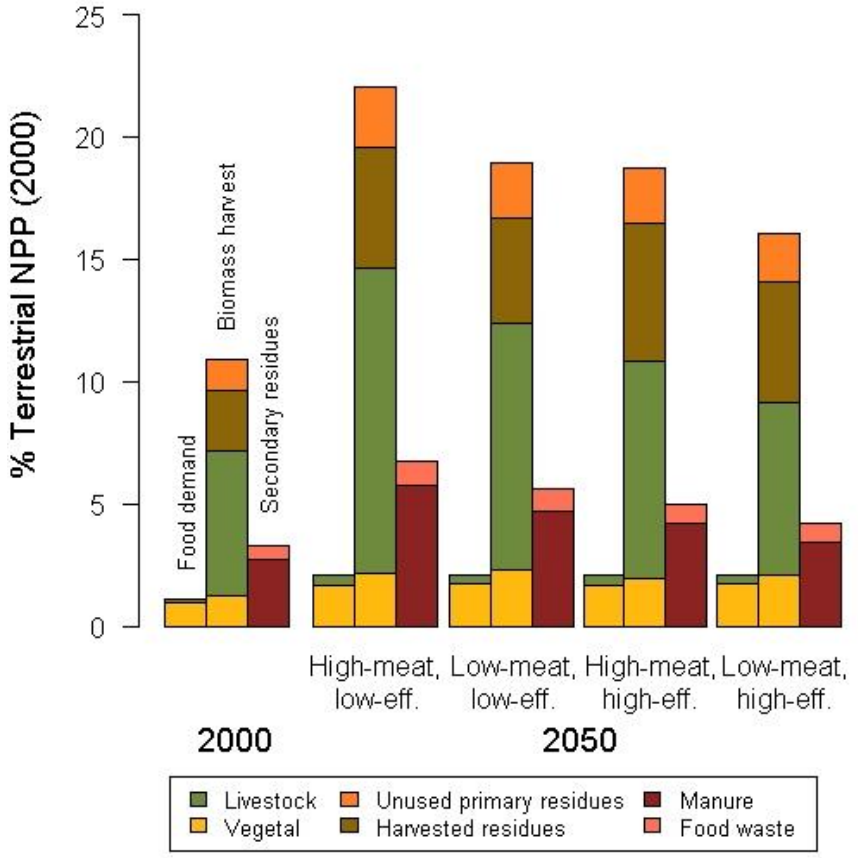

Figure 2.4: The demand for food energy, and associated biomass harvest and secondary residues in 2000 and under our four scenarios in 2050, as% global NPP from the year 2000. Food energy supplied by livestock products is less than 0.5% of global NPP, but requires disproportionately large harvest of biomass, almost half of which becomes manure. The livestock sector requires less biomass harvest in the high-efficiency scenarios due to changes in the mix of animals towards higher conversion efficiencies, more recycling of primary residues, and less food waste.

2.4.4 Land-use

To determine human land-use under the four scenarios (Figure 2.5) the yield increases and changes in diet were used to calculate livestock and vegetal crop land-use at 5 year time steps, as this is the resolution of the population estimates given by the UN Population Division. In addition, intensification is represented by a change in the proportion of biomass required by the livestock sector met by growth of fodder crops as opposed to grazing, from around 37% from fodder crops in 2000 to 60% in 2050 (Foley et al., 2011). This could also be interpreted as similar to intensification of grazing, which is difficult to include explicitly due to the very diverse nature of management approaches and land-uses classified as 'grazing' or 'pasture' (Ramankutty et al., 2008). Current trends were also extrapolated to estimate the expansion of managed forests to 400 Mha by 2050 (FAO, 2010). Two further categories of human land-use are also included; urban area and fibre crops, although these are very small compared to areas used for food production. Urban area is expected to increase at a rate slightly higher than that of the human population, due to a continuing trend of urbanisation in developing countries (UN, 2008 world urbanization prospects), and from an area of about 28 Mha in 2050 is estimated to reach around 53.3 Mha in 2050 (Erb et al., 2009a). Fibre crops occupied 35.2 Mha in 2000 (FAOSTAT); we assume an increase in demand for fibre proportional to that for food, which is driven by an increase both in population and in wealth, and allow the same 1% yr^{-1} increase in yield, resulting in 38.9 Mha of fibre crops in 2050. The total area used for food production in 2000 was around 4.9 Gha, comprising 1.46 Gha primary crops and 3.42 Gha pasture (based on FAOSTAT). Combined with the area of fibre and bioenergy crops, urban area and 0.2 Gha managed forest, we calculate total human land-use in 2000 as 5.17 Gha. To protect natural ecosystems, we only allow growth of bioenergy crops on land which has previously been occupied by food crops. The option to grow crops specifically for CDR is therefore limited to scenarios in which improving yields and efficiency within the food system is able to reduce the area required for food production.

The land-use implications of our four scenarios are just as striking as those for biomass harvest, and the same patterns are observable (Figure 2.5). As with biomass harvest, changes in the livestock sector have the greatest effects on agricultural land-use. Whether livestock are grazed or fodder fed makes a large

difference; for example if all of the biomass harvested to feed animals came from fodder crops the livestock sector would require only 0.95 Gha in 2000, rather than the 3.8 Gha it occupied, while to feed all animals by grazing would need 5.4 Gha.

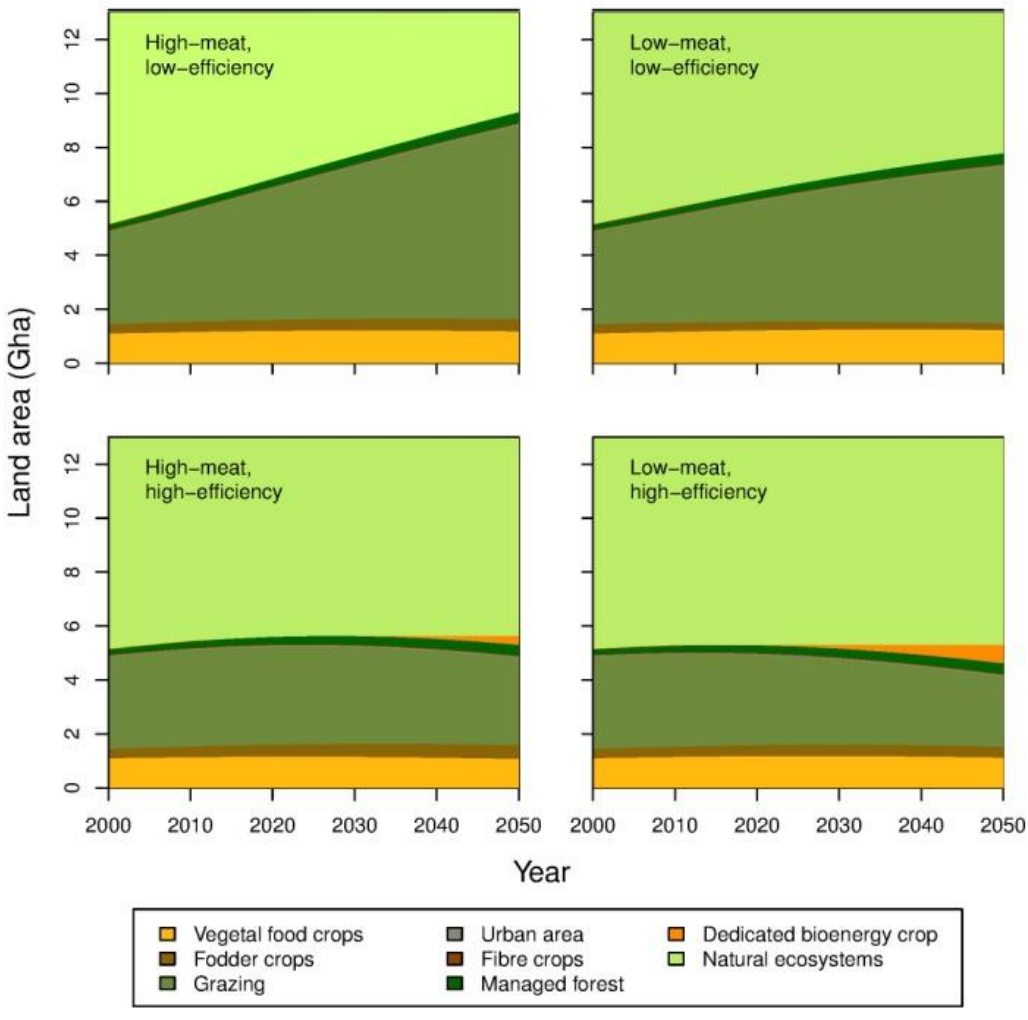

Figure 2.5: Land-use from 2000-2050 in the four scenarios.

Our high-efficiency scenarios ultimately lead to reductions in the area of land required for food production; the area of food-producing land in the 'low–meat, high-efficiency' scenario falls 15% by 2050 from 4.88 Gha to 4.13 Gha, having peaked at 4.97 Gha in 2010. In the 'high-meat, high-efficiency' scenario, although the area grows to reach a maximum of 5.26 Gha in 2025 as yield increases and efficiency improvements fail to keep up with demand from population growth and dietary change, it falls again to achieve an overall decrease of 1%, reaching 4.82 Gha in 2050. Food producing land in the 'high-meat, low-efficiency' scenario, i.e. under conditions in 2000, reaches a staggering 8.83 Gha in 2050, which must be considered an unsustainable land-

use change, and the ‘low-meat, low-efficiency’ scenario also sees an expansion of farmland by 49%, reaching 7.30 Gha in 2050. Only in the two high-efficiency scenarios is any land available for the growth of bioenergy crops; in the high-meat, high-efficiency case, bioenergy is squeezed out initially, but emerges again in the 2030s as improvements in efficiency allow expansion up to 332 Mha in 2050, within the range estimated using the LPJmL model (Beringer et al., 2011). In the low-meat, high-efficiency scenario, food-producing land starts to contract earlier, providing an increasing supply of abandoned land for growth of bioenergy feedstocks, with 686 Mha available in 2050.

Overall land-use increases throughout the fifty year period in both low-efficiency scenarios, reaching 9.3 Gha with high meat demand, and 7.8 Gha with a reduction in meat consumption. In the two high-efficiency scenarios land-use increases initially and then stabilizes as bioenergy crops occupy land abandoned by food production. Total land-use in 2050 is 5.64 Gha in the high-meat, high-efficiency scenario, and 5.31 Gha in the low-meat, high-efficiency scenario.

2.5 Effects on the global carbon balance

Here we assess the effects of our four scenarios on the global carbon balance, considering three key fluxes; CO_2 removal (CDR), offsets of fossil fuel CO_2 emissions, and land-use change CO_2 emissions. The CDR potential of each of our scenarios is determined by a combination of the feedstocks available and the way in which feedstocks are processed. Bioenergy generated can be used to offset fossil fuel use, offering further mitigation potential. However, increasing land-use in our scenarios generates net CO_2 emissions that counteract the CDR flux and offsets. The effect on atmospheric CO_2 concentration depends on the net flux to or from the atmosphere, integrated over time, taking into account that perturbations to atmospheric CO_2 concentration decay over time.

2.5.1 CDR potential

CDR potential of each scenario is determined by the supply and type of feedstocks, the efficiency of converting them to long-term storage, and the rate at which the relevant technology can be deployed. Feedstocks in our scenarios are generated by redirecting the unused waste streams from the harvesting and consumption of biomass in farming and forestry, supplemented where possible

with the growth of dedicated bioenergy crops. Residues are divided into crop residues, i.e. the parts of food crops not harvested for consumption; manure; food waste; and woody residues from forestry. Although we account for residue losses during recovery, we nevertheless assume that 100% of available feedstocks are diverted into carbon storage, and as such our figures should be seen as estimates of technical potential, and in the upper limits of what is achievable. We assume a delay in achieving full development of the infrastructure required for the collection of residues and establishing appropriate BECS facilities. The implementation of BECS increases linearly from non-existent in 2010 in all four scenarios, reaching full potential in 2030 in the high-efficiency scenarios, and 2050 in the low-efficiency scenarios (Figure 2.6). These implementation rates are relatively rapid, given the time required to plan and build new infrastructure such as power stations, especially on a scale able to process nearly 12 Pg yr^{-1} of biomass by 2050. However, global primary energy use increased by around 2.7 Btoe yr^{-1} (billion tonnes of oil equivalent per year) from 2000-2010 and is expected to rise by a further ~4.25 Btoe yr^{-1} by 2035, indicating that enormous expansions of infrastructure in the energy sector are achievable. Our assumed implementation rates result in average rates of increase in carbon storage of around 56 MtC yr^{-1} (reaching a maximum of 123 MtC yr^{-1} in 2045-2050) in the high-meat, high-efficiency scenario, and 84 MtC yr^{-1} (reaching 156 MtC yr^{-1}) in the low-meat, high-efficiency scenario. These growth rates are close to the constraining rate of 100 MtC yr^{-1} treated as 'realistic' in a cost analysis for BECCS by Azar et al. (2006). Higher rates close to 2050 in our low-meat, high-efficiency scenario might be considered reasonable since this scenario already assumes a strong drive for mitigation throughout, with citizens in Western countries prepared to consume half as much meat as we do currently. We are aware, of course, that implementation rates are highly sensitive to economic factors, and note that the mitigation costs associated with BIGCC + CCS are similar to conventional technologies in the electricity sector, though these costs can change considerably depending on carbon prices (i.e. the economic value placed on mitigation of climate change) (Rhodes and Keith, 2005). Since we lack the capacity to include a detailed economic analysis, we assume that a drive to rebalance our relationship with the carbon cycle in the next few decades will make carbon-negative technologies cost-effective.

Residues & Wastes

In the two 'low-efficiency' scenarios the unused residue fractions at harvest remain equal to those in 2000 based on the literature (Krausmann et al., 2008; Wirsenius, 2003) (Table 2.2), growing overall as the demand for food and fuel increases. Since we are concerned with maximizing CDR, we also assume that the fraction of manure currently recycled is available as a feedstock, while remaining aware that it provides a valuable soil amendment in itself. We assume this is compensated for by the subsequent addition of biochar generated not only from pyrolysis of manure but also other residues. Recovery factors are applied to the total residues available, based on assumptions about how much of each type of residue can be used while accounting for economic and ecological limitations (Table 2.3).

Table 2.3: Recovery factors for residues

Residue	**Recovery factor (2000 - 2050*)**	**Limitation**
Crop residues	0.5 – 0.35†	Must leave a portion (30-60%) on field to maintain soil structure + carbon etc. Useable portion declines in high-efficiency scenarios as some is recycled to be fed to livestock.
Manure	0.85 *(grain-fed, stabled)*‡ 0 *(grazed)*	Physical/economic limits to how much can be collected – a lot from barns, little on pasture.
Food waste	0.84 – 0.67††	Some is already recycled (depending on culture, laws etc), declines in high-efficiency scenarios as more is recycled into livestock system.
Woody residues	0.6‡‡	Physical/economic/ecological limits to how much can be collected.

*****2050 factors are those used by *high efficiency* scenarios. 2050 *low efficiency* scenarios use the same factors as 2000. **†** based on Lal (2005) and own estimates, **‡** my own estimate of likely losses in storage and collection, **††** own calculation based on FAOSTAT food-waste statistics, **‡‡** from Krausmann et al. (2008).

Studies of the technical potential of bioenergy tend to include the 'traditional' burning of biomass, since it provides energy to a large number of the world's poorer inhabitants (Chum et al., 2011; Erb et al., 2009a; Smeets et al., 2007). Since we are primarily concerned with CDR potential, however, and consider it unlikely that the infrastructure could be available to replace this portion of

energy generation with BECS in the near term, the feedstocks used in traditional energy generation are not considered here.

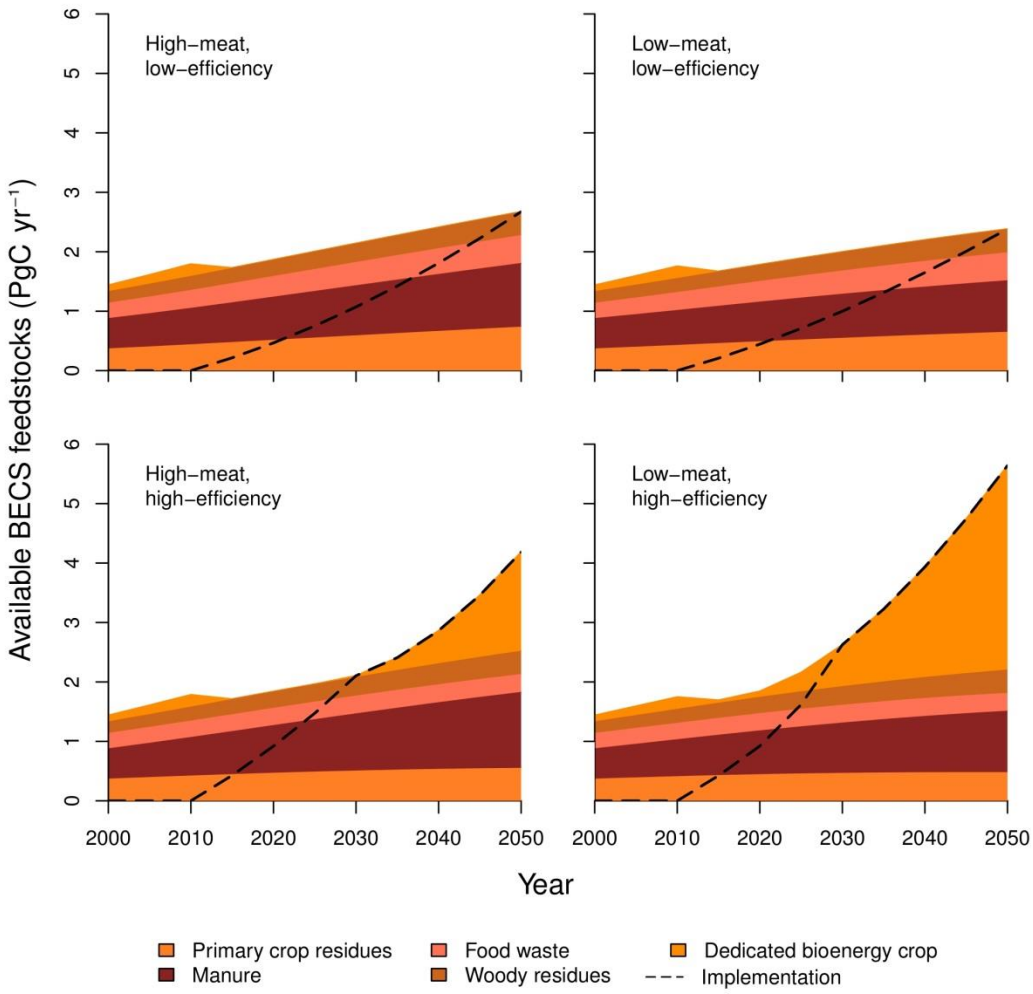

Figure 2.6: Available BECS feedstocks generated by our four scenarios, with the dashed line showing the mass actually processed in each year due to the rate of implementation of BECS facilities. Dedicated bioenergy crops are only available in the two high-efficiency scenarios.

In the high-efficiency scenarios, the portion of crop residues and food wastes useable as CDR feedstock diminishes through time, as some of what is available is recycled for the feeding of livestock (15% and 20% respectively by 2050). The amount of manure that can be recovered increases over time in these scenarios, as a result of a shift towards grain and fodder fed livestock rather than grazing. As with other elements of the food system, livestock production has a dominant role here, with manure contributing 37-51% of total residue feedstocks across the scenarios. More manure is available in the high-efficiency scenarios (51% vs. 31% of total manure production), since our high-

efficiency system features more intensive production of livestock with a larger proportion of fodder-fed, housed animals.

Crop residues and food waste are provided in greater quantities by the low-efficiency scenarios, as their lack of recycling means that not only are more wastes available, but that livestock in fodder fed systems require the growth of more primary crops than their counterparts in the high-efficiency scenarios. Overall the high-meat, low-efficiency scenario produces the largest residue stream (Figure 2.6,Table 2.4), providing 2.67 Pg C yr^{-1} BECS feedstocks in 2050; the low-meat, low-efficiency scenario produces 2.38 Pg C yr^{-1}; the high-meat, high-efficiency scenario produces 2.52 Pg C yr^{-1} and the low-meat, high efficiency scenario produces the smallest residue stream of 2.21 Pg C yr^{-1}.

Table 2.4: Calculated BECS feedstocks in 2050.

Feedstocks in 2050 (Pg C yr^{-1})	**High-meat, low-efficiency**	**Low-meat, low-efficiency**	**High-meat, high-efficiency**	**Low-meat, high-efficiency**
Crop residues	0.74	0.65	0.56	0.48
Manure	1.07	0.87	1.28	1.03
Food waste	0.47	0.47	0.30	0.30
Forestry residues	0.39	0.39	0.39	0.39
Bioenergy crop	0	0	1.66	3.43
Total	2.67	2.38	4.19	5.64

Dedicated bioenergy crops

We assume that all dedicated bioenergy crops grown in our scenarios are of the second generation, lignocellulosic variety. Crops such as willow, poplar, *Miscanthus* and switchgrass are fast growing perennial species with very efficient nutrient and water use, giving more than twice the energy yield for lower inputs than the previously used food crops (e.g. maize) (Heaton et al., 2008). These high-yield plants produce a large amount of lignocellulosic material better suited to energy generation than inedible food-crop residues such as corn stover or rice husks (Dohleman and Long, 2009; Heaton et al., 2008), and due to their lower nutrient and water requirements can be grown on marginal or abandoned lands, reducing the competition between food and bioenergy crops which has been a significant problem with the first generation of biofuels (Tilman et al., 2006). As a result, it is possible to allow for significant

growth of bioenergy crops on degraded or abandoned land thereby increasing overall the area of land under productive management, but limiting encroachment on natural ecosystems (Lenton, 2010). Furthermore, the perennial rootstocks of these crops can improve soil qualities by increasing stability, reducing runoff and fixing nutrients; it has even been suggested that damage to aquatic ecosystems could be reduced by planting *Miscanthus* as a 'buffer' around water courses sensitive to agricultural nutrient runoff (Heaton et al., 2008).

C_4 grasses and fast growing woody species such as these also respond well to CO_2 fertilization, with the LPJmL model results showing 20-30% yield increases by 2050, particularly in warm, dry climates where they use water far more efficiently than alternative crops (Beringer et al., 2011). This is particularly important in a world in which competition for water is likely to play an increasingly significant role in agricultural development (McIntyre, 2009). We have chosen, however, not to include yield increases as a result of CO_2 fertilization, as they remain uncertain, and in any case our analysis does not depend on the absolute future atmospheric CO_2 concentration.

We assume an average yield of 10 t ha^{-1} of dry matter; at the conservative end of the production achieved in field trials, but a more reasonable expectation of crops grown on marginal land and with low inputs (Beringer et al., 2011; Heaton et al., 2008; Lewandowski et al., 2000). The resulting bioenergy feedstocks available for BECS in 2050 are 1.66 Pg C yr^{-1} in the high-meat, high efficiency scenario and 3.43 Pg C yr^{-1} in the low-meat, high-efficiency scenario (Table 2.4).

Conversion to stored carbon

Several options to convert biomass to stored carbon are available, as outlined in Section 3.3, and our analysis is necessarily sensitive to the pathways we choose. Recognizing a need for solutions that reduce the footprint of agriculture across multiple dimensions rather than the more 'single issue' approach all too often taken, and following concerns about the effects on soil quality of removing of agricultural residues for bioenergy production, we choose pyrolysis as the fate for all residues (Lal, 2005). This allows recycling of the nutrients contained in otherwise unused parts of crop plants and manure, and has other potential benefits for soil (Clough et al., 2010; Lehmann et al., 2006; Singh et al., 2010;

Spokas et al., 2009; Zhang et al., 2012). These benefits include improvements in water retention, again likely to be a welcome feature of treated soils in a future world with severe limitations in availability of freshwater, and reduced need for fertilizer application, thereby lowering costs, and reducing runoff and N_2O emissions. A further benefit of pyrolysis is its relative ease of implementation at a localized scale, suitable for the collection and transport of farm and household wastes and the distribution of biochar products. For pyrolysis we take the conservative figures given in Section 2.3.3, assuming long-term sequestration of 50% of carbon content of the feedstock, with energy yield of 35%. For dedicated bioenergy crops, which are perennial species with their own beneficial stabilizing effects on soils, we choose BIGCC with CCS as it offers the greatest potential for fixing carbon from the feedstock; to this end we use a carbon yield of 90% as in other studies (Azar et al., 2006; Klein et al., 2011), whilst aware that this is an emerging technology with apparently very different results depending on the methods used. Since in other respects this study represents an exploration of the technical potential for biological carbon removal, we feel that it is reasonable to use this higher figure rather than the 55% efficiency described by Rhodes & Keith (2005). The carbon fluxes obtained from diverting the total biomass streams to their respective end uses should be taken as technical potentials, since they assume that all useable residues and crops that are harvested across the globe are used for the same purpose, which assumes an unprecedented level of worldwide decisiveness and cooperation.

Pyrolysis of the residues produced by each scenario fixes 1.1-1.34 Pg C yr^{-1} in 2050 (Table 2.5). In the high-efficiency scenarios BIGCC + CCS of bioenergy crops contributes substantial CDR fluxes 1.49 Pg C yr^{-1} in the high-meat version, and 3.09 Pg C yr^{-1} in the low-meat variant. The combined CDR fluxes for each scenario, as a function of time are shown in **Error! Reference source not found.** a.

2.5.2 Offsets of fossil CO_2 emissions

As well as actively removing carbon from the atmosphere, the generation of energy in BECS contributes to mitigation by offsetting fossil fuel CO_2 emissions. Although we deliberately opt for processes which maximize conversion of the feedstock to stored carbon, which necessarily entails a penalty in terms of reduced energy yield, offsets contribute a significant boost to the CDR fluxes

generated in our scenarios. Our calculations assume feedstock energy content of 37 MJ kg^{-1} C (i.e. 18.5 MJ kg^{-1} dry matter), as used in the IPCC SRREN report (Chum et al., 2011; Haberl et al., 2010).

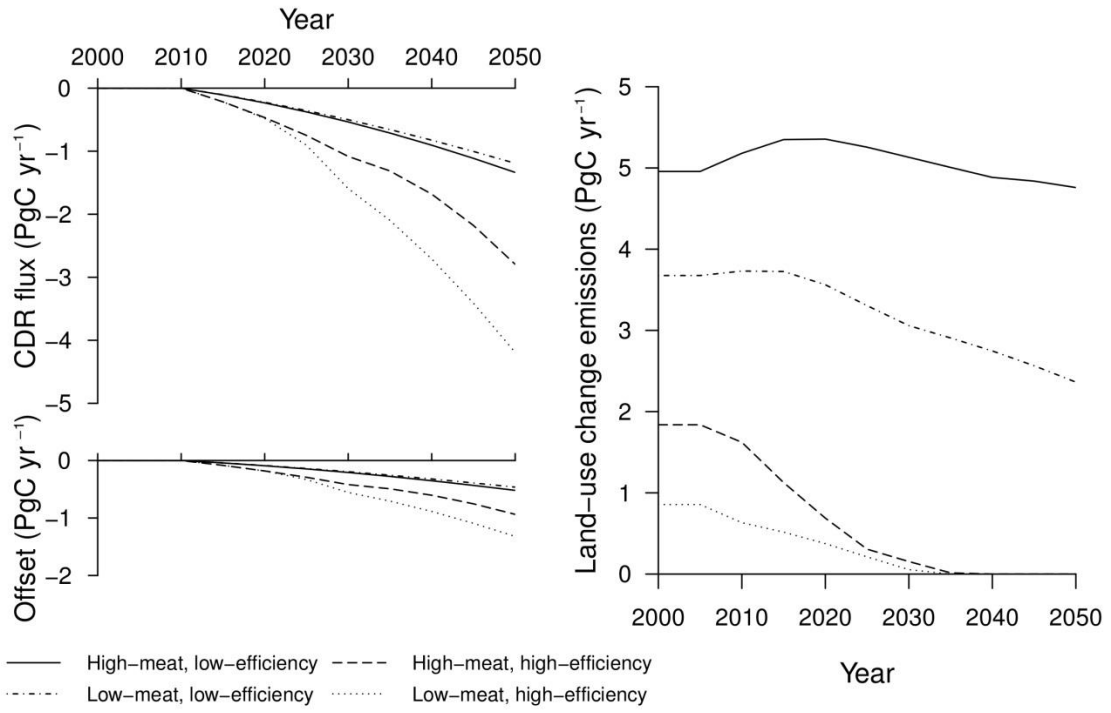

Figure 2.7: Carbon fluxes generated by the four scenarios. Negative fluxes are the result of BECS processing of feedstocks which both generates **(a)** a CDR flux and **(b)** offsets fossil fuel emissions. Positive fluxes come from **(c)** land-use change CO_2 emissions from the destruction of vegetation when natural ecosystems are converted to cropland or pasture.

Biogas produced in pyrolysis is able to offset emissions from the burning of natural gas at a rate of 0.015 Mg C GJ^{-1}. Assuming energy yield of 35% the energy content of the parent feedstock, this gives offsets of 0.43–0.52 Pg C yr^{-1} in 2050 (Table 2.5), and supplies 28.6–34.6 EJ yr^{-1} energy (between the low and mean estimates of the IPCC SRREN report) (Table 2.6). Energy generation from BIGCC is assumed to offset primary energy generation at a carbon intensity of 0.017Mg C GJ^{-1}, based on the value for 2000 given by the IPCC (Metz, 2007). With energy yield of 30%, BIGCC supplies 16.6–34.3 EJ yr^{-1} (Table 2.6) and offsets 0.28–0.57 Pg C yr^{-1} in 2050 (Table 2.5). The combined offset fluxes for each scenario, as a function of time are shown as a negative flux (equivalent to CDR) in **Error! Reference source not found.** b.

As others have found, the potential for energy generation from biomass responds strongly to projected trends in diets and land-use (Haberl et al., 2010;

McIntyre, 2009; Metzger et al., 2002; Slade et al., 2014; Smeets et al., 2007). In our scenarios biomass contributes 30.9 – 62.9 $EJyr^{-1}$ in 2050 (Table 2.6); very much in the lower end of the range predicted in the IPCC SRREN report, and lower even than those of similarly constrained studies, presumably at least in part due to our focus on increasing CDR fluxes at the cost of energy yields.

Table 2.5: Mitigation fluxes, land-use change emissions and net CO_2 fluxes in 2050 in the four scenarios.

	High-meat, low-efficiency	**Low-meat, low-efficiency**	**High-meat, high-efficiency**	**Low-meat, high-effciency**
CDR flux from pyrolysis of residues (Pg C yr^{-1})	1.34	1.19	1.27	1.10
Offset from biogas (Pg C yr^{-1})	0.52	0.46	0.51	0.43
Offset from biochar as soil amendment (Pg C-eq yr^{-1})*	0.60 – 1.20	0.54 – 1.07	0.58 – 1.17	0.50 – 0.99
CDR flux from BIGCC+CCS (Pg C yr^{-1})	-	-	1.49	3.09
Offset from electricity generation (Pg C yr^{-1})	-	-	0.28	0.57
Combined mitigation flux (Pg C yr^{-1})	**1.86**	**1.65**	**3.58**	**5.19**
Land-change emissions (Pg C yr^{-1})	4.76	2.36	0	0
Net CO_2 flux**	**2.9**	**0.79**	**-3.58**	**-5.19**

*****Not included in combined mitigation flux as this offset does not operate through C emissions to the atmosphere. **Here positive fluxes are net CO_2 emissions to the atmosphere, negative fluxes are net CO_2 uptake from the atmosphere.

The addition of biochar to soil also has the potential to mitigate against GHG emissions, by reducing fertilizer inputs and N_2O emissions, stabilizing soils and other indirect effects (Shackley et al., 2012). The scale of this mitigation potential is uncertain and difficult to quantify, but it is suggested that combined they could contribute 25–40% of the total mitigation potential of biochar. If this estimate is correct the use of biochar as a soil amendment could offset a further 0.5–1.2 Pg C-eq yr^{-1} in our scenarios (Table 2.5). However, since this potential does not operate through the direct removal of CO_2 from the atmosphere, or

offset of CO_2 emissions, we are unable to include it in our calculation of the effects on atmospheric CO_2.

Table 2.6: Energy generation potential from BIGCC and pyrolysis.

Energy supplied	High-meat, low-efficiency	Low-meat, low-efficiency	High-meat, high-efficiency	Low-meat, high-efficiency
Biogas from pyrolysis (EJ yr^{-1})	34.62	30.86	33.67	28.60
Electricity from BIGCC (EJ yr^{-1})	-	-	16.58	34.28

2.5.3 CO_2 emissions from land-use change

The destruction of vegetation and disturbance to the soil when natural ecosystems are converted to croplands and pasture causes significant CO_2 emissions, currently contributing to a net land-use change flux of 1.1±0.7 Pg C yr^{-1} (Friedlingstein et al., 2010). In fact the emission component of this flux is larger since it is counterbalanced by current afforestation (estimated at 0.21-0.42 Pg C yr^{-1} (Lenton, 2010)) and regrowth of forest on abandoned land (of order ~1 Pg C yr^{-1} (Churkina et al., 2007; Friedlingstein et al., 2010)).

Since there is no spatial element to this work, it is difficult to calculate carbon emissions from land-use change (LUC) with much certainty, as the exact fluxes depend on the type of vegetation being replaced, as well as local climatic conditions and the productivity of the managed vegetation that replaces them. Even with high-resolution spatial data the stocks and fluxes of carbon remain highly uncertain, in some cases with error bars of 75% (Eggleston et al., 2006). Average LUC emissions have been calculated, however, for croplands replacing natural ecosystems in a range of biomes in a study using a spatially explicit dataset (West et al., 2010). In order to produce a ballpark figure for LUC emissions in our scenarios, we are forced to assume that agricultural expansion occurs in exactly equal measure in all types of natural ecosystem (excluding boreal forest and tundra). For the expansion of croplands we then use the average of the values given by West *et al*., (2010)giving average emission of 83.8 tC ha^{-1}, while for expansion of pasture (treated as managed grassland) we apply the IPCC Tier 1 methodology (Eggleston et al., 2006) to the averaged carbon content of all sub-boreal vegetation types, giving average emission of

62.9 tC ha^{-1}. The resulting cumulative carbon emissions 2000-2050 from LUC in our four scenarios are: 254.7 Pg C in high-meat, low-efficiency, 159.3 Pg C in low-meat, low-efficiency, 29.9 Pg C in high-meat, high-efficiency, and 14.3 Pg C in low-meat, high-efficiency. The LUC CO_2 emissions as a function of time are shown in **Error! Reference source not found.** c and the fluxes in 2050 are given in Table 2.5.

We note that the estimated LUC emissions are very high in the low-efficiency scenarios, reflecting their massive expansion of agricultural land. Indeed they match or exceed estimated cumulative historical land-use change emissions 1850-2005 of ~156 Pg C (Houghton, 2008). This is plausible as the future increase in cropland and pasture area is comparable to the historical increase of ~3 Gha since 1850, with low-meat, low-efficiency increasing ~2.5 Gha and high-meat, low-efficiency increasing ~4 Gha. Furthermore such future increases would be biased to the tropics where natural vegetation stores more carbon on average when compared to historical increases in temperature regions. It also fits closely with the estimate that intensification rather than expansion of agriculture since the 1960s has saved emission of 161 Pg C (Burney et al., 2010). Considering the estimated land-use change CO_2 emission flux (**Error! Reference source not found.** c), clearly we are not on either of the low-efficiency trajectories at present and are probably closest to the high-meat, high-efficiency scenario.

2.5.4 Effects on atmospheric CO_2

To calculate the effects of our scenarios on atmospheric CO_2 concentration we follow an approach we have used previously, which captures the behaviour of a global carbon cycle model without having to run it (Lenton and Vaughan, 2009). The key point is that small CO_2 perturbations, whether increases or decreases, decay over time. This is because the atmosphere is continuously exchanging CO_2 with the ocean and the land surface, and additions or removals of CO_2 are shared out between the ocean, atmosphere and land ‘reservoirs’. The fraction of the original perturbation remaining after a given time, t (in years), is called the airborne fraction, $f(\Delta t)$. It is a complex function containing multiple decay timescales, related to multiple land and ocean carbon reservoirs. For small perturbations, it can be approximated, from the Bern carbon cycle model (Joos et al., 1996) by:

$$f(t) = 0.18 + 0.14e^{-t/420} + 0.18e^{-t/70} + 0.24e^{-t/21} + 0.26e^{-t/3.4} \quad \textbf{1}$$

According to this formula, for an instantaneous removal of carbon from (or release to) the atmosphere, 92% is still removed (or present) after 1 year, 64% after 10 years, 34% after 100 years, and 19% after 1000 years. Here we are concerned with relatively short timescales, but still we must take into account the fact that the 'value' of a given removal of CO_2 from the atmosphere 'depreciates' over time, and initially the depreciation is quite rapid. We can treat a reduction in CO_2 emissions to the atmosphere of a given magnitude as equivalent to a negative CDR flux of the same magnitude, because they have identical effects on the net anthropogenic CO_2 flux to the atmosphere – namely reducing it. However, we must add the positive CO_2 emissions due to land-use change to arrive at an overall net flux. What the background scenario of anthropogenic CO_2 emissions, atmospheric CO_2 concentration, and climate change is does not have to be specified, because to first order it does not affect the results. (This assumption begins to break down as climate change increases, but as we are only looking out to 2050 here it is reasonable.)

To make the calculations, the net CO_2 flux (i.e. LUC CO_2 emission minus CDR minus the offset of fossil fuel CO_2 emission) for each year of 2010-2050 is treated as an individual perturbation to atmospheric CO_2. Each of these perturbations is then decayed following equation 1 and the time elapsed. Here we simply implement this in a spreadsheet as the functions describing the perturbations are not of a simple form that would easily allow us to integrate precisely.

The estimated effect on atmospheric CO_2 concentration as a function of time for each scenario is illustrated in Figure 2.8, with the total net effect by 2050 also given. CO_2 emissions from land-use change have a dominant role in the low-efficiency scenarios, with net increases of atmospheric CO_2 from 2000-2050. The two high-efficiency scenarios, however, succeed in achieving net reductions in atmospheric CO_2 of 13.2 and 25.0 ppm by 2050. The greatest potential drawdown of atmospheric CO_2 is in our high-efficiency, low-meat scenario. Whilst 25 ppm might sound modest, it is equivalent to over 10 years of CO_2 rise at the current rate. This is significant, because it is around mid-century that the planet is expected to be approaching the widely-discussed policy

threshold of 2 °C warming above pre-industrial. Shaving off 25 ppm of CO_2, equivalent to ~0.3 W m^{-2} of radiative forcing (at an expected background concentration of ~470 ppm in 2050) could help reduce the risk of exceeding this threshold. Also, half of the estimated drawdown on atmospheric CO_2 kicks-in in the 2040s, consistent with the net CO_2 reduction flux itself growing rapidly in that decade (**Error! Reference source not found.**). This implies that much larger effects on atmospheric CO_2 and hence global temperature are conceivable in the second half of this century, as shown in our previous work (Lenton, 2010).

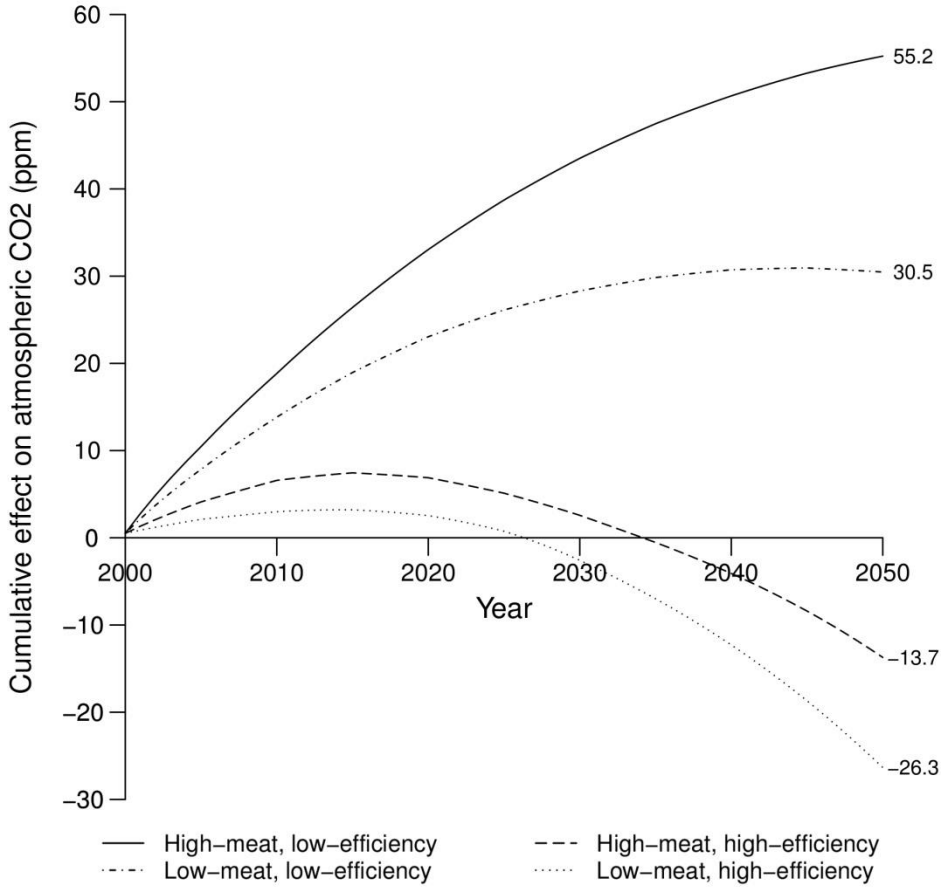

Figure 2.8: Cumulative effect on atmospheric CO_2 of combined mitigation fluxes and LUC emissions in our four scenarios.

2.6 Discussion and further research

Humanity cannot avoid appropriating a growing fraction of global NPP in the coming decades, so the challenge is to make this interaction with the terrestrial biosphere and the global carbon cycle a beneficial one. Our results show clearly that with the current dietary trend of increasing meat consumption, persisting with low-efficiency agricultural systems would be a catastrophe for natural

ecosystems, eliminating the majority of them. It would also make future land-use a major contributor to global CO_2 emissions, with the potential to increase atmospheric CO_2 in 2050 by over 50 ppm, equivalent to an additional ~0.6 W m^{-2} of radiative forcing. Only in our high-efficiency agriculture scenarios can managed land be turned from a carbon source to a carbon sink. Our high-meat, high-efficiency scenario is probably closest to what has actually happened over the last 12 years, with global growth in meat consumption being largely met by pork and poultry rather than beef, and estimated land-use change CO_2 emissions of ~1.8 Pg C yr^{-1} in the 2000s somewhat counterbalanced by forest regrowth and afforestation (which we do not account for here).

Our results can be compared to more detailed, spatial integrated assessments, in particular the new generation of 'Representative Concentration Pathway' (RCP) scenarios developed for the IPCC's 5th Assessment. The closest comparison is between our high-meat, high-efficiency scenario and the most extreme mitigation pathway; RCP2.6 (van Vuuren et al., 2011). In RCP2.6, overall agricultural land area increases to about 5.25 Gha in 2050, driven by an emerging bioenergy sector which grows to around 0.3 Gha by 2050, largely based on abandoned agricultural land. In high-meat, high-efficiency the bioenergy crop area of 332 Mha in 2050 is comparable, although agricultural land area is smaller, implying greater protection or restoration of natural ecosystems. The only RCP scenario with significant expansion of crop and grazing lands is the RCP8.5 storyline, which follows a high fossil fuel emissions pathway (Riahi et al., 2011). Its land-use is in the direction of our high-meat, low-efficiency scenario, but not as severe, with growth in the livestock sector compensated by intensification of grazing. In the middle emissions pathways, the area of farmland is either approximately stable over the next century in RCP6 (Masui et al., 2011), or declines in RCP4.5 (Thomson et al., 2011). In general, our scenarios explore a different dimension to the RCPs, which typically assume good efficiency gains but do not consider e.g. a shift to lower meat diets.

The efficiency of food production clearly has enormous implications for the future of the relationship between humans and the Earth. Grazing and fodder production currently account for around 60% of food related biomass harvest and 78% of agricultural land-use (Krausmann et al., 2008). Since the livestock

sector is the least efficient element of the food system, takes up the largest area of land, appropriates the largest portion of NPP, and generates the biggest residue streams, changes in this sector have the most significant impact. Minimising the land-use by the livestock sector and thus maximising potential bioenergy crops can be achieved in a number of ways; through changes in diet toward eating less meat; through altering the balance of livestock towards more efficient species like pigs and chickens; through replacing grazing with fodder feeding; and through the recycling of residues and food wastes. To illustrate this, we conduct sensitivity analyses varying our assumptions about these key controls and exploring the effects on land-use and cumulative bioenergy crop by 2050 (Figure 2.9).

The sensitivity analysis shows that dietary choice, mix of meat products, and the balance of grazing and fodder feeding can all exert huge leverage on future land-use and bioenergy potential. Also, the extra recycling of the high-efficiency scenarios can increase cumulative bioenergy crop by a similar amount to reducing livestock products to 15% of diet from 2000 levels. If average meat consumption could be reduced toward 10% of daily energy intake this could prevent further expansion of human land-use liberating up to ~1.5 Gha for bioenergy crops totalling ~150 Pg C by 2050. If the efficiency of livestock production could increase toward 6%, which corresponds to a system producing entirely pigs, (with dietary meat consumption at the 2000 level, 16.06% of daily energy) this too could prevent agricultural expansion and produce a comparable ~150 Pg C bioenergy crop. Alternatively, if all animals were fodder fed, up to ~3 Gha could be liberated and ~270 Pg C of bioenergy crop produced by 2050. These are of course unrealistic, extreme scenarios, but they emphasise that demand for meat and the methods of meeting that demand are the crucial determinants of future global land-use and future bioenergy potential (within the constraint of protecting natural ecosystems ahead of bioenergy production).

Intensification of farming, in particular the livestock sector, thus appears to be a necessary condition for global sustainable land-use and carbon cycling. However, we need to consider the broader environmental and animal welfare implications of this. Historically, intensification has led to increasing water demand for irrigation, with farming now responsible for up to 70% of global fresh water consumption (McIntyre, 2009). Increasing nitrogen and phosphorus inputs

as fertiliser have led to freshwater and coastal sea eutrophication and oxygen depletion as a result of nutrient runoff, and high N_2O emissions from soils. Inputs of chemical pesticides and herbicides have also had broader ecological impacts. Now there are concerns that phosphorus and potassium are non-renewable resources with finite geological reserves, and the price of rock phosphate has increased significantly over the last decade.

