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Review

Methods to Assess the Impacts and Indirect Land Use Change Caused by Telecoupled Agricultural Supply Chains: A Review

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Received: 15 December 2018; Accepted: 20 February 2019; Published: 22 February 2019

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Abstract: The increasing international trade of agricultural products has contributed to a larger diversity of food at low prices and represents an important economic value. However, such trade can also cause social, environmental and economic impacts beyond the limits of the countries directly involved in the exchange. Agricultural systems are telecoupled because the impacts caused by trade can generate important feedback loops, spillovers, rebound effects, time lags and non-linearities across multiple geographical and temporal scales that make these impacts more difficult to identify and mitigate. We make a comparative review of current impact assessment methods to analyze their suitability to assess the impacts of telecoupled agricultural supply chains. Given the large impacts caused by agricultural production on land systems, we focus on the capacity of methods to account for and spatially allocate direct and indirect land use change. Our analysis identifies trade-offs between methods with respect to the elements of the telecoupled system they address. Hybrid methods are a promising field to navigate these trade-offs. Knowledge gaps in assessing indirect land use change should be overcome in order to improve the accuracy of assessments.

Keywords: impact assessment; indirect land use change; telecoupling; agricultural commodities; food systems; life cycle assessment; sustainability; supply chain

1. Introduction

In current globalized economies, the stages of the life cycle of a product (from raw material extraction, manufacturing, distribution, consumption to end of life) occur across geographical scales. The increasing international trade of agricultural products brings high revenues but has increased environmental externalities across the globe [1–3]. The supply chain of agricultural products, defined as the set of processes and activities needed to produce and deliver a product [4,5], demands large quantities of resources such as water, land, energy, fertilizers and pesticides and generates large quantities of waste, pollutants and emissions [2,6]. The activities along the supply chain of these products can contribute significantly to climate change, eutrophication, land use change, biodiversity loss, resource depletion, water, soil and air pollution and other impacts that pose local and global environmental threats [2,7]. Global forces play an important role in modulating local impacts, therefore, a correct environmental impact assessment demands an improved understanding of the telecoupled nature of the drivers and effects involved [8]. We refer to the telecoupling framework where the term "telecoupled" is used to describe the socioeconomic and environmental interactions occurring across distances that influence a system [9] (in this case the system is an agricultural telecoupled supply chain). The multiple dimensions of the global sustainability challenges are not necessarily aligned and trade-offs between these dimensions may occur. Therefore, working towards more sustainable supply chains requires a deep understanding of the global telecoupled dynamics to ensure that negative trade-offs between sustainability dimensions or locations are limited.

The telecoupling framework [9] is helpful to conceptualize the relevant processes involved in international trade as it describes how the life cycle stages of a given product and the impacts generated might occur across temporal and geographic scales due to the complex socioeconomic and environmental interactions between the embedded, multiple systems. Beyond its geographic and temporal outspread, international trade entails complex dynamics such as cause-effect feedback loops, spillovers and leakage of impacts, legacy effects, time-lags, cascading effects and non-linearities [9,10]. These dynamics imply that agricultural systems embedded in international trade and their impacts are telecoupled. Notwithstanding their inherent presence, impact assessment methods usually describe these complex dynamics as external variables, mostly excluding them from the core analysis [11–13]. One reason for this is the lack of integration between methods coming from the social, environmental and economic sciences that are needed to evaluate such diverse impacts [13]. Another reason is the inexistence of suitable methods able to fully incorporate these telecoupling dynamics across different spatial and temporal scales into the analysis.

Previous reviews of environmental impact assessment methods applicable to agricultural products have had different focus. Ness and colleagues [14] categorized tools for sustainability assessments, including indicators, product-based assessment tools and integrated methods. Herva and colleagues [15] synthetized the environmental indicators commonly used by corporations to evaluate the environmental performance of their products and processes. Čuček and colleagues [16] reviewed several social, economic, environmental and composite footprints used to evaluate sustainability with the goal of clarifying definitions, calculation methods and units used. Henders and Ostwald [3] analyzed several methods used to account for land-related leakages caused by policy actions and international trade at global and aggregated scale without focusing on the specific supply chains of products. Bruckner and colleagues [17] analyzed the capacity of some physical, environmental, economic and hybrid assessment methods to account for the land footprints of agricultural, forestry and livestock products. Verburg and colleagues [18] reviewed and compared methods to model human-environment dynamics with special emphasis on feedbacks and teleconnections as key characteristics of the Anthropocene. Millington and colleagues [19] described the capacity of agent-based, system dynamics and equilibrium models to represent telecoupled food trade systems and proposed a method for their hybrid integration. Previous reviews referring to telecoupled dynamics have focused more on top-down approaches, arguing that they cover the global dynamics that characterize a typical telecoupled system. However, none has specifically analyzed the capacity of methods to assess, in a spatially explicit manner, the indirect land use changes (iLUC) caused by agricultural supply chains in specific locations (bottom-up approach) while also considering the non-local drivers (top-down approach) that shape the impacts of telecoupling systems.

This review identifies and compares the following methods that are available to assess the direct and indirect environmental impacts caused by the supply chain of traded agricultural products: life cycle assessment, environmental footprints and indicators, agent-based models, system dynamics models, equilibrium models and land use models. We aim to compare these methods on the extent to which they can evaluate the impacts of telecoupled systems. Agricultural production is inherently embedded in socio-ecological systems where humans and the environment interact. Since socio-ecological systems show high spatial variation, the methods to model them are better suited if they have a spatially-explicit character [20]. Socio-ecological systems also have high temporal variations, therefore, we also emphasize the temporal focus of these methods. The sustainable management of complex telecoupled production systems requires that methods have an adequate spatial and temporal coverage to be policy-relevant [12,18]. Because in agriculture, livestock and forestry, land systems play a central role [21], we use land use change as a bridge concept to analyze the social, economic and environmental impacts that are caused along supply chains. Therefore, we emphasize the capacity of methods to account for direct and indirect land use changes. We identify the strengths and weaknesses of the methods and highlight current knowledge gaps to propose future improvement pathways.

2. Materials and Methods

First, to frame the impact assessment methods, the representation of a telecoupled system for a generalized agricultural supply chain was elaborated based on different concepts available in the field. The diagram shown in Figure 1 was elaborated by including the main systems and agents representing the social, economic, political and environmental processes embedded in such telecoupled systems, and the flows, feedbacks and impacts arising from their interaction. This representation follows Liu et al. [9] for the general telecoupling framework and Meyfroidt et al. [22] for the particular case of land-related dynamics. Specific political and economic processes are based on Lambin et al. [23] and Albareda et al. [24]. Social processes are based on Lenzen et al. [25], Vermeir and Verbeke [26] and Cummins et al. [27]. Environmental processes are based on Rasmussen et al. [28] and Lambin and Meyfroidt [29].

Subsequently, a search of methods used to assess the environmental impacts of agricultural commodities was conducted in Science Direct, Web of Science and Google Scholar using a combination of the following words: impact assessment, telecoupling, agricultural supply chain, sustainability assessment, agricultural products, land use change, international trade, footprints, indicators, life cycle assessment, input-output analysis, deterministic equilibrium models and agent-based models. The first search round included the key word "review" to find review articles. The literature references from these reviews were also used to deepen the analysis of specific methods. A second search round did not include the word 'review' to include all existing relevant methods. Finally, a comparative description of the capacity of methods to assess telecoupled impacts was carried out. This analysis was based on the following criteria:

- System boundary definition: The ability of assessment methods to represent actual impacts highly depends on the broadness of the system boundaries. We evaluated the capacity of methods to account for both top-down (global scale) and bottom-up (local scale) dynamics. Truncation points either limit the capacity of models to capture specific global interactions affecting the system under study or cause models to lack granularity and the capacity to capture important fine-scale dynamics [13,30].
- Geographic and temporal approach: Because the impacts of telecoupled systems occur across distances and time, it is important to evaluate the spatial and temporal scope of methods. Methods can be better suited to local, regional or global scales and have a static or forecasting nature [18,31].
- Spatial explicitness: Landscape heterogeneity and local features of land systems largely condition the extent and intensity of environmental impacts [3,12]. Therefore, the capacity of methods to spatially allocate impacts is analyzed with the emphasis on direct and indirect land use change.
- Integratedness: The extent to which a given method is capable of incorporating social and economic dimensions along with the environmental ones, as suggested by the triple bottom line criteria of sustainability, is analyzed [31].
- Telecoupling dynamics: The capacity of methods to account for complex dynamics arising within telecoupled systems such as indirect impacts, feedback loops, spillovers, leakage, rebound effects, time lags, legacy effects and non-linearities is analyzed [9].

The methods were classified according to the following general method families: life cycle assessment (LCA), footprints and related indicators, rule and process-based models, deterministic equilibrium models and land use models (LUMs). Rule and process-based models include agent-based models (ABMs) and system dynamics models (SDMs). Deterministic equilibrium models include computable general equilibrium models (CGE), partial equilibrium models (PE) and input-output analysis (IO). These general method families were based on the categories previously set by Verburg et al. [18], Millington et al. [19], Herva et al. [15], Henders and Ostwald [3] and Bruckner et al. [17]. Finally, based on the analysis, some pathways for methodological improvements are proposed.

3. Results

A conceptual representation of the components and dynamics embedded in a generalized telecoupled supply chain is displayed in Figure 1. In this diagram, a simplified version of the most important agents embedded in consuming (receiving), producing (sending) and spillover systems are displayed. Moreover, the most important causes and impacts originating from the interactions between those systems are shown. The different color frames give the first indication of the components addressed by the different methods analyzed in this paper. Short definitions of the terms included in this graph are included in Table 1.

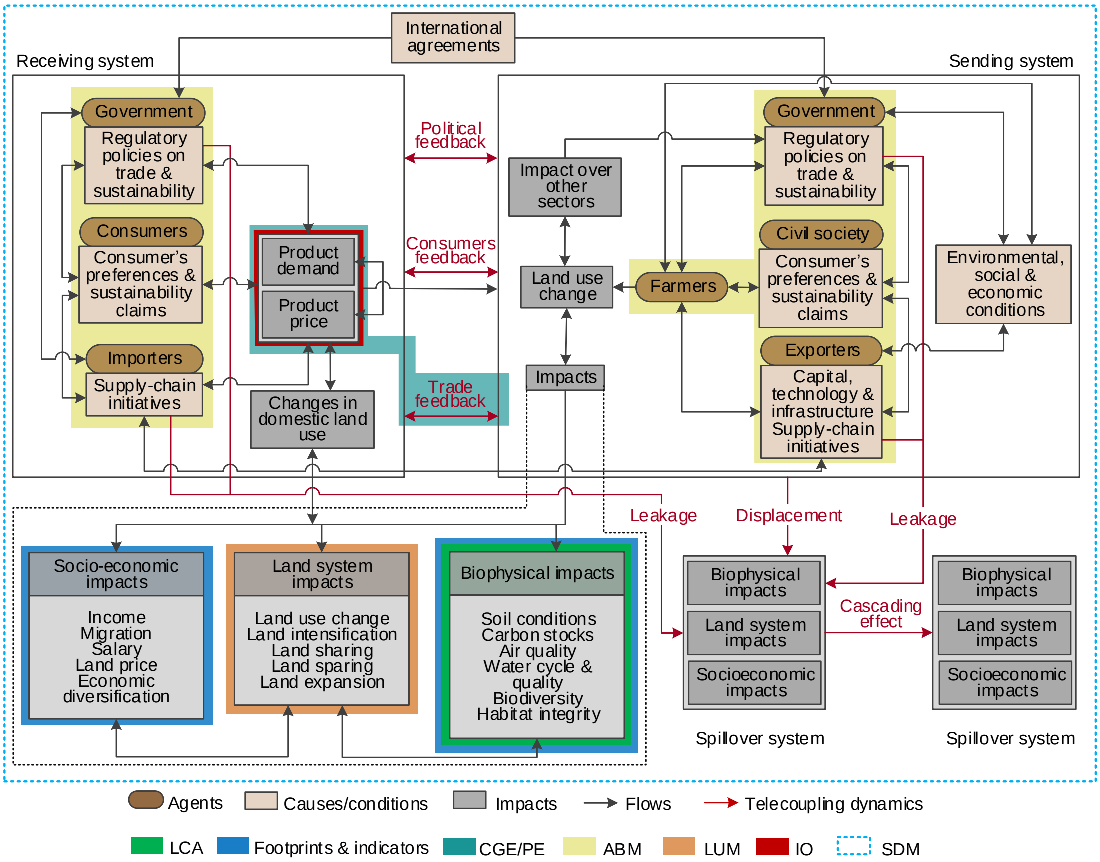

Figure 1. Representation of the main elements (systems and agents) and dynamics (flows, causes/conditions and impacts) embedded in a generalized telecoupled agricultural supply chain. Land system impacts, socio-economic impacts and biophysical impacts are represented repeatedly in spillover systems to describe the chain of impacts that can occur as a consequence of telecoupled systems. Different frame colors indicate the main focus of the methods reviewed in this paper to facilitate the understanding of the extent of their analytic capacities. LCA: life cycle assessment; CGE: computable general equilibrium models; PE: partial equilibrium models; IO: input-output analysis; ABM: agent-based models; LUM: land use models; SDM: system dynamics models.

