Bottom-up methodology

TEEB Animal Husbandry

True Price™

I. Methodological framework

II. Livestock snapshots valuations – Carbon, water and land

III. Landscape level valuation - Maasai pastoralism in Tanzania

True Price, 2016, *Bottom up methodology TEEB Animal husbandry*. Amsterdam, The Netherlands

TEEB asked for the assessment of the impact of global livestock production systems on human systems and ecosystems. Hidden costs and benefits of poultry, beef and dairy production systems have been studied. This report presents the valuation methodologies used by True Price for the analysis.

Authors (True Price): Pietro Galgani, Vincent Fobelets, Orfeas Agis Karachalios, Lana Krisanto, Leonardo Verkooijen, Adrian De Groot Ruiz

www.trueprice.org

 This study was subsidized by TEEB and was carried out by a consortium under the management of Wageningen UR. The biophysical data and the snapshot descriptions used as input for the assessment of greenhouse gas emissions, nutrient leaching and land occupation were provided by Wageningen UR. The monetary coefficients to be used in the assessment of water pollution were also provided by Wageningen UR. Information and views set out in this study do not necessarily reflect the official opinion of TEEB.

Front cover photo credits: Shutterstock

About this report

This report is a methodological appendix to the TEEB Agriculture and Food report on Livestock **"**Valuation of livestock eco-agri-food systems: poultry, beef and dairy**".** It contains the methodology used for the bottom-up valuation of externalities of livestock snapshots and for the landscape level valuation on Maasai pastoralism in Tanzania. The full citation of the main report is the following.

Baltussen W., T. Achterbosch, E. Arets, A. de Blaeij, N. Erlenborn, V. Fobelets, P. Galgani, A. De Groot Ruiz, R. Hardwicke, S.J. Hiemstra, P. van Horne, O. A. Karachalios, G. Kruseman, R. Lord, W. Ouweltjes, M. Tarin Robles, T. Vellinga, L. Verkooijen; *Valuation of livestock eco-agri-food systems: poultry, beef and dairy.* Wageningen, Wageningen UR (University & Research centre), Trucost & True Price, publication 2016-023.

About TEEB Agriculture and Food

The Economics of Ecosystems and Biodiversity (TEEB) for Agriculture & Food is an initiative led by the United Nations Environment Programme's TEEB Office. It brings together economists, business leaders, agriculturalists and experts in biodiversity and ecosystems to provide a comprehensive economic evaluation of the 'eco-agri-food systems' complex, and demonstrate that the economic environment in which farmers operate is distorted by significant externalities, both negative and positive, and a lack of awareness of dependency on natural capital.

www.teebweb.org/agriculture-and-food

About True Price

True Price helps organizations measure, monetize and improve their impact. True Price works with organizations – multinationals, SMEs, NGOs, governments – to quantify and valuate their economic, environmental and social impacts. This provides the information needed for sustainable risk management, strategic decision making and stakeholder engagement.

www.trueprice.org

Foreword

TEEB Agriculture and Food is a global initiative by UNEP-TEEB that engages leading institutes to identify and quantify the externalities (i.e. hidden costs and benefits) of the global food system. Understanding the hidden costs and benefits of our food production and consumption system is crucial to address the challenge of feeding the world sustainably.

True Price was asked by UNEP-TEEB, together with Wageningen UR and Trucost, to develop a comparative assessment and valuation of the impact of animal husbandry systems worldwide. The resulting study is to the best of our knowledge the first comprehensive analysis that quantified and monetizes the externalities of animal husbandry worldwide.

This report presents the methodology used by True Price in this assessment. It presents the general theoretical and methodological frameworks and then illustrates the valuation approaches used for the assessment of negative externalities and dependency on ecosystem services of livestock systems. It also illustrates the innovative method used for the valuation at the landscape level in a pastoral region of Tanzania where natural capital is especially at risk.

As the measurement and valuation of externalities of agricultural supply chains is such a new research area, it is crucial to be transparent about the methodology used as well as its strengths and limitations. We hope that the report will be useful to researchers and practitioners working on impact measurement and valuation.

Adrian De Groot Ruiz
Executive Director True Price

Contents

The report is divided in three parts. **Part I** presents the underlying valuation framework and general definitions. **Part II** explains the valuation methodologies used in the bottom up analysis of ten livestock systems ("snapshots"), in five different countries, for greenhouse gas emissions, water pollution, blue water dependency and land occupation. **Part III** describes the approach used in the landscape level study on ecosystem services and natural capital value of Maasai pastoralism in Tanzania.

List of abbreviations

BFM	Bone Free Meat
CBS	Central Bureau of Statistics Netherlands
CICES	Common International Classification of Ecosystem Services
DALY	Disability Adjusted Life Year
EAM	Eastern Arc Mountain
EB	Ecosystem Benefit
ES	Ecosystem
ESS	Ecosystem Services
FAO	Food and Agriculture Organization
FCR	Feed Conversion Ratio
GHG	Greenhouse Gases
GLEAM	Global Livestock Environmental Assessment Model
IAWG	Interagency Working Group on Social Cost of Carbon
IPCC	International Panel on Climate Change
KSH	Kenyan Shiling
LCA	Life Cycle Assessment
LU	Land Use
NC	Natural Capital
NTFP	Non Timber Forest Products
NP	National Park
MEA	Millennium Ecosystem Assessment
MNP	Manyara National Park
PEB	Present Ecosystem Benefits
PPP	Purchasing Power Parity
TEEB	The Economics of Ecosystems and Biodiversity
SCC	Social Cost of Carbon
SEEA	System for Environmental Economics Accounting
TEV	Total Economic Value
TLU	Tropical Livestock Unit
TNP	Tarangire National Park

TZS	Tanzanian Shilling
USD	United States Dollar
WUR	Wageningen University Research

Part I - Methodological framework

1 Introduction

In Part I of this report we provide a brief exposition of the underlying valuation framework that was used in the bottom-up valuations for the TEEB Animal Husbandry study.

The valuation of external effects is a classic topic in welfare and environmental economics. However, the systematic valuation of environmental impacts is relatively new. Methods to value environmental impacts such as pollution and resource use have emerged in the last 20 years in the areas of Life Cycle Analysis (eg. Steen, 2000) and policy (eg. ExternE, 2005). The social costs of global warming as a result of greenhouse gas emissions has received significant public attention (Stern, 2006). Costanza (1997) provided the first global estimate of the value of ecosystem services. Ecosystem services were first defined, identified and classified in a structured way in the Millennium Ecosystem Assessment (MEA) (2005) report.

The initial TEEB study (2010a, 2010b) provided a major push globally in the analysis, measurement and valuation of ecosystem services and biodiversity. It extended and further specified the list of services in the MEA report, added a comprehensive classification of ecosystem types and associated services (TEEB 2010a) and resulted in a database with thousands of values of ecosystem services based on academic research (Van der Ploeg and De Groot 2010, McVittie and Hussain 2013).

From a governmental perspective, a coalition of intergovernmental institutions led by the United Nations has developed a conceptual framework for a System for Environmental Economics Accounting framework “SEEA” (UN, 2014abc). This includes a formal Central Framework (UN, 2014a) that does not fully integrate ecosystem services as well as an experimental framework for ecosystem accounting (UN, 2014c).

From a business perspective, the Natural Capital Coalition comprised of business and civil society actors is developing a Natural Capital Protocol for businesses to conduct National Capital Assessments (NCC, 2013).

From an academic perspective, an appealing high-level valuation framework is provided by the research into “inclusive wealth.” This approach has been pioneered by leading economists (eg. Arrow et al 2012) and provides a consistent conceptual framework for wealth accounting that includes, in addition to manufactured capital, also natural and human capital. It has led to several reports with a global outlook (UNEP 2012, 2014). The inclusive wealth approach focuses on human well-being and has a broader scope than the SEEA framework, which restricts itself to environmental effects on marketable goods and mostly excludes externalities. The comprehensive wealth approach (Worldbank, 2011) takes a position in the middle: it has a similar theoretical structure as the inclusive wealth approach, but is aligned in terms of scope with the SEEA approach.

The current framework, which has been used for the bottom-up valuations of this study, builds mostly upon existing approaches. The underlying assumptions are in line with the True Price's Principles for Impact Measurement and Valuation (True Price, forthcoming). With respect to the characterization and classification of Natural Capital it follows the SEEA framework (UN, 2014abc). With respect to valuation, it is founded upon the inclusive wealth approach (UNEP 2012, 2014).

Chapter 2 describes the methodological framework. Chapter 3 provides a brief exposition of how the framework is applied. Chapter 4 provides a discussion of the limitations of the framework and a justification for it.

2 Methodological Framework

This section describes the framework used to measure and value the impact of human activity on Natural Capital. This framework follows True Price's Principles on Impact Measurement and Valuation (True Price, forthcoming). The underlying perspective is that measuring and valuing Natural Capital can be used to inform decisions. The starting point is a decision maker, such as a policy maker or a consumer, who faces a choice and whose choice has a certain impact on the state of the world. The framework follows the three steps that have to be taken to make an informed decision:

1. Identify the decision problem and in particular the decision set of choice alternatives
2. Measure the impact of the choice alternatives on the state of the world by characterizing a system and estimating the consequences of choice alternatives
3. Value the impact of choice alternatives by attaching a quantitative measure of the desirability of a choice alternative to the decision-maker

Traditionally, a valuation or welfare function is assumed to represent the complete preferences of the decision maker so that the desirability of a state of the world can be reduced to a single number. As shown in True Price (forthcoming), this assumption can be relaxed, so that the valuation function represents a partial order of the set of alternatives and the desirability of an option can be represented by several dimensions. These dimensions can represent several capitals but can also represent, on a higher level, various valuable characteristics such as wealth, equality, intrinsic value etc. These dimensions could be aggregated to one overall welfare function but need not be.

In particular, in this study we have valued Natural Capital in terms of inclusive wealth (UNEP, 2014). In terms of interpreting the results of the valuations, this can be considered as an important decision dimension but not necessarily the only one. Issues such as inequality or the intrinsic value of nature can be complementary decision criteria for decision makers.

2.1 Decision set

A natural capital valuation of livestock externalities can inform several decision makers. It can inform policy makers, who face several policy options with various potential impacts on Natural Capital. It can inform businesses that face, for example, various options to source and produce their products. It can also inform consumers who face decisions as to what products to buy. For example, the snapshot valuations can inform several types of decision makers, although one has to be careful that only those snapshots can be compared that present alternatives (substitutes) to consumers, businesses or policy makers. The Natural Capital valuation of the Maasai Steppe can inform choices of Tanzanian policy makers, in particular in relation to the issue of land conversion.

2.2 Impact measurement

To measure impact, first a system has to be defined and characterized. In characterizing the system of Natural Capital, we follow the approach described in the System of Environmental Economic Accounting (SEEA) 2012 (UN 2014abc) and in particular the Experimental Ecosystem Accounting framework. For the purposes of the current study, which focuses on products, agricultural practices and a region, we adapt and abstract from the national accounting focus of SEEA. In addition, the terminology used here is slightly different, mainly because we have chosen to use Natural Capital as the overarching concept.

In the current framework, Natural Capital is composed out of Natural Capital assets (Environmental assets in SEEA), which are “the naturally occurring living and non-living components of the Earth, together constituting the biophysical environment, which may provide benefits to humanity” (UN, 2014a). There are two types of Natural Capital assets

- Ecosystem assets
- Abiotic assets

An ecosystem is “a dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit” (UN, 1992). An important aspect of ecosystems is biodiversity, “the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part: this includes diversity within species, between species and of ecosystems” (UN, 1992).

Natural Capital assets, which are stocks, provide flows of Natural Capital Services. In classifying the ecosystem assets, we follow the CICES classification (SEEA, 2013). We distinguish three types of ecosystem services provided by ecosystem assets:

- Provisioning services
- Regulating services
- Cultural services

Analogously (although with a bit of a stretch), abiotic assets can be said to provide abiotic services. Through these services, Natural Capital assets provide natural goods (‘benefits’ in the SEEA Framework) to people.

The set of natural goods is defined here as all goods that require direct input of a Natural Capital asset and either have a market price or have direct consumption value. This includes goods like agricultural products, clean water, clean air, timber products, oil, gas, etc.

Once the system has been characterized, the impact of the different choice alternatives in the decision set can be estimated. This requires making a scenario describing the course of events following a choice and a reference scenario. All valuations in this chapter are absolute. In the snapshot valuations, the reference scenarios are supply chains with zero natural capital impact. In the valuation of Natural Capital, the reference scenario involves zero Natural Capital value.

2.3 Valuation

Building on the framework provided by UNEP's Inclusive Wealth Reports (2012, 2014), the starting point is the total wealth of a region.

Definition 1. The total wealth of a region is

$W(t) = \sum_A P_i^I(t)K_i(t)$

where

- T denotes the time period in years
- $W(t)$ is the wealth of the region at time t
- i a capital asset (including natural, human, social, reproducible capital assets, possibly other type of assets and time) and A the countable set of all capital assets
- $K_i(t)$ the stock quantity of capital asset i
- P_i^I the internal shadow price of asset i denoting the value of i to the (stakeholders inhabiting the) region

Furthermore, we define $P_i^E(t)$ as the external shadow price of an asset, denoting the value of the asset to all other regions. The total shadow price is $P_i(t) = P_i^I(t) + P_i^E(t)$. If the region under study is the world, then $P_i(t) = P_i^I(t)$.

Let N be a subset of A containing all Natural Capital assets. The internal Natural Capital Value $NC^I = \sum_N P_i^I(t)K_i(t)$ is the sum of the values of all Natural Capital Assets to the region. The external Natural Capital Value $NC^E = \sum_N P_i^E(t)K_i(t)$ is the sum of the value of these assets to all other regions and the (total) Natural Capital Value is $NC = NC^I + NC^E$.

Let F be the set of final consumption goods, i.e. all goods from which human individuals derive value (use and non-use). Let $C_k(t)$ be the quantity of consumption good k and $v_k(t)$ its consumption value. Let G be the set of natural goods and let $Q_j(t)$ be the quantity of natural capital good j. Observe that the quantity of good k supplied in an economy in a given year ($C_k(t)$) will be influenced by the supply of at least some natural goods. In addition, the production of natural good j ($Q_j(t)$) will depend on the input of at least some Natural Capital asset K_i and possibly the input of other type of capital assets (eg. manufactured and human capital).

The internal shadow price of any asset $K_i(t)$ can be valued by its contribution to final consumption goods:

$$P_i^I(t) = \sum_{s=t}^{\infty} \frac{1}{(1+\delta)^{s-t}} \left(\sum_F \frac{\partial C_k(s)}{\partial K_i(s)} v_k(s) \right) \tag{1}$$

where δ is the social discount rate.

Equation (1) can also be expressed as

$$P_i^I(t) = \sum_{s=t}^{\infty} \frac{1}{(1+\delta)^{s-t}} \left(\sum_G \frac{\partial Q_j(s)}{\partial K_i(s)} \pi_j^I(s) \right) \tag{2}$$

where $\pi_j^I(s) = \sum_F \frac{\partial C_k(s)}{\partial Q_j(s)} v_k(s)$ is the shadow price of natural good j.

3 Application of valuation framework

To apply the valuation framework, one must first characterize at least the relevant subset of Natural Capital assets, the relevant subset of other type of assets as well as the relevant subset of natural goods.

In the valuation of a Natural Capital Asset, one needs to identify two parts of equation (2).

(i) The marginal product of each natural good the asset provides

(ii) The shadow price of each natural good

Identifying the marginal product requires knowledge of the production function of natural good j as well as the quantities of all input assets. This will require knowledge about the ecosystem and/or abiotic services provided by the Natural Capital Asset. In case inputs such as labor or capital assets are used as well, such as in agriculture, also economic data is required. Estimating how the marginal product will develop over time is complex and it involves substantial environmental and economic assumptions.

How to identify the shadow price of a natural good will depend largely on the type of good. If it is a marketable good, standard economic techniques can be used (UNEP, 2014). A good approximation for the shadow price is the market price if markets are well-functioning (Dreze and Stern, 1990). In practice, even if markets are only reasonably well-functioning it may be the best approximation. The SEEA framework for example, strongly prefers to use market prices (UN 2014a).

If the natural good is not marketable, then it provides direct consumption value and revealed or stated preference elicitation techniques need to be used.

Ideally, to identify both the marginal product and the shadow price (through market or preference data) local data is used. In practice, this data is difficult to obtain for all goods and assets. Hence, benefit or value transfer of some sort is required. When doing so, the general guidelines by Brander (2013) are taken into account.

Note that the approach above is consistent with the unit resource rent approach described qualitatively in the SEEA 2012 Experimental Ecosystem Accounting (UN, 2014c). The marginal product times the shadow price of each natural good is broadly equivalent to the total product of a natural good by a Natural Capital asset times the unit resource rent.

The framework can be used to value impacts, dependencies and externalities.

To value the impact of human activity on Natural Capital, one needs to estimate how that human activity affects the quality and quantity of Natural Capital assets. Once that has been done, the valuation is straightforward.

To value the dependency of human activity (or product or business) on Natural Capital one needs to identify all natural capital goods that are required for that activity as well as the input intensity and shadow price of each good.

Finally, the externality of a market activity can be defined as the change in total wealth not reflected by market prices. The externality of a consumption good can be defined as the total externalities incurred in the activities involved in the production, consumption and disposal of such consumption good.

4 Discussion

4.1 Relation to previous literature

As described before in broad strokes, the current valuation framework follows the SEEA (UN, 2014abc) in its characterization of Natural Capital and builds on the inclusive wealth framework (UNEP 2012, 2014) for the valuation. It has adapted the SEEA framework to be applicable and manageable for smaller functional units than countries and provides a specific interpretation of the inclusive wealth framework. In particular, it makes a distinction between internal and external Natural Capital value to include transregional externalities and builds more structure into the valuation function.

We have chosen to follow the SEEA framework to characterize Natural Capital, as it provides the most robust characterization and classification of Natural Capital in our perception. We have chosen to build upon the Inclusive Wealth approach for the valuation framework, as that provides the most robust and sound economic framework.

4.2 Assumptions and Limitations

In terms of its characterization of Natural Capital, the current framework is compatible with a wide range of models for ecosystems and their services. At the same time, this means that it does not provide more guidance with respect to characterizing Natural Capital than the approaches it is built upon. Current approaches to describe Natural Capital, in particular ecosystems, face serious limitations due to the complexity of ecosystems and the many theoretical and data challenges remaining (UNEP, 2014, UN 2014abc).

One particular challenge is to integrate biodiversity in the production function of an ecosystem. In addition to the practical difficulty of measuring it, the concept covers so many aspects of ecosystems that it is difficult to include quantitatively in a systematic manner. Generally, the quantitative relationship between biodiversity, ecosystem components and processes and services is still poorly understood (De Groot et al 2010). In addition, most measures of biodiversity such as the mean species index are relative, whereas a valuation requires an absolute measure (Colwell, 2009). Most importantly, although it is highly intuitive that biodiversity should benefit ecosystem function, the actual empirical evidence for a causal link between measures of biodiversity and ecosystem functions is quite tenuous (Haynes Young and Potschin, 2010).

In terms of economic valuation, the current framework inherits the many fundamental and practical limitations of the economic theories and models it follows, among others:

- A valuation implicitly contains intrapersonal comparisons of utility, which is highly problematic (eg. Elster and Roemer, 1991);
- Market prices are only equal to shadow prices in perfect markets (eg. Dreze and Stern, 1990);
- Market prices only reflect shadow prices (from a utility perspective) even in perfect markets if all individuals earn the same income;
- Distributional effects of decisions are not easy to take into account and are therefore often not taken into account, although there is strong evidence that people have social preferences (Fehr & Schmidt, 1999, Fehr & Fischbacher 2003);
- Comparing income or wealth between countries or years based on a price index, which is implicitly done in the inclusive wealth framework, is problematic (Van Veelen, 2002);
- A valuation on the basis of marginal effects can only be extrapolated to a certain degree and provides limited insight in total or average effects in several cases;
- An economic valuation assumes a certain degree of substitutability between goods, which at the margin is a valid assumption but is not valid if the scale of substitution is too large (eg. one cannot substitute the entire planet in the foreseeable future);
- Most environmental and economic models assume a certain degree of smoothness, monotonicity or even linearity of production functions, which is not always a valid assumption, certainly for large impacts that would seriously affect the environmental or economic system (eg. Farley, 2012). In particular, appearance of critical thresholds is not accounted for;
- The outcome of a valuation is strongly dependent on the discount rate. There is no scientific manner to identify the social discount rate and discounting future generations is problematic from an ethical perspective (Beckerman and Hepburn, 2007);

- Intrinsic values of Nature can be difficult to take into account in an economic valuation (e.g. Farley, 2012);
- Translating various effects of a different nature into one single number can be problematic from a theoretical and ethical perspective;
- The necessary data for a precise valuation is typically lacking (Brander 2013, UNEP 2014, UN 2014abc), requiring value transfer and a considerable number of assumptions.

In general, due to the challenges on the environmental science and economic side, valuation outcomes have considerable uncertainty.

4.3 Justification

The most important reason to conduct Natural Capital valuations is that making decisions is inevitable. All the challenges mentioned above do not go away by not doing a valuation. A valuation does not create challenges of decisions making but just makes them visible.

From a theoretical point of view, a good case can be made that the preferences of a rational decision maker can be represented by a quantitative valuation function (eg. Von Neumann and Morgenstern, 1947; Blume and Easley, 2008). This is a normative result (not a descriptive result of actual choice behaviour), that shows that a valuation can serve as an ideal of informed decision making.

From a practical point of view, policy makers must make decisions that involve complex effects on Natural Capital and involve balancing interests of various individuals now and in the future. A valuation can help to make a decision more informed and more objective with clearly defined assumptions.

In addition, it can make the decision problem more clear and simple. This is relevant to policy makers, since people are by now well known to display a wide number of cognitive biases, in particular when faced with complex problems involving uncertainty (e.g. Kahnemann 2003).

Most fundamental concerns regarding valuation can be addressed by providing additional decision criteria (possibly valued in distinct dimensions) to accompany a valuation in terms of inclusive wealth.

The uncertainty, assumptions and limitations involved in a valuation should be addressed by providing transparency about these issues and providing an uncertainty analysis to the decision maker.

5 References

Arrow, K.J., P. Dasgupta, L.H. Goulder, K. J. Mumford and K. Oleson (2012). “Sustainability and the measurement of wealth.” *Environment and Development Economics* 17: 317-353.

Beckerman, W. and C. Hepburn (2007). “Ethics of the Discount Rate in the Stern Review on the Economics of Climate Change.” *World Economics* 8: 187-210.

Blume, L.E. and D. Easley (2008). “Rationality.” *The New Palgrave Dictionary of Economics* (2nd Ed). New York: Palgrave MacMillan.

Brander, L. (2013). Guidance Manual on Value Transfer Methods for Ecosystem Services. Nairobi: Publishing Services Section, UNON.

Colwell, R.K. (2009). “Biodiversity: Concepts, Patterns, and Measurement.” In: S.A. Levin (ed) The Princeton Guide to Ecology. New Jersey (NJ): Princeton University Press.

Costanza, R., R. d’Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, S. Naeem, K. Limburg, J. Paruelo, R.V. O’Neill, R. Raskin, P. Sutton, and M. van den Belt. (1997). “The value of the world’s ecosystem services and natural capital.” *Nature* 387: 253-260.

Dreze, J. and N. Stern (1990). “Policy reform, shadow prices, and market prices.” *Journal of Public Economics* 42: 1-45.

Elster, J. and J. Roemer, eds (1991). *Interpersonal Comparisons of Well-Being*. Cambridge University Press: London.

European Envrinmental Agency (2013). Common International Classification of Ecosystem Services version 4.3.

ExternE (2005). Externalities of energy. Methodology 2005 update. European Commission.

Farley, J. (2012). “Ecosystem services: The economics debate.” *Ecosystem Services* 1: 40-49.

Fehr and Fischbacher (2003). “The Nature of Human Altruism.” *Nature* 425: 785-791.

Fehr and Schmidt (1999). “A Theory of Fairness, Competition, and Cooperation.” *The Quarterly Journal of Economics* 114: 817-868

De Groot, R.S., R. Alkemade, L. Braat, L. Hein and L. Willemen (2010). “Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making.” *Ecological Complexity* 7: 260-272.

De Groot, R.S., L. Brander, S. van der Ploega, R. Costanza, F. Bernard, L. Braat, M. Christie, N. Crossman, A. Ghermandi, L. Hein, S. Hussain, P.Kumar, A. McVittie, R. Portela,L.C. Rodriguez, P. ten Brink, P. van Beukerin (2012). “Global estimates of the value of ecosystems and their services in monetary units.” *Ecosystem Services* 1: 50-61.

Haines-Young, R., and M. Potschin (2010). “The links between biodiversity, ecosystem services and human well-being”. In D. Raffaelli and C. Frid (eds) *Ecosystem Ecology: A New Synthesis*,. Cambridge, United Kingdom: Cambridge University Press.

Kahneman D. (2003), “Maps of bounded rationality: Psychology for Behavioral Economics,” *The American Economic Review*, 93: 1449-1475.

Landers, D., & Nahlik, A. (2013). Final Ecosystem Goods and Services Classification System (FEGS-CS). EPA Science Inventory.

McVittie, A. and S.S. Hussain (2013). The Economics of Ecosystems and Biodiversity – Valuation Database Manual. TEEB.

Millennium Ecosystem Assessment (2005). *Ecosystems and human well-being: synthesis*. Washington, DC: Island Press.

Natural Capital Coalition (2013). Valuing Natural Capital in business: Towards a harmonized protocol.

Steen, B. (2000). A systematic approach to environmental priority strategies in product development (EPS). Version 2000. CPM.

Stern, N. (2006). The Stern Review: The Economics of Climate Change. London: HM Treasury.

TEEB (2010a). The economics of ecosystems and biodiversity - Ecological and economic foundations. Edited by Pushpam Kumar. Earthscan: London and Washington.

TEEB (2010b), The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A Synthesis of the Approach, Conclusions and Recommendations of TEEB.

The World Bank (2011). The changing wealth of nations: measuring sustainable development in the new millennium. Washington: World Bank Publications.

True Price (Forthcoming). Principles for Impact Measurement and Valuation - Consultation version.

Tversky A. and D. Kahneman. "Judgment under Uncertainty: Heuristics and Biases." *Science*185, No: 1124-1131.

UN/EU/FAO/IMF/OECD/World Bank (2014a). System of environmental-economic accounting 2012 - Central framework.

UN (1992). Convention on Biological Diversity. Rio de Janeiro.

UN/EU/FAO/IMF/OECD/World Bank (2014b). System of environmental-economic accounting 2012 – Applications and extensions.

UN/EU/FAO/IMF/OECD/World Bank (2014c). System of environmental-economic accounting 2012 – Experimental Ecosystem Accounting.

UNEP-IHDP and UNEP (2012). Inclusive wealth report (2012). Measuring progress toward sustainability. Cambridge: Cambridge UP.

UNU-IHDP and UNEP (2014). Inclusive Wealth Report 2014: Measuring progress toward sustainability. Cambridge: Cambridge University press.

Van der Ploeg, S. and R.S. de Groot (2010). The TEEB Valuation Database – a searchable database of 1310 estimates of monetary values of ecosystem services. Foundation for Sustainable Development: Wageningen.

