Online Annexure

Chapter 8 Applications of the TEEBAgriFood Framework: Case Studies for Decision-makers

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1. Agriculture management systems

This family of examples focuses on agricultural management systems. We showcase two examples that demonstrate application of the framework, i) rice under different management practices, ii) mixed crops under organic and conventional management systems.

1.1. Rice management practices: agroecological versus conventional

Rice is central to the food security of half the world and the production of rice is essential to the food security and livelihoods of around 140 million rice farming households (FAO, 2014). Rice production provides a range of ecosystem services beyond food production (i.e. cereal grain) alone.

At the same time, rice production has been linked to a range of different environmental impacts such as high GHG (greenhouse gas) emissions, air and water pollution as well as an increase in water consumption. Information such as that provided here can inform decisions on how to manage and mitigate these impacts while providing affordable, nutritious, equitably accessible and safe food for a growing global population with changing patterns of consumption.

Objectives and scope

As these challenges are not independent, but rather interlinked, reaching them is likely to require trade-offs. The question of interest is therefore of how to reduce trade-offs between these different goals. Where possible, one should identify synergies that allow for a maximization of benefits, while minimizing costs to society and the environment, (i.e. negative externalities), and the wellbeing of the farmer him or herself through the degradation of natural capital from rice production. It is therefore crucial to know which types of farm management practices or systems offer the best options to reach these synergies, and reduce trade-offs. The specific objectives of the TEEB rice study (Bogdanski et al., 2016) were three-fold:

1. To identify visible and invisible costs and benefits of rice agro-ecosystems; i.e., externalities.
2. To identify and assess those rice management practices and systems which reduce trade-offs and increase synergies.
3. To make these trade-offs and synergies visible by assigning biophysical or monetary values to the different options.

The TEEB rice study set out to describe a variety of trade-offs and synergies that occur in rice agro-ecosystems in five case study countries all around the globe: the Philippines and Cambodia in Asia, Senegal in Africa, Costa Rica in Latin America and California/The United States in North America. The analysis makes a distinction between the three most common rice growing environments: irrigated lowland, rainfed lowland, and rainfed upland systems.

The different typologies are based on altitude (upland vs. lowland) and water source (irrigated or rainfed). The study consists of two parts, a biophysical quantification and an economic valuation. The biophysical framework for analysis details what should be included in the eventual monetary and non-monetary analysis.

Approach and methodologies
The rice production systems were categorized by 28 rice management systems and practices, which are starting with land preparation and finishing at harvest. The study has set out to identify those farm management practices that offer the best options to reach synergies, and reduce trade-offs between different management objectives. Several scenarios, i.e. pairwise comparisons, were applied to show the effect of the various farm management practices on different environmental and/or agronomic variables:

1. The baseline scenario describes a conventional management approach, for instance herbicide use to combat weeds.
2. The alternative scenario describes a farm management practice that is expected to decrease an environmental impact or to increase an ecosystem service. For instance, instead of herbicide use, hand weeding or biological control could be practiced.

The team extracted data from peer reviewed literature from all five case study countries and synthesized them in a vote-counting analysis. The final outcome was a statistical review of primary research, i.e., peer reviewed literature, on the effects of different agricultural management practices on different environmental, agronomic and ecosystem variables. Next, impacts were modelled biophysically, that are caused by changing physical conditions. This includes identifying factors such as the endpoint of nutrient run-off, which may be adjacent freshwater ecosystems for example, and quantifying the change in the biophysical indicator that is to be valued, such as the change in the quality of human health, measured in disability adjusted life years (DALYs).

The final step involved the economic modelling component of the valuation. This includes the identification of the final recipient of the impact, such as the local populations who experience the negative effects of eutrophication, and then selecting an appropriate valuation technique to monetize the change in biophysical conditions. A complementary study to the TEEB rice study is the LEGATO project (http://www.legato-project.net/index.php?P=7), which aims to quantify the dependence of ecosystem functions and the services they generate in irrigated rice systems in Southeast Asia.

Results
As the TEEB rice study has been designed to be a trade-off analysis, the results have been structured according to the effect of different management practices on two contrasting or synergistic ecosystem benefits or costs. The assumptions that underpin the analysis refer to rice production, on the one hand, and a range of different externalities, i.e., an environmental impact or ecosystem service, on the other, to show potential trade-offs or synergies between the two. Two examples are given below:

1. Increasing rice yields versus reducing water consumption.

Worldwide, about 80 million hectares of irrigated lowland rice provide 75% of the world's rice production. This predominant type of rice system receives about 40% of the world's total irrigation water and 30% of the world's developed freshwater resources (Bouman et al., 2007). The dependence on water of the rice farming sector is a huge challenge as freshwater resources are becoming increasingly depleted due to competing water uses from the residential and industrial sector and as rainfall is increasingly erratic due to climate change and variability. More efficient water use is therefore a must, yet given the importance of rice to food security around the globe, any trade-offs need to be carefully assessed.

Thus, this study sought undertook to assess and valuate trade-offs resulting from irrigation management, soil preparation and crop establishment on rice yields, on the one hand, and water consumption, on the other. The study analyzed the change in yield and water consumption under continuous flooding under conventional rice cultivation and water-saving practices such as alternate wetting and drying (AWD) and the system of rice intensification (SRI). The study further compared dry tillage to puddling, and direct seeding to the transplanting of seedlings; two other practices used in water-saving rice production systems. Box 1 shows the effects of water-saving rice production systems and conventional management on irrigated (IL) and rainfed lowland (RL) system in Senegal, Cambodia and the Philippines based on data from Krupnik et al., (2010), Krupnik et al., (2012a), Krupnik et al., (2012b), Miyazato et al., (2010) Dumas-Johansen (2009), Koma (2002), Ly et al., (2012), Ly et al., (2013) and Satyanarayana et al., (2007).

Box 1. Scenario analysis: water-saving rice production systems versus conventional management

Management systems to ensure water savings include intermittent flooding as part of the production package. Such systems advise transplanting of young (eight to ten days old) single rice seedlings, with care and spacing, and applying intermittent irrigation and drainage to maintain soil aeration. In addition, the use of a mechanical rotary hoe or weeder to aerate the soil and control weeds is encouraged.

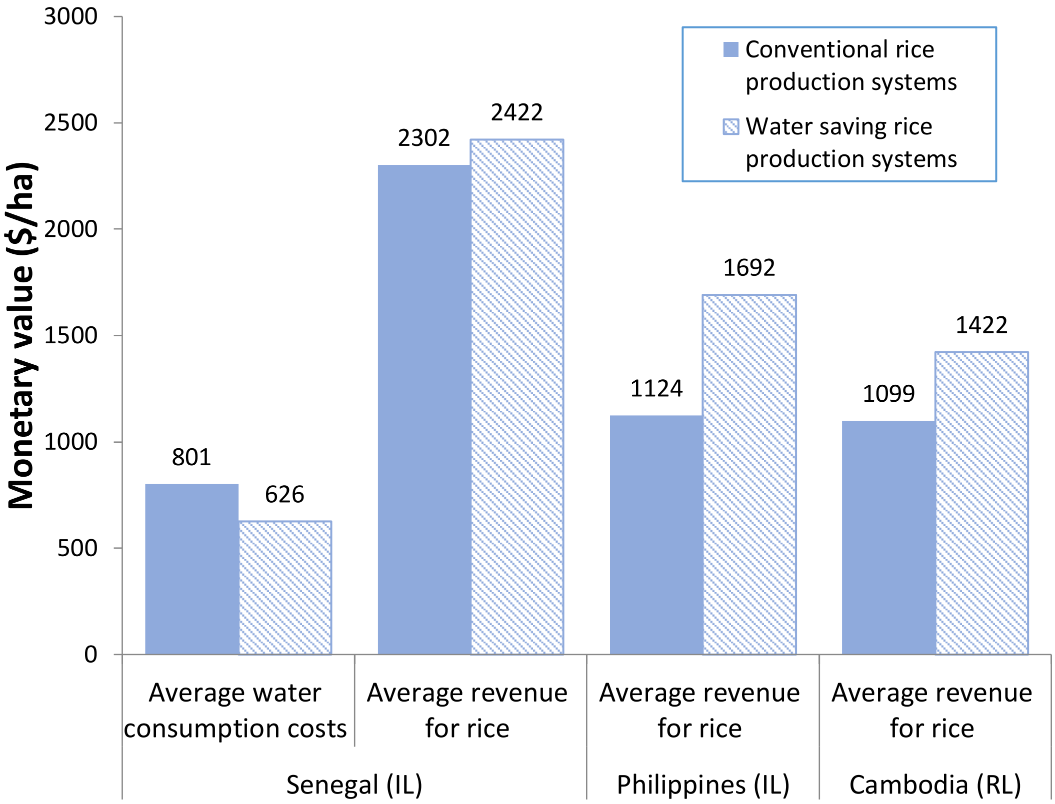

Figure 1. The comparison of conventional rice production systems and water-saving rice production systems for Senegal, Philippines and Cambodia in terms of average revenue for rice and environmental and health costs of water consumption. Note that the water consumption methodology used estimates a country specific monetary value for the health and ecosystem impact per cubic metre of water used for rice cultivation or any other purpose. (source: Bogdanski *et al.*, 2016)

If Senegal was to change all its irrigated lowland systems from conventional management to water-saving rice production systems, the society would save about US$ 11 million in water consumption related health and environmental costs. At the same time, the rice producer community would gain a total of US$17 million through yield increases – a clear synergy.
If the Philippines were to change all their rainfed lowland systems from conventional management to water-saving rice production systems, the rice producer community would gain a total of US$750 million through yield increases. Data on water consumption was not recorded.

If Cambodia was to change all its rainfed lowland systems from conventional management to SRI, the rice producer community would gain a total of US$801 million through yield increases. No irrigation water consumption costs result from this farming system as it is dependent on rainfall only.
While extrapolating the results from a few studies only for an entire country may show some general trends, one needs to be cautious about the context of each study. Yield increases with water-saving rice production systems are highly variable and mainly occur in

highly weathered soils, whereas in ideal rice soils yields tend to be the same or less with such systems (Turmel et al., 2010).

2. Increasing rice yields versus reducing GHG emissions.

Global estimates attribute about 89 percent of rice global warming potential to methane (CH_4) emissions, which are due to flooding practices in irrigated and rainfed lowland systems (RL) (Linquist et al., 2006). To a much smaller degree, the production and application of N-fertilizers contributes to the rice global warming potential. And also emissions from rice straw burning impact global climate change. In addition to rice production being a major emitter of GHGs, rice systems may also sequester carbon via soil organic carbon. Yet overall, rice production is a net producer of GHG emissions. The potential for alternative, modified water and nutrient management systems to generate positive benefits for both yields and the environment in studies in Cambodia are presented in Box 2.

Box 2. Scenario analysis: modified water and nutrient management systems versus conventional management in rainfed lowland rice production in Cambodia

The TEEB rice study sought to assess and monetize the trade-offs and synergies resulting from irrigation water management, residue management, fertilizer application and the choice of rice varieties on rice yields, on the one hand, and GHG emissions, on the other.

In studies in Cambodia, fields under modified water and nutrient management were kept moist during transplanting and drained several times during the growing season. Trade-offs were thought likely to occur between CH4 emissions when the fields are flooded and N2O emissions when fields are drained; but in fact synergies were found to occur between both yields and reduction of GHG emissions under modified water and nutrient management systems.

Data derived from Dumas-Johansen (2009), Koma (2002), Ly et al (2012), Ly et al., (2013) and Satyanarayana et al., (2007) collected in RL systems in Cambodia led to a value of rice production of US$1099 per hectare when conventional management was practiced and US$1422 when SRI was implemented (Figure 2). The value of rice production was estimated on the basis of the country specific revenue for rice grain received per ton of paddy rice. Primary data on GHG emissions as reported in the peer reviewed studies was used to model the GHG emission costs. The cost of GHG emissions were valued following the Trucost Greenhouse Gas methodology, which provides a valuation coefficient for CO_2 equivalent emissions, based on the social cost of carbon emissions.

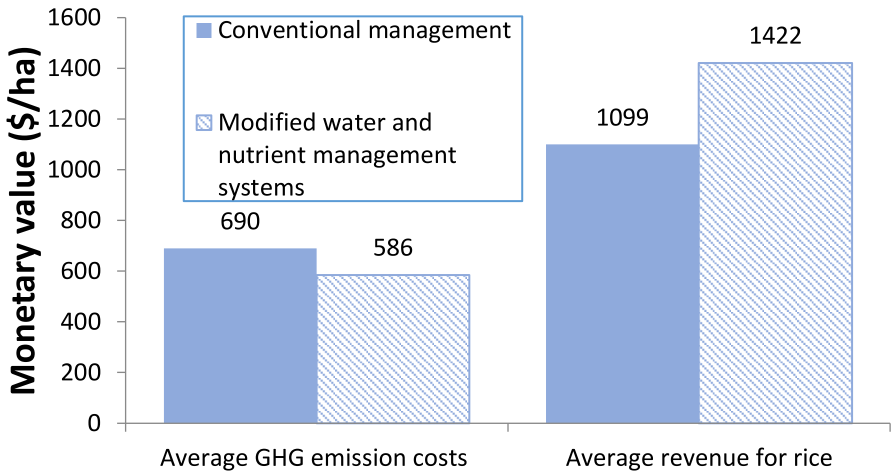

Figure 2. Valuation results comparing conventional management to modified water and nutrient management systems in terms of GHG emission costs and rice grain revenues in rainfed lowland systems in Cambodia. Monetary valuation for GHG emissions is based on primary research data (Source: Bogdanski et al, 2016).

The monetary valuation for GHG emissions in Cambodia's RL paddies resulted in an average cost of US$690 per hectare of rice production for conventionally managed systems and US$586 for systems with modified water and nutrient management – a reduction in costs of 15%

If all rice farmers in RL systems in Cambodia would adopt modified water and nutrient management, it is estimated that this would increase the revenue of rice by US$ 801 million. At the same time, society would have to spend US$ 258 million less in GHG emission costs.

3. Impacts of pest control management practices on regulating services.

The complexity of biological systems, which TEEBAgriFood seeks to capture, show that there are many intermediate factors and products in agricultural production systems, generated by biodiversity and ecosystem services. For example, ecological alternatives to pest control are not simply a matter of not applying pesticides, and thus avoiding such costs; it is an intricate process (on the part of nature, when not impeded by humans) of building a natural enemy community, through a management that encourages such ecological processes. This form of management has some value in an initial year and place, but adds value as it is allowed to flourish over time and space; thus not reflected adequately in a simple calculation of pesticide costs avoided. Methodology to quantitively assess the value added of long term investment in ecological infrastructure would be needed to incorporate this observation in TEEBAgriFood analyses.

As has been well documented in a landmark study on the ecology of Asian rice production systems (Settle et al., 1996) rice ecosystems under careful management are capable of sustaining important regulating services. Under such management, there may be no need of external inputs of pesticides, and there are important synergies related to nutrient management. The LEGATO project, previously mentioned, has shown invertebrates can contribute to soil fertility in irrigated paddy fields by decomposing rice straw, an ecosystem service that is degraded by the application of pesticides.