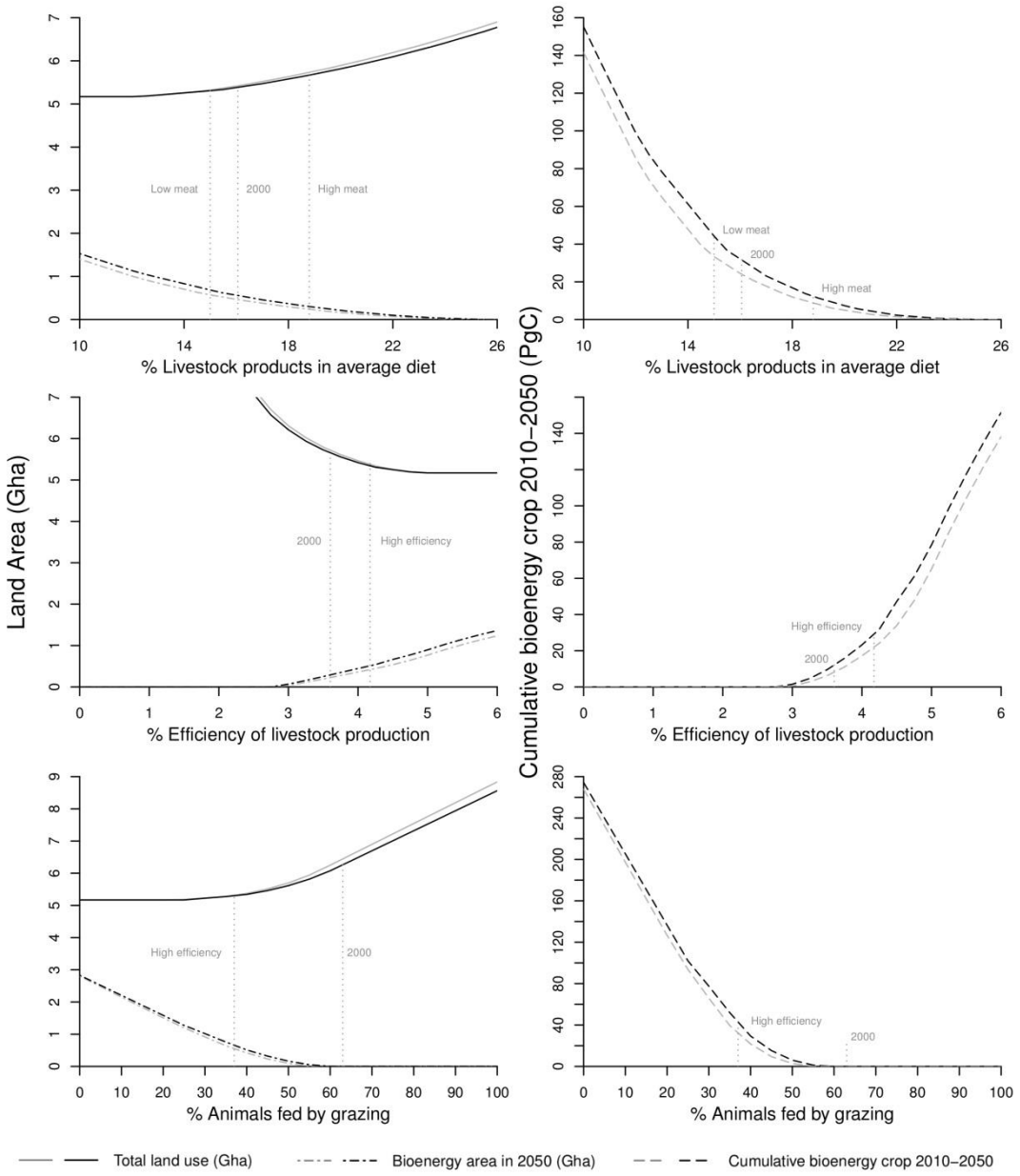

Figure 2.9: Sensitivity analysis showing how the extent and efficiency of livestock production has an enormous influence on **(a,c,e)** land-use and **(b,d,f)** bioenergy production potential. In all cases conditions by 2050 are equivalent to those of the high-efficiency scenario with meat consumption equal to that of 2000 (16.06% daily energy intake), except for the parameter being varied. Dependence on: **(a,b)**% livestock products in average diet; **(c,d)**% efficiency of livestock production (this corresponds to altering the

balance of animal species produced, with the highest efficiency of 6% corresponding to a system producing entirely pigs); **(e,f)**% animals fed by grazing (as opposed to fodder feeding). In all cases, grey lines show the effect of removing the extra recycling that occurs in high-efficiency scenarios.

Thus, a future vision of the agricultural system must be one which can cope with potentially limited and therefore costly supplies. There are also concerns about antibiotics used in intensive livestock production, about the nutritional implications of intensive production, and about the animal welfare implications. However, there are some good sides to the types of intensification we explore. For example, more manure can be collected (and separated from urine) for use either as a biochar feedstock or as fertiliser, thus increasing recycling of waste material and the nutrients it contains, and reducing problems of eutrophication, N_2O emissions, and finite rock phosphate supplies.

Further work should take a yet broader view of the holistic challenge of future land management, including nitrogen, phosphorus and water cycling, and multiple greenhouse gases in the analysis, with the emphasis on methods of intensification which reduce inputs and wastes rather than increasing them, and increase efficiency rather than relying on the economies of scale which have driven much of agricultural technology. The benefits of maximising carbon capture and storage (as we have) need to be weighed against those of maximising bioenergy yield and offsets of fossil fuel CO_2 emissions, together with more accurate estimates of land-use change emissions of CO_2 and other greenhouse gases. What is clear is that we need to develop and globally deploy the lowest input, highest efficiency agricultural systems if we are to prevent a land-use disaster for natural ecosystems, and stand a chance of rebalancing the global carbon cycle.

Chapter 3: Conflict between mitigating climate change and preserving biodiversity

Abstract

We assess the potential for future biodiversity loss due to three interacting factors: energy withdrawal from ecosystems due to biomass harvest, habitat loss due to land-use change, and climate change. We develop four scenarios to 2050 with different combinations of high or low agricultural efficiency and high or low meat diets, and use species–energy and species–area relationships to estimate their effects on biodiversity. In our scenarios, natural ecosystems are protected except when additional land is necessary to fulfil the increasing dietary demands of the global population. Biomass energy with carbon storage (BECS) is used as a means of carbon dioxide removal (CDR) from the atmosphere (and offsetting fossil fuel emissions). BECS is based on waste biomass, with the addition of bio-energy crops only when already managed land is no longer needed for food production.

Forecast biodiversity loss from natural biomes increases by more than a factor of five in going from high to low agricultural efficiency scenarios, due to destruction of productive habitats by the expansion of pasture. Biodiversity loss from energy withdrawal on managed land varies by a factor of two across the scenarios. Biodiversity loss due to climate change varies only modestly across the scenarios. Climate change is lowest in the 'low meat high efficiency' scenario, in which by 2050 around 660 million hectares of pasture are converted to biomass plantation that is used for BECS. However, the resulting withdrawal of energy from managed ecosystems has a large negative impact on biodiversity. Although the effects of energy withdrawal and climate change on biodiversity cannot be directly compared, this suggests that using bio-energy to tackle climate change in order to limit biodiversity loss could instead have the opposite effect.

Declaration of authorship:

This chapter was originally published as:

Powell, T.W.R and Lenton, T.M. (2013) Scenarios for future biodiversity loss due to multiple drivers reveal conflict between mitigating climate change and preserving biodiversity. *Environmental Research Letters*, **8**, 025024.

DOI: 10.188/1748-9326/8/2/025024

All scenario construction and biodiversity loss modelling was the work of TP, with TL providing the baseline C emissions scenario and carrying out modelling to determine effects of CDR scenarios on temperature. Paper was written by TP, with TL providing feedback and editorial guidance.

3.1 Introduction

Biodiversity is inextricably linked with human wellbeing; through its contribution to the functioning and resilience of ecosystems; as a resource with the potential for discovery of new compounds or processes; and a source of emotional wellbeing (Díaz et al., 2006). Natural ecosystems provide resources including food, fresh water, fibre etc., and biodiversity loss affects the ability of ecosystems to fulfil these roles. This in turn affects poorest communities the most, since they are least able to afford to substitute the lost ecosystem services.

At the same time, access to proper nutrition is fundamental to wellbeing. Around 1 billion people are currently in food poverty, and the global population is forecast to grow from the current ~7 billion to ~9.3 billion people in 2050. Providing everyone with an adequate diet is seen as one of the greatest challenges for human wellbeing in the coming decades. Yet the loss and fragmentation of natural habitats due to agricultural expansion has been a major cause of biodiversity loss to date.

Climate change also affects human wellbeing, for example through more extreme weather events, changing distributions of disease vectors, and forced migration, as well as being a key driver of biodiversity loss. Climate change is already removing unique habitats and contracting others faster than some species can disperse, and it will become an increasingly important driver of biodiversity loss in the future (Thomas et al., 2004a); indeed one of the key motivations for tackling climate change is to protect natural ecosystems and the goods and services they provide.

The joint pressures of climate change and expansion of agriculture set up a potential conflict between elements of human wellbeing; meeting food demand and improving diets requires the growth of agriculture, which in turn causes biodiversity loss and contributes to climate change. The use of bio-energy crops, in particular, is put in a highly ambiguous position among these interactions. They have the potential to mitigate climate change, but may also compete with food production, cause the destruction of natural habitats, and even cause CO_2 emissions as a result of land-use change.

Here we take an integrated view of global terrestrial biodiversity loss, seeking to quantify the multiple effects of different future land-use scenarios, and examining the interactions between different drivers. In particular, agriculture withdraws energy from ecosystems to feed us, is by far the largest driver of land-use change, and contributes around 30% of greenhouse gas emissions at present. Our scenarios (Chapter 2) are driven by increasing population and calorific intake, changing dietary demand for animal products, changes in the efficiency of food production, and a drive to use biomass to mitigate climate change, where it does not conflict with food production or the preservation of remaining natural ecosystems. In particular, we consider the potential for biomass energy with carbon capture and storage (BECCS) to remove carbon dioxide from the atmosphere and offset fossil fuel emissions. We focus on the use of waste biomass for BECS, only allowing dedicated bio-energy crops where land becomes abandoned from agriculture.

Our approach to estimating biodiversity loss follows previous work in using concepts from ecological theory, namely the species–area and species–energy relationships. The species–area relationship is a classic tenet of ecology describing a (non-linear) correlation between increasing area and the number of species to be found. The converse formula has been widely used to predict extinction rates from habitat loss (Pimm and Raven, 2000). Estimates of biodiversity loss from range shifts due to climate change also make use of the species–area relationship (Bakkenes et al., 2002; Thuiller et al., 2004), although only one study has attempted a quantitative global assessment (Thomas et al., 2004a).

Less widely recognized is that the withdrawal of energy from ecosystems, through the harvesting of biomass—for food, fibre, wood products or energy—also causes biodiversity loss. The corresponding species–energy relationship describes the correlation between species diversity and the energy available to organisms at a given spatial resolution (Gaston, 2000; Wright, 1983). While at a local scale the very highest levels of energy availability may lead to competitive dominance of a relatively low number of species causing a peaked distribution, at larger spatial scales there is generally a monotonic positive correlation between energy availability and diversity (Chase and Leibold, 2002; Evans et al., 2005; Mittelbach et al., 2001).

The species–energy relationship appears to be driven by high species occupancy at higher energy levels, with greater availability of energy allowing greater coexistence of species by providing a wider range and complexity of niches in space, time and in the number of possible community compositions (Bonn et al., 2004; Chase and Leibold, 2002; Jetz and Fine, 2012). Higher productivity may also lead to more species by increasing the probability of occurrence of resources that enable the persistence of viable populations (Storch et al., 2005). Species that occupy lower energy levels tend to be generalists with large ranges that are also present when more energy is available (except, of course, for those that specialize in extremely low energy environments), while higher energy levels support a greater range of specialists with small ranges (Bonn et al., 2004). Higher energy biomes, e.g. in the tropics, are therefore more sensitive to reductions in energy availability.

The withdrawal of energy from ecosystems can be quantified in terms of the human appropriation of net primary production (HANPP). This is defined as the combined effects of anthropogenic changes in productivity, and harvest of biomass, on the availability of NPP in ecosystems (Haberl et al., 2007). HANPP affects biodiversity via the species–energy relationship, since the anthropogenic removal of NPP constitutes a reduction in the energy available to other organisms.

Positive correlations have frequently been observed between intensity of agriculture and biodiversity loss, with the drivers being a combination of landscape effects, intensity of inputs and intensity of biomass extraction (Eglington and Pearce-Higgins, 2012; Haberl et al., 2004; Kleijn et al., 2009). These drivers are not independent of one another, and as such HANPP as well as being a cause of biodiversity loss in itself, may be considered a proxy for the other elements of agriculture which have a negative impact on biodiversity, including excessive nutrient application, chemical pollution and the creation of relatively homogeneous agro-ecosystems.

Very few studies have however characterized the impact of agriculture on biodiversity in terms of HANPP and the species–energy relationship (Haberl et al., 2005, 2004; Wright, 1990). Among these are observational studies of the relationship in several phylogenetic groups on Austrian farmland (Haberl et al., 2005, 2004). One pioneering study used a generalized species–energy

relationship to predict global species endangerment from 1990 to 2000 based on expected increase in food demand due to population growth (Wright, 1990), but did not include dietary trends.

Extant flows of biomass energy can be interpreted in terms of demand driven by population and demographics (Krausmann et al., 2008), and future demand can be forecast(Chapter 2). Such material and energy-flows analyses are powerful tools for the description of human impact on ecological energy and carbon fluxes, linked to the concept of a socio-ecological metabolism (Fischer-Kowalski and Haberl, 2007; Haberl, 2006). This framework describes the physical relationship between human society and our environment in terms of inputs and outputs of energy and resources, and provides a common accounting method for the intimately linked carbon and energy cycles of humanity and the Earth system as a whole.

3.2 Methods

3.2.1 Biomass flows model

We use a simple model of biomass flows and land-use in the global agricultural system, developed from a previous study of the carbon dioxide removal (CDR) potential from biomass energy generation with carbon storage (BECS) (Chapter 2). The model uses the predicted growth in human population to 9.3 bn in 2050 combined with an expected rise in average daily calorific consumption of ~20% (Smeets et al., 2007) to forecast the increase in global food demand to 2050. Conversion factors derived from literature sources (Krausmann et al., 2008; Wirsenius, 2003) and the FAOSTAT database are used to calculate the biomass harvest required to meet food demand each year, from a combination of primary crops and livestock fed by grazing or from market feed. Projected biomass harvest for fibre and forestry products are also included, the former driven by population growth and the latter based on FAO projections (FAO, 2010).

The demand for biomass harvest is combined with average yield data for primary crops, fodder crops and pasture, as well as non-food products, to calculate the land area required for each. Average crop yields are assumed to increase 1% annually, which in this biomass flows approach could be met either by increasing actual yields, or by reduction of the ‘yield gap’ that exists due to

environmental and management factors. A 10% increase in stocking intensity on pasture is also assumed between 2000 and 2050. We assume no increase in yields from forestry.

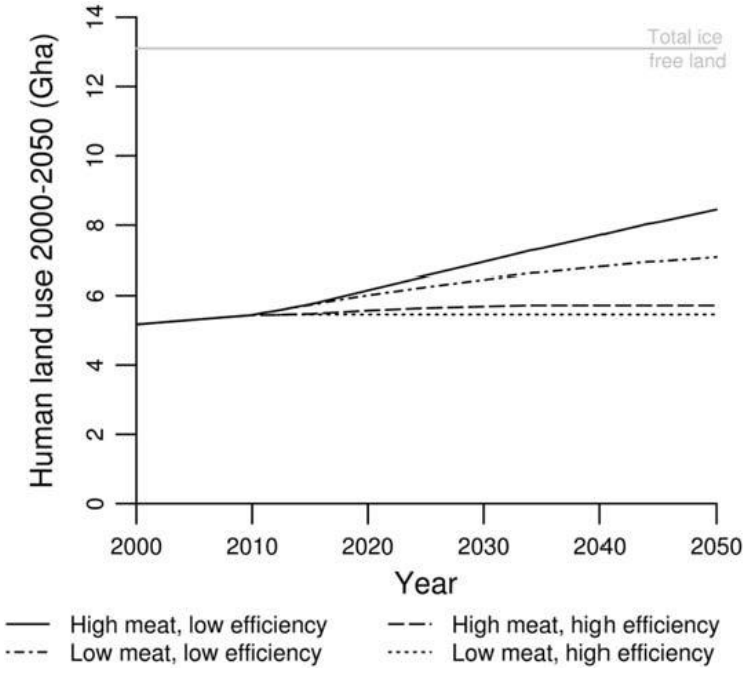

Figure 3.1: Projections of total global human land-use 2000-50 in our four scenarios.

Where possible, expansion of increasing land-use types is met by decreases in land-use of other sectors, with a hierarchy of allocation from food crops down to bio-energy crops. If, in a given time-step, the overall area under management is required to expand (Figure 3.1), land is appropriated from natural ecosystems. These are divided into five classes varying in carbon stocks based on IPCC guidelines for greenhouse gas inventories, and on data for above-ground net primary productivity (NPP) (Haberl et al., 2007). These five biome groups approximately equate to; tropical rainforest; deciduous forest; grassland and savanna; boreal forest; and desert and tundra (Table 3.1).

Expansion of croplands displaces pasture, which in turn appropriates land from the three most productive categories (tropical rainforest; temperate deciduous forest; grassland and savanna) in a ratio of 1:2:2, as long as 15% of each class is preserved. Managed forests also expand into natural biomes unless accommodated by shrinkage of other land-uses. Excess land requirement is taken up by the other classes or if necessary the next most productive unused class. This reflects the historical tendency for agricultural land to occupy the

most productive land types (Ramankutty and Foley, 1999). Indeed 80% of the expansion of cropland since the 1980s has been in the tropics (Gibbs et al., 2010).

Table 3.1: Classification of natural biomes.

Approximate biome type	**NPP** (t C ha^{-1} yr^{-1})	**Mean NPP** (t C ha^{-1} yr^{-1})	**Above ground carbon stock** (t C ha^{-1})	**Area** (Mha)
'Desert/Tundra'	0 – 2.5	0.6	2.9	1675.5
'Boreal Forest'	2.5 – 5.0	3.8	20.8	2060.7
'Savanna/Grassland'	5.0 – 7.5	5.8	5.2	983.4
'Deciduous Forest'	7.5 – 10.0	9.0	85.4	389.3
'Tropical Forest'	> 10.0	10.9	164.4	348.2

Based on calculations from datasets published by Haberl et al. (2007).

Of the four scenarios (Table 3.2), two meet the expected rise in the per-capita consumption of animal products from 16.06% of daily energetic intake in 2000 to 18.8% in 2050 (Smeets et al., 2007). Two alternative 'low meat' scenarios imagine a deliberate, or enforced, reduction in consumption of animal products to 15% of daily calorific intake by 2050. The scenarios are further separated into high and low agricultural efficiency variants. The high efficiency scenarios follow trends in the livestock system towards a greater contribution from pig meat, poultry and eggs, which are up to a factor of ten more efficient at converting primary biomass to food than are ruminants (Wirsenius, 2003). Indeed they are even more efficient in terms of land area since they can be fed on high quality feed crops rather than from extensive pasture (although this means that pigs and poultry compete directly with humans for primary crops). These scenarios are further focused on efficiency of land-use by an increase in the use of fodder crops rather than pasture as feed for ruminants, representing intensification of ruminant farming systems. This in turn allows a higher proportion of collection and recycling of manure. High efficiency scenarios also see increasing recycling of residues and reduction of food waste. In the low agricultural efficiency scenarios, efficiency gains slow after 2010 and cease altogether after 2015.

Table 3.2: Details of four scenarios.

	2000	2050			
		High meat, low efficiency	**Low meat, low efficiency**	**High meat, high efficiency**	**Low meat, high efficiency**
Population	6.12 bn	9.31 bn	9.31 bn	9.31 bn	9.31 bn
Average diet (Kcal/ca/day)	2760	3302	3302	3302	3302
Contribution of livestock products to daily calories	16.06%	18.8%	15%	18.8%	15%
Proportion of livestock calories from:	10.5%				
Ruminants	38.9%	10.5%	10.5%	5.2%	5.2%
Dairy	30.4%	38.9%	38.9%	19.4%	19.4%
Pig meat	11.7%	30.4%	30.4%	45.4%	45.4%
Poultry	8.5%	11.7%	11.7%	17.4%	17.4%
Eggs		8.5%	8.5%	12.7%	12.7%
Increase in stocking density 2000-2050	-	10%	10%	10%	10%
Livestock feed from fodder crops (%)	37.0%	39.7%	39.7%	50%	50%
Annual crop yield growth	-	1%	1%	1%	1%
Proportion of primary crop residues 'used'	66%	66%	66%	71%	71%
Reduction in food waste	-	0%	0%	50%	50%

as actively generating net negative carbon fluxes. All scenarios see an initial increase in land needed to meet food demand, but in those that see a subsequent decline in land-use area any land abandoned by food production is considered suitable for growing lignocellulosic biomass crops, which are fed into BECCS schemes. Perennial lignocellulosic crops such as *Miscanthus* sp. have

been shown to produce viable yields even on low quality, degraded land (Liu et al., 2012). In all of the scenarios here, BECS activity begins in 2010 and increases to full capacity in 2030.

The four scenarios thus include three drivers of terrestrial biodiversity loss; biomass harvest; conversion of natural habitats; and climate change, which we now detail.

3.2.2 Biomass harvest on managed land

Here we use %HANPP, defined as the harvested biomass as a percentage of the above-ground NPP of the potential vegetation before human influence (NPP_0), to predict the effect of intensifying and expanding agriculture on species richness. At each yearly time-step, %HANPP is calculated for each of seven main land-use types on actively managed land, and their relative areas are used to calculate a weighted average for %HANPP on managed land (Figure 3.2). This is thus affected by the intensity of agriculture (which tends to increase as yields rise and stocking densities increase), the relative demand for the different land-uses, and the NPP_0 of the land under use.

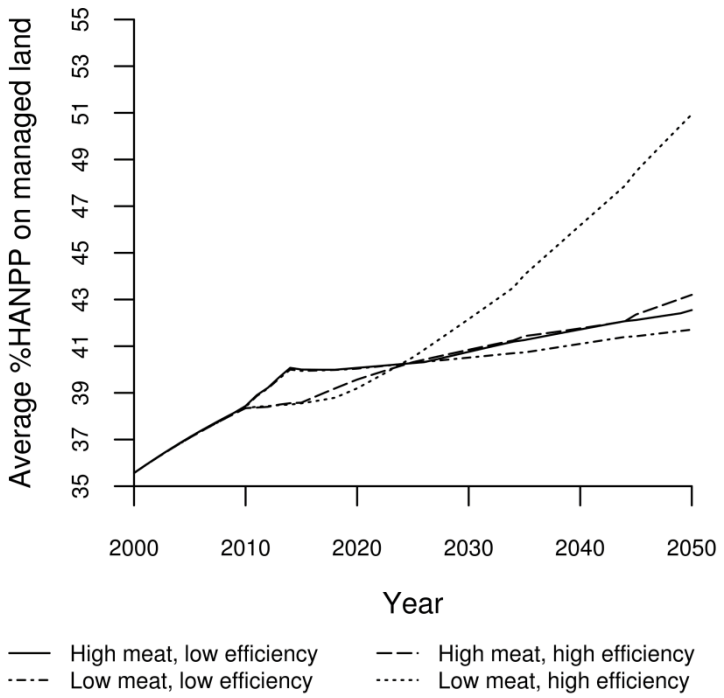

Figure 3.2: Projections of average %HANPP on managed land in our four scenarios.

As land-use demand changes within the categories of land already under management, managed land is redistributed according to an order of priority

with food crops the highest and bio-energy crops the lowest. Bio-energy crops are only allowed when overall human land-use would otherwise decrease, i.e. land is being abandoned from food production. In some cases, the local intensity of biomass harvest decreases, for example if land-used for growing crops is converted to pasture. Such decreases in %HANPP cannot lead to net increases in biodiversity, since an increase in global biodiversity can only be caused by speciation events. However, local easing of biomass harvest is considered to provide a buffer for loss of diversity elsewhere, by providing potential habitat for displaced species, and as such is included in the weighted average.

A species–energy relationship is used to estimate the effect of %HANPP on biodiversity. This is derived from an empirical study of the relationship between HANPP and biodiversity on managed lands in Austria (Haberl et al., 2004). This study is to our knowledge the only attempt to quantify the relationship in terms of HANPP, and does so across croplands, pasture, managed forests and urban environments, and across multiple taxonomic groups. Species richness (S) declines with %HANPP according to:

$$\log(S) = a + b \cdot \log(\%\text{HANPP}) \qquad \textbf{2}$$

where a is a constant, $b = -1.6$ for autotroph diversity and $b = -1.1$ for heterotroph diversity. We assume equal weighting of autotrophs and heterotrophs, combining them to give a total average effect on biodiversity. We note that species–energy curves can vary significantly among taxonomic groups, biota and spatial scales, and as such these curves derived from typical Austrian flora and fauna may not be representative of those across the world's agricultural systems.

3.2.3 Habitat loss via land-use change

The species–area relationship is used to determine the proportion of affected species on each natural land class as agriculture expands:

$$S = cA^{z} \qquad \textbf{3}$$

where S is species richness, c is a constant and $z = 0.25$ (Pimm and Raven, 2000; Rosenzweig, 1995; Thomas et al., 2004a). A weighted average for species loss from natural ecosystems is produced according to their relative

areas (A), and their differing species richness. Here we use a typical species–energy curve (Rosenzweig, 1995; Wright, 1990) to estimate the natural gradient in species diversity from less productive to more productive biomes:

$$S = dE^{z} \quad \textbf{4}$$

where S is species richness, d is a constant, E is energy taken to be NPP, and z = 0.5, reflecting the steeper curves usually associated with continental to global-scale trends (Mittelbach et al., 2001; Wright, 1983). This gives around an eight-fold variation in diversity between the most and least productive biome types.

Since this area is not destroyed as agriculture expands, merely transferred to a managed land-use type, expansion could be assumed to cause a corresponding increase in the species represented on managed land via the species–area relationship. We assume, however, that since the species present in low energy habitats tend to be generalists also found in high energy habitats (Bonn et al., 2004), any species surviving the transition are likely to be generalist species already present in managed environments. Our method for estimating global species loss shows low sensitivity to this assumption.

In addition to the area of habitat lost, we also account for the effect of differences in the productivity of appropriated land. At each time-step the remaining areas of each natural land class are used to calculate a weighted average NPP for unmanaged land. The resulting negative trend in remaining natural NPP, as high productivity land is favoured for agriculture, is then treated as further HANPP, and a species–energy curve applied to determine the associated species loss.

Habitat loss is thus defined by changes in area of habitat types, and the relative diversity of biomes in which habitat loss occurs.

3.2.4 Climate change

To calculate the consequences of our four scenarios on atmospheric CO2 and climate change, we use a simple Earth system model (Lenton, 2000; Vaughan and Lenton, 2012). The model is forced from 1800 to year 2000 with historical estimates of fossil fuel emissions (Boden et al., 2012) and land-use change emissions (Houghton, 2008), predicting 369.4 ppm CO_2 in year 2000 and global

warming of 0.89°C (from 1800) in good agreement with observations (Vaughan and Lenton, 2012).

From 2000 onwards, atmospheric CO_2 and global temperature change are determined by the combination of a common 'baseline' fossil fuel emissions scenario to which each of our four scenarios are added. Although absolute CO_2 and temperature change depends on the choice of baseline future fossil fuel emissions scenario, deviations from that baseline due to the scenarios are insensitive to the choice of baseline—as expected from theory (Lenton and Vaughan, 2009).

For our 'baseline' fossil fuel emissions after 2000 we follow an existing mitigation scenario (Vaughan and Lenton, 2012), using estimated fossil fuel (plus cement production) emissions for 2000–05 (Boden et al., 2012), followed by a 1.7% yr^{-1} increase from 2005 to 2015 (the long term mean growth rate over the last 25 years), after which mitigation activity begins in earnest and it takes 40 years to transition to a 1.7% yr^{-1} decrease in emissions. This scenario gives peak fossil fuel emissions of 11.35 Pg C y^{-1} in 2035, declining to 10.3 Pg C yr^{-1} in 2050.

Each scenario has contributions to the global CO_2 balance from land-use change emissions, carbon dioxide removal (CDR), and offsets of fossil fuel emissions by bio-energy. These three components are added together to get an overall CO_2 flux, either to or from the atmosphere, at each time-step. CO_2 emissions from land-use change are calculated using the IPCC tier 1 methodology, according to the carbon stocks of the vegetation on each land class and the land-use replacing it (Eggleston et al., 2006). CDR and offsets are calculated as described elsewhere(Chapter 2).

The combined flux is initially dominated by land-use change and is therefore a net CO_2 source to the atmosphere. It is identical in all four scenarios up to 2010, declining from 1.77 Pg C yr^{-1} in 2000 to 1.27 Pg C yr^{-1} in 2010, which is consistent with estimates that land-use change emissions declined markedly over that decade (Friedlingstein et al., 2010). Our land-use change emissions are above the estimated mean but well within the error range (Friedlingstein et al., 2010). In previous work (Vaughan and Lenton, 2012), a lower estimate of

land-use change emissions was used for 2000–05 (Houghton, 2008), but the absolute values agree well in 2005.

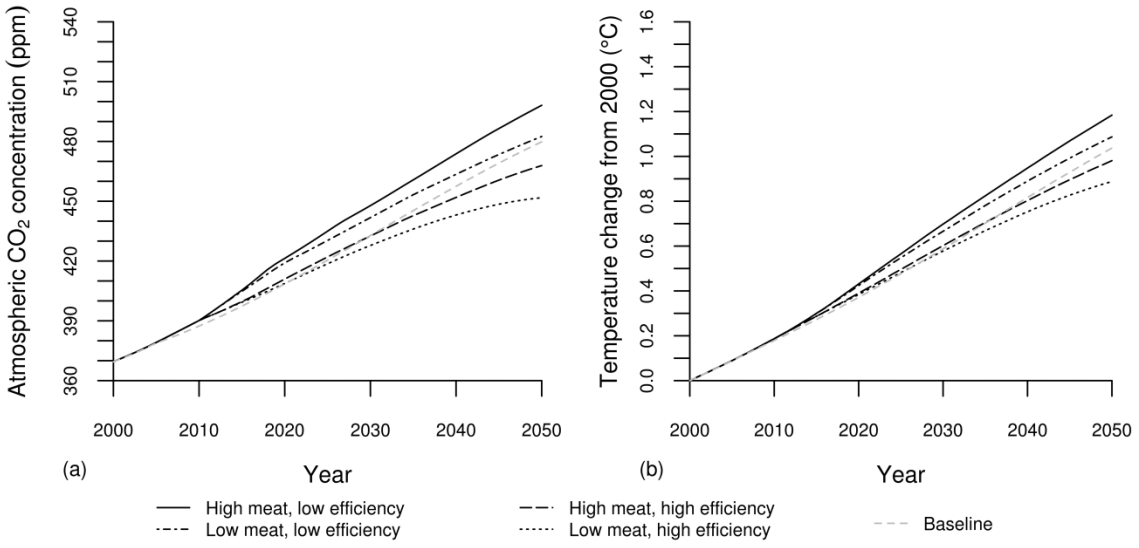

Figure 3.3: Calculated changes in **a)** atmospheric CO_2 concentration, and **b)** global temperature under our four scenarios, together with the baseline changes due to fossil fuel emissions only, which are the same in all scenarios.

Atmospheric CO_2 varies by ~50 ppm in 2050 from 452 ppm in the low meat high efficiency scenario to 498 ppm in the high meat low efficiency scenario (Figure 3.3 a). The corresponding global temperature range in 2050 is 0.29°C, from 0.89 °C to 1.18 °C warming above 2000 (Figure 3.3 b).

We base the sensitivity of terrestrial biodiversity to climate change on previous results (Thomas et al., 2004a), which use bioclimatic modelling of range shifts coupled with the species–area relationship to estimate overall species committed to extinction under a number of climate model projections. We fit a linear regression through their results, having found no significant difference between the fit of linear and polynomial models, producing a sensitivity of around 12.6% species loss per degree of climate change. This is between their two scenarios of ‘no dispersal’ and ‘full dispersal’, and we note that in reality dispersal is likely to be limited by anthropogenic land-use and habitat fragmentation, but is unlikely to be prevented entirely. We then use this value to forecast loss of terrestrial biodiversity due to temperature changes relative to the year 2000.

3.2.5 Sensitivity analysis

To analyse the sensitivity of our results to variation in the strength of the relationships driving biodiversity loss, we tested the effect of varying the key coefficients in each relationship on the final biodiversity loss caused by the relevant driver. In the case of the species–energy relationship on managed land (equation 2) and the species–area relationship (equation 3) this meant varying the pertinent coefficients (*b* or *z*) up to ±25% from the value used in our study, approximately spanning the range commonly found in the literature (Wright, 1990; Rosenzweig, 1995; Pimm and Raven, 2000; Thomas et al., 2004a). Since the gradient of the species–energy relationship appears to increase over larger spatial scales, and we were concerned about under-estimating the range in diversity across biomes, we varied *z* in equation 4 between 0.25 and 1.5, producing up to 100 fold differences in diversity between the most and least productive natural biomes. For the effect of climate change, sensitivity of the results was measured for a variation of ±50% of the climate change sensitivity coefficient, spanning the range of outcomes given by Thomas et al. (2004a). We thus obtained maximum and minimum variants for our forecasts of biodiversity loss from each scenario.

3.3 Results

We divide our forecasts of committed terrestrial biodiversity loss in 2050 from 2000 levels into three components (Figure 3.4); the effects of energy withdrawal due to biomass harvest on managed land, the effects of habitat loss due to land-use change from natural biomes, and the effects of climate change on all land types. Whilst the first two contributors to biodiversity loss could in theory be added up (because they refer to mutually exclusive fractions of the land surface) we have not done so as we are not able to appropriately weight the differences in diversity between managed land and natural biomes. Climate change as a cause of biodiversity loss cannot be considered additive to the other two contributors, because the same species may be vulnerable to both climate change and either energy withdrawal from managed lands or habitat loss from natural biomes. Conversely there are potential synergies between the effects of biomass harvest, habitat loss and climate change, which could make their effects on biodiversity greater than additive (Sala et al., 2000; Brook et al., 2008; Eglington and Pearce-Higgins, 2012).

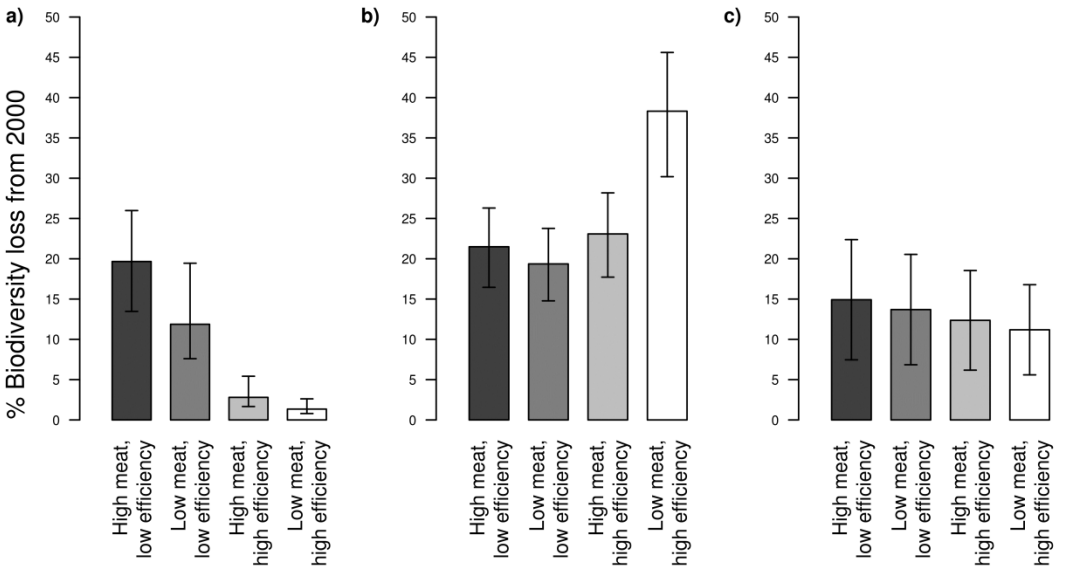

Figure 3.4: Forecasts of global terrestrial biodiversity loss 2000-50 under our four scenarios: **a)** in natural biomes due to habitat loss from land-use change, **b)** in managed land due to energy withdrawal in biomass harvest, and **c)** on all land types due to climate change. Error bars show the range produced by the sensitivity analysis.

Looking across our four scenarios, using our default parameter settings, forecast committed biodiversity loss in 2050 due to habitat loss from natural biomes ranges over 1–20% (Figure 3.4 a), that due to biomass harvest on managed land ranges over 19–38% (Figure 3.4 b), and that due to climate change on all land types ranges over 11–15% (Figure 3.4 c). Thus, the effect of variation across the four scenarios on biodiversity loss is largest for habitat loss due to land-use change and smallest for climate change. The effect of habitat loss on biodiversity (Figure 3.4 a) is largest in the high meat, low efficiency scenario and smallest in the low meat, high efficiency scenario. However, the effect of energy withdrawal on biodiversity (Figure 3.4 b) is greatest in the low meat, high efficiency scenario and comparable in the other three scenarios. The effect of climate change on biodiversity (Figure 3.4 c) is greatest in the high meat, low efficiency scenario and smallest in the low meat, high efficiency scenario, but variation between the scenarios is low.

Since the results for managed land and natural biomes refer to mutually exclusive sets of species, with potentially very different levels of diversity, and results for climate change driven biodiversity loss refer to all terrestrial species, no quantitative comparison can be made between the three sets of results. It is clear however, that 1% loss of diversity from productive natural biomes

represents a higher proportion of global biodiversity than the equivalent loss on human-dominated land. Furthermore, species at risk from climate change are likely to be specialized organisms with narrow niche-spaces, and therefore likely live in higher energy natural biomes (Thomas et al., 2004a; Storch et al., 2005), making biodiversity loss from natural ecosystems of particular significance.

Despite being unable to directly compare the consequences of the three drivers of biodiversity loss, the capacity of the four different scenarios to affect biodiversity in different ways is clear, and the differing ranges and patterns of each set of results allows us to draw some conclusions.

There is a clear difference in effect on habitat loss between our high and low efficiency scenarios, with biodiversity loss from natural biomes of 12.0–20.0% in low efficiency scenarios and only 1.3–2.8% in high efficiency scenarios (Figure 3.4 a). This is driven by the huge requirement for land in the low intensity, low efficiency livestock systems, which appropriate vast areas of productive natural biomes, leading to an increase in total human land-use from 5.17 Gha in 2000 to 8.45 Gha in the high meat variant and 7.08 Gha with a 'low meat' diet by 2050 (Figure 3.1). The preferential use of more productive ecosystem types for the expansion of agriculture increases this pressure, indeed in the high meat, low efficiency scenario the two most productive land classes are reduced as far as allowed in the model, leaving only 15% of their year 2000 area standing. In the high efficiency scenarios, very little growth in agricultural area is required, human land-use reaching its maximum in 2034 at 5.70 Gha under the forecast 'high meat' diet and 5.46 Gha in 2014 under 'low meat' (Figure 3.1). As a consequence, biodiversity loss from natural biomes is low. These results are clearly sensitive to the parameters assumed for the relationships used to produce them, the largest variations in the sensitivity analysis producing 2–3.5 fold variation in estimates of biodiversity loss - however, the effect of habitat loss on biodiversity if agricultural efficiency cannot be increased in future still stands out.

The intensification of biomass harvest required by high efficiency agriculture may carry some biodiversity cost (Figure 3.4 b), but this is relatively small. All scenarios see a significant loss of diversity from agricultural land due to increasing intensity of harvests from 2000 to 2050. The 'high meat, low efficiency' sees a biodiversity loss of 21.5%; 'low meat, low efficiency' a loss of

19.4%; and ‘high meat, high efficiency’ a loss of 23.1%, due to withdrawal of energy on managed land. Biodiversity losses are somewhat lower in the low efficiency scenarios, because although they require a huge land area the bulk of this is made up of pasture with relatively low biomass removal and relatively high diversity, leading to %HANPP of around 25–30%, as opposed to 65–85% for the fodder crops used more extensively in high efficiency scenarios. Increased use of fodder crops and higher stocking intensities in the high efficiency scenarios mean that, although biomass harvest is lower overall - as a result of reduced contribution of inefficient ruminants to livestock products, greater recycling and reduced food waste - it is more concentrated, with average %HANPP on managed land of 43.2% by 2050 in the ‘high meat, high efficiency’ scenario. However, the resulting increase in biodiversity loss on managed land is modest when compared to the huge reduction in species loss from natural biomes caused by this concentration of farming. By far the largest biodiversity loss due to biomass harvest occurs in the ‘low meat, high efficiency’ scenario because the roughly 660 Mha reduction in land required for food production allows for the conversion of pasture to a correspondingly large swathe of bio-energy plantation. The increase in %HANPP on this area from around 25% for pasture to almost 90% for bio-energy plantation contributes to a 38.3% species loss from managed land in this scenario.

The effect of climate change on biodiversity loss varies least across the four scenarios (Figure 3.4 c) primarily because the same baseline climate change of ~1 °C from 2000 to 2050 due to fossil fuel emissions occurs in all four scenarios. There is only a ~0.3 °C range in 2050 global warming across the scenarios. Climate change is least in the ‘low meat, high efficiency’ scenario thanks largely to the aforementioned bio-energy plantations, supporting a net carbon dioxide removal flux of ~5 Pg C yr^{-1} in 2050. However, inertia in the carbon cycle and the climate system means that the full climatic effect of this activity would not be felt until later. Climate change is greatest in the ‘high meat, low efficiency’ scenario, due primarily to CO_2 emissions from land-use change, however even in this scenario the conversion of biomass wastes to stored carbon is able to reduce the net flux of CO_2 from the land surface to close to zero in 2050.

The absolute percentage values of biodiversity loss forecast here should be viewed as highly uncertain, with many caveats accompanying them. The globalization of species–area or species–energy relationships has many potential flaws, several of which are highlighted by previous debate surrounding the estimated effect of climate change on biodiversity loss (Buckley and Roughgarden, 2004; Harte et al., 2004; Thomas et al., 2004b; Thuiller et al., 2004). Possible interactions between the species–area and *species–energy relationships are not considered* (Storch et al., 2005). Potential synergies between the effects of habitat loss and fragmentation, biomass harvest and climate change on biodiversity are ignored (Sala et al., 2000; Brook et al., 2008; Eglington and Pearce-Higgins, 2012). Our approach also fails to account for diverse management strategies according to regions, cultures, socio-economic conditions and history (Erb et al., 2012).

Despite these caveats, the relative effects of different drivers on biodiversity loss include some striking results that are robust to our sensitivity analysis. In particular, using bio-energy with carbon capture and storage to tackle climate change is likely to have a greater negative effect on biodiversity due to energy withdrawal from ecosystems, than the avoided biodiversity loss due to less climate change.

3.4 Conclusion

Although the absolute figures are highly uncertain, we are able to draw some tentative conclusions about the relative importance of different drivers of biodiversity loss. First, if current trends of increasing agricultural efficiency and intensification are not maintained, meeting food demand by expanding agricultural land area could become the dominant cause of future biodiversity loss, via the destruction of productive, and therefore species rich, natural habitat, with additional losses from the harvesting of biomass energy and the climatic consequences of land-use change CO_2 emissions. Second, although continued increases in agricultural efficiency could liberate land for dedicated bio-energy crops, the resulting withdrawal of energy from managed land would likely have a much greater negative impact on biodiversity than the positive effect of reducing climate change. Clearly biodiversity loss is not the sole reason for mitigating climate change, but it is a significant one, so this result

highlights a potentially important contradiction implicit in many climate change mitigation strategies.

Chapter 4: Development of the FALAFEL model

Abstract

Inadequacies in the biomass flows model used in the previous two chapters prompted a complete redevelopment of the model. This chapter describes the inputs and structure of a new version, referred to as the **F**lux **A**ssessment of **L**inked **A**gricultural **F**ood production, **E**nergy potentials & **L**and-use change (FALAFEL) model. FALAFEL is a significant upgrade of its predecessor, with significant disaggregation of the diet, crop yield, livestock management and waste-stream components in particular. Plant derived food products are divided into a mixture of the most important individual crops (e.g. *wheat*, *rice, maize*, *soybean*), and some aggregate groups (e.g. *other cereals, other oilseeds*). Livestock products are likewise divided into *bovine meat*, *pig meat*, *poultry meat*, *other meat*, *dairy*, *eggs* and *animal fats*. Long-term trends are derived from time-series datasets to describe the contribution of these groups to the average diet. Yield trends for each group are also derived from average annual yield data covering a 40 year period, and used to drive land-use demand.

Declaration of authorship:

I was originally prompted to begin rebuilding the model during a collaboration with Dr Anna Stephenson of the UK Government Department of Energy and Climate Change, on the Bioenergy Emissions and Counterfactuals (BEAC) model. The part of the BEAC model responsible for calculating land-use to meet food demand was based on my original biomass flows model, but AS adapted the structure and inputs by disaggregating the food groups and including nutritional information. Because of this, some elements of the structure of the BEAC model were incorporated into the design of FALAFEL, some of its skeleton is attributable to AS.

4.1 Introduction

During the attempts at using the biomass flows forecasting approach in the studies described in Chapters 2&3, several frustrating weaknesses became apparent in the simplistic model. The development of the biomass-flows modelling tool to overcome these has resulted in a new version referred to as the **F**lux **A**ssessment of **L**inked **A**gricultural **F**ood production, **E**nergy potentials & **L**and-use change (FALAFEL) model. Some of the improvements described in this chapter were also prompted in a significant way by collaborative work on the Bioenergy Emissions and Counterfactual Model (BEAC) (Stephenson and MacKay, 2014), with the UK Government Department of Energy and Climate Change (DECC), and indeed the structure of some components of FALAFEL is still based on elements of the BEAC.

Most significant among the changes is disaggregation of human dietary trends beyond the simplistic 'animal or vegetable' approach. Dividing these categories into key crop and animal product subgroups allows a more nuanced examination of the implications of current dietary trends, while giving the same treatment to sources of feed for livestock animals opens up further possibilities for exploring one of the largest consumers of biomass resources. Also crucial to this was internal calculations of significant trends in crop production, diets, yields and livestock characteristics based on a 40 year time-series of data, largely from FAOSTAT inventories. This inclusion of a bottom-up internal approach to key trends, rather than imposing external, literature derived parameters from baseline data describing a single year, as in previous iterations, vastly improves both the construction of a baseline scenario, and the design of internally consistent future scenarios.

FALAFEL is essentially a box model, tracing fluxes or flows of carbon through the human system of biomass appropriation and use, with the rates and sizes of fluxes controlled by input parameters including diet, waste production and management, and parameters associated with different technologies and management techniques of farming.

The structure of the key food system component of the model is given as a schematic in Figure 4.1. Model inputs and parameters, described in section 4.1,

are shown in the green columns on either side of the central column with their interactions depicted with pink arrows. These are used to set up a picture of the biomass harvest system at the year 2000 baseline, and define the trajectories of key trends and drivers through the modelling period of 2000-2050.

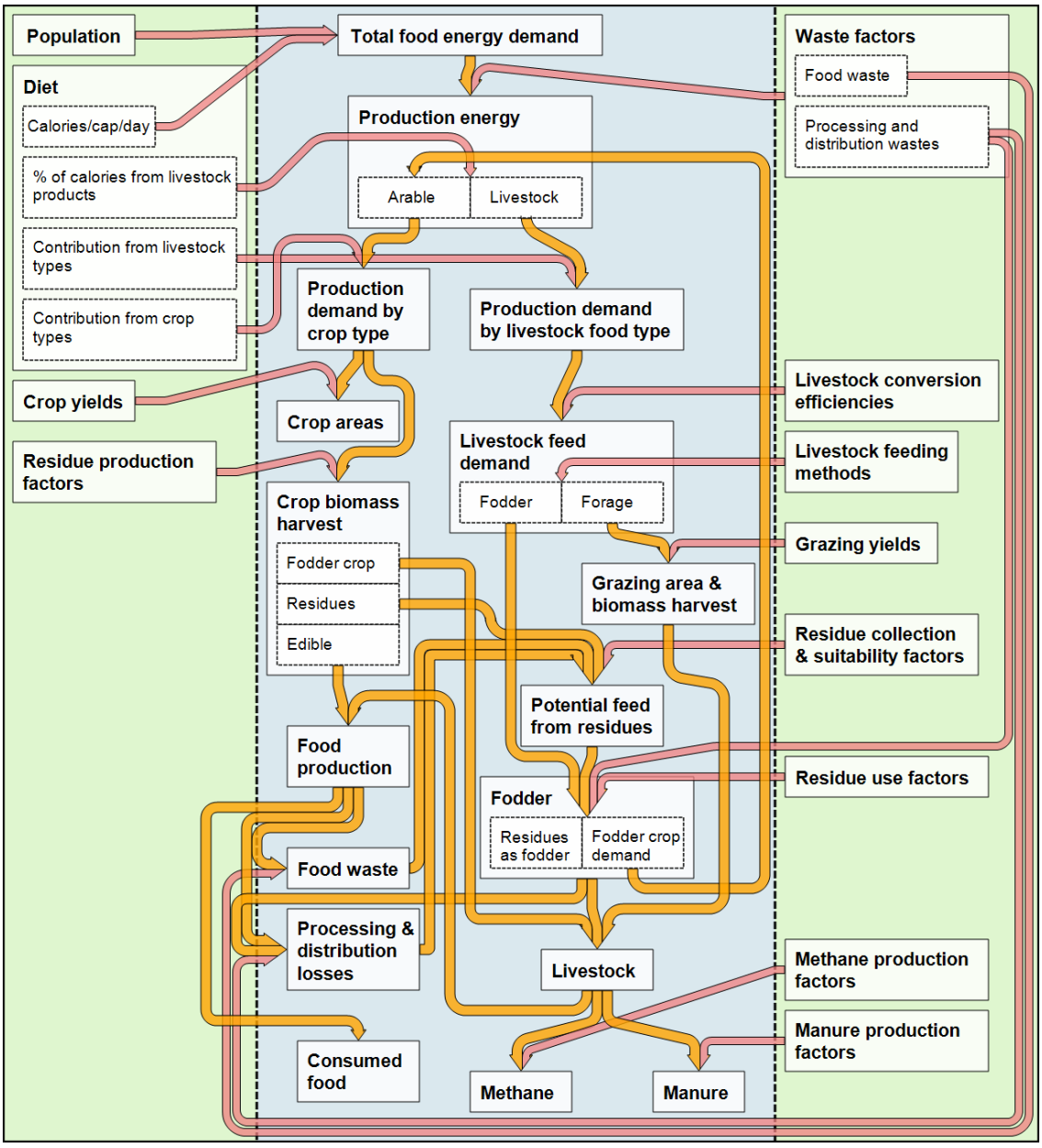

Figure 4.1: Schematic showing FALAFEL model structure. The central, blue area shows model processes, linked with orange arrows. The green areas show input data, linked with pink arrows.