The following subsections describe the main groups of methods to assess the environmental impacts caused by telecoupled agricultural supply chains. The main characteristics of the methods are summarized in Table 2. The description provided in this table refers solely to the main and most basic version of each method. Features of hybrid or integrated methods are analyzed along with the results and discussion.

Table 1. Definitions of main terms used related to the telecoupling framework. Based on Liu et al. [9].

Term	Definition
Feedback	Process by which an effect caused by one system to another system, impacts back to the first system.
Spillover system	System that is affected by/or affects the direct interaction of two other different systems (sending and receiving systems).
Leakage	Unintended negative effect of a sustainability action elsewhere than the target place.
Cascading effect	Process by which a system affects other multiple systems in sequence as a result of telecoupling dynamics.

3.1. Life Cycle Assessment

3.1.1. General Description

Life cycle assessment (LCA) is a quantitative screening tool used to identify environmental impacts occurring along the supply chain of a product or service starting from raw material extraction to end of life. Because it allows the identification of environmental hotspots, it has been used as a decision-tool for initiatives promoting sustainability [32,33]. LCA follows four steps: goal and scope definition, inventory analysis, life cycle impact assessment (LCIA) and interpretation [7,33]. The first phase defines the objectives of the study, sets the boundaries of the system and selects a functional unit to be used as a reference for all the impact calculations. The inventory phase compiles all data about the inputs and outputs of material and energy in each life cycle stage. LCIA uses this information to calculate indicators for the impact categories selected, which can include, for instance, global warming potential, biodiversity damage, eutrophication, ozone depletion and land use change. The conversion of data into impact units is done through weighting and standardization processes. The interpretation stage answers the questions set in the objectives of the study. LCA is attributional when it analyses current or past processes, and consequential (CLCA) if it aims to forecast the impacts of a given policy decision on the system under study [34–36].

3.1.2. General Limitations

LCA is mainly designed to perform fine-scale analysis on specific products or services; broader studies are constrained by the high data demand. Although in theory LCA analyses the entire supply chain of a product, in practice, it allows the exclusion of any input or output embedded in the life cycle and even excludes entire production stages that might significantly contribute to impacts. However, new applications with broader scope that allow the evaluation of sectors or entire economies are being developed [32]. The choice of impact categories and indicators is arbitrary [37] and depends on the goal of the assessment. This means that LCA studies lack standardization and comparability. Although guidelines of best impact indicators have been proposed [38–40], current LCA practices are still limited in their inclusion of important categories such as biodiversity, land-use change and social-economic aspects in an effective and righteous way [12,38,41,42]. Other comments in the literature on the limitations of LCA include the reliance on average (not place-specific) data of representative industries [17] and treating impacts as linear [43]. This non-differentiation of spatial heterogeneity limits the geographic-specificity of LCA. Spatially-explicit LCAs are needed to facilitate decision-making but they might be difficult to achieve because data about the location of suppliers and final consumers are rarely found [33]. Moreover, the use of pre-defined and year-specific conversion factors for the calculation of impacts [44] constrains the application of LCA to specific time periods, complicates comparison, and prevents the construction of long-time series. Furthermore, applications of LCA that integrate the social and economic factors that closely influence the environmental impacts need to be encouraged to provide better insights for sustainability [13,45–48].

Table 2. Comparative description of main desired attributes of methods to assess telecoupled systems.

Method Family	Telecoupling Aspects Analyzed	System Boundary Definition	Consider Landscape Heterogeneity & iLUC	Integratedness	Geographic Scale Suitability	Temporal Approach
LCA	Except by CLCA, it cannot account for feedbacks. Spillovers can be accounted with system boundary expansion.	Boundaries around a product or service usually exclude indirect impacts. Considers large-scale forces as external variables. Potential for expansion.	No	Usually only focus on biophysical impacts but the incorporation of social and economic ones is possible.	Local scale	Static
Footprints/Other Indicators	Feedbacks are not accounted. Spillovers can be accounted with system boundary expansion.	Boundaries strictly around territorial units or agents. Exclude several upstream and downstream impacts. Consider large-scale forces as external variables.	Limited because of the use of average transformation factors.	Indicators available for social, economic and environmental impacts.	Regional to global. Finer scale depends on data availability.	Static
CGE/PE/IO	CGE and PE analyze economic feedback loops and spillovers occurring at large scale. IO cannot include feedbacks.	Broad boundaries but poor granularity that ignores important intermediate causes and impacts. Boundaries around global economy or sectorial economies.	No. Some CGE and PE can account for iLUC from an economic perspective.	Based on economic factors but hybrid approaches can integrate social and environmental variables.	Regional to global	Forecast (CGE/PE). IO is static.
ABM/SDM	Can parameterize feedback loops and spillovers at least in a qualitative manner. Can analyze multi-temporal, multi-level and multi-disciplinary dynamics.	Flexible boundaries from narrow to broad ones. Boundaries around agents (ABM) or around the entire system (SDM). Multiple temporal and spatial scales.	No	Can parameterize environmental, social and economic factors.	From local to global depending on data availability.	Allow for scenario analysis.
LUM	Some models allow the integration of feedbacks and spillovers but only within the spatial extent of the study area.	Boundaries depend on the modelling approach but are more often broad. However, this means poor granularity that ignores important intermediate causes and impacts. Boundaries around the territory (ies) under study.	Yes	Depends on the model but they often emphasize more on biophysical factors.	Mainly regional to global depending on the model.	Forecast

Abbreviations stand for: LCA: life cycle assessment; CGE: computable general equilibrium models; PE: partial equilibrium models; IO: input-output analysis; ABM: agent-based models; SDM: system dynamics models; LUM: land use models.

3.1.3. Suitability for Telecoupled Systems

Despite the high flexibility of LCA, most current applications are product-centered and assume the dynamics occurring beyond the strict supply chain of a product (i.e., global economy, indirect impacts) as external variables to the model. Therefore, LCA applications need to expand the system boundaries to be able to account for upstream and downstream spillovers of impacts caused by telecoupling dynamics. With respect to that, the flexibility in the selection of impact categories is an attractive feature because it allows the consideration of several types of impacts. To quantify spillovers, first a deeper understanding of the cause-effect relationships arising along life cycle stages must be promoted, and the appropriate data to account for their impacts must be generated. The first issue is a challenge that extends beyond the LCA community, and the second one faces limitations regarding data transparency and accessibility. CLCA is a promising application because it extends beyond the purely biophysical focus of LCA by also analyzing the influence of the global economy on the system under study [33,34,36,49]. CLCA can also include non-linear impacts to study complex dynamics extended over time, such as time-lags and legacy effects. However, improved integration with tools having a forecast capacity would be necessary. The incorporation of other telecoupling dynamics such as feedbacks requires the integration of LCA with other methods capable of addressing these processes. Examples of input-output LCA and the integration of computable general equilibrium (CGE) and partial equilibrium models (PE) into LCA go in that direction [13,34,50]. Large-scale spatially-explicit analysis might be difficult to achieve or might carry high uncertainties due to the lack of place-specific data and the use of non-specific weighting and transformation factors [7]. The calculation of region and landscape-specific transformation factors needs to be encouraged. Studies such as van Zelm et al. [51] and Koellner and Scholz [52] shed light on this challenge. Data availability and quality are other related challenges, especially if LCA aims to assess the spillover impacts on land systems. Currently, most LCA studies that account for land use impacts are mainly based on indicators of land occupation and land transformation [12,53–55]. In this sense, the integration of LCA with land use models could contribute to improving the quantification and spatial allocation of land system impacts. Some recent applications in this direction include LUCI-LCA [12]. Based on land-change modelling and ecosystem services assessment methods, LUCI-LCA spatially assesses and forecasts the impacts of agricultural products on land-use and ecosystem services. There are several methodological approaches proposed to incorporate land use change in LCA [52,56–59]. However, most of them rely on area data of the occupied and transformed land and disregard the importance of iLUC caused by the interaction with other land uses, market forces and social dynamics. More explicit approaches to address iLUC and LCA include Di Fulvio and colleagues [60] who coupled LCA with the global land use model GLOBIOM (see Table S2 in the Supplementary Material) to quantify and allocate iLUC and biodiversity loss due to the international trade of biofuels. Other LCAs coupled with equilibrium models include [61,62]. Schmidt et al. [63] propose a conceptual framework to assess iLUC in LCA based on a biophysical model. Although the product-focus of LCA makes it an attractive tool to operationalize sustainable agricultural supply chains, there is no consensus about how to include iLUC in LCA, as is reflected in the variation in the approaches indicated above [64].

3.2. Footprints and Related Indicators

3.2.1. General Description

Environmental footprints are quantitative measures used to assess environmental performance and to track the human appropriation of natural resources [15,16,65]. Footprints are frequently used to assess human populations, countries, companies and, less frequently, products [6]. Footprint indicators have different definitions, scopes and calculation methods depending on the developer of a specific footprint measure [16]. Footprints calculate the amount of resources consumed (i.e., water, land, etc.) or released (i.e., greenhouse gases, nitrogen, etc.) and standardize them into particular units (usually area units or other units specific to the footprint) [16,21]. While LCA integrates different impact categories, most footprints account for a specific type of impact, such as impacts on water resources, greenhouse gases, biodiversity damage, land erosion, nitrogen pollution, among others and, as such, they can be incorporated into LCA as impact factors [16]. Footprints focused on social and economic aspects are in the early development stage [16]. The well-known ecological footprint (EF) is a composite measure aiming to evaluate sustainability in a comprehensive manner. EF accounts for the direct and indirect demand of resources and the required capacity to assimilate the waste and emissions generated by the subject under study in a given year. This evaluation focuses only on the following land use types: cropland, fishing grounds, grazing land, forest, built-up land and carbon uptake land [66,67]. EF calculations are based on biocapacity, which is the capacity of a certain type of land to regenerate its own resources and assimilate emissions [16,65,68]. To allow comparability, these specific biocapacities are later converted into global hectares by using equivalence factors that relate a them to the average global biocapacity. The human appropriation of net primary production (HANPP) is a footprint indicator that represents the capacity of the land to produce biomass (net primary productivity, NPP) accounting at the same time for the land depletion caused by human activities [69]. The embodied HANPP (eHANPP), measures the amount of HANPP caused by the supply chain of a product and it has been used to evaluate the impacts of trade. eHANPP accounts for the non-linear impacts of production activities because it uses NPP as basic measure, which is an attribute of land that can only be used once, and therefore, does not remain constant [1]. Contrary to EF, eHANPP is measured in biomass units (tons of carbon or dry-matter biomass) [70,71].