However, over the last few decades in many parts of the world, rice agriculture has become (or is perceived to be) heavily dependent on agricultural inputs. Synthetic fertilizers are used to boost yields, while pesticides and herbicides are applied to address pest outbreaks and weed manifestation. Weeds are a major challenge in rice production worldwide. With respect to insect outbreaks, it is still unclear whether pesticides (particularly insecticides) actually increase rice yields (Heong et al., 2015). Agricultural expansion and intensification can often lead to a change in the ratio of predatory invertebrates to herbivorous invertebrates. One of the main reasons is the misuse of pesticides. Not only is rice production itself affected but also the adjoining waterways, their wildlife such as fish and birds and the supply of drinking water. Finding alternative ways to address pests is therefore very important.

Another reason is the increased use of fertilizer. Increasing fertilizer use often leads to higher disease incidence and a greater abundance of herbivorous insects and mites. This, in turn, often leads farmers to apply higher levels of pesticides and thereby reduce ecosystem efficiency and reduce water quality (Horgan and Crisol, 2013; Spangenberg et al., 2015).

4. Cultural services of heritage rice systems.

The importance of cultural ecosystem services provided by rice systems is briefly mentioned in the rice feeder TEEB study and more elaborated in papers of the LEGATO study by Tilliger et al., (2015) and Castonguay et al., (2016).

That rice agro ecosystems are a source of cultural services is evidenced by the Ifugao province rice terraces in Philippines. In 1995 rice terraces of four municipalities of the Ifugao province have been designated to the UNESCO world heritage list (EEPSA, 2008; UNESCO, 2010). According to UNESCO the rice terraces were constructed 2000 years ago and illustrate ancient civilization with rituals, chants and symbols to enhance ecological balance: "…ensuring the authenticity of both the original landscape engineering and the traditional wet-rice agriculture". In 2009 in total over 102,000 tourists (domestic and international) visited the various municipalities in Ifugao, of which most (79,000) visited the Banuae rice terraces. This provides economic opportunities in the area (Vafadari, 2012). The rice terraces, including the irrigation channels, are being deteriorated. Major challenges are an increased pressure by tourists, inadequate water supply, poor/deteriorated irrigation systems and a lack of conservation of the terraces. This has led to farmers abandoning their terraces; many farmers have stopped farming to get jobs in the tourist sector (income from rice production is low), which also stopped the conservation of the terraces. In addition, the younger generation loses interests and migrates as well as pest invasions are increasing such as the Golden Apple Snail, putting pressure on the rice systems (Calderon et al. 2010; EEPSA, 2008; Tilliger et al., 2015; Castonguay et al., 2016).

According to the study of Castonguay et al. (2016) "several indicators of the social-ecological system of Banaue have remained constant through time. This is the case with the multi-functionality of forests, nutrition diversity, food self-sufficiency, and the primogeniture inheritance system, which shows that ecosystem services and related benefits have in a large part been conserved despite changes in the social and ecological subsystems". However, results of both studies (Tilliger et al., 2015; Castonguay et al., 2016) have shown that traditional rituals have decreased as well as the cultivation of several traditional rice varieties. Various reasons are given for this decrease. Cultural identify and cultural services, for example traditional knowledge as well as spiritual experiences, are decreasing due to a change in lifestyle and interests. Especially the younger generation lack the traditional knowledge and they are losing interest in agriculture. Commercialization of traditional culture for tourism, Christianisation and high cost to maintain rituals are also other reasons of a decrease in cultural services according to Castonguay et al. (2016). They stated: "The change in the practice of rituals may affect land-use practices, e.g., the synchronicity of the cropping season, which relates to the timing of rice transplanting and which was tightly accompanied by rituals in the past. Such changes may lead to greater pest problems because of poor synchronization of planting."

Study Recommendations

In a broad sense, this case study shows that by assessing farming systems as a whole, taking negative and positive externalities into focus along with standard production metrics, it is possible to highlight key synergies and trade-offs. Often where trade-offs are expected in rice production systems, alternative management practices may result in win-win outcomes. A TEEBAgriFood type analysis can bring such opportunities to the attention of decision makers and point out where trade-offs can be minimized, synergies can be maximised, and yields can be maintained while ecosystem services are being generated and enhanced.
On a more detailed level, the results show that the development of a solid typology that is further disaggregated into specific farming systems and practices is key to valuing externalities from the agriculture and food sector. The results have confirmed the need for practice and location specific typologies to show the full range of externality benefits and costs. As the results clearly show, environmental impacts and ecosystem services linked to rice farming strongly respond to the type of agricultural management practiced. Rice farming or agriculture in general, is too often categorized as one homogenous activity, when in reality farming is extremely diverse. This study has therefore made an attempt to go beyond production systems and rice growing environments, zooming in on the different ways in which rice is produced. While evidence gets scarcer the more detail is added to the typology – a true challenge indeed - the authors of this study are convinced that there is no other way if one aims to do justice to the diversity of the farming sector.

Further research

In order to provide a holistic assessment of a farming system it requires that experimental studies provide a comprehensive data set that goes beyond food production alone as is typically done in agronomic studies. Likewise, ecological and environmental studies need to record agronomic values, including yields, and widen their often restricted focus on natural resources and biodiversity alone. In addition, environmental and socio-economic benefits and costs are often studied in isolation from each other, despite them being closely interconnected.

Instead of relying on the scientific data alone, there may be large scope for applying a TEEB-type analysis to specific farms, and making greater use of on-farm, farmer-led research. Follow up studies must focus on integrating the available evidence into models that –using data were available, and expert knowledge where not- can provide insightful comparisons of conventional practices versus alternative, more ecologically-based production systems.

There is also a need to improve current valuation methodologies, as there is a clear lack of those that can value agroecosystem benefits, and the generation of these benefits over time, as opposed to costs. There is a need to link economic valuations to market costs, and avoided costs for the farmer. Methods are urgently needed to be able to assess and compare multi-dimensional values, as monetary analysis is not appropriate for all positive and negative externalities of agriculture. Furthermore, one needs to better adapt current models for valuation to the realities of developing countries. While ecosystem valuations usually focus on the local level, ecosystem accounting methods aim to aggregate information to produce statistical results at the national level. Since both areas of expertise are still in its infancy, it is timely to join forces now in order to follow a coherent approach in the future.

1.2. Organic versus conventional agriculture

Organic agriculture has evolved over time as an alternative to chemical intensive farming with a singular focus on productivity to a management system that aims to optimise the 'health' of soil, plants, animals and people. As an example of one application of TEEBAgriFood evaluation framework at the agricultural management system, the following study that compares organic and conventional agricultural system at field, region and global scale is discussed here.

Objectives, and scope of the study

The role of ecosystem services (ES) in two contrasting agricultural systems was investigated under organic and conventional arable systems in Canterbury, New Zealand by using an experimental 'bottom-up' approach comprising field experiments to quantify ecosystem services (Sandhu et al., 2008). It focuses on one sector (arable farming) of agriculture. It attributes economic values to a suite of ecosystem services, which were quantified experimentally, in contrast with earlier evaluations, which have used 'value transfer' approaches (Boyle and Bergstrom, 1992; Costanza et al., 1997). The total economic value of ES in arable land in the province of Canterbury, New Zealand is also calculated here by using 'bottom-up' approach and extrapolation using GIS techniques. It also provides information on the change in the economic value of ES in a scenario in which conventional farming shifts to organic farming at regional scale and at global scale.

The Canterbury region contains the largest area in New Zealand focused on the production of crops on about 125,000 ha of arable land. The rest of the agricultural land consists of land in horticulture, grasslands, forest plantations, tussock used for grazing, native bush and native scrub. In this work, 29 arable fields were selected, distributed over the Canterbury Plains and comprising 14 certified organic and 15 conventional fields with a mean area of 10 ha. The crops grown in these fields were wheat, barley, carrots for seed, process peas, field beans, white clover for seed and onions.

ES associated with arable farming in Canterbury, New Zealand were assessed by conducting a series of field experiments to assess each ES. The economic value of each ES (in 2005 US dollars; US $ 1=NZ $ 0.70) was then calculated for each of the 29 fields. The total economic value of ES for each field was calculated by summing the total of all the individual ES values measured. These were: biological control of pests, ES_1; soil formation, ES_2; mineralisation of plant nutrients, ES_3; pollination, ES_4; services provided by shelter- belts and hedges, ES_5;

hydrological flow, ES_6; aesthetics, ES_7; food, ES_8; raw material, ES_9; carbon accumulation, ES_{10}; nitrogen fixation, ES_{11}; soil fertility, ES_{12}, which are described in Sandhu et al. (2008, 2010). The market value of ES included the economic value of products and raw materials produced (grains, seed, peas for processing, onions, straw bales) and traded by farmers in the market. The rest of the ES comprised non-market values.
Assuming a shift of half of the conventional area to organic, the change in the value of ES for Canterbury arable land is calculated by using the value of organic and conventional areas, i.e., applying the value of ES generated by organic systems to half of the area of conventional farms.

Results

Comparison of organic and conventional agriculture management systems

Total economic value of ES in organic fields ranged from US \$1610 to US \$19,420 $ha^{-1} yr^{-1}$ and that of conventional fields from US \$1270 to US \$14,570 $ha^{-1} yr^{-1}$ (Table 1). All 12 ES were higher in organic fields as compared to the conventional ones (Figure 3).

Table 1 Summary of mean and range of economic value of ecosystem services in organic and conventional fields (Sandhu et al., 2008).

		Economic value (range) in US \$ $ha^{-1}yr^{-1}$	
	Ecosystem services	Organic fields	Conventional fields
1	Biological control of Pests	50 (0-100)	0 (0-0)
2	Mineralization of plant nutrients	260 (26-425)	142 (30-349)
3	Soil formation	6 (0.7-11)	5 (2-9)
4	Food	3990 (1150-18900)	3220 (840-14000)
5	Raw materials	22 (0-224)	38 (0-298)
6	Carbon accumulation	22 (0-210)	20 (0-210)
7	Nitrogen fixation	40 (0-92)	43 (0-92)
8	Soil fertility	68 (53-82)	66 (54-73)
9	Hydrological flow	107 (-111 - 190)	54 (-118-194)
10	Aesthetic	21 (21-21)	21 (21-21)
11	Pollination	62 (0-438)	64
12	Shelterbelts	880 (0-4725)	200 (0-617)

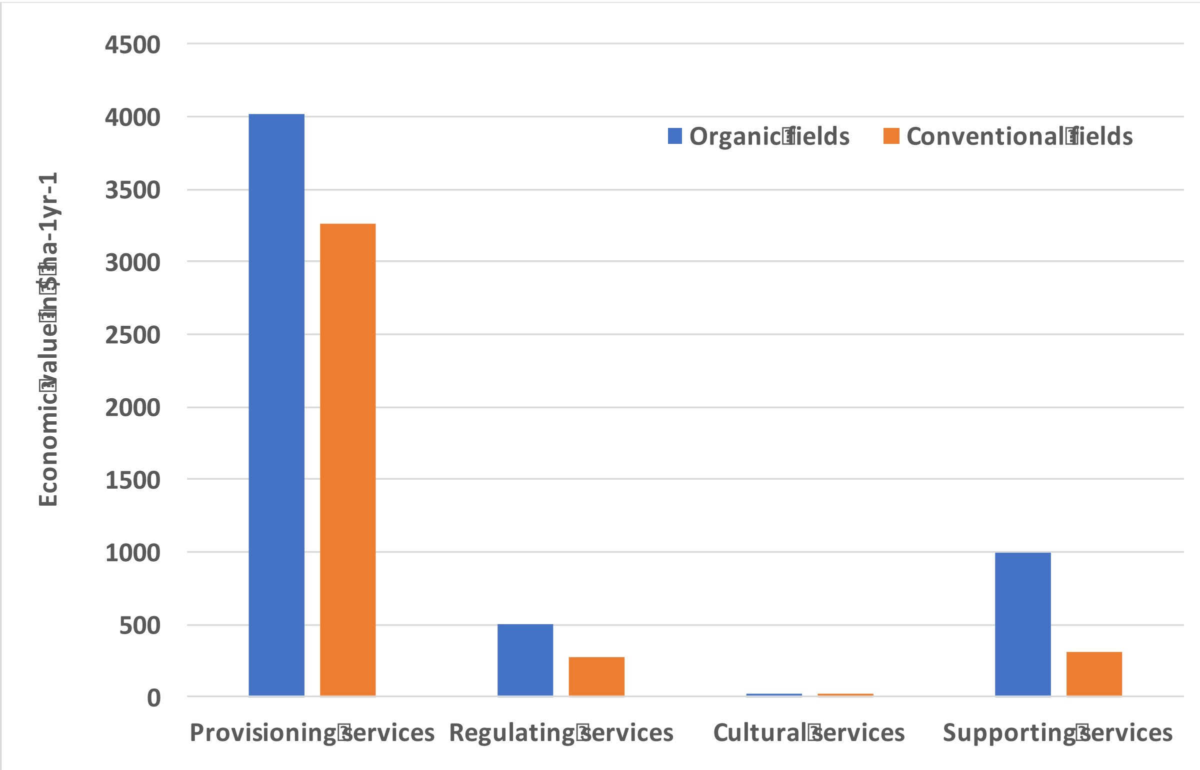

Figure 2 Economic value of ecosystem services under organic and conventional agriculture. (Sandhu et al., 2008)

Regional scale organic agriculture

Assuming the minimum and maximum values of total and non-market values of organic and conventional fields, the economic value of Canterbury arable land was calculated. Under the fully-conventional scenario, the GIS-based analysis produced an estimated total ES for Canterbury of US $468 million annually, with non-market ES accounting for c. US $100 million of the annual total. With a conversion to a half- organic scenario, the estimated total Canterbury ES was US $505 million annually, with non-market ES comprising US $142 million of total annual ES. This was an increase in total and non-market ES of US $37 million and US$ 42 million, respectively. It is estimated that a 1% to 45% increase in non-market ES would occur in Canterbury as a result of a conversion of half of the conventional arable farms to organic practices.

Global scale

To illustrate the potential of the relative magnitudes of two ES out of 12 investigated (Biological control of pests and mineralisation of plant nutrients) for world farming, extrapolations are provided, with appropriate caveats, to temperate arable areas in 110 countries in 15 global regions (Table 2) along with the economics of total N consumed and total pesticide use in those regions (in 2012 US $; Sandhu et al., 2015). The caveats included; regional climatic conditions, social-political factors, crop management changes and their costs, rate of uptake by farmers were not taken into consideration while extrapolating the results (Sandhu et al., 2015). The 15 regions were selected on the basis of temperate climatic conditions occurring in up to two-thirds of the total country area, as the field data used here were derived from New Zealand temperate conditions. Information on the area under four previously-selected crop types, their production and the amount of fertiliser and pesticide used for each of the 110 countries in the 15 regions were obtained from FAOSTAT (2014). The potential economic value of the two ES (biological control value for organic fields only, it was zero in conventional fields) and N mineralisation value for conventional and organic fields) was extrapolated from the New Zealand arable study (Sandhu et al., 2008; Sandhu et al., 2010).