Model processes, described in section 4.2, are shown in the central, blue column, with their interactions depicted using orange arrows. These describe the calculations made at each annual time-step throughout the modelling period. In this description the names of model processes (parameters, inputs,

boxes and outputs) are italicised, in order to distinguish them from the discussion of the concepts underlying them.

4.1 Trends driving biomass harvest

In order to develop an approach which internally linked forecasts of possible futures to observed trends in the past, rather than relying on externally parameterized trends derived from the literature, it was necessary to adapt the model to include time-series input data. By including input data for area, production and yield of all food, feed and fibre crops from the FAOSTAT database (FAOSTAT, 2014) for each year from 1970 – 2010, it was possible to derive yield trends for each major crop group over the 41 year period, as well as trends in the contribution of each crop group to the average diet, and the contributions of major fodder crop types to the diets of livestock animals. Data are taken primarily from *crop and livestock production inventories*, which detail primary production statistics, and *food balance sheets*, which are compiled nationally to account for net food import and export, consumption and the production of wastes and animal feed.

Although ultimately input data is converted to global averages, it is also collated as regional averages across eight regions; Sub-Saharan Africa, North Africa and Western Asia, Europe, Central and Southern Asia, East and Southeast Asia, Oceania, North America, and Latin America. This allows a comparison between global trends and trends across regions which, broadly speaking, differ from one another climatically, culturally and economically.

4.1.1 Diet - general trends

The forecast for overall food demand in each year is based on an idealised ‘global average diet’, derived from FAOSTAT production and consumption statistics. *Food supply* is defined as the total food energy available for human consumption (FAOSTAT, 2014), and is given on a per-capita basis in kilocalories per day (kcal cap^{-1} day^{-1}). This is the best available measure of the amount of food energy consumed by the average person in a given year, but is not in fact a direct measure, since it includes energy in foods that are available for consumption, but are subsequently lost as *post-production food waste*. Since little specific data is available for wasted food, it is assumed here that the

proportion of the food supply lost as food waste remained constant from 1970-2010, and therefore that the food energy actually consumed by the global population followed the same trend as that of the food supply. Global average food supply over this period increased at a rate of around 11 kcal cap^{-1} day^{-1} per year (Figure 4.2 a) from 2388 kcal cap^{-1} day^{-1} in 1970 to 2851 kcal cap^{-1} day^{-1} in 2010. Extending this trend to 2050 produces an expected further increase to 3353 Kcal cap^{-1} day^{-1} in 2050 (Figure 4.2 c); only slightly higher than the 3302 Kcal cap^{-1} day^{-1} used in previous work(Chapters 2&3).

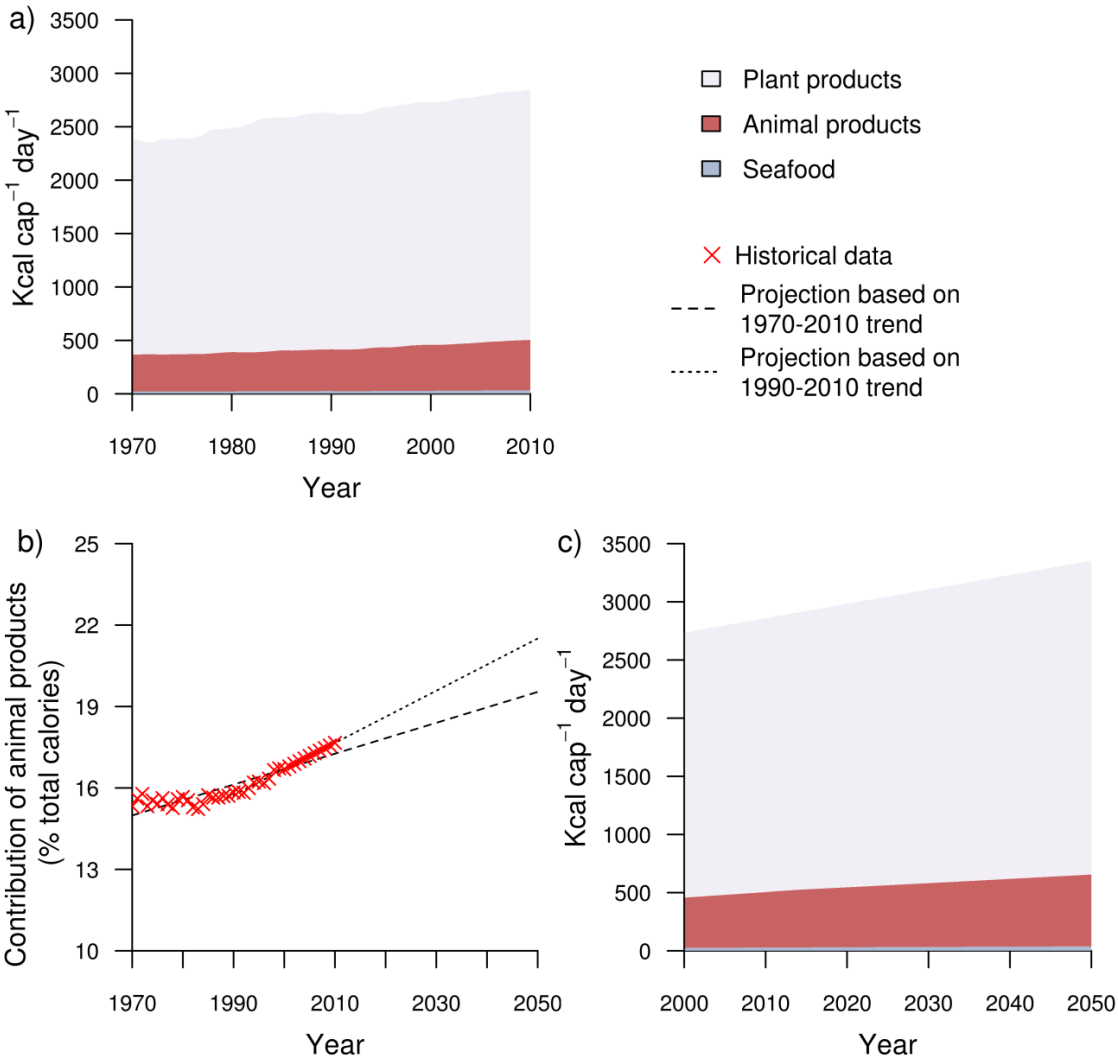

Figure 4.2: Trends in global average diet **a)** from 1970 – 2010, derived from the FAOSTAT databases, with **c)** projections to 2050. **b)** Trends in contribution of animal products to total calorific consumption, with two possible projections based on differing treatments of the historic data, also from FAOSTAT.

The calorific contribution of *animal products* to the average diet increased from 15.3% to 17.8% in the years between 1970 and 2010 (Figure 4.2 a,b). This is representative of changing diets in developing countries where animal products currently contribute as little as 7-15% of per capita calorific intake, but which are seeing a gradual shift from diets dominated by staple carbohydrates towards the protein, fat and sugar rich diets of developed countries (Kearney, 2010; Smeets et al., 2007). This increase appears to have been more rapid between 1990 and 2010 than for 1970-1990, perhaps reflecting accelerating economic development in large developing countries like China and Brazil. Projecting to 2050 using the average rate of increase for 1970-2010 (Figure 4.2 b) gives an expected contribution of *animal products* to the average diet of 19.7%; in itself quite an increase from around 16.8% in 2000. If, however, we interpret the increasing rate of growth between 1990 and 2010 as indicative of a sustained increase in the rate at which diets are changing, we could expect *animal products* to supply as much as 21.7% of calories in 2050.

Which trend is the more likely remains unclear as many factors may affect changes in diet; economic development is a key driver, but cultural differences are also significant. Economic development in China and Brazil, for example, has led to very significant increases in meat consumption, while in India and Africa the connection is weaker (Kearney, 2010). Some disagreement also exists over whether meat consumption has recently increased or decreased in North America and other developed regions where typically 25-35% of food energy is obtained from animal products, however it seems likely that upward trends in many of these regions are at least stagnating (Daniel et al., 2011; Kearney, 2010; Smeets et al., 2007).

The interactions between these trends in general point towards a continued rise in consumption of animal products globally, but at a slightly reduced rate (Alexandratos and Bruinsma, 2012; Kearney, 2010; Smeets et al., 2007). Here for the sake of consistency with the use of FAOSTAT data from 1970-2010 I use the 40 year average trend as a baseline, while the projections given by the faster growth rate fall well within the range of error used in sensitivity analyses (Section 5.2, p150). A baseline projection for the contribution of *animal products* in 2050 of 19.7% tallies well with the estimate given in a similar global study of the energetics of the biomass harvest system by Smeets et al.(2007), who use

an amalgamation of regional trends based on data from FAOSTAT and other sources to derive a global average trend which sees a 38% increase in per capita consumption for the period 1998-2050.

For 1970-2010 on average 6.2% (28.6 *Kcal cap*$^{-1}$ *day*$^{-1}$) of calories from animal products were provided by *fish and other seafood* (FAOSTAT, 2014), which are not considered beyond their contribution to diet as they do not affect land-use or the terrestrial carbon cycle. According to a collation of longer-term datasets fish and seafood consumption in *g cap*$^{-1}$ *day*$^{-1}$ increased by about 30% between 1963 and 1993, and then remained almost unchanged between 1993 and 2003 (Kearney, 2010). Although that study went on to predict a significant increase in consumption between 2003 and 2050, with a statement of some concern for fish stocks; since no very clear trend is show over the last 40 years in either dataset I assume that seafood remains a constant fraction of the dietary energy provided by animal products until 2050. It should however be noted that with the increase in consumption of animal products in general, and the growing world population, this represents more than a doubling of production from fisheries for 2000-2050; an increase likely to considerably increase pressure on marine ecosystems.

4.1.2 Diet – disaggregated trends

Plant and animal derived foods in the new approach are disaggregated into distinct subgroups consisting either of the single crop or animal species which have the highest production mass globally, or amalgamations of several crop or animal types. Crop plants are divided into *wheat, rice, maize, other cereals, soybeans, rapeseed, sunflower seeds, palm fruit, other oil crops, sugar cane, other foods* (an amalgamated group that includes *fruits*, *vegetables*, *pulses*, *roots and tubers*, *treenuts* and *mushrooms*). *Tea, coffee, herbs* and *spices*, and *cocoa* also form a distinct group, but these are excluded from calculations involving food energy demand and production, as they are considered luxury items which are not consumed for their energy content per-se. Animal products are separated into *bovine meat* (including *cattle* and *buffalo* meat), *pig meat*, *poultry meat* (including *chicken*, *turkey*, *ducks*, *geese* and other domesticated birds), *other meat* (which includes *sheep*, *goats*, *camelids* etc, and is assumed to behave the same as *bovine meat* unless otherwise stated), *milk* (which

includes milk produced for the manufacture of cheeses and other dairy products), *eggs* and *animals fats.* For full lists of the FAOSTAT primary production categories included in each subgroup, see Appendix I.

In order to determine the contribution of each major crop group to the overall calorie intake the *production mass* of each food group in each year (FAOSTAT, 2014), minus that which is used as *animal feed* (see section 4.1.6, p102), is converted to human-available energy using group-specific nutritional information (Table 4.1).

Table 4.1: Nutritional contents of crop groups.

	Nutritional contents (% of wet weight)			
	Moisture	Oil content	Protein and carbohydrate	Fibre (unavailable to humans)
Wheat	14%	0%	74%	12.2%
Maize	15%	0%	78%	7.3%
Rice	15%	0%	82%	3.5%
Other cereals	15%	0%	73%	12.2%
Soybeans	15%	18%	67%	0%
Rapeseed	10%	40%	50%	0%
Palm fruit	34%	25%	41%	0%
Sunflower seeds	10%	40%	50%	0%
Other oilseeds	10%	15%	75%	0%
Sugar cane	73%	0%	13%	15%
Other food *	56.4%	3.5%	35.1%	5.0%
Sugar beet	*70%*	*0%*	*16%*	*14.5%*
Roots and tubers	*65%*	*0%*	*32%*	*4%*
Fruit and vegetables	*85%*	*0%*	*14%*	*2%*
Pulses	*15%*	*5%*	*70%*	*10%*
Treenuts	*5%*	*55%*	*32%*	*8%*

* Nutritional content of *other food* is calculated as a weighted average of its subsidiary groups in the year 2000. Data from FAO standard conversion factors (FAO, 2003)..

It is assumed that the protein and carbohydrate content of crop products are available to humans with energy values of 18 MJ kg^{-1}, and fats with an energy value of 37 MJ kg^{-1} (FAO Agriculture and Consumer Protection department, 2003). The fibre content is assumed to provide no energy to humans, but fibre contained in oil crops (*soybean*, *palm fruit*, *rapeseed*, *sunflower seeds* and *other oil crops*) is made available as animal feed as part of the 'processing residues' waste stream, since 'residue cakes' formed in the pressing of oil from oil crops are commonly used as animal feed. The energy contributed to the human diet by each crop group is thus used to approximate the average diet of the global population (Figure 4.3).

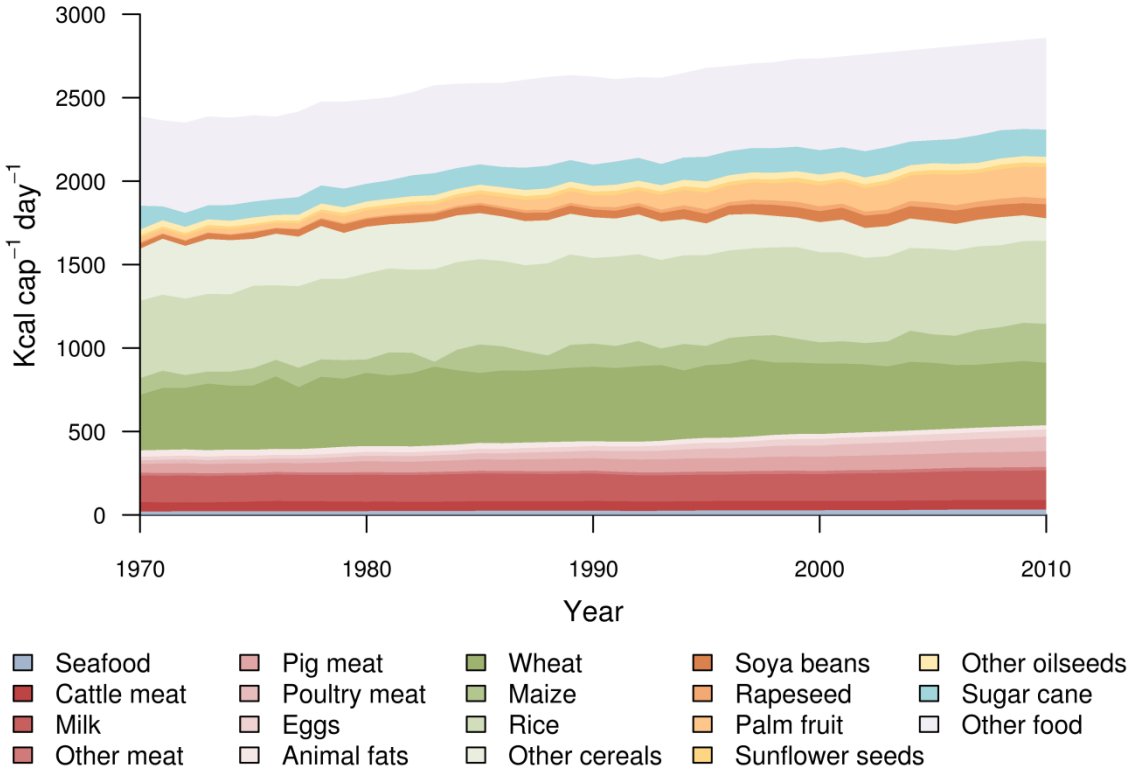

Figure 4.3: Disaggregated historical trends in the global average diet, from FAOSTAT data.

Trends in the average diet over the period 1970-2010 are then used to derive forecasts for the contribution of each food type from 2000-2050, which are used to generate gradually shifting food production demands as a model input. In order to accommodate differing dietary scenarios, the diet is divided into *animal products* and *crop products*, with the trends described in terms of % *contribution* to these two components. This allows variation in both the total contribution of animal products to the overall diet, and within the disaggregated groups; trends

described here are thus referred to in terms of % contribution to their respective portion of the overall diet. Full breakdowns of the contributions of crop and livestock groups are given in Appendix II.

Crop products for human consumption

Most of the trends in disaggregated crop-groups (Figure 4.4) are linear, but in the case of the declining consumption of *other cereals*, which are largely being replaced by the common cereals *maize*, *wheat* and *rice* an exponential decay is applied to prevent total exclusion from the diet.

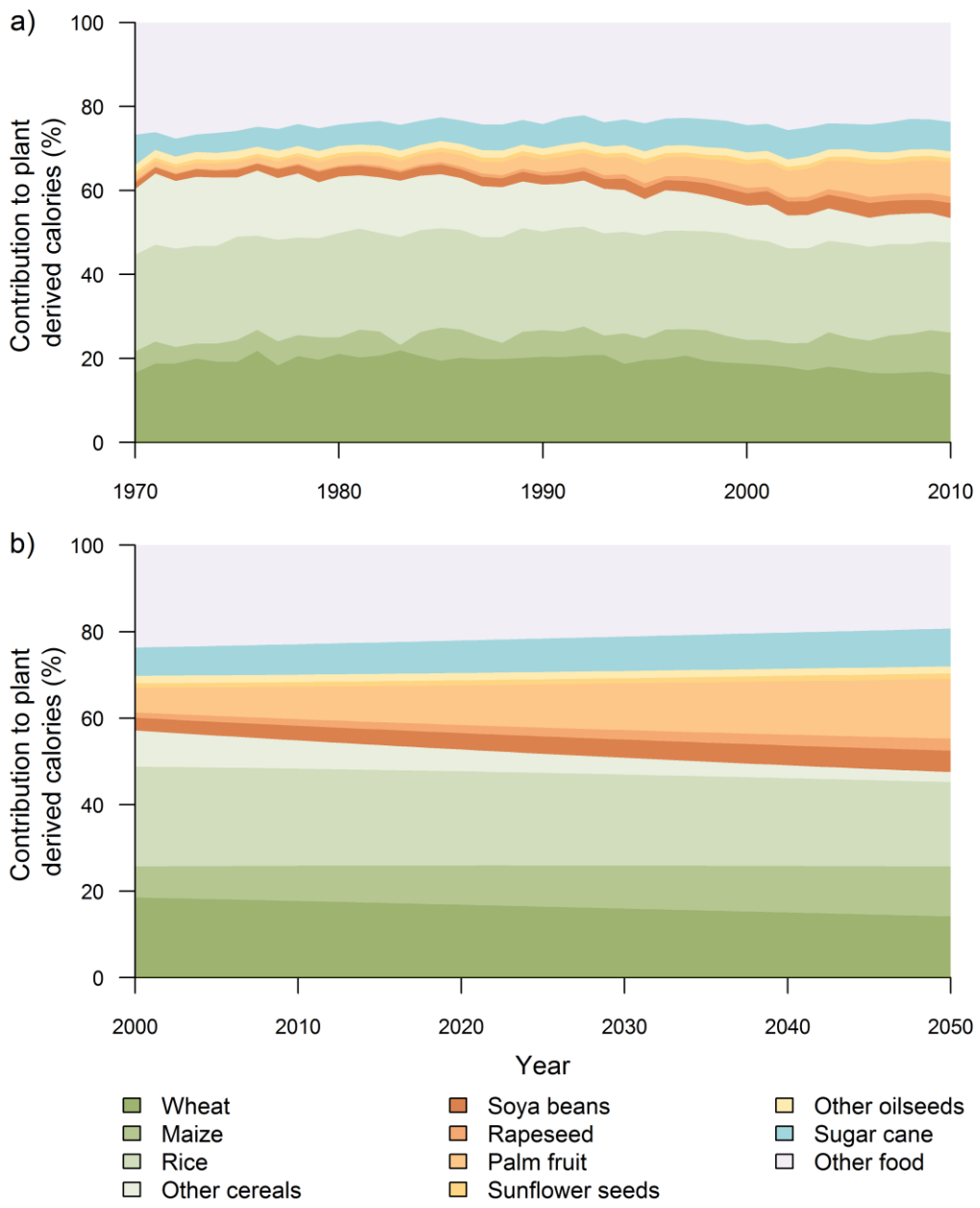

Figure 4.4: Trends in consumption of crop product groups; **a)** for 1970-2010 from FAOSTAT data, and b) projected trends for 2000-2050.

Perhaps most notable is the large increase expected in consumption of palm fruit from 5.8% to 13.8% of the non-livestock portion of the diet. This increase fits with rising consumption of vegetable oils in all economies (Kearney, 2010), and with the fruit of the oil palm being the highest yielding oil crop and becoming increasingly ubiquitous as a homogenizing agent in processed foods. For the same reason similar, though smaller, increases are expected in consumption of soybeans (2.9% - 5.0%) and rapeseed (1.2% - 2.8%).

Concurrent with the increase in consumption of vegetable oils is a decrease in the importance of cereals, with the group as a whole contributing 54.2% of calories from crop products in 2000, but only 47.5% in 2050. This is expected as part of a trend of increasing global wealth, as basic carbohydrates tend to be replaced with richer foods, and borne out by falling consumption of *other cereals* (from 8.4% to just 2.3%) and rice (23.1% - 19.6%). Also of note is the projected decrease in contribution of wheat from 18.6% of crop-products in 2000 to 14.2% in 2050. Wheat is typically associated with the diets of more economically developed populations, has been reported in other publications as the fastest growing cereal crop from 1963-2003 (Kearney, 2010), and has been forecast to maintain significant increases by economically driven models (Zuidema et al., 1994).

My exploration of FAOSTAT data is at odds with these accounts however, indicating a relative decline in wheat consumption for 1970-2010. Looking more closely at the data suggests that much of this decrease has occurred since the late 1990s, with wheat consumption as a proportion of diet remaining approximately level before then. It is difficult at this point to identify which of these trends most accurately represents the real world situation; it is possible for example that trend for the early 21st century has been skewed by very low yields caused by weather shocks in the key wheat producing regions of Europe, Oceania and North America in 2003, and 2005-2007) (own calculations, (Alexandratos and Bruinsma, 2012); for reasons of internal consistency with the input data and trends used I have opted to keep the trend derived for 1970-2010.

These decreases are offset slightly by increasing maize production (7.2% to 11.5%); although direct consumption of maize is in fact expected to decrease,

maize syrup is extremely widely used as a sweetener in processed foods and is likely to continue being so as developing economies consume more sweet foods (Kearney, 2010).

Animal products

The same process is applied to determine the contribution of livestock-product groups to food production, using FAOSTAT data on the production energy of each group from 1970-2010 (Figure 4.5). It is clear that a significant shift has been underway in the relative contributions of animal products (Figure 4.5 a), in particular with the declining importance of *cattle meat*, *animal fats* and *dairy products* and corresponding growth in *poultry meat* and *eggs.* The presence of *animal fats* was halved between 1970 and 2010 (10.7% to 5.2%) in parallel with the continued increase in consumption of vegetable oils; and is expected to do approximately the same in 2000-2050, decreasing from 6.3% to 3% (Figure 4.5 b). While *cattle meat, other meat* and *dairy products* (together comprising the total contribution of *ruminants*) made up 63.4% of consumed animal products in 1970, that contribution had fallen to 50.4% in 2010; if the same trajectory is maintained *ruminants* are expected provide only 37.1% of livestock derived calories in 2050. Meanwhile *poultry meat* is expected to become more significant, surpassing *dairy products* as the largest single share by 2050 at 29.4%.

Multiple factors are likely to lie behind this shift from ruminants towards monogastric species. Although red meat and animal fats have traditionally made up a very significant part of Western diets, there is some evidence that their prevalence may now be decreasing, associated with changing attitudes around diet and health (Alexandratos and Bruinsma, 2012; Daniel et al., 2011; Kearney, 2010). Probably of greater significance is the preference for poultry, eggs and pig meat in the diets of key developing nations, particularly China and India (Roy et al., 2002). This is likely driven by both cultural preferences (and indeed taboos in the case of India) and the relatively low cost and high efficiency with which pigs and poultry can be produced on an industrial scale.

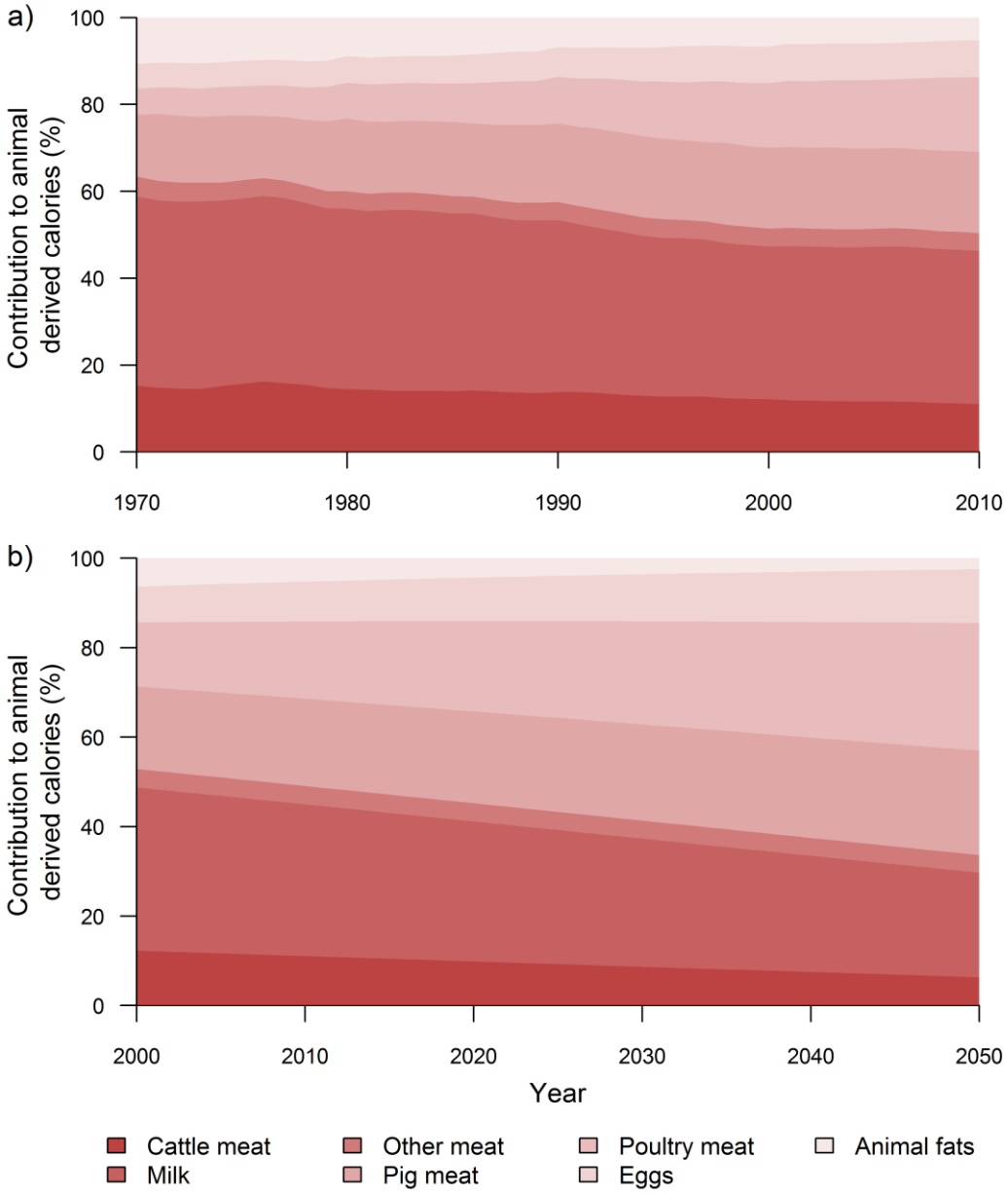

Figure 4.5: Trends in consumption of animal product groups ; **a)** for 1970-2010 from FAOSTAT data, and **b)** projected trends for 2000-2050.

Complete diet

Figure 4.6 shows the makeup of the baseline global average diet used to drive food demand in the FALAFEL model, constructed using the trends described above. The overall trend shows reduced cereal consumption, with increasing contributions from oil crops, sugar cane and animal products, concurrent with the observed trends globally, driven by shifts in developing countries towards diets richer in protein, fats and sugars, as well as a replacement of animal fats with vegetable oils in developed countries (Kearney, 2010).

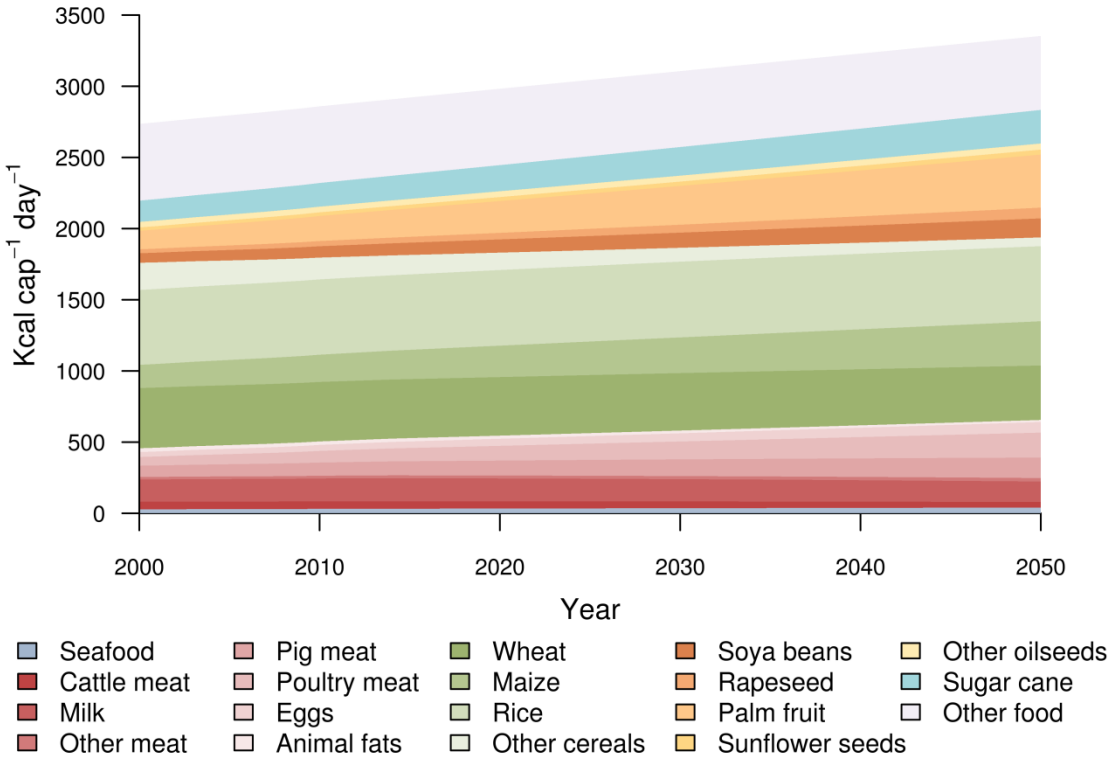

Figure 4.6: Makeup of the projected global average diet for 2000-2050.

Cereals provided 47.6% of all calories in 2000, but are expected to provide only 38.3% in 2050; a slightly steeper decline than that projected by the FAO (Alexandratos and Bruinsma, 2012), perhaps influenced by the unexpected forecast of a decline in wheat consumption. The contribution of oil crops is projected to almost double by 2050, from 10.5% to 19.7%, likely driven by very significant increases in consumption of vegetable oils in developing countries, and the strong trend in developed countries of replacing animal fats with vegetable oils (Kearney, 2010). Sugar cane is also expected to increase its contribution, from 5.4% of calories in 2000 to 7.0% in 2050, driven by increasing consumption of sugary foods in developing countries. The relative contribution of *other foods* is expected to decline from 19.7% to 15.5%, though this entails only a slight reduction in terms of actual calories (47 Kcal cap^{-1} day^{-1}). Further disaggregation would no doubt reveal trends in both directions among its constituent groups.

4.1.3 Yield

Average yields for each crop type or group in each year are used to define the global trend in yields for 1970-2010, by fitting regression lines. Yield trends

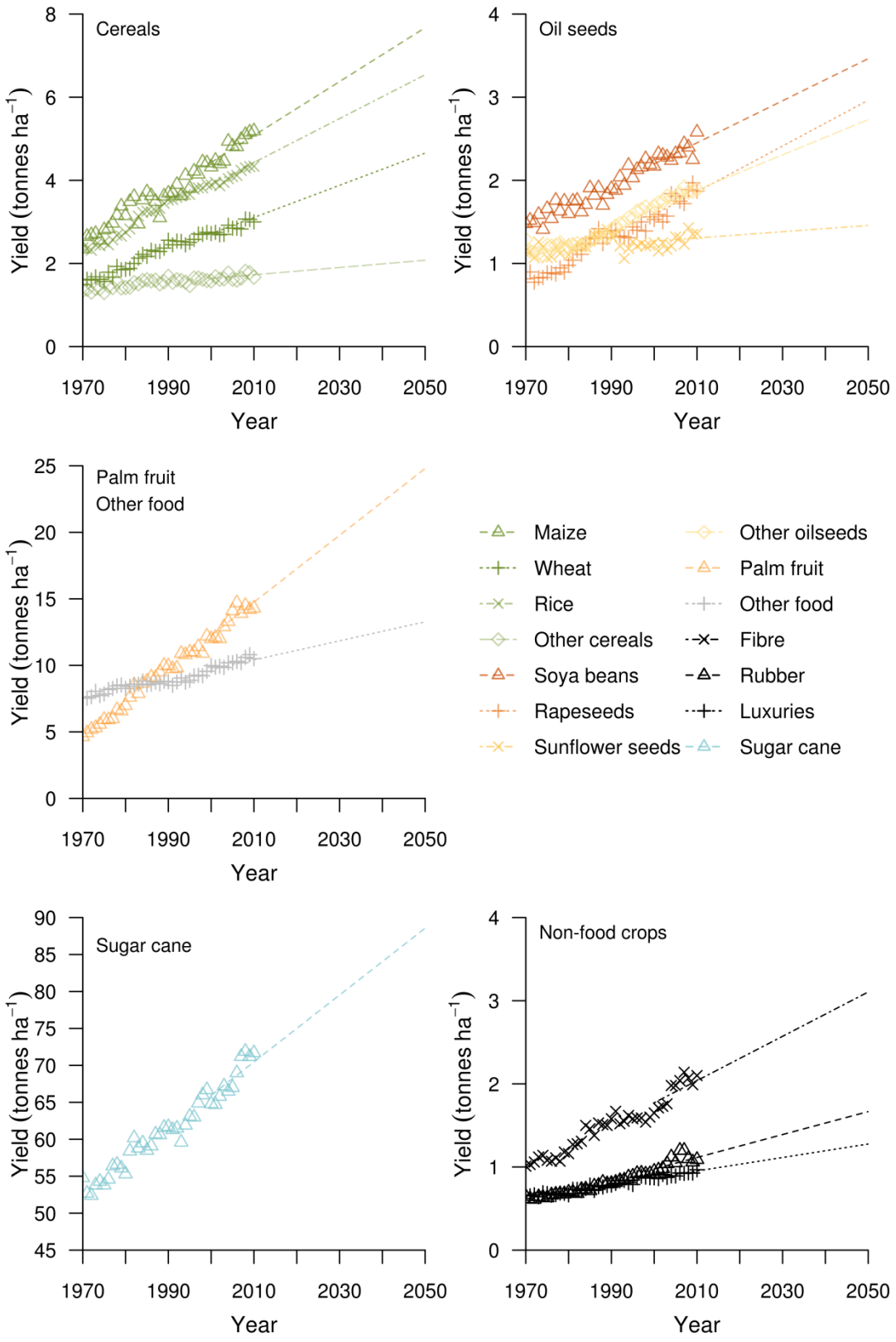

Figure 4.7: Historic global average yields for the major crop groups, with projections to 2050. Historical data as given by FAOSTAT.

based on this data are well described by linear increases in all crop types, as found in other studies (Alexandratos and Bruinsma, 2012; Grassini et al., 2013).

These trends are then used to extrapolate global average yields to 2050. In business-as-usual scenarios forward projections of yield trends are assumed to continue unchanged to 2050, although it is suggested that trends in some crops are unlikely to maintain such increases as many yields, especially those of cereals, have plateaued in developed countries (Grassini et al., 2013). It is also possible that reduced demand for some crops, cereals again cited as an example, may reduce the incentive to seek yield increases (Alexandratos and Bruinsma, 2012).

The recorded global average yields and forecast yield trends are shown here alongside the highest achieved regional average yield in the period 1990-2010 (Figure 4.7). Projected yield increases for each group can thus be compared with the highest currently achieved yields, in what are assumed to be the most suitable growing regions with the most developed farming systems. In all cases but one, projected global average yield in 2050 is significantly lower than the highest achieved regional average, giving a good indication that these forecasts are not excessively optimistic.

The only crop in which projected yield (24.8 t ha^{-1} in 2050) outstrips the highest achieved yield is *palm fruit*; the highest recorded regional yield being 20.3 t ha^{-1}. The yield trend for *palm fruit* is not currently treated any differently to other yield trends.

Livestock yield

Equivalent to crop yields, *livestock yield* values are also given by FAOSTAT; rather than production per unit area, livestock yields describe production per individual animal. For meat products these are given as *average carcass mass*, and for *egg* and *dairy* production as *annual production mass per animal*. Livestock yields are not necessary for the calculation of the land area necessary to produce a given food supply, but are used to determine the fluxes of the waste products *manure* and *methane* using production factors based on *production mass per animal*. Livestock yield trends are shown in Appendix III.

4.1.4 Waste factors

Three different waste factors are included as model inputs, describing wastes at the production, distribution and consumption stages of the food system.

Processing and distribution losses

Combined *processing losses* and *distribution losses* for the years 1990-2010 are calculated for plant derived food products as the disparity between the combined *production energy* of all crops allocated to human consumption (taken from the FAOSTAT food balance sheets (FAOSTAT, 2014)), and the *food supply* they actually provide. On average this amounts to around 40% of the energy content of food crops being lost before reaching consumers, meaning that primary crops are delivered as food products with only about 60% efficiency (Figure 4.8). No trend is evident in the level of processing and distribution losses in 1970-2010 (R^2<0.5); business-as-usual scenarios therefore maintain the same losses from 2000-2050.

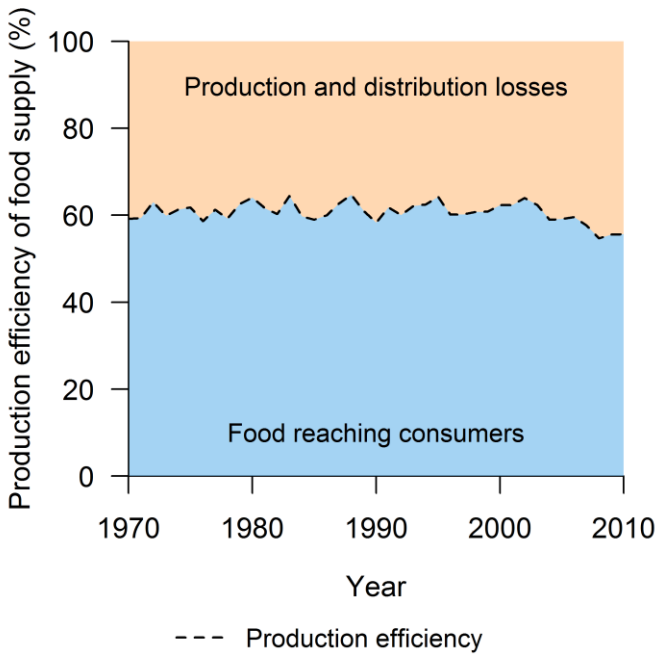

Figure 4.8: Production efficiency of crops for 1970-2010, from calculations based on FAOSTAT production data.

This fraction is divided into two separate streams to accommodate differences in the structure of the model between plant-based and livestock-based food production: a quarter (10% of total *production energy*) being apportioned to *distribution losses*, which are assumed also to apply to livestock products, and the remainder to *production losses* and in this context apply only to plant products. Processing losses likely include losses in storage of primary crop products and in the production of processed foods, for example the fibrous material left after pressing oilseeds. These are not applied to food energy

derived from the livestock sector because the types of waste are likely to be different, and to avoid double counting since the loss of non-edible parts of livestock is included in the *conversion factors* (see section 4.1.5).

Food waste

Food waste is caused by many different factors, from lack of adequate transport and storage infrastructure in developing countries, to aggressive marketing and consumer behaviour in more developed countries. Unfortunately FALAFEL is currently unable to distinguish between different sources of food waste, in part due to its use of global averages. As such the baseline rate of *post-production food waste* is assumed to be 30%, based on the lower end of the range reported in a recent major report (IMechE, 2013).

4.1.5 Crop residues

Group-specific crop *residue production* and *recovery* factors are derived from a comprehensive analysis of global biomass flows in the year 2000 (Krausmann et al., 2008), with regional values combined to produce global averages weighted according to the *production mass* in each region (Table 4.2).

Table 4.2: Residue production and recovery factors

	Residue production factor *(kg residue per kg yield)*		**Residue recovery factor**
	2000	**2050**	
Wheat	1.41	1.41	78.0%
Maize	2.42	1.20	75.7%
Rice	1.30	1.18	83.0%
Other cereals	1.58	1.45	76.6%
Soybeans	1.32	1.32	75.5%
Rapeseed	2.08	2.08	70.0%
Palm fruit	1.57	1.57	90.0%
Sunflower seed	2.07	2.07	50.0%
Other oilseeds	2.24	2.24	81.5%
Sugar cane	0.70	0.70	90.0%
Other food	1.53	1.45	73.7%

From Krausmann et al. (2008) and own calculations.

Regional residue production factors are also compared with yields, revealing negative correlation in some crop groups, implying that for these crops yield increases are concurrent with a reduction in the mass of residues produced. This is not unexpected, as yield increases and crop developments since the green revolution have frequently included shortening of stems and other reallocation of plant resources in order to increase resilience to poor weather, ease of harvesting and disproportionate growth of desired parts of the plant. In the crop groups that display this tendency linear regressions are fitted, and as global average yields increase from 2000-2050, so global residue production factors are reduced until they are equal to the lowest regional value for 2000. In other crop groups it is assumed that yield increases and residue production are not linked.

4.1.6 Livestock feed

The total energetic feed demand for livestock animals is back-calculated from the dietary contribution of *livestock products* using group-specific *conversion efficiency*, which describes the percentage of feed energy converted to edible product (Table 4.3). In general monogastric species (12-19%) are an order of magnitude more efficient than ruminants (0.95-2.3%) at producing meat products, though milk production (7.8-14%) can achieve similar levels of efficiency to non-ruminant species (Wirsenius, 2000, 2003). Total feed demand for livestock in 2000 was calculated as 1.46 x 10^{14} MJ.

Table 4.3: Conversion efficiency and feed preferences of livestock groups.

	Conversion efficiency * *(% feed converted to food product)*		**Fed with fodder *†** *(% feed energy)*
	Global average	Western Europe	
Cattle + other meat	1.5%	3.0%	45-55%
Dairy	7.8%	14.0%	40-45%
Pig meat	12.0%	15.0%	80-100%
Poultry meat	15.0%	19.0%	80-100%
Eggs	13.0%	16.0%	80-100%

Derived from *Galloway et al. (2007) or †Gill et al. (2010).

Feed requirements of livestock may be supplied by foraged or grazed biomass in pasture systems; or by *fodder*, which is sourced either from wastes and residues, or crops grown specifically as *fodder crops*. Although FAOSTAT food balance sheets provide data on crops grown for animal feed, they are not specific about which animals these crops are fed to; nor do they provide information on the amount of food energy livestock obtain from grazing. The feed intake of livestock groups within the FALAFEL model is therefore described by group-specific feed preferences (Table 4.3) gathered by Galloway et al. (2007).

In general, ruminant animals (e.g. cattle, sheep, goats) have a cellulose rich diet, derived from grazed grasses or silage, supplemented with fibre rich residue ‘cakes’ produced by vegetable oil and brewing industries, as well as some cereal grains and other fodder crops (Galloway et al., 2007): Mono-gastric species such as pigs and poultry have a broader diet, but are unable to digest grass and are generally fed on high energy food sources including cereal grains and vegetables. This sets up an interesting contrast between the two types of livestock species; although 69% of animal feed energy is consumed by ruminants, 72% of the feed crops grown on arable land, which might otherwise be growing crops for direct human consumption, are fed to monogastric species (Galloway et al., 2007).

Livestock feed sources differ considerably between regions, and particularly between more or less industrialized farming systems; industrial large-scale animal husbandry tending towards animals kept indoors and fed a higher proportion of *fodder* (Cassidy et al., 2013; Galloway et al., 2007; Wirsenius, 2000); while smaller scale livestock farming, and that in developing countries, tends to rely more on extensive pasture and other foraging opportunities. In general animals bred in intensive, industrialized systems tend to have higher conversion efficiencies, in part due to the higher nutritional value of their fodder-heavy diet, and in part due to a history of breeding for high output, rather than hardiness or other characteristics more suitable for extensively grazed animals (Galloway et al., 2007; Rosegrant et al., 1999; Wirsenius, 2000). This reflects two quite different roles of livestock within farming systems: Traditionally, livestock animals complement other agricultural and domestic processes; in the

case of ruminants by grazing otherwise unproductive grasslands, and pigs and poultry by foraging woodlands and consuming domestic wastes: In modern industrial societies, on the other hand, the market for livestock products has grown enormously, devaluing their original role in integrated farming systems, and promoting the intensive farming of animals on an enormous scale. In developing countries in which animals are still used in their traditional function, pigs fed on wastes can be a food source as efficient or even more efficient than cereal crops; while in Western Europe their efficiency is only a little more than a quarter that of cereals (Wirsenius, 2003).

Fodder

The upper ends of the ranges for feed intake from fodder (Table 4.3) are therefore taken as representative of modern industrial farming systems, and the lower ends as indicative of more traditional farming methods; default values are set at the median of the upper and lower bounds. In combination with the production energy of different livestock groups these values are used to determine a total feed intake from fodder of 6.7×10^{13} MJ in 2000, supplying 45.9% of total livestock feed demand with the remainder assumed to be provided by grazing.

Animal feed provided by fodder is further broken into feed from *fodder crops*, and feed from wastes or residue streams; possible waste streams which may be used to provide fodder being *agricultural residues*, *processing and distribution waste* and *food waste.* Agricultural residues consist largely of stems and leaves with high cellulose content, most of which are unsuitable for pigs and poultry, but are commonly used as fodder for ruminant species (Galloway et al., 2007; Wirsenius, 2000) (Table 4.4). Processing wastes may include a mixture of fibrous material (e.g. the husks of pressed oilseeds) and higher energy feeds such as brewery wastes, milling residues or fruit pulps, and are considered suitable for all livestock groups, though preferentially as feed for pigs and poultry, while post-production food waste is considered only suitable for pigs. Using these factors a total feed energy of 4.6×10^{13} MJ is calculated to have been provided by wastes and residues in 2000, or 69.7% of the total *fodder* requirement. Default projections use values at the lowest end of the range for the use of by-products as feed (Table 4.4).

Table 4.4: Suitability of wastes as feed for livestock groups.

	Agricultural residues	Food waste	Processing by-products
Cattle + other meat	25%	0%	5%
Dairy	25%	0%	5%
Pig meat	0-5%	0%	11%
Poultry meat	0%	0%	11%
Eggs	0%	15-45%	11-15%

Derived from (Galloway et al., 2007)

The remaining *fodder demand* must be met by crops grown especially for this purpose: In the same way as the diets of humans have been estimated from the relative production energy of food crop groups, the diets of livestock as provided by *fodder crop groups* can be estimated from *feed* production statistics from the FAOSTAT food balance sheets. Data are available for 13 individual crop types or groups; barley, maize, wheat, rice, other cereals, roots and tubers, *sugar beet and cane*, *fruit and vegetables*, *soybeans*, *other pulses*, *sunflower seeds*, *rape and mustard seeds*, and *other oilseeds* (Figure 4.9).