Van Veelen, M. (2002). "An impossibility theorem concerning multilateral international comparison of volumes." *Econometrica* 70: 369-375

Von Neumann, J. and and O. Morgenstern (1947). *Theory of Games and Economic Behavior.* (2nd ed). New Jersey: *Princeton University Press*.

Part II Livestock snapshot valuations: carbon, water and land

Part II contains an explanation on the valuation methods for Greenhouse gases (GHG), water dependency, water pollution and land occupation. The explanation includes a description of the scoping, valuation methods, data and analysis, results and conclusions.

1 Snapshots overview

Snapshots represent different production systems for beef, milk and poultry in specific regions and countries within the scope of the TEEB - Animal Husbandry project (Baltussen et al. 2016) to which this report represent an annex. The production systems are conventionally called farms[1]. Each farm is characterized by a set of technical parameters and environmental indicators. These include the output of the system in terms of product and pollutants and the inputs to the production process, namely feed imports. Parameters characterizing the herd kept at each farm fix the performance in terms of negative impact per unit output.

See Appendix A of the main report TEEB Animal Husbandry (Baltussen et al. 2016), for more details. Parameters used are:

- Beef/Milk/Meat/Eggs output, according to snapshot.
- Herd and farm size. The latter expressed as land use by the farm itself.
- Dressing, bone-free meat (BFM) fraction and protein content of each product.
- Feed conversion rate, land use of feed and feed composition.
- N and P leaching coefficients.

1.1 Retail prices

A comparison with retail prices is contained in the main report. When a farm produces more than one product, externalities are allocated between them in order to make the comparison. The allocation is done in terms of the protein output per farm. Specifically, the allocation factor is the protein contained in the total farm output of a certain product type (e.g. milk, beef) over the total protein output of the farm.

[1] Not all production systems listed here fit the term accurately. Pastoralists systems do not occupy a fixed area and landless farmers do not own land but still have their own farms. Additionally, in the dairy mixed feeding farms only the livestock production part of the farm is looked at and not other activities, for example arable farming unrelated to feeding animals.

$$Allocation\ factor_{type} = Product\ output_{type} * \ Protein\ content_{type} / \sum_{type'} Product\ output_{type'} * Protein\ Content_{type'}$$

Consumer prices of products in the country of each snapshot are collected in the table below for comparison with externalities. Expert opinion from livestock researchers at Wageningen UR who developed the database for the TEEB Animal Husbandry study (Baltussen et al. 2016) is marked as WUR.

Product	Country	Price location	Price ($/kg boneless meat or milk)	Source
Poultry	Tanzania	Rural	3.20	WUR
	India	Rural	2.38	WUR
	Netherlands	Rural	4.77	WUR
Milk	Tanzania	Rural	1.18	WUR
	India	Rural	0.65	WUR
	Netherlands	Rural	0.92	WUR
	Indonesia	Rural	1.17	WUR
Beef	Brazil	Retail price	6.71	Instituto de Economia Agricola 2015
	Tanzania	Average of urban and rural	3.26	Kadigi et al. 2013; Elinaza A., Tanzania Daily News 2015
	India	Rural	2.31	Ministry of Statistics & Programme Implementation 2012
	Netherlands	Urban	8.08	CBS consumer prices 2014
	Indonesia	Rural	7.68	National Bureau of Statistics 2014

Table 1: Retail prices of livestock products

1.2 Scope of the bottom up valuation

Table 2. Snapshot Scope. GHG = greenhouse gases, WD = Water dependency, WP = Water pollution, LU = Land use

Snapshot	Product	System specifics	Scope of Quantification	Scope of Valuation	Notes on scope
Tanzania backyard poultry	Meat, Eggs	Backyard	GHG	GHG	WD, WP, LU were not material.
Indonesia family farm broilers	Meat	Commerical	GHG, WD, LU	GHG, WD	WP was not material.
Netherlands industrial broilers	Meat	Commercial	GHG, WD, LU	GHG, WD	WP was not material.
Tanzania pastoralist cattle	Beef, Milk	Grass-fed	GHG, WP, LU	GHG, WP, LU	Natural capital valuation. WD was not material.
India pastoralist buffaloes	Beef, Milk	Grass-fed	GHG, WP, LU	GHG, WP	WD was not material.
Brazil beef grazing with feedlot	Beef	Mixed	GHG, WP, LU	GHG, WP	WD was not material.
Tanzania dairy mixed feeding	Milk, Beef	Mixed	GHG, WD, WP, LU	GHG, WD, WP	
India dairy mixed feeding	Milk, Beef	Mixed	GHG, WD, WP, LU	GHG, WD, WP	
Netherlands dairy mixed feeding	Milk, Beef	Mixed	GHG, WP, LU	GHG, WP	WD was not material.
Indonesia dairy mixed feeding	Milk, Beef	Mixed	GHG, WD, WP, LU	GHG, WD, WP	

2 GHG valuation methodology

2.1 Introduction

Animal husbandry systems emit greenhouse gasses (GHG) in various ways. Methane is emitted via enteric fermentation of ruminants and storage of manure. Nitrous oxide is emitted in all cases where nitrogenous compounds play a role such as manure storage and application and fertilizer use and production. Carbon dioxide is emitted in all cases where fossil fuels are used or where soil organic matter is lost due to loss of (soil) organic matter caused by land use and land use change. Carbon dioxide fixation by crops and exhale of animals is considered as part of the short Carbon cycle and not included in the emissions. The impact of these GHG emissions on society is valued by a social cost of carbon (SCC), which is a comprehensive estimate of climate change damages. Costs of GHG emissions of the various animal husbandry systems are compared taking 1 kg of protein as a functional unit or – wherever relevant – a kg of animal product.

2.2 Scope and Design

The scope of the GHG valuation is equal to the scope of FAO's Global Livestock Environment Assessment Model[2]. Tables 2, 3 and 4 based on Gerber, et al. (2013), gives an overview of the included and excluded GHG sources.

The model considers all the main sources of emissions along livestock supply chains; only emissions that are generally reported as marginal were omitted (Gerber, et al., 2013). Two sources of emissions that can be significant but are not included in the scope are (i) changes in soil and vegetation carbon stocks not involving land-use change, and (ii) emissions associated with the labour force and the provision of services and assistance to stakeholders along the chain. The carbon stock changes are excluded from the GLEAM model due to a lack of information and reliable frameworks.

For the purpose of readability, the various sources of GHG emissions in this study are clustered in 8 categories:

[2] Details can be found in Appendix A of the TEEB Animal husbandry study (Baltussen et al. 2016).

1. Organic and artificial fertilizer use (N_20 emissions)
2. Manufacturing of fuel and electricity
3. Transport
4. Enteric fermentation
5. Manure and organic waste storage (CH_4 emissions)
6. Manure and organic waste storage (N_2O emissions)
7. Land use change
8. Other

Table 3. Overview scope: sources of GHG emissions, Upstream.

Activity	GHG	Included	Excluded
Feed produc-tion	N_2O	Direct and indirect N_2O from: • Application of synthetic N & manure • Direct deposition of manure by grazing and scavenging animals • Crop residue management	• N_2O losses related to changes in C stocks • Biomass burning • Biological fixation • Emissions from non-N fertilizers and lime
	CO_2/ N_2O/ CH_4	• Energy use in field operations, feed transport and processing • Fertilizer manufacturing • Feed blending • Production of non-crop feedstuff (fishmeal, lime and synthetic amino-acids) • CH_4 from flooded rice cultivation • Land use change related to soybean cultivation (Brazil/Arg.)	• Changes in carbon stocks from land use under constant management practices
Non-feed produc-tion	CO_2	• Embedded energy related to manufacture of on-farm buildings and equipment	• Production of cleaning agents, antibiotics and pharmaceuticals

Table 4. Overview scope: sources of GHG emissions, Animal Production Unit

Activity	GHG	Included	Excluded
Livestock production	CH_4	• Enteric fermentation • Manure management	
	N_2O	• Direct and indirect N_2O from manure management	
	CO_2	• Direct on-farm energy use for livestock (e.g. cooling, ventilation and heating)	

Table 5. Overview scope: sources of GHG emissions, Downstream

Activity	GHG	Included	Excluded
Post farmgate	CO_2/ CH_4/ HFC's	• Transport of live animals and products to slaughter and processing plant • Transport of processed products to retail point • Refrigeration during transport and processing • Primary processing of meat into carcasses or meat cuts and eggs • Manufacture of packaging	• On-site waste water treatment • Emissions from animal waste or avoided emissions from on-site energy generation from waste • Emissions related to slaughter by-products (e.g. rendering material, offal, hides and skin) • Retail and post-retail energy use • Waste disposal at retail and post-retail stages (food losses are not included)

2.3 Methodology

The impact of direct and indirect GHG emissions caused by the animal husbandry systems is monetized via the social cost of carbon (SCC). The SCC is an estimate of the monetized damages associated with an incremental increase in carbon emissions in a given year. It is intended to include (but is not limited to) changes in net agricultural productivity, human health, property damages from increased flood risk, and the value of ecosystem services due to climate change (US IAWG, 2013). In the context of the general methodology described in Part I – Methodological Framework Bottom-up Valuations, a marginal increase in carbon emissions reduces natural, manufactured and human capital assets. The valuation of damage is then the decrease in the value of an asset associated with a decrease of their stock quantity (for property and human health) or their marginal productivity (for ecosystems and agricultural damage).

This study uses the SCC developed by the Interagency Working Group on Social Cost of Carbon (IAWG) under the United States Government. More specifically, the 95th percentile of the SCC estimate at a 3 percent discount rate of 2015 was chosen. The value of the SCC (inflated to January 2015) is 128 USD/ton CO_2. According to the IAWG, this value represents the higher-than-expected economic impacts from climate change. However, it remains a conservative estimate as this figure does not take into account all effects, something that will likely raise the value. SCC estimates have been rising over time, due to completer models. The 128 USD/ton value is in the middle of a range of other credible estimates (Table 6).

Table 6. SCC estimates, expressed in 2015-USD

SCC method	Year of publication	SCC	Source
Tol	2008	$26	(Tol, 2008)
IAWG (average, 3% discount rate)	2013	$43	(US IAWG, 2013)
Stern review	2008	$121	(Stern, 2006)
IAWG (95th percentile, 3% discount rate)	**2013**	**$128**	(US IAWG, 2013)
CPM report	1999	$208	(Steen, 1999)
Stanford University	2015	$220	(Moore and Diaz, 2015)

2.4 Data and analysis

GHG emission data were extracted from the GLEAM. Specifically, data used throughout the GHG, water pollution and land use analysis was collected and adapted by Livestock Research/Wageningen University from various sources. As the use of transport is negligible in both the Tanzania backyard chicken and pastoral system, GHG emissions caused by transport use were omitted for these two snapshots.

In order to allocate the emissions to the various output products, protein allocation was applied. Product output numbers (kg live weight, eggs and milk per farm), dressing factors (kg carcass/kg live weight), bone free meat (BFM) factors for beef (kg bone free meat/kg carcass) and protein contents for beef and milk (kg protein per kg BFM or milk) were provided by WUR (see Baltussen et al. 2016, Appendix A). The protein content of chicken meat and beef was taken from Lawrie and Ledward (2006). The dressing percentage and the fraction of bone free meat, the same percentages have been used as in GLEAM.

2.5 Discussion

Feed production and processing, and enteric fermentation from ruminants are the two main sources of emissions in both studies, representing at least 25 percent of emissions.

Two main limitations of the GHG valuation are the uncertainty of the applied SCC and the scope of GHG emissions (discussed in Scope and Design and Methodology).

Future research and reliable frameworks are needed on changes in soil and vegetation carbon stocks not involving land-use change. This source (or sink) of GHG emissions is not included in the scope of this study.

3 Water dependency valuation methodology

3.1 Introduction

According to Mekonnen and Hoekstra (2010) global animal production requires about 2422 Gm3 of water per year (87.2% green, 6.2% blue, 6.6% grey water). Most of this volume (98.07%) refers to the water footprint of the feed while drinking water for the animals, service water and feed mixing water account only for 1.1%, 0.8% and 0.03%, respectively. For this reason, the water valuation described in this chapter, focuses on the water footprint of the feed.

This chapter describes the methodology for valuing blue water dependency in livestock systems. The methodology is based on the valuation of irrigation water used for livestock feed production using a modified residual method. The analysis is based on "snapshots" which represent different production systems from various countries: Tanzania backyard poultry, Indonesia family farm broilers, Netherlands industrial broilers, Tanzania pastoralist cattle, India pastoralist buffaloes, Brazil beef grazing with feedlot, Tanzania dairy mixed feeding, Indonesia dairy mixed feeding, India dairy mixed feeding and Netherlands dairy specialised.

3.2 Scope and design

The geographical scope of the research is based on the water use in the country of origin of the crops used in the feed of each system. Feed consists of a combination of fresh grass, hay/silage, crop residues, grains (wheat and maize), oilseed meals (soybean meal and cottonseed meal) and a number of other agro industrial by products. Most of the feed used in pastoralist and mixed systems is composed of grass, hay, and crop residues. Fibrous materials as grass and crop residues are not utilised by the poultry systems.

The water footprint can be broken down into three parts: green, blue and grey water. Blue water consumption refers to "loss of water from the available ground-surface water body in a catchment area … when it is incorporated into a product" (Mekkonen & Hoekstra, 2010) such as livestock products. Green water refers to "consumption of green water resources (rainwater)" (Mekkonen & Hoekstra, 2010). Finally, grey water is defined as "the volume of polluted water that associates with the production of all goods and services for the individual or community" (Hoekstra, 2009) and is "an "indicator of the volume of freshwater pollution” (Mekonnen & Hoekstra, 2010). Grey water use is not relevant when measuring water dependency, as it is not an indicator of water dependency but of the water pollution impact.

In this study, it has been chosen to limit the scope to blue water consumption. Given the high data requirements to assess water dependency, a choice had to be made and for blue water consumption considering more data is available and is often considered as more relevant for decision makers as green water availability is only indirectly influenced by human action. (Fulton, J., Cooley, H., Gleick, P., 2014; Hoekstra, A. & Mekonnen, M.M., 2012; Pahlow, M. & Mekonnen, M.M., 2012).

According to Hoekstra (2010), crop residues and by-products such as bran, straw, chaff and leaves and tops from sugar beet have a water footprint of about zero, because the water footprint of crop growing is mainly attributed to the main crop products, not the low-value residues or by-products[3]. Some systems use crops grown entirely on green water. Since green water resources are out of scope in this water dependency analysis, only systems using crops grown (partially) with blue water are considered. This includes:

- Poultry systems (Indonesia family farm broilers, Netherlands industrial broilers) that have to import crops that use blue water.
- Dairy mixed systems that use locally grown blue water-fed crops (grown on farm or purchased) and/or imported feed (as in the case of Tanzania dairy mixed feeding, Indonesia dairy mixed feeding, and India dairy mixed feeding).

Systems that are out of scope:

- Tanzania backyard poultry (feed is mainly second grade food products and swill).
- Beef systems (Tanzania pastoralist cattle, India pastoralist buffaloes, Brazil beef grazing with feedlot,) where feed is mainly non-irrigated grass, hay/silage, grains and soy.

- Netherlands dairy specialised system that imports feed which is grown mostly under rain fed conditions.

3.3 Methodology

The methodology used in this chapter consists of two steps. The first step involves quantifying the water being used in crop production relevant to animal feed. By making an analysis of the feed composition in each snapshot, the country of origin is established for each crop used in the feed. Mekonnen & Hoekstra (2010) made an assessment of the green, blue and grey water footprint of global crop production. Their study takes a high resolution approach, estimating the water footprint of crops at a 5 by 5 arc minute grid, meaning their figures for crop water use at national level are relatively accurate. This database is used to calculate the blue water footprint per crop per snapshot.

The second part of the method consists of calculating the residual value of water for each crop used in each individual snapshot. A number of methods exist for the economic valuation of water resources in agriculture (FAO, 2004). Considering the scope of the analysis (using various crops from different countries) and the goal of valuing blue water, the choice was made for a combination of the residual or imputation method (FAO, 2004) and the residual rent method (Thompson & Johnson, 2012). The residual method is a frequently used method to value irrigation water (Hellegers, P. & Davidson, B., 2010; Berbel et. al 2011; Speelman et. al, 2011; Ziolkowska, J., 2015). FAO (2004) describes the residual method as a budget analysis used to estimate return attributable to water where the total returns are calculated and then all non-water expenses are subtracted. This means that the full net profit is attributed to water. To correct for this we use the residual rent method (Thompson & Johnson, 2012) where irrigated crop profits are compared with non-irrigated crop profits. The difference between these values is divided by total water use to end up with an estimate for the value for water for each specific crop. The value of one cubic meter of irrigation water for each crop in each relevant country is represented in the mathematical definition below:

$$Water\ value = \frac{(Revenues_{irrigated} - Input\ Costs_{irrigated}) - (Revenues_{non\ irrigated} - Input\ Costs_{non\ irrigated})}{Water\ use}$$

In the context of the general methodology in Part I this is an approximation to the shadow price of water, as it is based on the average increase in productivity instead of the marginal increase after irrigation is applied. Once the value of water has been estimated for each crop and country relevant to each snapshot, it is multiplied by the corresponding blue water footprint of each crop in the snapshot to come up with the valuation of water per snapshot.

Some snapshots import feed from several countries. This results in a blue water dependency composed of different water footprints and different residual values (because water in each country is valued separately). To keep the analysis manageable, the choice was made to take the top three importing countries and take a weighted average of each crop's water value (hence providing a water value as if the top-3 import countries represented the total import). Table 7 provides an overview of the crops used in each snapshot.

In line with the methodological framework as defined in Part I, this will reflect the internal shadow price of water by valuing its contribution to the final consumption of goods, in this case, animal products.

Table 7. Overview of crops used in feed

Snapshot	Crops used in feed	Origin		
Indonesia family farm broilers	Wheat	Australia	Canada	India
	Maize	India	Argentina	United States
	Soghum	Local (Indonesia)		
	Soybean meal	Argentina	Brazil	India
Netherlands industrial broilers	Wheat	UK	France	Germany
	Maize	France	Brazil	Germany
	Soybean	Brazil	Paraguay	United States
	Soybean meal	Brazil	Argentina	
	Sorghum	Local (Netherlands)		
	Rapeseed	Germany	France	
Brazil beef grazing with feedlot	Maize	Local (Brazil)		
	Soybean meal	Local (Brazil)		
Tanzania dairy mixed feeding	Cottonseed cake	Local (Tanzania)		
	Maize	Local (Tanzania)		
Indonesia dairy mixed feeding	Palm kernel expeller	Local (Indonesia)		
	Maize	India	Argentina	United States
India dairy mixed feeding	Cottonseed cake	Local (India)		
	Wheat	Local (India)		
	Rice	Local (India)		
Netherlands diary specialised	Wheat	UK	France	Germany
	Rapeseed	Australia		

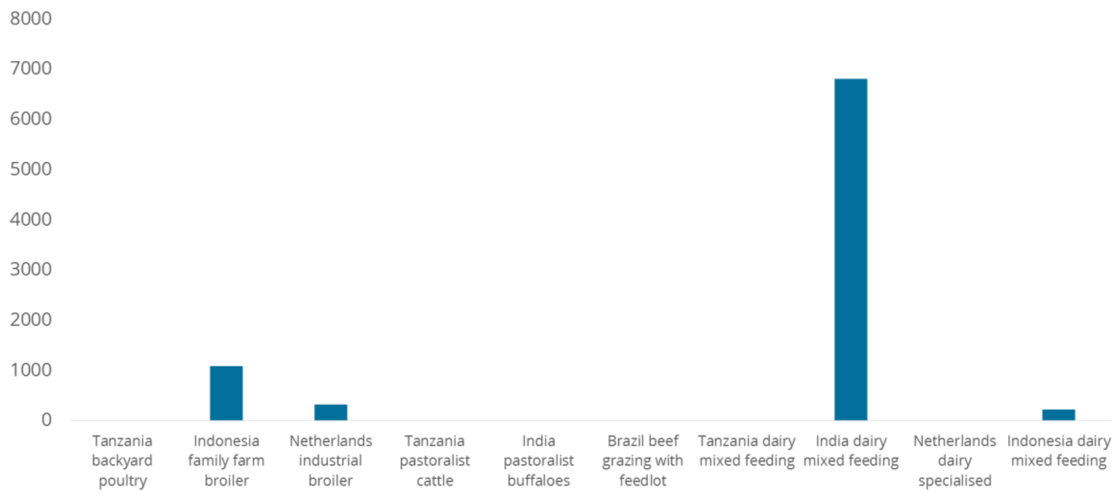
Significant blue water footprint (>1% of total)
Mostly rain fed (>99% green water)

3.4 Data and Analysis

Figure 1 gives an overview of the blue water footprint used in all the snapshots. It should be noted that most of the snapshots have a low or non-existent blue water footprint since they use crops grown with green water or use crop residues that do not count towards the total water footprint (Mekonnen, M. M. and Hoekstra, A. Y., 2010; Mekonnen, M. M. and Hoekstra, A. Y., 2011). It is expressed in liters per kilogram of protein produced to make it comparable. The conversion was made using data provided by WUR (see Baltussen et al,. 2016, Appendix A). The data shows that India has a high blue water footprint because its crops are highly irrigated in comparison to other countries.

Another interesting insight is that the Indonesia family farm broilers and Netherlands industrial broilers systems have a higher water footprint than the beef Tanzania pastoralist cattle, India pastoralist buffaloes and Brazil beef grazing with feedlot snapshots. This difference is mainly due to the amount of feed concentrate needed in the poultry systems, which is composed of crops that are partially grown using blue water. On the other hand, pastoralist and feedlot systems uses crops such as grass, hay/silage and maize/soybean meal; mainly grown under rain fed conditions.

Figure 1. Blue water footprint for all snapshots (liters/kg protein)



Next, Figure 2 shows the different residual values of water calculated for relevant crops. The residual value of water was calculated for each crop used in snapshots with significant blue water footprints. This was done by gathering data related to irrigated and non-irrigated crop yields, input costs, prices and water usage. In total, 15 different residual values for water were calculated using national averages for irrigated and non-irrigated crops, measuring the difference in profits and dividing by the average water use per hectare. References for each residual value are found in Table 8 and 9.

Table 8. Sources for residual value of water, soybeans, rice, cottonseed cake and sorghum

Crop	Country/region of origin	Sources
Soybeans	United States	Index Mundi (2015), Kansas Agricultural Statistics (2010), National Geographic (2015), Plastina, A. (2015), USDA National Agricultural Statistics Service (2014)
Soybeans	Argentina	Ghida Daza, C., (2013), Infobae, (2014), Econoagro, (2013), Global Yield Gap Atlas, (2011), Mercopress, (2014), Roulet, N., (2013), Arena, A. P., ca. (2009),
Rice	India	Duttarganvi, S. et. al (2014), Ramana Murthy, R. V. et. al (2012), Navadka, D. S et.al (2012), Fischer, R. A. et. al (2014), Siddiq, E. A., (2000), Singh, V. P. et. al (2000), XE (2011), Lagos, J. E. et. al (2015), WWF (2009), Facon, T. (2000)
Cottonseed cake	India	Aggarwal, P. K. et. al (2008), Singh, J. et. Al (2012), Ramasundaram, P. et. al (2001), Sood, D. (2015), Sood, D. (2014), Bhaskar, K. S. (2004), Netafim (n.d.), USDA Foreign Agriculture Service, (2015)
Sorghum	Europe	Department of Primary Industries (2013), Index Mundi (2015), KSU Department of Agricultural Economics (2014), Klocke and Currie (2009), Lloveras, et al. eds. (2006), Philip, Peake and McLean (2010), UNL (2013), University of Nebraska Lincoln, (2013)
Sorghum	Asia	NSW, (2013)

Figure 2. Residual rent value of blue water per crop and country ($/m3)

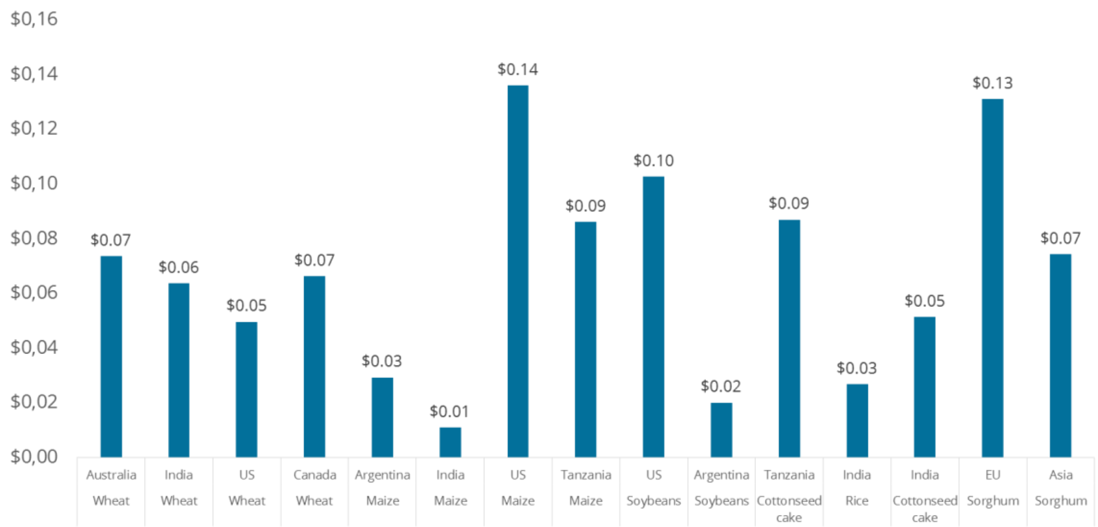


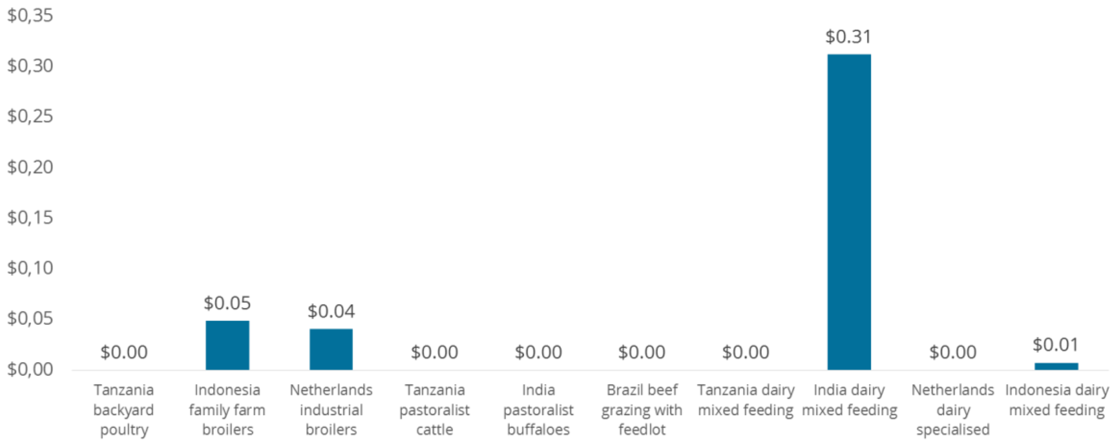
Table 9. Sources for residual value of water, wheat and maize

Crop	Country/region of origin	Sources
Wheat	Australia	Smith, D. J. et al., (2009), Iowa State University, (2015), AWB, (2015), Farrell, R., (2015), Scoot, F., (2012)
Wheat	India	Chouhan et. al (2014), Shirazi, S. et. Al (2014), EMCB-ENVIS (2013), FAO (2014), Kumar, S. (2008), Fischer, R. A. et. al (2014), Jalota, S. K. et. al (2007)
Wheat	United States	Kansas Agricultural Statistics (2012), KFMA (2011), USDA (2014), USDA National Agricultural Statistics Service (2012), National Geographic (2015)
Wheat	Canada	Agriculture and Agri-Food Canada (2015), Crozier, T. (2012), Ministry of Agriculture, Food and Rural Affairs (2015), Statistics Canada (2011), World Bank (2013).
Maize	Argentina	Ghida Daza, C., (2013), Infobae, (2014), Econoagro, (2013), Infocampo.com.ar, (2013), Global Yield Gap Atlas, (2011), Mercopress, (2014), Informa Economics, (2014), Teixeira, R., (2007)
Maize	United States	AG Decision Maker (2014; 2015), National Geographic (2015), Plastina, A. (2015), USDA National Agricultural Statistics Service (2012; 2014)

3.5 Results

An estimation of the value of blue water used for the production of feed for the relevant systems is expressed in terms of US$ per kilogram of protein produced (Figure 3). It is notable that various systems have no value, due to the fact that these systems mainly use crops cultivated under rain fed (green water) conditions. The valuation of blue (irrigated) water is based on a modification of the residual value method focusing on the change in residual profit as a result of blue water use.