Table 2 Total value of inputs in 15 global regions for target crops (PBBW; peas, beans, barley and wheat) and economic value of two key ecosystem services combined for 100% and 10% of the global arable area under organic management for above crops (Sandhu et al., 2015).

	Regions	Total value of pesticides and fertilisers in PBBW area (US million/yr)	Total value based on two ES in 100% of PBBW area (US million/yr)	Total value based on two ES in 10% of PBBW area (US million/yr)
1	Eastern Africa	0.3	0.8	0.3
2	Northern Africa	665.9	836.1	682.9
3	Southern Africa	28.9	115.7	37.6
4	South America	381.5	1165.7	459.9
5	Northern America	2872.4	5139.6	3099.1
6	Central Asia	154.1	1323.8	271.0
7	Eastern Asia	5347.6	6225.8	5435.4
8	Southern Asia	1347.2	2615.0	1474.0
9	South-eastern Asia	0.02	3.1	0.3
10	Western Asia	1994.6	2026.5	1997.9
11	Eastern Europe	1720.8	6487.5	2197.5
12	Northern Europe	1192.5	2191.4	1292.4
13	Southern Europe	1180.4	1731.2	1235.4
14	Western Europe	2871.8	4286.4	3013.2
15	Australia and New Zealand	360.5	531.8	377.7
	Total	20119.1	34680.9	21575.3

The extrapolations used here are only illustrations of the potential relative magnitudes of ES in conventional and organic fields and are not precise forecasts. This approach, however, can help improve the understanding of the potential contribution of ES provided by non-traded species to global agriculture. It does not advocate large-scale conversion to organic practices. However, if only 10% of the global arable area utilised such ES-enhancing techniques, then this study shows that the total ES value can then surpass the total cost of inputs.

Study recommendations

This study documents the impressive value of non-marketed ES, and the organic farmers who depend on these ES make savings for not using costly pesticides and fertilsers. There is a strong economic case for the organic agriculture as demonstrated here that can help change the role of farmers from primary providers of food and fibre to managers and providers of all ES. These studies also advocate the importance of economic valuation of ES in agriculture as the evidence of ecological disturbances sometimes does not generate much attention unless the evidence includes dollar values (Ghaley et al., 2015). At global level, conventional farming often suppresses the delivery of non-marketed ES whereas organic and other benign agricultural practices enhance it. This study strengthens the case for more diversified, ES–rich, integrated agricultural systems that enhance functional agricultural biodiversity, avoid expensive inputs, minimise external costs and are less energy intensive (Sandhu et al., 2015).

Further research

Future research should aim to develop mechanism to support ES rich farming. It should aim to include all costs and benefits (ES) related with the production, distribution and consumption of the food products. There is also need to consider regional climatic patterns and shifts to forecast future value of ES. In further elaborations of this type of study, it would be valuable to reflect on time dimensions. Ecosystem services in agriculture may require longer than one season to provide full levels of service (biological control, for example, or the building of soil fertility through cover crops), and yet can be reduced through one season of pesticide application or misuse of fertilizers. As current agricultural practices suppress vital ES thereby limiting the ability of agriculture to feed increasing human population, there is need to mainstream ES in global agriculture.

2. Business analysis

This family of example focus on products. We showcase two examples that demonstrate application of the framework, i) beef, ii) milk.

2.1. Grass fed versus grain fed beef

It is widely recognized that the current conventional systems of meat production and consumption produce tremendous quantities of meat, at relatively affordable prices, yet generate numerous negative externalities. Several reviews of the meat as a product have assembled information on the benefits and costs of different meat production systems, but none as yet bring such data into one focused “true cost accounting for agriculture” or TEEB-like analysis on the product level. Since an investigation of contrasting meat products includes many key questions and dimensions that are illustrative of a TEEB-like assessment, here we will draw from multiple sources to draw the outlines of the visible and invisible flows in two contrasting beef production systems- grain-fed and grass-fed beef production, in the United States. The United States serves a useful focus, since it is the largest beef producing country globally, and also has the third highest per capita level of beef consumption in the world (FAOSTAT, 2017).

Approach and methodologies

Before making a comparison of grass-fed vs. grain-fed beef it is necessary to first define these two beef types. According to the U.S. Department of Agriculture, beef labelled grass-fed means that the animal, with the exception of milk before weaning, ate grass and forage for its entire life. These animals cannot be fed grain or grain by-products and must have continuous access to a pasture during the growing season. Hay and other roughage may be included in their feed source. Routine mineral and vitamin supplementation may also be included in the feeding regimen" (AMS, 2007). In contrast, grain-fed beef are animals which were deliberately fed grain during their lifetime.

Results

Production and associated waste

Value captured

The livestock sector is a multi-billion dollar a year sector of the global economy, involving complex international trading patterns of both feedstock, live animals and carcasses. Within the US, cattle and calves garner $88.25 billion in farm gate receipts (USDA ERS, 2014). The value of total U.S. beef exports is about 7% of the national total, at $6.302 billion. The grass-fed beef sector is relatively small but growing. Most observers note, however, that it could never replace the productivity of the grain-fed beef sector; substantive changes in the way beef is produced in the United States would entail far smaller levels of per capita beef production.

Provisioning services

The production of grain-fed beef is inherently an inefficient system (Table 3), yet such comparisons are only fair when it is pointed out that cattle, as ruminants, are capable of processing different types of feed that such as grass that neither chickens, poultry or humans can convert.

Table 3 Feed, calories, and protein needed to produce one kg of chicken, pork, and beef.

Feed, calories and protein needed				**to produce**			
kg	calories	protein, g.		kg	calories	protein, g.	
2	6900	200	→	1	1090	259	chicken
3	10350	300	→	1	1180	187	pork
7	24150	700	→	1	1140	226	beef, low range conversion
16	55200	1600	→	1	1140	226	beef, high range conversion

(Source: Carolan, 2011)

Beef production in the United States has grown into the major industry that it is, in large part due to availability of inexpensive feedstuffs, corn and soy. The economic efficiency of grain-fed beef has been sustained in part by United States agricultural policy since 1996 that has supported the overproduction of corn and soy, driving the market price of these crops well below their cost of production (Starmer and Wise, 2007). Thus, the economic model of grain-fed beef fundamentally depends on government subsidy.

Until recently, grass-fed beef was not seen as competitive with grain-fed beef in the United States, due to higher production costs, and a lower quality of carcass. It was thought that cattle finished on forage alone cannot attain the higher quality grades because they have less fat deposition and "marbling" - a characteristic which commonly increases tenderness and sometimes flavour (Dinius and Cross, 1978). However, marbling is a feature of meat that was only became popular as corn became more abundant and cheap as a cattle feed; “visual marbling” was then integrated into the USDA grading system. As a new aesthetic has arisen around a leaner appearance of grass-fed beef and its advertised health benefits, the prices paid for grass-fed beef is substantially higher than grain-fed beef.

Grass-fed beef has a higher cost of production because of several significant factors: grass-fed cattle take longer to bring to market, require additional land, and require high quality pastures to finish cattle. The process of feeding grain in feedlots generally speeds the animal to slaughter weight much faster, reducing time and input costs. However, many comparisons are between feedlots and western U.S. rangeland, which is relatively arid and of low productivity. In a comparison of high potential pastureland versus feedlots in Ontario, Canada, cattle gained 0.2 kg less per day on pasture than in the feedlot. Feedlot cattle finished heavier and with a greater dressing percentage compared with pastured cattle (Jannasch et al., 2012). Grade or visual appearance of the cattle was not different and there was no indication of yellow fat (which is currently considered undesirable even if it reflects higher intake of carotenes in the grass-fed cattles’ diet). The cost of production was \$0.26/kg in the feedlot compared to \$0.10/kg on pasture. Pastured cattle netted \$0.13/kg of gain, or \$68.00 per head, while feedlot cattle broke even. Thus, depending on context, costs of production may vary.

Regulation and Maintenance Services

Grain-fed: The excessive nutrient loading and water contamination from Confined Animal Feedlot operations is known to cause simplification of ecosystems and loss of biodiversity and effective functioning of the ecosystem services.
Grass-fed: Perennial pastures provide continuous soil cover, thus protecting against soil degradation caused by annual cultivation and intensive cropping. Integrating rangeland into farming systems is seen as a key component of sustainable agriculture, using land which otherwise could not be productive (Jannasch et al., 2012). Well-managed intensive grazing approaches have been shown to supported greater numbers of soil microbes classified as heterotrophs, nitrifiers, and denitrifiers - functional groups that support greater soil enzyme activity, enhancing nitrification and nitrogen cycling relative to less intensive management (Patra et al., 2005). Grazing by livestock, under careful management, has been found to contribute to native grassland or prairie diversity (Collins et al., 1998).

Cultural Services

Grass-fed: Grass-fed cattle rearing traditions are a fundamental part of American culture, including country music, cowboy poetry, and many other works of art, literature and crafts focusing on Western culture. The earliest histories of ‘cowboy life” readily recognized the rich Hispanic traditions in cattle ranching, including many tools of the trade (western-style saddles, lariats, chaps, etc.) (Rollins, 1922). While most aspects of this life have changed or evolved dramatically in the last century, interest and pride in ranching culture remains high.

Health Impacts (Nutrition, Lifestyle diseases, Antibiotic resistance, etc.)

In grain-fed beef systems, usually large numbers of animals are raised together in confined spaces, known as industrial farm animal production- or in the case of facilities that discharge to navigable waters, Concentrated Animal Feedlot Operations (CAFOs). This mode of production increases the likelihood for health issues with the potential to affect humans, carried either by the animals or the large quantities of animal waste (PEW, 2008). Animal waste, which harbors a number of pathogens and chemical contaminants is often sprayed on fields as fertilizer, raising the potential for contamination of air, water, and soils, and in some documented cases, the outbreak of disease from microbial contamination (CDC, 2006). A diet rich in grain makes the pH of a beef cattle's stomach abnormally acidic, creating an acidosis and the proliferation of gut bacteria that could otherwise seriously impact the health of the animal. Specially formulated feeds that incorporate antibiotics are used to prevent this outcome. However, the concentrated nature of industrial farm animal production units facilitates the rapid evolution and proliferation of antibiotic-resistant strains of bacteria. A large portion of these end up excreted in urine or manure, presenting the possibility of the transfer of such resistant strains to people (Kumar et al., 2005). Transmission to employees handling such manure is one risk, while a smaller but still viable pathway may be through handling of raw meat purchased in grocery stores or used in restaurants.

Pollution Impacts (Nitrates, Pesticides, Heavy metals, etc.)

Grain-fed beef: The expansion of conventional feed crops for animal grain - usually produced inexpensively through large-scale monoculture maize and soy farming dependent on the intensive application of nitrogen fertilizers and agrochemicals - has its own consequences aside from pollution from the animals themselves, including amplifying air and water pollution, and degradation of soils.

Human sewage is treated to kill pathogens, while the same standard is not applied to animal waste. Animal waste from industrial farm animal facilities is generally stored in lagoons intended to reduce pathogenic elements, but even the best managed are estimated to kill off only 85 to 90% of viruses, and 45 to 50 % of bacteria (Carolan, 2011). A study of Iowa's manure storage structures found that over half leak above the legal limit (Osterberg and Wallinga, 2004). Manure applied to soils may have great value in recycling nutrients (often after a composting process). But the rates of application of manure to agricultural soils in developed countries such as the United States is too often determined by the need to get rid of the manure, rather than to enrich the soil, risking the health of the soil. Moreover, application of untreated animal waste on cropland contributes to excessive nutrient loading, contaminating surface waters, and stimulating bacteria and algal growth and subsequent reductions in dissolved oxygen concentrations in surface waters. The increased costs of requiring more sustainable manure application rates has been estimated for hog operations (between 1 to 5.5 percent increase of operating costs (Starmer and Wise, 2007) but not for grain-fed beef.

Grass-fed beef: Careless management of rangelands can contribute to overgrazing and degradation, particularly in riparian zones. Proposals for rangeland management include intensive rotational grazing in which livestock density is increased and animals are moved frequently through different grazing areas (Briske et al., 2008). One variation is holistic resource management (Savory, 1983), described as a whole farm systems approach that integrates social, ecological, and economic management factors. Holistic resource management of grazing has been associated with enhanced erosion control, which is expected to help maintain stream habitat by reducing sediment loading to waterways.

Rotational grazing reduced phosphorus loads to surface water relative to continuous cattle grazing at similar stocking rates (Haan et al., 2006).

GHG Emissions (CO2, CH4, etc.)
Animals produce greenhouse gases such as methane and carbon dioxide during the digestion, making them an important source of such gases (Table 4). Livestock production is inherently a large source of greenhouse gases. By some estimates, when emissions from land use and land-use change are included in the calculation, the livestock sector accounts for 18 per cent of CO_2 deriving from human-related activities (Steinfeld et al. 2006). It has been calculated that producing 1kg of cheap beef generates as much CO_2 as driving 250km in an average European car or using a 100W bulb continuously for 20 days. Animal agriculture is also responsible for roughly 37 per cent of all human-induced methane emissions, which has a global warming potential 23 times that of carbon dioxide (Steinfeld et al. 2006).

Table 4. US Greenhouse Gas Inventory for Agriculture Emissions (from PEW 2008).

Greenhouse Gas	**Source**	**Thousand Tons**	**Thousand Tons CO2 Equivalent**
Methane (CH_4)	Total	8,459.14	17,770
	Enteric fermentation	5,886.34	12,360
	Manure management	2,167.14	4,550
	Other	406.75	860
Nitrous Oxide (N_2O)	Total	1,333.80	41,350
	Agricultural soil management	1,298.52	40,250
	Manure management	34.17	1,050
	Other	2.20	60

The relative difference in enteric fermentation and manure emission levels per head between grain-fed and grass-fed beef is not well understood. However, there are important production differences, and areas requiring careful contextualization.

Grain-fed beef production: It has been suggested that fertilizer use to support animal agriculture will generate nearly twice as much N_2O as would its use for crops destined for direct human consumption. This is thought because, to quote the author, 'N_2O is first produced when the fertilizer is applied to the cropland for growing the animal feed grain and then is produced a second time when the manure-N, which has been re-concentrated by livestock consuming the feed, is recycled onto the soil or otherwise treated or disposed of' (Davidson, 2009, p. 662).

Grass-fed beef production: If well-managed and promoted by use of increased permanent cover of forage crops, pastured livestock can reduce soil erosion and emissions while sequestering carbon in pasture soils (Teague et al., 2016).