Of most significance here is the total dominance of maize, which accounts for around 50% of total fodder crops between 1970 and 1990; rising to around 58% by 2010. Also of note is the relative absence of soybeans, although they grow rapidly in significance from just 0.2% in 1970, they still supply only 2.5% of fodder crops in 2010. This is due to the nature of soybean production, and the accounting system of the FAOSTAT database: Since most soybeans are pressed, producing both soybean oil, which is used by humans as cooking oil and in processed foods, and oil and fibre rich soymeal cakes, which are fed to animals, FAOSTAT ascribes most soybean growth to crops grown for humans, and designates soymeal cakes as a byproduct of this growth which is fed to animals. Thus although whole soybeans supply only a small fraction of fodder, pressed soybeans make up a significant portion of the *wastes and residues* portion of fodder requirement mentioned above. It is very likely that the real-world drivers of soybean growth are not so clear cut, however since economic drivers do not play a part in the FALAFEL approach, and since adequate information is not available to effect a different treatment of combined

production of soybean oil and meal-cakes, I have adopted the FAOSTAT approach, with the assumption that whatever the market forces at work, the extrapolation of human and livestock diets, and associated production of residues from the same data implicitly describes the connection between the two products.

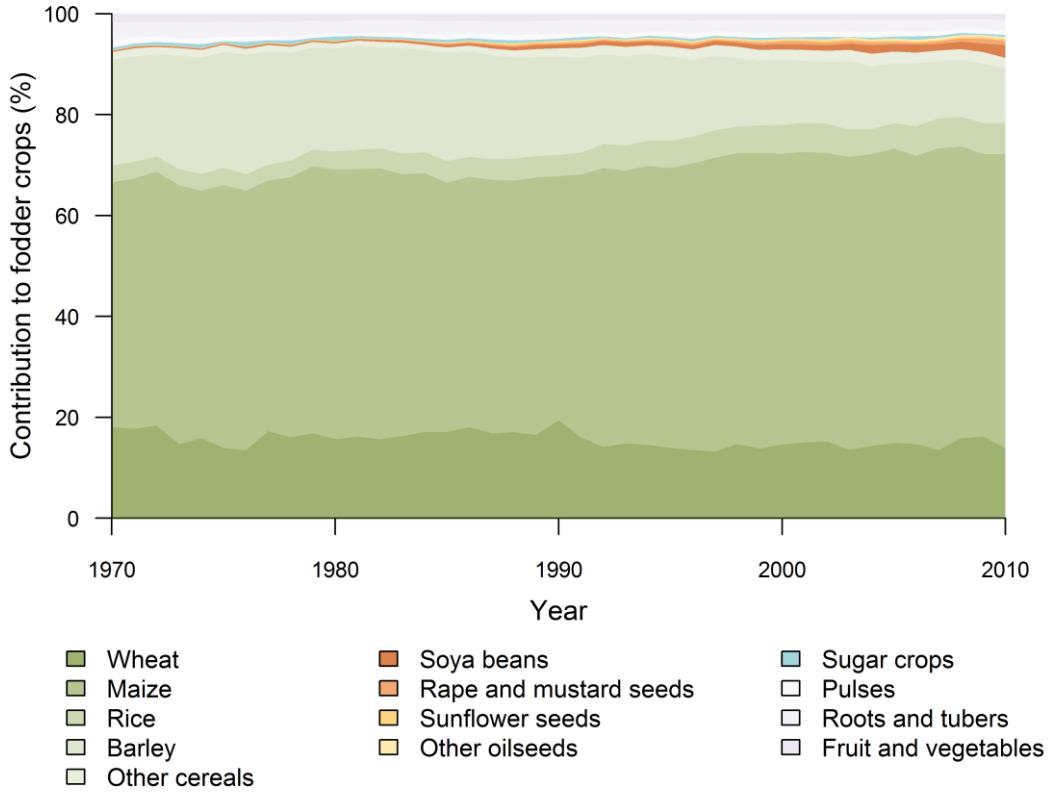

Figure 4.9: Historic trends in contribution of fodder crops for livestock, from FAOSTAT data.

Using these relative contributions to livestock diets in combination with yield data, the area of fodder crops required to meet demand in 2000 is 0.43 Gha; larger but within range of the estimate of 0.35 Gha produced by Foley et al. (2011) from gridded global datasets of crop distributions.

As for human diets, trends for 1990-2010 are used to derive forecasts for the relative production of different fodder crop groups for 2000-2050 (Figure 4.10). Most trends are linear, but again on the assumption that no crop group will disappear entirely, an exponential decline is applied to barley.

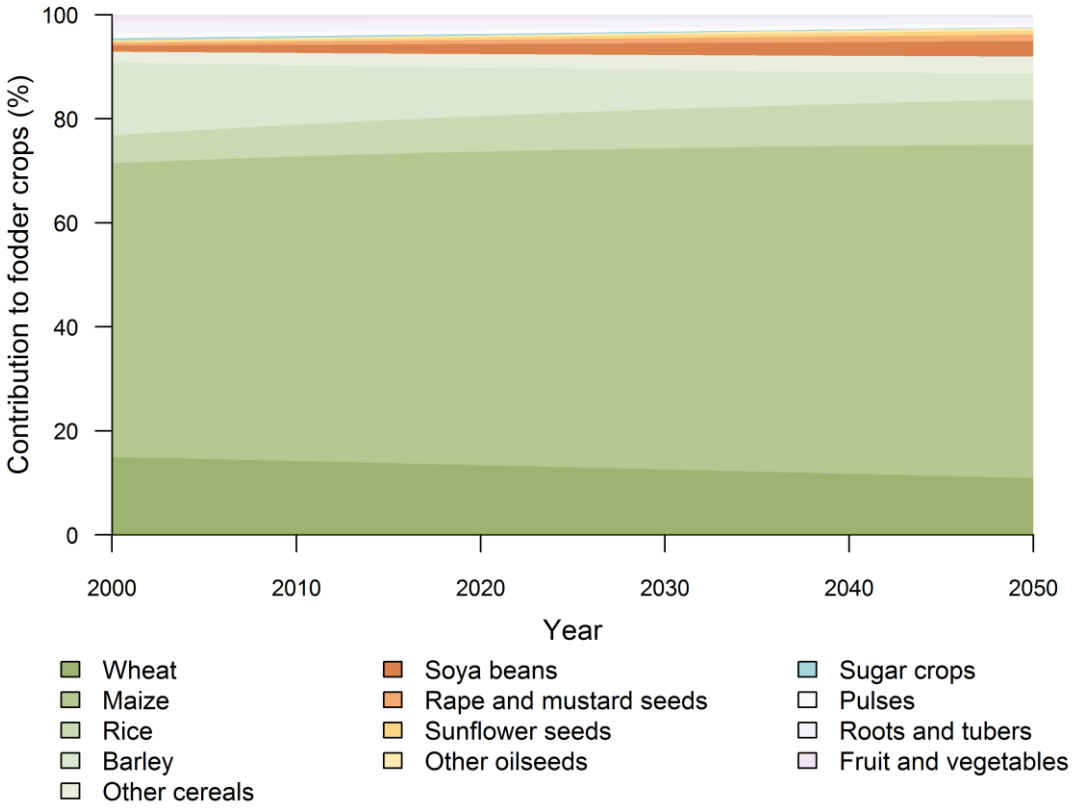

Figure 4.10: Projected trends in contribution of fodder crops.

As seen in human diets, maize and rice increasingly dominate the cereals used as fodder crops towards 2050, with declining use of wheat, barley and other cereals. Overall cereals provide 90.8% of fodder crops in 2000, decreasing slightly to 88.6% in 2050, while the contribution of fruit and root vegetables more than halves, from 4.0% to 1.8%. The use of pulses and oilseeds meanwhile doubles from 3.1% to 6.3%, with soybeans becoming an increasingly dominant constituent of that group.

Grazing

After accounting for the contributions of the various types of *fodder*, the remaining livestock *feed energy demand* is assumed to be met by grazed biomass from *pasture*. In 2000 this amounts to 6.67 x 10^{13} MJ; 45.8% of total feed demand.

Grazing land, however, is far from a single homogenous category of land-use, and includes a huge variety of systems from extensive semi-wild grazing of uplands or arid savannahs to the intensively grazed and fertilized grasslands common in lowland Europe. To assume that all grazed biomass was alike,

therefore, would be to miss important variation in the types of biomass harvest. Unfortunately, FAOSTAT definitions of *range-land and pasture* are necessarily rather constrained, and exclude certain management types (FAOSTAT, 2014; Krausmann et al., 2008; Ramankutty et al., 2008). In addition, FAOSTAT is unable, due to the near impossibility of aggregating such information globally, to provide data on the numbers of animals produced, or the amount of biomass that is ultimately consumed by livestock in different kinds of grazing systems.

To achieve some level of disaggregation in grazing types, I have used data generated by two studies for the year 2000. Ramankuttey et al. use agricultural statistics collected often at sub-national levels in combination with satellite imagery to produce gridded land-use maps for crop and pasture at 5 min resolution, with the data provided for each land-use type as % *coverage of gridcell* (Ramankutty et al., 2008). Haberl et al. (2007) use regional agricultural statistics and other information including estimated conversion efficiencies (Krausmann et al., 2008), in combination with a dynamic global vegetation model (DGVM) to produce spatial maps of HANPP relevant metrics (%HANPP, NPP_{act}, ΔNPP_{LUC}, NPP_0), also at 5 min resolution (Haberl et al., 2007). The final datasets, however, do not distinguish between biomass harvest in crop production or that consumed by livestock.

The map of pasture distribution was first used as a mask to isolate gridcells in the %HANPP dataset which contained pasture. In order to prevent biomass harvest data being skewed by the presence of cropland, with generally high %HANPP, the isolated data was then weighted by the area of pasture in each gridcell, resulting in a map approximating the global distribution and associated %HANPP of grazed pasture. The global pasture area was subsequently divided into two categories (Figure 4.11): 0.44 Gha of *intense pasture* covering 16.6% of total pasture area, with mean %HANPP of 41.9%, and 2.19 Gha of *low intensity pasture* with mean %HANPP of 21.9%.

ΔNPP_{LUC} was then subtracted from %HANPP to leave NPP_h as a percentage of NPP_0. This value was adjusted to account for the difference between NPP_0 and NPP_{act}, giving NPP_h as a percentage of NPP_{act}. On average *intense pasture* has ΔNPP_{LUC} almost twice that of *low intensity pasture* (20.6% and 11.5% respectively), indicating that intensively grazed pastures are more often the

result of significant land-use change, while less intense grazing has a lower ecological impact in terms of the removal of the original ecosystem.

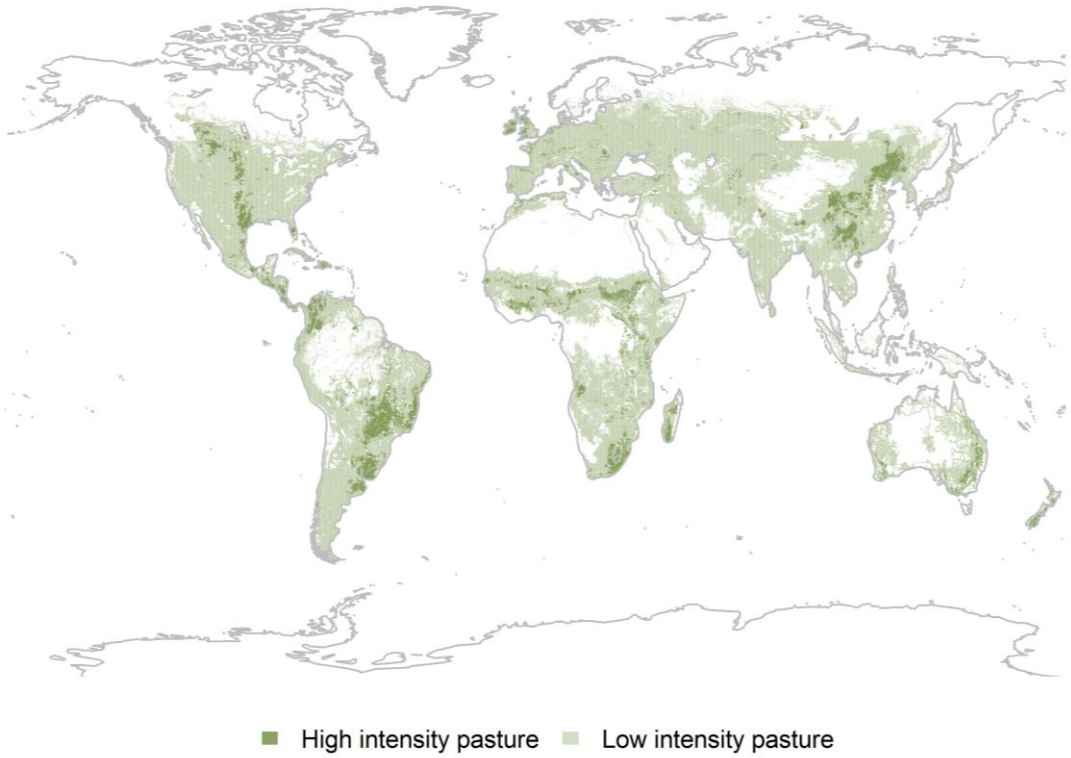

Figure 4.11: The distribution of *high* and *low intensity pasture* as determined according to the level of HANPP weighted by pasture area in each cell, based on datasets from Haberl et al. (2007) and Ramankutty et al. (2008).

An average of 26.9% of NPP_{act} (mean NPP_{act} 4.26 t C ha^{-1}) is consumed by grazing animals on *intense pasture*, and 11.8% on *low intensity pasture* (4.84 t C ha^{-1}). This results in total grazed biomass of 1.75 Gt C in 2000, 28.6% of which was consumed in *intense pasture* and 71.4% in *low intensity pasture*. This ratio is used to define a *grazing intensification factor*, which in business-as-usual scenarios remains constant to 2050.

4.1.7 Energy crops

Energy crops are not included in FAOSTAT production databases or food balance sheets, and although information is available for biomass energy generation (Chum et al., 2011; International Energy Agency, 2011), surprisingly little is to be found regarding the feedstock crops required. An approximation of

world bioenergy crop areas for 1990-2010 was therefore extrapolated from the little data available.

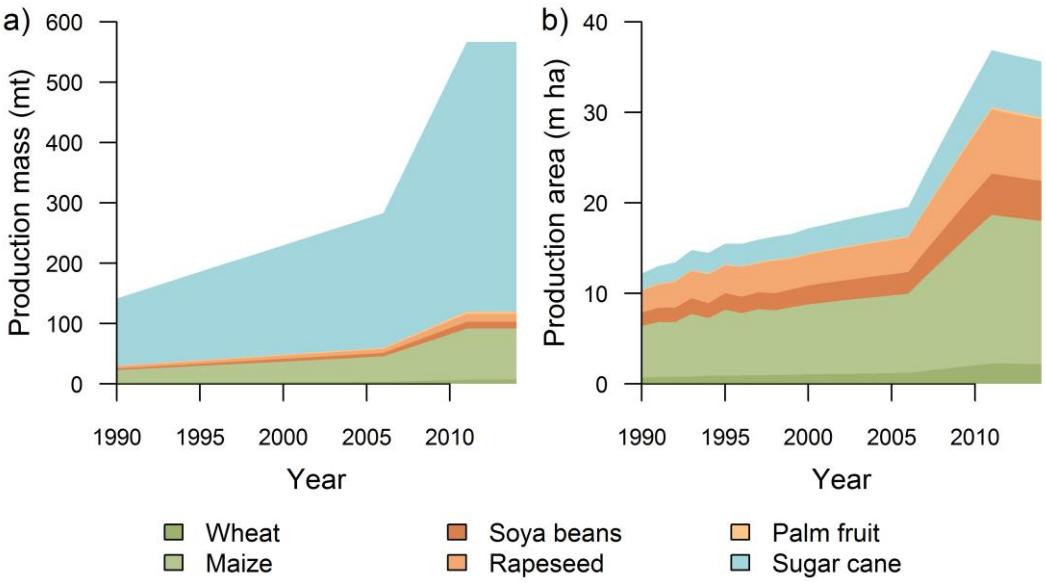

Figure 4.12: Estimated bioenergy crop production **a)** area and **b)** mass for 1990-2014, based on data for 2007 from the FAO (FAO, 2013), trends reported by the IEA and own calculations.

The area used to produce 6 key bioenergy crop plants (*maize, wheat, sugar cane, soybean, rapeseed, oil palm)* in 2007 (FAO, 2013), was used to infer their production mass assuming yields the same as plants grown for human consumption. IEA statistics describing a doubling of bioenergy generation for 2006-2011 followed by stagnation in 2012/13 provided an approximate linear growth rate of 20% of 2006 *production mass* per year for 2006-2011, which was used to estimate production mass for each crop. It was then assumed that bioenergy production had also doubled between 1990 and 2006, giving a second, slower rate of increase for that period. Production in 2011 was assumed to be followed by 3 years of continued production at constant levels (Figure 4.12 **a**). The calculation of bioenergy production after 2014 is described in section 4.2.7 (p124).

Production in the bioenergy sector is completely dominated by sugar cane during this time (78.8%, 0.27 Gt in 2007), with maize accounting for most of the rest (15.0%, 0.052 Gt). *Production mass* was also combined with food-crop *yields* to give bioenergy crop areas over this period (Figure 4.12 **b**). In terms of area maize is the dominant bioenergy crop, accounting for 10.4 Mha in 2007,

44.7% of total bioenergy area. Sugar cane, oilseed rape and soybean make up another 11.2 Mha, with smaller contributions from wheat and oil palm.

4.1.8 Other non-food biomass harvest

Biomass is harvested not only for its energy content, as food or fuel, but also for a range of other reasons. This section describes the trends associated with biomass harvested for use as fibres, timber products, rubber and other purposes not directly associated with calorific value.

Fibre, natural rubber & luxury consumables

Fibre crops comprise an amalgamation of all crop species used to produce fibres for which production statistics are given in FAOSTAT (Appendix I). Production of fibre crops increased by around 85% from 1970 to 2010 (Figure 4.13), driven by a global population growing both in number and in wealth. This relationship is unlikely to be straightforward however, as in that time there has also been considerable growth in production of synthetic fibres.

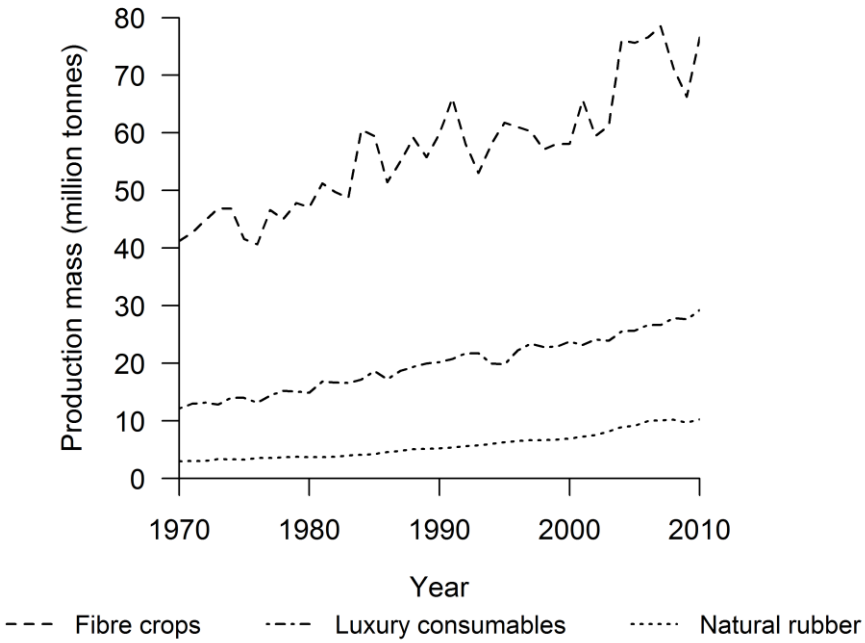

Figure 4.13: Production mass of *fibre crops, rubber*, and *luxury consumables* for 1970-2010, from FAOSTAT.

Production of *natural rubber* (Figure 4.13) saw an even larger increase of 244% during this time, despite being overtaken by the manufacture of synthetic

rubber. Rubber is used chiefly in the automotive industry, which has grown enormously over this time period.

A third category of biomass harvest listed in this section, designated *luxury consumables*, consists of biomass that is ingested by humans but for reasons other than its energy content. This group includes *tea*, *coffee*, *cocoa*, *spices*, and also *tobacco* (Appendix I), and are separated from the food categories because their consumption is likely to be driven more directly by economic factors than by food demand itself. Production of these crops also grew considerably from 1970 to 2010, increasing by 140% (Figure 4.13).

4.1.10 Natural biomes

The area and average NPP of 'natural biome' classes are calculated in much the same way as previously(Chapter 3), however for clarity a fuller explanation is given here than was allowed in the published article.

The challenge here is to give as nuanced an assessment as possible of the implications of expanding the land footprint human biomass harvest, within the FALAFEL model framework which relies on global averages and lacks a spatial context. While this is certainly not the perfect model framework for such an assessment, it has been possible to gain some degree of disaggregation in terms of the characteristics of biomes under threat from land-use change, and possible LUC emissions scenarios.

The area of land described as '*natural biomes*' into which human biomass harvest might potentially expand was determined using datasets of the spatial distribution of HANPP and other metrics published by the Institute for Social Ecology, University of Klagenfurt (Haberl et al., 2007), which combine biomass harvest statistics with DGVM output from the Lund-Potsdam-Jena model (LPJ). For the purposes of FALAFEL model inputs, *natural biomes* are defined as any terrestrial gridcell in which %HANPP is less than 10%. This cut-off point was chosen as almost all gridcells experience some degree of biomass harvest by humans, but very low levels of %HANPP are assumed to have relatively insignificant ecological impacts, leaving the biome in an approximation of its natural state. The resulting map was then overlayed onto a dataset showing

NPP_{act}, giving the NPP of all terrestrial gridcells relatively untouched by humans.

Table 4.5: Characteristics of natural biomes used in FALAFEL.

	Class boundary (Kg C m^2)	**Mean NPP** (kg C m^2)	**Mean above-ground NPP** (kg C m^2)	**Area** (G ha)
N1	>1	1.09	0.766	0.56
N2	0.75 - 1	0.900	0.630	0.62
N3	0.5 - 0.75	0.582	0.314	1.58
N4	0.25 - 0.5	0.384	0.207	3.30
N5	0 - 0.25	0.062	0.033	2.69

This dataset was then divided into five discreet NPP classes, with the boundaries between classes evenly distributed across the range of NPP values (Table 4.5). The sum of the gridcell areas, and the *average NPP* could then be calculated for each class. Average NPP was converted to *above-ground NPP* using ratios derived from the IPPC carbon inventory methods (Ravindranath and Ostwald, 2008).

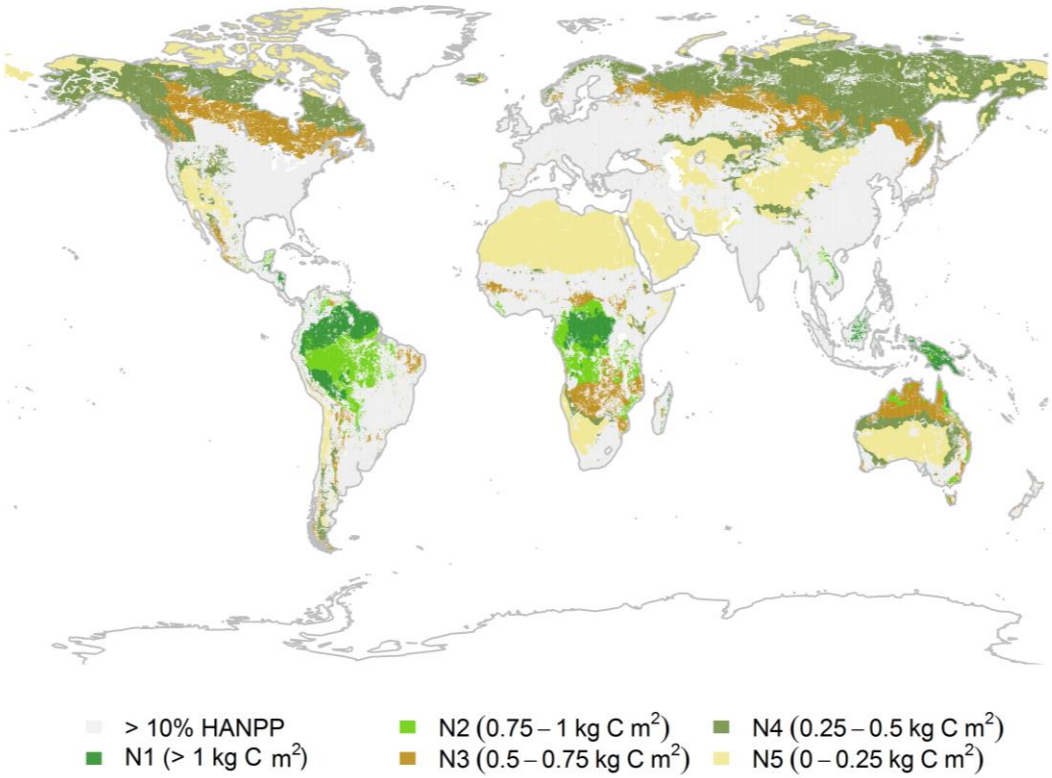

Figure 4.14: Global terrestrial 'unmanaged' ecosystems, zoned by NPP, derived from datasets published by Haberl et al. (2007).

The spatial distribution of the NPP classes generated by this exercise in fact maps quite well onto the distribution of the world's major biomes (Figure 4.14), with the five categories in order of productivity equating approximately to two classes of largely tropical forest, a grassland biome (combining savannah and steppe), boreal and alpine forests, and low productivity desert and tundra biomes. These approximations are used to allocate carbon stocks to each NPP class using IPCC tier 1 methodology (Eggleston et al., 2006).

Because mapping the distribution of cells in which HANPP is greater than 10% gives a slightly greater estimate of managed land area than that generated by FALAFEL, the total area of each biome class is scaled in order to make a total ice free land area of 13 Gha.

4.2 Model processes and flux calculations

This section describes the structure of calculations made in the FALAFEL model to describe fluxes of biomass carbon between model boxes or processes in each annual timestep. These are depicted in the central, blue highlighted area of the schematic provided at the beginning of this chapter (Figure 4.1). All biomass calculations are converted to tonnes of carbon assuming standard C content of biomass of 50% dry weight.

4.2.1 Food demand

As in previous iterations(Chapters 2&3), global demand for food energy is the ultimate driver of biomass harvest, with the underlying assumption that enough food will always be harvested to meet the predicted needs of the world population. *Food demand* is the equivalent of the FAOSTAT metric *food supply*, calculated annually as *average per capita consumption* (*Kcal cap*$^{-1}$) plus *food waste* (*%*), converted into KWh. As such, any decrease in food waste leads to a reduction in the food supply required to meet food energy demand, rather than leading to increased availability of food (Figure 4.15).

Food demand is then divided between energy from *livestock products* and energy from *plant products*. In the case of plant products demand for food energy is combined with *processing losses* and *distribution losses* to calculate the required *production energy*, i.e. the amount of plant based food products

actually required to meet food demands, after all wastes are taken into account. Animal products are also converted into *production energy* with the addition of *distribution losses,* but do not explicitly include *processing losses*, since in the livestock component these are included in the *conversion efficiency*, described later.

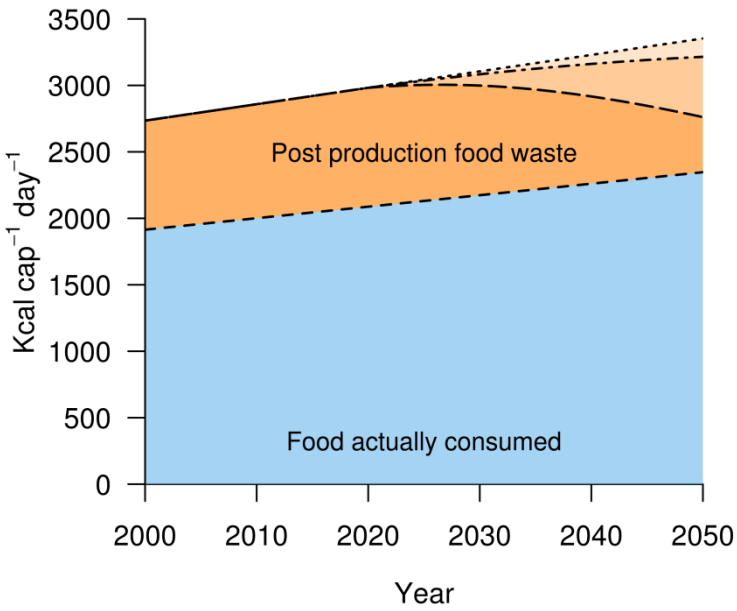

Figure 4.15: Projected per capita calorific consumption, with the required food supply depending on differing food waste trends. The food waste trends illustrated here are those used in the scenarios described in Chapter 5.

The required *production energy* for plant- or livestock-derived foods is divided among the different *crop-* or *livestock-product types* using forecasts of their contribution to the relevant component of diets as described in section 4.1.1 (p87).

4.2.2 Crop biomass harvest

Crop group specific *production energy* is converted to *dry production mass* using nutritional energy (*MJ kg^{-1} wet mass*) and moisture (%) contents (Table 4.1). *Residue production* is calculated using *residue production factors* (*tonnes residue per tonne dry yield*) and added to *dry production mass* to give *total biomass harvest.*

4.2.3 Biomass wastes & residues

Energy values for *food waste* and *processing & distribution losses* are all calculated as percentage of production energy, using the waste production factors described in section 4.1.4 (p99). The portion of *processing losses* generated by the non-human available fibre content of oilseed crops is also calculated for use as livestock feed.

Energy values are then converted to dry mass, with wastes treated as a homogenous mix of 'average biomass' with energy content of 18 MJ kg^{-1} (FAO Agriculture and Consumer Protection department, 2003), and subsequently to mass of carbon.

Crop residues as tonnes of dry mass are calculated for each crop group using *residue production factors* (section 4.1.5, p101), which are dependent on *production mass* and in some cases on *yield*. Crop residue production masses are also converted to *production energy* for use as animal feed, again assuming average energy content of 18 MJ kg^{-1}.

Residues are preferentially made available as animal feeds before other uses, because they currently tend to fetch a higher price as feed than as bioenergy feedstocks (Nonhebel and Kastner, 2011), and also because FALAFEL in general prioritizes efficient food production over the production of bioenergy feedstocks.

4.2.4 Livestock sector

Livestock feed demand is back calculated from the required *livestock production energy* using group specific *conversion efficiency.* The default assumption is that the efficiency of conversion from feed to food products will increase towards 2050, as feed quality improves and management becomes more intensive. Linear growth trends are therefore applied to *conversion efficiency* values from 2000, in the default scenario shrinking the gap by one third between the year 2000 global average and the high efficiencies reported for livestock systems in Western Europe (Galloway et al., 2007) (Table 4.3, p102).

Using *livestock feed factors* (section 4.1.6, p102) the total feed demand is divided into *feed from grazing* and *feed from fodder*: Again this balance is

expected to change towards 2050, with a greater focus on fodder feeding under more industrialized livestock systems (Table 4.3, p102).

Feed from wastes & residues

Using the factors describing maximum expected *feed from by-products*, the highest potential feed energy from *food waste, crop residues*, and *processing and distribution losses* is calculated for each animal product group and summed for each residue type. This is then compared with the size of the residue streams available (section 4.2.3): If *potential feed* from a particular by-product is smaller than the available residue stream then all of the potential feed energy is met by these by-products, with some edible biomass-waste remaining surplus to requirement; conversely if *potential feed* is larger than the available residue stream, the entire stream is consumed by animals, with the deficit in supply of *feed energy* met by extra production of *fodder crops*.

Feed from fodder crops

Any feed energy demand from fodder not met by residue streams is used to drive demand for *fodder crops*. This is calculated as the energy required from fodder crops plus *distribution losses*, based on the assumption that the system of fodder crop supply cannot be 100% efficient, but will not include the same *processing losses* as crops grown for human consumption since fodder crops are likely to be consumed ‘green’ rather than going through several stages of processing before being consumed.

The required *production energy* for fodder crops is then divided among the 13 *fodder crop* groups according to the trends described in section 4.1.6. These are subsequently fed back to the *crop biomass harvest* calculations, forming a separate *fodder crop* harvest category which takes into account the nutritional content available to animals in each crop type. Residues are produced from the growth of fodder crops using the same *residue production factors* as for crops grown for human consumption, but these are not made available as animal feed, mostly for reasons of modelling simplicity in the spreadsheet framework, which does not deal well with circularity in calculations.

Feed from grazing

Demand for animal feed energy from grazing is divided between *intense pasture* and *low intensity pasture* using the *grazing intensity factor*, making the assumption that the available energy per kg of grazed biomass is equal in both pasture types, but that in *intense pasture* a higher mass of biomass is consumed per unit area.

Grazed feed-energy demand and the associated biomass harvest of grazing for the year 2000 (section 4.1.6) are used to approximate factors describing *grazed biomass energy per unit mass*, which are used to convert grazing energy demand to biomass harvest for the two categories.

Manure production

To calculate manure production, the *production mass* of each livestock group is converted to the number of individual animals (usually called *head*) using livestock yield values. *Manure production factors* in *dry tonnes* yr^{-1} $head^{-1}$ for each group are then used to calculate total annual manure production (Table 4.6).

Table 4.6: Livestock excretion factors.

	Manure production *(dry tonnes* yr^{-1} $head^{-1}$*)*	**Methane production** *(kg* CH_4 yr^{-1} $head^{-1}$*)*	
Cattle meat	1.1	55	*±50%*
Dairy	1.8	89	*±50%*
Other meat	0.08	3.8	*±50%*
Pig meat	0.1	1.25	*±50%*
Poultry meat	0.01	-	-
Eggs	0.01	-	-

From IPCC guidelines for national greenhouse gas inventories (Eggleston et al., 2006).

4.2.5 Methane emissions

Methane emissions from livestock animals are calculated from emission factors in *kg* CH_4 yr^{-1} $head^{-1}$ (Table 4.6), although since there is a high degree of variability in methane production by animals, according to their diet, breed, living conditions other factors, error bars are given at ±50%.

Emissions from rice production are also highly variable and dependant on the method of land management; particularly the amount of time the soil spends waterlogged. Here emissions are calculated based on FAOSTAT global estimates (themselves based on IPCC tier 1 methodology) from 1990-2010, divided by the global rice production area to give a *CH_4 emission factor* in tonnes per hectare. The range of error is calculated using those given in the IPCC tier 1 methodology (Eggleston et al., 2006).

4.2.6 Fibre, forestry and other biomass harvest

Fibre, natural rubber & luxury consumables

Because these categories of biomass harvest are not driven directly by demand for food, it was necessary to link them to other input parameters. Typically the overall demand for biomass harvest is assumed to be a function of the population and economic development of the global population. Since the projected increase in per capita calorific food supply is also assumed to be driven by increasing economic development (Kearney, 2010), it is used here as a proxy for wealth.

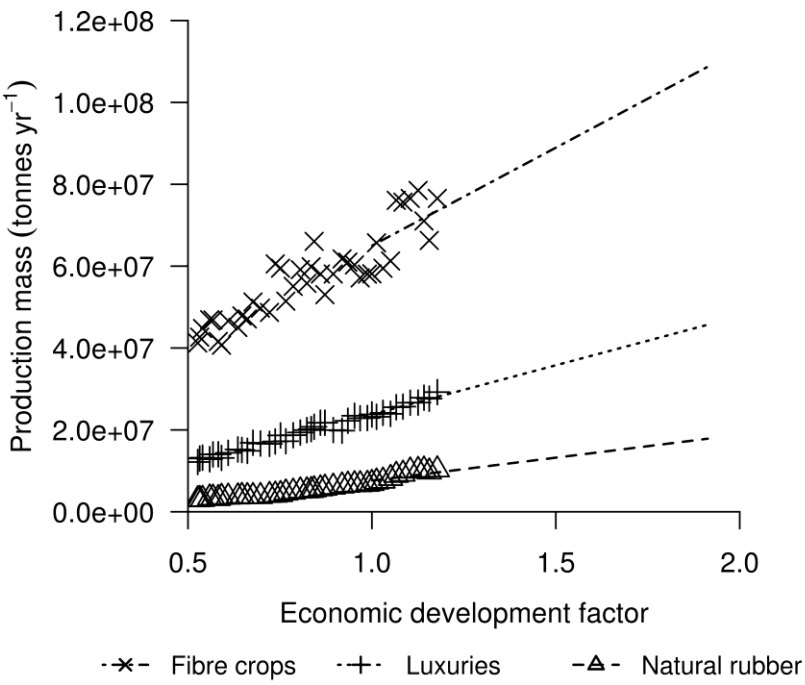

Figure 4.16: Production mass of non-food biomass harvest categories plotted against *economic development factor.* Historical data from FAOSTAT is plotted with points; lines show projected values.

An *economic development factor* is derived at each timestep by multiplying the *global population* by *per capita food supply*, normalized against the value for the year 2000. This factor correlates very strongly with the 1970-2010 production mass of the three categories (R^2 > 0.9), with the resulting regression lines providing trends to drive their production in FALAFEL (Figure 4.16).

Forestry

As in previous versions(Chapter 2), forecasting of biomass harvested in *forestry* is based on the FAO forestry division's forecast of a steady rate of increase in the area of *managed forest* to 400 Mha in 2050, assuming a constant rate of biomass harvest on forested land of 2.4 tonnes ha^{-1} yr^{-1} calculated for the year 2000 by Krausmann et al. (2008). This remains the same in all scenarios.

4.2.7 Agricultural land area & land-use change

Required area of agricultural land

Using *yield trends* described in section 4.1.3, the required land area for each crop group, including fodder crops, is calculated as *yield x production mass*. Year 2000 area and biomass harvests of the two categories of grazing land (section 4.1.6) are used to derive *harvest factors* describing average biomass harvest per unit area (equivalent to *yield*) for pasture types; in subsequent years these factors are combined with required biomass harvest for *intense* and *low intensity* pasture to give the required area of each.

Area of managed forest

As described above, the area of land on which *forestry* as a form of biomass harvest is projected to take place is forecast by the FAO forestry division to increase steadily to 400 Mha by 2050.

Urban area

Urban area is forecast using urbanization projections in the 'Shared Socioeconomic Pathways' (SSP) scenarios (Jiang and O'Neill, 2015; O'Neill et al., 2013), with the baseline rate of urbanization used in FALAFEL following the SSP2 'middle of the road' scenario, and the error range incorporating the other

scenarios (Figure 4.17). % *population living in urban areas* is converted into *urban area* using a declining trend in *city area available per capita*.

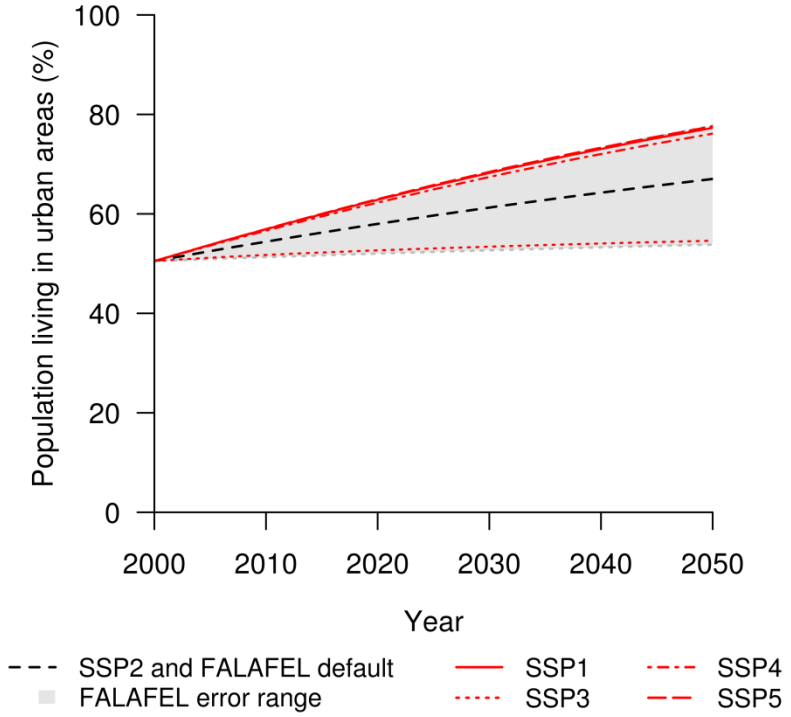

Figure 4.17: Default urbanization trajectory with error range used in FALAFEL, also showing the variation in SSP urbanization scenarios as given by O'Neill et al. (2013) and Jiang & O'Neill (2015).

Land-use change

In order to calculate land-use changes at each timestep, broader land-use categories are designated; *arable crops* comprises all food crops with the exception of *oil palm*, as well as *fodder crops* and *fibre crops*; the two categories of grazing land remain separated as *high-intensity-* and *low-intensity-pasture*; *managed forest* includes both forestry and the tree crops *rubber* and *oil palm*; *urban* land comprises its own land-use type, as does *energy crops*, and *unmanaged land* remains as five distinct categories. Changes within these categories, e.g. from one arable crop type to another, are treated as neutral; changes between categories constitute *land-use change* within the context of the FALAFEL model.

When the land area required by a particular category increases from one timestep to the next, land must be taken from one of the remaining categories. If the required land area decreases, land is given up and can be occupied by another category which may be undergoing expansion. Each land-use category

is given a set of rules about which other categories it is allowed to expand on to, based on assumptions about geographic location and suitability. *Urban* expansion, for example, is allowed to occupy only *arable* land or *intense pasture*. These land-use types are displaced on the basis that urban centres are by and large surrounded by cultivated land of relatively high quality, rather than by unmanaged land; an assumption also made in more complex, spatially resolved models such as the IMAGE suite (Zuidema et al., 1994).

Table 4.7: Suitability of land-use categories for expansion of managed land.

Expanding land-use category	**Land-use types available**					
	Urban	Arable	Intense Pasture	Low intensity pasture	Woody crops and Forestry	Natural ecosystems
Urban		50%	50%			
Arable				*		
Intense pasture				*		
Low intensity pasture						
Woody crops and forestry						†
Energy crop or other use						

Grey boxes indicate that expansion is not possible onto the indicated land-use type
Orange boxes indicate that expansion *must* occur on the indicated land-use type
Blue boxes indicate that the indicated land-use type can be occupied if it is no longer required by its current use

***** Only one third of low intensity pasture is assumed to be eligible for 'upgrading' to intense pasture or arable land-uses. † The tropical tree crops *natural rubber* and *palm fruit* are permitted to occupy only natural ecosystem classes *N1* and *N2,* while *foresty* crops are divided among classes *N1*, *N2*, *N3* and *N4* to cover the full range of forestry types from tropical to boreal.

For most types of *managed land* expansion is met by occupying other land-use categories in which the required area is declining; if after 'spare' managed land is allocated the full requirement is still not met the remaining portion is met by expansion into *natural ecosystems*. Where expansion of one land-use type is met by the land no longer required by others, land is allocated according to an

approximate hierarchy; the type of land-use requiring the most productive land having the primary claim to the most productive land being relinquished, the order of preference in general being *arable land* > *intense pasture* > *low intensity pasture*. A depiction of the rules and preferences allocated to each land-use type is given in Table 4.7.

When *managed land-use*s expand into *natural ecosystems*, the total expansion of each category is divided among the classes of natural ecosystem according to ratios derived from literature (Gibbs et al., 2010; Ramankutty et al., 2008) (Table 4.8). These were estimated by categorizing the historical (1985-2005) region level estimates for the given land-use types according to their dominant *natural ecosystem* class as depicted in Figure 4.14 (p113). A given portion of each biome class, expressed as a percentage of the original biome area, can be protected from LUC, with the requirement for expansion met on other classes which have not yet met their protected limit; default protection is set at 25% of the original area of N1, N2 and N3, with no protection for N4 and N5.

Table 4.8: Ratios of expansion for cropland, pasture and managed forest into the five natural biome classes.

Biome:	**N1**	**N2**	**N3**	**N4**	**N5**
Cropland	12%	26.9%	51.3%	8.6%	1.1%
Pasture	5%	10.2%	73.9%	3.1%	7.6%
Managed forest	10%	30%	30%	30%	-

Derived from (Ramankutty et al., 2008), numbers may not add to exactly 100% due to rounding

The methodology described in this section is clearly a vastly simplified treatment and does not allow for possible land-use combinations and transitions such as agroforestry or grazed woodland, but this discreet categorisation was necessary to keep the structure of land-use allocation simple enough to operate in the Excel framework. Also not explicitly included is the concept of rotation of crop types, or between pasture and arable land-uses; however given that land-uses in FALAFEL are not described in a spatial context but rather as the total required area in each class, it is not possible to describe shifts of land-uses in space which do not lead to a net change in area of land-use types.

Unused land

Following the rules applied to earlier work(Chapters 2&3), if, and only if, there is a net decrease in the total required area of *arable crops* and *pasture* from one year to the next, the surplus land is made available for an alternative land-use, with current options comprising either *second generation energy crops* or *afforestation*.

As before, *second generation energy crops* are assumed to be lignocellulosic crops such as *Miscanthus giganticus x.* or other similar crops suitable for low input cultivation on poor or degraded land, with an estimated average yield of 10 t ha^{-1} yr^{-1}. When *afforestation* is selected an average carbon accumulation curve derived from the BEAC calculator is used, assuming that disused agricultural land is distributed evenly between tropical and temperate zones. Since in many gridded land-use models abandoned land is assumed to return to the natural forest type (Zuidema et al., 1994), the afforestation treatment in FALAFEL could also be considered analogous to land abandonment.

CO_2 emissions from land-use change

Net carbon emissions from LUC are calculated according to the IPCC tier 1 methodology, wherein net C emissions are equal to the carbon stock of the original biomass minus the C content of the new vegetation; the underlying assumption being that when land-use change occurs the original vegetation is destroyed in its entirety, with the growth of replacement vegetation beginning in the same year. In the case of transitions to cropland replacement growth is assumed to be zero as arable crops are counted as annual growth, the carbon content of which is returned to the atmosphere within one year either through consumption, decomposition or burning. For transitions to grassland all regrowth is assumed to occur in a single year, while for transitions to managed forest carbon accumulation curves are applied representing either *tropical*, *temperate*, or *boreal* tree species depending on the biome class being replaced.

Carbon stocks for the different vegetation types are estimated using the IPCC guidelines for national greenhouse gas inventories (Eggleston et al., 2006) (Table 4.9). Within the same managed land-use, differing carbon stocks may be assumed depending on the class of the original *natural ecosystem*, so for

example the carbon stock of newly created *intense pasture* is calculated as a weighted average of the values given for pasture created on the *natural ecosystem* classes *N1*, *N2* and *N3*.

Table 4.9: Estimated aboveground carbon stocks of managed and natural ecosystems. All units are tonnes per hectare.

	Original biome type				
	N1	**N2**	**N3**	**N4**	**N5**
Original carbon stock	164.4	85.4	5.2	20.8	2.9
Cropland	1.3	1.3	1.3	1.3	1.3
Pasture	16.1	9.4	5.2	8.5	7.4
Managed forest	135.0	86.0	86.0	28.3	28.3
Energy crops	0.8	0.8	0.8	0.8	0.8

From IPCC guidelines for national greenhouse gas inventories (Eggleston et al., 2006).

4.2.8 Use of bioenergy crops and waste biomass streams

All biomass diverted to use in energy production or CDR are assumed to have an energy content of 36 MJ kg^{-1}, and carbon content 50% of dry mass.

Bioenergy crops

Two options are available for the processing of bioenergy crops; conventional use or bioenergy with carbon capture and storage (BECCS). The default setting is for 50% of available bioenergy crops to be used in each; however this can be varied between scenarios. Under conventional use, half of bioenergy crops are used as transport fuel and half for combined heat and power (CHP) generation. *Energy yield efficiencies* (Table 4.10) are used to determine the total *energy generated* by each. The *carbon intensity* of the fossil fuel equivalents is then used to calculate the carbon emissions *offset* by the use of biomass fuels. *Energy yield* and *carbon offset* are calculated in the same way for crops diverted to BECCS systems (Table 4.10), with the addition that *carbon capture efficiencies* are used to calculate the *CDR flux.* As in the previous chapters, it is assumed that all dedicated bioenergy crops diverted to BECCS are processed using the BIGCC technology described in Section 2.3.3 (p32).