Figure 3. Value of blue water (US$/kg protein)



The results show that the Indian dairy mixed feeding system has the highest dependency on blue water per kg of protein produced. Even though most of the feed in this system is based on crop residues, a small percentage of grains (wheat and rice) is used. Considering crops in India are highly irrigated, these small percentages still have a significant impact on the system's water dependency. Poultry systems that are dependent on imported feed also depend on blue water to a certain extent, although these dependencies are not as high because part of the crops, such as maize and soybeans, are partially grown under rain fed conditions. Aside from India, blue water dependency is generally immaterial within the snapshots assessed because most of these systems feed animals with crops that rely predominantly on green water (Mekonnen, M. M. & Hoekstra, A. Y., 2011).

3.6 Limitations

Considering the large scope of the study, some assumptions had to be made. The water footprint (Mekonnen, M. M. & Hoekstra, A. Y., 2011) scope is at national level, whereas several other factors such as yields, input costs for irrigated and non-irrigated crops can vary per region within a country. With regards to yields, national averages were taken, as far as possible, for irrigated and rain fed crops. Regarding input costs, it was not possible to find input costs for all irrigated and rain fed crops. To make up for it, the yield that corresponded to the input costs that were found was taken as a base and extrapolated using the difference in yield between irrigated and non-irrigated. With regards to irrigation methods, average application rates were used. Furthermore, additional factors that can influence yields, such as climate, rainfall, fertilizer application and soil type are assumed to be the same. This assumption was taken to keep the analysis manageable. Finally, it has to be noted that in this analysis blue water footprints were attributed fully to main crops, rather than crop residues, following the approach defined by Hoekstra (2010) which is assumed to be the mainstream method for blue water foot printing. This allocation method differs from the ones used in other parts of this study, where allocation between crop and byproducts based on feed digestibility has been used. If one would instead decide to allocate part of the water footprint to crop residues, the results could change significantly. This would apply in three snapshots (Tanzania dairy mixed feeding, Indonesia dairy mixed feeding and India dairy mixed feeding), where crop residues are about 70% of total feed composition.

3.7 Conclusion

The results provide the interesting insight that most animal productions systems studied are not too dependent on crops that use blue water, even those relying on imported feed. However, if we look at the various residual values of water, we reach the conclusion that blue water has the potential to contribute significantly to the value of the crops. This value might be better directed at crops that are meant for human consumption rather than animal production systems.

Besides high water use in India, these results cannot contribute to a discussion on policy implications for water management as animal production uses mostly feed grown with green water. If any debate on managing green water would come into scope, then it would be recommended to conduct additional research that looks at the dependency on green water which could shift due to climate change.

4 Eutrophication valuation methodology

4.1 Introduction

Animal husbandry systems produce varying amounts of dung and urine, containing a portion of the nutrients absorbed through the feed that animals consume. Depending on the amount of animals, the size of the farm, the soil types and waste water management, these nutrients can leach into surface and ground water and cause eutrophication. The costs of water pollution are calculated by measuring the total amount of leached nitrogen (N) and phosphorus (P) from each animal husbandry system and costing it with national water pollution coefficients for N and P.

4.2 Scope

The scope of the eutrophication valuation focuses on beef and mixed dairy systems that leach significant amounts of nutrients (nitrogen, N, and phosphorus, P) into water. These are defined as: Tanzania pastoralist beef, India pastoralist buffaloes, Brazil beef grazing with feedlot and its alternative systems, Tanzania dairy mixed feeding, Indonesia dairy mixed feeding, India dairy mixed feeding and Netherlands dairy specialised. Poultry snapshots will not be valued as these are landless systems. All water pollution that could result from the use of poultry manure in agriculture is therefore not attributed to the poultry system but to the cropland that makes use of it.

4.3 Methodology

The biophysical farm model described here was built by researchers at Wageningen University/Livestock Research (WUR) and more details can be found in Appendix A of the TEEB Animal Husbandry report (Baltussen et al. 2016). In broad terms, water pollution is quantified by applying a leaching factor of 30 % for N (IPCC 2006) to the total input of N to the land. Total input of N is calculated as the sum of N excretion of animals and N input via fertilisers. Total animal N excretion is calculated as the difference between N intake via feed and the retention in meat, milk and/or eggs. The same calculation is applied for P, albeit assuming 1% leaching. In mixed crop-livestock systems (Tanzania, India and

Indonesia dairy mixed feeding snapshots[4]) not all leaching can be attributed to livestock as crops also constitute a farm output. In these cases, one third of the leaching is attributed to livestock and two thirds to crops, based on standard economic allocation factors between crop and straw.

Fertilizer used at the supply chain level (N and P leaching at farms that cultivate crops used in livestock systems) are out of scope.

Once the amount of leached N and P has been quantified, the valuation is done by using a global water pollution coefficient provided by WUR. A full justification of the water pollution costing approach is described in a Ponsioen (2016). Monetary values are based on the Life Cycle Impact Assessment monetary coefficients presented in Weidema (2009). They represent the economic value of lost well-being due to environmental damage, quantified using a budget constraint approach.

4.4 Data and analysis

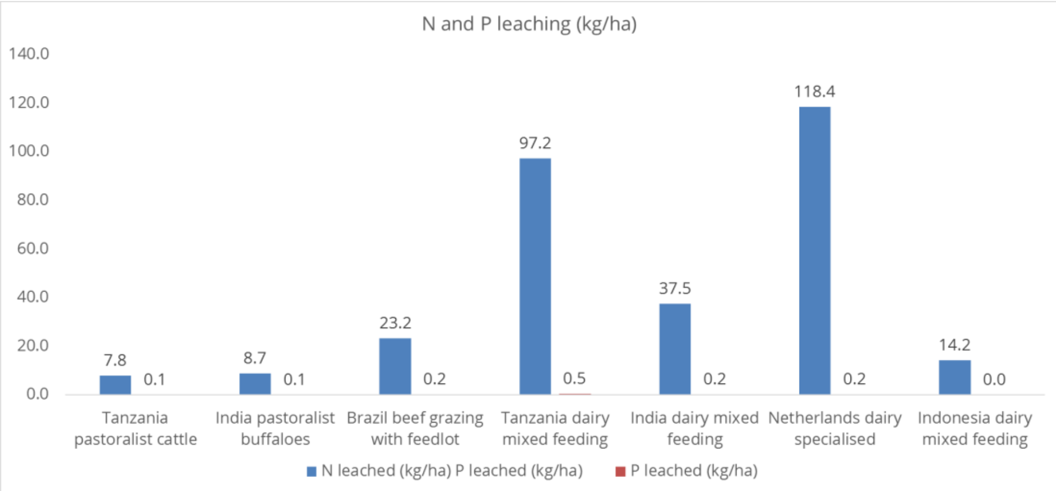
The cost of water pollution is compared between production systems by expressing it on a protein basis, i.e. an average protein produced by the livestock production system. Water pollution was defined in terms of leached N and P per hectare (ha) for each snapshot. To make this comparable across all systems, leached N+P was converted using total live weight output per ha to leached N+P per kilogram protein. The data used to make this conversion – namely, product output (kg live weight and kg milk), dressing factors (kg carcass weight/kg live weight), bone-free meat (BFM) factors for beef (kg BFM/kg carcass) and protein contents for beef and milk (kg protein per kg BFM or milk) – was also provided by WUR (see Baltussen et al. 2016, Appendix A). In mixed crop-livestock systems (Tanzania, Indonesia and India dairy mixed feeding systems) nutrient output as crops was estimated to be 60 kg N/ha and 11 kg P/ha based on average grain yields and protein contents.

The initial data analysis in biophysical terms presents some interesting insights into how the efficiency of each system has an impact on water pollution. Figure 5 gives an overview of the total leached N and P per snapshot. It shows that dairy systems have a higher

[4] The Netherlands dairy specialised system is described as a crop-livestock system but without crop output. The entirety of crop production and crop residues is fed to the animals and it is therefore modelled in the same way as a pure livestock system.

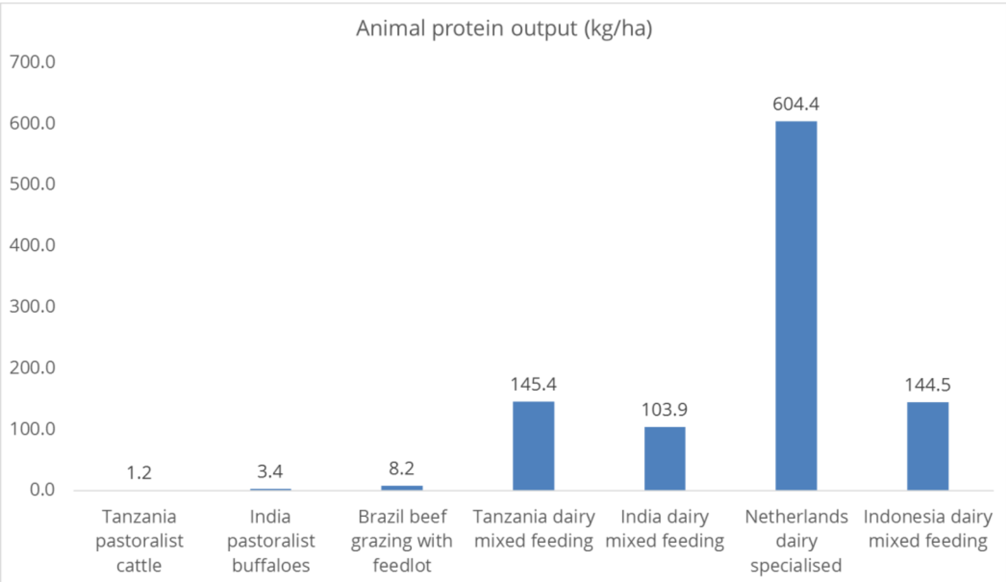
leaching of nutrients per hectare than pastoralist systems, as the latter are extremely extensive, with very low productivity and stocking density.

Figure 4. Leached N+P (kg/ha)



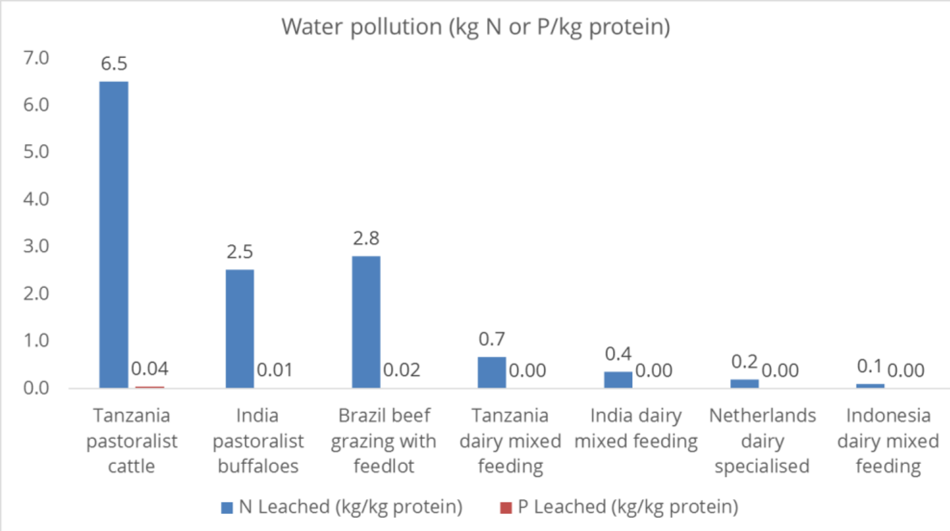
To get a better idea of system efficiencies, Figure 6 shows total protein output per farm. Figure 6 shows that the Netherlands dairy specialized system has the highest level of productivity, resulting in lower water pollution impacts per kilogram of output as can be seen in the next figure.

Figure 5. Farm protein output (kg protein/ha)



Combining the data in Figures 5 and 6, water pollution expressed in leached N+P per kilogram of protein output is derived (Figure 7).

Figure 6. Leached N+P per protein output (kg/kg protein)

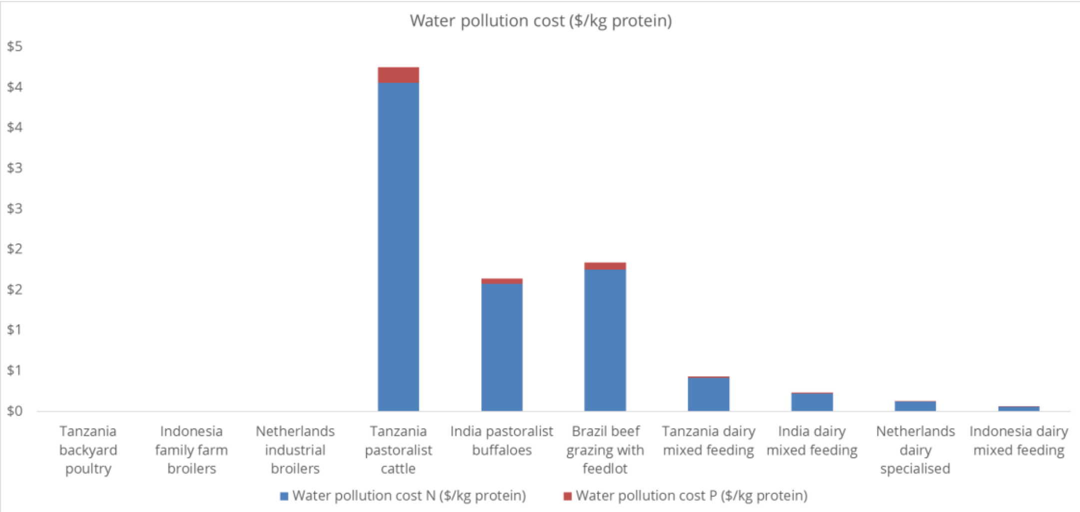


The results tell another story yet again when valuing the N and P leached using country-specific monetary coefficients for water pollution expressed in US$/kg protein produced (Figure 8). This is explained in the next section.

4.5 Results

Figure 8 shows that Nitrogen pollution is more important than Phosphorous. The Tanzania pastoralist cattle snapshot has the highest costs of water pollution per kilogram of protein produced among all snapshots (US$4). This is the snapshot with the lowest nutrient leaching per hectare, but also the least productive system, meaning that productivity is so low that water pollution, although limited is still high relatively to the proteins produced. Pastoral buffaloes in India also have a high cost of water pollution, for similar reasons. Brazil grazing with feedlot systems have a comparable eutrophication cost per kg protein, as they are more productive but they also lead to more manure and more nutrient deposition and leaching. Similarly, dairy systems, which are characterized by high leaching per hectare, also have relatively lower water pollution cost per kg of protein produced. This is mainly due to the high protein production represented by milk. Among them, Tanzania and India dairy mixed feeding systems have higher external costs due to, respectively, high use of imported feed and high use of mineral fertilizer. The Netherlands dairy specialised system has the highest leaching of water pollution per hectare but, being highly productive, also the second lowest eutrophication cost.

Figure 7. Water pollution costs for Animal Food Products in the Selected Snapshots (US$/kg protein)



The extended interpretation and discussion of the results are found in the main TEEB-Animal Husbandry report (Baltussen et al. 2016).

4.6 Limitations

The main limitations of the biophysical and monetary models underlying the water pollution analysis are discussed here. First of all, mixed farms are studied only for their livestock-related activities, as crop production was out of scope. Crop output was modelled roughly, with a standard allocation factor based on farm size, but it can be an important variable considering the importance of allocation. However, a full assessment of mixed farms would be a study on its own. Furthermore, a global leaching factor, rather than region specific, was used to quantify leaching from application of nutrients to soil. This is common practice (i.e. following IPCC guidelines) although a detailed bottom up assessment could in principle investigate leaching more in detail, taking into account parameters like distance from water bodies or rainfall. This is especially true for pastoral systems as the considered leaching factors were developed for managed grasslands and farmlands rather than natural grasslands. Finally, monetary valuation has some limitations. First of all it could also be done at country level, or even at a regional (i.e. watershed) level, but here it was chosen to apply a global monetary coefficient to guarantee comparability between snapshots in face of data scarcity (Ponsioen, 2016). Secondly, because estimation of societal costs of eutrophication vary widely in literature (Ponsioen, 2016), it is important to exercise caution when drawing policy recommendations from these results, for example by comparing valuations done with different approaches to gain insights on a specific regional context.

4.7 Conclusion

The on-farm water pollution analysis shows that water pollution costs depend on a number of factors including productivity, varying N+P inputs (both as fertiliser and as purchased feed) and national water pollution coefficients. By examining nutrient leaching per hectare it is shown that pastoralist systems have a low impact on water pollution and that the Netherlands dairy mixed feeding snapshot has a considerably higher impact than all other systems. However, looking at natural capital costs per kg of protein produced it is found that monetary damage related to nutrient leaching has sharp differences based on the relationship between feed and fertilizer use with animal productivity.

5 Land occupation quantification methodology

5.1 Introduction

The impact of animal husbandry on land is assessed by quantifying land use in terms of area, commonly known as the land footprint. As land use is not valued in monetary terms, the results of the analysis should be interpreted with care and always in relation to the impact of each land use type on biodiversity and the ecological and social context of each snapshot.

5.2 Scope and Design

The land use of a livestock production system consists of the total area used for feed production, both pasture and crops[5]. As feed is often produced in pastures or croplands that have multiple economic functions, land use has to be attributed between these services. Land use originating from other activities, like processing, is out of scope due to low materiality (see the scope section in the main report Baltussen et al. 2016).

Each snapshot represents a production system, the outputs of which are animal protein contained in livestock products. The land under consideration are grassland and cropland.

Grassland use is divided into three types: ranging where animals graze vast areas, grazing and roadside grazing. Ranging in pastoralist systems in Tanzania and India allow wild herbivores to graze next to livestock and their impact on the ecosystem is small or even positive. Grazing is done with fenced grassland in Brazil, although not very intensive, this system has already a relatively large impact on ecosystems and biodiversity. Finally, smallholders with little or no land in India, Tanzania and Indonesia partly rely on utilizing roadside grass by grazing or by cut-and-carry systems.

Cropland can provide different types of animal feed. Primary feed crops can be grown, such as feed grains, maize silage etc., but also agro industrial by products of food crops can be used as animal feed. The most important source of feed from cropland are the

[5] Land to keep animals is either the pasture or special housing such as stables, which is negligible compared to pasture.

residues annual and perennial crops as e.g. cereals, sugar cane, bananas and many others. Land use is allocated to the different types of feed, applying LCA allocation rules. The different snapshots are compared on the basis of how efficiently they produce protein in terms of land use. This depends on how intensive the system is, feed quality, the animal breed and associated productivity of animals. Animals produce more than one product in some of the systems and land use should be allocated between them. However, all results are expressed per unit of protein independent of product.

5.3 Methodology

Livestock consume feed and convert it into protein (in this case meat, milk and eggs). The rate of conversion is called the feed conversion ratio (FCR = kg feed/kg protein) and it is specific to each snapshot. It is derived from kilos of feed consumed per kilo of product produced by the animal, where it is assumed that feed is converted to protein at the same ratio as it is converted to product, independent of the type of feed.

Each type of feed requires a certain area to be produced. The area depends on the type of main crop, the region of origin and the type of feed. The type of main crop and the region of origin specify the yield of the crop, which determines the total land use. The type of feed (crop, crop residue and food processing by-product) specifies how land use is allocated between different output products of the land where the main crop is produced.

Land use (LU, m^2) in each snapshot results from the above mentioned land use, which can occur on the farm itself, but also on other locations (even other continents). The feed ration and origin is assessed for each snapshot on the basis of GLEAM databases and FAOstat information about import of feed products. Land use is calculated for all feeds on the basis of net yields and allocation rules for crop residues and agro industrial by products (AIBP). The general formula is

LU = 1/yield * 10,000.

The 10,000 is used for converting from hectares to m2.

Specifically, allocation of land use between crops and crop residues, which are produced on the same land, is done based on digestibility.

Allocation of land use to by-products from processing is based on economic value, as it is the most accepted allocation mechanism in the feed production chain.

The total land use associated with a livestock product containing protein (kg) is the product of the feed conversion rate and the average land use of the used feeds The Feed Conversion Rate is taken from the GLEAM database and calculation results.

LU total/kg protein = FCR * (average land use of 1 kg of ration)

5.4 Data and analysis

Farm level data was provided WUR (see Appendix A of the main TEEB Animal husbandry study, Baltussen et al. 2016) for each snapshot, including farm size and herd size. Herd parameters that were used include animal productivity, FCR for all livestock products and protein content in livestock products for each region.

In certain snapshots modifications were made to the methodology to accommodate for the snapshot context and data being inadequate. This is the case for the Tanzania pastoralist cattle, Tanzania backyard poultry and Netherlands dairy specialized snapshots.

In the Tanzania pastoralist cattle system, it is not feasible to estimate a farm size, as the system is transhumant and the grazing area changes over time to where there is grass. An estimate was made of the animal density (FAO, 2014).

In the Tanzania backyard poultry system, feed is composed of products found while scavenging (snails, worms, insects etc.), swill and second grade food crops. No land use is attributed to scavenging material and swill and with a low allocation factor the second grade food products, used as feed. The meat products are converted to protein content by the successive application of processing factors. Specifically, the animal productivity is defined in terms of animal live weight or the weight of the animal at slaughter. This is converted to carcass weight by applying the dressing factor and subsequently to bone free meat (BFM) by applying the BFM factors summarized in Appendix A of the TEEB Animal husbandry study (Baltussen et al. 2016).

Protein content of milk was not available in the pastoralist snapshots and the same estimate as in the mixed-dairy snapshots was used.

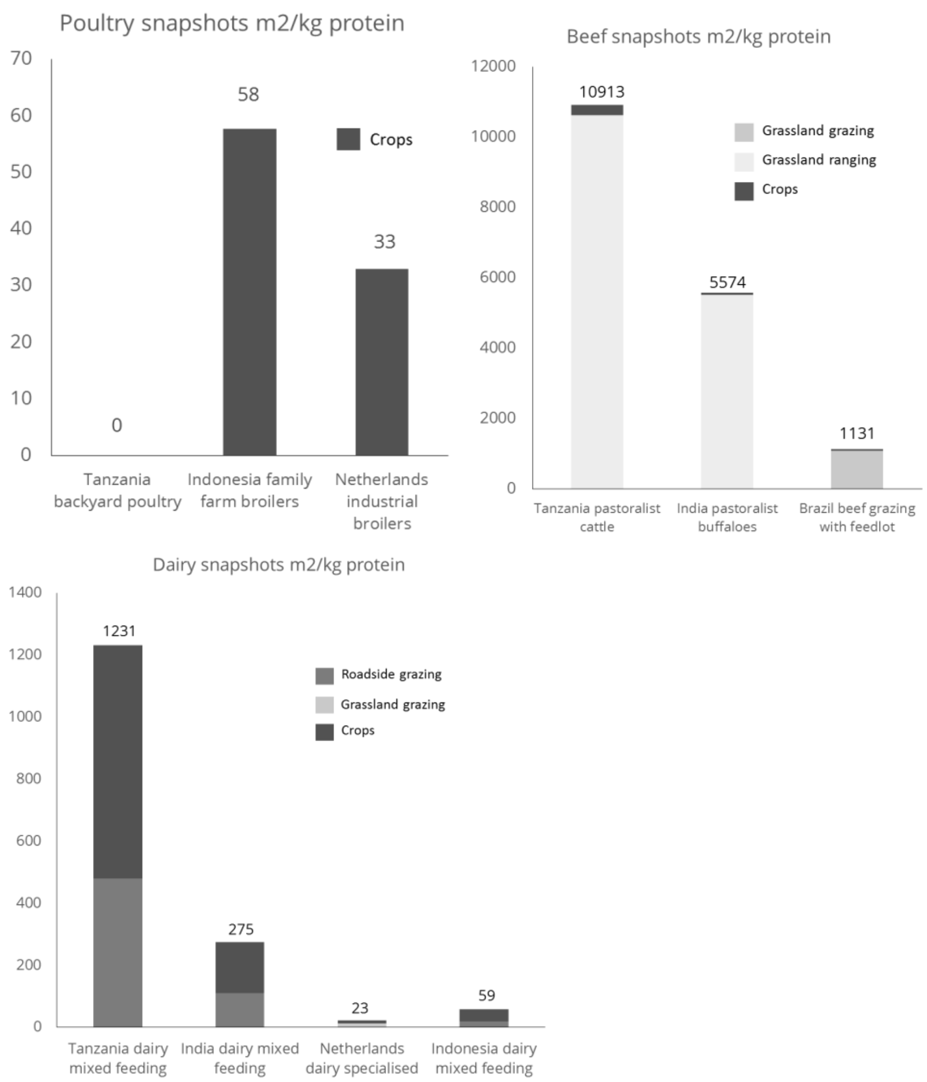
The methodology and data sources for the estimation of land use for feed production, including digestibility factors, crop yields, crop residue yields, trade matrices and by-products are summarized by WUR (Baltussen et al. 2016, Appendix A).

5.5 Results

Results are compared with literature in Table 11. The Tanzania pastoralist cattle and India pastoralist buffaloes snapshots have land occupancies much higher than the rest. This is explained by the extremely high feed conversion rate (FCR) associated with both snapshots, which reflects poor feed basis of the system: problems with availability and quality of feed, leading also to animal health problems.

A comparison of the all snapshots is shown in Fig. 10.

Figure 8. Poultry, beef and dairy snapshot comparison (m2/kg protein)



Snapshots based on grassland grazing and ranging result into much higher land-use than intensive systems Land use is zero in the case of Tanzania backyard poultry because it is economically allocated. Intensive systems do minimize land use by using feed grown from crops on- or off-farm. Animal productivity also has a large impact on land-use, as the most productive systems, Netherlands industrial broilers and Netherlands dairy specialized, have a very small footprint compared to all other snapshots. This is because of the small FCR that characterizes both systems.