However, grass-fed beef in the Midwestern United States must be fed hay in the winter months when pastures are under snow. Both the production and transportation have costs and greenhouse gas implications. In addition, managed pastures are may require intensive inputs of fertilizers and other amendments. In one study, it was concluded in the Upper Midwest US pasture fed beef from managed grazing systems is more greenhouse gas intensive per kg of meat produced than feedlot finished (Pelletier et al., 2010). However, it should be noted that this calculation is made on a per unit of product basis. Industrial agriculture will always perform better than more agroecological approaches when emissions are expressed on per kg of produce, given the higher levels of productivity of the former in the global scheme of agricultural production. Yet what causes global warming is the total net emission of greenhouse gases per area, regardless of yields. Grain-fed livestock's overall contribution to greenhouse gases is substantial, and stems from the rate that this intensive meat production system has been able to vastly increase in the last decades (Carolan, 2011). Efficiencies in production will not offset increases in total emissions, if livestock production continues to expand in the same way it has through industrial animal feedlot operations.

Social values (Food security, Gender equality, etc.)

While many aspects of beef production fit well into the TEEBAgriFood framework, it is not clear where to place some others that may be more global or "underlying". The overall impact of meat production on global food security is an example of this. Collectively, cattle, pigs and poultry consume roughly half the world's wheat, 90% of the world's corn, 93% of the world's soybeans, and close to all the world's barley not used for brewing and distilling (Tudge, 2010). The discourse on how to address the challenges of feeding a growing world population often focuses on a perceived imperative to simply increase production; yet simple production of calories is not the fundamental issue, as world agricultural production of calories is more than sufficient to feed each person more calories than are needed per day. One of the "underlying" issues of why the global food system is not as yet delivering full food security is due to the amount of land and resources devoted to grain-fed animal-based agriculture. The extent of croplands devoted to producing grain and soy-based animal feed is estimated at about 350 million hectares; in the United States an estimated 50% of all grain produced goes to animal feed. Meat production systems, - particularly on grazing land that is otherwise not suitable for agriculture - and mixed crop-livestock systems are capable of producing both critical calories and proteins. However, the use by intensive livestock production systems of highly productive croplands to produce animal feed imposes a negative force on the world's potential food supply (Foley et al., 2011). The conversion of tropical rain forests in Latin America to produce soy feed for animal agriculture, much of it in other continents including the USA, is equally an issue of social values in conflict.

Processing and distribution (and associated waste)

Processing facilities for beef production should be distinguished between feedlots and slaughterhouses; the latter are used equally by grain-fed and grass-fed beef producers, while feedlots are unique to grain-fed beef production and grass-fed beef is finished on decentralized pasture. While slaughterhouses may differ in practices related to aspects of waste, consumption of power, etc., we will not contrast such differences as they are not as yet aligned with beef production systems. Thus, this section relates only to industrial farm animal feedlot visible and invisible flows.

Value captured
It has been noted that there is a distinct economic disparity between farm communities that include industrial farm animal production units and those that retain locally owned farms where animals are finished on-farm (PEW, 2008). This is attributed, to some extent, to the degree to which money stays in the community. Locally owned and controlled farms tend to buy their supplies and services locally, supporting a variety of local businesses. It has been estimated that every dollar earned on a locally-owned farm generates seven times that value to the local community. In contrast, industrial farm animal facilities have a much lower multiplier effect because their purchases of feed, supplies, and services tend to leave the community, going to suppliers and service providers mandated by the vertical integrators in the meat processing business (PEW, 2008).

Health Impacts (Nutrition, Lifestyle diseases, Antibiotic resistance, etc.)
The greater the possibility for transmission events, between livestock, or between livestock and people, the greater the risk of infectious agents evolving to become more virulent. The co-existence of numerous strains of pathogens also increases such risks. Industrial farm animal production facilities that house large numbers of animals in tight quarters can be a source of new or more infectious agents. Healthy or asymptomatic animals may carry microbial agents that can infect and sicken humans, who may then spread the infection to the community before it is discovered in the animal population (PEW, 2008).

Some estimates of the costs of disease transmission and bacterial resistance from grain-fed beef production have been made in the grey literature, but need more thorough documentation. Sones (2006) estimates the costs associated with the 2001 foot-and-mouth disease outbreak in the UK to be more than US$15 billion; while, the cost of microbial resistance to antibiotics in the US was estimated to be US$4 billion per year in 2010 (Wang, 2010).

In addition to the risks of disease transmission and bacterial resistance from industrial farm animal production units, there may be other serious occupational health impacts. High concentrations of bio-aerosols (airborne particles that are biological in origin) and gases, which are common environmental contaminants in livestock buildings may cause temporary or chronic respiratory irritation among workers and operators (Carolan, 2011). The agitation of liquid manure slurries in industrial farm animal production facilities (an operation carried out to empty out manure pits) can causes levels of hydrogen sulfide to soar to toxic levels that have proven lethal (Mitloehner and Calvo, 2008; PEW, 2008).

Communities near industrial farm animal production units are also subject to air and water emissions, with documented instances of exposure impacting respiratory health or neurological functions (PEW 2008). Residents living near industrial farm animal production facilities often have overall levels of exposure that far exceed that of a 40-hour-a-week worker (Carolan, 2011). Noxious odors can diminish both property values and a household's ability to entertain guests, with impacts upon a sense of community belonging and personal identity (Carolan, 2008).

Social values (Food security, Gender equality, etc.)

Social capital—built through strong cooperation, mutual trust, reciprocity, and shared norms and identity—is considered a foundation of community and an important aspect of the quality of life. Communities with higher levels of social capital tend to have many associated characteristics—lower poverty rates, fewer incidents of violent crime, and stronger democratic institutions. Social capital also emerges as an internal resource in instances of controversy.

Grain-fed: The social fabric of communities undergoes significant change as industrialized farms replace family farms. Capital-intensive agriculture relies more on technology than on labor, there are fewer jobs for local people and more low-paid, itinerant jobs, which often go to migrant laborers who are willing to work for low wages (PEW, 2008). Industrial farm animal production facilities frequently generate controversy and thus threaten community social capital—and as noted in the PEW report (2008) the rifts that develop among community members can be deep and long-standing.

Industrial farm animal facilities are often located near to low-income and non-white communities; poor housing and unprotected sources of drinking water may make such communities even more exposed to the air and water pollutants that may emanate from such facilities. Most such data, however, comes from documentation of swine operations in the United States (Carolan, 2011).

Risks and uncertainties (Resilience, Health, etc.)
Grain-fed beef: Industrial farm animal production facilities have a number of inherent and unique risks that may affect their sustainability. While some of these have been sited properly with regard to local geological features, watersheds, and ecological sensitivity, others are located in fragile ecosystems, such as on flood plains in North Carolina and over shallow drinking water aquifers in the Delmarva Peninsula and northeastern Arkansas. The waste management practices of industrial farm animal production facilities can have substantial adverse effects on air, water, and soils, in ways that are not entirely predictable, for example in times of floods. Another major risk stems from the routine use of specially formulated feeds that incorporate antibiotics, other antimicrobials, and hormones to prevent disease and induce rapid growth. The use of low doses of antibiotics as food additives facilitates the rapid evolution and proliferation of antibiotic-resistant strains of bacteria. The resulting potential for “resistance reservoirs” and interspecies transfer of resistance determinants is a high priority public health concern. Finally, industrial farm animal feedlot operations rely on selective breeding to enhance specific traits such as growth rate, meat texture, and taste. This practice, however, results in a high degree of inbreeding, which reduces biological and genetic diversity and represents a global threat to food security, according to the Food and Agriculture Organization (FAO) of the United Nations (Steinfeld et al., 2006).

Consumption (and associated waste)

Provisioning services
There have been multiple taste comparisons carried out over the last decades comparing grain-fed versus grass-fed beef. In general there have not been clear-cut, consistent taste differences between the two, although grass-fed beef is generally leaner and less

“marbled”. Some chefs note that with less fat, grass-fed beef must be very carefully cooked to avoid overcooking.

Health Impacts (Nutrition, Lifestyle diseases, Antibiotic resistance, etc.)
As noted above, an infectious agent that originates at an industrial farm animal facility may persist through meat processing and contaminate consumer food animal products in homes or restaurants, resulting in potentially serious disease outbreaks far from the facility (PEW, 2008)

Considerable discussion has focused on differences in the nutrition and health impacts of grass-fed versus grain-fed beef. Grass fed beef has been found to be lower in calories, contains healthier omega-3 fats, more precursors for vitamins A and E, higher levels of antioxidants, and up to seven times the beta-carotene (Carolan, 2011). However, many of these claims need contextualisation.

While levels of omega-6 fatty acids are roughly the same in the meat of corn-fed and grass-fed cattle, levels of omega-3 are higher in the fully pastured cow. The ratio of omega-6 to omega-3 in grass-fed beef is roughly 1.56:1, while in grain-fed beef it averages about 7.65:1. A healthy diet is believed to supply these fats in the range of 1:1 to 4:1. Diets in the West, however, tend to have ratios in the range of 11:1 to 30:1, which is hypothesized to be a significant factor in the rising rate of inflammatory disorders in the US (Daley et al., 2010, p5). Ratios in the ‘healthy’ range have been associated with inflammation suppression in patients with rheumatoid arthritis, decreased colorectal cancer cells and reductions in the risk of breast cancer among women.

Other experts minimize such differences by pointing to the fact that meat, in general, is a relatively poor source of “good fats”, with walnuts, edible seeds, algal different plant oils, and fish and eggs being far richer sources. If indeed grass-beef is leaner in fats, it stands to reason that the actual provisioning of “good” fats, along with “bad” fats, is less, or marginal. In terms of calories it is estimated that a 6-ounce grass-fed beef tenderloin may have 92 fewer calories than the same cut from a grain-fed cow; if an American eats a typical amount of beef per year (66.5 pounds or 30.1 kg), switching to lean grass-fed beef will reduce that person’s calorie intake by 17,733 calories a year (Carolan, 2011).

Social values (Food security, Gender equality, etc.)
Costs to taxpayers: As noted earlier under production systems, the availability of inexpensive feed sources results from government subsidies of corn and soy. This inevitably has impacts on consumers, and taxpayers. One early study, admittedly from a journalistic standpoint, found that the real cost of a hamburger (now around $5.00) should be over $200, if all environmental and social costs that taxpayers ultimately will pay for, were accounted for (Dunne, 1994).

A more in-depth study of subsidies in animal agriculture in the United States found that between 1997 and 2005, animal producers were able to purchase corn and soybeans at subsidized prices prices, saving themselves saved approximately US$3.9 billion. This resulted in reducing the operating costs for livestock producers by 5 to 15 per cent. Of this, beef feeders saved US$0.5 billion, or around 5 per cent of their production costs (Starmer and

Wise, 2007). More recent trends have shown that while farm incomes in the US have declined over time and are substantially lower than the 10 year average, government payments for commodities have risen 18% since 2013 (Schnepf, 2017). For livestock in particular, the ratio of livestock output prices to feed costs is at a high and increasing level, indicating that conditions governing grain-fed beef continues to make it highly profitable. The savings provided by government support programs for animal feed, in addition to variable and in some cases lax environmental regulations may offset from 2.4 to 10.7 per cent of an industrial farm animal production system's operating costs.

But the sustainability and efficiency of such production systems can be questioned, versus midsized diversified farming system growing and producing their own feed if all full costs are accounted for, including pollution and health externalities (Carolan, 2011).

Animal welfare:
Increasingly animal welfare is becoming a concern to consumers, who endorse the application of certain basic standards, and recognize societal costs of ignoring the needs of non-human animals. The core standards, as articulated by Grandin (2010) have found resonance among many consumers, in ways that are best answered by pastured as opposed to industrial farm animal production units.

2.2. Palm oil case study

With 56 million tonnes consumption in 2013, palm oil is the world's most popular vegetable oil, widely used in the food, personal care, chemicals and energy sectors (Raynaud et al., 2016). Its consumption is expected to double by 2050. However, the rapid growth of palm oil production in some countries is having serious environmental and social impact costs. These costs include carbon dioxide emissions and air pollution from using fire to clear rainforest and peatland for new plantations, water pollution and harm to health from applying fertilizers and pesticides to crops, methane released from palm oil mill effluent processing facilities, land property rights violations during land expansion and substandard wages and working conditions (http://www.teebweb.org/agriculture-and-food/palm-oil/).

TEEB considers that the root cause of these problems is that the agriculture sector is too often considered in isolation from the society that it feeds, and the environment that supports it. Instead, business and society need to shift their thinking towards a systems-based approach, which recognizes the reality that agriculture, society and the environment, are all connected. Natural and human capital accounting are used to reveal these mutual inter-dependencies and show how they can be reflected in production costs and market prices.

Objectives, and scope of the study
The Palm oil study (Raynaud et al., 2016) is organized in two parts. First, a materiality assessment quantifies and monetizes the main natural capital impacts of palm oil across the 11 leading producer countries. This is followed by a case study that quantifies and monetizes natural capital impacts in more detail in Indonesia, the largest palm oil producer, and also quantifies and monetizes a selection of human capital impacts. A scenario analysis illustrates how natural and human capital accounting can be used in

Indonesia to compare a selection of alternative techniques for growing palm oil, which may lower impact costs.

Approach and methodologies

The materiality assessment studies the visible and invisible natural capital costs linked to the growing, milling and refining stages of palm oil production. It does not include the transportation, food processing and consumption stages. Palm oil and palm kernel oil were included within the scope of the analysis; other by-products such as fatty acid distillate or palm kernel expeller were excluded. The Indonesia case study looks at the visible and invisible natural and human capital costs associated with five specific growing and milling practices.

The analysis combines the use of secondary global life-cycle assessment studies and the application of country-specific valuation coefficients, where data availability and quality is sufficient (see chapter 7 for methods). The first step is to understand the drivers of change by devising appropriate key performance indicators that measure the relationship between palm oil systems, human systems, and ecosystems and biodiversity.

The second step is to understand the consequence of the impact to a specific end-point. An end point is the primary receptor of this impact–society, the environment, or the business itself. Impacts are quantified in biophysical terms (see chapter 7 for methods). Examples of metrics, or valued attributes, are changes in life expectancy or changes in species richness due to the emission of pollutants. Biophysical models are used to estimate these metrics, based on a thorough literature review, and adapted to reflect local conditions (see chapter 7 for methods).

The last step consists of converting the biophysical metrics into monetary terms that reflect the costs and benefits to specific beneficiaries of the change in valued attribute using a valuation coefficient. The output of this step is a valued impact that reflects cost or benefit of specific practices and associated use of inputs and emissions on human health and ecosystems. In this sense, the valuations reflect the damage on different endpoints: the damage to ecosystems and/or the damage to human health.

Results

Materiality assessment across 11 producer countries

The results show that palm oil production in the 11 countries assessed has a natural capital cost of $43 billion per year compared to the commodity's annual value of $50bn. Of this cost, crude palm oil accounts for $37.5bn while palm kernel oil accounts for $5bn. Indonesia has by far the biggest share of the total natural capital cost at 66%, while Malaysia is second at 26%.

Overall, producing one tonne of crude palm oil (CPO) has a natural capital cost of $790 while one tonne of palm kernel oil costs $897. If these costs were added to the weighted average market price of $837 per tonne of palm oil in 2013, the overall cost per tonne would almost double. The natural capital intensity of palm oil production varies widely between countries, which may have implications for siting palm oil operations or sourcing palm oil.