3.2.2. General Limitations

EF and eHANPP are indicators usually calculated for territories with political boundaries. These indicators do not account for upstream or for downstream resource demands and emissions generated beyond the studied system. They are usually better suited for regional or global studies because they rely on highly aggregated data (normally at national level) that lack geographic specificity. Fine-scale data at product or corporate levels are usually not available [16,65]. EF and eHANPP are static, meaning that they measure environmental performance only at a given point in time, and thus, cannot consider long-term effects [66,72]. EF has limitations in incorporating impacts from dynamic processes affecting the biocapacity of land, such as land degradation, intensive land use and resource depletion [66,72]. Moreover, EF does not consider the contribution of intensive agriculture and other technological improvements to productive systems and waste assimilation [72]. These methods provide easy-to-understand single measures but the trade-offs generated by the highly aggregated approach used to achieve them is a subject of debate [72].

3.2.3. Suitability for Telecoupled Systems

Most footprint studies set the boundaries of the system as political borders and exclude the telecoupling dynamics interacting with national accounts. EF, for instance, accounts for the resources consumed and emissions generated within a territory in a certain year without considering the impacts from exports and other external dynamics [72]. However, there are recent applications of EF based on input-output data that account for the impacts generated by international trade [65,72], thus capturing economic dynamics. eHANPP is based on the differential HANPP consumed and produced by a nation, and as such, it accounts for the impacts of international trade. The incorporation of spillovers in the calculations would demand enhanced traceability of the primary products used for the consumption or production of a country or agent and the waste and emissions generated. A concrete example in this direction is provided by Kastner et al. [73] who introduce an algebraic algorithm to trace the origin of the primary products used in a product consumed elsewhere based on bilateral trade data. Moreover, the inclusion of spillovers requires broadening the environmental parameters so far considered in these footprint indicators that can have significant impacts in supply chains (such as emission flows) [21]. The same applies for iLUC spillovers because the land footprint calculation methods only account for the direct land used and disregard the indirect land use changes caused, for example, by market and social dynamics and the competition between different land uses. The inclusion of long-term effects into the calculations would need an improved understanding of the dynamics occurring beyond the biophysical ones and would probably demand the calculation of prospective time series. These methods are not designed to account for feedbacks but could be coupled with methods able to address them. The assessment of non-linear impacts requires the improvement of conversion and equivalence factors, which in turn requires an improved understanding of cause-effect mechanisms. Finer-scale spatial analysis needs to overcome data limitations and the creation of place-specific conversion factors. This would allow the operationalization of landscape heterogeneity and will provide better tools for spatially-explicit analysis. Nevertheless, it is important to note that footprints and indicators can help to incorporate the specific environmental dimensions that other methods lack.

3.3. Deterministic Equilibrium Models

3.3.1. General Description

The economic and environmental impacts embodied in international trade have been modelled with economy-based methods such as input-output analysis (IO), computable general (CGE) and partial equilibrium (PE) models. IO is an empirical method to model market dynamics by calculating linear equations to describe inter-industry relationships in a given economy based on demand data [74,75]. It is traditionally based on transaction tables of yearly monetary flows between economic sectors of countries [74]. Recent IO analysis based on biophysical input-output tables have been proposed [76]. IO can be considered as a component of CGE and PE models [74,77]. CGE and PE are dynamic models that are built on the conceptual basis of IO but with important differentiations [75,77]. CGE and PE model markets and economic sectors, respectively, and provide future economic projections for a defined time frame based on optimized equilibriums (long term economic equilibrium solutions) between demand, supply and price [3,77]. CGE uses the technical coefficients obtained with IO but incorporates, among other things, supply and price data [74,75]. CGE and PE consider that both supply and demand regulate each other in perfect equilibrium through feedback loops (market feedbacks), which allows them to model international economic competition [75]. IO is better suited for small-scale analysis (i.e., national) whereas CGE is more appropriate for larger scales (i.e., regional or global). Both methods use input-output tables of global databases such as EoRA, GTAP, EXIOBASE or WIOD [78–82] as their core data (see Table S1 in the Supplementary Materials for more details). These tables report on the monetary transactions between countries and economic sectors including exports, capital formation and final consumption [83].

3.3.2. General Limitations

Due to the highly aggregated input data (non-differentiated production sectors) and the large set of assumptions, these methods often carry large uncertainties and lack granularity for fine-scale studies (i.e., product level) [2,3,11]. IO is static (no forecast capacity) because it only analyses past data and because it is based on constant coefficients that do not incorporate dynamics (i.e., price changes, technological changes and capital instability) that would provide future projections [77]. Moreover, IO assumes unlimited supply of products and homogeneous global prices [77]. CGE and PE assume rational economic behavior, economic equilibrium between supply and demand, homogeneous global prices and perfect economies (perfectly competitive markets, zero transaction costs and homogeneous product quality) [3,18,75]. Additionally, IO databases are available only for certain years, for certain countries and with distinct sector-detail information (usually highly aggregated). Finally, because they come from the economic field, these methods mostly do not include environmental and social interactions that can feedback on the economic dynamics.

3.3.3. Suitability for Telecoupled Systems

The system boundaries of IO, CGE and PE are set at broad scales (national, global and sectoral) which allows the incorporation of large-scale economic dynamics into the analysis and makes them very appealing for studying telecoupled systems. However, one disadvantage of such an approach is that these methods cannot consider place-specific dynamics, so fine-scale studies are difficult to undertake with these models. Because the CGE and PE models integrate non-linear economic dynamics by using complex solution algorithms [77], they are capable of accounting for market feedbacks loops and non-linear responses. Single IO analysis (at country level only) cannot integrate feedback loops but multi-regional IO (MRIO) analysis can [84]. Therefore, CGE, PE and MRIO are promising methods to assess telecoupled systems at global scale. Moreover, by considering the broader economic spectrum, CGE, PE and IO help to calculate economic spillovers and indirect, economy-linked impacts. These features have inspired LCA practitioners to integrate IO into their analysis with the goal of expanding the product-centered analysis of a normal LCA with the impacts of international trade on a supply chain [83,84]. Improvement in the resolution of these methods would need more disaggregated data about production sectors in databases. Continuous time-series and data for more countries are also needed. CGE and PE provide forecasts but to account for time-related telecoupling dynamics (such as time-lags, legacy effects and cascading effects) they need improved algorithms. The integration of environmental and social variables would also improve the forecasting practices and would allow them to reflect the full spectrum of dynamics. There are several hybrid approaches documented in this direction, such as the environmentally extended input-output analysis method that aims at analyzing the impacts that international trade has on the environment [84,85]. Besides, by integrating IO, footprints could estimate the embodied environmental impacts of production, consumption, imports and exports [21,84]. Regarding land systems, CGE, PE and IO have been combined with land use allocation models to analyze the iLUCs caused by international trade in a spatially-explicit manner [84,86–89]. These methods are highly suitable to evaluate feedbacks and spillovers (including iLUC) in a spatially-explicit manner if coupled with land use models and methods accounting for specific environmental impacts [3,17] as done by [60–62] with LCA. Such combinations are often made by downscaling the aggregated results with simple spatial algorithms following some kind of land suitability map [90]. However, transformations from monetary data to land use change values are based on global or regional average yields that deny the importance of land heterogeneities [3]. Moreover, downscaled land change patterns do not feedback on the global equilibrium calculations.

3.4. Rule and Process-Based Models

3.4.1. General Description

Two groups of rule and process-based models that are relevant to telecoupling are agent-based models (ABMs) and system dynamics models (SDMs). ABM is a computer-simulated method designed to understand the dynamics of a system and make forecasts about it. They model agents' behavior (agency) (i.e., humans, institutions or any social structure) and their interactions with their environment based on a set of decision rules that represent assumed or observed behavior [18,91,92]. These decision rules are represented in a finite space and time in a quantitative or qualitative manner in the model [93,94]. ABMs allow the parameterization of human interactions, adaptation and learning processes and the diversity and uncertainty of human behavior in a flexible and context-specific way [92,94]. On the other hand, SDMs are flexible computer-modelling frameworks to understand the behavior of a given system by representing the processes and relationships occurring between their elements. SDMs aim to go beyond the representation of cause-effect relationships towards a more holistic understanding of the functioning of systems [95]. To do so, SDMs use mathematical equations and decision rules to parameterize processes and relationships [30,96]. Because SDMs and ABMs are general modelling frameworks they can be applied at local, regional or global scales [19].

3.4.2. General Limitations

Although both ABMs and SDMs are very flexible modelling frameworks, in practice, their data demand can limit the expansion of the system boundaries, the integration of multi-level data and their application at broad geographic scales [18]. Therefore, ABMs are better suited for fine-scale studies and global applications are limited. The flexibility of ABMs and SDMs has been criticized for including several decision rules and assumptions that do not rely on any economic, physiological or sociological theory [92]. Due to the strong bottom-up approach of ABMs, the integration of exogenous dynamics operating at larger scales (i.e., global trade and price development of agricultural commodities) is limited [18]. Although in theory, ABMs and SDMs allow forecasting based on past trends, most of them are not used for this purpose but for understanding systems [92,97].

3.4.3. Suitability for Telecoupled Systems

The possibility of parameterizing agents' behavior has made ABMs useful for modelling socio-ecological systems (i.e., to model land use change) [20,91–94,98,99]. Recent articles argue that ABMs are a highly valuable tool to parameterize the complex dynamics occurring in telecoupled systems because they can parameterize feedbacks and address spillovers [93,100]. This is because ABMs can represent the external forces (such as climate change, global market influences, etc.) playing outside the boundaries of a defined system, and they can integrate data across multiple spatial and temporal scales [93,101]. SDMs are also suitable to incorporate these and other telecoupling dynamics (i.e., feedback loops, rebound effects and indirect impacts) [13,19,102]. This is because the conceptualization and parameterization of feedback loops with decision rules is a central component of SDMs [19,103]. The feedback loops determine the behavior (response) of a system; hence, by altering the parameters, SDMs are advantageous for scenario analysis [19]. Moreover, because the data demand of these methods is very flexible (ranging from qualitative to quantitative), the inclusion of the multiple variables embedded in telecoupled systems find less constraints than purely quantitative methods [19,104]. ABMs and SDMs also allow the simultaneous parameterization of several processes affecting human interactions such as biophysical, socioeconomic and demographic processes [93]. This can be done through the integration of footprint measures or environmental indicators [102]. This feature is appealing for methods to assess the impacts occurring in telecoupled systems, but the assumptions set in models must be improved if the integration of multi-level and multi-disciplinary variables is to be done consistently. There is no analytical framework for forecasting in SDMs and the strong fine-scale focus of ABMs limits their forecasting capacity, however, their integration with other methods such as general equilibrium models and land use models could help to overcome this limitation. While many ABMs provide a spatially explicit representation of impacts [20,92,105], SDMs usually lack this type of representation. Spatially-explicit ABMs are important to capture the spatial heterogeneity of the behavior and factors parameterized in the model [20], and are key for capturing iLUC caused by the agents' behavior. Examples of this are available in [98,106,107] and have been reviewed [20,92,105]. Moreover, Millington et al. [19] have proposed a conceptual framework for the integration of ABMs, SDMs and CGE models to simulate the dynamics between international food trade and land use change originating under different social, political, economic and environmental scenarios. This hybrid proposal could certainly improve the analysis of multi-temporal and multi-level dynamics and feedbacks. However, the spatially-explicit allocation of impacts (i.e., iLUC) and the operationalization of land heterogeneity remains a challenge that could potentially be overcome with land use models.