Table 1. Comparison with Literature

Extensive systems	**m2/kg protein**	**Source**
Brazil beef grazing with feedlot	1131	This study
India pastoralist buffaloes	5574	This study
beef, extensive	1765	(Nijdam, et al., 2012)
beef, BR	2302	(Marieke, et al., 2011)
grass-fed steers	2318	(Ridoutt, et al., 2014)
Intensive systems, dairy	**m2/kg protein**	**Source**
Indonesia dairy mixed feeding	59	This study
Netherlands dairy specialized	23	This study
India dairy mixed feeding	275	This study
culled dairy cows	37	(Mollenhorst, et al., 2014)
milk, peat area	32	(Marieke, et al., 2011)
milk, cow, full-cream	26	(Marieke, et al., 2011)
Poultry systems	**m2/kg protein**	**Source**
Netherlands industrial broilers	33	This study
Indonesia family farm broilers	58	This study
chicken industrial	32	(Nijdam, et al., 2012)
chicken, conventional, NL	31	(Marieke, et al., 2011)
chicken, corn, NL	20	(Marieke, et al., 2011)

5.6 Discussion and Conclusion

The most land efficient systems are dairy and poultry intensive systems, while extensive systems have land demands two orders of magnitude larger. The land degradation associated with each is outside the scope of this section. This is partially studied in the case study on land valuation for the Tanzania pastoralist cattle snapshot (see Part III of this report).

The results indicate that scaling-up intensive systems to satisfy the rising demand for food is the best policy decision to reduce land occupation. However, the land occupation associated with feed crops could lead to more severe degradation of the natural environment than for extensive systems. A study that takes into account land degradation is required to compare with other impacts, such as GHG, and reliably answer the policy question.

6 References

6.1 Retail prices references

Centraal Bureau voor de Statistiek 2014. Retrieved from: http://statline.cbs.nl/StatWeb/publication/?DM=SLNL&PA=80346NED

Elinaza A., *Tanzania Daily News 2015. Tanzania: Beef Prices Down By 20 Per Cent 31 March 2015*.
Available at: http://allafrica.com/stories/201503310164.html

Instituto de Economia Agricola, 2015. Precos Medios Mensais no Varejo [Online].
Available at: http://ciagri.iea.sp.gov.br/nia1/precos_medios.aspx?cod_sis=4

Kadigi et al., Academia Journal of Agricultural Research 1(8): 145-155, 2013. Retrieved from:
http://www.academia.edu/8037416/Value_chain_of_indigenous_cattle_and_beef_products_in_Mwanza_region_Tanzania_Market_access_linkages_and_opportunities_for_upgrading

Meytha W. CEIC 2013. *Import Curbs Raise Food Prices*. Available at:
http://www.ceicdata.com/en/blog/import-curbs-raise-food-prices

Ministry of Statistics & Programme Implementation, Government of India, 2012. *Prices and Rural Wages in India, October 2011 to December 2011.* Retrieved from:
http://mospi.nic.in/mospi_new/upload/price-wage-rur-india-oct11-dec11.pdf

6.2 GHG references

Baltussen W. , A. de Blaeij, P. van Horne , T. Achterbosch , G. Kruseman, E.J.M.M. Arets, T. Vellinga, S.J. Hiemstra, W. Ouweltjes , V. Fobelets, O. A. Karachalios , L. Verkooijen , A. De Groot Ruiz, P. Galgani, N. Erlenborn, M. Tarin Robles , R. Lord, R. Hardwicke (2015) *Valuation of livestock eco-agri-food systems: poultry, beef and dairy* , Wageningen University and Research center, Trucost & True Price, The Hague

Gerber, P.J., Steinfeld, H., Henderson, B., Mottet, A., Opio, C., Dijkman, J., Falcucci, A. and Tempio, G., 2013. *Tackling climate change through livestock – A global assessment of emissions and mitigation*. [pdf] Rome: Food and Agriculture Organization of the United Nations (FAO). Available at: <http://www.fao.org/docrep/018/i3437e/i3437e.pdf> [Accessed 7 April 2015].

Heinz, G. and Hautzinger, P., 2007. *Meat processing technology for small- to medium-scale producers*. [pdf] Bangkok: Food and Agriculture Organization of the United Nations (FAO). Available at: <http://www.fao.org/3/a-ai407e.pdf> [Accessed 7 April 2015].

Lukmanji, Z. and Hertzmark, E. eds., 2008. *Tanzania food composition Tables*. [pdf] Dar es Salaam: MUHAS - TFNC, HSPH. Available at: <http://cdn1.sph.harvard.edu/wp-content/uploads/sites/30/2012/10/tanzania-food-composition-tables.pdf> [Accessed 7 April 2015].

Moore, F. and Diaz, D., 2015. Temperature impacts on economic growth warrant stringent mitigation policy. *Nature Climate Change (5), pp. 127-131*. [online] Available at: <http://www.nature.com/nclimate/journal/v5/n2/full/nclimate2481.html> [Accessed 7 April 2015].

Steen, B., 1999. A systematic approach to environmental priority strategies in product development (EPS). Version 2000 – Models and data of the default method. [pdf] Göteborg: Centre for Environmental Assessment of Products and Material Systems. Available at: <http://msl1.mit.edu/esd123_2001/pdfs/EPS1999_5.pdf> [Accessed 7 April 2015].

Stern, N., 2006. *The Stern Review: The Economics of Climate Change*. [pdf] London: HM Treasury. Available at: <http://mudancasclimaticas.cptec.inpe.br/~rmclima/pdfs/destaques/sternreview_report_complete.pdf> [Accessed 7 April 2015].

Suriani, N. W., Purnomo, H., Estiasih T. and Suwetja, I.K., 2014. Physicochemical Properties, Fatty Acids Profile and Cholesterol Content of Indigenous Manado Chicken, Broiler and Spent Hen Meat. [pdf] *International Journal of ChemTech Research, 6(7), 3896-3902*. Available at: <http://sphinxsai.com/2014/ch_vol_6_7/7/%283896-3902%29O14.pdf> [Accessed 7 April 2015].

Tol, R., 2008. *The Social Cost of Carbon: Trends, Outliers and Catastrophes*. Economics: The Open-Access, Open-Assessment E-Journal. [online] Available at: <http://www.economics-ejournal.org/economics/journalarticles/2008-25> [Accessed 7 April 2015].

US IAWG, 2013. Technical Support Document: Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866. [pdf] Washington, DC: Interagency Working Group on Social Cost of Carbon, United States Government. Available at: <https://www.whitehouse.gov/sites/default/files/omb/inforeg/social_cost_of_carbon_for_ria_2013_update.pdf> [Accessed 7 April 2015].

Westhoek, H., Rood, T., van den Berg, M., Janse, J., Nijdam, D., Reudink, M. and Stehfest, E., 2011. *The Protein Puzzle - The consumption and production of meat, dairy and fish in the European Union*. [pdf] The Hague: PBL Netherlands Environmental Assessment Agency. Available at: <http://www.fao.org/fileadmin/user_upload/animalwelfare/Protein_Puzzle_web_1.pdf> [Accessed 7 April 2015].

6.3 Water dependency references

AG Decision Maker, 2014. *Historical Corn Yields by County*. [pdf] Ames: Iowa State University of Science and Technology. Available at: <http://www.extension.iastate.edu/agdm/crops/pdf/a1-12.pdf> [Accessed 0 3 March 2015].

AG Decision Maker, 2015. *Iowa Cash Corn and Soybean Prices*. [pdf] Ames, IA: Iowa State University of Science and Technology. Available at: <http://www.extension.iastate.edu/agdm/crops/pdf/a2-11.pdf> [Accessed 03 March 2015].

Aggarwal, P. K., Hebbar, K. B., Venugopalan, M. V., Rani, S., Bala, A., Biswal, A. & Wani, S. P., 2008. *Quantification of Yield Gaps in Rain-fed Rice, Wheat, Cotton and Mustard in India*. Global Theme on Agroecosystems, Report no. 43. [online] Available at: http://www.indiaenvironmentportal.org.in/files/ICRISATReport43.pdf [Accessed 03 March 2015].

Agriculture and Agri-Food Canada, 2015. *Weekly Price Summary*. [online] (Last update: 2 April 2015). Available at: <http://www.agr.gc.ca/eng/industry-markets-and-trade/statistics-and-market-information/by-product-sector/crops/crops-market-information-canadian-industry/weekly-price-summary/?id=1378745200250> [Accessed 03 March 2015].

Arena, A. P., Piastrellini, R., Civit, B., ca. 2009. *Water footprint of soybean production in Argentina*. [pdf] Argentinian Universidad Technologica Nacional, Consejo Nacional de Investigaciones Cientificas y Tecnicas.

AWB, 2015. AWB Site Based Prices. [online] Available at: <http://www.awb.com.au/growers/awbgrainprices/dailygrainprices/> [Accessed 03 March 2015].

Bandyopadhyay, P. K., Mallick, S., 2003. *Effect of soil moisture levels on root distribution, water uptake and crop coefficient of winter maize in a humid tropic region*. Food, Agriculture & Environment. [online] Available at:

<https://www.academia.edu/3444249/Effect_of_soil_moisture_levels_on_root_distribution_water_uptake_and_crop_coefficient_of_winter_maize_in_a_humid_tropic_region> [Accessed 03 March 2015].

Berbel et. Al 2011. *Value of irrigation water in Guadalquivir Basin (Spain) by Residual Value Method*. Water Resources Management 25. p. 1565-1579. [online] Available at: <http://link.springer.com/article/10.1007%2Fs11269-010-9761-2> [Accessed 03 March 2015].

Bhaskar, K. S., Rao, M. R. K., Mendhe, P. N., Suryavanshi, M. R., ca 2004. *Micro irrigation management in cotton*. [pdf] Central Institute for Cotton Research Nagpur. Available at: <http://www.cicr.org.in/pdf/micro_irrigation.pdf> [Accessed 03 March 2015].

Chapagain, A. K., Hoekstra, A. Y., Savenije, H. H. G., Gautam, R., 2005. *The water footprint of cotton consumption*. [pdf] UNESCO-IHE Available at: <http://www.waterfootprint.org/Reports/Report18.pdf> [Accessed 03 March 2015].

Chouhan et. al 2014. *Maximizing water productivity and yields of wheat based on drip irrigation systems in clay loam soil*. [pdf] International Journal of Engineering Research & Technology. Vol. 3 Issue 7. Available at: <http://www.ijert.org/view-pdf/10518/maximizing-water-productivity-and-yields-of-wheat-based-on-drip-irrigation-systems-in-clay-loam-soil> [Accessed 03 March 2015].

Crozier, T., 2012. *Yield correlations for major field crops in Alberta*. [pdf] Yield Alberta 2012. Available at: <http://www.afsc.ca/doc.aspx?id=5576> [Accessed 03 March 2015].

Department of Primary Industries, 2013. Surface Irrigated Sorghum (diesel pump from bore). [pdf] Sydney: NSW Government. Available at: <http://www.dpi.nsw.gov.au/__data/assets/pdf_file/0011/175916/Irrigated-sorghum-12-13.pdf> [Accessed 03 March 2015].

Duttarganvi, S., Tirupataiah, K., Yella Reddy, K., SAndhyrani, K., Mahendra Kumar, R., Kadasiddappa, M., 2014. *Yield and water productivity of rice under different cultivation practices and irrigation regimes*. International Symposium on Integrated Water Resources Management. Available at: <https://www.academia.edu/6277250/Yield_and_Water_Productivity_of_Rice_under_Diffe

rent_Cultivation_Practices_and_Irrigation_Regimes_INTRODUCTION> [Accessed 03 March 2015].

Econoagro, 2013. Agricultura 2013/2014: comparacion de resultados y rentas entre cultivos y modalidades de produccion. Econoagro. [online] Available at: <http://www.econoagro.com/555-agricultura-2013/2014-comparacion-de-resultados-y-rentas-entre-cultivos-y-modalidades-de-produccion> [Accessed 04 March 2015].

EMCB-ENVIS, 2013. *Annual average rainfall by districts in Punjab (in mm)*. Chandigarh: Punjab State Council for Science & Technology. Available at: <http://punenvis.nic.in/index3.aspx?sslid=3870&subsublinkid=3599&langid=1&mid=1> [Accessed 03 March 2015].

Facon, T., 2000. *Water management in rice in Asia: Some issues for the future*. Bangkok: FAO. Available at: < http://www.fao.org/docrep/003/x6905e/x6905e0g.htm> [Accessed 03 March 2015].

FAO, 2004. Economic valuation of water resources in agriculture: From the sectoral to a functional perspective of natural resource management. [pdf] FAO Water Reports 27 Available at: < ftp://ftp.fao.org/agl/aglw/docs/wr27e.pdf> [Accessed 03 March 2015].

FAO, 2014. *FAOSTAT*. [online] Available at: <http://faostat.fao.org/site/703/DesktopDefault.aspx?PageID=703#ancor> [Accessed 03 March 2015].

Farrell, R., 2015. *Australia: Grain and feed update*. USDA Foreign Agriculture Service. [online] Available at: <http://gain.fas.usda.gov/Recent%20GAIN%20Publications/Grain%20and%20Feed%20Update_Canberra_Australia_1-27-2015.pdf> [Accessed 03 March 2015].

Fischer, R. A., Byerlee, D., & Edmeades, G. O., 2014. *Crop yields and global food security: will yield increase continue to feed the world?*. Canberra: Australian Centre for International Agricultural Research. [online] Available at: <http://aciar.gov.au/publication/mn158> [Accessed 03 March 2015].

Fulton, J., Cooley, H., Gleick, P.H., 2014. Water Footprint Outcomes and Policy Relevance Change with Scale Considered: Evidence from California. Water Resources Management.

Volume 28, Issue 11. pp 2627-3649. [online] Available at: <http://link.springer.com/article/10.1007%2Fs11269-014-0692-1> [Accessed April 6 2015]

Ghida Daza, C., 2013. Evaluación económica de maíz. Campaña 2012/13. [pdf] *EEA INTA Marcos Juarez*.

Global Yield Gap Atlas, 2011. Argentina: Description of cropping systems, climate, and soils. Global Yield Gap Atlas [online] Available at: <http://www.yieldgap.org/argentina> [Accessed 03 March 2015].

Hellegers, P. & Davidson, B., 2010*. Determining the disaggregated economic value of irrigation water in the Musi sub-basin in India*. [pdf] Agricultural Water Management 97. p. 933-98. Available at: <http://www.sciencedirect.com/science/article/pii/S0378377410000442> [Accessed 03 March 2015].

Hoekstra, A. Y., 2009. *Human appropriation of natural capital: a comparison of ecological footprint and water footprint analysis*. [pdf] Ecological economics 68 (2009) p. 1963-1974. Available at: < http://www.waterfootprint.org/Reports/Hoekstra2008-Ecological-versus-WaterFootprint.pdf> [Accessed 03 March 2015].

Hoekstra, A.Y. and Mekonnen, M.M. (2012) From water footprint assessment to policy, Proceedings of the National Academy of Sciences, 109(22): E1425. Available at: <http://www.pnas.org/content/109/22/E1425.full> [Accessed 6 April 2015]

Index Mundi, 2015. *Soybeans Monthly Price - US Dollars per Metric Ton*. [online] Available at: <http://www.indexmundi.com/commodities/?commodity=soybeans&months=12> [Accessed 16 February 2015].

Infocampo.com.ar, 2013. *Arrendamientos: Afirman que la caída en los precios beneficiará a la producción*. Infocampo.com.ar. [online] Available at: <http://infocampo.com.ar/nota/campo/44379/arrendamientos-afirman-que-la-caida-en-los-precios-beneficiara-a-la-produccion> [Accessed 04 March 2015].

Infobae, 2014. *Los costos de los alquileres y la caída de precios de granos afectarán el nuevo año agrícola*. [online] Available at: <http://www.infobae.com/2014/08/22/1589532-los-

costos-los-alquileres-y-la-caida-precios-granos-afectaran-el-nuevo-ano-agricola> [Accessed 04 March 2015].

Informa Economics, 2014. Argentina February Production Update and Weekly Crop Condition Report. [pdf] Informa Economics South American Crop Reporting Service. Available at: < http://www.informaecon.com/SampleReports/ArgentinaSampleReport.pdf> [Accessed 03 March 2015].

Iowa State University: extension and outreach, 2015. *Metric conversions, Iowa State University*. [online] Available at: <http://www.extension.iastate.edu/agdm/wholefarm/html/c6-80.html> [Accessed 03 March 2015].

Jalota, S. K., Sood, A., Vitale, J. D. & Srinivasan R., 2007. Simulated crop yields response to irrigation water and economic analysis: increasing irrigated water use efficiency in the Indian Punjab. Agronomy Journal, vol. 99. [online] Available at: < http://ssl.tamu.edu/media/11491/cropyield-punjab-epic.pdf#page=1&zoom=auto,-109,797> [Accessed 03 March 2015].

Kansas Agricultural Statistics, 2010. *2009 Kansas Soybean County Estimates*. [pdf] Topeka, KS: Kansas Department of Agriculture. Available at: <http://www.nass.usda.gov/Statistics_by_State/Kansas/Publications/County_Estimates/Coest/soybean09.pdf> [Accessed 03 March 2015].

Kansas Agricultural Statistics, 2012. *2011 Kansas Wheat County Estimates*. [pdf] Topeka, KS: Kansas Department of Agriculture. Available at: <http://www.nass.usda.gov/Statistics_by_State/Kansas/Publications/County_Estimates/Coest/wheat11> [Accessed 03 March 2015].

Kansas State University Department of Agricultural Economics, 2014. *Grain Sorghum Cost-Return Budget in Southeast Kansas*. [pdf] Manhattan, KS: Kansas State University. Available at: <http://www.ksre.ksu.edu/bookstore/pubs/mf995.pdf> [Accessed 03 March 2015].

KFMA, 2011. *Will wheat be profitable in 2012?*. Abilene, KS: Kansas Farm Management Association Volume 5, Issue 9. [online]. Available at:

<http://www.agmanager.info/kfma/Newsletters/September_11.pdf> [Accessed 03 March 2015].

Klocke, N. and Currie, R., 2009. *Corn and grain sorghum production with limited irrigation*. In: CPIA, the Annual Central Plains Irrigation Conference. [online] Colby, Kansas, February 24-25, 2009. Colby, KS: CPIA. Available at: <https://www.ksre.ksu.edu/irrigate/OOW/P09/Klocke%2009CS.pdf> [Accessed 03 March 2015].

Krishi Jagran, 2014. *All India area, production and yield of maize under irrigation*. Krishi Jargan. [online] Available at: <http://www.krishijagran.com/farm-data/farm-statistics/2014/12/All-India-Area-Production-and-Yield-of-Major-Crops/All-India-Area-Production-and-Yield-of-Maize-under-irrigation> [Accessed 03 March 2015].

Kumar, S., 2008. *Raising wheat production by addressing supply-side constraints in India.* [pdf] New Delhi: National Centre for Agricultural Economics and Policy Research. Available at: <http://www.ncap.res.in/upload_files/PME_notes/pmenotes16.pdf> [Accessed 03 March 2015].

Lagos, J. E., Singh, S. K., 2015. *India: Grain and Feed Annual*. [pdf] USDA. Available at: <http://gain.fas.usda.gov/Recent%20GAIN%20Publications/Grain%20and%20Feed%20Annual_New%20Delhi_India_2-24-2015.pdf> [Accessed 03 March 2015].

Lloveras, J. et al. eds., 2006. *Sustainable Grassland Productivity*. [pdf] 21st General Meeting of the European Grassland Federation. Badajoz, Spain, 3-6 April 2006. Zurich: European Grassland Federation. Available at: <http://www.europeangrassland.org/fileadmin/media/EGF2006_GSE_vol11.pdf> [Accessed 03 March 2015].

Mekonnen, M. M. and Hoekstra, A. Y., 2010. *The green, blue and grey water footprint of crops and derived crop products, Value of Water Research Report Series No. 47*. [pdf] UNESCO-IHE, Delft, the Netherlands. Vol.1 & Vol.2 Available at: < http://www.waterfootprint.org/Reports/Report47-WaterFootprintCrops-Vol1.pdf> [Accessed 03 March 2015].

Mekonnen, M. M. and Hoekstra, A. Y., 2010. *The green, blue and grey water footprint of farm animals and animal products, Value of Water Research Report Series No. 48*. [pdf]

UNESCO-IHE, Delft, the Netherlands. Vol.1 & Vol.2 Available at: <http://www.waterfootprint.org/Reports/Report-48-WaterFootprint-AnimalProducts-Vol1.pdf> [Accessed 03 March 2015].

Mekonnen, M. M., Hoekstra, A. Y., 2011. National water footprint accounts: The green, blue and grey water footprint of production and consumption, Value of Water Research Report Series No. 50. [pdf] UNESCO-IHE. Available at: <http://www.waterfootprint.org/Reports/Report50-NationalWaterFootprints-Vol1.pdf> [Accessed 03 March 2015].

Mercopress, 2014. *Cosecha argentina de soja y maíz alcanzará más de 80 millones de toneladas*. Mercopress. [online] Available at: <http://es.mercopress.com/2014/06/23/cosecha-argentina-de-soja-y-maiz-alcanzara-mas-de-80-millones-de-toneladas> [Accessed 03 March 2015].

Ministry of Agriculture, Food and Rural Affairs, 2015. *2015 Field Crop Budgets Publication 60*. [pdf] Toronto: Ministry of Agriculture, Food and Rural Affairs. Available at: <http://www.omafra.gov.on.ca/english/busdev/facts/pub60.pdf> [Accessed 03 March 2015].

Mishra, H., Rathore, T., Savita, U., 2001. *Water-use efficiency of irrigated winter maize under cool weather conditions of India.* Irrigation Science. [online] Available at: <http://link.springer.com/article/10.1007%2Fs002710100037> [Accessed 03 March 2015].

National Geographic, 2015. *Corn Remains King in USDA Irrigation Survey*. [online] Available at: <http://voices.nationalgeographic.com/2015/02/10/corn-remains-king-in-usda-irrigation-survey> [Accessed 03 March 2015].

Navadka,r D. S., Amale A. J., Gulave C. M. & Nannaware V. M., 2012. *Economics of Production and Marketing of Kharif Maize in Ahmednagar District of Maharashtra State*. [pdf] Rahuri: Department of Agricultural Economics Mahatma Phule Krishi Vidyapeeth. Available at: <http://eands.dacnet.nic.in/Publication12-12-2012/4605-september12/4605-1.pdf> [Accessed 03 March 2015].

Netafim, n.d. *Cotton in India*. Netafim. [online] Available at: <http://www.netafim.com/story/cotton-in-india> [Accessed 03 March 2015].

NSW, 2013. Surface irrigated sorghum (diesel pump from bore). [pdf] NSW. Available at: <http://www.dpi.nsw.gov.au/__data/assets/pdf_file/0011/175916/Irrigated-sorghum-12-13.pdf> [Accessed 05 March 2015].

Pahlow, M. and Mekonnen, M. M., 2012. Using the water footprint as a tool for sustainable appropriation of freshwater resources. John Hopkins Water Institute. [online] Available at: <http://water.jhu.edu/magazine/using-the-water-footprint-as-a-tool-for-sustainable-appropriation-of-freshw/> [Accessed 6 April 2015].

Philip, T., Peake, A., and McLean, G., 2010. *Improving Profitability of Irrigated Sorghum through Partial Irrigation of Larger Areas.* Edited paper. In: Proceedings of the 1st Australian Summer Grains Conference [online] Gold Coast, Australia, June 21-24 2010. Available at: <https://www.grdc.com.au/uploads/documents/2010ASGCEditedPapersPDF/Philp_Irrigation_edited_paper.pdf> [Accessed 03 March 2015].

Plastina, A., 2015. *Estimated Costs of Crop Production in Iowa - 2015*. [pdf] Ames, IA: Iowa State University of Science and Technology. Available at: <http://www.extension.iastate.edu/agdm/crops/pdf/a1-20.pdf> [Accessed 03 March 2015].

Rajaskekhara Rao, B. K., Sahrawat, K. L., Wani, S. P., Pardhasaradhy, G., 2010. *Integrated nutrient management to enhance on-farm productivity of rain fed maize in India*. [pdf] International Journal of Soil Science. Available at: <http://oar.icrisat.org/127/1/IJSS5_4_216-225_2010.pdf> [Accessed 03 March 2015].

Ramana Murthy, R. V., Misra, R., 2012. *Pricing of Paddy: A Case Study of Andhra Pradesh*. Mumbai: Reserve Bank of India. [online] Available at: <http://www.rbi.org.in/scripts/PublicationsView.aspx?id=14604> [Accessed 03 March 2015].

Ramasundaram, P., Gajbhiye, H., 2001. *Constraints to cotton production in India*. [pdf] Central Institute for Cotton Research Nagpur. Available at: <http://www.cicr.org.in/pdf/constraint_cotton.pdf> [Accessed 03 March 2015].

Roulet, N., 2013. *Maíz: "El precio del maíz en Argentina es más alto que en Chicago".* Agrositio. [online] Available at:

<http://www.agrositio.com/vertext/vertext.asp?id=150174&se=14> [Accessed 04 March 2015].

Scoot, F., 2012. *Irrigated northern winter crop gross margins 2012: Durum wheat (surface irrigated)*, NSW Department of Primary Industries. [online] Available at: <http://www.dpi.nsw.gov.au/agriculture/farm-business/budgets/winter-crops> [Accessed 03 March 2015].

Sharma, B. R., Rao, K. V., Vittal, K. P. R., Amarasinghe, U. A., ca 2007. *Converting rain into grain: Opportunites for realizing the potential of rain-fed agriculture in India*. International Rainwater Harvesting Alliance. [online] Available at: < http://www.irha-h2o.org/?wpfb_dl=353> [Accessed 03 March 2015].

Shirazi, S. M., Yusop, Z., Zardari. N. H. & Ismail, Z., 2014. *Effect of Irrigation Regimes and Nitrogen Levels on the Growth and Yield of Wheat*. Advances in Agriculture. [e-journal] 2014 (6). Available at: <http://dx.doi.org/10.1155/2014/250874> [Accessed 03 March 2015].

Siddiq, E. A., 2000. *Bridging the rice gap in India*. Bangkok: FAO. [online] Available at: < http://www.fao.org/docrep/003/x6905e/x6905e09.htm#TopOfPage> [Accessed 03 March 2015].

Smith, D. J. et al., 2009. Managing stored soil moisture and irrigation frequency to benefit irrigated winter cereals. In: Irrigation Australia Limited (IAL), the Irrigation and Drainage Conference. Swan Hill, Vic, Australia, 18-21 October 2009. Mascot, NSW: Irrigation Australia Limited. [online] Available at: <http://irrigation.org.au/documents/publications-resources/conference_papers_2009/201009_Session11_DavidSmith2.pdf> [Accessed 03 March 2015].

Singh, V. P. & Singh, R. K., 2000. *Rainfed Rice: A Sourcebook of Best Practices and Strategies in Eastern India*. [pdf] Los Baños: International Rice Research Institute. Available at: http://books.irri.org/8186789022_content.pdf [Accessed 03 March 2015].

Singh, S. K., 2013. *India: Grain and Feed Annual*. [pdf] USDA. Available at: <http://gain.fas.usda.gov/Recent%20GAIN%20Publications/Grain%20and%20Feed%20Annual_New%20Delhi_India_2-15-2013.pdf> [Accessed 03 March 2015].