The cost of Indonesia's palm oil industry is driven by the large size of its production and its high natural capital intensity. The total natural capital cost of palm oil production in Indonesia is almost $28billion while its natural capital intensity is $950 per tonne. Land-use change is the biggest single impact in Indonesia, mostly due to GHG emissions from peatland drainage and clearing rainforest.

Palm oil production in Malaysia has much lower natural capital intensity than Indonesia due to the lower cost of land conversion. Only 12% of Malaysia's plantations are planted on peatland and 30% on forested land. Climate change due to GHG emissions from palm oil production, mostly as a result of land-use change, is responsible for 89% of the natural capital cost per tonne of palm oil. The use of fertilizers is responsible for 22% of the cost. Palm oil mill effluent contributes 12% of the cost, largely as a result of the climate change impacts of methane emissions. The impacts of pesticides contributes 3% of the cost per tonne. The upstream impacts from manufacturing fertilizers, pesticides and other raw material inputs are responsible for 3% of the cost.

Indonesian case study results

The case study on Indonesia shows how natural and human capital accounting can be used to assess alternative palm oil production practices to reduce the impact costs of the sector. The case study illustrates this approach by focusing on three practices with the largest natural capital costs and two practices with substantial expected human capital costs. These are land selection and clearing, fertilizer application, palm oil mill effluent remediation, wages and occupational health and safety.

The results show that converting primary forest on peat soil using burning techniques has highest natural capital cost due to GHG emissions and air pollution. Converting grassland and already-disturbed forest using mechanical means has a natural capital benefit as the palm oil plantation sequesters more carbon than the previous land use. The results also show that converting forest or peatland by burning appears less financially costly than mechanical means, but entails a higher natural capital cost.

Over the lifetime of the plantation, using an optimized mix of organic fertilizer containing pruned palm oil fronds, empty fruit bunches and palm oil mill effluent combined with chemical fertilizers has the lowest natural capital cost at $1,640 per tonne palm oil, compared to $3,080 per tonne palm oil where chemical fertilizer use is not optimized. Optimization also has the lowest financial cost due to the lower quantity of fertilizer needed.

Installing methane capture equipment on palm oil mill effluent treatment processes to generate energy is also identified as best practice to reduce natural capital costs. It also results in a 17% financial cost saving due to the sale of Certified Emissions Reduction credits. The results also show that underpayment and occupational health impacts have a total human capital cost of $592 per full-time employee, or $34 per tonne of palm oil and $53 per tonne of palm kernel oil. This is comparable in size to the combined natural capital impact of fertilizer manufacturing and pesticide application.

If plantation owners paid a living wage to casual workers, the human capital cost of underpayment would be reduced to zero, while plantations remain profitable with margins reducing from 28% to 24%. The human capital return on investment for this intervention is 11%, which means that the decrease in human capital costs is higher than the decrease in the net cash flow of the plantation.

Wearing personal protective equipment reduces instances of pesticide poisoning, cutting the human capital cost of occupational health by 6%. The human capital return on investment for this intervention is 130%. As these results do not take into account positive effects of improved labor conditions on net cash flow or projected financial losses due to reputational and other risks, they should not be considered as a complete financial business case analysis for these interventions, but as a means to include human capital costs in business decision making.

Study recommendations
Businesses can act to improve the sustainability of palm oil production through implementing more sustainable production practices such as increasing yield and conversion rates and optimizing the quantity and quality of inputs used, and by relocating to areas less vulnerable to social and environmental impacts. These elements should be considered together to identify trade-offs and ensure that the overall natural and human capital impact of the system is minimized.

This can be done for example via voluntary commitments, environmental or social taxation or environmental and social regulation. These measures should however not increase food prices for vulnerable shares of the population. Efforts to improve palm oil production to reduce human and natural capital costs should also be made through policies.

Further research
Palm oil plantations have significant social and natural components that were not explored in this study. Palm oil landscapes provide a number of important ecosystem services such as soil erosion control, biodiversity, water regulation, other agricultural production that support subsistence livelihood. Moreover, it covered only the production side of palm oil and did not account for any costs or positive benefits associated in distribution and consumption side of the equation, as well as food security aspects, access, distribution, markets, agribusiness, supply chain, waste reduction that are all important parts of food systems. These are important areas for future research. Relatedly, other qualitative social impacts such as food security, the role of gender in agroforestry systems, cultural values, labour conditions, land dispossession etc. should also be examined further.

3. Dietary comparison

This family of examples focuses on two examples that compares diets i) diet study from France, ii) Ten diet scenarios and carrying capacity of agricultural land in US.

3.1. Welfare and sustainability effects of diets in France

This study conducted in France is an ex-ante assessment of dietary recommendations in multiple sustainability dimensions such as taste cost, welfare effect, deaths avoided, reduction in greenhouse gas emissions and acidification (Irz et al., 2016).

Approach and methodologies
A model of rational behaviour is developed, building on microeconomic theory of the consumer under rationing (dietary constraints), with the goal of identifying diets compatible with both dietary recommendations and consumer preferences. This model is calibrated using KANTAR Worldpanel data from a panel of 19,000 representative consumers of the French population. Food consumption is aggregated into 22 food group categories. The nutrient content of these aggregates is calculated by combining the food consumption database of the French dietary intake survey INCA2 and average adult intakes of the components of each aggregate drawn from INCA2 (Irz et al., 2015). This model is new in the sense that it includes taste cost by taking into account consumer preferences in the model. Six different sustainable diet recommendations, expressed via dietary constraints are considered. The dietary constraints assessed are between current values? and a 5% relative variation in the level of constraint of its baseline level. The constraints derive from nutrient based (salt intake, saturated fat acids SFA) and food-based (fruit and vegetables, meat), health (added sugar) and environmental (CO_2 emissions) recommendations. For these restrictions, the percentage change in consumption of the 22 food groups was calculated. To deal with health impacts, an epidemiological model to estimate the effects in terms of chronic disease prevalence and mortality was applied. The effect on environmental indicators was estimated as well, making use of a Life Cycle Analysis (LCA)- based approach and a top-down input-output approach. These estimates include each stage of the production, transformation, packaging, distribution, use, and end-of-life products.

Results
Irz et al. (2016) did the analysis for three different income groups. The main results did not differ between these income groups.

Based on the restrictions, the percentage change in consumption of the 22 food groups was calculated. Due to the complementarity and substitutability among the food products captured in the model, a decrease in meat consumption of 8 grams (5%) results in relatively important changes in consumption of starchy foods (-2.2%) and dairy products (+3.4%). Also within subgroups substitutions occur, for example more fish (+7.5%) and less eggs (-3.3%). The restriction on only red meat results in smaller adjustments in food consumption.
The results of changes in the other constraint are varied. For example, for CO_2, large changes in food consumption occur as products with a high CO_2 impact will be replaced with products with a low impact. The overall result reveals that restrictions may result in large changes in consumption patterns. This makes it relevant to consider adjustments in the whole diet.

The analysis of the shadow prices shows equal or higher prices for almost all products, with as exception the fruits and vegetables (F&V) constraint. For the nutrient based constraints and the CO_2 constraints the shadow prices are relatively high, suggesting that substitution required satisfying the constraint is difficult. This makes intuitive sense. For example, a high shadow price of oil, margarine and condiments makes sense with a restriction on saturated fat acids (SFA).

In the next step, the short run welfare costs of satisfying these constraints is estimated. The red meat constraint is associated with the lowest costs, while the CO_2 constraint is associated with the highest welfare costs. Because of a recommendation and its promotion, the consumer modifies his arbitrage between short-term rewards from food consumption and long-term reward from improved health. In the short term, this adjustment has a welfare cost, which we measure and identify as 'taste cost'. The long-term health impact is measured at the aggregate level of the population in terms of deaths avoided (DA). Compared with the total food budget, these costs are small (but percentage change is also small, only 5%).

Effects on nutrition and dietary indicators
The constraint result in desirable and undesirable health effects is based on the DIETRON nutritional model indicators. The environmental effect indicators used in this analysis were greenhouse gas emissions (CO_2 equivalents) and acidification (SO2 equivalents). With the exception of SFA and added-sugar, all recommendations lead to a decrease in the environmental impact of the diet.

To assess simultaneously the economic, health and environmental effects of the recommendations, the overall benefits and cost-effectiveness of the recommendations are calculated. The consumer costs varies from 45,000 Euro for the meat restriction to 412,000 Euro for the CO_2 restriction. The health effects are estimated with the DIETRON model as the avoided deaths within the whole population due to the reduced incidence of chronic heart diseases (CHD), strokes and then different types of cancer. The constraint on salt intake results in the highest number of deaths avoided (2852), the constraint on fruit and vegetables, SFA and CO_2 also results in more than 2000 deaths avoided. The reduction in meat consumption results in less than 250 annually saved lives. Also the effects on CO_2 reduction and on SO_2 reduction are calculated, based on LCA coefficients. The restriction on CO_2 has the biggest effect, followed by the restriction on F&V. Calculating the partial cost-effectiveness per indicator gives as result that most restrictions are very cost-ineffective restrictions.
A more complete cost-effectiveness analysis, in which the benefits and costs of the measures can be considered jointly is necessary. Valuing the positive effects with the value of carbon (32 Euro/ton), the value of an avoided death (240,000 Euro), justifies spending considerable amounts of resources to promote the recommendations targeting F&V, Salt, SFA, added-sugar and red meat. With higher values for carbon (185 Euro/ton) and a value for an avoided death closer to the value of a statistical life (1 million Euro), the benefits of targeting GHGs and consumption of all meat appear to be cost-effective as well. This way of reasoning makes it possible to establish a ranking of the recommendations to be promoted. To test the robustness of the results, the analysis is done again with other LCA coefficients. This does not have an impact on the overall results.

The model developed in this study weighs the taste cost (or short term welfare costs) incurred by consumers against the health and environmental benefits induced by their adoption. Based on the analysis, the diet restrictions can be ranked in different ways. Based on the shadow price analysis, it can be concluded that a tax on the health based restrictions and on CO_2 are unlikely to be very effective, while a tax on all meat of red meat consumption would be relevant. Based on the complete cost-benefit analysis the authors conclude that; i) measures focused on intakes of F&V, SFA, sodium, and to some extent, added-sugar, provided that they lead to at least a 5% change in the consumption of the targeted food or nutrients, would be a valuable investment: ii) informational measures to promote a reduction of red meat or all meat consumption would be valuable investment only for high values of CO_2 market prices and that this result is sensitive to the value of a death avoided. A last conclusion: the values of health benefits induced by dietary recommendations are often much greater than those of environmental benefits (except in the case of a very high CO_2 price).

3.2. Ten diet scenarios and carrying capacity of agricultural land in US

Assessment of human carrying capacity (persons fed per unit land area) is essential to fully understand current and potential productivity of a land base. Estimates of carrying capacity represent the productive output of many crops grown across a heterogeneous land base in a single indicator, the number of people fed. Therefore, the purpose of the study highlighted here was to compare the per capita land requirements and potential carrying capacity of the land base of the continental United States (U.S.) under a diverse set of dietary scenarios (Peters et al., 2016).

Objectives and scope

The study focuses on an analysis of how dietary change might impact land use and carrying capacity. It uses a “Foodprint model” to estimate land requirements for complete diets, accounting for three important interactions: the multiuse nature of certain grain and oilseed crops, the suitability of multiple land types to grazing, and the relationship between dairy production and beef production. Finally, it explores how assumptions about the partitioning of agricultural land and the suitability of cropland for cultivated crops influences estimates of carrying capacity.

Approach and methodologies

A biophysical simulation model (the U.S. Foodprint Model based on Peters et al., 2007) was designed to calculate the per capita land requirements of human diets and the potential population fed by the agricultural land base of the continental United States. To do this, three sets of calculations were performed (Figure 4).

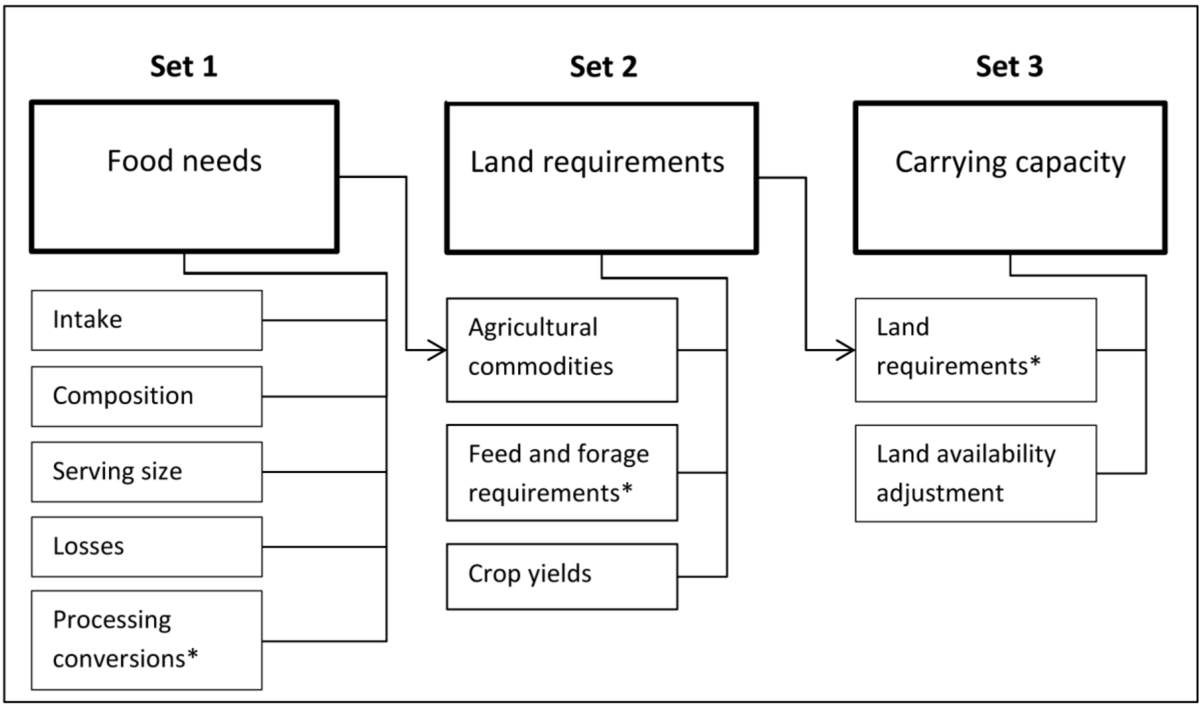

Figure 4. Flow diagram of the sets of calculations performed in the U.S. Foodprint model. Adapted from Peters et al., (2016).

Set 1 calculations estimates the annual, per capita food needs of the population based on daily food intake, the individual food commodities that comprise each food group, the weight of a serving of food, losses and waste that occur across the food system, and the conversion of raw agricultural commodities into processed food commodities. The second set of calculations estimated the individual land area required for each agricultural commodity in the diet based on yield data for each component crop and the feed requirements of all livestock. The third set of calculations estimated the potential carrying capacity of U.S. agricultural land, accounting for the aggregate land requirements of a complete diet, the area of land available, and the suitability of land for different agricultural uses.