Table 4.10: Factors associated with bioenergy feedstock pathways.

	Transport fuel	CHP	Pyrolysis	BigCC
Energy yield	35%	35%	35%	30%
Carbon capture	-	-	50%	90%
Fossil fuel equivalent	Petrol + diesel	Natural gas CHP	Natural gas CHP	Coal + gas electricity generation
Carbon intensity (kg C MJ^{-1})	0.017	0.015	0.015	0.017

Values for pyrolysis are based on Azar et al. (2006) and Klein et al. (2011), for biochar from Woolf et al. (2010), and carbon intensity of fossil fuels are from Metz et al. (2007).

Waste biomass streams

Currently up to 75% of all biomass waste streams within FALAFEL (comprising *processing and distribution wastes*, *food waste*, *crop residues*, *forestry residues* and *manure*) are deemed suitable for bioenergy with carbon storage (BECS), with the exact portion being determined by a variable *waste recovery factor*. As above, *energy yield*, *carbon offset* and *CDR flux* are calculated, assuming that all biomass wastes with the exception of *forestry residues* are processed using pyrolysis, for the reasons discussed in Chapter 2 (Section 2.3.3, p31). *Forestry residues* are deemed suitable feedstock for BIGCC systems.

Implementation of BECS

In order to compare scenarios in which CDR strategies are adopted at different times and rates, the year in which diversion of biomass from conventional uses to BECS begins can be varied, as can the number of years taken for BECS to reach the full allocated capacity. Prior to the set *implementation date* it is assumed that all biomass waste streams destined for energy generation are used in conventional systems, with biomass wastes being used in CHP systems.

4.2.9 CDR fluxes and effect on atmospheric CO_2

Carbon fluxes are treated exactly the same as in previous work (section 2.5, p42), with the offset of fossil fuel emissions and the CDR flux generated by

BECS combined to give a total CDR flux, with a negative value. This is added to LUC emissions to give the total net carbon flux.

As before, the net carbon flux in each year is then decayed for the rest of the modelling period by applying the decay function derived from the Bern carbon cycle model (equation 1). In any given timestep the effect on atmospheric CO_2 is thus determined by the cumulative effect of the net fluxes in all previous years, but with the effect of earlier emissions decreasing with each year.

4.2.10 HANPP and other macroecological indicators

While not currently set up to make the full analysis carried out in Chapter 3, FALAFEL calculates a number of indicators of the macrecological effects of the global biomass harvest system, based on net primary productivity (NPP).

Potential vs. Actual NPP

The difference between the expected NPP of an ecosystem in the absence of any human interference (here referred to as potential NPP, or NPP_0), and the actual NPP (NPP_{act}) is an important indicator of the magnitude of the anthropogenic influence in that ecosystem (Haberl et al., 2007). In FALAFEL, NPP_0 for managed land categories is estimated using the NPP of the 5 natural biome classes (NPP_B), weighted by the historic ratios of expansion derived from (Ramankutty et al., 2008) (Table 4.8), as shown in 5 equation 5;

$$NPP_0\ (t\ C\ ha^{-1}) = \sum_{B=1}^{5} (NPP_B \times expansion\ ratio) \qquad \textbf{5}$$

Using these estimates global NPP_0 is calculated using equation 6:

$$NPP_0\ (PgC\ yr^{-1}) = \sum_{\substack{managed\\landuse\\types}} (NPP_0 \times area) + \sum_{B=1}^{5} (NPP_B \times biome\ area) \qquad \textbf{6}$$

This produces an estimate for global aboveground NPP_0 of 38.5 Pg C yr^{-1}, slightly higher than the 35.4 Pg C yr^{-1}, calculated by Haberl et al. using the LPJ DVGM, but close considering the relatively crude approach used here.

In each timestep after 2000 local NPP_0 in each land-use category is recalculated based on its previous value, and the NPP_0 of any new land it has

expanded into. If the area occupied by a particular land-use shrinks, the NPP_0 for that category is assumed to remain unchanged. Global NPP_0 is unaffected by the sharing of land between categories.

Estimates for NPP_{act} on each managed land category were derived from different sources: Those for managed forest and urban areas were borrowed directly from Haberl et al. (2007), while estimated NPP_{act} of the pasture categories were based on the NPP data originally used to calculate their area and intensity of biomass harvest (section 4.1.6, p107); NPP_{act} for crops and energy crops is estimated as their harvested biomass (including residues) with the addition of an estimated unharvested fraction (NPP_t). Equation 6 is then used, with NPP_0 substituted for NPP_{act}, to calculate global NPP_{act} which is estimated at 36.6 Pg C yr^{-1} in 2000, again close enough to the 33.5 Pg C yr^{-1} estimated by Haberl et al. (2007) using much more sophisticated techniques. The difference between NPP_0 and NPP_{act} is the result of replacing natural biomes with managed ecosystems, referred to as ΔNPP_{LUC} and expressed as a percentage of NPP_0: ΔNPP_{LUC} in 2000 is 5.0%.

Local NPP_{act} remains constant throughout the modelling period in *managed forest*, *pasture*, *urban* and in *bioenergy crops* once they are entirely replaced with lignocellulosic crops. In *crops*, however, NPP_{act} changes proportionally to *yield*, with 50% of all yield gains assumed to come directly from an increase in total productivity, while the remaining 50% is assumed to be as a result of other changes such as redirecting plant resource allocation. Global NPP_{act} is recalculated in each year to account for the changing areas of managed land and natural biome categories; ΔNPP_{LUC} is thus also recalculated.

On each type of managed land NPP_{act} in each year is divided into two parts; the NPP harvested by humans (NPP_h), and unharvested NPP (NPP_t). NPP_h is calculated as the sum of *production mass* and *residue production* for each type of biomass harvest; both *harvested* and *unharvested residues* are included in NPP_h, as even if residues are left in the field their role in local ecosystem processes has been substantially altered by human activity, changing biomass from living primary phytomass into something more akin to leaf litter, accessible mainly to decomposers. NPP_t is the living biomass remaining on managed land after harvest; for example grass that goes ungrazed in pasture, or the living

remains of crop and other plants in cropland. NPP_t for cropland and energy crops is estimated from a number of sources (FAOSTAT, 2014; Haberl et al., 2007; Krausmann et al., 2008), and also influenced by yield increases; for other land-uses NPP_t is simply calculated as $NPP_{act} – NPP_h$.

The NPP that remains available to natural ecosystems after human activity (NPP_n) is calculated as the sum of NPP_t on managed land and NPP in natural biomes. Global NPP_n is thus calculated using equation 6, substituting NPP_0 for NPP_t, as 29.1 Pg C in 2000.

Chapter 5: Model output and sensitivity analysis

Abstract

In this chapter I use the FALAFEL model to generate scenarios which can be compared with those described in Chapters 2 and 3. Total land-use for food production is generally lower than previous estimates due to changes in assumptions and methodology. In the most comparable scenarios, CDR fluxes are also smaller, due largely to changes in assumptions about the availability of residue streams as BECS feedstocks. However, the new model structure also allows for the generation of more extreme *high efficiency* and *high BECS* scenarios, which achieve very large CDR fluxes.

I then conduct a sensitivity analysis to show the full range of possible FALAFEL output and determine the dominant drivers of variability in possible outcomes. In general these are associated with total food demand (e.g. *population*, per capita *food supply*, contribution of *livestock products* to the average diet), or the production of livestock, particularly ruminants (e.g. *conversion efficiencies*, *intensification*, feeding of residues).

5.1 Comparison with previous scenarios

In order to display model output from the finalized FALAFEL model, I have constructed a set of scenarios based on those presented in Chapters 2&3. In this version the *low efficiency* (LE) and *business as usual* (BAU) scenarios are roughly equivalent to the two efficiency variants previously used, while *high efficiency* scenarios are now rather hypothetical examples of a world in which *efficiency* and *intensity* parameters are set much closer to those of advanced industrial systems, and *wastes* are reduced significantly; giving a technically feasible upper limit to developments in those areas. Values in 2050 for the key parameters in each scenario are shown in Table 5.1; as before each assumes the same default assumptions on diet and population, but varies in terms of waste production and recycling, and intensity of livestock management. All scenarios follow the BAU trajectories from 2000-2014, diverging thereafter.

Table 5.1: Parameter values in 2050 for three scenarios.

		2050		
Variable *(value in 2050)*	**2000**	**Low efficiency (LE)**	**Business as usual (BAU)**	**High efficiency (HE)**
Population	6127700000	9550945000	9550945000	9550945000
Food consumption *(Kcal cap^{-1} day^{-1})*	1914.5	2347.2	2347.2	2347.2
Processing losses *(% of total production)*	28.2%	28.2%	25.4%	14.1%
Production & distribution wastes *(% of total production)*	10%	10%	10%	5%
Food waste *(% of food supply)*	30%	30%	27%	15%
Food waste available as animal feed *(% of food waste)*	15%	15%	15%	40%
Grazing intensity *(% grazed biomass from high-intensity pasture)*	28.6%	28.6%	35%	40%
Livestock intensification *(% feed from fodder)*				
Cattle meat	45%	45%	50%	55%
Dairy	40%	40%	42.5%	45%
Pig meat	80%	80%	90%	100%

Poultry meat	80%	80%	90%	100%
Eggs	80%	80%	90%	100%
Livestock conversion efficiency *(% feed converted to food product)*				
Cattle meat	1.3%	1.5%	2%	2.5%
Dairy	7.0%	7.8%	9.9%	11.9%
Pig meat	11.6%	12%	13%	14%
Poultry meat	14.5%	15%	16.3%	17.7%
Eggs	12.6%	13%	14%	15%
Feed from agricultural residues *(maximum% contribution to total feed)*				
Cattle meat	25%	25%	25%	25%
Dairy	25%	25%	25%	25%
Pig meat	-	-	-	5%
Poultry meat	-	-	-	-
Eggs	-	-	-	-
Feed from processing wastes *(maximum% contribution to total feed)*				
Cattle meat	5%	5%	5%	5%
Dairy	5%	5%	5%	5%
Pig meat	11%	11%	11%	15%
Poultry meat	11%	11%	11%	11%
Eggs	11%	11%	11%	11%
Feed from food waste *(maximum% contribution to total feed)*				
Cattle meat	-	-	-	-
Dairy	-	-	-	-
Pig meat	15%	15%	15%	45%
Poultry meat	-	-	-	-
Eggs	-	-	-	-

Each of the three basic scenarios also features *projected meat* (PM) and *low meat* (LM) variants ('meat' in fact referring to all livestock products including dairy and eggs); *projected meat* diets follow the observed trends as discussed in Chapter 4 (Section 4.1.1, p 87, 4.1.2, p 90), while *low meat* variants follow previous work in reducing consumption of livestock products to 15% of per capita food supply by 2050. The BAU PM scenario is thus the closest to simulating likely real world trajectories. Changes in the contribution of groups within the livestock and plant product sectors follow the projected trend in all scenarios.

In all of these scenarios the recovery of residues (i.e. agricultural residues, manure, processing and distribution wastes, forestry residues and food waste) for bioenergy generation is set at 25% of the available residue stream. Half of this feedstock is diverted to pyrolysis for BECS, with the remainder used for conventional energy generation combined heat and power generation. Similarly, half of all dedicated bioenergy feedstock is diverted to BigCC for BECCS, and half is used for conventional electricity generation. In addition, two extra *high BECS* (HiBECS) variants of the BAU scenarios are constructed, in which 75% of available residues are recovered, and 100% of all feedstocks (i.e. residues + dedicated bioenergy crops) are diverted to BECS via the appropriate process. Again the purpose of this is to provide a 'technical potential' rather than a realistic scenario.

5.1.1 Biomass harvest for food production

As in Chapter 2, the estimated plant biomass harvest required to meet food demand (i.e. the overall efficiency of food production) varies considerably between scenarios but invariably dwarfs the initial demand for food energy (Figure 5.1). As expected food related biomass harvest is highest in the LE PM scenario (29.3% of NPP_{2000}) and lowest in HE LM (18.0%), with the BAU PM scenario requiring 26.5%; as in previous work the LE LM and BAU PM scenarios are close to one another, indicating that the effect of continuing current efficiency and intensification trends is roughly equivalent to a cut in consumption of livestock products to a global average of 15%.

Estimated food-driven plant biomass harvest is somewhat higher than in the original model, which gave estimates between 16% and 23% of NPP_{2000}; probably as a consequence of disaggregating the production of plant and animal products of different types as well as explicitly describing a greater diversity of waste streams. Overall efficiency of food production in LE and BAU scenarios is also slightly higher than in the previous work, at 12.3-14.5% rather than ~10%, and is as high as 14.7% in the HE PM scenario and 16.5% in HE LM.

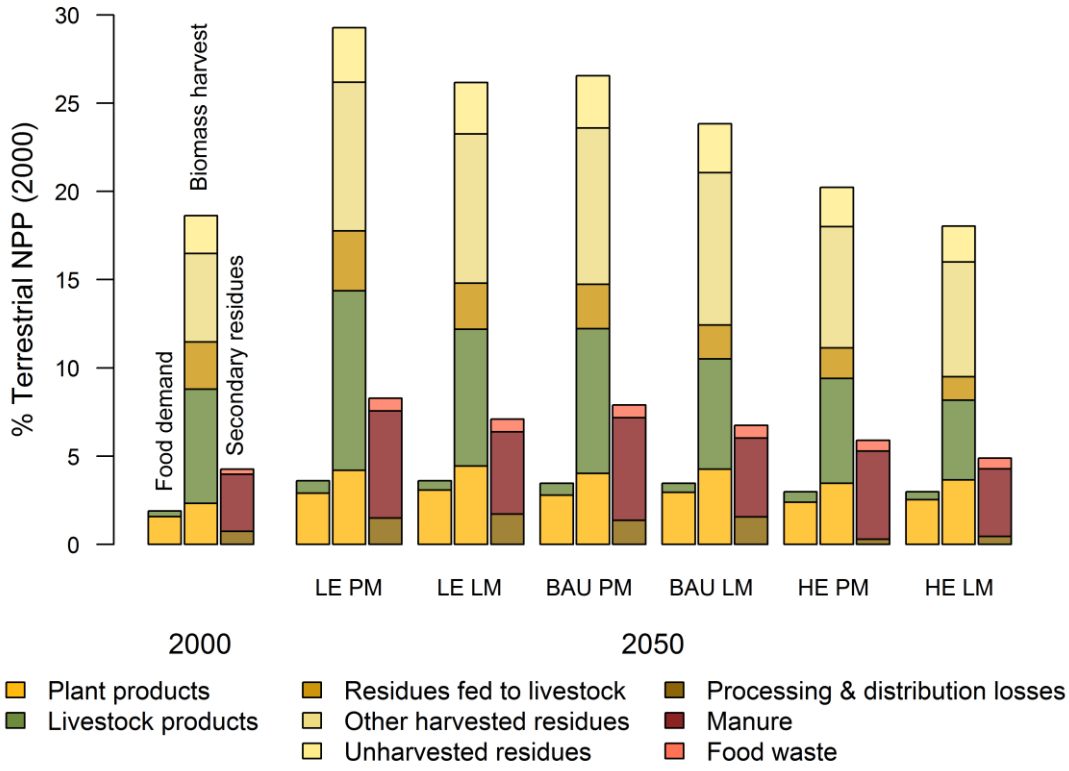

Figure 5.1: Food energy demand, associated plant biomass harvest and secondary biomass wastes as% of global aboveground NPP in the year 2000 (NPP_{2000}). The left hand group of bars shows values for 2000, while the remaining bars show values for 2050 in each of the 6 scenarios. Note this figure does not depict non-food biomass harvest categories.

Livestock production remains the dominant driver of biomass harvest, responsible for 66.0% of food-driven biomass harvest in 2000 (including residues fed to livestock, and residues generated in the production of fodder crops). When non-food biomass harvest is included, livestock accounts for 61.4% of total biomass harvest in 2000; a value very close to the 'almost 60%' calculated in the most comprehensive study of global biomass flows to date (Krausmann et al., 2008).

For all scenarios the livestock sector is slightly less dominant in 2050 than in 2000, ranging from 58.6 - 61.7% in *projected meat* scenarios and 49.0-51.8% in *low meat scenarios*. This is due to increases in conversion efficiency of the livestock groups even in the *low efficiency* scenarios; as well as the changing composition of the average diet towards animals with higher conversion efficiencies. Also of note is that in relative terms livestock drives a greater fraction of food biomass harvest in *high efficiency* scenarios than in *business as usual* scenarios (60.9% vs 58.6% (PM); 51.3% vs 49.0% (LM)): rather than

being caused by differences in the livestock sector this is largely down to the considerable reduction in *production losses* in HE scenarios, which significantly improves the efficiency of production of plant products, thus decreasing their relative contribution to biomass harvest.

Manure is by far the largest secondary waste stream, amounting to a carbon flux in 2000 equivalent to 3.2% of NPP_{2000}, and between 3.8% and 6.1% in 2050 dependent on the scenario. Manure production is highest in *low efficiency* scenarios, probably as a result of losses in the production system which mean that a larger number of animals is required to ensure the eventual consumed food product. Production and distribution losses are also significant; amounting to 0.7% of NPP_{2000} in 2000, and 0.3-1.7% in 2050. Post production food waste tends to constitute a slightly smaller actual flux (0.3% of NPP_{2000} in 2000, 0.6-0.7% in 2050), however it has a very significant effect on biomass harvest as a whole, since an increase in food waste feeds back to drive increased food production, with all the extra biomass harvest and waste streams entailed.

5.1.2 Land-use

Total land area under human management (Figure 5.2) in the year 2000 is calculated as 4.28 Gha, of which 3.96 Gha are used to produce food; a total markedly lower in FALAFEL than the 5.17 Gha estimated in Chapter 2. This is largely due to a key difference in the method used to estimate the area of pasture between the two models: the work presented in Chapter 2 begins with the FAOSTAT estimate of pasture area (3.42 Gha) and back-calculates the associated biomass consumption based on livestock feed requirements, whereas FALAFEL uses estimates of HANPP (Haberl et al., 2007) overlayed on a map of pasture area derived from satellite data as well as national and sub-national reported statistics (Ramankutty et al., 2008). The area of pasture in the Ramankutty et al. (2008) dataset (2.8 Gha) is considerably lower than the FAOSTAT estimate, for reasons stemming largely from the difficulty involved in classifying shifting land-uses in very low productivity areas and problems with the reporting of statistics in many areas. Comparing the two methodologies it seems clear that the land area which is included in the FAOSTAT database but missing from the Ramankutty et al. (2008) dataset, is minimally productive in terms of its grazing potential; for example 80% of the extremely arid land area

of Saudi Arabia is reported by FAOSTAT as pasture due to its use by nomadic peoples, while Ramankutty et al. (2008) estimate an area 3500 times smaller for the same region. The vast majority of biomass 'harvested' in grazing must therefore come from the area mapped by Ramankutty et al. (2008), making it suitable for use here.

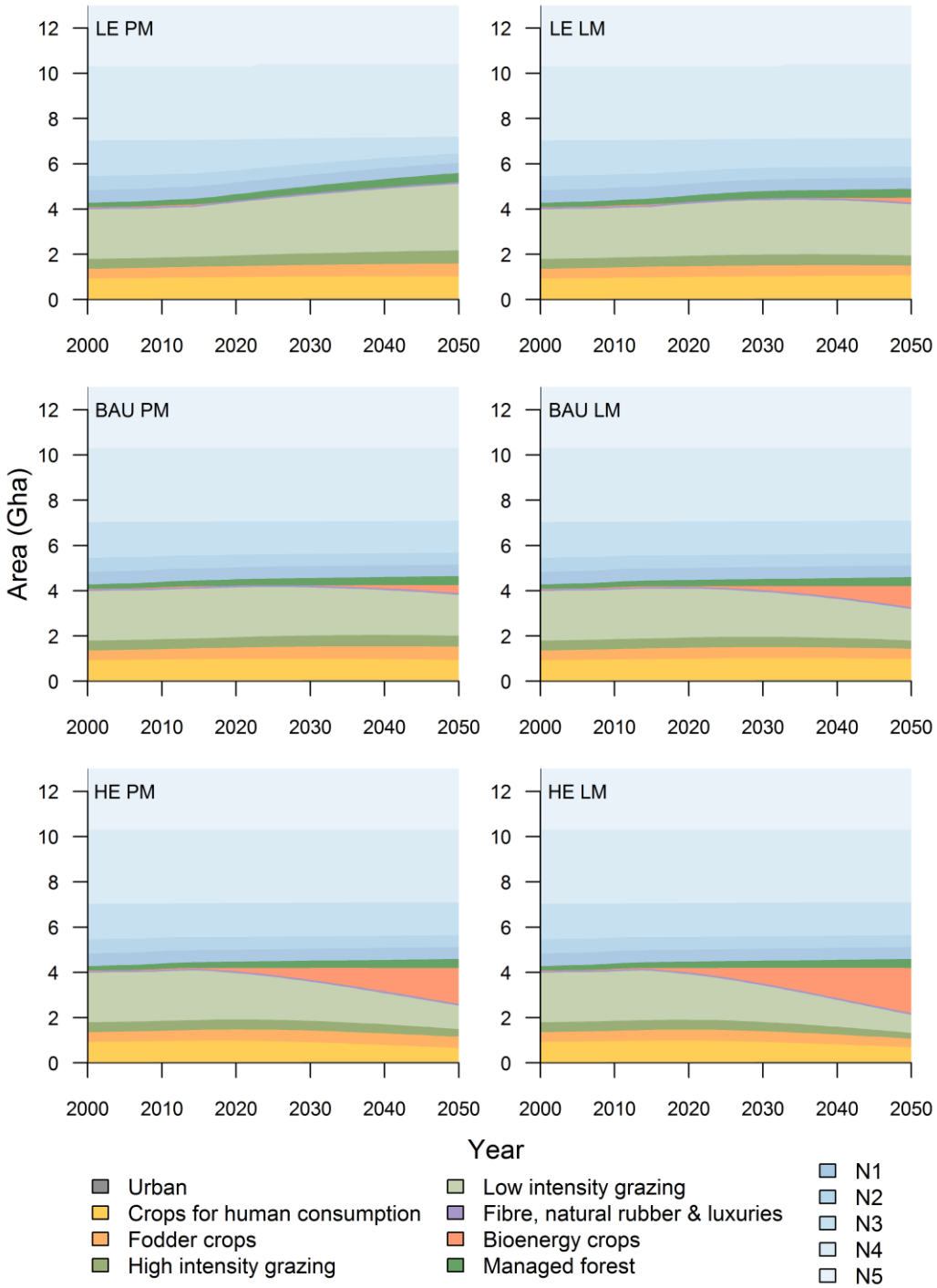

Figure 5.2: Land-use trajectories in six scenarios; land-use in the two *BAU HiBECS* scenarios is the same as in the *BAU* scenarios shown.

The bottom up calculation of the land area required by primary crops also results in a land area slightly lower than the FAOSTAT estimates, at 1.33 Gha vs. 1.46 Gha in 2000. Of this 0.9 Gha produces crops directly for human consumption, and 0.43 Gha produces fodder crops for livestock. The remaining discrepancy in total managed land area is due to small changes in the areas occupied by non-food biomass harvest, including fibre crops and rubber.

The land area required for food production (i.e. the sum of *crops for human consumption, fodder crops* and the two types of *grazing* land) increases in all scenarios by 2.2% from 2000 to 2014 to reach 4.05 Gha, at which point the scenarios diverge. The area of cropland during this period is simulated by FALAFEL to increase at an average rate of 0.42% per year; slightly higher than the 0.24% per year reported in the FAOSTAT database for 2000-2012 (2012 is the most recent date for which most FAOSTAT statistics are available). I do not make the same comparison with pasture area due to the differences in definitions described above.

In all but the *low efficiency, projected meat* scenario the land area required for food production peaks between 2014 and the mid-2030s; after which peak it begins to fall, making room in these scenarios for lignocellulosic bioenergy crops on abandoned land. Land area for food production in the *low efficiency* scenarios is considerably lower than in their equivalents in Chapters 2 & 3, since although most efficiency, intensity and waste factors are frozen at 2010 values the relative contributions of different livestock groups are still tending towards the dominance of groups with higher conversion efficiencies: in the LE PM scenario however this is not enough to prevent the continued expansion of food producing land to reach 5.06 Gha in 2050, a 28% increase between 2000 and 2050. In the LE LM scenario the improvement in average conversion efficiency in the livestock sector, coupled with the reduction in demand for livestock products causes the area of food producing land to peak at 4.37 Gha in 2035; by 2050 this has fallen to 4.15 Gha, allowing around 210 Mha of lignocellulosic energy crops. In the BAU PM scenario, likely to be the closest to the real world situation, land-use requirement for food production peaks in 2025 at 4.12 Gha, making 347 Mha available for bioenergy crops in 2050; an area very similar to the 332 Mha forecast in the *high meat, high efficiency* scenario in Chapter 2, and close to other estimates (Beringer et al., 2011; Campbell et al.,

2008; Haberl et al., 2010). The overall land-use trajectory in this scenario agrees well with that forecast in other studies, which tend to predict a small increase in managed land area in the first half of this century, but most production increases coming from intensification (McIntyre, 2009).

The remaining three scenarios are indicative of what may be achieved if special measures are implemented to reduce the land area footprint of food production. If current waste, efficiency and intensity trajectories are followed but the average consumption of livestock products is reduced, as in BAU LM, land area for food production peaks at 4.06 Gha in 2018 and falls to just 3.14 Gha in 2050, allowing for 929 Mha of bioenergy crop in 2050. Making the significant increases in *intensity* and *efficiency* and reductions in *waste* described in the HE scenarios leads to an immediate decrease in the area required for food production; both scenarios peaking in 2014 at 4.05 Gha with bioenergy crops replacing 1,578 Mha of food producing land in HE PM and 1,989 Mha in HE LM.

5.1.3 Effects on the global carbon balance

The effect of each scenario on the global carbon balance is determined by the interaction of three key carbon fluxes; emissions from land-use change (LUC), the sequestration flux generated by BECS, and the offset of fossil fuel emissions through energy generation from biomass feedstocks. The latter is not strictly speaking an actual flux of carbon, but the reduction of a fossil fuel emissions flux which is not itself represented in FALAFEL; in the context of the model it is logical to think of it as a negative flux counteracting an assumed baseline level of fossil fuel emissions.

Land-use change

Carbon emissions from land-use change, in the form of CO_2 released when vegetation in destroyed, vary significantly between scenarios (Figure 5.3 a). From 2000-2014 emissions are estimated to average 0.66 Pg C yr^{-1}, though with considerable year to year variability within the range of 0.5 - 1.2 Pg C yr^{-1}. These emissions are in the lower end of the range given by the Global Carbon Project (Friedlingstein et al., 2010); though it should be noted that FALAFEL currently accounts only for the emissions caused by the destruction of aboveground vegetation, and actual emissions are likely to be higher because they also include losses of soil carbon. The considerable increase in emissions

in 2009 and 2010 is probably as a result of the increase in area of bioenergy crops around that time, which is assumed to be divided between existing cropland and newly cleared natural ecosystems (Section 4.1.7, p109).

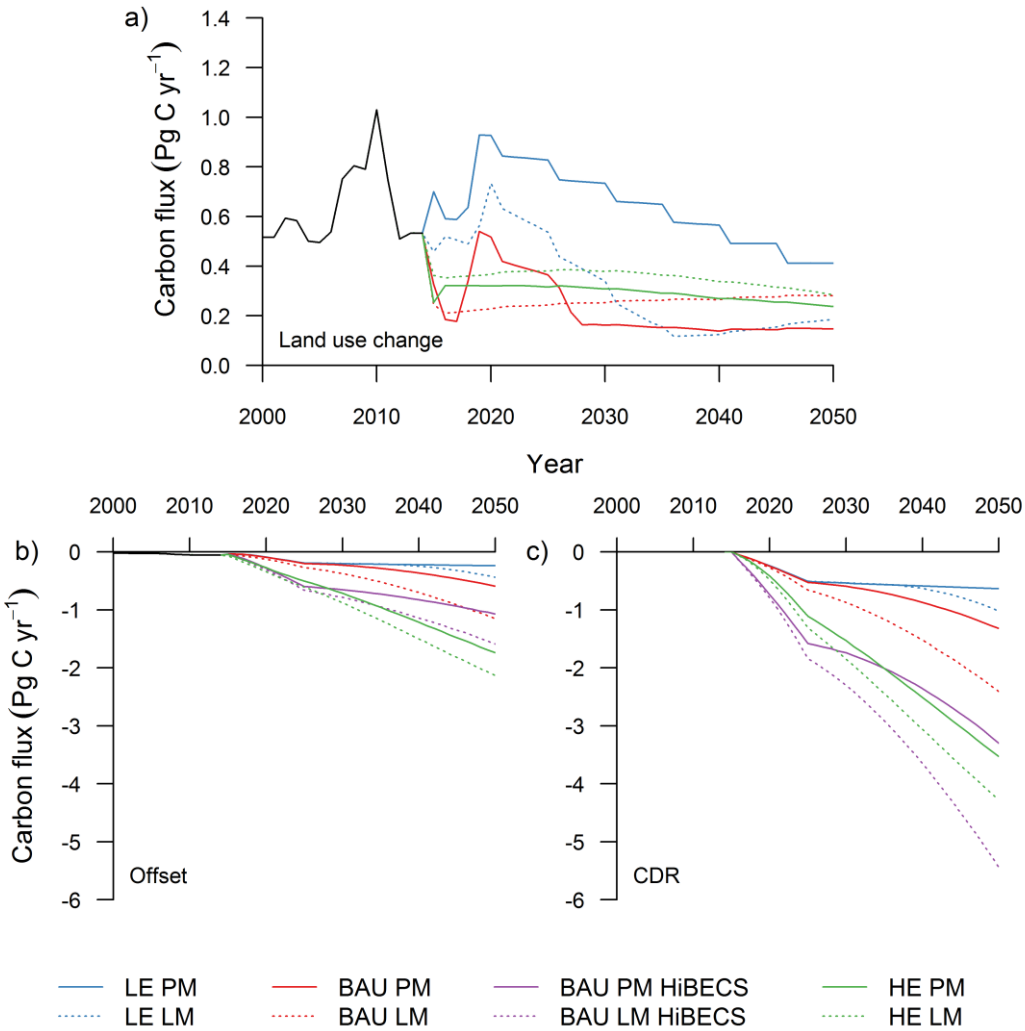

Figure 5.3: Carbon fluxes in each scenario from **a)** land-use change; **b)** offset emissions from fossil fuels; and **c)** carbon dioxide removal fluxes generated by BECS systems. Land-use change emissions from the two *BAU HiBECS* scenarios are not shown, as they are identical to those from the other *BAU* scenarios.

In the *low efficiency* scenarios, LUC emissions are initially high, due to the high rate of expansion of agricultural land into natural ecosystems. In LE PM emissions remain high, though have returned to 2000 levels by 2050 as land-use change slows slightly and is concentrated in ecosystems with lower carbon stocks. Total cumulative LUC emissions in 2050 are 43.5 Pg C in LE PM (Figure 5.5). LE LM follows a similar trajectory to a low point in 2035 when the

area of food producing land peaks; however after this point emissions begin to grow again as although only the expansion of managed forest is contributing to the removal of natural vegetation, the replacement of pasture land with bioenergy crops also begins to constitute a significant source of emissions (0.13 Pg C, 35% of total LUC emissions in 2050). Total cumulative LUC emissions in this scenario are 27.6 Pg C.

Emissions in BAU PM initially fall, as the growth in land required for food production is mitigated by replacing land initially used for bioenergy crops; once this is entirely replaced emissions resume a similar trajectory to those in LE LM. Although a larger area of bioenergy crops are converted in this scenario than LE LM, the LUC emissions produced as a result are lower (0.12 Pg C in 2050) since the differences in diet and waste production mean that in BAU PM demand for cropland is shrinking faster than that for grazing land and consequently providing more of the area on which bioenergy crops are grown. In the current model structure there is no cost to converting cropland to bioenergy crops in terms of LUC emissions, as there is no residual vegetation to be cleared. Cumulative emissions in BAU PM for 2000-2050 are 23.2 Pg C.

The pattern observed in LE LM and BAU PM is repeated in the remaining scenarios: BAU LM sees low emissions after 2014 as very little clearance of natural ecosystems is required, but by 2050 LUC emissions are double those of BAU PM at 0.28 Pg C due to the area of grazing land being turned over to bioenergy crop. Total cumulative emissions in this scenario are 23.9 Pg C. The LUC emissions trajectories in the two HE scenarios are very similar to those of BAU LM, with cumulative emissions in HE PM at 24.9 Pg C and at 27.1 Pg C in HE LM.

LUC emissions in the *low efficiency* scenarios in FALAFEL are considerably lower than in previous iterations of the model (Chapter 2), with cumulative total emissions of 27.5 – 43.4 Pg C as opposed to the previous estimate of 159.3 – 254.7 Pg C. This is a direct result of the significantly reduced LUC required to meet food demand in the newer scenarios, as described above. Emissions in the remaining scenarios are similar; at 14.3 - 29.9 Pg C Chapter 2, and 23.2 – 27.1 Pg C in FALAFEL.

Offset of fossil fuel emissions

16.0 – 132.7 EJ yr^{-1} of energy generated by the combustion of biomass is able to offset fossil fuel emissions of 0.24 - 2.13 Pg C yr^{-1} across the scenarios in 2050 (Figure 5.3 b); constituting a potentially significant contribution to mitigating emissions from fossil fuels. Offset in the LE and BAU scenarios, which are most comparable to the scenarios used in previous work, provides the equivalent of a CDR flux of 0.24 – 1.15 Pg C yr^{-1} in 2050; a range very close to the 0.46 – 1.0 Pg C yr^{-1} estimated in Chapter 2. Energy generation in these scenarios is 16.0 – 72.3 EJ yr^{-1}, which assuming a basic efficiency of energy capture at 30% puts the upper limit close to the technical potential found in other studies (Beringer et al., 2011; Chum et al., 2011; Haberl et al., 2010; Smeets et al., 2007).

Carbon dioxide removal flux

CDR fluxes also differ significantly between scenarios; in 2050 ranging from 0.64 Pg C yr^{-1} for LE PM to 5.44 Pg C yr^{-1} in BAU LM HiBECS (Figure 5.3 c). Aside from the HiBECS variants, the size of the CDR flux across the other scenarios is approximately proportional to the area of bioenergy crop available; waste and residue streams provide a significant source of BECS feedstocks in all scenarios (0.92 – 1.28 Pg C yr^{-1} in 2050), but with relatively little variation between scenarios. A full breakdown of BECS feedstocks is given in Appendix IV; most notable in comparison with previous work is that *manure* is no longer the largest source of waste biomass, presumably as a result of the disaggregation and restructuring of the livestock sector in FALAFEL. The total feedstock resource from wastes and residues is roughly half that estimated in Chapter 2, with *agricultural residues* now providing the largest source.

BECCS feeedstock provided by dedicated bioenergy crops ranges from 0 in LE PM to 4.23 Pg C yr^{-1} in HE LM (

Figure A.2), of which 90% contributes to the CDR flux due the high assumed carbon capture efficiency of BIG CC (section 2.3.3, p32). This large CDR flux is obviously dependent on the very high assumed carbon capture efficiency and in reality could be somewhat lower (Rhodes and Keith, 2005), however as in Chapter 2, this initial description of model results is based around exploring

technical potentials, and so it is reasonable to assume the higher efficiency here.

The HiBECS scenarios here indicate the very high potential under a truly concerted effort to use biomass to sequester atmospheric CO_2: Under BAU PM, in which 12.5% of harvested wastes and residues and 50% of bioenergy crops are used as BECS feedstocks, the potential CDR flux in 2050 is 1.32 Pg C yr^{-1}; in the HiBECS variant of the same scenario, in which 75% of wastes and residues and 100% of bioenergy crops become BECS feedstocks, the CDR flux in 2050 is increased by 250% to 3.30 Pg C yr^{-1}.

Net carbon flux

Combining the LUC, offset and CDR fluxes depicted in Figure 5.3 gives the net carbon flux for each scenario (discounting emissions of methane), and shows that even with the relatively high LUC emissions of the LE PM scenario, all scenarios manage to achieve net negative emissions by around 2030 (Figure 5.4). Net C fluxes in 2050 range from -0.46 to -6.74 Pg C yr^{-1}, with the net flux in the BAU PM scenario in the lower end of the range at -1.77 Pg C yr^{-1}. Both the HiBECS and *high efficiency* scenarios hugely increase the net negative flux of carbon, and it is interesting to note that that the two strategies ultimately generate rather similar net effects on the global carbon balance.

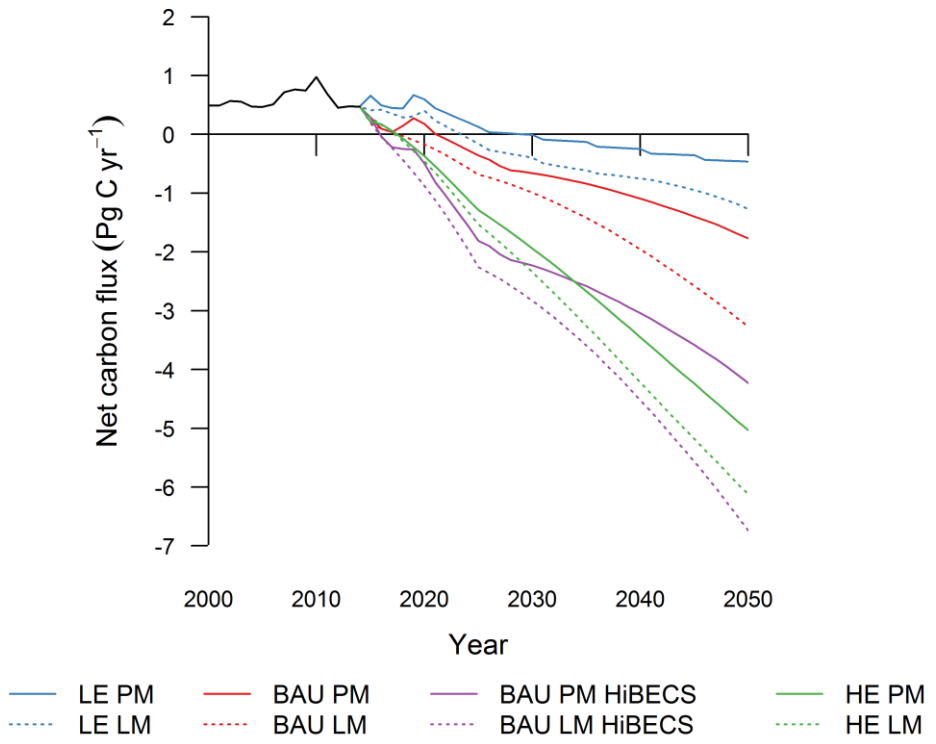

Figure 5.4: Net carbon flux from land to atmosphere for each scenario.

When these net carbon fluxes are added cumulatively (Figure 5.5), net negative emissions after 2030 in the LE PM scenario are not sufficient to offset the total LUC emissions, leading to a net positive emission to the atmosphere of 8.0 Pg C in this scenario.

In all the remaining scenarios the net cumulative carbon flux is negative, suggesting that even with minimal development of global biomass harvest systems, a concerted effort to develop a large BECS infrastructure could bring a considerable benefit in terms of rebalancing the global carbon cycle. The scale of this benefit is relatively small in LE LM (-6.7 Pg C) and BAU PM (-17.0 Pg C), but the reduction in consumption of livestock products in the *low meat* variant of the BAU scenario more than doubles the overall removal of CO_2, giving a net cumulative carbon flux of -39.2 Pg C.

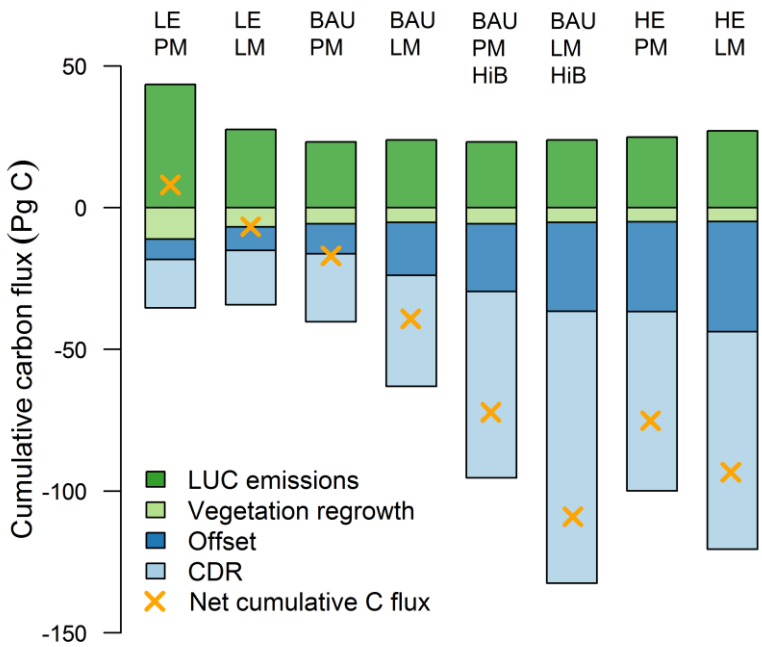

Figure 5.5: Cumulative carbon fluxes for 2000-2050 in each scenario.

The BAU HiBECS and HE scenarios achieve very high net cumulative carbon dioxide removal, with net fluxes in the PM variants of -72.2 Pg C and -75.1 Pg C respectively, and in the LM variants of -108.6 Pg C and -93.3 Pg C. These amount to the removal from the atmosphere of up to 10 years-worth of total anthropogenic carbon emissions at current rates.

Effect on atmospheric CO_2

The effect of net carbon fluxes on atmospheric CO_2 concentration (Figure 5.6) is proportional to the size of the net carbon flux in a particular timestep, and the cumulative effect of emissions in the years before subject to the decay function described in section 2.5.4 (p126). These effects should be considered in isolation as the effects of the fluxes explicitly represented in the FALAFEL model framework; they therefore amount to the potential for each scenario to deflect an assumed baseline trajectory for atmospheric CO_2 concentrations.

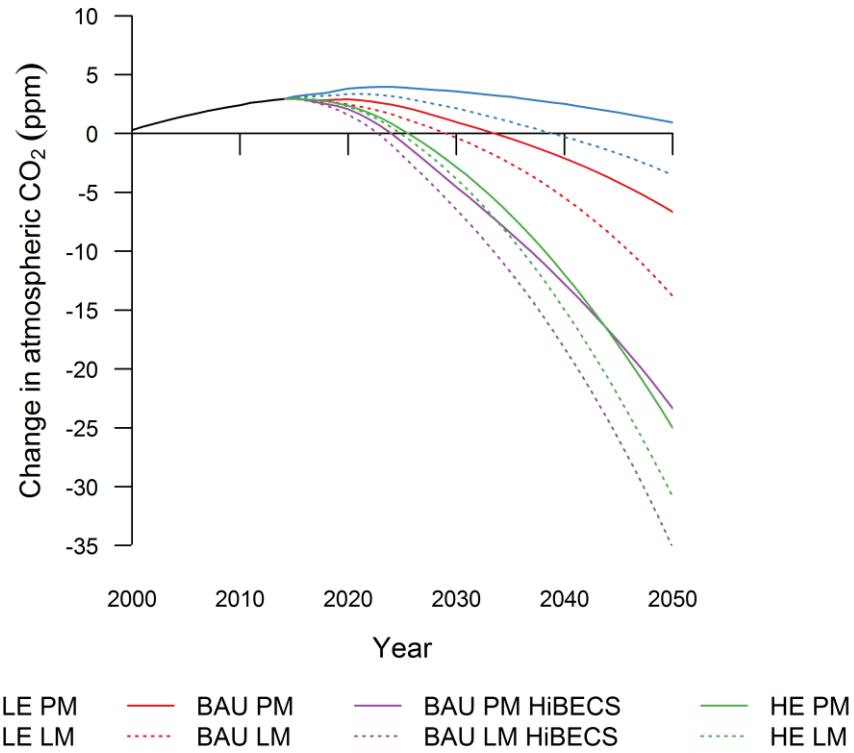

Figure 5.6: Overall effect on atmospheric CO_2 in each scenario.

LE PM, as the only scenario with net positive cumulative carbon emissions, is the only scenario which has the effect of increasing the atmospheric CO_2 concentration in 2050. Even in this scenario, the relatively large addition of around 4 ppm in the 2020s has been reduced to 0.9 ppm in 2050 due to the negative emissions generated after 2035. The range of effects on atmospheric CO_2 by 2050 in LE and BAU scenarios is much smaller here (0.9 to -13.8 ppm) than in Chapter 2, in which very high LUC emissions lead to large increases in the *low efficiency* scenarios (30.5 – 50.2 ppm) and the *high efficiency* scenarios achieved larger decreases (-13.2 to -25.0 ppm).

Only the HE and BAU HiBECS scenarios in FALAFEL manage to achieve these very significant reductions in atmospheric CO_2 concentration, giving decreases of -23.3 to -35.2 ppm by 2050.

Methane emissions

In addition to the fluxes described above, FALAFEL also calculates methane emissions from livestock production and rice paddies. Methane emissions are not currently integrated with the rest of the carbon cycle, since carbon emitted as CH_4 has a higher radiative forcing potential and a shorter residence time in the atmosphere than CO_2 and so cannot be treated in the same way. Cumulative CH_4 emissions for 2000-2050 vary little between scenarios, with the main difference driven by the reduced livestock production of the *low meat* scenarios (Figure 5.7). Total cumulative emissions range from 4.48 Pg C (HE LM) to 5.04 Pg C (LE PM), although the range of error given by the IPCC emissions factors implies very significant uncertainty in the absolute values.

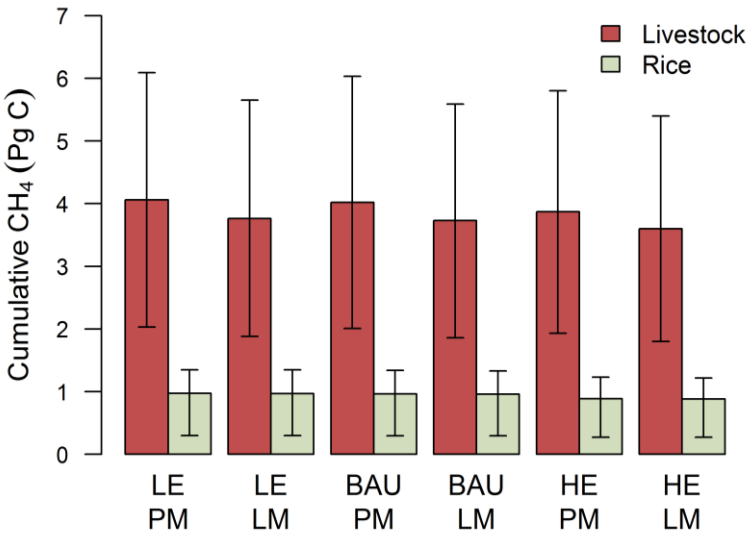

Figure 5.7: Cumulative methane emissions for 2000-2050 in each scenario. Methane emissions in HiBECS scenarios are identical to those in BAU scenarios. Error bars show the range of error given in IPCC emissions factors.

Emissions from rice cultivation are extremely consistent, varying from 0.88 Pg C (HE LM) to 0.98 Pg C (LE PM). Emissions from livestock production in *projected meat* scenarios range from 3.87 Pg C (HE PM) to 4.06 Pg C (LE PM), with the *low meat* scenario variants seeing a reduction in emissions of about 7 %, at 3.60-3.76 Pg C.

5.1.4 Human appropriation of NPP

The human appropriation of NPP (HANPP) associated with these scenarios is a means by which to gain a sense of their macroecological implications, and effects outside of the context of balancing anthropogenic carbon fluxes. HANPP in FALAFEL is the sum of the aboveground net primary productivity harvested by humans (NPP_h), and the change in global NPP caused by replacing natural ecosystems with those under human management (ΔNPP_{LUC}) (Figure 1.2 and section 4.2.10, p127). This is expressed here as a percentage of the global *potential* aboveground NPP (NPP_0), i.e. the NPP of the vegetation that would be present in a world without any human interference (Haberl et al., 2007). The NPP available to non-human ecosystems (NPP_n) is calculated as the sum of the aboveground NPP of natural biomes, and the NPP remaining after harvest in managed systems (NPP_t). HANPP in 2000 is calculated as 24.5% of NPP_0, to which NPP_h contributes 19.4%, representing a biomass harvest of 7.48 Pg C, and ΔNPP_{LUC} contributes 5.0% (Figure 5.8). These numbers agree closely with Haberl et al., who estimate NPP_h at 20.4% of aboveground NPP_0, and put ΔNPP_{LUC} at 5.2%; their study, however, also includes vegetation destroyed in anthropogenic fires as a third category of HANPP, increasing their total estimate of HANPP to 28.8%.