Singh, J., Babar, S., Abraham, S., Venugopalan, M. V., Majumdar, G., 2012. *Fertilization of high density, rainfed cotton grown on vertisols of India*. Better Crops. [online] Available at:

<http://www.ipni.net/publication/bettercrops.nsf/0/1CD4CC8EECD05C9606257A0900610914/$FILE/Better%20Crops%202012-2%20p.26.pdf> [Accessed 03 March 2015].

Sood, D., 2014. *Cotton and products update: Arrivals slow and with lower prices… MSP operations underway*. [pdf] New Delhi: United States Department of Agriculture. Available at: <http://gain.fas.usda.gov/Recent%20GAIN%20Publications/Cotton%20and%20Products%20Update_New%20Delhi_India_12-1-2014.pdf> [Accessed 03 March 2015].

Sood, D., 2015. *Cotton and products update December 2014*. [pdf] New Delhi: United States Department of Agriculture. Available at: <http://gain.fas.usda.gov/Recent%20GAIN%20Publications/Cotton%20and%20Products%20Update%20December%202014_New%20Delhi_India_1-6-2015.pdf> [Accessed 03 March 2015].

Speelman et. al, 2011. *Variability in smallholders' irrigation water values: study in North-West Province, South Africa.* [pdf] Irrigation and Drainage 60. p. 11-19. Available at: < http://hdl.handle.net/1854/LU-781915> [Accessed 03 March 2015}.

Statistics Canada, 2011. *Irrigation volume per hectare by crop type, 2010 — Province or region.* [online] (Last update: 10 April 2013). Available at: <http://www.statcan.gc.ca/pub/16-402-x/2011001/t005-eng.htm> [Accessed 03 March 2015].

Teixeira, R., Domingos, T., Simoes, A., Rodrigues, O., ca. 2007. *Local vs. global grain maize production:* Where should you get your maize from?. [pdf] Portugal: Instituto Superior Tecnico.

Thompson, C. & Johnson, B., 2012. *The Value of Water in Agricultural Land Markets: The Nebraska Case. Journal of the ASFMRA*. [pdf] University of Nebraska, Department of Agriculture Economics. Available at: <http://ageconsearch.umn.edu/bitstream/161610/2/362_Thompson.pdf> [Accessed 03 March 2015].

United Nations, 2014. *System of Environmental-Economic Accounting 2012 – Central Framework* [pdf] United Nations. European Union. Food and Agriculture Organization of the United Nations. International Monetary Fund. Organization for Economic Cooperation and Development. The World Bank. New York, 2014. Available at: <http://unstats.un.org/unsd/envaccounting/seeaRev/SEEA_CF_Final_en.pdf> [Accessed 03 March 2015].

University of Nebraska Lincoln, 2013. *Sorghum Irrigation and Water Information*. [online] Available at: <http://cropwatch.unl.edu/sorghum/water> [Accessed 03 March 2015].

USDA, 2014. Kansas Wheat History. [pdf] Washington, D.C.: United States Department of Agriculture. Available at: <http://www.nass.usda.gov/Statistics_by_State/Kansas/Publications/Crops/whthist.pdf> [Accessed 03 March 2015].

USDA National Agricultural Statistics Service, 2012. *2012 Census of Agriculture - Iowa State Data*. [pdf] Washington, D.C.: United States Department of Agriculture. Available at: <http://www.agcensus.usda.gov/Publications/2012/Full_Report/Volume_1,_Chapter_1_State_Level/Iowa/st19_1_034_036.pdf> [Accessed 03 March 2015].

USDA National Agricultural Statistics Service, 2014. *Iowa AgriNews Vol. 14-26*. [pdf] Washington, D.C.: United States Department of Agriculture. Available at: <http://www.nass.usda.gov/Statistics_by_State/Iowa/Publications/Agri-News/2014/AgriNews26_2014.pdf> [Accessed 03 March 2015].

USDA Foreign Agriculture Service, 2015. *Cotton: World markets and trade*. [pdf] Available at: <http://apps.fas.usda.gov/psdonline/circulars/cotton.pdf> [Accessed 03 March 2015].

World Bank, 2013. Cereal yield (kg per hectare). [online] Available at: <http://data.worldbank.org/indicator/AG.YLD.CREL.KG> [Accessed 03 March 2015].

WWF, 2009. *Water footprint of rice: Quantifying the rainbow of virtual water fluxes related to rice trade*. [pdf] Ashok Chapagain, WWF – UK. Available at: http://www.rac.es/ficheros/doc/00740.pdf [Accessed 03 March 2015].

XE, 2011. *Current and Historical Rate Tables*. [online] Available at: http://www.xe.com/currencytables/?from=INR&date=2011-01-02 [Accessed 03 March 2015].

Ziolkowska, J., 2015. *Shadow price of water for irrigation – A case of the High Plains.* [pdf] Agricultural Water Management 153. p. 20-31 Available at:

<http://www.sciencedirect.com/science/article/pii/S0378377415000426> [Accessed 03 March 2015].

6.4 Water pollution references

Baltussen W. , A. de Blaeij, P. van Horne , T. Achterbosch , G. Kruseman, E.J.M.M. Arets, T. Vellinga, S.J. Hiemstra, W. Ouweltjes , V. Fobelets, O. A. Karachalios , L. Verkooijen , A. De Groot Ruiz, P. Galgani, N. Erlenborn, M. Tarin Robles , R. Lord, R. Hardwicke (2015) Valuation of livestock eco-agri-food systems: poultry, beef and dairy , Wageningen University and Research center, Trucost & True Price, The Hague

IPCC (2006) IPCC Guidelines for national greenhouse gas inventories. Intergovernmental Panel on Climate Change.

FAO. 2007. Gridded livestock of the world 2007, by G.R.W. Wint and T.P. Robinson. Rome, pp 131.

Robinson, T.P., Thornton P.K., Franceschini, G., Kruska, R.L., Chiozza, F., Notenbaert, A., Cecchi, G., Herrero, M., Epprecht, M., Fritz, S., You, L., Conchedda, G. & See, L. 2011. Global livestock production systems. Rome, Food and Agriculture Organization of the United Nations (FAO) and International Livestock Research Institute (ILRI), 152 pp.

MacLeod, M., Gerber, P., Mottet, A., Tempio, G., Falcucci, A., Opio, C., Vellinga, T., Henderson, B. & Steinfeld, H. 2013. *Greenhouse gas emissions from pig and chicken supply chains – A global life cycle assessment.* Food and Agriculture Organization of the United Nations (FAO), Rome.

Opio, C., Gerber, P., Mottet, A., Falcucci, A., Tempio, G., MacLeod, M., Vellinga, T., Henderson, B. & Steinfeld, H. 2013. *Greenhouse gas emissions from ruminant supply chains – A global life cycle assessment.* Food and Agriculture Organization of the United Nations (FAO), Rome.

Ponsioen, T. 2016. *Monetary valuation methods in life cycle impact assessment – focus on eutrophication.* LEI Working paper, LEI, Wageningen UR.

Weidema, B. P. 2009. *Using the budget constraint to monetarise impact assessment results.* Ecological Economics, 68 (6), p. 1591-1598.

6.5 Land occupation references

Baltussen W. , A. de Blaeij, P. van Horne , T. Achterbosch , G. Kruseman, E.J.M.M. Arets, T. Vellinga, S.J. Hiemstra, W. Ouweltjes , V. Fobelets, O. A. Karachalios , L. Verkooijen , A. De Groot Ruiz, P. Galgani, N. Erlenborn, M. Tarin Robles , R. Lord, R. Hardwicke (2015) Valuation of livestock eco-agri-food systems: poultry, beef and dairy TEEBAgFood Feeder Rerport. Wageningen University and Research center, Trucost & True Price, The Hague

FAO (2014) Global Livestock Production and Health Atlas (GLIPHA), available at http://kids.fao.org/glipha/

Marieke, H., Sevenster, M. & Croezen, H., 2011. *Life Cycle Impacts of Protein-rich Foods for Superwijzer,* Delft: CE Delft.

Mollenhorst, H. et al., 2014. *A novel approach to assess efficiency of land use by livestock to produce human food.* San Francisco, Proceedings of the 9th International Conference on Life Cycle Assessments in the Agri-Food Sector.

Nijdam, D., Rood, T. & Westhoek, H., 2012. The price of protein: Review of land use and carbon footprints from life cycle assessments of animal food products and their substitutes. *Food Policy,* 37(6), pp. 760-770.

Ridoutt, B. et al., 2014. Carbon, water and land use footprints of beef cattle production systems in southern Australia. *Journal of Cleaner Production,* Volume 73, pp. 24-30.

Part III – Landscape level valuation: Maasai pastoralism in Tanzania

Abstract

Part III of this report explains the methodology used in the valuation of natural capital in the Maasai Steppe in Tanzania, as presented in chapter 5 of the TEEB Animal Husbandry study. In order to quantify the loss of natural capital caused by land degradation, a model is built to value ecosystem services under different land conversion scenarios. The annual value of ecosystem services in the region is calculated using primary valuations and value transfer. The aggregated value is then extrapolated into the future. The value per hectare of ecosystem benefits in rangeland is compared with that of farmland and national parks, using an attribution approach that allows the value added by ecosystems to be separated from that added by human labour and other inputs. Additionally, alternative future scenarios are developed to take into account ecosystem changes that have an impact on ecosystem services in the long run. In each scenario land conversion is assumed to happen at a different pace with different consequences for biodiversity and the value of ecosystem services. The results show that farming creates a higher short-term value at the cost of natural capital in the long-term. Specifically, soil degradation and the negative effects on tourism and pastoral livelihoods offset the additional value created by agriculture. A sensitivity analysis of the model shows which key assumptions influence the results the most. Finally, the value of ecosystems, calculated only for local beneficiaries, is compared with the value of carbon stocks in the region which have value to the global community. The value of carbon stocks is shown to be almost five times higher, highlighting the fact that global community has an important stake in land conversion dynamics in the Maasai steppe.

1 Introduction

The Maasai Steppe region in northern Tanzania is an area characterized by rich biodiversity, containing two of the most visited national parks in Tanzania[6] (Sekar, Weiss, & Dobson, 2014),. It also sustains the pastoralist livelihoods of indigenous Maasai communities, who largely rely on herd mobility to find grazing areas in a resource-scarce region (Homewood, Kristjanson, & Trench, 2002).

The rapid growth of farmland has been identified as a threat to the pastoralist livelihoods and biodiversity (FAO, 2009; Kshatriya, 2007). A comprehensive valuation of each ecosystem provides an insight into the hidden costs and benefits of the evolving dynamics in the area.

In this study, the key trade-off between extracting more value from expanding one ecosystem (land converted for agriculture) and preserving other types of ecosystems (national parks and pastoralist rangelands) is assessed quantitatively.

The purpose of this research is to quantify the benefits that ecosystems provide to local stakeholders in the Maasai Steppe, and to understand the costs and benefits of the ongoing conversion of grasslands and woodlands into agricultural land. A model that links the value of ecosystems to land conversion is used. Based on the definitions presented in Part I, the value of ecosystems for local stakeholders is defined as Internal Natural Capital[7].

The key research question of this study is: What is the Internal Natural Capital value of the Maasai Steppe region, and how is land conversion to farmland affecting that value?

Part III has the following chapters. Chapter 2 describes the region under study in terms of ecosystem characteristics, local livelihoods and policy issues and defines the scope of the land valuation in terms of region, population and ecosystem services. In chapter 3, the methodology used for the Natural Capital valuation is described. Chapter 4 describes firstly the model of the Maasai steppe, including land cover change and the time dependence of land value, and presents the inputs used in the model in detail. In chapter 5 the results are shown, including the Natural Capital valuation, losses in carbon stocks of

---

[6] Namely, Tarangire and Manyara national parks
[7] Distinct from the external value, the value for non-local stakeholders.

the Maasai steppe and a comparison between the two. Chapter 6 provides an uncertainty analysis of the model. Chapter 7 discusses the model and its results, including the limitations of the model. Chapter 8 contains the conclusions and suggest directions for future research.

2 Scope of the analysis

2.1 Geographical scope

The geographical scope consists of the relatively homogenous districts, in terms of climate, economy, and ecosystems, of Monduli and Simanjiro in Northern Tanzania, plus the adjacent Lake Manyara and Tarangire national parks. These regions roughly comprise the "Maasai Steppe."

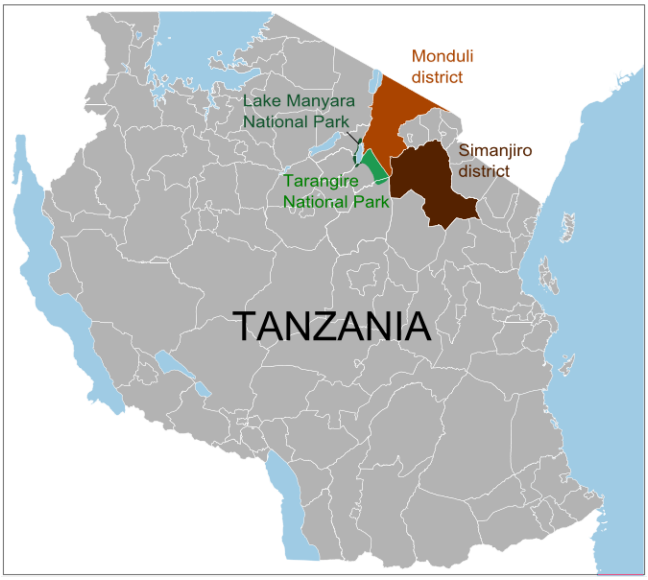


Figure 9. Geographical scope

Table 2. Key characteristics of the region studied

Considered area	2.994.000 ha	The area of Monduli and Simanjiro district is from Brinkhoff (2015)The area of National parks is from TANAPA, (2014)
Agricultural land	15%	Based on data from the Tanzanian agricultural survey, Manyara and Arusha regions (URT, 2012)
National parks	11% (two parks)	(TANAPA, 2015)
Livelihoods in the region	Pastoralism, agro-pastoralism,	(Homewood, Kristjanson, & Trench, 2002)
Population density	0,13 pp/ha	(Brinkhoff, 2015)

2.2 Land use types

Four types of land use are identified within the specified geographical boundaries:

- **Rangeland**. Natural grasslands mixed with woodlands occupy most of the region. This land is utilized not only as grazing territory for pastoralist Maasai populations, but also as provisioning grounds for useful plants (i.e. food, medicinal herbs) by sedentary populations (Homewood, Kristjanson, & Trench, 2002) . This area is very rich in wildlife, including large populations of elephants, giraffes, zebras, and wildebeests, which use the lands both for grazing and as migratory corridors between wet and dry seasons (Msoffe, 2010) .
- **Agricultural land[8].** This type of land use includes both cropland and pasture for sedentary livestock keepers. The two main crops cultivated in this area are maize and beans, and are utilized as both food crops for subsistence (household consumption) and cash crops. Yields in the region are low because most farming is done by hand, without irrigation or any significant amount of agricultural inputs (Castel, 2006 and URT, 2012). Farm sizes are small, between 1 and 5 hectares, although larger farms that are more commercially-oriented also exist (Homewood, Kristjanson, & Trench, 2002). Because of low soil fertility and unsustainable farming practices, farmland is usually abandoned after some years of cultivation, and is subsequently unfit for both crop and livestock production (FAO, 2009). This cycle results in a continuous land conversion rate, which is also driven by population growth and poverty (Msoffe, 2010).
- **National park**. National parks are areas of land reserved for wildlife and tourism. Two of the main Tanzanian parks, Tarangire and Lake Manyara, are located in this region. Although the parks are not fenced, local populations are not allowed to access the parks. Wild animals spend part of the year within the parks and part outside them, in the Maasai rangeland, for wet season grazing (Nelson, 2009).
- **Degraded land** is land that was farmed in the past, but has been abandoned because it became unfit for either farming or grazing. Reports of land degradation

[8] In the text the terms agricultural land and farmland are used interchangeably.

generally confirm that degraded land remain infertile indefinitely if no interventions are organized (see FAO, 2009, Kshatriya, 2007, and, for a case study in the Tabora region, Majule, 2012).

Urban land use is not included in the study analysis, as there are no major urban centres in the geographical area reviewed (Brinkhoff, 2015). The scope of ecosystem services of each land cover type is described in the next subsection.

2.3 Ecosystem services

The land uses identified provide utility to local populations in a number of ways. A qualitative materiality and data availability assessment prompts the inclusion of several ecosystem services in the present study (See Table 13 below; Ecosystem services materiality, data availability and scope selection).

With the aim of avoiding double counting, the recognized approach of focussing the valuation on *final* ecosystem services, those that provide a direct benefit to humans, is adopted (Landers and Nahlik 2013, Fisher et al. 2009, Balmford et al. 2008). *Intermediate* ecosystem services are excluded because these processes underpin final benefits rather than constituting benefits in themselves[9]. More information about the definition of ecosystem services examined can be found in Chapter 3.3 Methodology - Ecosystem services.

[9] For example, by looking at the value of harvested crops and useful plants the value added by pollination or nutrient cycling is already taken into account.

Table 3. Ecosystem services materiality, data availability and scope selection

Type of land use	Ecosystem service	Beneficiaries	Materiality (estimate)	Data availability	In scope
All land use types	Carbon storage	Global community			
	Carbon sequestration	Global community			
	Water cycle regulation	Local and national population			
	Water purification	Local population			
Rangeland	Wild fruit and vegetables	Households involved in trade of wild fruit and vegetables			
	Livestock products	Households keeping livestock			
	Medicinal herbs	Households involved in trade of medicinal herbs			
	Traded forest products (honey, gum)	Households involved in trade of forest products			
	Water	Local population			
	Wood products (fuelwood, construction)	Local population			
	Tourism	Households involved in tourism			
	Soil erosion prevention	Local population			
Rangeland and National park	Preservation of genetic diversity	Local population and global community			
National park	Recreation	National and international visitors			
Farmland	Crops	All farming households			
	Livestock products	All farming households			
Degraded land	None	None			

The list of ecosystems services in the Maasai steppe is based on a survey regarding ecosystem services in the world's biomes (De Groot, 2012), on literature about the value of pastoralism in Sub-Saharan Africa (Davis, 2007), and on publications by the International Institute for Environment and Development (IIED) on undervalued drylands (Hesse & MacGregor, 2009). Services provided by the National Parks are limited to tourism, as access to other benefits is not allowed. The impact of National Parks on the water cycle is only indirectly included through the water provisioning service.

In order to keep the uncertainty of the results low the choice was made to consider only those services for which primary data specific to Tanzania and Kenya is available. Some of the ecosystem services left out of scope, i.e. carbon sequestration, soil erosion prevention, and those relating to the water cycle, are potentially highly material, according to the few existing valuations of these services in grasslands (De Groot, 2012). However, the lack of fundamental research about these ecosystem services in this region results in them being left out of scope[10].

On the other hand, because more research has been conducted about carbon storage in grasslands, the value of the Maasai steppe's carbon stocks is estimated using data from an arid grassland in South Africa (Petz, Glenday, & Alkemade, 2014)[11].

Valuation of the services listed above follows the methodology presented in the next section which is in turn based on the general methodology detailed in Part I.

---

[10] To our knowledge, no quantitative research exists regarding the ecological processes underlying carbon sequestration, soil erosion or the water cycle in the Maasai steppe, the provisioning of water to local populations and the interrelations between land cover change and water cycling and purification in the region. Such studies would berequired in order to quantify the value of carbon sequestration, water provisioning and water cycling in the Tanzanian rangeland, and especially in the context of the conversion of land to agriculture.

[11] Carbon stocks are not, however, an ecosystem service that has an annual flow of benefits, but rather a stock of capital assets. As such their value is more comparable to the value of Natural Capital as a whole than to that of ecosystem services.

3 Methodology

3.1 Methodological context

To quantify Natural Capital we look at the value of ecosystem services, following the System of Environmental-Economic Accounting (SEEA) framework of the United Nations Statistical Division (UN 2014abc). The framework describes Natural Capital in terms of stocks (ecosystem assets) and flows (ecosystem processes and services) and states that capital stocks can be measured through expected flows of ecosystem services. The value of such assets is then defined in terms of their contribution to Inclusive Wealth (UNU-IHDP, UNEP, 2014).

Ecosystem services were first defined, identified and classified[12] in the context of the Millennium Ecosystem Assessment (MEA 2005). The Economics of Ecosystems and Biodiversity (TEEB) built on top of MEA's work, extended and further specified the list of services, added a comprehensive and rigorous classification of ecosystem types and associated services (Kumar, 2010), and created a database with thousands of values of ecosystem services from academic literature (Foundation for Sustainable Development, 2014).

Variations to the classification proposed by TEEB have later been proposed by other organizations in response to specific research needs. In particular the most recent frameworks, including SEEA's (UN 2014c) and the U.S. Environmental Protection Agency's (Landers & Nahlik, 2013), consider only regulating services that provide a direct benefit to humans, as opposed to those that only represent ecological processes underpinning provisioning services[13]. This approach was developed in response to concerns by economists that including regulating services may lead to double-counting (Fisher B., 2009; Balmford, 2008) and focusses on valuing only final services that have direct beneficiaries. We adopt this more recent approach in the methodology that follows.

[12] Classified as provisioning, regulating and cultural services.
[13] Supporting services and a broader definition of regulating services were included in the MEA before the approach was refined to the current level and they are therefore not included in most recent frameworks.

3.2 Formal definitions

Natural capital (NC) value of a region is defined as the sum of average *NC* values per hectare ($\overline{NC}$) of each land cover type *k* multiplied by the extent of a specific land cover type A_k.. $\overline{NC}$ is the price *P* of the natural capital asset *K* measured in hectares as defined in the general methodology. *NC* is then the contribution of natural ecosystems to inclusive wealth.

If A_k does not change over time

Equation 1 $$NC = \sum_k \overline{NC}_k * A_k$$

The average natural capital value of 1 hectare of land ($\overline{NC}$) is defined as the sum of the present value of ecosystem benefits (PEB) provided by 1 hectare of this land. For a given type of land use, the following definition applies.

Equation 2 $$\overline{NC} = \sum_{t=1}^{\infty} PEB_t = \sum_{t=1}^{\infty} \frac{EB_t}{(1+d)^t}$$

Where

d = discount rate.

EB_t = value of all ecosystem benefits provided by one hectare in year *t*. It is the sum of all benefits $EB_{t,j}$, specifically $EB_t = \sum_j EB_{t,j}$.

The **value of ecosystem benefit** $EB_{t,j}$ is defined as the economic value of a product or service[14] *j* derived from ecosystems (including agricultural produce) that can be attributed to land, as opposed to human inputs. It does not include non-use values, as defined in the Total Economic Value context. It corresponds to the j-th component of marginal land value $\frac{\partial C_j(t)}{\partial K(t)} v_j$ in equation (1) of Part I, when the asset *K* is hectares of land. It is redefined here as

Equation 3 $$EB_{t,j} = att_{t,j} \times q_{t,j} \times v_{t,j}$$

Where

[14] The product can be a consumption good or natural good. The terminology used depends on whether the product is directly consumed or used as input in the production of another product.

$v_{t,j}$= the value of consumption good *j* generated from the land in year *t*

$q_{t,j}$ = the quantity of consumption good *j* generated from one hectare of land in year *t*

$att_{t,j}$ = the attribution to ecosystems coefficient of a consumption good *j* in year *t*

The attribution coefficient $att_{t,j}$ excludes value added by inputs of human capital and prevents overestimation of services provided by ecosystems under different land managements. It approximates the marginal productivity of land together with $q_{t,j}$, specifically $\frac{\partial C_j(t)}{\partial K} \approx att_{t,j} \times q_{t,j}$ with respect to good *j*.

Attribution to ecosystems is defined as follows, for each type of land use, for a given year *t*

Equation 4 $$attr_t = \frac{r_t}{\sum_j (l_{t,j}+k_{t,j})+r_t}$$

Where

r_t= land rent shadow price for one hectare of land

$l_{t,j}$ = cost of labour requirements for ecosystem benefit *j* on one hectare of land

$k_{t,j}$ = cost of capital inputs required for ecosystem benefit *j* on one hectare of land

In the absence of a functioning market for natural land, shadow prices used by the specific beneficiary of the ecosystem benefits are applied[15].

3.3 Ecosystem services

The ecological processes, which are considered to provide services to humans, depend on the definition of ecosystem services adopted. Different definitions have been developed (Balmford, 2008; Landers & Nahlik, 2013). Here only *final* ecosystem services are considered. Following the U.S. Environmental Protection Agency's definition of Final Ecosystem Goods and Services, a **final ecosystem service** is identified wherever ecological processes yield direct benefits to humans (Landers & Nahlik, 2013).

---

[15] In this case, rural populations for rangeland and farmland and the government for national parks.

The choice to include only final ecosystem services is made to avoid double counting. For example, if the value of pollination would be included separately, this value would be exaggerated because pollination increases crop yields, and harvested crops, and is therefore included in those crop values. Another example is groundwater recharge, which allows a steady supply of water, and water provisioning (Balmford, 2008; Fisher B., 2009; Landers & Nahlik, 2013). As explained above (chapter 3.1 Methodological context) this is becoming a mainstream approach in ecosystem valuation.

Intermediate regulating ecosystem services and supporting services, in other words ecological processes that benefit humans only indirectly, are however not excluded: they are indirectly included in the value of those final services they support.

3.4 Valuation approach

Unit value transfer of direct market prices of ecosystem services or their substitutes is applied. Direct market prices are collected from the literature, focussing on sources that describe the specific area under examination in Tanzania, or Kenya's comparable Maasai region.

The choice was made to avoid literature using contingent valuation techniques, based on the consideration that this valuation approach is very sensitive to survey characteristics and the preferences of local stakeholders, and as such is hard to transfer between different sites accurately (Barrio M., 2009; Lindhjem, 2007). Furthermore market-based valuation methods are the most commonly used for quantifying direct use value of final ecosystem services (Barbier, 2013; Balmford, 2008).

3.4.1 Value transfer

The annual value generated by economic activities based on ecosystems is calculated using value transfer techniques as described by Brander (2013). Furthermore, we applied value transfer at the most granular level feasible: individually for each ecosystem service or subservice. In some instances value transfer was performed separately for biophysical quantities (i.e. quantities of crops produced) and prices (i.e. market price of the crop)[16].

[16] This was done whenever biophysical quantities were available and market prices were not considered representative.

Study sites

To examine the value of the selected ecosystem services in the Maasai Steppe, representative studies done in the area or other comparable areas are gathered, from which values of ecosystem sub-services are extracted.

To be selected as representative, studies had to fulfil the following criteria:

i. Study site and policy site match in terms of population and ecosystem characteristics.
ii. Population in the study site is homogeneous in terms of livelihoods and income.

Choice of transfer unit

Values of ecosystem services and benefits can be expressed in a variety of units i.e. per hectare, per household, per district or per person.

In order to transfer the value of the benefit from the original study site to the policy site, the value should be expressed in terms of the appropriate transfer unit. The appropriate transfer unit is the unit in terms of which the value varies the least between sites, for example a hectare or a beneficiary.

As a rule of thumb, the following guidelines are used in the choice of a transfer unit:

- Provisioning services: beneficiary. The size of the benefit depends on the amount of people that make use of the service, whenever the service is harvested below the maximum carrying capacity.
- Tourism: visitor. The benefits increase with increasing number of visitors.
- Final regulating services: beneficiary. The size of the benefit depends on the amount of people that make use of the service.
- Agricultural services: hectare. The benefit is proportional to the amount of area under cultivation.