Scenarios of food consumption

Ten distinct diet scenarios were analyzed in this study [BAS (baseline), POS (positive control), OMNI 100 (100% healthy omnivorous), OMNI 80 (80% healthy omnivorous), OMNI 60 (60% healthy omnivorous), OMNI 40 (40% healthy omnivorous), OMNI 20 (20% healthy omnivorous), OVO (ovolacto vegetarian), LAC (lacto vegetarian), and VEG (vegan)]. The reference diet (Baseline) reflects contemporary food consumption patterns based on loss-adjusted food availability data from 2006–2008 (USDA Economic Research Service, 2010). The first isocaloric diet is identical to the baseline for the major food groups, but contains fewer discretionary calories in the form of added fats and sweeteners to prevent energy intake from exceeding caloric needs (Positive control, POS). The scenarios focused solely on differences in food consumption patterns; parameters for food losses and waste, processing conversions, livestock feed needs, crop yields, land availability, and land suitability were held constant.

Partitioning of agricultural land
Productive agricultural land was divided into two pools, cropland and grazing land.

Model calculations
The principal calculations in the model were: food needs, land requirements, and carrying capacity. Diet scenarios were structured based on intake of food groups. The first set of calculations performed on the U.S.Foodprint model translated each of the diet scenarios into estimates of the mass of primary food commodities needed to supply each diet, as well as the equivalent quantities of agricultural commodities from which the foods are derived. The second set of calculations determined the land requirements for individual foods and for complete diets. Third set of calculations were of potential carrying capacity that was calculated based on per capita land requirements, the areas of cultivated cropland, perennial cropland, and grazing land available in the U.S, and the suitable uses for each pool of land.

Results
Land requirements of diet
The baseline scenario had the highest total land use requirement, 1.08 ha person^{-1} year^{-1}, followed closely by the positive control, 1.03 ha person^{-1} year^{-1}. Land requirements decreased steadily across the five healthy omnivorous diets, from 0.93 to 0.25 ha person^{-1} year^{-1}, and the total land requirements for the three vegetarian diets were all similarly low, 0.13 to 0.14 ha person^{-1} year^{-1}.

Utilization of available land
The aggregate area available for food production was estimated to be 95 million ha cultivated cropland, 134 million ha total cropland, and 299 million ha grazing land. Not all diets equally exploited each pool of land. The five diets containing the largest quantities of meat (baseline, positive control, 100% health omnivorous, 80% healthy omnivorous, and 60% healthy omnivorous) used the entire available area, both cropland and grazing land. The five diets containing the least meat (or no meat) used the maximum allowable area of cultivated cropland and varied widely in their use of the remaining agricultural land. The 40% healthy omnivorous diet and the 20% healthy omnivorous diet used some of the available grazing land (214 and 75 million ha, respectively) and most of the cropland restricted to perennial forages (35 and 24 million ha, respectively). The ovolacto- and lacto-vegetarian diets used about half of the cropland restricted to perennial forages, while the vegan diet used none of the cropland so restricted. None of the vegetarian diets used any grazing land (dairy rations were modeled with cows fed only harvested feeds and forages).

Potential carrying capacity
All dietary changes increased estimated carrying capacity relative to the baseline. Reducing excess discretionary calories (positive control diet) resulted in a small increase in potential to feed people, 19 million persons (about 5% of the 2010 U.S. population). Reducing meat in the diet, as shown by the five healthy omnivorous diet scenarios, further increased carrying capacity relative to the baseline: 63 to 367 million persons (16% to 91% of the 2010 U.S. population). Switching to an entirely vegetarian diet also increased carrying capacity relative to the baseline, though ovolacto- and lacto-vegetarian diets had higher carrying capacities

than the vegan diet. Indeed, the carrying capacity of the vegan diet fell between the 60% omnivore and 40% omnivore diet.

Diet composition greatly influences overall land footprint. Five of the diets operate under conditions in which the total footprint of agriculture does not change, even though carrying capacity differs widely. However, the 40% healthy omnivorous, the 20% healthy omnivorous, and the three vegetarian diets all have aggregate footprints smaller that the area currently used in the U.S. This finding is significant in light of recent calls to contain the footprint of agriculture (Godfray et al., 2010; Foley et al., 2011). Provision of food, while essential, is not the only important ecosystem service provided by land. Some of these services, such as carbon capture, may be compatible with grazing (Havstad et al., 2007). Other services, such as wildlife habitat (Knight and Johnson–Nistler, 2004), may be impinged where domesticated species compete for biomass with wild ruminants and ungulates. Finally, the use of perennial cropland for grazing or hay production could conceivably compete with bioenergy production where biomass energy or draft animals are possible alternatives to fossil fuels (Concostrina-Zubiri et al., 2016).

The estimates of carrying capacity for each scenario suggest that dietary choices can greatly influence the ability of agriculture to meet human food needs, while simultaneously generating ecosystem services. Reducing meat in the diet clearly resulted in increased carrying capacity, as evidenced by the fact that carrying capacity increased across the five healthy omnivorous diets as the amount of meat consumed decreased. Likewise, the ovolacto- and lacto-vegetarian diets had the highest estimates of carrying capacity overall. However, the influence of dietary changes are not always obvious, as shown by the fact that the relative position of the vegan diet varied depending on starting assumptions regarding the proportion of cropland available for cultivation. Similarly, removing 700 kcal person^{-1} day^{-1} from the baseline diet caused just a small jump in carrying capacity as shown in the positive control diet.

Study Recommendations

The findings of this study support the idea that dietary change towards plant-based diets has significant potential to reduce the agricultural land requirements of U.S. consumers and increase the carrying capacity of U.S. agricultural. The differences between the scenarios suggest that the dietary changes could free up capacity to feed hundreds of millions of people around the globe. To meet global food needs in 2050, a potential of this magnitude is significant. Of perhaps greater relevance though, similar studies in countries around the world could help policy makers strategize on diet interventions to attain food security.

Further work

Future work is needed to determine the potential for dietary change to influence land requirements and carrying capacity around the world. Diet composition matters.

4. Policy evaluation

Here we present two examples, i) Pesticides tax in Thailand, ii) Sloping Land Conversion Program in China

4.1. Pesticide tax in Thailand

Over the period from 1987 to 2010 agricultural pesticide use in Thailand increased from 1 kg/ha to 6 kg/ha, while the pesticide productivity (gross output per unit of pesticide use) decreased from 400 USD/kg to 100 USD/kg. The increase in pesticide use has been attributed to subsidized farm credit programs amongst other causes; Thai policies have been supporting the use of pesticides from 1950 till the 1990s (Praneetvatakul et al., 2013). Besides the negative effect of pesticides on the environment, farmers', farm workers' and consumers' health are also exposed to risks. These risks are considered higher in lower income countries, because of incorrect use of pesticides and the use of more hazardous pesticides. In addition, many lower income countries do not have the institutional capacity to manage these risks.

Policy debates in Thailand have focussed on banning certain highly hazardous pesticides. However, data on costs and benefits is missing to support such decisions and therefore the debates are often focussed on ideology and commercial interests. A study was undertaken Praneetvatakul et al., 2013 to provide a quantitative analysis of the external costs of pesticides, to help policy makers understand who was bearing these costs, and where policy might intervene to reduce or eliminate these.

Objectives and scope

In this quantitative analysis on the externalities of pesticide use in Thailand external costs were defined as follows: "pesticides can harm organisms other than pests, such as beneficial insects and soil organisms, aquatic life and humans. This potential harm brings costs to society and the environment in the form of pest resurgence and pesticide resistance, chronic and acute health problems for people taking in pesticide residues, the pollution of water resources—including drinking water, and also costs in terms of having to monitor food systems". Two different approaches are used to calculate external costs. The outcomes are used to illustrate how this kind of data can be used to develop policies related to pesticides. Here we apply the TEEBAgriFood framework to the variables used in the study, to demonstrate how policy makers might use such studies to make these external costs visible, and thus help to defining economic policies (e.g. taxes or incentives) for pesticide use.

Approach and methodologies

Two approaches have been used in this study to quantify external costs:

1. **Pesticide Environmental Accounting (PEA)** based on methodology developed by Leach and Mumford (2008, 2011). A set of base values for external costs (EC) associated with the application of one kg of active pesticide ingredients is calculated. The PEA tool adjusts the base values for economic costs to differences in the relative toxicity of pesticides using the Environmental Impact Quotient (EIQ) tool developed by Kovach et al. (1992). The eight categories are:
    - Farm worker health: the effect on pickers and applicators
    - Consumer health: the effect of residues on groundwater and food consumption
    - Environment: the effect on aquatic life, birds, bees and beneficial insects

   External costs of pesticide use in Thailand were calculated for 1997 and 2010, divided over the 8 categories. The external costs as published by Leach and Mumford (2008) were projected to 2010 values by the authors and adjusted to the level of income of

Thailand and average share of agricultural employment, for two production systems: rice and intensive horticulture.

2. **Actual cost studies:** Data on costs related to pesticide use are collected from government agencies, using the methodology of Jungbluth (1996), the first to do an actual cost study, is followed. Though, the authors provide this caveat: "However, neither actual cost studies, nor willingness to pay studies, are suitable for quantifying the external costs of a particular active ingredient or a particular production system, which is what policy makers need to know when considering intervention, such as banning a chemical for use in agriculture".

Results

1. Pesticide Environmental Accounting (PEA)

The external costs calculated using the PEA method are shown in Table 5. The results demonstrate the effect on consumers is only 11%, while the effect on farm workers is 83% in 2010. External costs of pesticides use can be mainly explained by an increase in quantity of pesticide use (10.6%). The trend in the average pesticide toxicity from 1997 to 2010 was calculated and decreased annually with 1.6%. This trend was calculated by weighting the Environmental Impact Quotient (EIQ) by the quantity of pesticides used, and then using a least square regression.

Table 5. External costs of pesticide use in Thailand as based on the PEA method (USD/ha in constant 2010 prices) (Praneetvatakul et al., 2013).

EIQ category	1997	2010
Total farm worker health	5.56	22.42
Applicator effects	3.43	13.3
Picker effects	2.13	9.13
Total consumer health	0.55	2.91
Consumer effects	0.40	2.17
Ground water	0.14	0.74
Total environment	0.35	1.8
Aquatic effects	0.22	1.13
Bird effects	0.05	0.23
Bee effects	0.04	0.19
Beneficial insect effects	0.05	0.25
Total	6.46	27.13

Table 6 gives an overview of the external costs per production system for rice production and intensive horticulture. The pesticide input and external costs are higher for the horticulture system compared to the rice system. In addition, pesticide productivity (gross output per unit of pesticide use) is higher in the rice system as well as the external cost/gross output. However, per dollar of pesticide the rice production system results on average in 0.66 USD of external costs, while the horticulture system results in 0.23 USD of external costs. According to the authors: "Internalizing these external costs would hence require a rise in the average retail price of pesticides of between 11 and 32%, depending on the price and toxicity of pesticides. This would increase the average variable cost in rice cultivation by about 6%".

Table 6 Comparison of pesticide use and external costs for rice cultivation and intensive horticulture (at constant 2010 prices) (Source: Praneetvatakul et al., 2013)

Variable	Rice cultivation	Intensive horticulture
Pesticide application rate (kg/ha)a	1.3	13.3
Herbicides (%)	49	11.4
Insecticides (%)	45.6	29.4
Fungicides (%)	4.6	54.4
Pesticide expenditure (USD/ha)	60.01	962.64
External cost (USD/ha)	19.29	105.75
Gross output (USD/ha)	465.26	12,010.06
External cost/gross output	0.041	0.009

2. Actual cost studies

Table 7 shows the estimates of the actual cost approach, including information on source and information. In this case, actual cost estimates permits the valuation of specific policy measures, such government budgets for pest outbreaks, pesticide research and food safety standards.

Table 7 Estimates of actual costs related to pesticides in Thailand in 2010 (million USD) (Source: Praneetvatakul et al., 2013)

Cost category	Million USD	Source and calculation
1. Health costs due to acute pesticide poisoning		
a) Registered cases	0.13	Registered cases: 8546 cases of pesticide poisoning recorded in the National Health Insurance Database in 2010 (Biothai, 2011). Average cost per case was 494.12 Baht (Jungbluth, 1996).
b) All cases	2.79	All cases: Cost transfer approach (Jungbluth, 1996; Whangthongtham, 1990). Number of poisoning cases/kg of pesticide use × total amount of pesticide use in 2010
2. Pesticide contamination of:		15% of fruit and vegetables exceeded maximum residue limits in 2006/2007 (Athisook et al., 2006). Multiplied by fruit and vegetable output valued at farm gate prices (Anonymous, 2008)
a) fruit	155.25	
b) vegetables	72.88	
3. Costs related to the BPH outbreak in 2010	15.77	Data obtained from summary of a government cabinet meeting on 1 February 2011a
4. Budget for research related to pesticide issues	38.85	Budget for pesticide research at the Entomology Division. Estimated at 40% of the total budget of the DOA in 2010 (MoAC, 2011).

5. Budget for R&D on agricultural production inputs (related to pesticides)	0.48	Budget in 2010 at Agricultural Production Science Research and Development Office (DoA, 2011) (projects 4–7 related to pesticides)
6. Budget of the Q-GAP program	60.34	Annual Report, Department of Agricultural Extension, 2009–2010 (DoAE, 2010)
7. Food safety standards	5.89	Food safety standards set by the National Bureau of Agricultural Commodity and Food Standards (ACFS). Summary of 2010 Budget Report (ACFS, 2010)
Total	352.7	

Study recommendations
Until the late 1990s policies in Thailand were supporting the use of pesticides, as in other lower income countries in East and Southeast Asia, in order to stimulate agricultural production. There is a need for a change from an institution framework that promotes pesticides to an institutional framework that takes into account the risks and is adjusted to the true costs and benefits. The authors of the study come up with the following recommendations:

"Pesticide externalities exist because pesticides create costs for society and the environment that are not transmitted to the farmers who choose to apply them. From an economic point of view, efficiency could be improved by internalizing these external costs into the price that farmers pay for pesticides, for instance through an environmental tax on pesticides. It is most practical to levy such tax on importers and producers of pesticides, which are few in number relative to retailers and farmers. Yet an environmental tax on pesticides is not enough to address the problem. Research from various countries shows that the demand for agricultural pesticides is typically inelastic and that a tax would only have a weak effect on pesticide demand, though generating considerable government revenues (Falconer and Hodge, 2000). Based on the results, an environmental tax would raise pesticide prices by 11-32%".

In addition to the environmental tax on pesticides the study recommends the introduction of measures supporting non-chemical pest management methods, focusing on on-farm practices, such as IPM methods, FFS, programs on awareness raising on pesticides, farmer training and education. As the actual costs method shows, priority of the government is on consumer safety, although the PEA method shows 83% of the external costs of pesticide use accrue to farmworkers and only 11% to consumers.