3.5. Land Use Models

3.5.1. General Description

The modelling of land use changes can be done with integrated assessment models (IAMs) that often include simplified land use modules, or with more specialized land use models (LUMs) [18].

Land use change modelling can be based on economic outputs (usually on CGE and PE) and hybrid data (using economic data, biophyisical data and spatial explicit allocation) [108], and the modeling approach can be process-based (i.e., representing agent behavior) or pattern-based (i.e., based on algorithms to describe changes) [109]. Different models simulate land use changes using similar rationale but have different allocation procedures. LUMs have forecasting capacity and are based on scenarios. They are also based on different data sources, have different spatial and temporal resolution and are based on different assumptions. For instance, CLUMondo, GLOBIOM, IMAGE and MagPIE use the outputs of CGE and PE models for crop demand data [88,108,110–113] (see Table S2 in the Supplementary Materials for more details). All models have some kind of calculation of location suitability to determine where land use changes are happening. For instance, in CLUMondo, this is largely based on empirical analysis of current land use patterns in relation to location factors, and in LandShift the location suitability is determined by models of plant growth and hydrology.

3.5.2. General Limitations

Each LUM is designed to answer specififc questions at a specific scale. Therefore, the system boundary defined in the models is either very narrow to study fine-scale changes, or very broad to study global or regional changes. Models integrating these cross-scales face several technical and practical challenges. Due to the complexity of IAMs, they are subject to very high uncertainties [108] and are difficult to validate due to the lack of historical observational data [114]. These uncertainties come from the underlying assumptions, input data, scenario assumptions, scale mismatches and differences in initial land cover classes [114]. Models include multiple variables but there are some that are still very difficult to incorporate. For instance, few models incorporate land use management categories as drivers of land use change [115]. Fine-scale models integrate actors' behavior but large-scale ones often do not [109]. In many cases, the complexity of the underlying processes leads to different simplifications and the exclusion of certain social, economic and environmental variables.

3.5.3. Suitability for Telecoupled Systems

Land systems reflect the result of the interaction of social, economic and environmental dynamics, and as such, are important for telecoupling analysis. Land use models are of special interest to telecoupled systems because they can account for land-related spillovers (iLUC), can be used for scenario analysis, to evaluate policy decisions, as learning-tools to test different drivers of change, and for spatially-explicit impact assessment [10]. Land use models with global coverage are relevant because they can analyse multiple and large scale processes [8]. However, the poor granularity of large-scale models leads to limited applicability at the scale needed by decision makers [116]. Models with broad boundaries usually lack granularity and representation of variations in local responses (such as adaptation) and they face limitations in integrating feedback mechanisms, for instance, with economic global dynamics. Hybrid land use models that can represent human decisions, socioeconomic and environmental factors simultaneously are available and reviewed in Brown et al. [109]. Although most LUM are very capable of calculating and allocating iLUC, the full parameterization of cross-scale processes and feedbacks is limited due to lack of understanding of the embedded processes, computation capacity and the empirial data that is available [10]. However, despite this advantageous capacity, LUMs are not product-centred and as such have limited capacities to analyse specific supply chains of products. The integration of different modelling approaches to fill some of these gaps faces challenges regarding the integration of multi-scale processes [109]. IAMs are designed to incorporate feedbacks within the studied systems, however, the simulation of feedbacks between causal mechanisms and impacts beyond them is still limited [18]. Regarding other impact measures (such as biodiversity loss, carbon release, or other related to ecosystem services), these can be calculated independently using the simulated direct and indirect land use changes as a basis [12]. Moreover, in some models the data for demand is based on aggregated groups of products, so analysis ofspecific products is not possible. However, in novel applications of models, the demands for subsistence commodities are distinguished from marketed commodities which allowsprocesses that are driven locally and those that are caused by telecoupled systems to be distinguished [117,118].

4. Discussion

In the previous section we identified the challenges related to each method type. In this section we discuss the overarching challenges that need to be tackled to help design meaningful and applied assessment methods to account for telecoupled impacts of agricultural supply chains.

4.1. Systems Boundaries

The definition of the boundaries of the studied systems has an important effect on the capacity of methods to account for the impacts caused by telecoupling dynamics [30,119]. Despite its relevance, the definition of the system boundaries is often arbitrary and non-science based. One of the most common and systematically applied over-simplifications is the definition of narrow truncation points that either exclude large-scale global interactions or exclude highly important fine-scale responses. Setting the correct system boundary depends on the goals of the study and the scale of analysis. Top-down approaches (such as CGE and PE) have the advantage of capturing large-scale processes but lack the capacity to account for place-specific impacts. Therefore, for this type of method, system boundary expansion means allowing the inclusion of place-specific factors to adjust the usual global averages they rely on. Most top-down approaches still have limitations to including telecoupling dynamics, so they also need to undergo improvements as described in Section 3 for each method. Bottom-up approaches (such as LCA and ABM) are well suited to capture place-or product-specific dynamics but have limitations in accounting for large-scale dynamics influencing the impacts of a supply chain. For this type of method, system boundary expansion means capturing large-scale dynamics and could be achieved by coupling them with other methods that have this capacity. Hence, it is necessary to integrate top-down and bottom-up modelling approaches [33,120]. The several hybrid initiatives previously mentioned are a good starting point.

4.2. Hybrid Models to Assess Telecoupled Impacts

The multi-disciplinary nature of telecoupled systems demands the integration of different types of methods rather than aiming at one single method able to address a broad variety of issues with insufficient detail [18]. Hybrid approaches could bridge the gaps between different methods and could be capable of assessing the impacts of multiple telecoupling dynamics in one single analysis. An overview of hybrid methods available, detailing their contribution to the assessment of telecoupled impacts and some examples is provided in Table 3.

Footprints and impact indicators used within LCA are useful tools to demonstrate, with a high level of specificity, the environmental impacts that are considered important by the practitioner. As such, LCA and footprints can be used to add the environmental dimension that other methods lack. Concrete examples of footprints coupled to IO analysis [21,121–123] and SDMs [124] exist. LCA is usually coupled with IO to expand the system boundaries in the inventory phase, and with equilibrium models to address economic global feedbacks influencing supply chains and final impacts. Examples of LCA coupled with equilibrium models [34,50,60–62,125], IO [126–129] and other methods [13] are available. Regarding land use change, practitioners have coupled LCA with LUMs to add spatial dimensions and assess direct and indirect land use impacts [12,50,60,130]. However, the correct application of those place-specific LCAs demands the calculation of spatially-explicit conversion factors for each impact category, and such efforts are already being developed for some environmental impacts [51,56,131]. LCA is an adequate method to evaluate the telecoupled environmental impacts of supply chains from the producer perspective. However, the high flexibility allowed in the selection of impact indicators in LCA implies that practitioners need a high level of system understanding to provide an adequate represention of reality while avoiding a selection of measures aimed at achieving convenient results.

Deterministic equilibrium models are an appropriate tool to incorporate economic feedback loops and account for economic spillovers at a large scale. Several equilibrium models that incorporate the environmental dimension are already able to address direct and indirect land use changes (i.e., GLOBIOM and MagPIE) [108]. Efforts to add the environmental dimension to IO include the environmentally extended-IO [83,85] and coupling exercises of IO with environmental footprints and indicators [83,85,86,121,122].

On the other hand, human behaviour and agency are important modulating factors of telecoupled impacts; however, LCA, footprints, CGE, PE and IO lack or simplify their representation. Here, ABMs have the capacity to explicitly address decision making and its variations amongst actors, and the interactions between actors and biophysical systems. ABM and SDM could benefit from LUMs to more explicitly address the spatial variations in the environment and agencies, and could even provide the possibility of calculating indirect land use changes. Examples of coupling ABMs with environmental and spatially-explicit methods are available [20,92,105]. The very fine scale of analysis of ABM and SDM complicate their application to large scales where telecoupling dynamics are abundant, but there are some methodological suggestions to couple them with equilibrium models to incorporate global economic dynamics [19].

From a producer perspective, value chain analysis (VCA) is an important tool to evaluate the performance and sustainability of supply chains. VCA is a method for identifying hotspots where resource use, efficiency, coordination and profitability might be problematic along the supply chain of a product [132,133]. Industry has a long tradition of using VCA to improve the strategic and operational steps of their supply chains [134]. Increased awareness of the environmental dimension has triggered the use of VCA as a tool to improve the environmental sustainability of supply chains. This has usually been done by coupling VCA with other methods such as LCA, material flow analysis and footprints [5,134,135]. Because the factors affecting the quality and efficiency of supply chains can heavily influence the extent and intensity of the environmental impacts, VCA can play an important role in the identification of those driving factors/agents and contribute to understanding the chain of actions that give rise to telecoupled impacts in agricultural supply chains. This is possible because VCA goes beyond the product-level and adopts a multi-dimensional approach by integrating vertical and horizontal elements of supply chains [132,133,136,137]. Special attention would be needed to link VCA and LUMs to analyze indirect land use changes in a spatially explicit manner.

Finally, although there are several coupling exercises available, scientists have warned that some hybrid models have an unmanageable complexity or are dysfunctional because they disregard the conceptual, technical and semantic differences between the methods they couple [19,138,139]. Technical differences might include geometry and spatial resolution, data scales, non-standardized ontologies and conceptual mismatches that could lead to the loss of important individual properties of models when coupled with others [139]. Therefore, coupled exercises are encouraged but must be done with caution. Although hybrid proposals are presented in this paper, it is important to mention that improving the methods themselves, has to go hand-in-hand with achieving more hybrid approaches to avoid overwhelmingly complex methods where the individual tools still have difficulty in addressing basic questions.

4.3. Long-Term Impacts

The inclusion of long-term impacts in methods is important because agricultural activities can cause soil depletion and toxicity from agrochemicals over time, and can also be affected by long-term phenomena such as climate change. The economic and social impacts of agricultural activities can also be extended over time and the socio-ecological dynamics occur simultaneously at multiple temporal scales [22]. Additionally, socio-ecological systems are prone to experience abrupt structural changes (regime shifts) that can be transmitted in the system structure as a cascade effect [30]. Therefore, disregarding the long-term impacts in the analysis only limits the reliability of assessments. Because the dynamics of telecoupled systems occur at diverse temporal and spatial scales, methods should be able to reconcile these scales [18]. Some studies have tried to incorporate them, but there is little consensus about how to integrate short- and long-term dynamics in the same study [18,140]. The inclusion of long-term dynamics would improve the forecasting capacity of methods and would highly favor decision making processes by providing information about foreseen impacts that would enable contingency measures. This goal could be achieved by expanding the system boundaries of methods and by increasing the understanding of cause-effect mechanisms in a multi-spatial and multi-temporal manner. Initiatives related to land systems are going towards this direction [22].

Table 3. Examples of hybrid methods to analyze the impacts of telecoupled agricultural supply chains.