Although HANPP in 2050 is invariably substantially higher than in 2000 it varies considerably between scenarios, revealing an interesting story behind the intensification of biomass harvest. In LE PM, which takes a relatively low intensity approach to food production, NPP_h is lower than in most other scenarios at 31.2%. Despite this, due to the large area converted from natural biomes to managed land to provide this biomass, NPP_n is lower than in any other scenario at 65.0% of NPP_0. For the same reason ΔNPP_{LUC} remains significant at 3.8%, although it is slightly reduced as yield increases on arable land increase the overall productivity of land under human management (NPP_{act}). Together these put total HANPP in LE PM at 35.0% of NPP_0.

The LE LM scenario, in which the area of land under human management is much smaller than in LE PM, preserves a higher value of NPP_n (67.7%). NPP_h is also lower, at 30.9%, due to the reduced demand for livestock products which require a much larger harvest of biomass to achieve the same food supply: In

fact, NPP_h would be lower still if it were not for the growth and harvest of relatively high productivity bioenergy crops, mostly on land which was previously low intensity pasture.

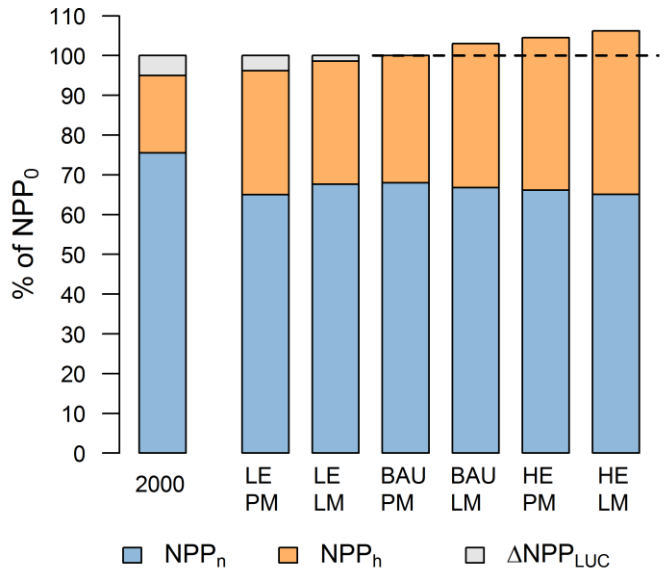

Figure 5.8: HANPP in 2050 as% of NPP_0 associated with each scenario. Scenarios in which the total calculated productivity exceeds 100% of NPP_0 display negative ΔNPP_{LUC}, the value of which is indicated by the amount by which they exceed the dotted line. In these scenarios the net effect of human land management has increased NPP_{act} beyond that of the natural potential.

This replacement of low productivity, low intensity pasture with high yielding bioenergy crops has a second obvious effect here, in further reducing ΔNPP_{LUC} to 1.4%. It is important to note, however, that this reduction of ΔNPP_{LUC} does not undo the ecological damage caused by replacing natural ecosystems with managed ones in the first place; increasing productivity on managed land increases NPP_h in proportion with the reduction in ΔNPP_{LUC}, producing no net change in HANPP and returning none of the original lost productivity to NPP_n. Total HANPP here is 32.4%.

In BAU PM, the effect of this increasing productivity on managed land is even larger, with ΔNPP_{LUC} reduced to virtually 0%, with total HANPP equal to NPP_h at 32.0%. Another effect of expanding high yielding bioenergy crops onto low productivity land also becomes apparent here: On low intensity pasture only about 22% of the estimated NPP_{act} is consumed by animals, leaving the remaining 78% as unharvested biomass (NPP_t) which contributes to NPP_n;

when this land is replaced with bioenergy crops, with NPP_t of around 12%, the contribution of NPP_t to NPP_n is correspondingly diminished. The same is true, although to a lesser extent, of transitions to bioenergy crops from all other land-use types, as the assumed yield and harvest of bioenergy crops make them the most intensely harvested land-use type in FALAFEL (i.e. that which has the lowest NPP_t). The result of this in BAU PM is that despite a peak food-producing land area some 250 Mha lower than that in LE LM, with a corresponding area of natural biomes preserved, the total NPP_n is virtually unchanged at 68.0%.

The patterns described above continue in the remaining scenarios with even more exaggerated effects, with bioenergy crops replacing low productivity ecosystems to such an extent that the global aboveground NPP is increased beyond its natural potential, expressed as negative ΔNPP_{LUC} (Haberl et al., 2007). ΔNPP_{LUC} in these scenarios thus ranges from -3.0% to -6.2%, with NPP_h increasing from 36.3% to 41.1%, and NPP_n falling from 66.8% to 65.0%. The realities and costs of achieving such increases in productivity are discussed in Chapter 6.

5.2 Sensitivity analysis

In order to explore the full range of results that can be generated using FALAFEL, I have conducted an analysis of the sensitivity of 16 key outputs to variation in 39 individual model parameters. This makes it possible to place the outcomes of the scenarios described above within a wider range of possible outcomes, and also to explore the sensitivity to input parameters which did not change between scenarios like *population*, *yield*, and *per capita calorific intake*.

5.2.1 Method

Input parameters to test (Table 5.2) were chosen based on observations made during the construction of the model, as well as those suggested by previous work (e.g. livestock conversion efficiencies, waste factors) and in the literature (e.g. population, urbanization, protection of natural ecosystems) (Wirsenius, 2000; Haberl et al., 2010; O'Neill et al., 2013 and others). Probable error ranges for each variable were then defined, based where possible on ranges reported in the literature or on the degree of variation in the available historical data; if no

range was given in the literature an assumed range of either 33% or 25% was applied, depending on the variable.

Table 5.2: Input parameters tested in the sensitivity analysis, with default values and range of variation.

Input parameter	Default (2050)	Range
Population	9,550,945,000	UN forecast high and low scenarios
Per capita food supply (Kcal cap^{-1} day^{-1})	3,353	±20%
Contribution of livestock products	19.6%	±25%
Reduction in food waste	25%	±60%
Food waste available as animal feed	25%	±60%
Distribution losses	10%	±50%
Processing wastes	25%	±60%
Yield increase	Projected increase	±75%
Crop residues removed from field	Projected value	±20%
Livestock intensification (energy from non-grazing)		
Ruminants/other meat	50%	±15%
Dairy	42.5%	±15%
Pigs	90%	±10%
Poultry meat	90%	±10%
Eggs	90%	±10%
Livestock conversion efficiency		
Ruminants/other meat	2%	±50%
Dairy	9.9%	±25%
Pigs	13%	±25%
Poultry meat	16.3%	±25%
Eggs	14%	±25%
Feed from agricultural residues (% of non-forage)		
Ruminants/other meat	25%	±33%

Dairy	25%	±33%
Pigs	5%	±33%
Feed from food waste		
Pigs	30%	±50%
Feed from processing residues		
Ruminants/other meat	50%	±33%
Dairy	50%	±33%
Pigs	15%	±33%
Poultry meat	11%	±33%
Eggs	11%	±33%
Grazing intensity	35%	±33%
Crop expansion into N1 (% of total expansion)	12.1%	±50%
Pasture expansion into N1 (% of total expansion)	5.3%	±100%
Minimum preservation of biome		
N1	25%	±95%
N2	25%	±95%
N3	25%	±95%
Residues diverted to BECS	37.5%	±100%
Bioenergy crop use in BECCS	50%	±100%
Implementation date for BECS	2025	±10 years
Implementation time	10 years	±10 years
Urbanization	SSP2	~range of SSP scenarios

The output variables selected fall into three broader groups: Land-use, consisting of crop area, pasture area, total food producing area and bioenergy crop area: Carbon fluxes, comprising cumulative LUC emissions, cumulative CDR flux, cumulative net C flux and effect on atmospheric CO_2, and also including annual methane emission and cumulative methane emission: and macroecological indicators, consisting of NPP_h, ΔNPP_{LUC}, NPP_n, HANPP, average NPP_t and average NPP of natural biomes.

In order to simultaneously test sensitivity to multiple varying input parameters, I used a latin hypercube sampling approach to generate a matrix containing 300 possible values for each of the 39 parameters, randomly varying within the defined error range. The effect of this is essentially to create 300 randomly generated scenarios, in which the 39 parameter values vary both randomly and independently of one another. These scenarios were then run through the model, each time recording the values of the 16 selected output variables in 2010, 2030 and 2050. Results from the model runs were then used to calculate mean values at these three dates, as well as the 90% confidence interval.

Multiple linear regression analysis was used to statistically determine the sensitivity of each output to variation in all possible inputs. This was repeated for output values in 2030 and in 2050, in order to compare how the sensitivity of outputs might change over the course of the modelled period. A 'maximal' model to describe each output variable was defined by including all possible predictor (input) variables: predictor values that can be said with certainty to be irrelevant to a particular output variable are excluded from the maximal model, for example the proportion of waste streams diverted to BECS cannot possibly influence the area of cropland, and so is not included in the maximal model for *crop area*. A manual stepwise approach was then use to sequentially remove non-significant predictors ($P > 0.05$) until a minimum adequate model was reached, in which all remaining predictors have a significant effect on the value of the output variable, or until removing a predictor has a significant effect on the model's adjusted R^2 value. Tables showing the top 10 predictors of each output variable for 2030 and 2050 are given in Appendix V, ranked by t-value to indicate the magnitude of their effect.

5.2.2 Results

Area of managed land

The area of *cropland*, which includes crops for human consumption and also fodder crops, shows considerable variation in 2050 (Figure 5.9 a); with the potential either to increase or decrease from a starting area of 1.33 Gha in 2000. The 90% confidence interval gives an area of 0.98 – 1.83 Gha in 2050, with the mean of all model runs increasing to 1.48 Gha in 2030 before falling back to 1.39 Gha in 2050. The area of cropland in the LE LM scenario in fact

follows this trajectory almost exactly, with those in the LE PM, and both BAU scenarios falling just above, though close to, the mean. The area of cropland in the *high efficiency* scenarios falls very much in the lower end of the possible range predicted by the sensitivity analysis.

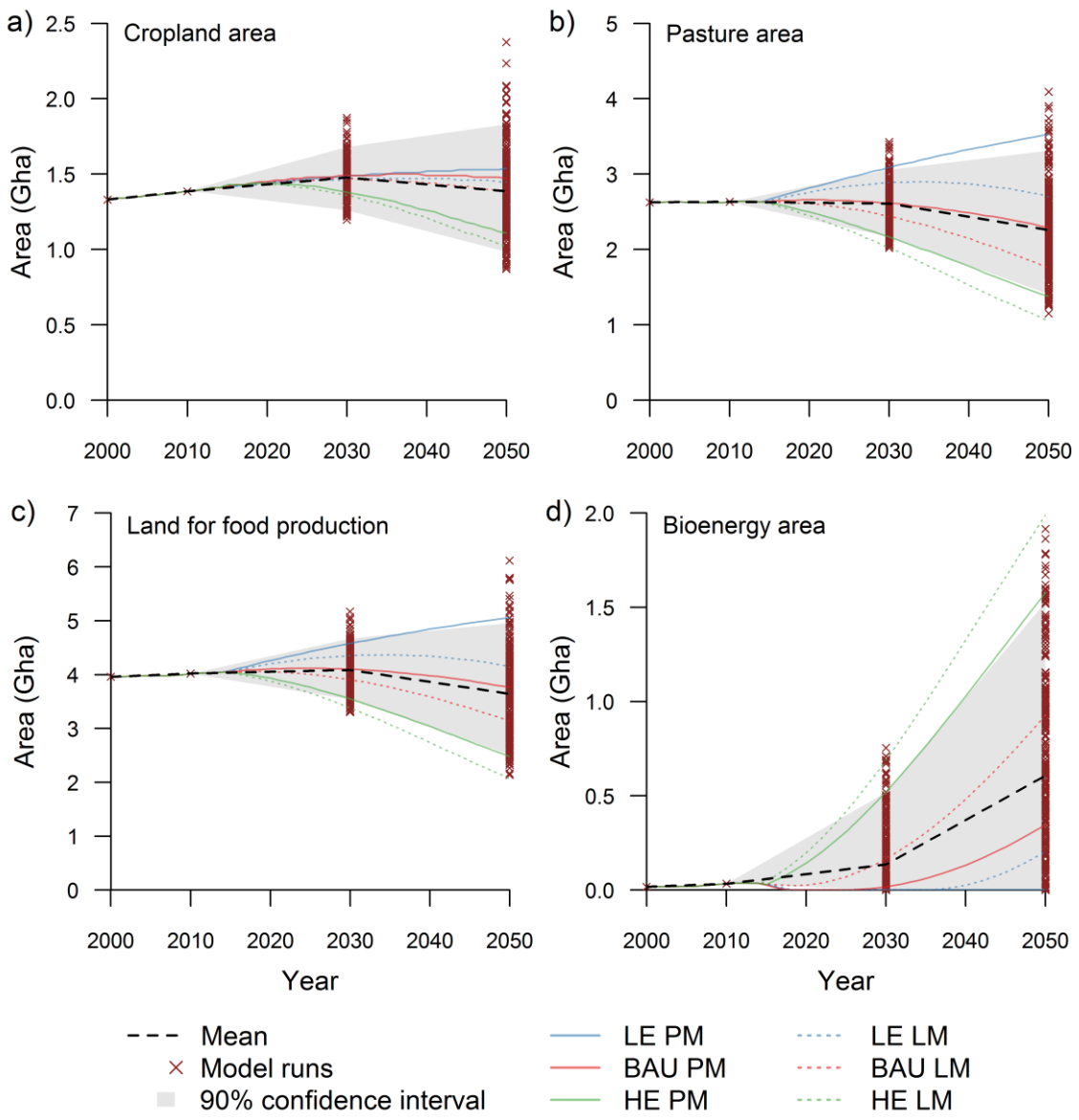

Figure 5.9: Sensitivity analysis results for key land-use outputs, with results from the scenarios also shown.

Multiple linear regression analysis suggests that the total calorific energy demand of the average diet is the most powerful predictor of cropland area in both 2030 and 2050 (Appendix V, pXI). Following this, the rate of crop yield increases is the second most important influence; supporting the argument that

closing yield gaps is an important way to meet food demand while limiting environmental damage (Mueller et al., 2012).

Population is, unsurprisingly, the next most significant driver in both years. After population, in 2030 the diet of dairy animals are also important predictors of cropland area; both in terms of the ratio of grazing to fodder (*intensification*), and the contribution of agricultural residues to that fodder. These are followed by the total contribution of livestock products to the average diet as sixth most important driver, though of similar influence are the conversion efficiency and diets of ruminants bred for meat production; factors that directly influence the amount of fodder crops required each year. In 2050 the key drivers are largely the same, however the contribution of livestock products to diet gains even more importance as the fourth most influential predictor; similarly the level of post-production food-waste moves from eleventh in 2030 to sixth in 2050.

As shown in the scenarios described in section 5.1, the area of pasture necessary for grazing animals also has the potential either to expand or to shrink from an area of 2.63 Gha in 2000. The 90% confidence interval given in this sensitivity analysis puts the likely range in 2050 at 1.41–3.31 Gha, with the mean predicted area remaining approximately constant to 2030 before falling to 2.26 Gha in 2050 (Figure 5.9 b). Interestingly, only the LE LM and BAU scenarios forecast pasture areas which fall within the 90% confidence interval, serving to highlight the roles of the worst- and best-case scenarios in demonstrating the extremes of possibility, rather than as likely outcomes.

The statistical analysis indicates dietary demand for livestock products as the most important predictor of pasture area in 2030 and 2050, followed by the total energy demand of the average diet (Appendix V, pXI). After these, the conversion efficiency of ruminant meat and dairy products have the largest effect, followed by their intensification factors.

Total food producing land area (Figure 5.9 c) is the sum of cropland and pasture areas. As such it is driven by the same factors, with average dietary energy demand, contribution of livestock products, global population and the conversion efficiency of dairy and ruminant meat production the most influential predictors, and crop yields and food waste also playing a role (Appendix V, pXII). The average area occupied by food production increases slightly from

3.96 Gha in 2000 to 4.10 Gha in 2030, before falling to 3.64 Gha in 2050; the 90% confidence interval shows similar uncertainty to the areas of cropland and pasture, ranging from 2.52 to 4.96 Gha in 2050.

Bioenergy production is permitted in FALAFEL only when the area required for food production is shrinking; the area of bioenergy crop (Figure 5.9 d) is thus very closely related to the factors affecting the area of food producing land, with essentially the same model parameters having the greatest influence (Appendix V, pXII). The average area of bioenergy crops increases significantly between 2030 and 2050, growing from 0.14 to 0.61 Gha, with the 90% confidence interval giving a range of 0 – 1.53 Gha in 2050. The mean predicted area sits between those forecast in the two *business as usual* scenarios, slightly lower than the 0.78 Gha estimated as the global technical potential in the IPCC Special Report on Renewable Energy and Climate Change Mitigation (Chum et al., 2011) and well within the range of 0.23 – 0.99 Gha suggested by Erb et al. (2012), which also considered different dietary scenarios. The area of bioenergy crops produced in the *high efficiency* scenarios sits outside the 90% confidence interval, and in fact that given in HE LM is higher than any produced by the sensitivity analysis, suggesting that it is very unlikely to be achievable.

Carbon fluxes

Cumulative land-use change emissions are affected both by expansion into natural biomes and by changes between managed land types. The results of the scenarios suggested that expansion into natural biomes was likely to be a significant factor in the early part of the modelling period, with changes between different categories of managed land becoming significant later; particularly when pasture is replaced with bioenergy crops. The results of the sensitivity analysis support this as mean LUC emissions continue to accumulate between 2030 and 2050, growing from 20.1 to 28.6 Pg C, even though the area of land required for food production shrinks during this time (Figure 5.10 a). The statistical analysis highlights average daily calorie consumption, population, contribution of livestock products, bioenergy crop area and crop yield as key drivers of LUC emissions, followed by factors associated with ruminant and dairy feed demand, and post production food waste (Appendix V, pXIII). These are virtually identical in 2030 and 2050, offering no further insight into the

differences in drivers; however all the factors listed could be associated with either expansion of managed land area or with changes between managed land types.

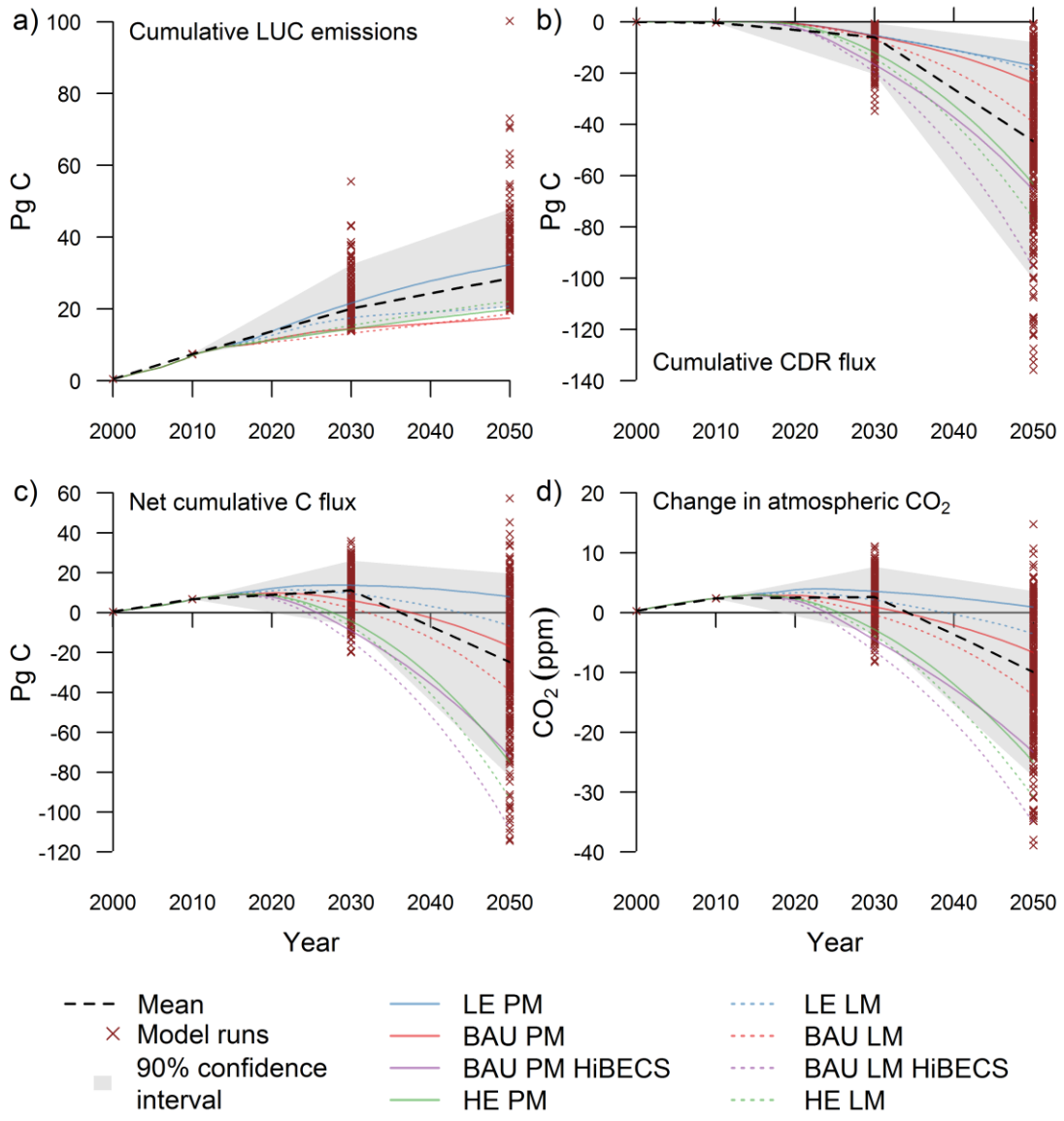

Figure 5.10: Sensitivity analysis results for cumulative carbon flux outputs, with scenario results also shown.

Although estimated mean cumulative LUC emissions are fairly modest in 2050, the 90% confidence interval puts the total between 20.8 and 47.8 Pg C, with the largest predicted values as high as 100.2 Pg C. Despite this potential for very high emissions, surprisingly the LUC emissions generated in the scenarios described in section 5.1 all sit well towards the bottom of the range, or even outside it in the case of the BAU scenarios. Only LE PM generates emissions

close to the mean of the sensitivity analysis, why this should be the case is unclear at this point.

Cumulative carbon dioxide removal (CDR) is composed of the negative flux generated by BECS alongside the carbon emissions offset by energy generated from biomass feedstocks. As such CDR is strongly correlated with bioenergy crop area, and also with the collection of waste and residue streams as BECS feedstocks and with the fraction of bioenergy crops processed in BECCS systems (Appendix V, pXIII). Also important are the year in which BECS strategies are implemented and the time taken for them to reach full capacity, with the later and slower the implementation the lower the CDR flux. Cumulative CDR ranges from 0 - 135.7 Pg C in 2050, and even in 2030 reaches as much as 34.8 Pg C (Figure 5.10 b). The 90% confidence interval puts the more probable range in 2050 at 7.9 – 100.0 Pg C in 2050, and although this clearly still shows a very high degree of uncertainty it does suggest that it might be possible to generate very considerable CDR fluxes using biomass. The spread of CDR fluxes produced by the eight scenarios is very similar at that shown by the 90% confidence interval, indicating that in this respect the scenarios are a good illustration of the range of results that FALAFEL is able to produce.

The cumulative net carbon flux, as the sum of LUC emissions and CDR flux, also shows a high degree of uncertainty, with the 90% confidence interval at -82.3 – 19.7 Pg C in 2050 (Figure 5.10 c). The mean result shows a net emission to the atmosphere of 11.2 Pg C for 2000-2030 which is more than compensated by negative fluxes after 2030 to reach net removal of 24.8 Pg C by 2050. This is a similar trajectory to those produced by the *business as usual* scenarios, sitting midway between the PM and LM variants, with the LE scenarios also sitting well within the 90% confidence interval. The HE and HiBECS scenarios produce cumulative carbon fluxes very much towards the extremes of potential net removal, as much as a result of their relatively low LUC emissions as their very large CDR fluxes. In 2030 the key drivers are those associated with LUC emissions, such as *food supply*, *population*, *yield*, *contribution of livestock products* and factors describing livestock diets, although the harvest of residue streams as BECS feedstocks also plays a role (Appendix V, pXIV). By 2050 all the main drivers are those associated with the production and processing of bioenergy and BECS feedstocks.

Since cumulative net carbon flux is the sole driver of the ultimate effect on atmospheric CO_2 concentration, the key drivers and the pattern shown in the results are identical to those described above (Figure 5.10, Appendix V, pXIV). The mean trajectory for CO_2 sees a net addition to the atmosphere of 2.6 ppm in 2030 converted to net removal of 9.9 ppm in 2050, with the 90% confidence interval between -27.4 and 3.6 ppm.

Methane emissions in FALAFEL are caused by livestock and rice production, with 81.4 Mt C emitted as CH_4 in 2000 (Figure 5.11 a). Mean annual emissions in the sensitivity analysis peak in 2030 at 102.8 Mt C before falling to 97.9 Mt C in 2050, presumably largely as a result of the changing mix of livestock products, with low methane emitters such as chickens and pigs replacing ruminants in the average diet. As in other outputs sensitive to the quantity and method of livestock production the uncertainty is quite high, with the 90% confidence interval putting the likely annual emission anywhere between 70.5 and 129.5 Mt C in 2050. In terms of cumulative emissions this amounts to the addition to the atmosphere as methane of 4.4 – 5.5 Pg C in 2050, with the mean accumulation at 4.9 Pg C (Figure 5.11 b).

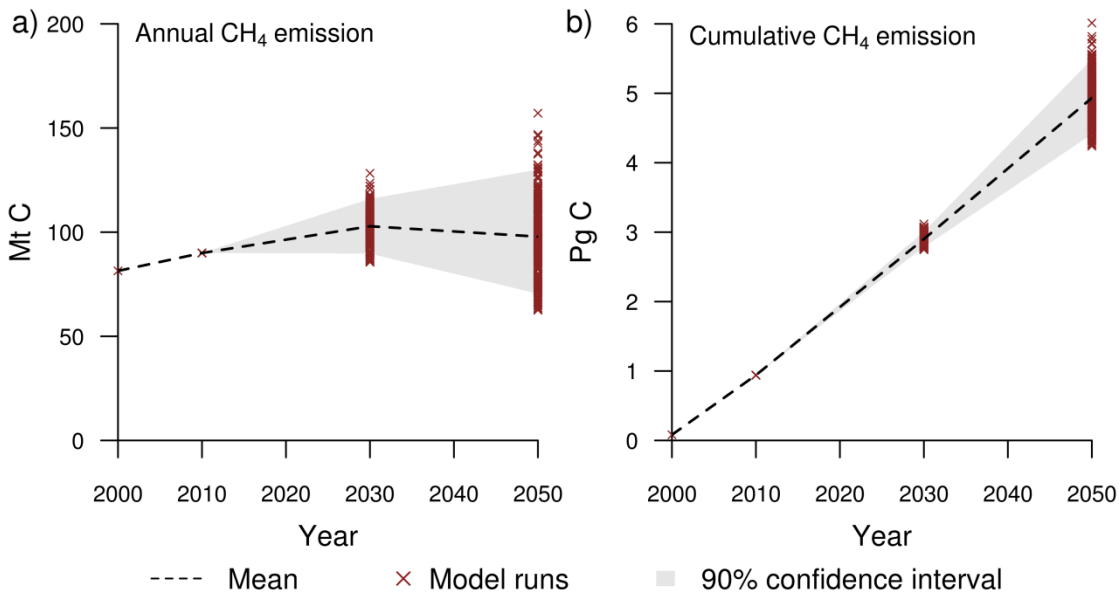

Figure 5.11: Sensitivity analysis results for CH_4 fluxes.

Key predictors of methane emissions in the statistical analysis are *per capita food supply*, *contribution of livestock products*, *population*, *food waste* and *crop yield*, followed by dairy and ruminant *intensification factors* and *processing* and

distribution losses (Appendix V, pXV). The role of crop yield is significant here, as the current method for calculating CH_4 emissions from rice production is based on an emission factor per unit area, meaning that as yields increase the methane emission per unit mass of rice decreases.

Macroecological factors

The fraction of NPP_0 harvested by humans (which includes both used and unused residues) is expected to increase as the growing global population and improved average diet drive demand for increasing biomass harvest. Mean NPP_h in the sensitivity analysis increases approximately linearly, rising from 19.4% of NPP_0 in 2000 to 27.2% in 2030 and 33.1% in 2050 (Figure 5.12 a). The most influential driver in both 2030 and 2050 is the area of bioenergy crops (Appendix V, pXVI), probably explained as much by the variability in biomass harvest for energy crops as by the magnitude of harvest, with biomass for food and other uses displaying greater consistency between model runs. Following bioenergy area the predictors with the highest t-values are the usual trio of *total food supply*, *population*, and *contribution of livestock products*, followed by ruminant and dairy *conversion efficiency* and feed factors, and *crop yield. Food waste* also becomes significant in 2050. NPP_h in 2050 varies between 25.2% and 41.1% of NPP_0, with the 90% confidence interval at 28.3-38.2%, suggesting that although humans will certainly need to harvest a great deal more biomass in the future than we do currently, pathways exist which limit that growth.

As shown in section 5.1 ΔNPP_{LUC}, or the anthropogenic difference between actual NPP (NPP_{act}) and potential NPP (NPP_0) (Figure 5.12 b), is also extremely sensitive to the production of bioenergy crops as defined by FALAFEL, which tends to have the effect of significantly increasing the productivity of otherwise marginal land. Following *bioenergy crop area*, the statistical analysis highlights ruminant and dairy *intensification*, *conversion efficiency*, and other livestock dietary factors as key drivers, with *contribution of livestock products to diet* also featuring prominently (Appendix V, pXVI). ΔNPP_{LUC} in 2050 varies very considerably between model runs, with some behaving similarly to the LE PM scenario in which ΔNPP_{LUC} barely changes from its initial value of 5.0% and others following the trajectory of the BAU LM and HE scenarios in which ΔNPP_{LUC} becomes negative between 2030 and

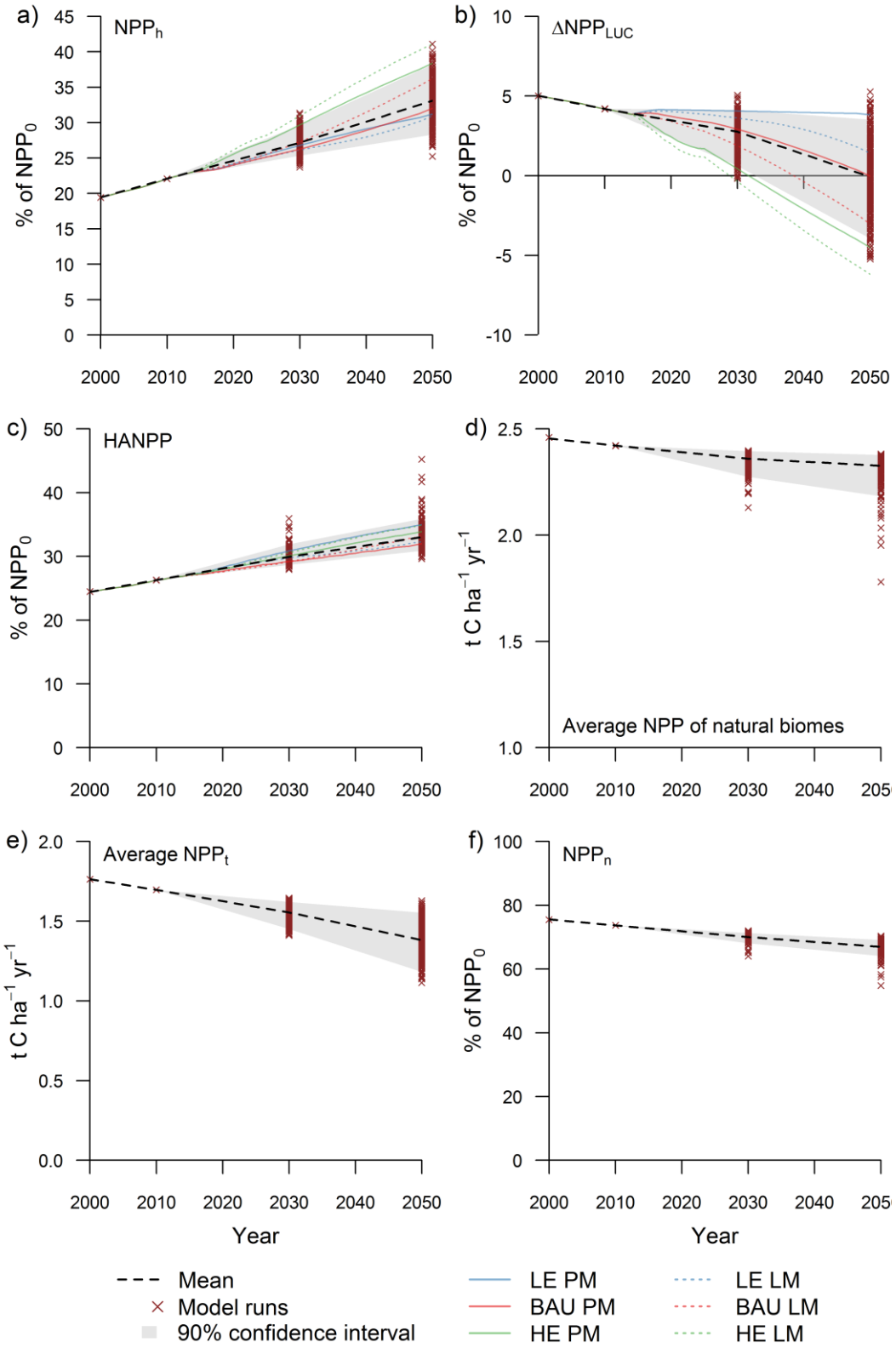

Figure 5.12: Sensitivity analysis results for macroecological indicators, with results from the scenarios also shown.

2050, signifying that human activity has actually increased global NPP_{act} beyond its potential. The 90% confidence interval reflects this multitude of

possible outcomes, putting ΔNPP_{LUC} between -3.9% and 3.5% in 2050, although this does suggest that at the least increases in productivity on managed land will undo some of the current suppression of biological potential; the mean result follows BAU PM in effectively negating this suppression, putting ΔNPP_{LUC} at -0.1% in 2050. This trajectory shows at a global level a transition already observed in a study of national-scale HANPP in six countries including the UK (Krausmann et al., 2012): Typically initial population growth and early industrialization drive LUC, which increases HANPP via ΔNPP_{LUC}; in the later stages of industrialization increases in yield lead to increased production mass per unit area, diminishing ΔNPP_{LUC}.

Mean total HANPP (Figure 5.12 c) increases from 24.5% in 2000 to 33.0% in 2050, showing remarkably low uncertainty with the 90% confidence interval at 30.9 – 35.9%. This apparent lack of variability seems surprising considering the high uncertainty displayed by its constituent parts, NPP_h and ΔNPP_{LUC}, and is likely being confounded by the presence of negative values of ΔNPP_{LUC}. Given that NPP_n is relatively well constrained, much of the high degree of variability in NPP_h must be caused by changing intensity of production and harvest over the same areas of land; since scenarios which display very high values of NPP_h also tend to feature negative ΔNPP_{LUC}, and vice versa, it seems that most scenarios are likely to reach approximately the same value of HANPP but could vary significantly in their intensity of production and actual quantity of biomass harvested.

Whether this affects the value of HANPP as a measure of the impact of human biomass harvest on a global scale is an interesting point of discussion. Viewed in isolation it could be argued that since increases in NPP_h achieved by increasing the productivity of managed land come at minimal cost in terms of reduction in NPP_n, effectively decoupling HANPP from increasing biomass harvest (Krausmann et al., 2012). Since, however, the purpose of HANPP as a metric is to integrate economic and socio-ecological activity with ecosystem and Earth system processes, it would be a mistake to neglect the extra cost in terms of energy and inputs required to reduce ΔNPP_{LUC} or even turn it from a suppression to an enhancement of potential net primary productivity. HANPP alone does not convey these externalities, and so can conceal some impacts of intensification.

The average NPP of the remaining natural biomes, weighted by biome area, is a generic indicator of the impact of land-use change on natural ecosystems which is sensitive to both the total area of managed land and to the preference for particular biome types when managed land area expands. As expected, since most managed land expansion occurs on natural biome classes N1:N3, the weighted average NPP of natural biomes decreases in all model runs from the initial 2.46 t C ha^{-1} yr^{-1}, with a mean result in 2050 of 2.33 2.46 t C ha^{-1} yr^{-1} (Figure 5.12 d). The mean is very much in the upper end of the distribution however; with the 90% confidence interval at 2.18 – 2.38 t C ha^{-1} yr^{-1}, and one run producing a result of 1.11 t C ha^{-1} yr^{-1}. Very low figures such as this imply the destruction of substantial portions of the more productive ecosystems. The statistical analysis suggests much the same dominant factors as those that control the area of managed land and the harvest of biomass (Appendix V, pXVII).

The average NPP that goes unharvested on land managed by humans (NPP_t) also decreases in all model runs, the mean decreasing from 1.76 t C ha^{-1} yr^{-1} in 2000 to 1.38 t C ha^{-1} yr^{-1} in 2050 (Figure 5.12 e), with the 90% confidence interval puts NPP_t at 1.18 – 1.55 t C ha^{-1} yr^{-1}. As discussed in the earlier scenarios, the presence and extent of bioenergy crops are a major influence on average NPP_t, with *grazing intensity*, *crop yield*, and the intensification of ruminant and dairy production also important (Appendix V, pXVIII). NPP_t is an important indicator of the intensity of biomass harvest, and closely linked with important ecosystem services like biodiversity, as discussed in Chapter 3.

NPP_n, the total portion of NPP_0 which goes unharvested by humans, shows relatively little variation between individual model runs (Figure 5.12 f); as it did in the earlier scenarios. Mean NPP_n shows a decline from 75.5% in 2000 to 67.0% in 2050, due to a combination of destruction of natural biomes and intensification on managed land, with the 90% confidence interval at 64.1 - 69.1%. Some model runs see NPP_n as low as 54.8% however, showing that certain pathways have truly disastrous results, removing almost a third of the remaining unharvested vegetation globally. Key drivers are the usual culprits of *food supply*, *bioenergy area*, *population*, *contribution of livestock products to diet*, and ruminant and dairy factors (Appendix V, pXVII).

Bioenergy vs. Afforestation: Sensitivity to a non-continuous variable

In addition to the input parameters tested above, FALAFEL takes some inputs as binary choices which were not possible to incorporate into the sensitivity analysis. Of particular interest is the choice between growing lignocellulosic bioenergy crops or afforestation on land vacated by food production (Section 4.2.7, p124), especially in light of the ecological and soil quality implications of the very high intensity of harvest associated with bioenergy crop production (Chapter 3). In order to test the model sensitivity this choice I therefore repeated the sensitivity analysis for each option and contrasted the results. The full results of these analyses are shown in Appendix IV; here I briefly describe the key differences they present.

Afforestation on average reduced the cumulative CDR flux by about 40% in comparison with model runs growing bioenergy crops; producing a mean cumulative CDR flux of 26.9 Pg C by 2050, much of which is still generated by the unaffected BECS processing of waste streams. Although LUC emissions were also slightly lower under afforestation, this still meant a significantly reduced cumulative net C flux of -13.1 Pg C by 2050; a reduction of 47%, with a corresponding difference in effect on atmospheric CO_2. While the CDR flux achieved is lower than in bioenergy/BECS scenarios, the macroecological effects are also lower, with mean HANPP reduced by almost 2 percentage points to 31.2%. Although afforestation also frequently drives a considerable increase in the productivity of vacated land, the much lower intensity of harvest means that mean NPP_t is 9.6% higher than in bioenergy systems at 1.51 tC ha^{-1} yr^{-1}, implying a much lower negative impact on biodiversity and other ecosystem functions. It should be noted, however, that this implication is frequently not borne out by reality, and long-term afforestation projects in northern China have shown that if afforestation policies are not sensitive and flexible about planting native species with appropriate water and nutrient demands, they can cause soil degradation and other ecological problems (Cao et al., 2010; Hu et al., 2007).

Sensitivity to an alternative dietary scenario

A second comparison of sensitivity analyses was also made in order to assess the effect of an apparent nonlinearity in some of the historical data used to

forecast the global average diet. As pointed out in Section 4.1.1 (p87) and Figure 4.2, the historical trend for contribution of livestock products to the average diet appears to ramp up sharply in around 1990, showing a much steeper gradient for 1990-2010 than for 1970-1990. This change seems to be approximately mirrored in some other historical trends, for example Figure 4.4 (p93) suggests that the increase in contribution of *oilcrops* becomes more rapid in 1990-2010, as well as a possible decrease in the importance of wheat during this time. I therefore ran a second sensitivity analyses to compare the effect of using the 1990-2010 trends as inputs rather than the original 1970-2010 trends.

Unsurprisingly, due to the higher contribution of livestock products, using the 1990-2010 trends produces an increase of 12.2% in the mean land area required for food production in 2050, putting it at 4.10 Gha, largely due to increases in pasture area. Significantly, although mean food producing land area declines after a peak of 4.35 Gha in 2030, it does not fall below the 2000 area by 2050, as the mean result using the 1970-2010 trend did. Other outputs associated with the change in required land area and increased livestock numbers are also affected, with a 22% increase in cumulative LUC emissions and a 15% increase in cumulative CH_4 emissions. Area of bioenergy crops (or afforestation) is also negatively affected, with the mean decreased by 27.2% to 0.44 Gha (vs. 0.61 Gha using 1970-2010 inputs), and consequently the cumulative CDR flux is also reduced by 13.0% to 40.6 Pg C in 2050. Many of the macroecological factors are also affected, with total HANPP increased by 3.4% in 2050 to 34.2% of NPP_0, and NPP_n reduced by 1.7% to 65.8%.

Chapter 6: Discussion

Abstract

In this chapter I give a brief summary of the key differences between the scenarios discussed in Chapter 5, by examining the network of biomass flows depicted in FALAFEL. The major inefficiencies in the food production system are discussed, with the livestock sector by far the most significant, and the scale of carbon fluxes in anthropogenic biomass flows are shown to be equivalent in scale to entire natural biomes. This latter comparison is useful in indicating the sheer scale of the effort needed to produce the bioenergy feedstocks described as achievable in other studies.

Following this I discuss some key elements that are currently missing from FALAFEL, and ways in which the model could be improved in order to develop a more nuanced understanding of the drivers and impacts of global biomass harvest. Chief among these are; breaking down the model into geographical regions with differing demands and production capacities; disaggregation of waste streams and tracing their carbon content all the way through their lifecycle, particularly focussing on methods of waste disposal; and addressing FALAFEL's tendency to systematically underestimate carbon emissions from land-use change.

6.1 A global perspective of biomass flows.

The work described in the preceding chapters helps to build a picture of global anthropogenic biomass flows, and places them in the context of the aboveground terrestrial carbon cycle. My work has built on that of others (particularly Wiresius (2000); Smeets et al. (2007); Krausmann et al. (2008); Erb et al. (2009a) & Haberl et al. (2007)), and in fact FALAFEL does not match its nearest equivalent (the model developed by the Institute of Social Ecology, University of Klagenfurt (Krausmann et al., 2008; Erb et al., 2009a, 2012)) in terms of the complexity of its regional disaggregation, and tracing of certain biomass streams. FALAFEL has, however, attempted to integrate the human harvest of biomass more strongly with Earth system processes and climate change mitigation approaches by focusing on the carbon content rather than the energy content of biomass. FALAFEL also has an advantage in terms of its temporal resolution, mapping biomass flows on an annual basis rather than the single output for 2030 or 2050 produced by other models. This is of particular importance when tracing the cumulative effects of emissions to the atmosphere, where influences build up or decay over time, and in terms of the year to year variability and changing patterns in land-use and intensification. A model which only produced output for 2000 and 2050, for example, would miss the tendency shown in FALAFEL for the area of managed land to peak between 2015 and 2035 and then recede, and would thus miss important LUC emissions or ecological effects of habitat loss.

The fluxes of biomass carbon described by FALAFEL can be depicted neatly using ‘Sankey’ diagrams, a type of flow diagram in which the width of the arrows is proportional to the size of the flow (in this case measured in Pg C yr^{-1}). Figure 6.1 shows the fluxes of aboveground biomass in natural and human systems for the year 2000, as mapped by FALAFEL. This format shows very clearly the scale of human appropriation of net primary production (HANPP) relative to natural biological carbon fixation, with the biomass harvested by humans forming a carbon flux larger than the NPP of any individual natural biome.

Figure 6.1 also highlights the inefficiency of food production as a whole, with the very large initial harvest via crops and grazing reduced by more than 85% in the final flux to food products. The inefficiency of food production using livestock is

also well shown here. Livestock feed constitutes almost 50% of total biomass harvest, but the output from livestock to useful products is such a small flux that it has to be inflated to make it visible. This is an excellent representation of the trophic energy loss involved in producing food using livestock, and is the reason that elements of the livestock production system are consistently the most influential drivers of the indicators described in Chapter 5.

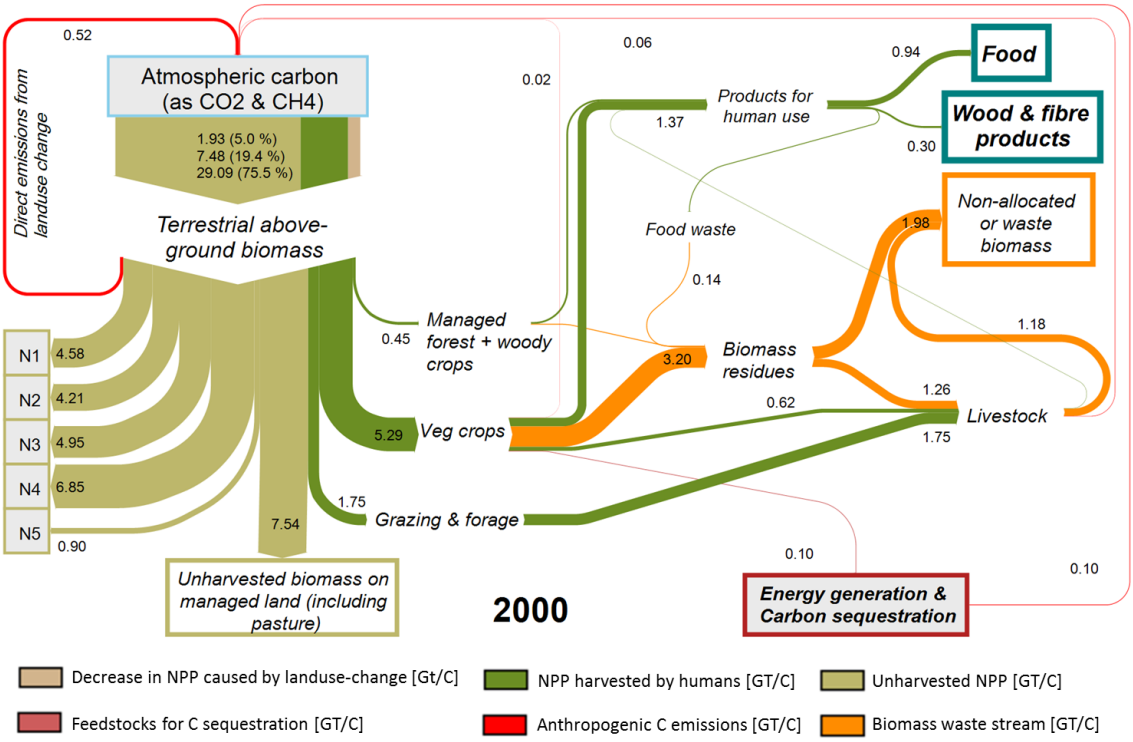

Figure 6.1: Sankey diagram showing global anthropogenic above ground biomass flows in the year 2000. The width of arrows is proportional to the scale of the flux they represent, except for *livestock* to *products for human use*, which is actually far smaller than shown.

The flow of biomass through the livestock component also shows up a flux that is currently missing from FALAFEL, since far more biomass is fed to livestock than the mass that is produced as useful products or 'wastes'. The missing flux is that of CO_2 produced by respiration, which is not strictly necessary to include as it should have no net effect on atmospheric carbon content and does not represent a useable resource. For FALAFEL to become completely internally consistent, however, and generate a fully mass-balanced account of biomass flows, fluxes such as this would have to be included. Methane emissions from livestock are included, depicted by the red flux from livestock to the

atmosphere, since they constitute a net contribution to anthropogenic greenhouse gas (GHG) emissions.