Since the analysis shows that the greatest costs are currently being incurred on the farm, amongst pesticide appliers and pickers, it can be questioned if a pesticide tax will actually address these costs. It is unlikely that the tax will be applied to dealing with farmworker health (or funding research into production methods that use less pesticides) unless it is explicitly formulated to do so. To be effective, policies and social institutions addressing the true costs of agricultural inputs are needed that effectively address areas of greatest costs and benefits along the food system; the TEEBAgriFood framework has utility in identifying these areas.

Further research
The PEA method that can be used for formulating effective pesticide policies. However, some limitations were addressed by the authors:
- The method captures toxicity, but not risk exposure
- The method does not clearly differentiate between highly toxic and less toxic pesticides
- The method does not capture external effect of pesticides of which no immediate monetary payment were made

Finally, in the methods used, benefits of pesticides are not taking into account (e.g. yield increase). A tool should be developed taking into account both the benefits and costs of pesticide use, but also comparing the costs and benefits of alternative, more ecological practices.

4.2. The China Ecosystem Assessment: Sloping Land Conversion Program
China has become the second largest economy in the world since the "reform and opening up," begun in the 1970s. However, its rapid economic development has resulted in high levels of environmental degradation such as massive deforestation and erosion has contributed to severe flooding along the Yangtze River. This resulted in loss of lives and rendered 13.2 million homeless, which caused about U.S. $36 billion in property damage (Ye et al., 1998). This environmental and economic crisis led to the creation of the world's largest government-financed payment for ecosystem services (PES) programs: the Natural Forest Conservation Program (NFCP) and the Sloping Land Conversion Program (SLCP) (Zhang et al., 2000; Liu et al., 2008). By 2009, the cumulative total investment through the NFCP and SLCP exceeded U.S. $50 billion and directly involved more than 120 million farmers in 32 million households in the SLCP alone (Ouyang et al., 2016). These programs aim to reduce natural disaster risk by restoring forest and grassland, while improving livelihood options and alleviating poverty. China has never evaluated such conservation programs at the national level. Therefore, in 2012, China's Ministry of Environmental Protection and Chinese Academy of Sciences launched a national ecosystem assessment to quantify ecosystem status and trends, and ecosystem service provision between 2000 and 2010.

Objectives and scope
The China ecosystem assessment (CEA) was designed to evaluate government financed PES schemes and aimed to address;
1. how ecosystem services are changing,
2. where important services originate, and
3. what should be protected and restored to increase ecosystem services.

PES works by compensating those who provide ecosystem services (ES), e.g. specific land uses (e.g. afforestation is frequently promoted, particularly in developing countries) (Engel et al., 2008), while payments come from ES users, government revenues, or third-party donors. PES have been increasingly used to reduce the negative environmental effects of farming activities (Landell-Mills and Porras, 2002; Scherr et al., 2003). PES schemes are institutionally simpler and more cost-effective than traditional conservation programs (Pagiola et al., 2005; Pattanayak et al., 2010). One of the foremost objectives of PES schemes (NFCP and SLCP) was improvement of livelihoods and poverty alleviation of rural population (Uchida et al., 2009).

The study showcased here reports on the results of the first CEA, which covered all of mainland China from 2000 to 2010 (Ouyang et al., 2016). The CEA is the first assessment of various ecosystems and ecosystem services and hence relevant in part to the TEEB-like study.

Approach and methodologies

The assessment used data from a variety of sources, including >20,000 multisource satellite images, recorded biophysical data [such as soil, digital elevation models (DEMs), hydrology, and meteorology], >100,000 field surveys; historical records of biodiversity; and special assessments from several government ministries (e.g., surveys of desertification, soil erosion). All lands were classified using a newly established ecosystem classification system for China. The CEA collected data on food production by crop converted to kilocalories (kcal) and modelled the level of provision for six other important ecosystem services: carbon sequestration (metric tons), soil retention (metric tons), sandstorm prevention (metric tons), water retention (metric tons), flood mitigation (m^3), and habitat provision for biodiversity (total habitat area of endemic, endangered, and nationally protected species per county)] using InVEST (a suite of free, open-source software models designed for Integrated Valuation of Ecosystem Services and Tradeoffs) (Kareiva et al., 2011; Sharp et al., 2015) and other bio-physical models.

Results

All ecosystem services evaluated increased between 2000 and 2010, with the exception of habitat provision for biodiversity (Figure 9). Food production had the largest increase (38.5%), followed by carbon sequestration (23.4%), soil retention (12.9%), flood mitigation (12.7%), sandstorm prevention (6.1%), and water retention (3.6%), whereas habitat provision decreased slightly (–3.1%).

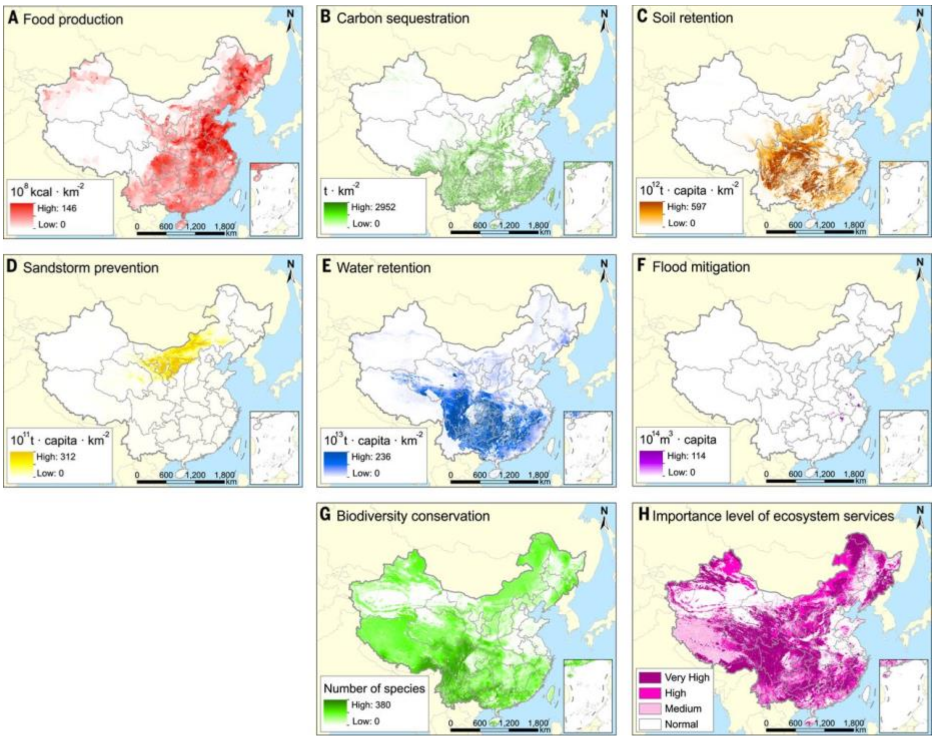

Figure 5 Spatial pattern of ecosystem service provision in China in 2010.
A. Food production (10^8kcal km^{-2}, 0-146); B. Carbon sequestration (tkm^{-2}, 0-2952); C. Soil retention (10^4tkm^{-2}, 0-133); D. Sand storm prevention (10^2 t km^{-2}, 0-684); E. Water retention (10^4 t km^{-2}, 0-216); F. Flood mitigation (10^6 m^3, 0-843); G. Provision of habitat for biodiversity (total species richness of endemic, endangered, and nationally protected species) (0-380 species county^{-1}); H. Index of relative importance of regulating services.

The changes in the provision of ecosystem services from 2000 to 2010 are the result of natural capital investment policies, changes in biophysical factors, and socioeconomic development. Overall, our results suggest that China's national conservation policies contributed significantly to the increases in four key ecosystem services: carbon sequestration, soil retention, sand fixation, and water retention.

Table 8 Impact of different policies on four ecosystem services. X indicates positive impacts.

SLCP_Forest	X	X		
SLCP_Grassland			X	
NFCP	X	X		X

Study recommendations
Although continuing these programs provides good opportunities for restoring and conserving ecosystem services, there are also many challenges and unexpected outcomes. However, the experiences and lessons learned from the policies on payments for ecosystem services in the past several years have laid a good foundation for their continuation and expansion. It is expected that systematic planning, diversified funding, effective compensation, interdisciplinary research and comprehensive monitoring will make future endeavours more successful (Liu et al., 2008).

Although the CEA documented improvement in ecosystem services, there remain serious environmental challenges, including deteriorating air and water quality, increasing greenhouse gas emissions, and impacts on mental and physical health. There is also need to directly link ecosystem services to human well-being, such as economic measures of value and direct measures of impact on health, livelihoods, happiness, or other aspects of well-being. Better understanding of human behavioral responses to changes in policy or market conditions could improve policy effectiveness.

Further research
SLCP and NFCP significantly reduces agricultural production activities and the consumption of self-produced products, which could reduce food security at the local level (Cao et al., 2010; Liu and Henningsen, 2014). However, the negative impact of the SLCP on agricultural production at the national level is rather small (around -2.8%). The program has reduced

poverty in the Yellow River basin by increasing the income of participating households through the compensation payment and shifting the labor force from farm activities to non-farm work. However, in the Yangtze River basin, the SLCP does not significantly increase non-farm work and total consumption, which could be caused by lower off-farm work opportunities in the Yangtze River basin than in the Yellow River basin. Thus, measures that facilitate the households' access to the non-farm labor market–including employment training and information services–could strengthen the positive socio-economic effects of the SLCP, particularly in the Yangtze River basin.

5. National accounts

Here we highlight two examples, i) Agriculture development in Senegal, ii) Australian Environmental Economic Accounts in agriculture.

5.1. Agriculture development in Senegal

Senegal has high rates of poverty, food and nutrition insecurity and environmental issues despite it being one of the most promising countries in the West African region. A global report has identified agriculture development to address some of these problems (IAASTD, 2009). The majority of the population living in extreme poverty live in rural areas (IFAD, 2010) and depends on agriculture and livestock directly or indirectly for their livelihoods (IFAD 2012). For sustainable environmental and economic development, Senegal has developed a vision - the Plan Sénégal Emergent (PSE). It envisions transformation of the structure of the economy. The first policy lever for this transformation is the development of agriculture, fisheries and the food industry, which contributes to reaching multiple goals, including food and nutrition security, rebalancing trade, and revitalizing the rural economy (Gouvernement de la République du Sénégal 2014).

A synthesis of a study conducted by the Millennium Institute is presented here, as it addresses many of the issues that a TEEB-type study will need to do on the level of national accounts (Millennium Institute, 2015). This study aims to provide analysis of the socio-economic and environmental impacts of the agriculture development through provision of World Bank's loan to the Government of Senegal. It provides scenarios for the social, economic and environmental development based on investment at national scale, and considers the impacts of alternative investment in small-scale ecological and knowledge-intensive approaches, as opposed to high external-input, agricultural systems.

Objectives and scope

For the 'sustainable and inclusive agribusiness development project', the World Bank proposed to implement a credit of USD 80 million from the International Development Association and a grant in the amount of USD 6 million from the Global Environment Facility Trust Fund in six years (2014-2020). The main objective is to develop inclusive commercial agriculture and sustainable land management in project areas thorough investments in infrastructure (irrigation in particular, 70% of assistance), technical assistance to key public institutions (rural communities in particular, 20%), and support to the private sector (including smallholders, 10%) all along the agribusiness value chains (World Bank, 2013). The project aims to build irrigation infrastructure for 10,000 ha within the St Louis and Louga regions, the Ngalam Valley and Lac de Guiers. The assessment of the potential impact of such a project and potential alternatives is important to support the development of coherent policies and to ensure that the development goals are reached in the most effective way (Millennium Institute, 2015).

Approach and methodologies
The Millennium Institute used its Threshold-21 (T21) simulation model – an integrated and dynamic planning tool – that enables transparent cross-sectoral analyses of the impacts of policies, and enables exploration of their direct and indirect long-term consequences on social, economic and environmental development (Pedercini, 2005). The framework is implemented with the System Dynamics method, which is well-suited to capture the elements of dynamic complexity, such as feedback loops, delays, and non-linearity (Forrester, 1961; Sterman, 2000) that make public policy analysis in this area particularly difficult. That is why System Dynamics has proven very effective for the analysis of a variety of development issues over the last decades (Parayno, 1993; Qureshi, 2008; Saeed, 1987). However, it has failed in many instances, for example the case of the Club of Rome report, where, System Dynamics approach was limited to model the complex interactions of the world economy, population and ecology when they grow linearly (Meadows et al., 1972). The method also facilitates the integration of knowledge from different sectors and stakeholders into a single framework (Pedercini, 2005).

Four scenarios are analyzed in this study: the Base Run scenario (without the World Bank loan), the World Bank loan scenario (in which the World Bank loan is implemented as suggested mainly focusing on investment into irrigation infrastructure), and two alternative scenarios in which the World Bank loan is implemented but its focus is changed towards the support of small producers and training investment. This section describes the assumptions of the four scenarios.

Base run scenario: This scenario is a business-as-usual scenario without the World Bank loan, which assumes no major changes in external conditions and a continuation of current government policies.

World Bank loan scenario: This scenario assumes that World Bank loan and the GEF grant are implemented as suggested in the project document (World Bank 2013). Based on the disbursement plan of that document, the amount of the World Bank loan is divided into investment for training and investment for irrigation infrastructure. The irrigation investment distribution over the six-year project period is based on the planned amount of infrastructure per year, while for the training investment, a homogenous distribution over the six years is assumed.

Small scale World Bank scenario: In this scenario, the World Bank loan and the GEF grant are implemented but in an adjusted version, assuming that all of the investment is directed towards small producers.

Small scale and training World Bank scenario: This scenario assumes that the World Bank loan and the GEF grant are implemented but in a further adjusted version. The whole investment is not only directed towards small producers but in addition also used for training in low external input techniques instead of investing it in irrigation infrastructure.

Results

1. Base Run

Economic indicators

In this scenario, crop production accounted on average for around 60% of total agriculture GDP between 1980 and 1990, decreased to around 55% between 2005 and 2015 and declines to less than 45% between 2040 and 2050. In the same periods, value added from livestock increases from around 23% to around 30% to 44%.

Social indicators

The base run carries trends over the past into the future, at more or less current or declining rates. Average life expectancy increases from less than 50 years in 1980 to around 60 years around 2010 and nearly 90 years at the end of the simulation in 2050. The fast growth is among others due to a fast decrease of food insecurity and increase of per capita GDP. The HDI does not grow on such a fast rate because adult literacy rate grows at a slower rate. Rural poverty decreases from 71% in 1995 to 57% in 2011, and overall poverty from 68% to 47% during the same period. For the future, the Base Run shows a decrease in overall poverty to 11% in 2050, and for rural poverty following the same trend down to 17%. The reduction is due to the steady improvement of income distribution (Gini coefficient) and is affected by the changes in average per capita GDP. While the decrease of per capita GDP until 1995 counteracted the improvement of income distribution, the increase of per capita GDP after 1995 further strengthens the decrease of poverty.