Hybrid Method	Description	Main Contribution to the Assessment of Telecoupled Impacts	Examples
LCA and LUM	Uses LUM to predict, calculate and allocate the impacts of land use change in LCA	Spatially-explicit forecasting of land-related spillovers (iLUC change)	Chaplin-Kramer et al. [12] Geyer et al. [59] De Rosa et al. [133]
LCA and CGE/PE-based LUM	Couples LCA with CGE/PE-based LUMs (i.e., GLOBIOM) to quantify and spatially allocate direct and indirect LUC and calculate other environmental impacts caused by international trade.	System boundary expansion (to the global economy), integration of economic feedbacks, analysis of land-related spillovers (iLUC).	Di Fulvio et al. [60] Searchinger et al. [61] Leip et al. [62] Kloverpris et al. [50]
IO and footprints or indicators	Uses simple or multi-regional IO tables coupled with environmental data, footprints and indicators to calculate the environmental impacts caused by trade.	System boundary expansion and integration of economic feedbacks (only for the case of MRIO).	Kitzes [85] Tukker et al. [83] Prell et al. [133] Ewing et al. [121] Hertwich et al. [122] Weinzettel et al. [123] Turner et al. [21]
IO and LCA	Uses input-output tables to track resources used in the life cycle of a product to calculate the environmental impacts caused in response to market changes.	System boundary expansion.	Hawkins et al. [126] Igos et al. [127] Kennelly et al. [128] Yi et al. [129]
SDM and footprints or indicators	Representing wider system dynamics and link it to environmental indicators to represent the relationship between environmental impacts and socio-economic drivers.	System boundary expansion, integration of feedback loops and spillovers.	Mavrommati et al. [102]
ABM, SDM and CGE	Uses ABM to represent land use decision-making, CGE to represent markets and SDM to represent flows.	System boundary expansion (to the global economy), integration of feedback loops and spillovers caused by agents.	Millington et al. [19]

Abbreviations stand for: LCA: life cycle assessment; CGE: computable general equilibrium models; PE: partial equilibrium models; IO: input-output analysis; ABM: agent-based models; SDM: system dynamics models; LUM: land use models.

4.4. Geographic Heterogeneity

Land heterogeneity and land management practices implemented in a production system get little attention in most methods. Pongratz and colleagues [115] describe the importance of representing land management practices in models to significantly increase their comprehensiveness [95]. Critical aspects of land heterogeneity and land management practices need to be first identified, understood, prioritized and parametrized to be included in methods. Methods would need to implement place-specific information to increase their accuracy [3]. Moreover, the use of baseline information (i.e., land cover maps, biome maps, etc.) should ideally be homogenized to allow comparability between studies using the same scale. One of the most extended practices facilitating the exclusion of landscape-specific considerations in methods is the use of generalized assumptions and highly aggregated data [1,65].

To overcome these limitations, increased understanding of the importance of landscape-specific phenomena and the construction of disaggregated databases is needed. Empirical studies would play an important role to fill this gap and would contribute to the improvement of the available methods [18,20,141].

4.5. Suitability for Different User Types and Hands-On Approach

There is a wide range of stakeholders using impact assessment studies to help improve the environmental performance of a given product, territory, service or supply chain. Since the choice of a given impact assessment method carries different implications [142], this selection must be carried out carefully. Regardless of the technical criteria described in this paper, the choice of an adequate method is strongly influenced by the ultimate practical goals of the analysis, which in turn are closely related to the target audience. Different stakeholders rule over different subjects (products, supply chains, territories, consumers, laws, etc.), and therefore, may need distinct approaches. The methods analyzed in this article have either a consumption, production, geographical or a system approach. The relative importance of these approaches is closely linked with the concept of responsibility allocation, and for instance, has been widely discussed for the case of carbon emissions. For consumption-based approaches (i.e., EF), the responsibility of a given agent relies solely on the products consumed regardless of all the impacts caused through own production activities [143,144]. Therefore, this approach assesses the impacts embedded in products and attributes them to the agents consuming them. While this approach accounts for the impacts caused by demand, it is limited for promoting management strategies because consumers might have no interference power above the producers of the services or products demanded [1]. For production-based approaches (i.e., LCA), the responsibility relies on the impacts caused by the production processes of goods and services regardless of the final consumer or the origin of the inputs used [143]. This approach accounts for the impacts of supply but it can be problematic when it comes to using it for effective management, as it can negatively incentivize producers to outsource the most harmful activities or inputs to avoid responsibility [65]. The geographical approach (i.e., LUM) refers to methods which focus on a spatially defined area where diverse human and natural forces interact and cause changes (impacts) across that territory. Therefore, its main goal is to spatially allocate impacts caused by a set of activities. One limitation of this approach is that it does not provide explicit decision-support information, neither to producers nor to consumers because it describes changes over territorial areas where multiple producers and consumers are responsible for impacts but are not explicitly identified in the models. The systems approach (i.e., ABM and SDM) includes methods whose ultimate goal is to understand the dynamics and processes embedded in telecoupling systems that lead to impacts without necessarily emphasizing the quantification of impacts or allocating responsibility [19]. An advantage is that these methods are flexible enough to emphasize both the consumption and production sites.

Additionally, it is important to note the trade-offs between the applicability and comprehensiveness of methods. Single impact scores (i.e., from EF) have a communicative advantage for decision-making because they ease comparisons. At the same time, a problem is that they can be based on oversimplified analysis. At the same time, nowadays, the use of assessment studies that have a large spatial coverage (i.e., LUM and CGE) by decision makers is limited because they do not provide information at the scale needed for practical actions [18]. Therefore, the specific application goals and the local context should be considered in the choice of methods. Inter-institutional science-policy collaborations should be encouraged to achieve meaningful and hands-on assessments.

4.6. Reference Points for Sustainability

"A given indicator does not say anything about sustainability, unless a reference value or threshold is given to it" [145]. LCA for instance, is mainly designed for comparison between products but it does not provide information about the sustainability of the products themselves. EF does not provide a reference point for sustainability either and is used also for comparison. These methods are strongly criticized for oversimplifying the concept of sustainability and authors have discouraged their use for that purpose [72]. The damages to land systems can be calculated from LUMs, but no reference point to sustainability is provided. Similar to the previous models, interpretations about sustainability are left to personal judgements. ABMs and SDMs are more focused on understanding the functioning of systems. CGE and PE, when coupled with other methods, can calculate the environmental damage but again without references to sustainability. To solve these limitations, Heck and colleagues [146] proposed differentiated maximum land-use capacities to ensure sustainability based on the planetary boundaries. Bjorn and Hauschild [147] proposed the use of carrying capacities as reference points for environmental sustainability. Zhang and colleagues [66] proposed the re-definition and re-calculation of differentiated biocapacities for the calculation of EF. Hoekstra and Wiedmann [11] proposed the definition of maximum environmental-specific footprints. Nonetheless, although this limitation could be improved at regional or local scales, the application of reference points of sustainability at the corporate level still represents a complex challenge because it demands the allocation of fair shares to limit resource use [148]. The cited initiatives are important steps towards increasing the application of impact assessment studies but empirical studies to analyze the adequacy of them must be encouraged. Finally, given the holistic nature of sustainability, more studies about the trade-offs between sustainability dimensions are needed.

5. Conclusions

The implementation of sustainable trade can be supported by the use of methods capable of allocating the negative direct and indirect social, environmental and economic impacts occurring along the supply chain of products [15]. Although there is a wide range of tools available to assess these different impacts of telecoupled systems, there is no method that is able to fully assess these impacts in an integrated manner while considering the telecoupling dynamics in a spatially explicit manner. This is not necessarily a single, desired goal, but rather the confluence of independent achievements to improve methods and their smart hybridization. The well-known technical challenges to obtaining hybrid models described in this paper have to be surpassed to succeed in this path. Due to the nature of agricultural supply chains and the actions needed to pursue their sustainability, spatially-explicit methods that are able to account for direct and indirect land use changes are a pivotal part of such a challenge. The improvement in the methods themselves demands the expansion of system boundaries to capture bottom-up and top-down dynamics, improving the geographic resolution and time-coverage of databases, integrating landscape heterogeneity, creating location-specific transformation factors, improving data transparency, and improving the assumptions embedded in methods. Especially important is the task of improving the understanding of cause-effect mechanisms that modulate the impacts of supply chains. Multi-disciplinary collaborations are encouraged in order to succeed. Additionally, it is important to acknowledge the trade-offs between the straight forward interpretation of some methods versus the comprehensiveness of others when making a selection of methods. Finally, the definition of sustainability reference points is an urgent task in order to go beyond product benchmarking towards methods that provide straight forward advice about the sustainability of supply chains and responsibility allocation.

Supplementary Materials: The following are available online at http://www.mdpi.com/2071-1050/11/4/1162/s1, Table S1: Additional information of input-output databases mentioned in the main text, Table S2: Additional information of land use models mentioned in the main text.

Author Contributions: Conceptualization, writing-original draft preparation, C.P.P.; Writing-review & editing, P.H.V.

Funding: This work was funded by the Marie Skłodowska-Curie actions (MSCA) grant agreement No 765408 from the European Commission: COUPLED 'Operationalising Telecouplings for Solving Sustainability Challenges for Land Use'. PV would like to acknowledge support from European Research Council grant GLOLAND (Grant No. 311819).

Acknowledgments: This work was supported by the Marie Skłodowska-Curie actions (MSCA) grant agreement No 765408 from the European Commission: COUPLED 'Operationalising Telecouplings for Solving Sustainability Challenges for Land Use'.

Conflicts of Interest: The authors declare no conflicts of interest.

References

1. Schaffartzik, A.; Haberl, H.; Kastner, T.; Wiedenhofer, D.; Eisenmenger, N.; Erb, K.H. Trading land: A review of approaches to accounting for upstream land requirements of traded products. *J. Ind. Ecol.* **2015**, *19*, 703–714. [CrossRef] [PubMed]
2. Ramos, S.; Larrinaga, L.; Albinarrate, U.; Jungbluth, N.; Ingolfsdottir, G.M.; Yngvadottir, E.; Landquist, B.; Woodhouse, A.; Olafsdottir, G.; Esturo, A.; et al. SENSE tool: Easy-to-use web-based tool to calculate food product environmental impact. *Int. J. Life Cycle Assess.* **2016**, *21*, 710–721. [CrossRef]
3. Henders, S.; Ostwald, M. Accounting methods for international land-related leakage and distant deforestation drivers. *Ecol. Econ.* **2014**, *99*, 21–28. [CrossRef]
4. Webber, M.; Labaste, P. *Using Value Chain Approaches in Agribusiness and Agriculture in Sub-Saharan Africa: A Methodological Guide. Tools that Make Value Chains Work: Discussion and Cases*; The World Bank: Washington, DC, USA, 2007.
5. Fasse, A.C.; Grote, U.; Winter, E. *Value Chain Analysis: Methodologies in Context of Environment and Trade Research*; Hannover Economic Papers (HEP); Leibniz Universität Hannover, Wirtschaftswissenschaftliche Fakultät: Hannover, Germany, 2009.
6. Borsato, E.; Tarolli, P.; Marinello, F. Sustainable patterns of main agricultural products combining different footprint parameters. *J. Clean. Prod.* **2018**, *179*, 357–367. [CrossRef]
7. Rebitzer, G.; Ekvall, T.; Frischknecht, R.; Hunkeler, D.; Norris, G.; Rydberg, T.; Schmidt, W.P.; Suh, S.; Weidema, B.P.; Pennington, D.W. Life cycle assessment Part 1: Framework, goal and scope definition, inventory analysis, and applications. *Environ. Int.* **2004**, *30*, 701–720. [CrossRef] [PubMed]
8. Lambin, E.F.; Turner, B.L.; Geist, H.J.; Agbola, S.B.; Angelsen, A.; Folke, C.; Bruce, J.W.; Coomes, O.T.; Dirzo, R.; George, P.S.; et al. The causes of land-use and land-cover change: Moving beyond the myths. *Glob. Environ. Chang.* **2001**, *11*, 261–269. [CrossRef]
9. Liu, J.; Hull, V.; Batistella, M.; DeFries, R.; Dietz, T.; Fu, F.; Hertel, T.W.; Izaurralde, R.C.; Lambin, E.F.; Li, S.; et al. Framing Sustainability in a Telecoupled World. *Ecol. Soc.* **2013**, *18*, 26. [CrossRef]
10. Verburg, P.H.; Crossman, N.; Ellis, E.C.; Heinimann, A.; Hostert, P.; Mertz, O.; Nagendra, H.; Sikor, T.; Erb, K.H.; Golubiewski, N.; et al. Land system science and sustainable development of the earth system: A global land project perspective. *Anthropocene* **2015**, *12*, 29–41. [CrossRef]
11. Hoekstra, A.Y.; Wiedmann, T.O. Humanity's unsustainable environmental footprint. *Science* **2014**, *344*, 1114–1117. [CrossRef]
12. Chaplin-Kramer, R.; Sim, S.; Hamel, P.; Bryant, B.; Noe, R.; Mueller, C.; Rigarlsford, G.; Kulak, M.; Kowal, V.; Sharp, R.; et al. Life cycle assessment needs predictive spatial modelling for biodiversity and ecosystem services. *Nat. Commun.* **2017**, *8*, 1–8. [CrossRef]
13. Onat, N.; Kucukvar, M.; Halog, A.; Cloutier, S. Systems Thinking for Life Cycle Sustainability Assessment: A Review of Recent Developments, Applications, and Future Perspectives. *Sustainability* **2017**, *9*, 706. [CrossRef]
14. Ness, B.; Urbel-Piirsalu, E.; Anderberg, S.; Olsson, L. Categorising tools for sustainability assessment. *Ecol. Econ.* **2007**, *60*, 498–508. [CrossRef]
15. Herva, M.; Franco, A.; Carrasco, E.F.; Roca, E. Review of corporate environmental indicators. *J. Clean. Prod.* **2011**, *19*, 1687–1699. [CrossRef]
16. Čuček, L.; Klemeš, J.J.; Kravanja, Z. A review of footprint analysis tools for monitoring impacts on sustainability. *J. Clean. Prod.* **2012**, *34*, 9–20. [CrossRef]
17. Bruckner, M.; Fischer, G.; Tramberend, S.; Giljum, S. Measuring telecouplings in the global land system: A review and comparative evaluation of land footprint accounting methods. *Ecol. Econ.* **2015**, *114*, 11–21. [CrossRef]
18. Verburg, P.H.; Dearing, J.A.; Dyke, J.G.; van der Leeuw, S.; Seitzinger, S.; Steffen, W.; Syvitski, J. Methods and approaches to modelling the Anthropocene. *Glob. Environ. Chang.* **2016**, *39*, 328–340. [CrossRef]
19. Millington, J.; Xiong, H.; Peterson, S.; Woods, J. Integrating Modelling Approaches for Understanding Telecoupling: Global Food Trade and Local Land Use. *Land* **2017**, *6*, 56. [CrossRef]