The size of the 'natural' biological carbon fluxes is also revealing. Even in 2000, the largest single contribution to global natural NPP (NPP_n) is the unharvested biomass on land managed by humans (NPP_t). Most of this flux is the unharvested biomass in pastures, which also comprise the largest single biome in terms of area. The increase in NPP_t by 2050 in the *low efficiency, projected meat* (LE PM) scenario (Figure 6.2) reflects the increasing area of pasture at the expense of natural biomes, in a scenario in which the projected global food demand is met with the farming system of the year 2000.

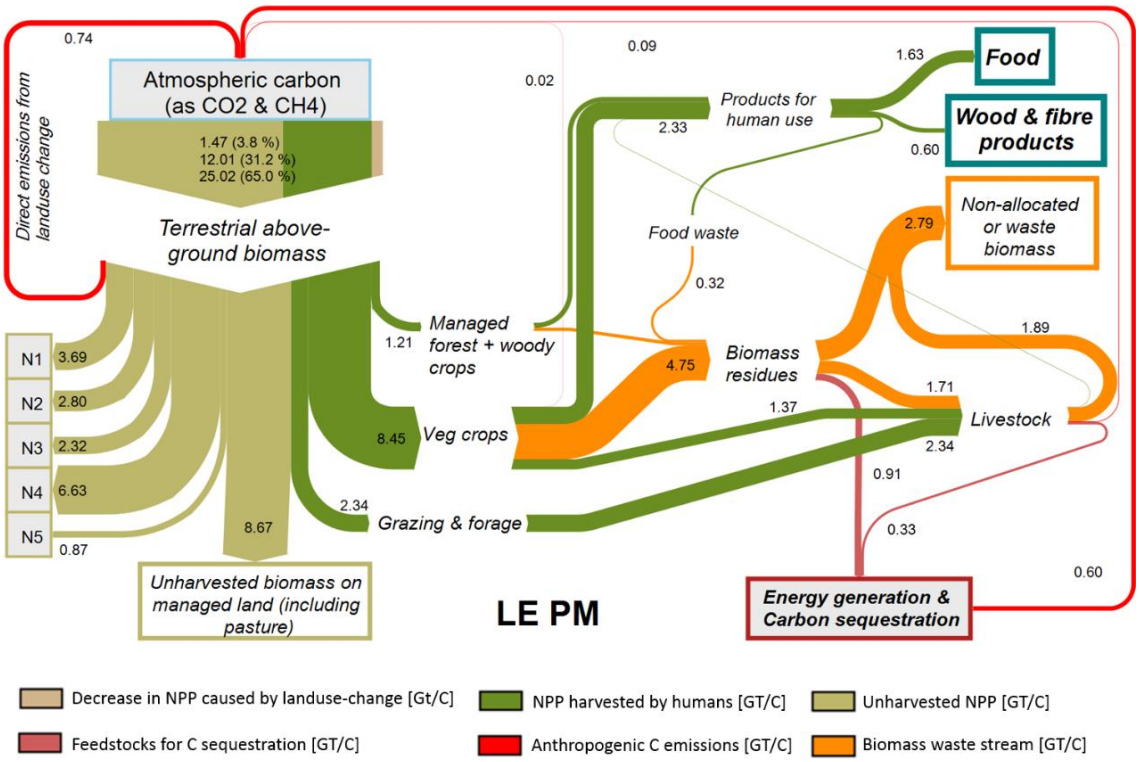

Figure 6.2: Global biomass flows in 2050 for the LE PM scenario.

The sheer scale of the projected increase in food demand between 2000 and 2050 is also apparent here, with the final flux to food growing by 73%. That this growth is met by an increase in in total biomass harvest of 'only' 60% shows that the projected change in diet alone has a non-trivial role to play in increasing the efficiency of global food production, probably largely due to the declining role of ruminants.

In the *business as usual, projected meat* (BAU PM) scenario (Figure 6.3), the intensification of livestock production significantly reduces the biomass harvest of grazing, meaning that the NPP_t flux is correspondingly smaller, while the lower land area required for food production leaves the fluxes to natural biomes less diminished than in LE PM. The harvest of biomass for crops, however, is far larger than in LE PM, in part due to the shift towards fodder crops in livestock production, and in part driven by the intense harvest of bioenergy crops on vacated land.

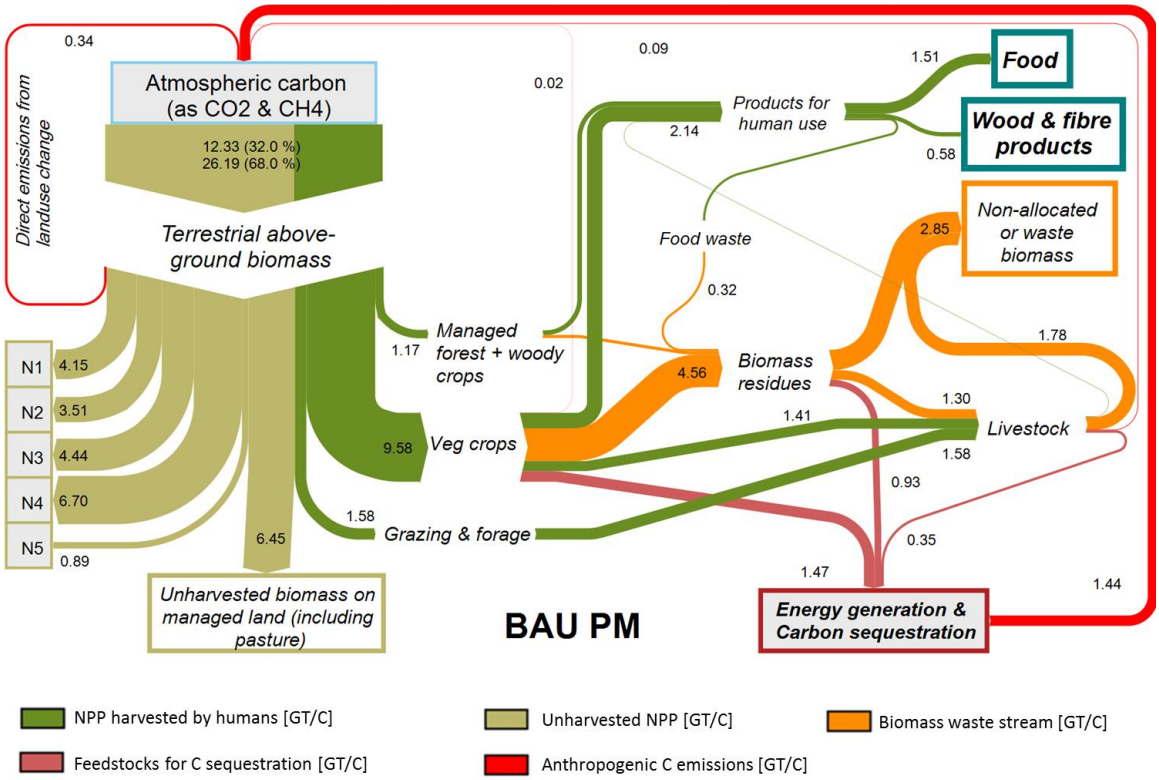

Figure 6.3: Global biomass flows in 2050 for BAU PM.

In BAU PM bioenergy crops, in combination with fluxes of residues and manure constitute a considerable source of feedstocks for energy generation or BECS. Because some bioenergy crops are used in conventional energy generation, and due to the less than 100% carbon capture efficiency of BECS processes, a significant portion of the carbon contained in the original feedstock flux is returned to the atmosphere. As it is generated from carbon fixed by photosynthesis in the same year, in theory (at least in the context of the model) this should not constitute a net emission to the atmosphere. It is useful to show this flux, however, as the difference between the feedstock flux and the emission flux represents the total CDR achieved in the scenario. Particularly

notable in Figure 6.3 is the size of the feedstock flux required to generate the relatively modest CDR flux of just over 1 Pg C yr^{-1} in 2050 for BAU PM; which at 2.75 Pg C yr^{-1} constitutes a larger stream than that providing food from crop products.

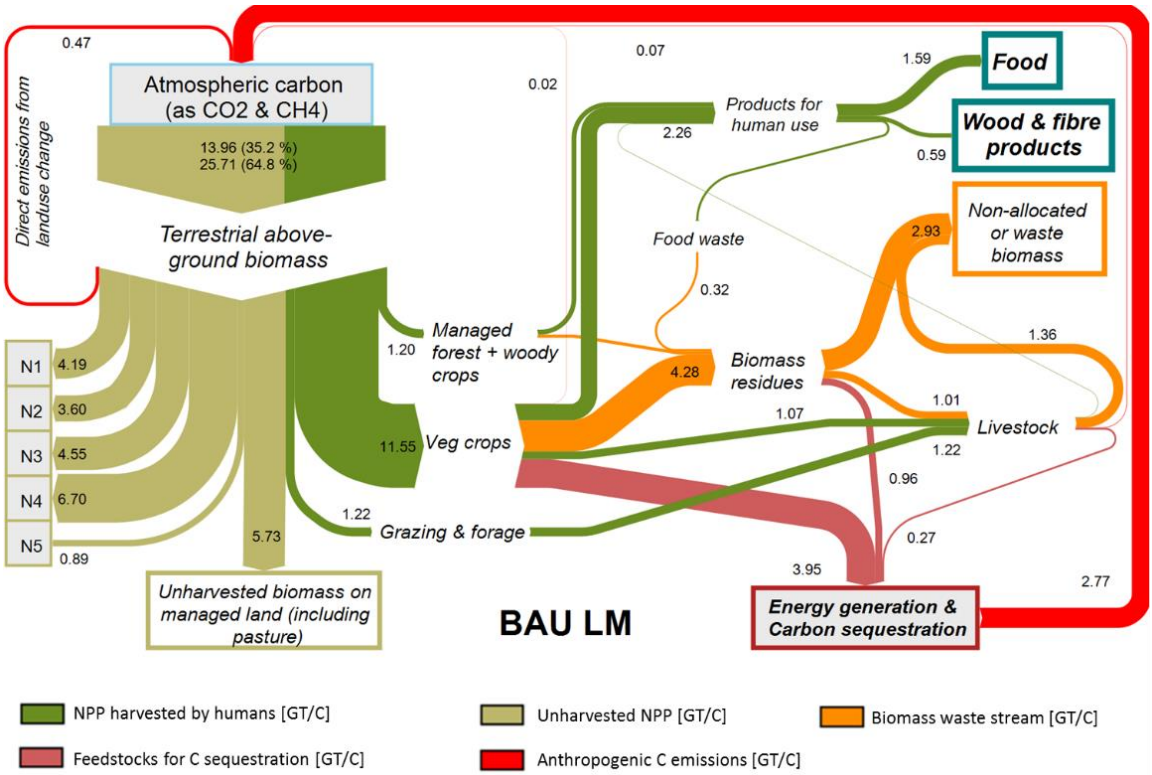

Figure 6.4: Global biomass flows in 2050 for BAU LM.

In the *business as usual, low meat* variant (BAU LM) the mass of available feedstock is hugely increased (Figure 6.4), due to the larger area of bioenergy crops, and in fact constitutes a carbon flux equivalent in scale to the NPP of the individual natural biome types, and close to that of food production.

The *HiBECS* variants of the BAU scenarios (Figure 6.5, Figure A.11) manage to generate much larger CDR fluxes under the same food production scenario, by a combination of redirecting biomass waste streams, and increasing the ratio of BECS to conventional biomass energy generation in the processing of feedstocks. Although residues are still preferentially fed to livestock before being made available as bioenergy or BECS feedstocks in these scenarios, the appropriation of up to 75% of the remaining available residue streams would in practice make biomass unavailable to important uses such as bedding for animals, which are not currently allocated in FALAFEL but constitute significant

fluxes in their own right (Krausmann et al., 2008). This is discussed further in section 6.2.

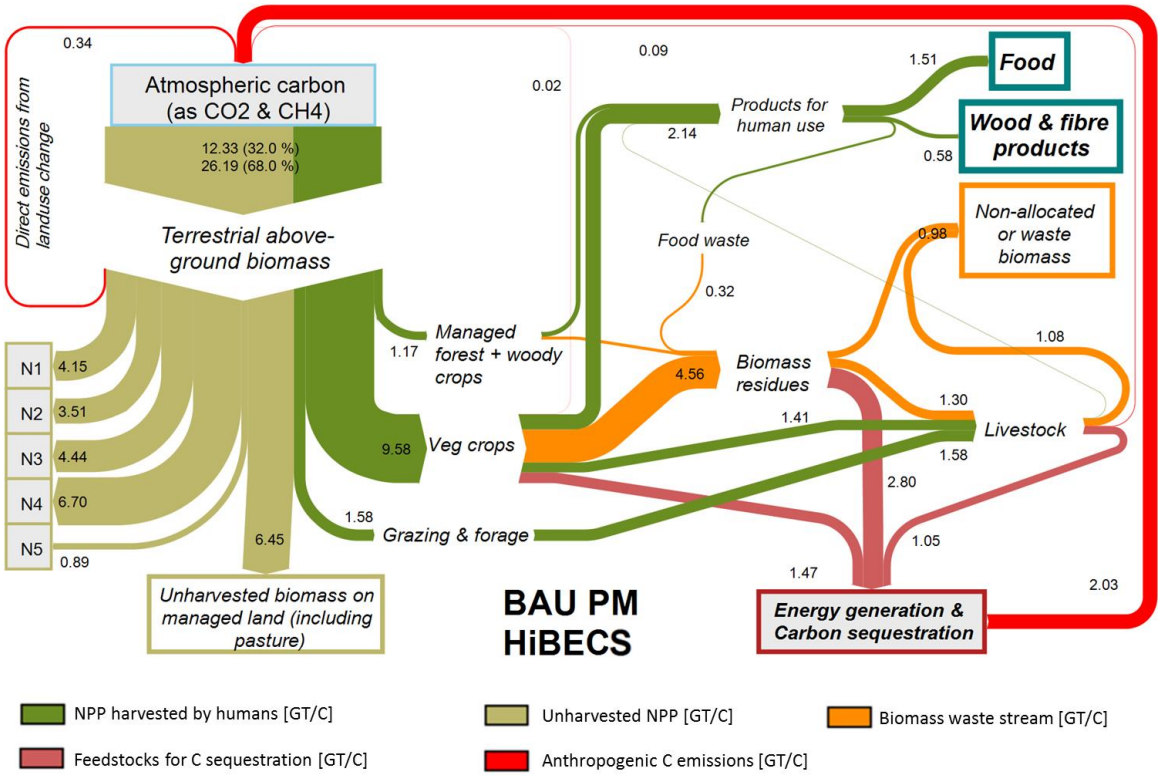

Figure 6.5: Global biomass flows in 2050 for BAU PM HiBECS.

The enormous scale of bioenergy crop production entailed in the *high efficiency* scenarios (Figure 6.6, Figure A.12) is immediately obvious, as it represents almost half of the total anthropogenic biomass harvest, and a carbon flux equal in size to the NPP of the entire N4 biome (approximating boreal forest). This is enabled by the increase in overall efficiency of the food production system; in HE PM if bioenergy crops are discounted the anthropogenic biomass harvest in 2050 is 8.1 Pg C yr^{-1}, only 8% higher than that in the year 2000. This is achieved by two main strategies: First, substantial reductions in the wastes associated with food production, which reduce the primary biomass input per mass of food product produced; and second, considerable alterations to the livestock system which essentially improve the efficiency of all animal production to match the current industrial systems of Western Europe and North America.

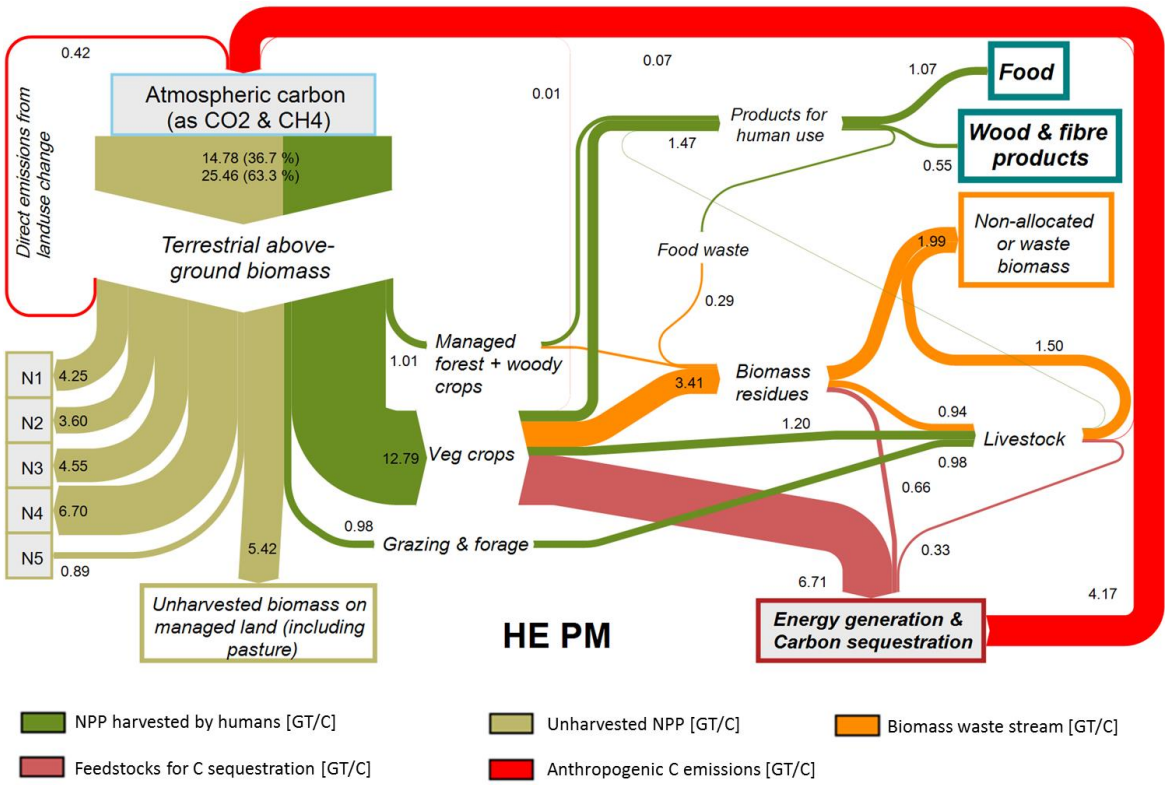

Figure 6.6: Global biomass flows in 2050 for HE PM.

While these 'improvements' to livestock production appear positive from a carbon cycle perspective, and indeed in preventing further destruction of natural biomes, they also imply trade-offs not currently captured by FALAFEL. Intense industrial farming of animals currently requires very large inputs of pharmaceuticals, particularly antibiotics (Landers et al., 2012) and has significant waste management problems (Oenema et al., 2007), not to mention animal welfare issues which should certainly give us cause for concern.

Likewise the very large areas of monoculture bioenergy crops produced in these scenarios generate significant CDR fluxes, but are likely to come at a considerable cost in terms of biodiversity impacts and may increase chemical and water inputs to marginal land. It should be noted that although several papers describe the relatively high biodiversity and low inputs of *Miscanthus* cultivation (Bellamy et al., 2009; Dauber et al., 2015), these usually feature comparisons with other monoculture food crops such as wheat. While in Europe this may be a fair comparison, the scenarios developed here mostly see *Miscanthus* replacing low productivity pasture, which is likely to support considerably higher species diversity. On the other hand, bioenergy crops are quite crudely characterized in FALAFEL, and parameters used could be

adjusted to describe a range of possible options, such as short rotation coppice or mixed native perennial grassland species. These potentially support higher diversity than *Miscanthus*, for similarly low inputs, and may also promote better storage of soil carbon (Dauber et al., 2015; Tilman et al., 2006).

The CDR flux generated from waste streams is fairly constant, at about 0.5 Pg C yr^{-1} by 2050 in the main scenarios, in which 25% of eligible wastes are processed as feedstocks, and around 1.5 Pg C yr^{-1} in the HiBECS scenarios in which 75% are collected. The generation of this flux, although relatively assured in comparison with that from bioenergy crops, assumes the existence of a substantial infrastructure associated with the collection, transport, drying and processing of feedstocks, along with the redistribution and spreading of biochar in the case of pyrolyzation (Shackley et al., 2012). Although these wastes are well suited to a distributed network of small scale processing facilities, the development of such an infrastructure is costly, and the economics have to stack up to make the system work. There is evidence that this is beginning to be the case, with combined heat and power plants running in municipal solid waste in the UK and Europe, and pyrolyzing plants running successfully on human sewage waste in Japan (Koga et al., 2007). The effects of systematically removing crop residues can have negative impacts on soil health (Lal, 2005), although in some systems large amounts of residue can be removed at relatively low cost (Sheehan et al., 2003). It is often argued that the negative effects of residue removal can be mitigated by the return of biochar to the soil, though the effects of biochar addition in fact seem to vary considerably (Clough et al., 2010; Singh et al., 2010; Spokas et al., 2009), and the ultimate capacity of soils to store biochar may not be as high as originally suggested.

Bioenergy/BECS potential in FALAFEL scenarios

Since most studies of bioenergy potential focus on energy generation rather than carbon storage from biomass, comparing FALAFEL outputs directly with the potentials found in other studies is slightly misleading. For that reason I give here a brief comparison using the primary energy potential of biomass streams, which refers to the total energy content of biomass feedstocks before they are used in energy generation or BECS.

Primary energy potential from feedstocks in the *low efficiency* and *business as usual* scenarios (45.9-217.1 EJ yr^{-1} in 2050) fall in the range of other studies which are more constrained by land-use and sustainability criteria (Haberl et al., 2010; Beringer et al., 2011; Chum et al., 2011), suggesting that FALAFEL agrees well with other approaches despite its coarse treatment of spatial issues.

The primary energy potential available in the more extreme scenarios (*HiBECS* and *high efficiency*) is similar to the upper ranges found in other scoping studies (Hoogwijk et al., 2003, 2005; Smeets et al., 2007; Chum et al., 2011; Erb et al., 2012), at 206.8-401.8 EJ yr^{-1} in 2050. The work described here also does an important job in showing the significance of this biomass harvest relative to other global biomass fluxes, and gives an indication of the scale of effort and investment that would be required to attain these high potentials.

FALAFEL appears to stand up reasonably well against other models in forecasting global land-use and bioenergy potentials, but its main strengths are in linking these potentials with the wider system of anthropogenic biomass flows. The results described in this thesis make it clear that although the potential exists for generating bioenergy or biomass based carbon dioxide removal fluxes with significant power to mitigate climate change, the ability to do this without driving extra destruction of natural biomes is highly dependent on other drivers of human biomass harvest.

Principle among these are the base drivers of total food demand of the human population; population itself, and per capita *food supply*. Since population growth is locked in, at least for the near future (United Nations, Department of Economic and Social Affairs, Population Division, 2013), and access to sufficient nutrition is a basic human right and the first Millennium Development Goal, the provision of an adequate food supply must be prioritized above all else.

How this food demand is met, however, is only marginally less significant in its effect on the availability of biomass for CDR or energy generation. In particular, the contribution of livestock to the total food supply is an enormously powerful determinant of the required biomass harvest for food production. A small reduction in consumption of livestock products globally (in reality a reduction in consumption in western diets, but a small increase for many others), could very

significantly increase the potential to use biomass for climate change mitigation. Unfortunately, this is likely to be a difficult cultural shift to achieve; although the almost entirely vegetarian diet of over one billion Indian people shows that cultural attitudes to what is acceptable to eat can certainly vary.

Aside from the sheer quantity of consumption of livestock products, the efficiency with which they are produced also has a powerful effect, especially in the production of food from ruminant species. As already mentioned, the intensification of livestock production carries significant problems of its own, but considerable opportunities may exist in some systems to improve production efficiency with relatively little cost. Of course the level of disaggregation of the sector achieved in FALAFEL, though improved, cannot pretend to capture anything like the complexity and variety of real livestock production systems.

Efficiency in other parts of the food production system is also important, in particular reducing production losses and post-production food waste could have a significant impact. In the developing world this would require considerable improvement in infrastructure, with improved distribution and storage of perishable goods, and protection from disease and water shortages. In the developed world, much could be achieved with cultural and legislative changes around the acceptability of wasting food, and indeed these kind of shifts do currently seem to be gathering momentum.

Finally, a key assumption underlying most scenarios in which considerable CDR fluxes are achieved is that of sustained yield increases. In fact, the increases afforded by industrial agriculture appear to be stagnating in many areas (Ray et al., 2012; Grassini et al., 2013). In addition, intensive farming practices can deplete soil fertility (Sanchez, 2002), and disrupt soil micro-organisms (Weese et al., 2015), and global crop production seems to be becoming less nutrient efficient (Tilman et al., 2011). Because of this it is difficult to reconcile the assumption that yields will maintain an upward trajectory with the reality of observations.

There is a danger, when undertaking a necessarily simplistic global level study such as this, to be similarly simplistic in proposing possible ‘solutions’ such as improving efficiency, or relying less on livestock. The reality, of course, is that ‘the food production system’, although globalized to a certain degree, is an

extremely complex network of interactions at all scales. This system is influenced by the decisions and behaviours of individual actors as well as national and international markets and policies, and is constrained by local environmental variables or cultures as much as it is by the global climate. Correspondingly, there is no suite of global solutions which will address the issues discussed here. Global level studies do, however, help to point out the larger scale implications of the behaviour of complex systems such as this, and can highlight issues that are relevant at all levels.

One of the key motivations for this work was to look for trade-offs implicit in strategies for meeting biomass demand while reducing environmental impacts or mitigating climate change. The preceding chapters have highlighted some of the important issues in pointing out connections between diet and climate change mitigation potential, biodiversity preservation and bioenergy production, and intensive versus extensive agriculture, albeit at a coarse scale.

6.2 Issues in FALAFEL, and opportunities for further work

Although capable of producing useful output, FALAFEL currently lacks some elements necessary to dig deeper into the questions raised by the work I have presented. Here I briefly describe a few of the current inadequacies of the model, and ways in which it could be improved to allow a more nuanced understanding of the drivers and impacts of human biomass harvest.

Lack of regional differentiation

By dealing in global averages, FALAFEL simplifies its calculations and eliminates certain complexities involved in a system that is in fact extremely spatially heterogenous. In doing so, however it also misses the potential impacts of differing demands and pressures in different regions of the world. Different stages of development are likely to drive differing demands for biomass in different regions of the world, while the spatial disconnect between production and consumption means that the impacts of biomass harvest can be exported to other places (Erb et al., 2009b; Haberl et al., 2009; Nonhebel and Kastner, 2011). These spatial ‘teleconnections’ between elements of the system of biomass harvest and consumption are also connected to issues of the inequality of food distribution, with prices for global commodities such as wheat

and maize driven by the economics of wealthier nations. The prices paid by European farmers for wheat to feed to cattle may exclude the citizens of poorer countries from being able to buy enough to feed themselves (Rosegrant et al., 1999)

Breaking FALAFEL into multiple regional models would allow the separation of demand and production in different regions, as well as dealing in more detail with the different land-use constraints and yield trajectories of different geographical areas. On the face of it, this kind of regionalization would be relatively easy, albeit bulky in the current spreadsheet format. In practice the complexity would lie in establishing approximations of the key economic drivers of biomass trade between regions, although this is not insurmountable.

Nutritional aspects of diet

Using only the energy content of food products to compare their contributions to the total food supply misses important differences between their content of other nutritionally important variables, such as protein. Although less important when using dietary scenarios based on past trends, if FALAFEL is used to examine the impacts of deliberately adjusting diets, by reducing consumption of livestock products for example, it may be important to ensure that adjusted diets are nutritionally consistent, rather than simply energetically consistent.

Underestimation of LUC emissions

As discussed in Chapter 5 (section 5.1.3, p140), FALAFEL is certainly missing a significant portion of CO_2 emissions from land-use change, as the model currently only deals with the carbon content of above ground vegetation. This was based originally on a decision to simplify the model by not dealing with the complex interactions between plant and soil carbon pools, and crude assumptions about the differences in biomass allocation to roots in different biomes. Now that the rest of the model structure is completed, it may be time to return to this issue and account for the remaining portion of biomass carbon, especially as differences in root and soil carbon storage between forests and grasslands can significantly affect the way the carbon storage of the two biomes is viewed.

Another likely source of the underestimation of land-use emissions is the rather over-efficient method of calculating land-use allocation used by FALAFEL. In the model, land is only vacated if demand declines for a particular category of land-use, and when this happens it is immediately occupied by another use, with bioenergy crop or afforestation as a final option when it is no longer required by any other category. In reality, shifting cultivation, slash and burn practices in tropical regions and the degradation of land by soil erosion mean that a significant area of cultivated land is abandoned each year, and even in the EU changing politics and economics can cause land to be abandoned. For this reason FALAFEL significantly underestimates deforestation rates for 2000-2010, with the model putting the loss of natural biome classes N1, N2 and N4 (roughly equivalent to tropical and temperate forests (N1 & N2) and boreal forests (N4)) at an average rate of 7.2 Mha per year in this period, while observational studies put the loss at 13-23 Mha yr^{-1} (FAO, 2010; Hansen et al., 2013).

Interaction with other global biogeochemical cycles

FALAFEL's biomass flows approach presents a very good mechanism with which to investigate the stoichiometry of the global agricultural system. By using the ratios of C:N:P contained in different biomass products, one could model fluxes of key nutrients and interactions with their respective biogeochemical cycles alongside the current overview of carbon cycle interactions. This could provide important insight into the nitrogen and phosphorous cycle implications of different food production or bioenergy scenarios, and may highlight trade-offs in terms of N_2O emissions and other forms of nitrogen pollution, or depletion of phosphate rock reserves. Conversely opportunities for using waste streams to recycle nutrients and thus reduce the impacts associated with waste disposal and resource extraction could be explored.

Inadequate tracing of waste fluxes

Although waste streams are relatively well disaggregated by FALAFEL in terms of their origins and suitability as livestock feed or bioenergy feedstocks, their default treatment is somewhat crude. Unharvested residues and manure are not deemed available as BECS or energy feedstocks, or any other use, and their carbon content is assumed to be a 'backflow to nature' (Krausmann et al.,

2008). This is an adequate assumption in the current structure of FALAFEL, but in reality it constitutes an important part of human impacts on carbon and nutrient processes in the soils of managed lands (Lal, 2005).

The remaining wastes end up in a pool of biomass with no particular allocation, of which a certain portion is diverted for use as fuel for energy generation or BECS. This includes what might be considered genuine wastes, such as food waste and some crop residues which are likely to be sent to land-fill or incinerated, but also includes biomass fluxes which cannot be considered wastes, such as manure from housed animals which is in fact usually returned to the land, or straw used as bedding for animals.

Unpacking the likely destinations of the carbon or other elements contained in these fluxes would be a valuable exercise, as many of them are substantial sources of greenhouse gas emissions or other pollutants in their own right. When food waste or other biomass is buried in landfill sites, for example, it becomes a source of methane, while storage of manure is an important source of nitrogen pollution (Oenema et al., 2007). Diverting some of these waste streams for use as energy feedstocks could offset these emissions, as well as offsetting fossil fuels emissions and potentially sequestering carbon. In cases where the fluxes described as waste streams in fact already have a use, for example as bedding for animals, or as a source of nutrients, it is also important to consider the counterfactuals associated with diverting them for use as bioenergy/BECS feedstocks. If manure is pyrolysed instead of being return to the land, for example, the cost in terms of fossil fuel use and N_2O emissions should ideally be accounted for in order to work out whether there is a net benefit.

In addition to missing greenhouse gas emissions from the storage and processing of wastes, some entire waste streams are also missing from FALAFEL. Most notably among these is human sewage, the nutrient content of which especially should be considered a potential resource, which could help to close loops and reduce the need for synthetic fertilizer application.

Methane

Methane is produced from various sources associated with biomass production, harvest and processes. The effect of CH_4 emissions to the atmosphere is not currently quantified in FALAFEL, since because methane has a higher radiative forcing potential and shorter residence time than CO_2, carbon emitted to the atmosphere as CH_4 cannot be treated as analogous to carbon emitted as CO_2. In order to reconcile this difference, a similar decay function to that used to describe the behaviour of CO_2 emissions needs to be derived for CH_4. The effects of CO_2 and CH_4, and indeed any other greenhouse gas fluxes, such as N_2O, could be combined as a total effect on radiative forcing potential.

Climate change feedbacks in long term scenarios

The scenarios described in chapters 2-5 set up worlds in which the climate by 2100, if not by 2050 could be quite different from one another. One element that is currently missing from FALAFEL in terms of comparing these worlds is the effect of the climate itself on crop production. In practical terms, this type of work is the job of spatially explicit, process-based models, which can assess the effects of interacting feedbacks like temperature, precipitation and CO_2 fertilization on crop yields and distributions, or of crop distributions on the albedo of the Earth's surface (Singarayer and Davies-Barnard, 2012).

That FALAFEL cannot address this issues in a meaningful way is the price of taking a non-spatial approach, and is hopefully offset by some of the advantages of taking a global perspective.

6.4 Conclusions

FALAFEL has been shown to make forecasts of land-use change and bioenergy production that fall within the ranges predicted by other studies, which is a good indication that its output is relatively sensible. The biomass flows modelling approach highlights the importance of the connection between the efficiency of food production and the availability of feedstocks for bioenergy and BECS, and makes it clear that meeting growing food demand and attempting to reduce anthropogenic perturbation to Earth's life support systems will necessarily involve some compromises and trade-offs. That said, it is also clear that under most circumstances there is significant potential for the generation of carbon dioxide removal fluxes from waste streams if an infrastructure could be developed with which to collect and process them. The FALAFEL model is well set up to be developed into a tool for further investigation of regional differences in biomass harvest systems, and for investigating implications for CDR strategies in terms of nutrient cycling and other impacts.

Discussing global scale scenarios can seem a little like the orchestration of a Soviet style planned economy, and it is important to remember that the global biomass harvest system is in fact constructed of the interactions of many individuals, environments, corporations and governments. The drive for sustainable food production and other biomass harvest must also include strategies to address the sustainability and resilience of the social structures in the food system; for example issues around the accessibility of food, or the resilience of farming communities to climate change and fluctuating markets (McIntyre, 2009; Loos et al., 2014; Hodbod and Eakin, 2015).

I

Appendices

Appendix I: Make up of crop and livestock groups

The following tables list the FAOSTAT production categories included in each major food type group.

Table A.1: Production categories forming cereals, oil seeds and luxuries. Categories in bold constitute a group of their own in the disaggregated diet.

Cereals	Oil seeds	Luxuries
Barley	*Castor oil seed*	*Anise, badian, fennel, coriander.*
Buckwheat	*Coconuts*	*Cloves*
Canary seed	*Groundnuts, with shell*	*Cocoa beans*
Cereals, nes	*Hempseed*	*Coffee, green*
Fonio	*Jojoba Seeds*	*Ginger*
Maize	*Kapok Fruit*	*Natural rubber*
Millet	*Karite Nuts (Sheanuts)*	*Pepper (Piper spp.)*
Mixed grain	*Linseed*	*Peppermint*
Oats	*Melonseed*	*Spices, nes*
Quinoa	*Mustard seed*	*Sugar beet*
Rice, paddy	***Oil palm fruit***	*Sugar cane*
Rye	*Oilseeds, Nes*	*Sugar crops, nes*
Sorghum	*Olives*	*Tea*
Triticale	*Poppy seed*	*Tobacco, unmanufactured*
Wheat	***Rapeseed***	*Vanilla*
	Safflower seed	
	Sesame seed	
	Soybeans	
	Sunflower seed	
	Tallowtree Seeds	
	Tung Nuts	

Table A.2: Production categories forming *other food*, part 1.

Fruit		Vegetables	
Apples	*Lemons and limes*	*Artichokes*	*Lettuce and chicory*
Apricots	*Mangoes, mangosteens, guavas*	*Asparagus*	*Maize, green*
Avocados	*Oranges*	*Beans, green*	*Mushrooms and truffles*
Bananas	*Other melons (inc.cantaloupes)*	*Cabbages and other brassicas*	*Okra*
Berries Nes	*Papayas*	*Carrots and turnips*	*Onions (inc. shallots), green*
Blueberries	*Peaches and nectarines*	*Cassava leaves*	*Onions, dry*
Carobs	*Pears*	*Cauliflowers and broccoli*	*Peas, green*
Cashewapple	*Persimmons*	*Chillies and peppers, green*	*Pumpkins, squash and gourds*
Cherries	*Pineapples*	*Cucumbers and gherkins*	*Spinach*
Citrus fruit, nes	*Plantains*	*Eggplants (aubergines)*	*String beans*
Cranberries	*Plums and sloes*	*Garlic*	*Tomatoes*
Currants	*Pome fruit, nes*	*Leeks, other alliaceous veg*	*Vegetables fresh nes*
Dates	*Quinces*	*Leguminous vegetables, nes*	
Figs	*Raspberries*		
Fruit Fresh Nes	*Sour cherries*		
Fruit, tropical fresh nes	*Stone fruit, nes*		
Gooseberries	*Strawberries*		
Grapefruit (inc. pomelos)	*Tangerines, mandarins, clem.*		
Grapes	*Watermelons*		
Kiwi fruit			

Table A.3: Production categories forming *other food*, part 2.

Pulses	Treenuts	Roots and tubers
Bambara beans	*Almonds, with shell*	*Carrots and Turnips*
Beans, dry	*Brazil nuts, with shell*	*Cassava*
Broad beans, horse beans, dry	*Cashew nuts, with shell*	*Potatoes*
Chick peas	*Chestnuts*	*Roots and Tubers, nes*
Cow peas, dry	*Hazelnuts, with shell*	*Sweet potatoes*
Lentils	*Nuts, nes*	*Taro (cocoyam)*
Lupins	*Pistachios*	*Yams*
Peas, dry	*Walnuts, with shell*	*Yautia (cocoyam)*
Pigeon peas		
Pulses, nes		
Vetches		

Animal products

Table A.4: Production categories for meat products.

Bovine	Other meat	Poultry meat	Pig meat
Buffalo meat	*Camel meat*	*Bird meat, nes*	*Pig meat*
Cattle meat	*Game meat*	*Chicken meat*	
	Goat meat	*Duck meat*	
	Horse meat	*Goose and guinea fowl meat*	
	Meat nes	*Turkey meat*	
	Meat of Asses		
	Meat of Mules		
	Meat of Other Rod		
	Meat Other Camelids		
	Rabbit meat		
	Sheep meat		
	Snails, Not Sea		

Table A.5: Production categories of non-meat livestock products.

Milk	Eggs
Buffalo milk, whole, fresh	*Hen eggs, in shell*
Camel milk, whole, fresh	*Other bird eggs,in shell*
Cow milk, whole, fresh	
Goat milk, whole, fresh	
Sheep milk, whole, fresh	

Fibre crops

Table A.6: Production categories of fibre crops

Fibre crops
Agave Fibres Nes
Fibre Crops Nes
Flax fibre and tow
Hemp Tow Waste
Jute
Kapok Fruit
Manila Fibre (Abaca)
Other Bastfibres
Ramie
Seed cotton
Sisal
Agave Fibres Nes

Appendix II: contributions of crop and livestock to global average diet in 2000 and 2050

Table A.7: Contributions of food groups to the average diet in 2000 and 2050 (baseline scenario).

	% of total diet		KCal cap^{-1} day^{-1}	
	2000	2050	2000	2050
Cattle Meat	1.9	1.2	52.7	38.9
Pig Meat	2.9	4.3	79.1	143.6
Chicken Meat	2.2	5.2	61.3	175.6
Other Meat	0.6	0.7	17.8	24.3
Milk	5.7	4.3	156.4	143.9
Eggs (with Shell)	1.3	2.2	34.5	74.3
Animal fats	1.0	0.4	27.1	14.9
Fish	1.0	1.2	28.2	40.4
Wheat	15.5	11.4	422.6	382.2
Maize	6.0	9.3	163.1	311.0
Rice	19.2	15.7	525.9	527.4
Other Cereals	7.0	1.8	191.4	61.7
Soybeans	2.4	4.0	66.8	133.5
Rapeseed	1.0	2.2	27.4	75.4
Palm Fruit	4.8	11.1	130.0	372.6
Sunflower Seeds	0.9	1.0	23.7	34.9
Other Oilseeds	1.4	1.3	39.3	43.3
Sugar Cane	5.5	7.0	149.1	235.6
Other Food	19.7	15.5	538.9	519.6

Appendix III: Livestock yields

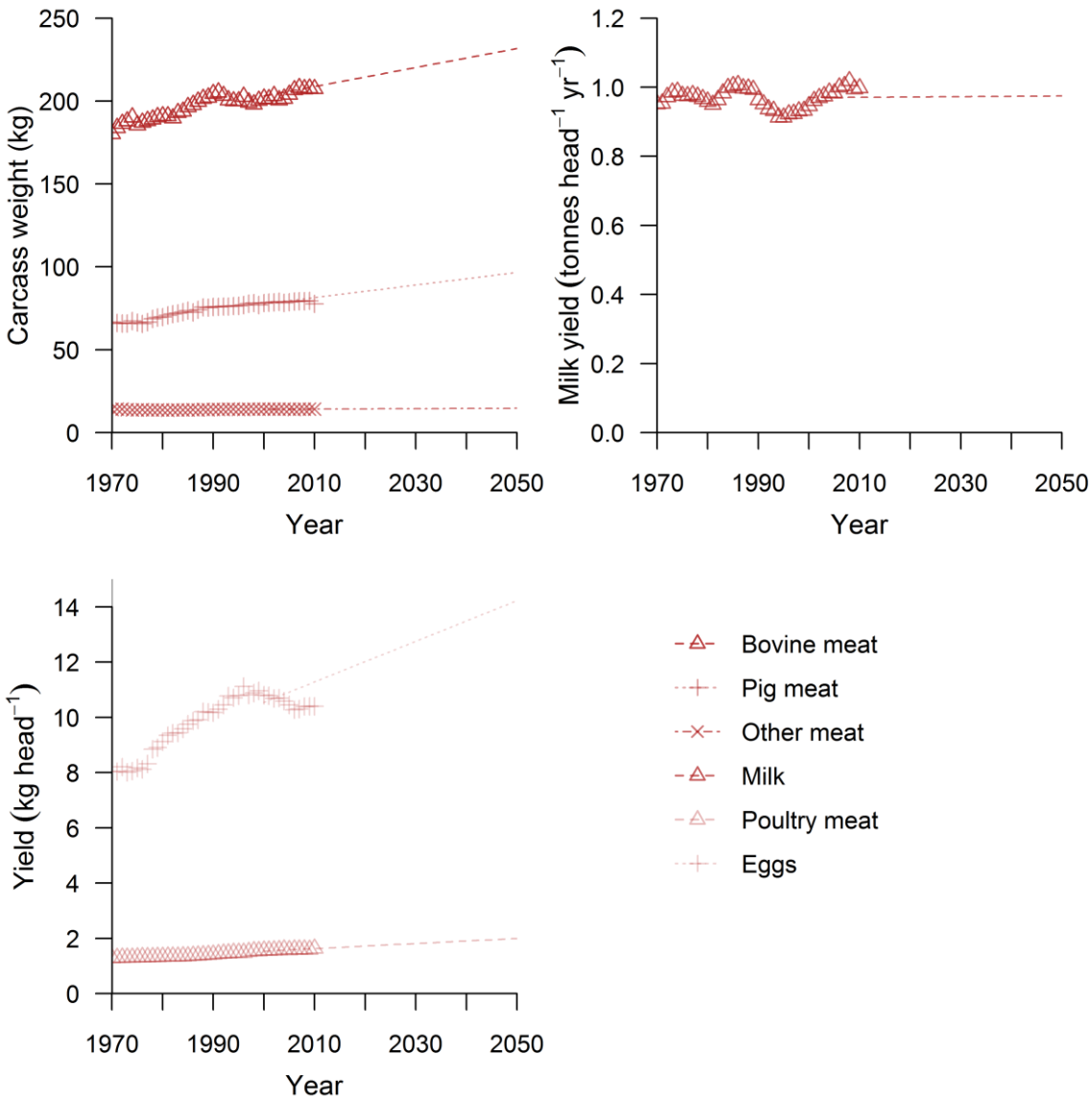

Figure A.1: Yields of livestock product groups with historical data from FAOSTAT, and projected trends.

Appendix IV: BECS feedstocks

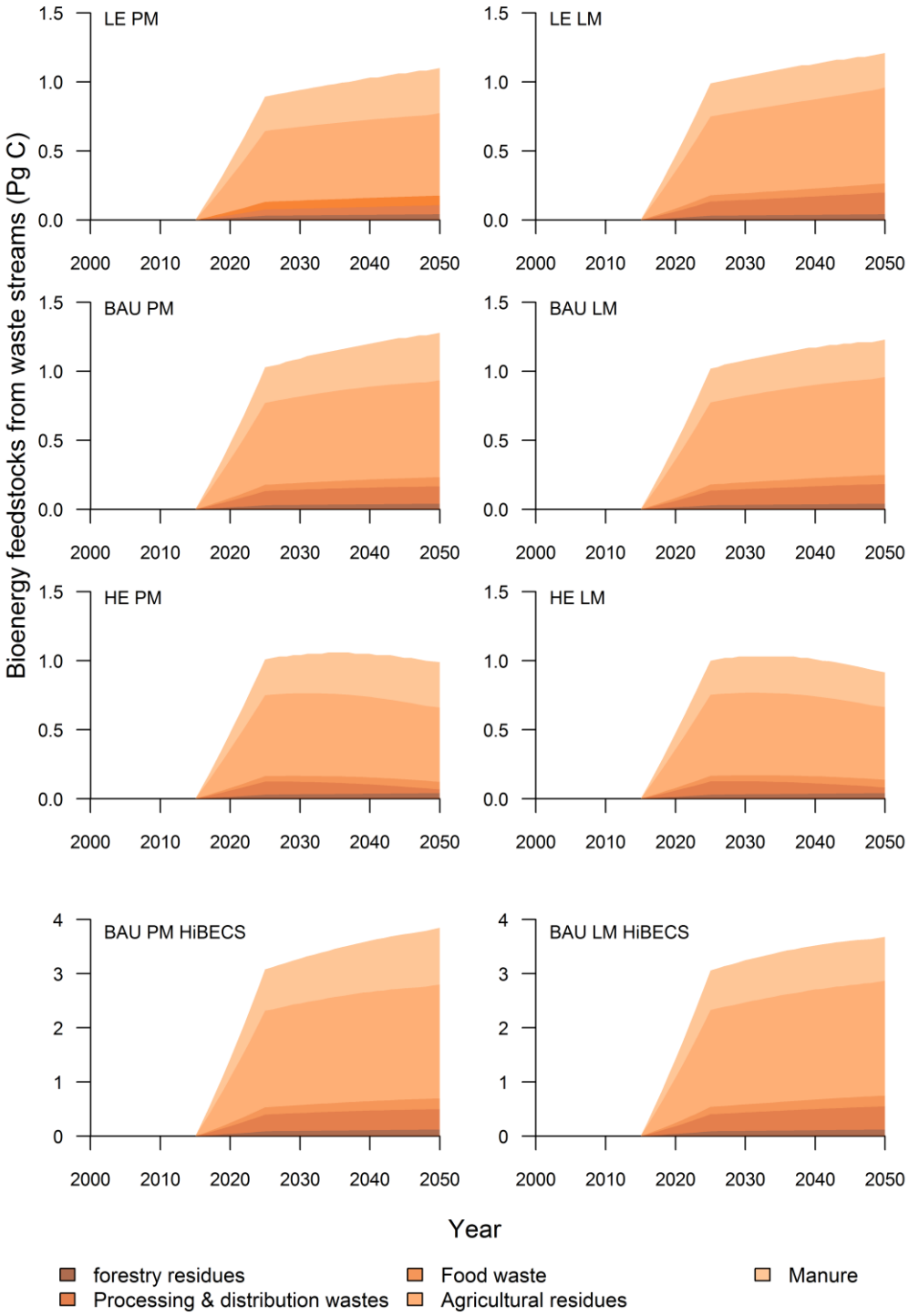

Figure A.2: BECS feedstocks derived from waste and residue streams in each scenario. Note the different Y axis scale for HiBECS scenarios, which produce more than double the feedstock resource.