Finally the value is expressed per hectare using the area density of invariant units at the policy site.

Price adjustment

The most recent values of ecosystem services are used after some unit adjustments. Monetary values are adjusted for inflation and converted to US$ Purchasing Power Parity (PPP) to reduce distortions due to inefficient currency markets and make the results more internationally comparable. The conversion values applied for Tanzanian and Kenyan Shillings are respectively 768 TZS/USD PPP and 43 KSH/USD PPP, extrapolated to2014 based on historical series (World Bank, 2014).

The methodology is implemented by the model specific to the Maasai steppe as explained in the next section.

4 Land valuation model

4.1 Overview

As explained above, the value of Natural Capital depends on the flows of ecosystem benefits over time. In practice these depend on how beneficiaries harvest this value and on how the extent and quality of ecosystems changes with time[17]. In any real life situation it is uncertain how these two dynamics will play out in the foreseeable future, so a model is built to study how patterns of land cover and value of ecosystem benefits will result in a different value of Internal Natural Capital in the region.

The model takes as inputs land conversion speed, land degradation speed, the value of ecosystem benefits provided by each land cover type and their projected growth rates in the future. The output of the model is the Internal Natural Capital value of the Maasai steppe.

Finally, the model is structured in two periods, with different variations in land cover of the region and productivity of each land use type.

- in the first period (from 1 to 20 years) both land cover and land productivity can vary
- in the second period (from 20 years onwards) land cover is constant while productivity can change.

[17] The ecosystem benefits that humans can draw from rangelands and national parks is related to the size of rangelands that are left in a pristine state, as both wild and cattle herds require wide spaces as a habitat for migration and grazing.

4.2 Input parameters

Table 14 shows which parameters are used as input for modelling Natural Capital and which ones have been left out of scope.

Table 4. Parameters included in the model

	Input	Out of scope
Rangeland	Changes in animal density Changes in number of beneficiaries (population growth) Current rates of growth / decrease in ES harvesting per beneficiary	Productivity improvements for livestock Increased involvement in trade New economic activities related to woodlands
Farmland	Lower soil fertility of newly converted land Abandoning of degraded farmland	Productivity improvements Switch to new crops
National park	Current rates of growth in visitor numbers Impact on visitor numbers due to wildlife corridors encroachment	Changes in revenue per visitor

Changes in observed productivity trends, or to the number of beneficiaries, are input parameters for each land use type. It is assumed that no new socio-economic trends will emerge that would strongly influence the relation between ecosystems and beneficiaries. Surveys do show that economic development in this area has been slow to progress up to this point, be it for farming (URT, 2012), livestock keeping (CGIAR, 2011), access to water (Jacques, 2014), and trading of rangeland products (UNCTAD, 2014). In fact the Maasai steppe are far from the main urban centres of development in Tanzania, and the main socio-economic trends observed in the region are the steady growth of visitors to National Park (TANAPA 2013) and the transfer of livelihoods from pastoralism to sedentary farming (Homewood, Kristjanson, & Trench, 2002).

4.3 Scenarios

The relation between the extent of rangeland left intact and the amount of value that pastoralists and national parks can generate from ecosystems is non-linear. There are critical thresholds that, once reached, lead to high drops in ecosystem values. Based on this observation, three scenarios were developed to explore what occurs when thresholds are reached in the short-term or long-term, or never reached.

The value of Natural Capital value in the Maasai steppe is therefore assessed under the following three scenarios.

Low rate of conversion to agriculture. Agricultural land growth slows down. This allows pastoralist populations to continue their nomadic lifestyle and keeps the remaining wildlife corridors open. Wildlife is preserved and tourism continues to grow at the current high rate.

Medium rate of conversion to agriculture. Agricultural land increases at a medium rate, all arable land is converted within 20 years and conversion stops. Abandonment of degraded land continues. Pastoralists and national parks suffer consequences due to encroachment of migratory corridors, but agriculture suffers from declining yields.

High rate of conversion to agriculture. Agricultural land conversion continues at a fast rate. Pastoralism shrinks as the rangeland is no longer large enough to support transhumance. More people settle down and initiate farming, but average yields are even lower. Land conversion is so fast that all fertile land is converted within 10 years. Wildlife corridors are also converted leading to a decline in animal populations and, in the long run, tourism revenues.

4.4 Structure

This section explains the structure of the model developed to quantify the value of Natural Capital in the Maasai steppe based on the changes in land cover and value of ecosystem services.

4.4.1 Natural capital value

The following approach for modelling natural capital is derived from the aforementioned Equation 1 and Equation 2. In the Maasai steppe the area of each land cover type *k* with area A_k changes over time, so NC is calculated as follows:

Equation 5
$$\mathrm{NC} = \sum_{\mathrm{t,k}} \mathrm{PEB_{t,k}} * \mathrm{A_{t,k}}$$

Where t is one year in the interval studied.

Based on Equation 2, equation 5 is reformulated for each type of land use:

Equation 6
$$\mathrm{NC} = \sum_{t=1}^{20} PEB_t A_t + \sum_{t=21}^{\infty} PEB_t A_t = \sum_{t=1}^{20} \left(\frac{EB_t}{(1+d)^t}\right) A_t + \frac{EB_{21}}{(d-g)} A_{21}$$

Where g is the growth rate of *v* after year 20, and EB_{21}, A_{21} is the value of ecosystem benefits and area of a specific land use type in the steady state. The average natural capital value of one hectare $\overline{NC}$ is calculated in a similar fashion.

In the next sections it is explained how A_t and EB_t are modelled to calculate the Natural Capital Value described above. It contains the data used and motivates the key assumptions underpinning the model.

4.4.2 Land cover change

As explained above, the total land area, A = 2.9 million hectares, is characterized by four main types of land use (Rangeland, Agricultural land, Degraded land and National Parks).

Land use change is modelled explicitly for the first 20 years and assumed to be in a steady state after 20 years. In the first period the areas of agricultural and degraded land grow at the expenses of rangeland. In the second period land conversion is assumed to stop. Agricultural land area and degraded land area are assumed to be constant and equal in size.[18]

[18] In reality what would happen after land conversion stops is that the area of abandoned land would gradually continue to increase until all former farmland is degraded. Here for simplicity we assume that at steady state 50% of the land that was once farmed is degraded, and we test this assumption in the sensitivity analysis.

Table 5. Land cover change

		Years t=1…20	Steady state t=21…∞
A	Total area	constant	constant
$\frac{A_{NP\,t}}{A}$	National park share in year t	constant	constant
$\frac{A_{Ag\,t}}{A}$	Farmland share in year t	For $\frac{A_{Ag\,t-1}}{A} < M$ $A_{Ag\,t-1} * (1 + LUC)$ For $\frac{A_{Ag\,t-1}}{A} = M$ $A_{Ag\,t} = A_{Ag\,t-1}$ Grows at a constant annual rate LUC until a maximum limit M is reached	$\frac{1}{2} * \frac{(A - A_{R\,20} - A_{NP\,20})}{A}$
$\frac{A_{Deg\,t}}{A}$	Abandoned land share in year t	$\frac{A_{Ag\,t-20}}{A}$ Land is abandoned after 20 years of farming (Msoffe, 2010)	$\frac{1}{2} * \frac{(A - A_{R\,20} - A_{NP\,20})}{A}$
$A_{R\,t}$	Rangeland in year t	$A - A_{Ag\,t} - A_{Deg\,t}$	$A_{R\,20}$

4.5 Input data

The land cover in the Maasai steppe is described in detail. It is followed by an explanation of productivity growth or decline coefficients used to model the three scenarios. Finally the annual costs and benefits for three land use types are discussed, namely Rangeland, agricultural land and national parks.

4.5.1 Land cover

Background facts

- 222.000 hectares of land were under agricultural use in the two districts according to the 2007-2008 Tanzania Agricultural Survey (URT, 2012).

- Although two of the five major wildlife migration corridors in the region have already been closed by agriculture, 35% of the remaining rangeland still has a high or medium probability to be converted to agriculture (FAO, 2009).
- Land conversion to agriculture proceeds at a high rate, but no recent data is available on the current speed of conversion in these districts.
- Studies of changes of land cover in the region are all based on a single study (Kshatriya, Kifugo, Msoffe, Neselle, & Said, 2007) that assessed land cover changes between 1986 and 2000 by analysing satellite images. This study showed that in the Monduli and Simanjiro districts (representing 74% of the region) the annual growth rate of farmland had been 11%.
- This is a very high rate that would result in reaching the limit of 35% in 6 years (and if continued at that pace, would convert the whole region by 2032). It is safe to assume that the rate of conversion to farmland has already decreased, as land has become scarcer.
- Agricultural land is regularly abandoned. According to the above mentioned study by Kshatriya et al (2007), 75% of the land that was farmed in 1986 had been abandoned by 2000.

Initial land use partition

The following values in Table 16 were used for land cover in the Maasai steppe in year 2014 (t=0).

Table 6. Initial land use partition

Land use type	Hectares	%	Source
Farmland	461,372	15%	Estimation based on Tanzania regional agricultural survey (URT, 2012)
National Park	318,000	11%	Area of Lake Manyara NP and Tarangire NP (Nelson, 2009)
Rangeland	1,839,170	72%	Calculated.
Degraded land (unproductive)	57,225	2%	Extrapolated value for extent of agricultural land 20 years earlier.
TOTAL	2,993,769	100%	Area of Monduli and Simanjiro district and the two above mentioned national parks.

Future land conversion

As explained in the land use model section (chapter 4.4.2 Land cover change), future changes in land use were modelled as growth of farmland and growth of abandoned land. Due to uncertainties about rates of growth of farmland per year, separate scenarios have been developed.

None of the three scenarios is classified as business-as-usual, as the current rate of land conversion is not calculated in the surveyed literature. The only data available (Msoffe, 2010) suggest that at the time of the study (2006) it was above 10%, a rate that if sustained would have resulted in the whole region being converted in 20 years. It is expected that the rate slows down[19].

The amount of land within the Maasai Steppe that is fit for agricultural use is limited. It is estimated that about 600 000 hectares of rangeland could still be turned into agricultural land today (around 30% of remaining rangeland). In total (including the land already farmed) the maximum amount of farmland area in the region is assumed to be around 1 million hectares, or 35% of the total region's area.

It is assumed also that abandoned land is neither productive for crops nor for grazing, as suggested by FAO (2009).

The values applied to model land conversion are summarized Table 17 below

Table 7. Future land conversion values

Parameter	Value	Explanation
Rate of expansion of farmland: *High conversion scenario*	8%	Three scenarios are developed
Medium conversion scenario	4%	
Low conversion scenario	2%	
Annual expansion of degraded land	11%	Based on speed of land conversion to agriculture at t=-20
Maximum share of arable land in the region	35%	Assumption based on (FAO, 2009). See text for explanation.

[19] as the land available becomes marginal and locals diversify their livelihood in other directions to complete their income, for example by migrating to urban centers.

The sensitivity of the results to the assumptions in the values above is tested in a sensitivity analysis.

The next subsection describes the input data used to model trends in ecosystem services value over time.

4.5.2 Growth rates of ecosystem services

Changes in ecosystem services are modelled per group of services that have similar underlying biophysical factors driving changes in value per hectare over time. For example all the ones relating to pastoralist livestock herds are assumed to follow the same growth trends, as do those related to other traded rangeland products (gum, beeswax and honey) and different crops (maize and beans) in agricultural land.

Growth rates

As explained in chapter 4.4 *Model Structure* above each type of ecosystem benefit is modelled with up to four growth rates: one for each of the two time periods, both for growth of transfer unit density (i.e. beneficiaries per hectare, or heads of livestock per hectare) and growth of value per transfer unit. Since the modelling is carried out for three scenarios, there are in total twelve growth rates per group of ecosystem services.

Two parameters that are assumed not to be affected by the changes in land cover are equal across all scenarios: population growth is assumed to be constant at 5.8% in the first period (National Bureau of Statistics, Tanzania, 2015) and to slow down to 0.5% in the second period[20]; wood fuel use per person is assumed to decrease in the first period by 1.5% per year and remain constant in the second period.[21] Parameters have been assigned

---

[20] Population growth is used as a basis to calculate the growth of beneficiaries per hectare for water, wood fuel and other non-livestock rangeland products, both traded and subsistence ones. It is considered smaller after year 20, assuming the arid landscape of the Maasai steppe cannot support a high growth of human populations indefinitely.

[21] Although wood and charcoal fuel consumption is expected to grow in Tanzania (SAFMA, 2004), woodfuel collection per person is steadily decreasing as the population switches to more efficient stoves and fuels (FAO, 2009). Commercial charcoal production is one of the main drivers of deforestation in Africa (Mwampamba, 2007 & SAFMA 2004), but it is not expected to play a significant role in the Maasai steppe, as it is normally observed near major urban centers (FAO 2009b).

zero value wherever no growth trends are currently observed.[22]Only a handful of parameters are therefore considered to be influenced by the speed of conversion to arable land in the region. These have different values in each scenario and they are summarized and explained in Table 18.

Table 8. Parameter change across scenarios by land use

Key parameters changing between scenarios		Scenario			
	Parameters	LOW	MID	HIGH	Explanation
AGRICULTURE	Expansion of agricultural land	2%	4%	8%	Hitting the maximum threshold of arable land in 10, 20 and 40 years.
	Revenue/ha change per year	-1%	-2%	-3%	Yields are currently stagnating. In the future, as land conversion progresses less fertile land is cultivated (FAO 2009, Msoffe 2010)
RANGELAND	Animal density growth(t=1...20)	0%	-1%	-6,7%	LOW scenario: no change, HIGH scenario: decline rate of populations of wildebeest between the 70s and 90s when migratory corridors were closed (Msoffe 2010, Nelson 2009). MID scenario: sow decrease.
NATIONAL PARK	Increase in visitors/year (t=1...20)	6,3%	6,3%	4%	6.3 is the current growth rate (WTTC, 2014). In HIGH scenario it is assumed that wildlife populations start to decline earlier than 20 years and the growth of tourism is affected.
	Increase in visitors/year (t=21...)	1%	0%	-1%	It is assumed that after 20 years tourism continues to grow if wildlife corridors are preserved (LOW), it reaches a steady state in the MID scenario and starts to slowly decrease in the HIGH scenario.

[22] For agricultural land: share of cropland to pasture and livestock density. For rangeland: water harvesting per person, productivity of pastoral herds, share of the population involved in harvesting gum or beekeeping and income per person involved in these trades, consumption of subsistence rangeland products such as food and medicinal herbs. For national parks revenue per visitor.

The valuation of ecosystem services for each land use type follows in the next subsection.

4.5.3 Rangeland

Introduction

The rangeland of the Maasai steppe considered here is a semi-arid to arid savannah region. Specifically, it is a patchwork network of savannah grassland and acacia tree woodlands. Maasai pastoralists live in temporary villages (bomas) that are usually inhabited by elders, women and young children while adults and older children travel the savannah with their livestock in search of grazing pasture. Typical livestock kept include the indigenous zebu cattle, goats and sheep. Maasai rely almost exclusively on their surrounding ecosystem to survive. The habitat provides them with income, food, clothing, medicine, construction materials and fuel. A major threat to pastoralist livelihoods, apart from land-conversion to agriculture, are droughts that occur every few years in an unpredictable fashion (Msangi, Rutabingwa, Kaiza, & Allegretti, 2014).

Key literature

The regional context and characteristics of ecosystem and local livelihoods were extracted from (Homewood, et al., 2002, Davis, 2007 and Mdoe & Mnenwa, 2007). These sources offered key insights on how locals secure their income, the historical evolution and current state of the system as well as key drivers of land conversion and livelihood diversification. This allowed careful choice of study sites and values.

Inventory

Services typically provided by grasslands (Table 13) are well-known in the literature (Honigova, et al., 2012). In the context of a bottom-up analysis, the list of ecosystem services) is customized for the Maasai steppe region based on relevant studies[23] (Davis, 2007).

The beneficiaries in the rangeland are primarily the Maasai pastoralists, but also sedentary agro-pastoralists benefit by extracting wild food, medicine, fuel and wood from the ecosystem. External beneficiaries outside of the region are not considered. However,

---

[23] Certain services are not included explicitly since they are not final services while other services are not represented by a comparable valuation with a sufficient level of detail. Specifically, drought prevention is not considered explicitly, since its exact valuation requires advanced modelling and no existing valuation was found in the literature.

carbon stocks are included in the overall analysis and are described in chapter 5.3. Services used by beneficiaries are divided into five groups: livestock, wood, subsistence, traded goods and water products.

- Livestock products are meat, milk, skins and hides. Milk is rarely sold and mainly consumed for subsistence[24].
- Wood products include firewood and charcoal used as fuel and poles and thatch used as construction material. Timber production values are considered negligible due to the low forest cover in the Maasai Steppe and are not included (Msoffe, 2010).
- Subsistence goods include fruits and vegetables harvested from the woodland in the region, especially during the dry season[25] (Monela, Chamshama, R., & D.M., 2005).
- Traded products are products that have a large demand in global markets, specifically honey and beeswax, gum and medicinal herbs. Their harvesting is underdeveloped (Mdoe & Mnenwa, 2007) and their option value is assessed to be much larger than the direct value collected at the moment (Davis, 2007).
- Water provisions include both surface-water and groundwater extracted from the rangeland. Households sometimes own their own borehole but water is scarce and household members often have to travel long distances to secure it (The Whole Village Project, 2010).
- Recreational services were not considered as they are not developed in the Maasai rangeland[26]. Tourism is concentrated in nature reserves and few or no evidence

[24] The livestock system is known for its low productivity, as feed is scarce and owners have little access to markets. Additionally, cattle is considered an asset and is rarely sold for meat production. More details can be found in the Tanzanian pastoralism snapshot description in the main report.

[25] Wild foods would usually include wild mushrooms and edible insects, however the value in this case was negligible.

[26] The option value of recreation is however considered to be high and tightly connected with biodiversity conservation (Mohammed, Naini, & Douglis, 2009).

was found of the involvement of Maasai households in tourism outside national parks (Nelson, 2009).

Study sites

Each subservice valuation was carefully selected from regional studies found in scientific literature and non-governmental-organization reports. Livestock services were valued based on data specific to the Maasai steppe. This was compared to more recent literature of experts about the region and some disagreement between livestock densities was found[27] (Homewood, Trench, & Brockington, 2012).

Study sites were also chosen in Kenya based on studies located in Turkana and elsewhere (Davis, 2007). These livelihood systems have remained closer to their traditional roots and are considered representative of pastoralism in Tanzania as well. Although market prices can be different in Kenya compared to Tanzania (Homewood, et al., 2001) we adjust for this difference by using PPP equivalent currency.

Another study site chosen was the agro-pastoral Shinyanga region. An exhaustive report was available that documented the benefits those locals extract from their natural environment, specifically woodlands and forest. The high quality of data and similar dependencies of locals on their natural environment between the two regions is sufficient reason to choose this as a study site despite of the differences between sedentary and transhumant livelihoods.

The region of the Eastern Arc Mountains (EAM) was also chosen, as it has been extensively studied and advanced literature exists that describes benefit extraction there. EAM is a global biodiversity hotspot and it suffers from unsustainable and illegal timber extraction and trade (Schaafsma, 2014). Concurrently, local agro-pastoral livelihoods largely depend on the forests to support their income and cover their needs for fibers and fuel.

Finally, agro-pastoral villages in the Maasai steppe were also chosen as study sites (Castel, 2006). Locals practice mixed farming but also harvest other commodities from the natural

[27] Recent regional data is not representative of traditional pastoralism as former pastoralists have diversified their livelihoods to adapt to the new land tenure laws banning communal grazing areas (Homewood, Trench, & Brockington, 2012).

environment such as honey and beeswax. This source is used extensively in the Agricultural land input data section (chapter 4.5.4).

Valuation of benefits

Footprints in biophysical terms per transfer unit and unit prices of ecosystem subservices are presented in Table 19. The footprint is given in terms of either production unit, such as cattle, or beneficiary, such as person and household.

The goat milk and non-timber forest products (NTFP) services are monetized directly as a benefit since a footprint was not available in the surveyed literature.

Livestock footprints are based on animal productivity data as described in the Tanzanian pastoralism snapshot of this report[28]. Pricing was averaged between different sites. Specifically, beef prices were averaged between the pastoralist Turkana region in Kenya and the agro-pastoralist Mara region in Tanzania (Davis, 2007 and Rentsch & Damon, 2013).[29] Milk prices were similarly averaged between the pastoral Turkana region in Kenya and the pastoral to agro-pastoral Usangu region in Tanzania (Davis, 2007) (Mdoe & Mnenwa, 2007). Goat milk was monetized as a benefit based on the Turkana region in Kenya (Davis, 2007).

NTFPs monetary values are based on a literature review by Schaafsma (2014). In this study the benefit is calculated using a household production function that is based on various sites in the EAM region and then transferred across and aggregated over the entire region. The value used here corresponds to a poor household and it is assumed that it is similar to that of an average household in the much drier rangeland of the Maasai steppe.

[28] An assumption was made that only half of the destocked cattle is harvested for their skin, as pastoralists do not usually trade hides but make private use of them (Davis, 2007).
[29] Additionally, the value added by the butchery was subtracted based on studies in agro-pastoral regions of Tanzania (Kadigi, Kadigi, Laswai, & Kashaigili, 2013).

Table 9. Rangeland ES services footprint and pricing

Subservice	Region	Subtype	Footprint (FP) units	FP	Price units	Price/unit
Beef	Maasai Steppe	Livestock	kg/cattle	15	TZS2013/kg	3,195
Milk	Maasai Steppe	Livestock	kg/cattle	35	TZS2007/kg	328
Hides	Maasai Steppe	Livestock	piece/cattle	0.1	TZS2005/piece	421
Goat milk	Turkana, Kenya	Livestock	goat	1.0	KES2004/goat	126
NTFPs	EAM	NTFP	pp	1.0	TZS2010/pp	12,000
Fruit & vegetables	Shinyanga	Subsistence	kg/pp	4	TZS2004/kg	76
Water	Tanzania	Subsistence	m3 water/pp	19	TZS2011/m3	751
Honey	Monduli and Simanjiro	Traded products	kg/hh	61	TZS2011/kg	2,532
Beeswax	Monduli and Simanjiro	Traded products	kg/hh	4	TZS2011/kg	5,064
Gum	Turkana, Kenya	Traded products	kg/hh	52	KES2009/kg	39
Medicinal plants	Shinyanga	Traded products	kg/pp	0.9	TZS2004/kg	2,423

Subsistence products are composed of wild foods that are easy to harvest but here we look only at fruits & vegetables. The fruit & vegetables footprint is derived from the Shinyanga region and is an average of the region's districts (Monela, Chamshama, R., & D.M., 2005). Monetization of the benefit is based on market substitutes and household surveys[30].We also classify water as a subsistence service, as its collection for other uses is not considered relevant for beneficiaries living in the rangeland. The water footprint follows from country level data from AQUASTAT and the pricing is taken from DAWASCA water company in Dar es Salaam (Bayliss & Tukai, 2011).

Traded products are products harvested from the rangeland that have a high demand on national and international markets. Honey and beeswax footprints were collected from the villages in the mixed farming study (Castel, 2006), so that we assume an agro-pastoralist and a pastoralist collect similar amounts. Pricing is an average over a pastoral

[30] Pricing is deduced from the footprint and the value of the benefit.

forest region in Kenya (Kipkoech, Mogaka, Cheboiywo, & Kimaro, 2011), the pastoral to agro-pastoral Usangu region (Mdoe & Mnenwa, 2007) and village prices in the Maasai steppe (Castel, 2006).[31] Arabic gum footprint and price was collected from a study on gum collection in the Turkana and other regions in Kenya (Vellema W., 2014). Finally, the benefits derived from medicinal herbs are valued based on the study in the Shinyanga region (Monela, Chamshama, R., & D.M., 2005).

Value transfer to policy site

Benefits are transferred from study sites to the policy site based on the unit that has the smallest variance between sites, listed in Table 20. Animal productivity is expected to be similar between different pastoralist regions in the drylands as management, breed and food availability are fixed by tradition and environmental factors. Cattle density is chosen to be the transfer density and its value is provided from FAO's Global Livestock Production and Health Atlas (GLiPHA) (FAO, 2002)

People with similar needs in the same environment and low incomes are assumed to derive similar benefits from the rangeland if access to the resource is not limited by supply or institutions. We transfer values of NTFP, medicinal herbs and wild fruits and vegetables based on the Maasai steppe population density. This was derived from a database (Brinkhoff, 2015)and excluded urban centres larger than 10 000 people in order to be representative of the rural region.

Because households tend to specialize in certain economic activities, the value was transferred per household for traded products such as honey, beeswax and gum.[32] Therefore the transfer density is modified by the fraction of households collecting in the region. The fraction of households collecting honey, beeswax and gum is from the same sources as the corresponding footprints.

[31] The value of Beeswax is estimated based on the ratio of beeswax value to honey in the Usangu region assuming the two are linearly related.

[32] Transferring the benefit per person would be an overestimation if all households are considered to collect or an underestimation if more households collect at the policy site.

Table 10. Transfer unit density

Subservice	Transfer unit (t.u.)	T.u. density/ ha	Explanation
Beef	Animal	0,14	Based on FAO glipha data
Milk	Animal	0,14	Based on FAO glipha data
Hides	Animal	0,14	Based on FAO glipha data
Goat milk	Animal	0,03	Goat to cattle ratio
NTFPs	Person	0,13	Population density
Fruit & vegetables	Person	0,13	Population density
Water	Person	0,13	Population density
Honey	People in household involved	0,001	7% of households
Beeswax	People in household involved	0,001	7% of households
Gum	People in household involved	0,002	16% of households
Medicinal plants	Person	0,13	15% of households

Input costs

Human input in the rangeland is small, as the system uses little infrastructure and technology as well as external resources. The cost is calculated for each service or group of services and is then aggregated to a total. Costs are summarized in Table 21.

Livestock services are provided by the animal herd, so that the cost for all services together is the human labor put into caring for and managing the herd. Labor is provided by the pastoralist household members that manage and care for the herd and an average labor per tropical livestock unit (TLU)[33] was calculated based on a FAO survey of pastoralism in Kenya (Bekure, Leeuw, Grandin, & Neate, 1991). The labor is then converted to labor per

[33] A Tropical Livestock Unit allows comparison between values of livestock in areas with varying herd sizes and types of animals, through the conversion of each livestock type based on bodyweight (1 cow = 0,7 TLU, 1 sheep or goat = 0.1 TLU, 1 chicken 0.01 TLU).

hectare using cattle density. Labor was priced according to the minimum wage in Tanzania in 2013 (WageIndicator Foundation, 2015)

NTFP collection requires that locals travel to the site, harvest and process the material. Cost per item collected is provided by a study of pastoralism in a Kenyan forest area (Kipkoech, Mogaka, Cheboiywo, & Kimaro, 2011).[34] It is assumed that the labour costs are proportional to value between sites.

Subsistence services costs are assumed to be utilized at zero cost. Little labor is required to harvest wild foods and water that originates from a public tap or borehole[35]. Medicinal herbs are also assumed to have zero production costs.