20. Filatova, T.; Verburg, P.H.; Parker, D.C.; Stannard, C.A. Spatial agent-based models for socio-ecological systems: Challenges and prospects. *Environ. Model. Softw.* **2013**, *45*, 1–7. [CrossRef]
21. Turner, K.; Lenzen, M.; Wiedmann, T.; Barrett, J. Examining the global environmental impact of regional consumption activities—Part 1: A technical note on combining input-output and ecological footprint analysis. *Ecol. Econ.* **2007**, *62*, 37–44. [CrossRef]
22. Meyfroidt, P.; Chowdhury, R.; De Bremond, A.; Ellis, E.C.; Erb, K.-H.; Filatova, T.; Garrett, R.D.; Grove, J.M.; Heinimann, A.; Kuemmerle, T.; et al. Middle-range theories of land system change. *Glob. Environ. Chang.* **2018**, *53*, 52–67. [CrossRef]
23. Lambin, E.F.; Gibbs, H.K.; Heilmayr, R.; Carlson, K.M.; Fleck, L.C.; Garrett, R.D.; Le Polain De Waroux, Y.; McDermott, C.L.; McLaughlin, D.; Newton, P.; et al. The role of supply-chain initiatives in reducing deforestation. *Nat. Clim. Chang.* **2018**, *8*, 109–116. [CrossRef]
24. Albareda, L.; Lozano, J.M.; Ysa, T. Public policies on corporate social responsibility: The role of governments in Europe. *J. Bus. Ethics* **2007**, *74*, 391–407. [CrossRef]
25. Lenzen, M.; Murray, J.; Sack, F.; Wiedmann, T. Shared producer and consumer responsibility—Theory and practice. *Ecol. Econ.* **2007**, *61*, 27–42. [CrossRef]
26. Vermeir, I.; Verbeke, W. Sustainable food consumption: Exploring the consumer "attitude–Behavioral intention" gap. *J. Agric. Environ. Ethics* **2006**, *19*, 169–194. [CrossRef]
27. Cummins, S.; Reilly, T.M.; Carlson, L.; Grove, S.J.; Dorsch, M.J. Investigating the Portrayal and Influence of Sustainability Claims in an Environmental Advertising Context. *J. Macromark.* **2014**, *34*, 332–348. [CrossRef]
28. Rasmussen, L.V.; Coolsaet, B.; Martin, A.; Mertz, O.; Pascual, U.; Corbera, E.; Dawson, N.; Fisher, J.A.; Franks, P.; Ryan, C.M. Social-ecological outcomes of agricultural intensification. *Nat. Sustain.* **2018**, 275–282. [CrossRef]
29. Lambin, E.F.; Meyfroidt, P. Land use transitions: Socio-ecological feedback versus socio-economic change. *Land Use Policy* **2010**, *27*, 108–118. [CrossRef]
30. Filatova, T.; Polhill, J.G.; van Ewijk, S. Regime shifts in coupled socio-environmental systems: Review of modelling challenges and approaches. *Environ. Model. Softw.* **2016**, *75*, 333–347. [CrossRef]
31. Sala, S.; Ciuffo, B.; Nijkamp, P. A systemic framework for sustainability assessment. *Ecol. Econ.* **2015**, *119*, 314–325. [CrossRef]
32. Guinée, J.; Heijungs, R.; Huppes, G.; Zamagni, A.; Masoni, P.; Buonamici, R.; Ekvall, T.; Rydberg, T. Life Cycle Assessment: Past, Present, and Future. *Environ. Sci. Technol.* **2011**, *45*, 90–96. [CrossRef] [PubMed]
33. Hellweg, S.; Canals, L.M.I. Emerging approaches, challenges and opportunities in life cycle assessment. *Science* **2014**, *344*, 1109–1113. [CrossRef] [PubMed]
34. Earles, J.M.; Halog, A. Consequential life cycle assessment: A review. *Int. J. Life Cycle Assess.* **2011**, *16*, 445–453. [CrossRef]
35. European Commission, Joint Research Centre, Institute for Environment and Sustainability. *International Reference Life Cycle Data System (ILCD) Handbook—General Guide for Life Cycle Assessment—Detailed Guidance*; European Commission: Ispra, Italy, 2010; ISBN 978-92-79-19092-6.
36. Yang, Y.; Heijungs, R. On the use of different models for consequential life cycle assessment. *Int. J. Life Cycle Assess.* **2018**, *23*, 751–758. [CrossRef]
37. Reap, J.; Roman, F.; Duncan, S.; Bras, B. A survey of unresolved problems in life cycle assessment. Part 2: Impact assessment and interpretation. *Int. J. Life Cycle Assess.* **2008**, *13*, 374–388. [CrossRef]
38. UNEP/SETAC Life Cycle Initiative. *Global Guidance for Life Cycle Impact Assessment Indicators: Volume 1*; Frischknecht, R., Jolliet, O., Eds.; United Nations Environment Programme: Nairobi, Kenya, 2016; ISBN 978-92-807-3630-4. Available online: https://www.lifecycleinitiative.org/training-resources/global-guidance-lcia-indicators-v-1/ (accessed on 5 December 2018).
39. Hauschild, M.Z.; Goedkoop, M.; Guinée, J.; Heijungs, R.; Huijbregts, M.; Jolliet, O.; Margni, M.; De Schryver, A.; Humbert, S.; Laurent, A.; et al. Identifying best existing practice for characterization modeling in life cycle impact assessment. *Int. J. Life Cycle Assess.* **2013**, *18*, 683–697. [CrossRef]
40. Steinmann, Z.J.N.; Schipper, A.M.; Hauck, M.; Huijbregts, M.A.J. How Many Environmental Impact Indicators Are Needed in the Evaluation of Product Life Cycles? *Environ. Sci. Technol.* **2016**, *50*, 3913–3919. [CrossRef] [PubMed]

41. Jolliet, O.; Antón, A.; Boulay, A.-M.; Cherubini, F.; Fantke, P.; Levasseur, A.; McKone, T.E.; Michelsen, O.; Milà i Canals, L.; Motoshita, M.; et al. Global guidance on environmental life cycle impact assessment indicators: Impacts of climate change, fine particulate matter formation, water consumption and land use. *Int. J. Life Cycle Assess.* **2018**. [CrossRef]
42. Curran, M.; De Baan, L.; De Schryver, A.M.; Van Zelm, R.; Hellweg, S.; Koellner, T.; Sonnemann, G.; Huijbregts, M.A.J. Toward meaningful endpoints of biodiversity in Life Cycle Assessment. *Environ. Sci. Technol.* **2011**, *45*, 70–79. [CrossRef]
43. De Haes, U.; Heijungs, R.; Suh, S.; Huppes, G. Three strategies to overcome the limitations of LCA. *J. Ind. Ecol.* **2004**, *8*, 19–32. [CrossRef]
44. Castellani, V.; Benini, L.; Sala, S.; Pant, R. A distance-to-target weighting method for Europe 2020. *Int. J. Life Cycle Assess.* **2016**, *21*, 1159–1169. [CrossRef]
45. Guinée, J.B. Life Cycle Sustainability Assessment: What Is It and What Are Its Challenges? In *Taking Stock of Industrial Ecology*; Clift, R., Druckman, A., Eds.; Springer: Basel, Switzerland, 2016; pp. 1–362, ISBN 9783319205717.
46. Dreyer, L.; Hauschild, M.; Schierbeck, J. A Framework for Social Life Cycle Impact Assessment. *Int. J. Life Cycle Assess.* **2006**, *11*, 88–97. [CrossRef]
47. Hutchins, M.J.; Sutherland, J.W. An exploration of measures of social sustainability and their application to supply chain decisions. *J. Clean. Prod.* **2008**, *16*, 1688–1698. [CrossRef]
48. UNEP Setac Life Cycle Initiative. Guidelines for Social Life Cycle Assessment of Products. 2009. Available online: http://www.unep.fr/shared/publications/pdf/dtix1164xpa-guidelines_slca.pdf (accessed on 5 December 2018).
49. Curran, M.A. Life Cycle Assessment: A review of the methodology and its application to sustainability. *Curr. Opin. Chem. Eng.* **2013**, *2*, 273–277. [CrossRef]
50. Kloverpris, J.; Wenzel, H.; Nielsen, P.H. Life cycle inventory modelling of land use induced by crop consumption—Part 1: Conceptual analysis and methodological proposal. *Int. J. Life Cycle Assess.* **2008**, *13*, 13–21. [CrossRef]
51. Van Zelm, R.; van der Velde, M.; Balkovic, J.; Čengić, M.; Elshout, P.M.F.; Koellner, T.; Núñez, M.; Obersteiner, M.; Schmid, E.; Huijbregts, M.A.J. Spatially explicit life cycle impact assessment for soil erosion from global crop production. *Ecosyst. Serv.* **2018**, *30*, 220–227. [CrossRef]
52. Koellner, T.; Scholz, R.W. Assessment of land use impacts on the natural environment. Part 1: An analytical framework for pure land occupation and land use change. *Int. J. Life Cycle Assess.* **2007**, *12*, 16–23. [CrossRef]
53. Loiseau, E.; Aissani, L.; Le Féon, S.; Laurent, F.; Cerceau, J.; Sala, S.; Roux, P. Territorial Life Cycle Assessment (LCA): What exactly is it about? A proposal towards using a common terminology and a research agenda. *J. Clean. Prod.* **2018**, *176*, 474–485. [CrossRef]
54. Mattila, T.; Helin, T.; Antikainen, R. Land use indicators in life cycle assessment A case study on beer production. *Int. J. Life Cycle Assess.* **2012**, *17*, 277–286. [CrossRef]
55. Arodudu, O.; Helming, K.; Wiggering, H.; Voinov, A. Towards a more holistic sustainability assessment framework for agro-bioenergy systems—A review. *Environ. Impact Assess. Rev.* **2017**, *62*, 61–75. [CrossRef]
56. Milà i Canals, L.; Bauer, C.; Depestele, J.; Dubreuil, A.; Knuchel, R.F.; Gaillard, G.; Michelsen, O.; Müller-Wenk, R.; Rydgren, B. Key elements in a framework for land use impact assessment within LCA. *Int. J. Life Cycle Assess.* **2007**, *12*, 5–15. [CrossRef]
57. Udo de Haes, H.A. How to approach land use in LCIA or, how to avoid the Cinderella effect? Comments on 'key elements in a framework for land use impact assessment within LCA1. *Int. J. Life Cycle Assess.* **2006**, *11*, 219–221. [CrossRef]
58. Saad, R.; Koellner, T.; Margni, M. Land use impacts on freshwater regulation, erosion regulation, and water purification: A spatial approach for a global scale level. *Int. J. Life Cycle Assess.* **2013**, *18*, 1253–1264. [CrossRef]
59. Geyer, R.; Lindner, J.P.; Stoms, D.M.; Davis, F.W.; Wittstock, B. Coupling GIS and LCA for biodiversity assessments of land use. *Int. J. Life Cycle Assess.* **2010**, *15*, 692–703. [CrossRef]
60. Di Fulvio, F.; Forsell, N.; Korosuo, A.; Obersteiner, M.; Hellweg, S. Spatially explicit LCA analysis of biodiversity losses due to different bioenergy policies in the European Union. *Sci. Total Environ.* **2019**, *651*, 1505–1516. [CrossRef] [PubMed]