Environmental indicators

Water demand increases for the most of the simulation period and a stabilizes shortly after 2045, based on recent patterns. Total water demand, for example, increased from the end of the nineties until beginning of the new century by more than 60%, and can be expected to more than doubles in the coming 35 years. The water stress index represents the proportion of water supply that is used. Hence, it corresponds to the water resources vulnerability index and is calculated as the ratio between water demand and available water. At the beginning of the simulation period less than 5% of total water supply is demanded while it more than triples until 2050. Since water supply is rather constant the increasing trend and its stabilization at the end of the simulation is mainly due to the behavior of water demand which is highly affected by the existent irrigation infrastructure. Average nitrogen content in the soil compared to its value in 1980 continuously decreases until it more or less stabilizes at the end of the simulation.

2. Different World Bank Loan Scenarios

Economic indicators

Crops value added is around 7% higher in the adjusted ‘Small Scale and Training World Bank loan’ scenario while it is only around 1% in the other two scenarios. This difference is also observable in agriculture production in monetary terms (agricultural GDP) with crop production being an important contributor. For total GDP, the difference between the adjusted ‘Small Scale World Bank loan’ and the ‘World Bank loan’ is higher than for agriculture production. This is due to the fact that in the ‘Small Scale World Bank loan’, improvements in poverty reduction as well as food and nutrition security lead to better health conditions, increasing total factor productivity also in the service and industry sectors, generating higher industry and services production facilitating higher savings and reinvestments, which eventually reinforces GDP growth.

For the indicators concerning the government's debt and interest payments levels the ratio of foreign debt over GDP and interest payments on foreign debt over government revenue is the highest in the 'World Bank loan' scenario. Whereas it is the lowest in the adjusted 'Small Scale and Training World Bank loan' scenario although the amount of the credit is the same for all the three scenarios. This is due to the fact that GDP and consequently government's revenue are higher in the adjusted 'Small Scale and Training World Bank loan' scenario. Accordingly, both ratios are smaller.

Social indicators

For the social indicators in 2050 agriculture employment is 27% greater in the adjusted 'Small Scale and Training World Bank loan' scenario than in the 'Base run' while this difference is only around 15‰ in the adjusted 'Small Scale World Bank loan' scenario and only 1‰ in the 'World Bank loan'. Similarly, both poverty levels, rural and total poverty, in the adjusted 'Small Scale and Training World Bank loan' scenario are significantly lower than in the 'Base Run' (around 20%). Whereas the difference is smaller for the 'Small Scale World Bank loan' scenario (around 10% compared to 'Base Run') and very small in the 'World Bank loan' scenario (around 1% compared to 'Base Run'). The positive results in the adjusted 'Small Scale and Training World Bank loan' scenario are due to the combination of differences in production and effects on income and its distribution. Consequently average income increase, and in income distribution, which is more equitable in the adjusted 'Small Scale World Bank loan' and 'Small Scale and Training World Bank loan' scenario due to broader involvement of the rural poor in the production, process (see agriculture employment). Similarly, prevalence of undernourishment in the adjusted 'Small Scale and Training World Bank loan' scenario is around 20‰ lower than in the 'Base Run' although crops production and therewith availability of food does not increase as much. Nevertheless, access to food does change due to the change in poverty rates. In contrast, prevalence of undernourishment in the 'World Bank loan' scenario does not decrease significantly because neither availability of food nor access to food improves in a relevant way. Also for average life expectancy, there is barely a difference of the 'World Bank loan' scenario compared to the 'Base Run' while there is at least a small improvement in the adjusted 'Small Scale and Training World Bank loan' scenario due to the improvements of GDP and food security. Similarly, these differences are also observable in the Human Development Index with average life expectancy and per capita GDP being two of the three components of the indicators.

Environmental Indicators

The water stress index, the ratio between water demand and available water, in 2020 is 40% higher in the scenarios in which the World Bank loan is mainly invested into irrigation infrastructure since this increases the agricultural water demand. However, in 2050 there is no difference compared to the 'Base Run' since at this point irrigation infrastructure is the same in all the four scenarios because the limit of 350,000 ha has been reached. In contrast the improvement of soil fertility is higher in the adjusted 'Small Scale and Training World Bank loan' scenario than in the other two scenarios due to the application of low external input techniques that restore soil organic matter and increase the use of organic fertilizer.

Since in the adjusted 'Small Scale and Training World Bank loan' scenario more money is spent for this training, the results are better in this scenario. Hence, soil matter is around 4% higher in the adjusted 'Small Scale and Training World Bank loan' scenario and around 1% higher in the other two scenarios compared to the 'Base Run'. Similarly, the adjusted 'Small Scale and Training World Bank loan' scenario also generates higher levels of soil nutrients although they are also improved in the other two scenarios but at a lower scale.

Study recommendations

Simulation results indicate that looking at the aggregated national scale the impact of the project as planned by the World Bank is rather small. This is because the loan comprises only around 1.5% of the agricultural budget for 2015-2020, and the amount of new irrigated area of 10,000 ha is only around 8% of the irrigated area and only 0.4% of the total harvested area, but still might be very important for the livelihood of thousands of farmers. For example, model analysis shows that overall poverty in the 'World Bank loan' scenario in 2050 is around 0.1% (1 % lower than in the 'Base Run' scenario), but this means that the project helps more than 3,000 people to overcome the poverty line compared to the 'Base Run' in 2050. Further improvements on a similar scale can be observed for Crops production in tons, value added from crops production, Agriculture GDP, employment and undernourishment. As negative side effects, simulation results indicate that the level of foreign debt (as share of GDP) and of interest payments on foreign debt (as share of government revenue) are slightly higher in 2050 (0.7% and 2%, respectively). In addition, the water demand and therewith the water stress index are periodically (after 2020) with 4% (40%) significantly higher than in the 'Base Run' scenario. From the analysis of the alternative scenarios, results indicate that more support of small producers has more positive impact on social indicators (around 10-15%, which is 1-1.5%, in 2050 compared to the 'Base Run' scenario), such as employment, poverty reduction and food security. Similarly, combining the increased support of small producers with training on sustainable agriculture leads to better results for employment, poverty, undernourishment and soil nutrients of around 2-3% (20-30%) in 2050 compared to the 'Base Run' scenario. And resulted in less agriculture water demand for the first decade after implementation compared to the 'World Bank loan' scenario, and to higher increase of crops production, crops value added and agriculture GDP of around 4-7% in 2050 compared to the 'Base Run' scenario. Hence, focusing the investment on training instead of irrigation infrastructure seems to have higher positive effects in the long run than the focusing the investment mainly on irrigation infrastructure. Based on this analysis, it is recommended to direct the support of such a loan towards small producers and training on low external input techniques.

Further research

This study provides an initial analysis of the impacts of the WB project and of possible alternatives. The analysis can be improved and the underlying model strengthened in various ways in order to gain more insights on such impacts. Areas for further modeling activities and expansion of model structure that would enable more detailed analysis of loan's impacts include:

- Include the effect of irrigation on biodiversity, on soil nutrients leaching / pesticide runoff, fertilizer / pesticide use / CO_2 emissions / land use change / salinization / private investment

- Allow for interactions between forest and water availability
- Assess the impact of low external input techniques on water availability
- Include the distinction between primary, secondary and tertiary irrigation

In addition, interviews with sector's experts involved with the implementation of the project would be useful to further improve the model and gain additional insights on this especially important project for the agriculture sector in Senegal.

5.2. Australian Environmental-Economic Accounts for agriculture

In Australia, agriculture is an important sector of economy and contributes about 2.3% (with exports worth $41 billion annually) to the national GDP. About half of the total 400 million hectare land area in the country, is used by agriculture, comprising 26 million hectare under crops, while the rest is mixed livestock-crop, livestock only or rangeland agriculture (ABS, 2015). However, agriculture consumes about 65% of natural water resources in Australia (Hochman et al., 2013). Australian agriculture will be able to meet the national demand for food for the projected population in 2050 and beyond. However, the ability to contribute towards the global food security in coming decades will be a challenge, as the recent trends indicate decline in the agricultural productivity rate from 1.95% (between 1977-1999) to 0.4% (between 1999-2007). Along with global agricultural research and farming community, Australian agriculture is also looking for alternative approaches for its economic and environmental sustainability (Sandhu et al., 2012).

There are negative and positive externalities in agriculture, which are not realised by current market environment, and thus they remain 'invisible' in farm economy. Moreover, dependency of agricultural production on healthy ecosystems is not being recognized either. These information gaps need to be filled to provide the right incentives for managing agricultural systems for productivity and environmental sustainability.
In Australia, the Australian Bureau of Statistics (ABS) realised this and produces a set of environmental-economic accounts (ABS, 2017) each year. The value of Australia's environmental assets (in current prices) increased 95% over the period 2005-06 to 2014-15 from $2,999.5billion to $5,837.5billion. The value of Australia's produced capital also increased over this period, although to a lesser extent (70%), rising from $3,276.7b to $5,564.1b. Environmental assets now make up the largest share of Australia's capital base. Here the environmental-economic accounts (2017) related with agriculture sector and reflected in national accounts of Australia are summarised using the TEEBAgriFood framework.

Objectives and scope
The ABS produce a range of environmental accounts on water, land, energy, environmental assets as a part of its national accounts (ABS, 2017). However, in order to provide a complete picture of the interactions between environment and economy, ABS has produced the environmental-economic accounts, which is a work in progress. These are underpinned by the System of Environmental-Economic Accounting (SEEA).

Approach and methodologies
Australian Environmental-Economic Accounts (AEEA) follow the System of Environmental-Economic Accounting 2012—Central Framework (SEEA Central Framework). This

multipurpose conceptual framework describes the interactions between the economy and the environment, and the stocks and changes in stocks of environmental assets (UNSD, 2012). The SEEA Central Framework was adopted by the UN Statistical Commission as an international statistical standard in 2012.

The SEEA Central Framework uses a systems approach to organise environmental and economic information, covering, as completely as possible, the stocks and flows that are relevant to the analysis of environmental and economic issues. The SEEA Central Framework applies the accounting concepts, structures, rules and principles of the System of National Accounts (SNA).

SEEA defines environmental assets as being 'the naturally occurring living and non-living components of the Earth, together comprising the bio-physical environment that may provide benefits to humanity'. Within the SEEA, assets are measured in both physical and monetary terms, whereas the SNA relates only to monetary information.

The notion of environmental assets used in this study is consistent with the SEEA definition and has the potential to include the following resources in agriculture, forestry and fishing sector:

- Land
- Soil resources
- Timber resources
    - Cultivated timber resources
    - Natural timber resources.
- Aquatic resources
    - Cultivated aquatic resources
    - Natural aquatic resources.
- Other biological resources (excluding timber resources and aquatic resources)
    - Water resources
        - Surface water
        - Groundwater
        - Soil water.

Results

Australia's economic production, as measured by Gross Value Added (GVA) in chain volume terms, rose 73% over the period 1996-97 to 2013-14. Over the same period, indicators of environmental pressure related to the total production of waste, energy consumption and greenhouse gas (GHG) emissions all increased, while water consumption fell. Waste production rose 163%, energy consumption increased 31% and GHG emissions increased 20%. Water consumption in Australia has fallen by 16% since 1996-97. However, the increase in water availability over the most recent years, due to higher rainfall, has supported a rise in water consumption (an increase of 40% between 2010-11 and 2013-14) and in turn led to a recent increase in the intensity of water use by industry.

The value of production generated by the agriculture industry (including forestry and fishing), as measured by its GVA, rose from $24b to $36b between 1996-97 and 2013-14. The agriculture industry's contribution to total GVA across all industries fell from 3% in 1996-97 to 2% in 2013-14.

The agriculture industry witnessed a steady trend downwards in water intensity, decreasing 67% over the period 1996-97 to 2009-10. In response to the drought dominated climatic conditions of the early 2000s, the agriculture industry became more efficient with water use through infrastructure improvements, technology advancements and changes to crop selection. Between 2009-10 and 2013-14, however, increased water availability resulting from higher rainfall accompanied a 56% rise in the volume of water consumed per unit of economic output produced by the agriculture industry.

Energy intensity

The energy intensity of the agriculture industry increased 40% over the 18 years to 2013-14. Energy consumed per unit of economic production by agriculture was variable over the whole period from 1996-97 to 2013-14, primarily due to swings in the industry's economic output. GHG emissions intensity was similarly variable, rising 18% in the decade to 2007-08, before falling thereafter to finish down by 26% across the entire recorded period. In contrast, waste production by agriculture recorded a 32% increase in intensity between 1996-97 and 2013-14.

Water consumption

The agriculture industry was the largest consumer of water throughout the six years from 2008-09 to 2013-14, consuming 11,814GL of water in 2013-14. Between 2008-09 and 2010-11 water consumption by the agriculture industry was steady at around 7,300GL per annum. Water consumed by the agriculture industry increased by 4,464GL between 2010-11 and 2013-14, with the three most significant contributors to this increase being: dairy cattle farming; sheep, beef and grain farming; and other crop growing. In combination, these three activities made up 83% of total water consumed by the agriculture industry in 2013-14.

Water intensity

Water intensity is a measure of the water consumed to produce one unit of economic output. It is calculated by dividing water consumption by Industry Gross Value Added (GL/$m GVA). The volume of water required by the agriculture industry to produce one unit of economic output fell by 68% between 1996-97 and 2010-11 to 0.21GL/$m GVA. Since then, the water intensity of Agriculture has increased by 58% to 0.33GL/$m GVA against a backdrop of easing drought conditions.

Gross value of irrigated agricultural production

Total gross value of irrigated agricultural production (GVIAP) for Australia in 2013-14 was $14.6b, up 22% from 2008-09. The three commodities with the highest GVIAP in Australia in 2013-14 were dairy ($2.7b, up 21% from 2008-09) fruit excluding grapes ($2.7b, up 14% from 2008-09), and vegetables ($2.5b, down 4% from 2008-09).

Greenhouse gas emissions

Agriculture industry (including forestry and fisheries) which recorded a fall in emissions of 31.6Mt (or 24%) between 2007-08 (132.3Mt) and 2012-13 (100.7Mt).

Carbon stocks

Carbon stock accounts report 239,581Mt of carbon stored in Australia's geosphere. In comparison, 14,270Mt of carbon is stored as biomass carbon and 16,811Mt is stored as soil organic carbon.

Employment

There was a 20% drop in employment in the agriculture industry, from 403,500 in 1996-97 to 324,500 in 2013-14. Following table shows the quantitative and qualitative indicators addressed in the study.

Study recommendations

The ABS first published environmental accounts in 1995, beginning with monetary estimates for a number of environmental assets within scope of the SNA asset boundary. In particular, estimates for subsoil assets, and forests and land were developed within the ABS national accounts program and these are now an established feature of the national balance sheet within the Australian System of National Accounts (ASNA). During the 1990s the ABS also commenced a program of environmental accounts development within its environmental statistics area and this program continues to drive the development of these accounts within the ABS - often in partnership with other agencies. Australian Environmental-Economic Accounts is work in progress and the estimates will improve as more stocks and flows of ecosystem services are captured in national balance sheets.

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