The rest of the traded products are collected with almost no processing involved, so that the production cost is the labor involved in collecting them. Arabic gum is collected without tapping and later sold to agents (Wekesa, Luvanda, Muga, Chikamai, & Makenzi, 2013). Honey and beeswax are collected together and the value added by processing by the collector is considered small (Kipkoech, Mogaka, Cheboiywo, & Kimaro, 2011).

Table 11. Rangeland input costs: original and final values

Service	Cost in original study		Transferred cost	
Livestock	FTE/TLU	0,02	FTE/ha	0,002
	TZS2005/month	5500	PPP2014/FTE	133
Beekeeping	costs / revenue	0,33	PPP2014/ha	0,08
Wood collection	cost/revenue	0,30	PPP2014/ha	0,8
Gum collection	KES2013/kg	2	PPP2014/ha	0,005

Finally, because rangeland land does not have a market value, as a shadow cost of land the market rent of land asked by locals in agro-pastoral villages of the Maasai steppe is used (Castel, 2006).

[34] The value transferred is the ratio of aggregated cost to market value and cost at the policy site is the product of this ratio with the value at the policy site.
[35] It is often the case that this infrastructure is not available in pastoralist villages and people may travel many hours to collect drinking water (The Whole Village Project, 2010).

Results

The total annual value from rangelands, the part attributed to ecosystems and the costs incurred to gain those benefits for a hectare of the Maasai land are shown in Table 22.

Table 12. Ecosystem benefits and input costs for 1 hectare of rangeland in the Maasai Steppe

Ecosystem service USD2014 PPP/ha/yr	Benefit before attribution to ecosystem	Ecosystem benefit	Cost 1 ha other inputs
Wild food	0.10	0.10	0.00
Livestock (beef, cow milk, goat milk, skins)	12.62	11.99	2.68
Medicinal herbs	0.72	0.68	0.00
Traded products (honey, beeswax, gum)	0.41	0.39	0.08
Drinking water	2.96	2.81	0.00
Wood products (fuelwood, charcoal, poles, thatch)	2.64	2.51	0.79
TOTAL	**19.46**	**18.48**	**3.55**

The largest part of the value is attributed to livestock products and in particular beef and milk production, as expected from the main occupation of beneficiaries in the region. Water and NTFPs represent the next largest benefits and are a significant part of the value extracted from the rangeland. Finally, the attributed value to ecosystems is 18.48 PPP2014/ha, 95% of the value before attribution, as labor and capital costs are minimal in the production system.

4.5.4 Agricultural land

Introduction

Agricultural land is modelled taking into account the value extracted from land by populations that are sedentary, hold a small quantity of land, and are involved in agriculture (marking a shift from pastoralism towards agro-pastoralism).

Key literature

Literature gathered included a socio-economic baseline survey of 3 villages in the Maasai Steppe (Castel, 2006), Tanzania National Census of Agriculture 2007/2008 Arusha Region (URT, 2012), FAO data on tropical livestock unit (TLU) in sub-Saharan Africa conversion factor (Otte & Chilonda, 2002), FAO data on animal density in the Mara region (FAO, 2002), agricultural labour wage data from Wageindicator Foundation (2015), inflation rate data from (OECD, 2014) and labour cost data from literature on cattle production (Kadigi, et al., 2013 and Mlote S.N., 2012)).

Inventory
The ecosystem services reviewed in the mixed farming system consist of crop production (maize, beans) and livestock production (cattle, goat, sheep, chicken, cow milk, goat milk, eggs) (Table 13).

Study site
Data of the study site is mainly derived from a socio-economic baseline survey conducted by Vincent Castel (2006) in three villages in the Monduli and Simanjiro districts in Northern Tanzania: Lolkisale, Naitolya, and Loiborsoit 'A'. This study is chosen as a main source because the sample population surveyed reflected an area within the policy site in which most of the population has shifted from pastoralism to agro-pastoralism, thus making the data more specific and representative of the Maasai Steppe than, for example, the Tanzania national-level agricultural survey, which does not distinguish between pastoralist and agro-pastoralist households. The study also provides comprehensive data on most land inputs and outputs and their values.

In this study, 80% of the 363 households surveyed practice crop farming, and 68% of the households own livestock. The main crops produced in this area are maize and beans. This is slightly different from national-level data for main crops where maize is followed by cassava, beans and paddy (Castel, 2006 p. 71). 100% of producers in the area plant maize, which has become the staple food, and 64% of producers plant beans, which are principally marketed. Livestock owned by producers are cattle, goats, sheep, and chicken.

Cultivation activity in this area is characterized by low technical capacity, low input rates, and low yields. Revenue comes from selling part of the maize and beans harvest that is not consumed by the household. Use of irrigation technologies and chemical fertilizer were not observed (Castel, 2006 p. 4). Only few producers apply pesticides, and the efficiency is low because the volume applied is low. The only significant costs involved are seed costs and labour costs.

Livestock-keeping is characterized by a low stocking and destocking rate, low fertility ratio, low milk yield, and the use of a drought-resistant breed with low-productivity (Castel, 2006

p. 4). Sources of revenue are stock marketing, livestock slaughtering, and milk and eggs production.[36] Costs involved in keeping livestock are for livestock inputs, stock buying, and labour. Most (82%) of livestock producers use the communal lands to feed their animals, and 20% use cut-and-carry (grass, maize, beans) to supplement feed. The use of livestock input (labour, dewormers, drugs, and vaccines) is high for cattle, probably due to the significantly higher value of cattle compared to the other livestock. Stock buying remains marginal.

Valuation of benefits

Due to the minimal inputs, annual maize productivity in this area at 990 kg/hectare (Castel, 2006) is lower than the national average at 1096 kg/hectare (FAOSTAT, in Castel, 2006), but higher than the 2008 average yield of the Monduli and Simanjiro districts at 685 kg/hectare (URT, 2012). This is also observed for beans, with a yield of 625, 763, and 345 kg/hectare for the baseline survey (2006), national average (2006), and Monduli and Simanjiro (2012) respectively. We therefore estimate that the baseline survey values are representative of the crop productivity in the Maasai Steppe. The mean price per kilogram of crops are taken from the baseline survey, inflated according to the annual inflation rate of Tanzanian Shillings through 2014 and converted to USD 2014 purchasing power parity (PPP). Value in USD 2014 PPP for a hectare cultivated by a single type of crop is obtained by multiplying the yield with the market price.

Livestock revenue and costs are calculated per tropical livestock unit (TLU). Value per type of animal is converted using the TLU conversion factor for sub-Saharan African livestock, adopted from Otte & Chilonda, FAO (2002).

The survey (Castel, 2006) unfortunately does not provide precise footprints for calculating the livestock revenue. Instead of calculating based on footprints, we therefore decide to take the percentage of income and consumption derived from each livestock and the total household gross income and consumption to calculate the total value of livestock per farm. An average farm in the study site holds 8,32 TLU of livestock, and from these data we derive the value of livestock per TLU in the study site.

[36] Livestock are slaughtered for ceremonies, consumption, or when the animal is sick or dying, but at a very marginal rate.

Value transfer to policy site

Values for crop production are transferred by hectare of cultivated land. Values for livestock are transferred by animals expressed in Tropical Livestock Unit (TLU).

The value of crops produced on an average hectare of farmland is estimated by looking at the average ratio of cropland to pasture in each farm and determining the partition of cropland between maize and beans.

Of all cultivated area, maize is planted on 68,5%, and beans 31,5%. These percentages are based on the crop land share of the baseline survey (Castel, 2006); other regional-specific crops are also planted but in marginal proportions, so the ratio is collapsed into the two main crops. In the Arusha region, as a comparison, maize constitutes 63% of the area planted, and beans constitute 26%, while in the Manyara region, the ratio of maize and beans to the total planted area is 56% and 17% respectively. (URT 2012).

Crops are assumed to take 41% of all land holdings, so maize is estimated to be planted on 29% of the entire land holding and beans on 13% on the policy site. The remaining land is assumed to mostly consist of pastures. These land cover percentages were taken from the Naitolya Village in the Monduli District, one of the three villages in the considered study (Castel, 2006) where the land holding per capita is highest and crop production is the main occupation (90%). Land cover and utilization in Naitolya were therefore considered most representative of mixed farming in the region.

For the value transfer, we refrain from using the livestock density in the study site (Castel, 2006) because with a range of 3-12 TLU/hectare, the density at the study site is too high for a closed farming system. Such high livestock density is only possible for a mixed agro-pastoralist system where livestock also graze outside of the land holding of each farm on a common pasture.

Livestock density is therefore assumed to be similar to animal density in the Mara region, which is the region in Tanzania with the highest livestock density, or 0,53 TLU/hectare (FAO, 2002). This value was calculated based on regional FAO cattle density data and converted using the average ratio of cattle to total farm TLU in Castel 2006.

The Mara region was selected because it is more densely populated and further converted to farmland than the study area, and this condition is expected to determine the size of livestock herds able to be maintained by a typical mixed farm. The livestock density of this region therefore represents the density that can be maintained by the farms within their farm boundaries. The value also matches the livestock density in the snapshot Tanzania dairy mixed feeding in the main report of TEEB-Animal Husbandry which is 0,5 cattle head/hectare.

Table 13. Transfer unit densities for agricultural land use types

Transfer Unit densities			
Cropland ha/Farm ha		0.42	Natolya village
Maize cropland	68.54%	0.29	Average Monduli study
Beans cropland	31.46%	0.13	Average Monduli study
Livestock TLU/Farm ha		0.53	Mara region, FAO livestock density in agricultural land

Revenue per hectare for each crop in the policy site is obtained from multiplying the yield of one hectare of crop with the land share allocated for each crop. In the policy site, one hectare of farm therefore provides a value of 123,26 USD 2014 PPP from maize planted on 0,29 hectare, and 97,25 USD 2014 PPP from beans planted on 0,13 hectare.

Livestock value per hectare in the policy site, USD 54,70 2014 PPP, is derived from multiplying the livestock density of 0,53 TLU/hectare in the policy site with the income generated from one TLU livestock calculated in the study site. Table 24 summarizes the value transfer of farm revenues that has been described in this section.

Table 14. Breakdown and Value Transfer of Farm revenues (Study vs Policy Site)

Category	Product	Transfer unit (t.u.)	Original value USD2014 PPP/t.u.	T.u. density t.u./ha farm	Value USD2014 PPP/ha farm
Crops - revenue	Maize	ha maize cropland	432.24	0.29	123.26
	Beans	ha beans cropland	743.02	0.13	97.25
Livestock - revenue	All Livestock	TLU	102.91	0.53	54.70

Input costs

Value for land rent is taken from Castel (2006), and converted to rent per hectare per year in USD 2014 PPP. This is the same land price used for rangeland.

Crop cultivation costs can be divided into labour and non-labour costs. Labour costs include costs for employing labour for land preparation, planting, weeding, and harvesting. The baseline survey indicates that farms typically employ members of their households to work in the fields and tend livestock. For the purpose of this valuation, we value the entire labour input, regardless of whether it is done by family or hired labour. For labour cost, footprints such as the types of land preparing methods, days needed to complete each cultivation stages, and wages paid for planting, weeding, and harvesting workers, are taken from Castel (2006). Labour wages for all methods of land-preparation are taken from the agricultural manual labour hourly rate reported by htttp.WageIndicator.com, deflated to Tanzania Shilling 2005 to match the other input prices. The prevalence rate for each land-preparation method is taken from the Arusha region agricultural survey to better represent the technical and logistical capacity of a greater sample population. From these calculations, the cost of labour for one hectare of each crop per year in the study site expressed in USD 2014 PPP is obtained.

The non-labour costs of cultivation in the study area consists of seed and pesticide costs, both converted to USD 2014 PPP per hectare of cropland. Costs of seeds used per hectare are derived from the average expenses per farm spent on maize and beans seeds as reported in the household survey, divided by the area of the mean land holding allocated for croplands (1,94 hectares). The average cost for pesticides per hectare is calculated by multiplying the percentage of households using pesticides (14%) with the usage rate (litre/hectare) and the price of pesticides per litre, thus averaging the cost out with the 86% of households that do not use pesticides.

Livestock cost is divided into labour and non-labour costs. Labour cost for tending livestock is calculated by averaging the values of labour per cattle head in two other studies on cattle production (Kadigi et al., 2013; Mlote et al., 2012). Converting it using the FAO TLU conversion factor (0,7 TLU per cattle head), we derive the labour cost per TLU of livestock in the study site and use it for all livestock, assuming labour costs per TLU are on average similar.

Livestock non-labour costs consist of costs of buying stock and livestock inputs such as drugs, dewormers, and vaccines. Costs of buying stock per farm is calculated by multiplying the percentage of households that bought stock over the year, multiplied by the quantity of stock bought (product of the ratio of stock bought to the overall herd population with the herd size of each stock), multiplied by the average buying price of each type of livestock. Cost of livestock input per farm per year is calculated by multiplying the average price of input for each livestock averaged out over the whole population (percent of producers procuring each inputs multiplied by the average cost per animal per year) with the herd size for each livestock. The sum of the costs, divided by the number of TLU per farm and converted to USD 2014 PPP, is the non-labour costs per TLU.

The value transfer of production costs is summarized in Table 25.

Table 15. Breakdown and Value Transfer of Farm Costs (Study vs Policy Site)

Category	Product	Transfer unit (t.u.)	Original value USD2014 PPP/t.u.	T.u. density t.u./ha farm	Value USD2014 PPP/ha farm
Crops - costs	Labour cost, maize	ha maize cropland	295.27	0.29	84.20
	Labour cost, beans	ha beans cropland	331.05	0.13	43.33
	non labour costs, crops	ha cropland	69.04	0.42	28.73
Livestock - costs	Livestock Inputs	TLU	34.25	0.53	18.20
	Labour Cost, livestock	TLU	20.41	0.53	10.85

Results

The total annual value from farmland, the part attributed to ecosystems and the costs incurred to gain those benefits for a hectare of agricultural land in the Maasai region are shown in Table 26. Crops contributed 220.51 USD PPP and livestock 54.70 USD PPP annually to the total value of a hectare farmland, while the two services incur 156.26 and 29.05 USD PPP in costs respectively. The coefficient of attribution of benefits to ecosystems is 26.6% for agricultural land.

Table 16. Total revenue and Costs of a Hectare Agricultural land in the Policy Site

Ecosystem service USD2014 PPP/ha/yr	Benefit before attribution to ecosystem	Ecosystem benefit	Cost 1 ha other inputs
Crops	220.51	58.70	156.26
Livestock	54.70	14.56	29.05
TOTAL	**275.21**	**73.26**	**185.31**

4.5.5 National Park

Introduction
National parks are a major type of land-use in Tanzania, being over 30% of the country's land cover (Kidegehesho & Mtoni, 2008). Two major parks exist within the Maasai steppe, the Tarangire National Park (TNP) and the Manyara national park (MNP).

Ecosystem services valuation
The benefits received from the NPs can be seen as recreation for national and international visitors or tourism revenues for the Tanzanian government. Here we take the part of tourism revenue that can be attributed to ecosystems using the attribution method described in the Method section (chapter 3.2 Formal definitions). Revenues of the year 2006/2007 and the size of each park were extracted from a discussion of the benefits NPs receive from pastoralism (Nelson, 2009). In this case value transfer is not necessary as data is available for the policy site itself.

Costs are based on a formula developed by the African national park expert V. Booth (Booth, 2000). It includes maintenance costs and personnel costs required to run the park as a function of park size. Costs related to wildlife damage to the farms surrounding the park and the reciprocal damage of poachers killing wildlife are not included.

As for rangeland, no market rental value of land is available for nature reserves. Because the user of the land in this case is the Tanzanian government, as a shadow price of land we take the annual rental rate of off-village land that the government charges to foreign investors. This rate is fixed annually and published by the government. It amounts to 5000 TZS/acre, equivalent to 16.07 USD PPP/ha (Tanzania Investment Center, 2014)

Results

Table 17. Results for one hectare of National park

item	unit	value
Visitor expenditure	PPP2014/vis	95
Visitor density	vis/ha	0,8
Benefit before attribution to ecosystem	PPP2014/ha	72.61
% attributed to ecosystem		71.6%
Costs	PPP2014/ha	6.4
Ecosystem benefit	PPP2014/ha	51.96

Benefits and costs from the NPs are to be compared to the rangeland and mixed farming services. The visitor density, the revenue per visitor and cost per area of NP are averaged over the two parks, weighted by the size of each. Results are shown in Table 27.

5 Results

5.1 Value of ecosystem services

This section presents a summary of the ecosystem valuation results. Revenues per hectare are much higher for farmland than for other land use types, so the value of ecosystem services on a per hectare basis is the highest in agricultural land, followed by national park and rangeland.

Livestock keeping in rangelands and crop production are the two main sources of ecosystem value in the Maasai steppe, accounting respectively for 28% and 30% of the total annual ecosystem benefits in the region, followed by tourism (18%), water and wood products (6 and 5% respectively).

On a per hectare basis livestock accounts for as much as two thirds of the overall value creation from ecosystems in rangeland. Agriculture, however, creates as much as five times more value per hectare than livestock in rangelands. Crop production has the absolute highest value per hectare across all types of land use, as agriculture is a land efficient way of creating value from ecosystems.

The value of crops calculated here is in fact not the revenue from crop production, but the share of that value that can be attributed to ecosystems. This crop value indirectly represents the value of all ecosystem services (including pest control, pollination, water and nutrient cycling) that contribute to crop yields[37].

The recreational value of national parks, represented by tourism revenues, has the second highest value per hectare in the region. This value indirectly represents the most marketable form of value of the habitat and biodiversity that support big mammals such as elephants, giraffes and wildebeests in the Maasai steppe.

Overall rangeland creates the lowest value per hectare, but due to its size relative to the other land allocations on the steppe, it contributes the most to the total annual benefits for the whole Maasai Steppe.

Table 18. ES services for all land use types

Land use type	**Ecosystem Service**	**Annual value per hectare** (USD 2014 PPP/ha/yr)	**Whole Maasai steppe** (USD 2014 PPP/yr)
Rangeland	Food	0.10	209,250
Rangeland	Livestock	11.99	25,839,388
Rangeland	Medicine	0.68	1,470,472
Rangeland	Traded products	0.39	838,002
Rangeland	Water	2.81	6,067,115
Rangeland	Wood products	2.51	5,410,885
Agricultural land	Crops	58.70	27,082,540
Agricultural land	Livestock	14.56	6,717,582
National park	Tourism	51.96	16,522,622
TOTALS:			
Rangeland	**Total**	**18.48**	**39,835,112**
Farmland	**Total**	**73.26**	**33,800,121**
National park	**Total**	**51.96**	**16,522,622**

[37] A review of the link between regulating ecosystem services and agriculture can be found in Barbier (2013)

The value of rangelands is in reality tightly connected to the value of national parks, as the same population of wild animals that attract tourism migrate to graze outside the boundaries of the park every year. Interrelations of this kind between events on one type of land use and the effects on neighbouring ones and the implications on sustainable extraction of value can be studied by looking at Natural Capital values.

5.2 Value of Internal Natural Capital

Table 29 presents the results of the Natural Capital valuations, including the modelling of future changes in flow of benefits according to different scenarios.

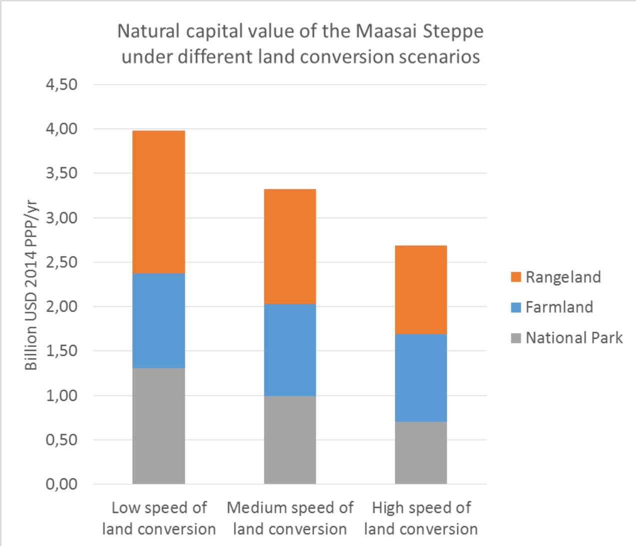
The internal value of Natural Capital in the Maasai steppe is calculated to be in between 2.6 and 4 Billion USD PPP, depending on the considered scenario.

Table 19. Maasai Steppe Natural Capital value under different land conversion scenarios

Land use type (Billion USD2014 PPP/yr)	Low speed of land conversion	Medium speed of land conversion	High speed of land conversion
National park	1.31	0.99	0.70
Farmland	1.06	1.03	0.99
Rangeland	1.60	1.29	1.00
Total	3.98	3.32	2.69

The Natural Capital value of the whole region as well as that of each type of land use are lower in the high conversion speed scenario than in the low speed scenario. Even agricultural land, that is expected to more than double in the foreseeable future in all scenarios decreases in overall value. This is a result of an increase in farmland area that cannot compensate for a decrease in farmland productivity due to land degradation. The value of the rangeland drops even more dramatically between scenarios. This results from the steep rates of decline in livestock productivity following a further expansion of farmland in the region. The value of national parks is also affected by the rate of conversion to agriculture, as a consequence of encroachment upon migratory corridors by agriculture. The total loss of Natural Capital between the High conversion and Low conversion speed is estimated to be 1.3 Billion USD PPP.

Figure 10. Natural Capital under different scenarios in billion USD



5.3 Carbon losses

Values of carbon stocks per hectare are derived from a literature survey, and the difference in terms of land cover between now and after 20 years is translated into losses of carbon stocks. Values of carbon stocks used for rangeland and degraded land are 87 tC/ha and 31 tC/ha respectively, based on values for pristine and degraded grassland in South African rangeland (Petz et al. 2014). Carbon stocks in agriculture are expected to be in between this range as gradually they decrease from the former to the latter in the years of farming. we assume the average difference between farmland and degraded land to be at the middle point between pristine rangeland and farmland, or 59.1 tC/ha. In practice the loss of carbon in farmland is not likely to happen linearly with time, but can follow more complex dynamics which depend on climate and farming techniques (FAO 2004), but a more accurate estimate was not available.

Carbon stocks differences between t = 0 and t = 21 were calculated and monetized using the social cost carbon of 128 $/tCO2eq (U.S. IAWG 2013, see also Part II Chapter 2.3).

The following table shows the monetized values of carbon losses under three different land conversion scenarios. As expected, higher conversion rate to farmland results in the highest value of carbon losses. The value of carbon stocks is approximately one order of magnitude higher than the internal value of Natural Capital. An attempt to curb conversion to farmland in the Maasai steppe could save up to 70 billion US$ to the global economy. These results highlight that the global community has an important stake in ecosystem degradation dynamics in this region.

Table 20. Carbon losses due to land conversion under different scenarios

Scenario	Unit	Rate of conversion to farmland		
		HIGH	MID	LOW
C losses due to land conversion in the Maasai steppe	USD2014PPP	8,51E+10	2,53E+10	1,69E+10

6 Sensitivity analysis

6.1 Scope and design

The exact relation between ecosystem and land benefits on land use and degradation patterns require advanced modelling that is not within the scope of this study. Land degradation is modelled here as a moderate reduction in land productivity under different rates of land conversion. The rates are chosen based on literature surveys of land conversion drivers in the Maasai steppe. In order to test the robustness of the model outcomes and their sensitivities to the initial assumptions a sensitivity analysis is carried out.

Parameters in scope for the sensitivity analysis are:

- Productivity growth rates of each type of land use and in each scenario.
- Extent and speed of land degradation.
- Maximum area of arable land in the region.
- Discount rate.

Excluded parameters, such as the current level of ecosystem services or land attribution coefficients, are considered to have a much lower uncertainty compared to growth rates and are not tested as part of the sensitivity analysis. Scenarios that include major improvements on land management and infrastructure are not in scope for the sensitivity analysis.

6.2 Method

In order to check the robustness of the model clusters of parameters are changed and the value of Natural Capital in the three scenarios is recorded. Each cluster represents a group of related parameters that operationalize expected changes in ecosystem services.

Changing the parameters of entire clusters allows the assessment of:

1. The effect of a relative (between scenarios) parameter change on the relative value of Natural Capital
2. The effect of an absolute (across scenarios) parameter change on the absolute value for Natural Capital in all of the three scenarios within a cluster

Six clusters of 1-6 parameters each are changed as illustrated in Table 31 and Table 32.

Each cluster of parameters has a number of possible values (options). Each option for each cluster is tested against all other options for all other clusters. The results for each of the 3750 combination are compared, in terms both of ranking between scenario and total Natural Capital value, to identify the most sensitive parameters and assumptions.

Table 21. Key for table 32

Parameter cluster		Option 1 ID	Option 2 ID	Option 3 ID	Option 4 ID	Option 5 ID
Period	Scenario	Value	value	Value	value	value
Period	Scenario	Value	value	Value	value	value
Period	Scenario	Value	value	Value	value	value

Table 22. Values used in the sensitivity analysis

Rangeland animal density		RL_used	RL_tight _min	RL_more	RL_tight_ max	RL_zero
1-20 y	LOW	0%	0%	1%	0%	0%
1-20 y	MID	-1%	-1%	0,5 %	-0,5%	0%
1-20 y	HIGH	-6%	-2%	0%	-1%	0%
Farmland crop productivity /ha		FL_steady	FL_used	FL_rain	FL_zero	FL_wide
1-20 y	LOW	-1%	-1%	1%	0%	1%
1-20 y	MID	-2%	-2%	0%	0%	0%
1-20 y	HIGH	-3%	-3%	-1%	0%	-2%
20+	LOW	0%	-1%	-1%	0%	1%
20+	MID	0%	-2%	0%	0%	0%
20+	HIGH	0%	-3%	-1%	0%	-2%
National Parks Visitors/ha		NP_less_now	NP_used	NP_less_later	NP_more	NP_steady
1-20 y	LOW	4%	6%	6%	8%	6%
1-20 y	MID	4%	6%	6%	8%	6%
1-20 y	HIGH	2%	4%	4%	6%	4%
20+	LOW	1%	1%	0%	1%	0%
20+	MID	0%	0%	-1%	1%	0%
20+	HIGH	-1%	-1%	-2%	0%	0%
Discount Rate	All	Disc_low		Disc_med		Disc_high
	All	2.5%		3%		5%
Farmland Maximum	All		Fl_max_less		Fl_max_more	
	All		35%		70%	
Abandoned land		DD_used	DD_less	DD_more	DD_less_more	DD_more_less
Initial share	All	2%	1%	4%	1%	3%
Expansion rate (0-20 yr)	All	11%	3%	11%	7%	11%
Share of converted land at steady state A_{Deg} / (A_{Ag} + A_{Deg})	All	50%	10%	60%	30%	55%

6.3 Results

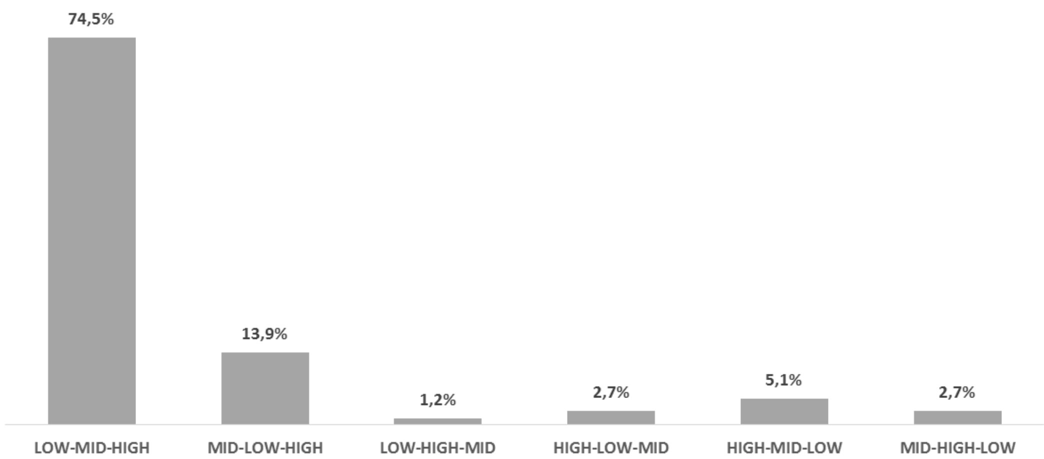
The sensitivity analysis validates the main result of the model, namely that a low rate of conversion to agriculture has the highest ranking in terms of Natural Capital value. Under 3750 different combinations of parameter clusters, the variation of the Natural Capital value under different parameter values is within one order of magnitude, as can be seen in Table 33.