61. Searchinger, T.; Heimlich, R.; Houghton, R.A.; Dong, F.; Elobeid, A.; Fabiosa, J.; Tokgoz, S.; Hayes, D.; Yu, T. Emissions from land-use change. *Science* **2008**, *423*, 1238–1240. [CrossRef] [PubMed]
62. Leip, A.; Weiss, F.; Wassenaar, T.; Perez, I.; Fellmann, T.; Loudjani, P.; Tubiello, F.; Grandgirard, D.; Monni, S.; Biala, K. Evaluation of the Livestock Sector's Contribution to the EU Greenhouse Gas Emissions (GGELS). Final Report. 2010. Available online: https://ec.europa.eu/agriculture/sites/agriculture/files/external-studies/2010/livestock-gas/full_text_en.pdf (accessed on 7 February 2019).
63. Schmidt, J.H.; Weidema, B.P.; Brandão, M. A framework for modelling indirect land use changes in Life Cycle Assessment. *J. Clean. Prod.* **2015**, *99*, 230–238. [CrossRef]
64. Schmidt, J.H. Life cycle assessment of five vegetable oils. *J. Clean. Prod.* **2015**, *87*, 130–138. [CrossRef]
65. Galli, A.; Wiedmann, T.; Ercin, E.; Knoblauch, D.; Ewing, B.; Giljum, S. Integrating Ecological, Carbon and Water footprint into a "footprint Family" of indicators: Definition and role in tracking human pressure on the planet. *Ecol. Indic.* **2012**, *16*, 100–112. [CrossRef]
66. Zhang, L.; Dzakpasu, M.; Chen, R.; Wang, X.C. Validity and utility of ecological footprint accounting: A state-of-the-art review. *Sustain. Cities Soc.* **2017**, *32*, 411–416. [CrossRef]
67. Wackernagel, M.; Cranston, G.; Morales, J.C.; Galli, A. Ecological footprint accounts. In *Handbook of Sustainable Development*; Atkinson, G., Dietz, S., Neumayer, E., Agarwala, M., Eds.; Edwar Elgar Publishing Limited: Cheltenham, UK, 2014; pp. 371–396, ISBN 9781782544708.
68. Brooks, S.T.; Jabour, J.; Bergstrom, D.M. What is "footprint" in Antarctica: Proposing a set of definitions. *Antarct. Sci.* **2018**, *30*, 227–235. [CrossRef]
69. Krausmann, F.; Erb, K.-H.; Gingrich, S.; Haberl, H.; Bondeau, A.; Gaube, V.; Lauk, C.; Plutzar, C.; Searchinger, T.D. Global human appropriation of net primary production doubled in the 20th century. *Proc. Natl. Acad. Sci. USA* **2013**, *110*, 10324–10329. [CrossRef] [PubMed]
70. Haberl, H.; Erb, K.H.; Krausmann, F.; Berecz, S.; Ludwiczek, N.; Martínez-Alier, J.; Musel, A.; Schaffartzik, A. Using embodied HANPP to analyze teleconnections in the global land system: Conceptual considerations. *Geogr. Tidsskr.* **2009**, *109*, 119–130. [CrossRef]
71. Erb, K.H.; Krausmann, F.; Lucht, W.; Haberl, H. Embodied HANPP: Mapping the spatial disconnect between global biomass production and consumption. *Ecol. Econ.* **2009**, *69*, 328–334. [CrossRef]
72. Fiala, N. Measuring sustainability: Why the ecological footprint is bad economics and bad environmental science. *Ecol. Econ.* **2008**, *67*, 519–525. [CrossRef]
73. Kastner, T.; Kastner, M.; Nonhebel, S. Tracing distant environmental impacts of agricultural products from a consumer perspective. *Ecol. Econ.* **2011**, *70*, 1032–1040. [CrossRef]
74. Miller, R.E.; Blair, P.D. *Input–Output Analysis. Foundations and Extensions*, 2nd ed.; Cambridge University Press: Washington, DC, USA, 2009; ISBN 9780521517133.
75. Rose, A.; Rose, A.; Rose, A. Input-output economics and computable general equilibrium models. *Struct. Chang. Econ. Dyn.* **1995**, *6*, 295–304. [CrossRef]
76. Weisz, H.; Duchin, F. Physical and monetary input-output analysis: What makes the difference? *Ecol. Econ.* **2006**, *57*, 534–541. [CrossRef]
77. West, G.R. Comparison of Input–Output, Input–Output + Econometric and Computable General Equilibrium Impact Models at the Regional Level. *Econ. Syst. Res.* **1995**, *7*, 209–227. [CrossRef]
78. Lenzen, M.; Moran, D.; Kanemoto, K.; Geschke, A. Building Eora: A Global Multi-Region Input-Output Database At High Country and Sector Resolution. *Econ. Syst. Res.* **2013**, *25*, 20–49. [CrossRef]
79. Inomata, S.; Owen, A. Comparative Evaluation of Mrio Databases. *Econ. Syst. Res.* **2014**, *26*, 239–244. [CrossRef]
80. Moran, D.; Wood, R. Convergence Between the Eora, Wiod, Exiobase, and Openeu'S Consumption-Based Carbon Accounts. *Econ. Syst. Res.* **2014**, *26*, 245–261. [CrossRef]
81. Wood, R.; Stadler, K.; Bulavskaya, T.; Lutter, S.; Giljum, S.; de Koning, A.; Kuenen, J.; Schütz, H.; Acosta-Fernández, J.; Usubiaga, A.; et al. Global sustainability accounting-developing EXIOBASE for multi-regional footprint analysis. *Sustainability* **2015**, *7*, 138–163. [CrossRef]
82. Dietzenbacher, E.; Los, B.; Stehrer, R.; Timmer, M.; de Vries, G. The Construction of World Input-Output Tables in the WIOD Project. *Econ. Syst. Res.* **2013**, *25*, 71–98. [CrossRef]
83. Tukker, A.; Huppes, G.; Van Oers, L.; Heijungs, R. Environmentally Extended Input-Output Tables and Models for Europe. 2006. Available online: ftp://ftp.jrc.es/pub/EURdoc/eur22194en.pdf (accessed on 7 February 2019).

84. Wiedmann, T.; Lenzen, M.; Turner, K.; Barrett, J. Examining the global environmental impact of regional consumption activities—Part 2: Review of input-output models for the assessment of environmental impacts embodied in trade. *Ecol. Econ.* **2007**, *61*, 15–26. [CrossRef]
85. Kitzes, J. An Introduction to Environmentally-Extended Input-Output Analysis. *Resources* **2013**, *2*, 489–503. [CrossRef]
86. Prell, C.; Sun, L.; Feng, K.; He, J.; Hubacek, K. Uncovering the spatially distant feedback loops of global trade: A network and input-output approach. *Sci. Total Environ.* **2017**, *586*, 401–408. [CrossRef]
87. Yu, Y.; Feng, K.; Hubacek, K. Tele-connecting local consumption to global land use. *Glob. Environ. Chang.* **2013**, *23*, 1178–1186. [CrossRef]
88. Havlik, P.; Valin, H.; Herrero, M.; Obersteiner, M.; Schmid, E.; Rufino, M.C.; Mosnier, A.; Thornton, P.K.; Bottcher, H.; Conant, R.T.; et al. Climate change mitigation through livestock system transitions. *Proc. Natl. Acad. Sci. USA* **2014**, *111*, 3709–3714. [CrossRef]
89. Van Asselen, S.; Verburg, P.H. Land cover change or land-use intensification: Simulating land system change with a global-scale land change model. *Glob. Chang. Biol.* **2013**, *19*, 3648–3667. [CrossRef]
90. Irwin, E.G.; Geoghegan, J. Theory, data, methods: Developing spatially explicit economic models of land use change. *Agric. Ecosyst. Environ.* **2001**, *85*, 7–24. [CrossRef]
91. MacPherson, B.; Gras, R. Individual-based ecological models: Adjunctive tools or experimental systems? *Ecol. Model.* **2016**, *323*, 106–114. [CrossRef]
92. Groeneveld, J.; Müller, B.; Buchmann, C.M.; Dressler, G.; Guo, C.; Hase, N.; Hoffmann, F.; John, F.; Klassert, C.; Lauf, T.; et al. Theoretical foundations of human decision-making in agent-based land use models – A review. *Environ. Model. Softw.* **2017**, *87*, 39–48. [CrossRef]
93. An, L.; Zvoleff, A.; Liu, J.; Axinn, W. Agent-Based Modeling in Coupled Human and Natural Systems (CHANS): Lessons from a Comparative Analysis. *Ann. Assoc. Am. Geogr.* **2014**, *104*, 723–745. [CrossRef]
94. Hare, M.; Deadman, P. Further towards a taxonomy of agent-based simulation models in environmental management. *Math. Comput. Simul.* **2004**, *64*, 25–40. [CrossRef]
95. Ercan, T.; Onat, N.C.; Tatari, O. Investigating carbon footprint reduction potential of public transportation in United States: A system dynamics approach. *J. Clean. Prod.* **2016**, *133*, 1260–1276. [CrossRef]
96. Ramankutty, N.; Coomes, O.T. Land-use regime shifts: An analytical framework and agenda for future landuse research. *Ecol. Soc.* **2016**, *21*. [CrossRef]
97. Mai, T.; Smith, C. Scenario-based planning for tourism development using system dynamic modelling: A case study of Cat Ba Island, Vietnam. *Tour. Manag.* **2018**, *68*, 336–354. [CrossRef]
98. Valbuena, D.; Verburg, P.H.; Bregt, A.K.; Ligtenberg, A. An agent-based approach to model land-use change at a regional scale. *Landsc. Ecol.* **2010**, *25*, 185–199. [CrossRef]
99. Judson, O.P. The rise of the individual-based model in ecology. *Trends Ecol. Evol.* **1994**, *9*, 9–14. [CrossRef]
100. Tonini, F.; Liu, J. Telecoupling toolbox: Spatially explicit tools for studying telecoupled human and natural systems. *Ecol. Soc.* **2017**, *22*. [CrossRef]
101. An, L. Modeling human decisions in coupled human and natural systems: Review of agent-based models. *Ecol. Model.* **2012**, *229*, 25–36. [CrossRef]
102. Mavrommati, G.; Bithas, K.; Panayiotidis, P. Operationalizing sustainability in urban coastal systems: A system dynamics analysis. *Water Res.* **2013**, *47*, 7235–7250. [CrossRef] [PubMed]
103. Onat, N.C.; Egilmez, G.; Tatari, O. Towards greening the U.S. residential building stock.A system dynamics approach. *Build. Environ.* **2014**, *78*, 68–80. [CrossRef]
104. Coyle, R.G. Qualitative and Quantitative Modelling in System Dynamics: Some Research Questions. *Syst. Dyn. Rev.* **2000**, *16*, 225–244. [CrossRef]
105. Matthews, R.B.; Gilbert, N.G.; Roach, A.; Polhill, J.G.; Gotts, N.M. Agent-based land-use models: A review of applications. *Landsc. Ecol.* **2007**, *22*, 1447–1459. [CrossRef]
106. Schouten, M.A.H.; Polman, N.B.P.; Westerhof, E.J.G.M. Exploring Green Agricultural Policy Scenarios with a Spatially Explicit Agent-Based Model. Available online: http://edepot.wur.nl/260251 (accessed on 7 February 2019).
107. Happe, K.; Hutchings, N.J.; Dalgaard, T.; Kellerman, K. Modelling the interactions between regional farming structure, nitrogen losses and environmental regulation. *Agric. Syst.* **2011**, *104*, 281–291. [CrossRef]