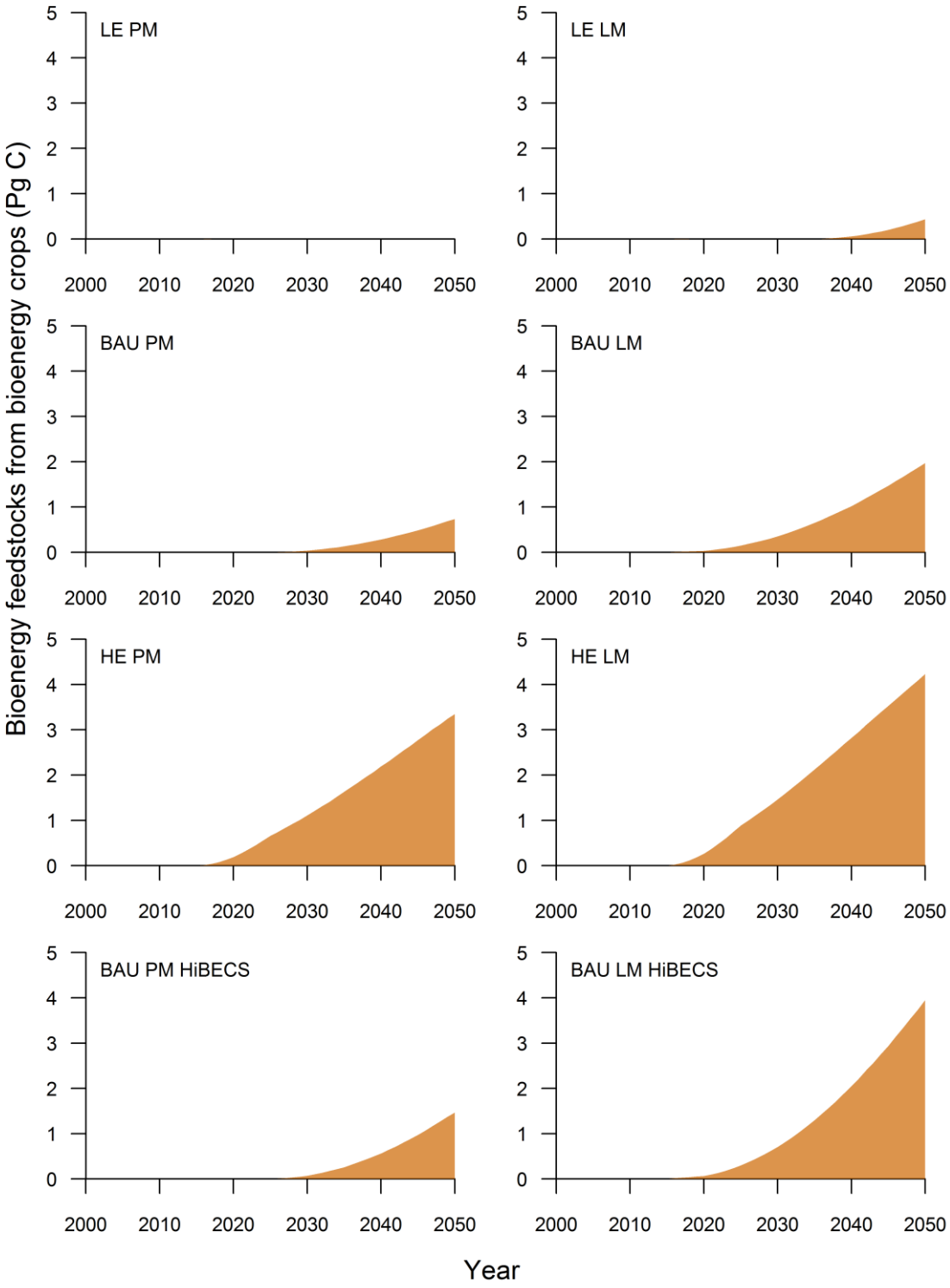

Figure A.3: BECCS feedstocks from dedicated bioenergy crops in each scenario.

Appendix V: Minimum adequate regression model tables

Crop area

2030

	t value	Pr(>|t|)
Calories	147.688	< 0.001
Yield	-113.649	< 0.001
Population	83.939	< 0.001
Dairy - agricultural residues	-54.083	< 0.001
Dairy intensification	38.213	< 0.001
% meat	35.120	< 0.001
Ruminants - agricultural residues	-34.940	< 0.001
Ruminant intensification	29.714	< 0.001
Distribution losses	27.965	< 0.001
Ruminant conversion efficiency	-25.377	< 0.001

2050

	t value	Pr(>|t|)
Calories	61.953	< 0.001
Yield	-46.914	< 0.001
Population	42.695	< 0.001
% meat	23.327	< 0.001
Dairy - agricultural residues	-18.851	< 0.001
Food waste	15.545	< 0.001
Dairy intensification	14.398	< 0.001
Ruminant conversion efficiency	-13.827	< 0.001
Ruminants - agricultural residues	-13.057	< 0.001
Ruminant intensification	12.460	< 0.001

Pasture area

2030

	t value	Pr(>|t|)
% meat	113.442	< 0.001
Calories	109.724	< 0.001
Dairy conversion efficiency	-83.288	< 0.001
Ruminant conversion efficiency	-79.956	< 0.001
Population	62.799	< 0.001
Grazing intensity	-34.883	< 0.001
Dairy intensification	-33.433	< 0.001
Ruminant intensification	-22.875	< 0.001
Food waste	10.977	< 0.001
Pig intensification	-8.980	< 0.001

2050

	t value	Pr(>|t|)
% meat	49.865	< 0.001
Calories	38.542	< 0.001
Ruminant conversion efficiency	-34.934	< 0.001
Dairy conversion efficiency	-31.036	< 0.001
Population	27.728	< 0.001
Grazing intensity	-13.656	< 0.001
Dairy intensification	-13.463	< 0.001
Food waste	9.534	< 0.001
Ruminant intensification	-8.284	< 0.001
Pig intensification	-5.646	< 0.001

Land-use for food

2030			2050		
	t value	Pr(>|t|)		t value	Pr(>|t|)
Calories	154.506	< 0.001	Calories	54.025	< 0.001
% meat	114.116	< 0.001	% meat	50.068	< 0.001
Population	87.842	< 0.001	Population	37.919	< 0.001
Ruminant conversion efficiency	-81.131	< 0.001	Ruminant conversion efficiency	-33.767	< 0.001
Dairy conversion efficiency	-79.479	< 0.001	Dairy conversion efficiency	-28.041	< 0.001
Yield	-41.822	< 0.001	Yield	-15.333	< 0.001
Grazing intensity	-31.686	< 0.001	Food waste	13.499	< 0.001
Dairy - agricultural residues	-20.339	< 0.001	Grazing intensity	-11.951	< 0.001
Dairy intensification	-15.644	< 0.001	Dairy intensification	-6.576	< 0.001
Food waste	15.510	< 0.001	Dairy - agricultural residues	-6.545	< 0.001

Bioenergy area

2030			2050		
	t value	Pr(>|t|)		t value	Pr(>|t|)
Calories	-21.728	< 0.001	Calories	-30.493	< 0.001
% meat	-14.492	< 0.001	% meat	-29.341	< 0.001
Ruminant conversion efficiency	11.408	< 0.001	Population	-21.412	< 0.001
Dairy conversion efficiency	9.534	< 0.001	Ruminant conversion efficiency	20.280	< 0.001
Population	-9.146	< 0.001	Dairy conversion efficiency	13.705	< 0.001
Yield	5.193	< 0.001	Food waste	-9.660	< 0.001
Food waste	-3.343	< 0.001	Yield	7.946	< 0.001
Distribution losses	-2.997	< 0.01	Dairy intensification	4.977	< 0.001
Urbanization	2.575	<0.05	Distribution losses	-4.225	< 0.001
Dairy intensification	2.513	<0.05	Grazing intensity	4.210	< 0.001

Cumulative LUC emissions

2030

	t value	Pr(>|t|)
Calories	50.904	< 0.001
Population	38.494	< 0.001
% meat	37.608	< 0.001
Bioenergy area	33.196	< 0.001
Yield	-31.251	< 0.001
Dairy conversion efficiency	-30.118	< 0.001
Ruminant conversion efficiency	-29.060	< 0.001
Dairy - agricultural residues	-12.501	< 0.001
Grazing intensity	-9.572	< 0.001
Food waste	8.843	< 0.001

2050

	t value	Pr(>|t|)
Calories	25.265	< 0.001
Population	22.379	< 0.001
% meat	20.590	< 0.001
Bioenergy area	18.582	< 0.001
Yield	-18.557	< 0.001
Ruminant conversion efficiency	-18.277	< 0.001
Dairy conversion efficiency	-17.615	< 0.001
Food waste	11.008	< 0.001
Grazing intensity	-6.869	< 0.001
Dairy - agricultural residues	-6.766	< 0.001

Cumulative CDR flux

2030

	t value	Pr(>|t|)
Bioenergy area	-16.228	< 0.001
Residue harvest for BECS	-15.301	< 0.001
Implementation date	10.061	< 0.001
Implementation time	5.684	< 0.001
Residue removal	-2.926	< 0.01
Poultry - processing residues	2.876	< 0.01

2050

	t value	Pr(>|t|)
Bioenergy area	-27.216	< 0.001
Residue harvest for BECS	-25.575	< 0.001
Implementation date	15.365	< 0.001
Bioenergy crop use	-12.531	< 0.001
Implementation time	8.440	< 0.001
Population	-3.760	< 0.001
Dairy - agricultural residues	3.518	< 0.001
Residue removal	-3.327	< 0.001
Food waste	-3.086	< 0.01
% meat	-2.945	< 0.01

Net cumulative C flux

2030	t value	Pr(>|t|)	**2050**	t value	Pr(>|t|)
Calories	20.370	< 0.001	Residue harvest for BECS	-24.897	< 0.001
Population	15.577	< 0.001	Implementation date	14.915	< 0.001
Yield	-15.229	< 0.001	Bioenergy crop use	-12.835	< 0.001
% meat	13.247	< 0.001	Calories	9.542	< 0.001
Residue harvest for BECS	-12.227	< 0.001	Implementation time	8.508	< 0.001
Dairy conversion efficiency	-10.728	< 0.001	Yield	-7.367	< 0.001
Implementation date	9.390	< 0.001	Bioenergy area	-7.061	< 0.001
Ruminant conversion efficiency	-9.281	< 0.001	Population	5.724	< 0.001
Grazing intensity	-6.771	< 0.001	Dairy conversion efficiency	-5.520	< 0.001
Bioenergy area	6.158	< 0.001	% meat	5.394	< 0.001

Effect on atmospheric CO_2

2030	t value	Pr(>|t|)	**2050**	t value	Pr(>|t|)
Calories	19.661	< 0.001	Residue harvest for BECS	-25.169	< 0.001
Population	14.786	< 0.001	Bioenergy crop use	-14.226	< 0.001
Yield	-14.713	< 0.001	Implementation date	12.241	< 0.001
Residue harvest for BECS	-14.147	< 0.001	Bioenergy area	-8.969	< 0.001
% meat	12.707	< 0.001	Calories	7.623	< 0.001
Dairy conversion efficiency	-10.447	< 0.001	Implementation time	7.585	< 0.001
Implementation date	9.506	< 0.001	Yield	-5.588	< 0.001
Ruminant conversion efficiency	-8.934	< 0.001	Dairy conversion efficiency	-4.433	< 0.001
Grazing intensity	-6.640	< 0.001	Population	4.206	< 0.001
Bioenergy area	5.378	< 0.001	% meat	4.186	< 0.001

Annual CH4 emission

2030			**2050**		
	t value	Pr(>|t|)		t value	Pr(>|t|)
Calories	263.385	< 0.001	%meat	101.314	< 0.001
% meat	222.534	< 0.001	Calories	95.524	< 0.001
Population	148.020	< 0.001	Population	65.884	< 0.001
Yield	-55.388	< 0.001	Food waste	26.711	< 0.001
Food waste	27.958	< 0.001	Yield	-18.591	< 0.001
Distribution losses	11.768	< 0.001	Dairy - agricultural residues	-4.712	< 0.001
Dairy - agricultural residues	-11.059	< 0.001	Processing losses	4.652	< 0.001
Dairy intensification	7.442	< 0.001	Ruminant intensification	4.013	< 0.001
Ruminant intensification	7.279	< 0.001	Dairy intensification	3.284	< 0.01
Ruminants - agricultural residues	-6.644	< 0.001	Distribution losses	3.198	< 0.01

Cumulative methane

2030			**2050**		
	t value	Pr(>|t|)		t value	Pr(>|t|)
Calories	331.163	< 0.001	Calories	137.749	< 0.001
% meat	272.883	< 0.001	% meat	126.607	< 0.001
Population	206.538	< 0.001	Population	84.798	< 0.001
Yield	-72.340	< 0.001	Yield	-28.591	< 0.001
Food waste	17.350	< 0.001	Food waste	24.387	< 0.001
Distribution losses	14.991	< 0.001	Dairy - agricultural residues	-5.844	< 0.001
Dairy - agricultural residues	-13.528	< 0.001	Distribution losses	4.886	< 0.001
Dairy intensification	9.310	< 0.001	Ruminants - agricultural residues	-4.307	< 0.001
Ruminants - agricultural residues	-9.084	< 0.001	Processing losses	3.982	< 0.001
Ruminant intensification	6.894	< 0.001	Dairy intensification	3.907	< 0.001

NPP_h

2030	t value	Pr(>|t|)	2050	t value	Pr(>|t|)
Bioenergy	167.408	< 0.001	Bioenergy	44.713	< 0.001
Calories	143.516	< 0.001	Calories	38.160	< 0.001
Population	111.922	< 0.001	Population	33.528	< 0.001
% meat	70.369	< 0.001	% meat	20.201	< 0.001
Dairy - agricultural residues	-64.130	< 0.001	Dairy - agricultural residues	-17.530	< 0.001
Ruminant conversion efficiency	-54.943	< 0.001	Food waste	17.249	< 0.001
Yield	-47.287	< 0.001	Ruminant conversion efficiency	-16.696	< 0.001
Dairy conversion efficiency	-45.378	< 0.001	Yield	-15.800	< 0.001
Ruminants - agricultural residues	-42.601	< 0.001	Ruminants - agricultural residues	-12.993	< 0.001
Distribution losses	34.017	< 0.001	Dairy conversion efficiency	-12.590	< 0.001

ΔNPP_{LUC}

2030	t value	Pr(>|t|)	2050	t value	Pr(>|t|)
Bioenergy area	-23.260	< 0.001	Bioenergy area	-27.868	< 0.001
Dairy conversion efficiency	-17.057	< 0.001	Dairy intensification	-18.565	< 0.001
%meat	16.572	< 0.001	Dairy conversion efficiency	-17.651	< 0.001
Dairy intensification	-14.343	< 0.001	% meat	13.053	< 0.001
Ruminant conversion efficiency	-12.371	< 0.001	Ruminant conversion efficiency	-13.030	< 0.001
Ruminant intensification	-8.661	< 0.001	Ruminant intensification	-12.478	< 0.001
Dairy - agricultural residues	7.358	< 0.001	Dairy - agricultural residues	9.064	< 0.001
Ruminants - agricultural residues	5.917	< 0.001	Ruminants - agricultural residues	7.190	< 0.001
Yield	-4.676	< 0.001	Distribution losses	-7.060	< 0.001
Calories	4.277	< 0.001	Poultry intensification	-6.827	< 0.001

NPP_n

2030			2050		
	t value	Pr(>|t|)		t value	Pr(>|t|)
Calories	-58.564	< 0.001	Calories	-29.916	< 0.001
Bioenergy area	-46.356	< 0.001	Population	-26.410	< 0.001
Population	-44.072	< 0.001	Bioenergy area	-25.639	< 0.001
% meat	-41.236	< 0.001	% meat	-21.708	< 0.001
Ruminant conversion efficiency	32.276	< 0.001	Ruminant conversion efficiency	19.214	< 0.001
Dairy conversion efficiency	32.017	< 0.001	Dairy conversion efficiency	17.644	< 0.001
Yield	21.948	< 0.001	Food waste	-13.106	< 0.001
Dairy - agricultural residues	17.655	< 0.001	Yield	12.363	< 0.001
Ruminants - agricultural residues	10.747	< 0.001	Dairy - agricultural residues	9.577	< 0.001
Food waste	-10.630	< 0.001	Ruminants - agricultural residues	7.013	< 0.001

HANPP

2030			2050		
	t value	Pr(>|t|)		t value	Pr(>|t|)
Calories	58.564	< 0.001	Calories	29.916	< 0.001
Bioenergy area	46.356	< 0.001	Population	26.410	< 0.001
Population	44.072	< 0.001	Bioenergy area	25.639	< 0.001
% meat	41.236	< 0.001	% meat	21.708	< 0.001
Ruminant conversion efficiency	-32.276	< 0.001	Ruminant conversion efficiency	-19.214	< 0.001
Dairy conversion efficiency	-32.017	< 0.001	Dairy conversion efficiency	-17.644	< 0.001
Yield	-21.948	< 0.001	Food waste	13.106	< 0.001
Dairy - agricultural residues	-17.655	< 0.001	Yield	-12.363	< 0.001
Ruminants - agricultural residues	-10.747	< 0.001	Dairy - agricultural residues	-9.577	< 0.001
Distribution losses	9.144	< 0.001	Poultry conversion efficiency	-6.860	< 0.001

Average NPP of natural biomes

2030			**2050**		
	t value	Pr(>|t|)		t value	Pr(>|t|)
Calories	-58.446	< 0.001	Calories	-25.109	< 0.001
% meat	-46.664	< 0.001	Population	-22.568	< 0.001
Population	-44.923	< 0.001	% meat	-22.022	< 0.001
Dairy conversion efficiency	36.848	< 0.001	Ruminant conversion efficiency	19.287	< 0.001
Ruminant conversion efficiency	36.398	< 0.001	Dairy conversion efficiency	18.730	< 0.001
Bioenergy area	-35.636	< 0.001	Bioenergy area	-17.531	< 0.001
Yield	32.765	< 0.001	Yield	16.949	< 0.001
Dairy - agricultural residues	13.885	< 0.001	Food waste	-11.043	< 0.001
Grazing intensity	13.103	< 0.001	Grazing intensity	7.130	< 0.001
Food waste	-10.188	< 0.001	Dairy - agricultural residues	6.756	< 0.001

Average NPPt

2030			**2050**		
	t value	Pr(>|t|)		t value	Pr(>|t|)
Bioenergy area	-160.225	< 0.001	Bioenergy area	-46.143	< 0.001
Grazing intensity	-106.641	< 0.001	Grazing intensity	-38.471	< 0.001
Yield	-88.638	< 0.001	Yield	-29.203	< 0.001
Dairy intensification	-59.597	< 0.001	Dairy intensification	-21.572	< 0.001
Dairy - agricultural residues	50.892	< 0.001	Calories	-17.464	< 0.001
Ruminant intensification	-43.715	< 0.001	Ruminant intensification	-16.340	< 0.001
Calories	-41.289	< 0.001	Distribution losses	-15.463	< 0.001
Dairy conversion efficiency	-40.651	< 0.001	Dairy - agricultural residues	14.541	< 0.001
% meat	36.244	< 0.001	Population	-14.467	< 0.001
Distribution losses	-31.826	< 0.001	Processing losses	-14.396	< 0.001

Appendix VI: Sensitivity analysis testing non-continuous variables

As described in section 5.2.2, in order to test sensitivity to two separate non-continuous variables, the original sensitivity analysis was repeated twice more, with these inputs in their alternative states. Figures showing the results of these analyses are presented here, also showing the mean values produced in the original sensitivity analysis.

Afforestation

Instead of bioenergy crops grown on vacated food producing land, carbon accumulation curves based on forest growth are applied.

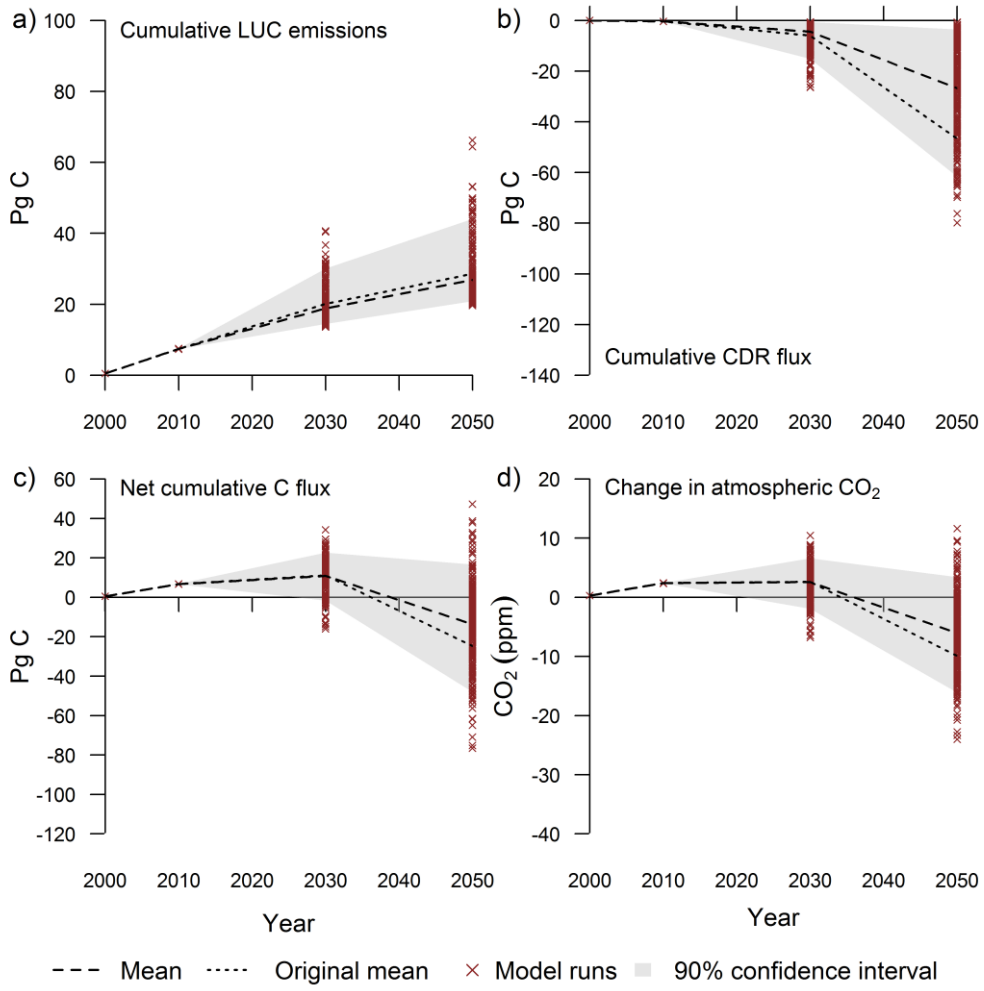

Figure A.4: Cumulative carbon fluxes under afforestation. CDR is much lower.

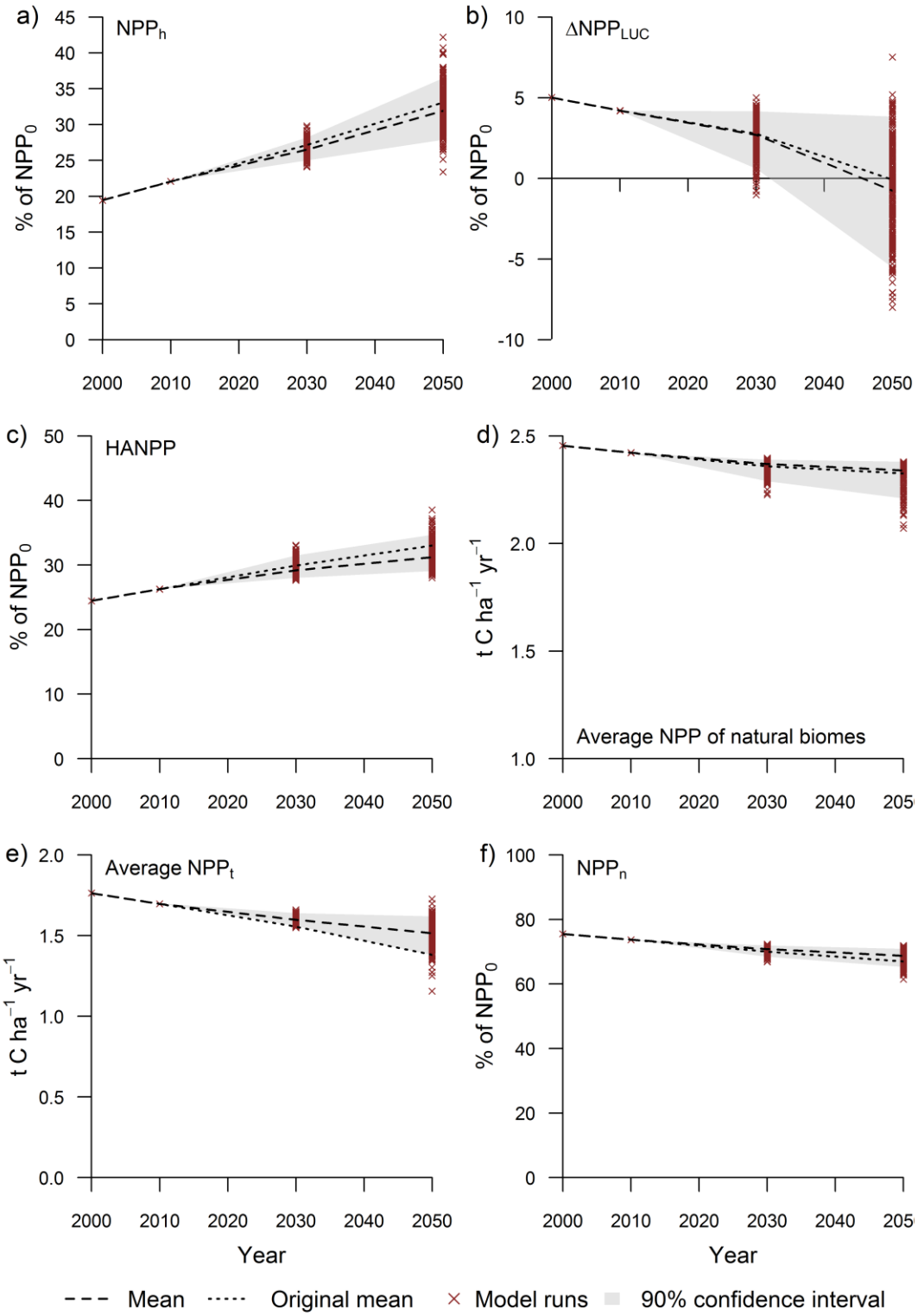

Figure A.5: Macroecological indicators under afforestation. In general more NPP remains in ecosystems, due to less intensive biomass harvest.

Alternative dietary scenario

Instead of trends derived from 1970-2010 data used to drive diet, trends from 1990-2010 are used, with the main difference being a larger predicted growth in consumption of livestock products.

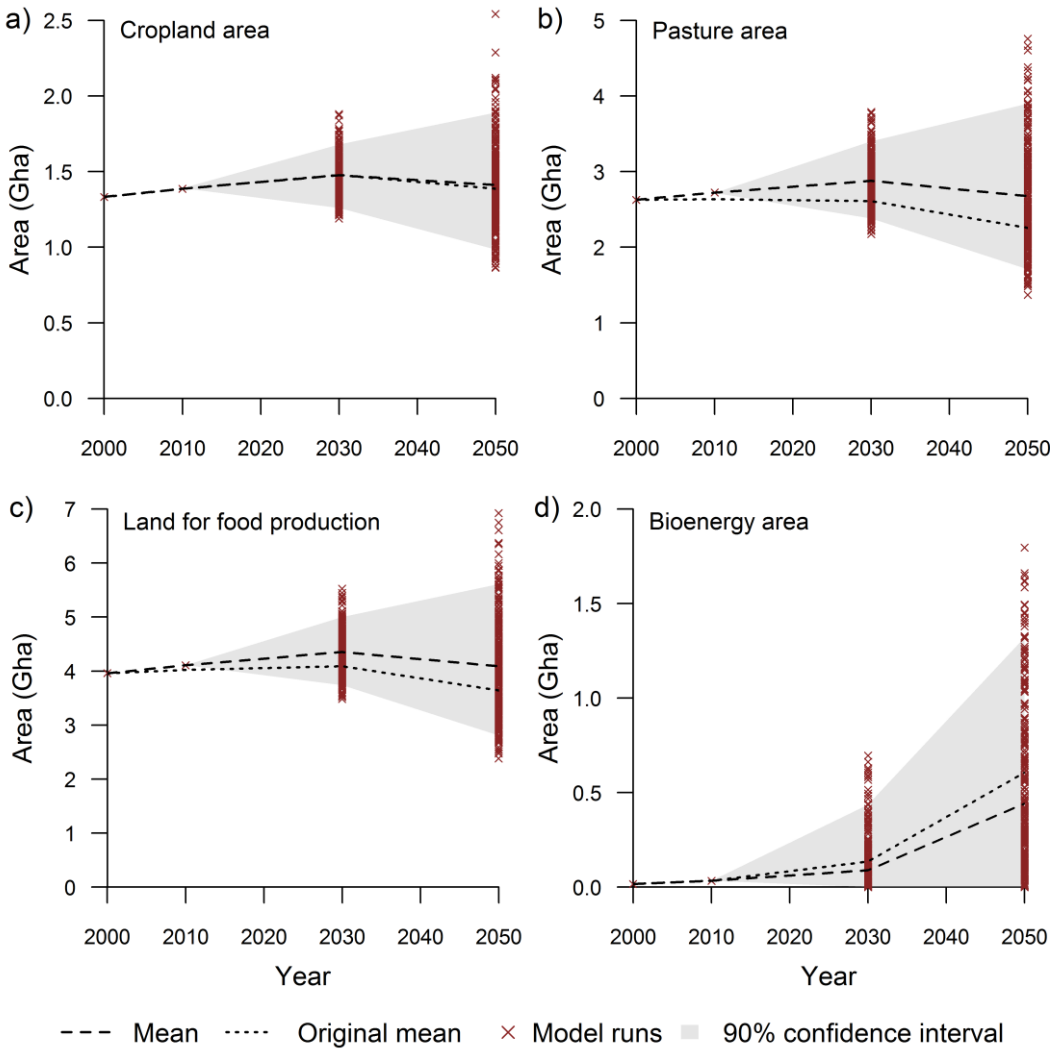

Figure A.6: Land-use under diet based on 1990-2010 trends. The area of land required for food production is higher under these inputs, due to more rapid growth in demand for livestock products.

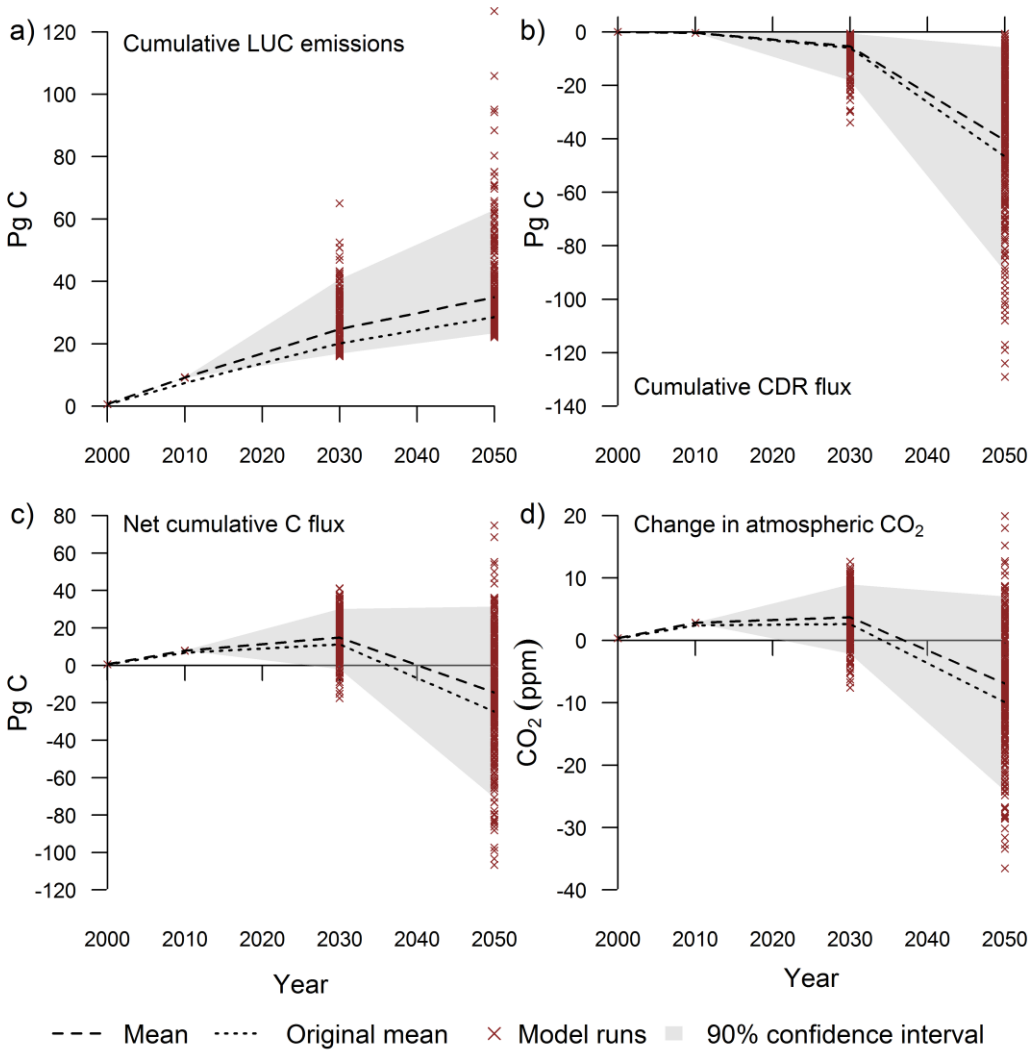

Figure A.7: Cumulative C fluxes under alternative diet inputs.

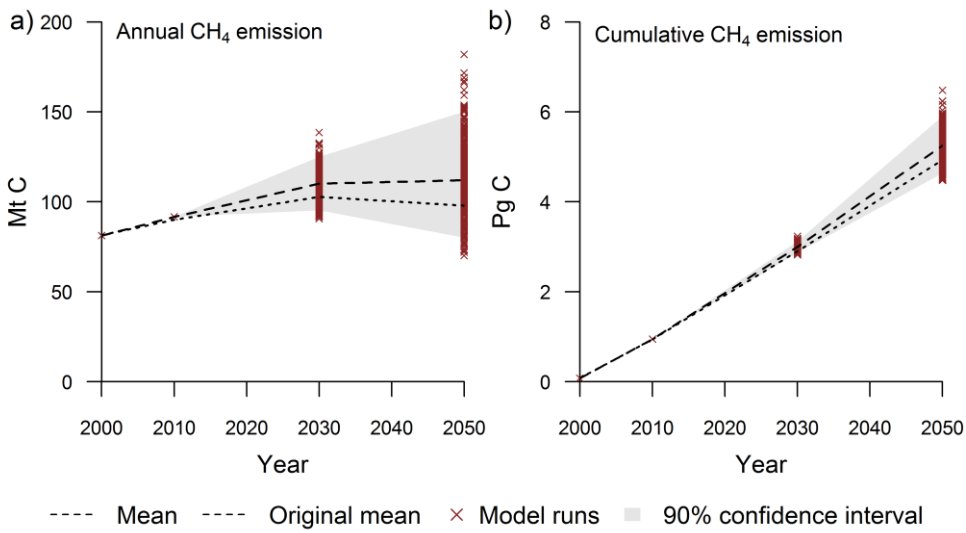

Figure A.8: CH_4 fluxes under alternative diet inputs.

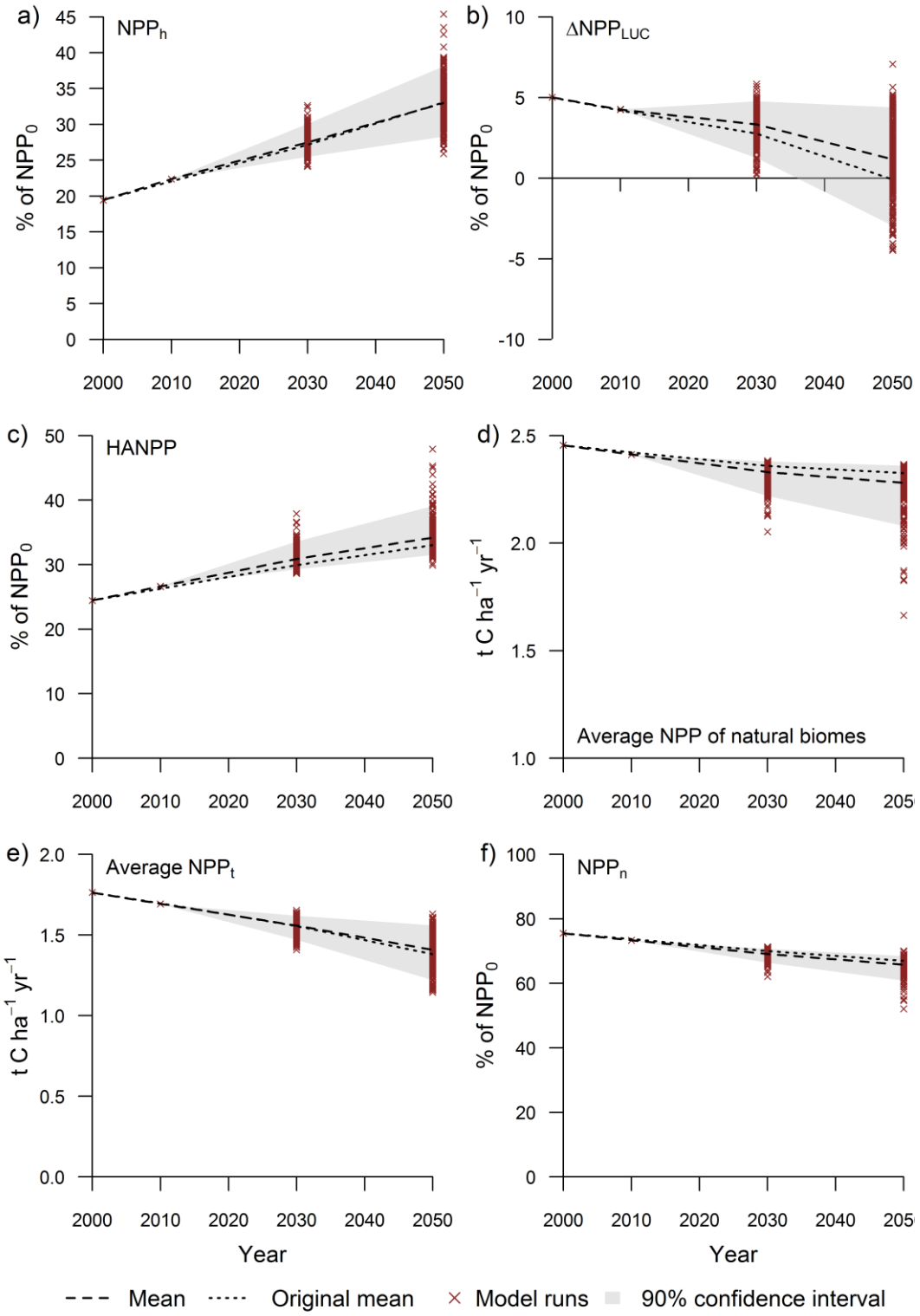

Figure A.9: Macroecological indicators under alternative diet inputs. Total harvest is higher, but actually slightly less intense than under the original analysis.

Appendix VII: Global biomass flows; Sankey diagrams not included in the text

This appendix contains Sankey diagrams for 2050 of the scenarios not shown in Chapter 6.

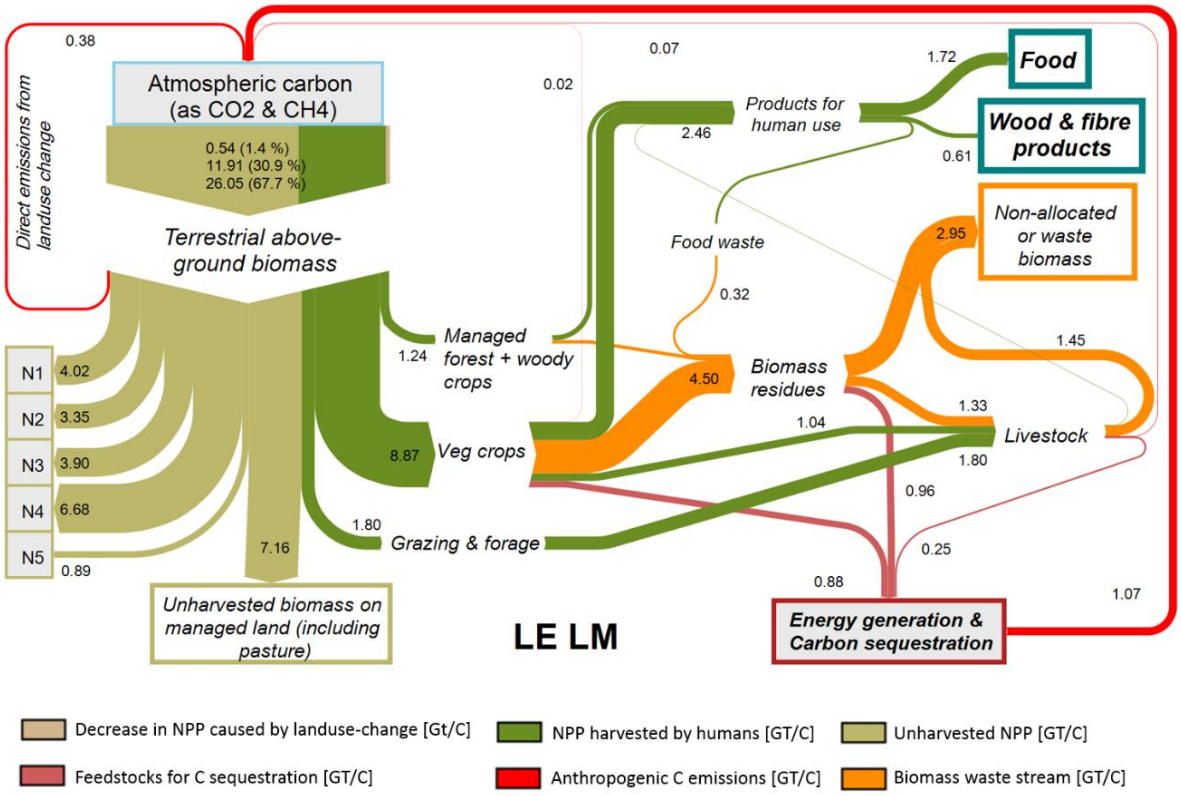

Figure A.10: Global biomass flows in 2050 for the LE LM scenario.

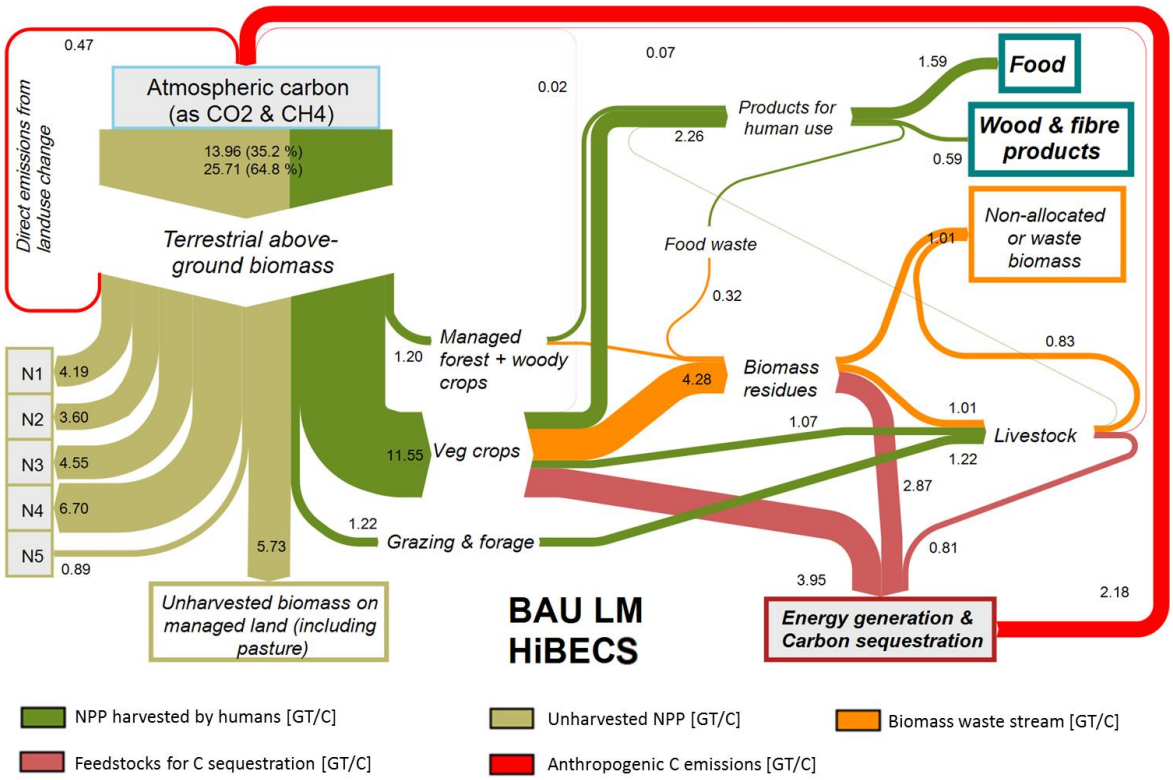

Figure A.11: Global biomass flows in 2050 for BAU LM HiBECS.

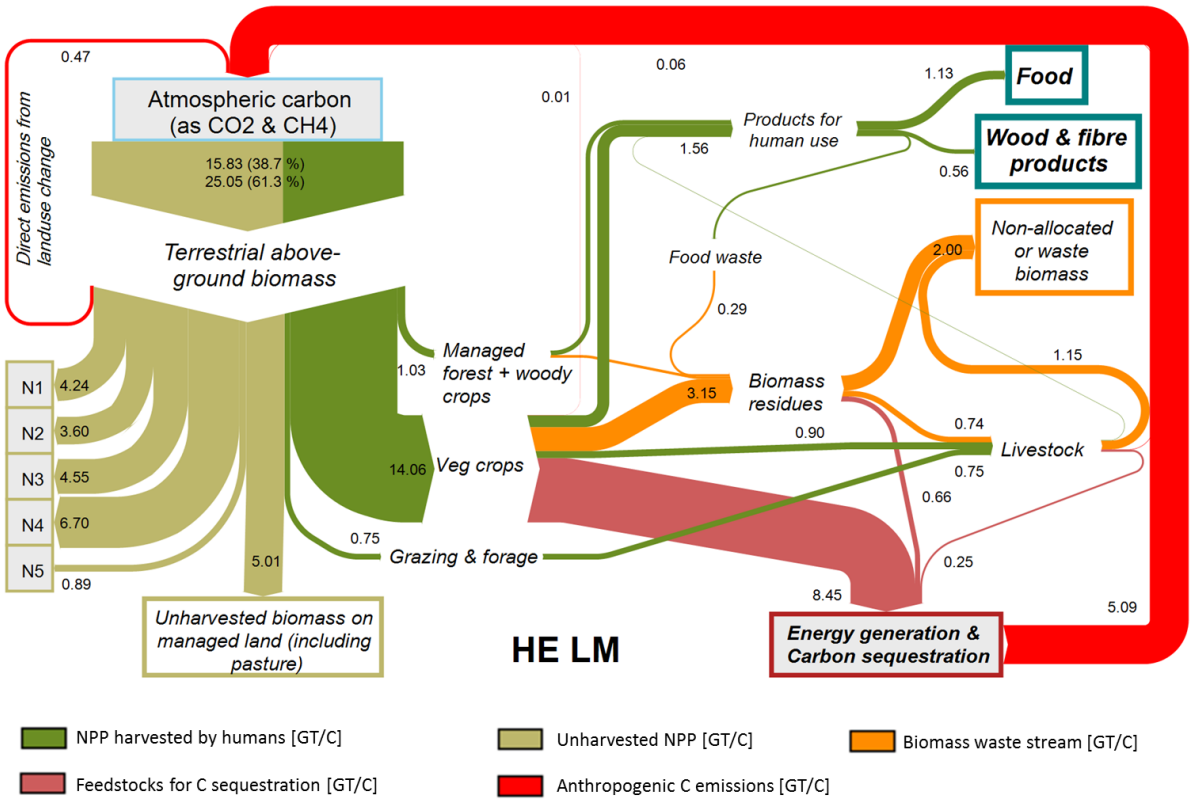

Figure A.12: Global biomass flows in 2050 for HE LM.

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