Table 23. Natural Capital value variance

Scenario	Natural capital value (US$ billion)	5th percentile (US$ billion)	95th percentile (US$ billion)
High speed of land conversion	2.7	1.6	4.9
Medium speed of land conversion	3.3	1.8	5.2
Low speed of land conversion	4.0	2.0	6.0

This is due to the discount rate, which has a major impact on the total Natural Capital value. This does not influence the ranking, as the discount rate is constant across scenarios. The ranking is sensitive to certain clusters of parameters. In Figure 16 the frequency of the different rankings is shown.

Figure 11. Ranking under different combinations of parameter clusters



The probability that the ranking of scenarios would not change under a different set of assumptions was found to be 74,5%, meaning that one in four of the combinations tested have a different ranking among scenarios. However the probability that the Low conversion rate scenario would have a natural capital value above the High conversion rate scenario is 89%. The sets of assumptions that were found to most likely have high natural capital value in the High conversion scenario are those where the productivity trends of farmland (soil fertility trends), rangeland (average animal density in the remaining pastoralist areas) or national parks (revenues) would be considered the same in all scenarios. This means that agriculture could well expand without reducing natural capital, when it would do so without causing land degradation or posing threats to pastoralism and tourism.

7 Discussion

Part III presented an in-depth analysis of the internal value of Natural Capital in the Maasai steppe performed as part of a broader study on the impact on Natural Capital of the livestock sector worldwide. The underlying methodology developed for valuation of ecosystem services was also illustrated. In this section the results are reviewed in the context of existing literature, the main methodological limitations are discussed and the relation with the rest of the analyses presented in this work is explained.

7.1 Discussion of results

The value per hectare calculated for rangeland is within the same order of magnitude as studies in other dryland regions of Sub-Saharan Africa, namely 59 PPP2014/ha in Jordan and 3 PPP2014/ha in Botswana. Similar services were considered in both studies.

The average value of 1244 USD2007/ha provided by TEEB includes many regions with higher productivities. In addition, this average includes regulating services that are excluded, to prevent double counting (as they regulate future serve provisioning services), or are out of scope in this analysis.

Table 24. Rangeland annual benefit, comparison with literature

Rangeland	unit	value	Source
Maasai steppe	PPP2014/ha	19	This study
Global average grasslands	USD2007/ha	1244	van der Ploeg S., de Groot D., Wang Y., The TEEB Valuation Database: overview of structure, data and results, FSD (2010)
HIMA-Jordan, IUCN	PPP2014/ha	59	IUCN, Natural Resource Economic Valuations: Environmental Economic Valuation of the HIMA System, The Case of Zarqa River Basin – Jordan, IUCN (2011).
Botswana - IIED	PPP2014/ha	3	Arntzen J., Economic Valuation Of Communal Rangelands In Botswana: A Case Study, IIED London (1998).

The value per hectare of mixed farming is extracted from a comprehensive review of mixed farming in the Maasai steppe and is considered accurate (Castel 2006). A comparison with national scale values provided by ILRI show that the value of 275 PPP2014/ha is below the average of 1615 PPP2014/ha. However, the Maasai steppe is a relatively dry region and with very limited used of irrigation and other inputs compared to other agricultural regions in Tanzania. Finally, the National park land value included only cultural services and the data used is provided by the official authority of NPs in Tanzania, namely TANAPA. Regulating services provided by National Park were not within the scope of this study.

7.1.1 Relation to other components of this research project

This analysis is part of a broader TEEB study on livestock systems comprised of a top-down view on the Natural Capital impacts of the animal industry worldwide and a bottom-up valuation of environmental externalities of selected production systems, or snapshots, in a number of countries.

The valuation of Natural Capital in the Maasai steppe offers a deep dive into one of the snapshots of the bottom-up analysis, a pastoralist livestock system in Tanzania, and its linkages with neighboring ecosystems and related economic activities, such as farming and tourism, to uncover the hidden benefits of East African rangeland ecosystems and the hidden costs of agricultural intensification. This deep dive is a complementary approach to the analysis of selected environmental externalities, such as greenhouse gas emissions or water pollution, which facilitates comparisons across different systems on a product basis. It focuses on interconnections between the economy and Natural Capital at the local scale and on how region specific social-ecologic systems and trends determine the value of ecosystems.

This valuation, in particular, complements the analysis of the land occupation of different types of animal husbandry systems. That study highlighted that Tanzanian pastoralism has the highest land requirement per unit of protein produced among the investigated snapshots, because it is a low animal density, low productivity livestock system. The in-depth analysis of the Maasai steppe colors this picture in a different tone, by showing that pastoralist land management is also related to tourism revenues in the neighboring National parks and that, although less efficient, it is more sustainable over time than the alternative of sedentary farming, which depletes soil fertility and leads to land degradation. Giving a value to these interrelations demonstrates how Tanzanian pastoralism is preserving Natural Capital for local populations.

As a final point, there is an important difference between this analysis and the bottom-up valuation of snapshots and their externalities. Mixed agriculture as described in this Part III of the report consists of both crop farming and cattle keeping but it does not correspond to the Tanzanian mixed dairy system presented in Chapter 4 of the main report. That system refers to a highland system in a region with higher rainfall than the Maasai steppe, slightly higher use of agricultural inputs and use of improved cattle breeds, which was not observed in the Monduli and Simanjiro district studied here.

7.2 Discussion of methodology

The model here described puts in relation the speed of deterioration of agricultural and natural ecosystems with the speed of expansion of agriculture, based on a thorough literature review on land use patterns and ecosystem services in the Tanzania Maasai steppe and similar neighboring regions.

A key feature of the methodology used, compared to other ecosystem services analyses, is the valuation of agro-ecosystem services attributing part of the agricultural value to human inputs and part to ecosystems. This allows (1) the valuation of intermediate services as a part of the final value of their benefit and (2) a comparison between natural land and managed land (like farmland). Another one is the inclusion of future trends in ecosystem services value to look beyond the present situation and take into account the sustainability over time of the studied land management types.

Land degradation trade-offs are usually studied using Total Economic Value (TEV). Compared to the TEV methodology, the one introduced here has a more limited scope, focusing on direct value only, but a longer time scale, looking not only at economic value in a given year but also at whether it can be sustained over time. Indirect value can be considered as equivalent to the External Natural Capital value, the value for non-local stakeholders, which was out of scope for this analysis. The two methodologies could be combined, limited to types of value that can be quantified with a market based valuation, for example calculating option value as the Natural Capital value in a best case scenario.

7.3 Limitations

There are limitations in the methodology used here to analyze the land conversion trade-offs.

- Some regulating ecosystem services that could have high materiality for rangeland are not included in the analysis. This is connected to the methodological choice of focusing on transfer of values from study sites closely related to the Maasai steppe to drive down uncertainty of the results. The lack of fundamental scientific research on the benefits (let alone the underlying biophysical mechanisms) from water cycling, water purification, carbon sequestration in the region led to the decision to focus mainly on provisioning services. A number of regulating ecosystem services that are considered intermediate are however included in the valuation of final ones[38]: the value of soil fertility and nutrient cycling is essentially valued by looking at the contribution of ecosystems to value creation as crops, animal protein and wood products.
- Future trends in ecosystem services value, be it for pastoralist systems, mixed farming or tourism, were not modelled biophysically in relation to land use change but estimated for discrete land use scenarios, based on relevant literature. An actual investigation into the relations between land conversion and increases or decreases in value creation from ecosystems has never been carried out to the author’s knowledge and it was out of scope for this analysis.

[38] For a distinction between final and intermediate ecosystem services see (SEEA, 2012)

- Land degradation is not modelled as a gradual process, but in a simplified model based on the little scientific knowledge available on this phenomenon in the Maasai steppe. As for the previous point, a biophysical model relating land degradation with farming practices would have been preferable. Furthermore land degradation is assumed to be irreversible and definitive. Although research seems to point out that degraded land is unfit for both farming and livestock keeping (see Land Cover section, chapter 4.5.1 of this Part III), a more refined analysis of land degradation dynamics, for example using a spatial model, would allow to reach new and useful insights especially relating to the possibilities for land restoration.
- Current socio-economic trends besides land conversion, such as agricultural productivity, population growth, access to water, tourism or trading of grassland products, were assumed to remain constant for the foreseeable future. The only exception was made for high growth trends, which were assumed to slow down with time to stay within the biophysical limits of the ecosystems. What this analysis does not look into is therefore what would happen to Natural Capital under different possible outcomes than business as usual (i.e. investment in intensive farming, in making pastoralism more efficient, combining pastoralism with eco-tourism, introduction of agro-ecological practices or sustainable land management, development of industrial activity or other natural resource-related economic activities).
- An analysis of the social implications of land cover change dynamics lies outside the scope of this study. Because the subject is the total value generated from ecosystems questions of fair distribution of the benefits and number of beneficiaries reached are not addressed. Similarly, broader economic questions, related for example to income generated under different scenarios were out of scope for this study.

8 Conclusions

The results presented in this document show that the internal value of Natural Capital is highly influenced by land use and degradation dynamics taking place in the region. In particular slowing down the rate of conversion to agriculture will preserve the value of the remaining rangeland and national parks, which are approaching critical thresholds as migratory corridors for wild animals and cattle herds are being replaced by farms. This, together with the fact that the local agricultural system is unsustainable and leads to land degradation and abandonment, means that land conversion erodes the value of Natural Capital in the Maasai steppe. Therefore, even if farming in practice does make a more efficient use of land, more farming does not lead to more Natural Capital, as the price to pay for every additional hectare of agricultural land gets higher and higher with the shrinkage of the remaining rangeland.

8.1 Policy recommendations

The loss of internal Natural Capital means that ecosystem services that today produce wealth for local populations in a number of ways, from food provisioning to income generation, and fuelwood to fertile ground, will be lost in the foreseeable future. Some of the services will have to be replaced by the use of imported products and other capital goods while others, such as the cultural value of biodiversity which materializes as National parks' revenues, will be difficult to replace.

Creating incentives for farmers to avoid converting land in key migratory corridors and transhumance areas can lead to preservation of Natural Capital, but the fact that farming is practiced in an unsustainable way is another key driver of the loss of Natural Capital in the region. This suggests that investing in a form of farming that makes a sustainable use of land and can co-exist with pastoralism would be necessary to compensate for the value lost by reducing the size of rangelands.

8.2 Directions for future research

Based on the findings of this study and its limitations, the following directions for future research on ecosystem services, Natural Capital, land conversion and degradation in the Maasai steppe are identified.

- Carbon sequestration, hydrological processes, land degradation and ecosystem thresholds related to migratory and transhumance corridors should be studied from a biophysical perspective.
- The economic value of the benefits for humans of these ecological processes should be quantified.

- Social and economic implications for local and stakeholders of land conversion and degradation dynamics should be investigated.
- Studying the impact on Natural Capital value of the adoption of other farming and land management systems than those currently in place can shed light on possibilities for land restoration and sustainable options for the Maasai steppe.

9 References

Balmford, A. R. (2008). The Economics of Ecosystems and Biodiversity: scoping the science. European Commission. Cambridge, UK: European Commission.

Baltussen W. , A. de Blaeij, P. van Horne , T. Achterbosch , G. Kruseman, E.J.M.M. Arets, T. Vellinga, S.J. Hiemstra, W. Ouweltjes , V. Fobelets, O. A. Karachalios , L. Verkooijen , A. De Groot Ruiz, P. Galgani, N. Erlenborn, M. Tarin Robles , R. Lord, R. Hardwicke (2015) Valuation of livestock eco-agri-food systems: poultry, beef and dairy , Wageningen University and Research center, Trucost & True Price, The Hague

Barbier, B. E. (2013). Economics of the Regulating Services. Encyclopedia of Biodiversity (Second Edition), 45–54.

Barrio M., L. M. (2009). A meta-analysis of contingent valuation forest studies. Ecological Economics, 1023–1030.

Bayliss, K., & Tukai, R. (2011). Services and supply chains: The role of the domestic private sector in water service delivery in Tanzania. New York: UNDP.

Bekure, S., Leeuw, P. d., Grandin, B., & Neate, P. (1991). Maasai herding - An analysis of the livestock production system of Maasai pastoralists in eastern Kajiado District, Kenya. Addis Ababa: International livestock centre for africa. Retrieved from FAO CORPORATE DOCUMENT REPOSITORY: http://www.fao.org/wairdocs/ILRI/x5552E/x5552e08.htm#6.1.2%20division%20of%20responsibility%20and%20labour%20in%20livestock%20production

Booth, V. (2000). Options for the future status and use of saadani/mkwaja game reserve. Dar Es Salaam: Saadani conservation and development program. Retrieved from http://www.wildlife-baldus.com/download/Saadani%20-%20Options%20for%20Use.pdf

Brander, L. (2013). Guidance Manual on Value Transfer Methods for Ecosystem Services. Nairobi: Publishing Services Section, UNON. Retrieved from http://www.gwp.org/Global/ToolBox/References/Guidance%20Manual%20on%20Value%20Transfer%20Methods%20for%20Ecosystem%20Services%20%28UNEP,%202013%29.pdf

Brinkhoff, T. (2015). City population. Retrieved January 2015, from http://citypopulation.de

Castel, V. (2006). Novel forms of livestock and wildlife integration adjacent to protected areas in Africa, Socio-economic baseline survey. Rome: FAO. Retrieved from http://www.fao.org/fileadmin/templates/lead/pdf/tanzania/SocioEconomicSurvey-baseline.pdf

CGIAR. (2011). Dairy value chain in Tanzania: Background proposals for the CGIAR Research Program on Livestock and Fish. Nairobi: International Livestock Institutet CGIAR.

Davis, J. (2007). Total economic valuation of kenyan pastoralism. Nairobi: WISP. Retrieved from http://cmsdata.iucn.org/downloads/kenya_tev.pdf

De Groot, R. B. (2012). Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Services, 1(1), 50-61.

ELD Initiative. (2013). The rewards of investing in sustainable land management. Interim report of ELDI. Bonn: ELD.

FAO. (2002). Global Livestock Production and Health Atlas (GLiPHA). Retrieved from http://kids.fao.org/glipha/

FAO (2004). Carbon sequestration in dryland soils. World Soil Resources Reports. Rome: FAO.

FAO. (2006). Livestock's long shadow : environmental issues and options. Rome: FAO.

FAO. (2009). Criteria and indicators for sustainable woodfuel, case studies from Brazil, Guyana, Nepal, Philippines and Tanzania. Rome: FAO.

FAO. (2009). Sustaining communities, livestock and wildlife in the Maasai Steppe: vital facts, observations and policy actions. Rome: FAO.

FAO. (2009). The state of food and agriculture: Livestock in the balance. Rome: FAO.

FAO. (2014, 12 22). FAOSTAT. Retrieved from FAOSTAT: http://faostat.fao.org/site/291/default.aspx

FAO. (2014). GLIPHA. Retrieved November 2014, from FAO Glipha: http://kids.fao.org/glipha/

Fisher B., T. R. (2009). Defining and classifying ecosystem services for decision making. Ecological economics, 68(3), 643–653.

Foundation for Sustainable Development. (2014). Retrieved from The Ecosystem Services Partnership: http://www.fsd.nl/esp/80763/5/0/50

Gerber, P. J., Steinfeld, H., Henderson, B., Mottet, O. C., Dijkman, J., . . . Tempio, G. (2013). Tackling climate change through livestock – A global assessment of emissions and mitigation. Rome: Food and Agriculture Organization of the United Nations (FAO).

Hesse, C., & MacGregor, J. (2009, June). Arid waste? Reassessing the value of dryland pastoralism. Retrieved from IIED: www.iied.org/pubs/display.php?o=17065IIED

Homewood, K., Kristjanson, P., & Trench, P. C. (2002). Staying Maasai? Livelihoods, Conservation and Development in East African Rangelands. New York: Springer.

Homewood, K., Lambin, E. F., Coast, E., Kariuki, A., Kikula, I., Kivelia, J., . . . Thompson, M. (2001). Long-term Changes in Serengeti-Mara Wildebeest and Land Cover: Pastoralism, Population or Policies? PNAS, 12544-12549.

Homewood, M. K., Trench, P. C., & Brockington, D. (2012). Pastoralist livelihoods and wildlife revenues in East Africa: a case for coexistence? Pastoralism: Research, Policy and Practice, 2, 19.

Honigova, I., Vackar, D., Lorencova, E., Melichar, J., Gotzl, M., Sonderegger, G., . . . Chobot, K. (2012). Survey on grassland ecosystem service: Report on the European Topic Centre on Biological Divesity. Prague: Nature Conservation Agency of the Czech Republic.

Jacques, M. (2014, November). Tanzania: Water is life, but access remains a problem. Retrieved from World Bank - Africa Can End Poverty Blog: http://blogs.worldbank.org/africacan/tanzania-water-is-life-but-access-remains-a-problem

Kadigi, R. M., Kadigi, I. L., Laswai, G. H., & Kashaigili, J. J. (2013). Value chain of indigenous cattle and beef products in Mwanza region, Tanzania: Market access, linkages and opportunities for upgrading. Academia Journal of Agricultural Research, 1(8), 145.

Kidegehesho, J. R., & Mtoni, P. E. (2008). Who compensates for wildlife conservation in Serengeti? International Journal of Biodiversity Science and Management, 4, 112-125.

Kipkoech, A., Mogaka, H., Cheboiywo, J., & Kimaro, D. (2011). The total economic value of Maasai Mau, Trans Mara and Eastern Mau Forest blocks of the Mau Forest, Kenya. Nairobi: Lake Victoria Basin Commision Secretariat.

Kshatriya, M., Kifugo, S., Msoffe, F., Neselle, M. O., & Said, M. Y. (2007). Mapping Land Cover Changes in Simanjiro and Monduli Districts. International Livestock Research Institute.

Kumar, P. (2010). The economics of ecosystems and biodiversity: ecological and economic foundations. London and Washington: Earthscan.

Lal, R. (2004). Soil carbon sequesration impacts on global climate change and food security. Science, 204(5677), 1623-1627.

Landers, D., & Nahlik, A. (2013). Final Ecosystem Goods and Services Classification System (FEGS-CS). Retrieved from EPA Science Inventory: http://cfpub.epa.gov/si/si_public_record_report.cfm?dirEntryId=257922

Levinsohn, J., & Petrin, A. (2000). Estimating production functions using inputs to control for unobservables. NBER Working paper series.

Lindhjem, H. (2007). 20 years of stated preference valuation of non-timber benefits from Fennoscandian forests: A meta-analysis. Journal of Forest Economics, 251-277.

MacLeod, M., Gerber, P., Mottet, A., Tempio, G., Falcucci, A., Opio, C., . . . Steinfeld, H. (2013). Greenhouse gas emissions from pig and chicken supply chains – A global life cycle assessment. Rome: FAO.

Majule, A. E., Yanda, P. Z., Kangalawe, R. Y., & Lokina, R. (2012). Economic value assessment of land resources, ecosystems services and resource degradation in Tanzania. Dar Es Salaam: The Global Mechanism and University of Dar Es Salaam.

Maxwell, D., McKenzie, E., & Traldi, R. (2014). Valuing natural capital in business: Towards a harmonised protocol. Natural Capital Coalition. Retrieved from http://www.naturalcapitalcoalition.org/js/plugins/filemanager/files/Valuing_Nature_in_Business_Part_1_Framework_WEB.pdf

Mdoe, N., & Mnenwa, R. (2007). Study on options for pastoralists to secure their livelihoods: assessing the total economic value of pastoralism in Tanzania. TNRF. Retrieved from http://www.tnrf.org/files/E-INFO-RLTF_VOL2-PART5_Mdoe-N_Mnenwa-R_2008_Assessing_the-total_economic_value_in_Tanzania.pdf

Mlote S.N., M. M. (2012). Value addition of beef cattle fattening in the Lake Zone in Tanzania: Challenges and opportunities. Livestock Research for Rural Development, 24(6).

Mohammed, Y. S., Naini, I., & Douglis, C. (2009). Sustaining communities, livestock and wildlife: a guide to participatory land-use planning. Rome: FAO.

Monela, G., Chamshama, S., R., M., & D.M., G. (2005). A study on the social, economic and environmental impacts of forest landscape restoration in shinyanga region, tanzania. Dar Es Salaam: IUCN. Retrieved from http://cmsdata.iucn.org/downloads/flr_final_report_shinyanga_june2005.pdf

Msangi, A., Rutabingwa, J., Kaiza, V., & Allegretti, A. (2014). Community and government planning together for climate resillient growth: issues and opportunities from Longido, Monduli and Ngorongoro Tanzania. London: IIED. Retrieved from http://pubs.iied.org/pdfs/10075IIED.pdf

Msoffe, F. U. (2010). Land use change in maasailand : drivers, dynamics and impacts on large-herbivores and agro-pastoralism. Edinburgh: The University of Edinburgh.

Mwampamba, T. (2007). Has the woodfuel crisis returned? Urban charcoal consumption in Tanzania and its implications to present and future forest availability. Energy Policy.

National Bureau of Statistics, Tanzania. (2015, January). Retrieved from National Bureau of Statistics - NBS: Statistics for Development: http://www.nbs.go.tz/

Nelson, F. (2009). Natural Conservationists? Evaluating the Impact of Pastoralist Land Use Practices on Tanzania's Wildlife Economy. Arusha: Maliasili Initiatives.

Nijdam, D., Rood, T., & Westhoek, H. (2012). The price of protein: Review of land use and carbon footprints from life cycle assessments of animal food products and their substitutes. Food Policy, 37(6), 760-770.

OECD. (2014). Prices - Producer Prices Index. Retrieved December 2014, from https://data.oecd.org/price/inflation-cpi.htm

Opio, C., Gerber, P., Mottet, A., Falcucci, A., Tempio, G., MacLeod, M., . . . Steinfeld, H. (2013). Greenhouse gas emissions form ruminant sources - a global life cycle assessment. Rome: FAO.

Otte, M. J., & Chilonda, P. (2002). Cattle and small ruminant production systems in sub-Saharan Africa. A systematic review. Rome: FAO.

Petz, K., Glenday, J., & Alkemade, R. (2014). Land management implications for ecosystem services in a South African rangeland. Ecological Indicators 45, 692-703.

Rentsch, D., & Damon, A. (2013). Prices, poaching, and protein alternatives: An analysis of bushmeat consumption around Serengeti National Park, Tanzania. Ecological Economics, 91, 1.

SAFMA. (2004). Ecosystem services in Southern Africa: a regional assessment. Pretoria: Council for Scientific and Industrial Research.

Schaafsma, M. (2014). Mapping ntfp collection in Tanzania: A comparison of surveys. Norwich: CSERGE.

Sekar, N., Weiss, J., & Dobson, A. (2014). Willingness-to-pay and the perfect safari: Valuation and cultural evaluation of safari package attributes in the Serengeti and Tanzanian Northern Circuit. Ecological Economics 97, 31-41.

Steen, B. (1999). A systematic approach to environmental priority strategies in product development (EPS). Version 2000 – Models and data of the default method. Göteborg: Centre for Environmental Assessment of Products and Material Systems.

Stern, N. (2006). The Stern Review: The Economics of Climate Change. London: HM Treasury.

Tanzania Investment Center. (2014). Tanzania Investment Guide 2014-15. Dar Es Salaam: Tanzania Investment Center, TIC.

Tanzanian National Parks Authority (TANAPA). (2014, December). The official site of Tanzania National Parks. Retrieved from http://www.tanzaniaparks.com/

The Whole Village Project. (2010). Village reports for Engaruka, Migombani, Naitolia and Selela. Minneapolis and Arusha: University of Minnesota.

Tol, R. (2008). The Social Cost of Carbon: Trends, Outliers and Catastrophes. Economics: The Open-Access, Open-Assessment E-Journal.

UN/EU/FAO/IMF/OECD/World Bank (2014a). System of environmental-economic accounting 2012 - Central framework.

UN/EU/FAO/IMF/OECD/World Bank (2014b). System of environmental-economic accounting 2012 – Applications and extensions.

UN/EU/FAO/IMF/OECD/World Bank (2014c). System of environmental-economic accounting 2012 – Experimental Ecosystem Accounting.

UNCTAD. (2014, December). Gum Arabic commodity profile. Retrieved from UNCTAD.org: http://www.unctad.info/en/Infocomm/AACP-Products/COMMODITY-PROFILE---Gum-Arabic/

UNU-IHDP, UNEP. (2014). Inclusive Wealth Report 2014: Measuring progress toward sustainability. Cambridge: Cambridge University press.

URT. (2012). National Sample Census of Agriculture 2007/2008, Arusha and Manyara Regions. Dar Es Salaam: United Republic of Tanzania.

US IAWG. (2013). Technical Support Document: Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866. Washington, DC: Interagency Working Group on Social Cost of Carbon, United States Government.

Vellema W. (2014). Gum arabic collection in northern Kenya: unexploited resources, underdeveloped markets. Africa focus 27, 69-86.

WageIndicator Foundation. (2015). Wage Indicator. Retrieved January 2015, from www.wageindicator.org

Wekesa, C., Luvanda, A., Muga, M., Chikamai, B., & Makenzi, P. (2013). Market chain analysis of gum arabic trade in Kenya. Octa J. Environ. Res. 1, 93.

Westhoek, H., Rood, T., van den Berg, M., Janse, J., Nijdam, D., Reudink, M., & Stehfest, E. (2011). The Protein Puzzle - The consumption and production of meat, dairy and fish in the European Union. The Hague: PBL Netherlands Environmental Assessment Agency.

World Bank. (2014). PPP conversion factor, private consumption . Retrieved November 2014, from http://data.worldbank.org/indicator/PA.NUS.PRVT.PP

WTTC. (2014). Travel & Tourism economic impact 2014 Tanzania. London: World Travel & Tourism Council. Retrieved from http://www.wttc.org/-/media/files/reports/economic%20impact%20research/country%20reports/tanzania2014.pdf

True Price, 2016, *Bottom up methodology TEEB Animal husbandry*. Amsterdam, The Netherlands

Authors (True Price): Pietro Galgani, Vincent Fobelets, Orfeas Agis Karachalios, Lana Krisanto, Leonardo Verkooijen, Adrian De Groot Ruiz

www.trueprice.org