108. Alexander, P.; Prestele, R.; Verburg, P.H.; Arneth, A.; Baranzelli, C.; Batista e Silva, F.; Brown, C.; Butler, A.; Calvin, K.; Dendoncker, N.; et al. Assessing uncertainties in land cover projections. *Glob. Chang. Biol.* **2017**, *23*, 767–781. [CrossRef] [PubMed]
109. Brown, D.G.; Verburg, P.H.; Pontius, R.G.; Lange, M.D. Opportunities to improve impact, integration, and evaluation of land change models. *Curr. Opin. Environ. Sustain.* **2013**, *5*, 452–457. [CrossRef]
110. Nelson, G.C.; Valin, H.; Sands, R.D.; Havlík, P.; Ahammad, H.; Deryng, D.; Elliott, J.; Fujimori, S.; Hasegawa, T.; Heyhoe, E.; et al. Climate change effects on agriculture: Economic responses to biophysical shocks. *Proc. Natl. Acad. Sci. USA* **2014**, *111*, 3274–3279. [CrossRef] [PubMed]
111. Stehfest, E.; van Vuuren, D.; Kram, T.; Bouwman, L.; Alkemade, R.; Bakkenes, M.; Biemans, H.; Bouwman, A.; den Elzen, M.; Janse, J.; et al. *Integrated Assessment of Global Environmental Change with IMAGE 3.0. Model Description and Policy Applications*; PBL Publishers: The Hague, The Netherlands, 2014; ISBN 978-94-91506-71-0.
112. Popp, A.; Humpenöder, F.; Weindl, I.; Bodirsky, B.L.; Bonsch, M.; Lotze-Campen, H.; Müller, C.; Biewald, A.; Rolinski, S.; Stevanovic, M.; et al. Land-use protection for climate change mitigation. *Nat. Clim. Chang.* **2014**, *4*, 1095–1098. [CrossRef]
113. Schmitz, C.; van Meijl, H.; Kyle, P.; Nelson, G.C.; Fujimori, S.; Gurgel, A.; Havlik, P.; Heyhoe, E.; d'Croz, D.M.; Popp, A.; et al. Land-use change trajectories up to 2050: Insights from a global agro-economic model comparison. *Agric. Econ.* **2014**, *45*, 69–84. [CrossRef]
114. Prestele, R.; Alexander, P.; Rounsevell, M.D.A.; Arneth, A.; Calvin, K.; Doelman, J.; Eitelberg, D.A.; Engström, K.; Fujimori, S.; Hasegawa, T.; et al. Hotspots of uncertainty in land-use and land-cover change projections: A global-scale model comparison. *Glob. Chang. Biol.* **2016**, *22*, 3967–3983. [CrossRef] [PubMed]
115. Pongratz, J.; Dolman, H.; Don, A.; Erb, K.-H.; Fuchs, R.; Herold, M.; Jones, C.; Kuemmerle, T.; Luyssaert, S.; Meyfroidt, P.; et al. Models meet data: Challenges and opportunities in implementing land management in Earth system models. *Glob. Chang. Biol.* **2017**, 1470–1487. [CrossRef] [PubMed]
116. Schaldach, R.; Alcamo, J.; Koch, J.; Kölking, C.; Lapola, D.M.; Schüngel, J.; Priess, J.A. An integrated approach to modelling land-use change on continental and global scales. *Environ. Model. Softw.* **2011**, *26*, 1041–1051. [CrossRef]
117. Debonne, N.; van Vliet, J.; Heinimann, A.; Verburg, P. Representing large-scale land acquisitions in land use change scenarios for the Lao PDR. *Reg. Environ. Chang.* **2018**, *18*, 1857–1869. [CrossRef]
118. Ornetsmüller, C.; Verburg, P.H.; Heinimann, A. Scenarios of land system change in the Lao PDR: Transitions in response to alternative demands on goods and services provided by the land. *Appl. Geogr.* **2016**, *75*, 1–11. [CrossRef]
119. Friis, C.; Nielsen, J.Ø. On the system. Boundary choices, implications, and solutions in telecoupling land use change research. *Sustainability* **2017**, *9*, 974. [CrossRef]
120. Sala, S.; Farioli, F.; Zamagni, A. Life cycle sustainability assessment in the context of sustainability science progress (part 2). *Int. J. Life Cycle Assess.* **2013**, *18*, 1686–1697. [CrossRef]
121. Ewing, B.R.; Hawkins, T.R.; Wiedmann, T.O.; Galli, A.; Ercin, A.E.; Weinzettel, J.; Steen-Olsen, K. Integrating ecological and water footprint accounting in a multi-regional input-output framework. *Ecol. Indic.* **2012**, *23*, 1–8. [CrossRef]
122. Hertwich, E.G.; Peters, G.P. Carbon Footprint of Nations: A Global, Trade-Linked Analysis. *Environ. Sci. Technol.* **2009**, *43*, 6414–6420. [CrossRef]
123. Weinzettel, J.; Hertwich, E.G.; Peters, G.P.; Steen-Olsen, K.; Galli, A. Affluence drives the global displacement of land use. *Glob. Environ. Chang.* **2013**, *23*, 433–438. [CrossRef]
124. Jin, W.; Xu, L.; Yang, Z. Modeling a policy making framework for urban sustainability: Incorporating system dynamics into the Ecological Footprint. *Ecol. Econ.* **2009**, *68*, 2938–2949. [CrossRef]
125. Lenzen, M.; Murray, S.A.; Korte, B.; Dey, C.J. Environmental impact assessment including indirect effects—A case study using input-output analysis. *Environ. Impact Assess. Rev.* **2003**, *23*, 263–282. [CrossRef]
126. Hawkins, T.; Hendrickson, C.; Higgins, C.; Matthews, H.S.; Suh, S. A mixed-unit input-output model for environmental life-cycle assessment and material flow analysis. *Environ. Sci. Technol.* **2007**, *41*, 1024–1031. [CrossRef] [PubMed]
127. Igos, E.; Rugani, B.; Rege, S.; Benetto, E.; Drouet, L.; Zachary, D.S. Combination of equilibrium models and hybrid life cycle-input-output analysis to predict the environmental impacts of energy policy scenarios. *Appl. Energy* **2015**, *145*, 234–245. [CrossRef]

128. Kennelly, C.; Berners-Lee, M.; Hewitt, C.N. Hybrid life-cycle assessment for robust, best-practice carbon accounting. *J. Clean. Prod.* **2019**, *208*, 35–43. [CrossRef]
129. Yi, I.; Itsubo, N.; Inaba, A.; Matsumoto, K. Development of the interregional I/O based LCA method considering region-specifics of indirect effects in regional evaluation. *Int. J. Life Cycle Assess.* **2007**, *12*, 353–364. [CrossRef]
130. De Rosa, M.; Knudsen, M.T.; Hermansen, J.E. A comparison of Land Use Change models: Challenges and future developments. *J. Clean. Prod.* **2016**, *113*, 183–193. [CrossRef]
131. Saad, R.; Margni, M.; Koellner, T.; Wittstock, B.; Deschênes, L. Assessment of land use impacts on soil ecological functions: Development of spatially differentiated characterization factors within a Canadian context. *Int. J. Life Cycle Assess.* **2011**, *16*, 198–211. [CrossRef]
132. Bolwig, S.; Ponte, S.; Toit, A.; Riisgaard, L.; Halberg, N. Integrating Poverty and Environmental Concerns into Value-Chain Analysis A Conceptual Framework. *Dev. Policy Rev.* **2010**, *28*, 173–194. [CrossRef]
133. Kaplinsky, R.; Morris, M. A Handbook for Value Chain Research. Available online: http://www.fao.org/fileadmin/user_upload/fisheries/docs/Value_Chain_Handbool.pdf (accessed on 7 February 2019).
134. Fearne, A.; Garcia Martinez, M.; Dent, B. Dimensions of sustainable value chains: Implications for value chain analysis. *Supply Chain Manag. Int. J.* **2012**, *17*, 575–581. [CrossRef]
135. Fasse, A.C.; Grote, U.; Winter, E. Recent developments in applying environmental value chain analysis. *Environ. Econ.* **2011**, *2*, 74–86.
136. De Marchi, V.; Di Maria, E.; Micelli, S. Environmental Strategies, Upgrading and Competitive Advantage in Global Value Chains. *Bus. Strateg. Environ.* **2013**. [CrossRef]
137. Mahutga, M.C. When do value chains go global? A theory of the spatialization of global value chains. *Glob. Netw.* **2012**, *12*, 1–21. [CrossRef]
138. Panichelli, L.; Gnansounou, E. Impact of agricultural-based biofuel production on greenhouse gas emissions from land-use change: Key modelling choices. *Renew. Sustain. Energy Rev.* **2015**, *42*, 344–360. [CrossRef]
139. Voinov, A.; Shugart, H.H. "Integronsters", integral and integrated modeling. *Environ. Model. Softw.* **2013**, *39*, 149–158. [CrossRef]
140. Hellweg, S.; Frischknecht, R. Evaluation of Long-Term Impacts in LCA. *Int. J. Life Cycle Assess.* **2004**, *9*, 339–341. [CrossRef]
141. Magliocca, N.R.; Rudel, T.K.; Verburg, P.H.; McConnell, W.J.; Mertz, O.; Gerstner, K.; Heinimann, A.; Ellis, E.C. Synthesis in land change science: Methodological patterns, challenges, and guidelines. *Reg. Environ. Chang.* **2014**, *15*, 211–226. [CrossRef]
142. Franzen, A.; Mader, S. Consumption-based versus production-based accounting of CO_2 emissions: Is there evidence for carbon leakage? *Environ. Sci. Policy* **2018**, *84*, 34–40. [CrossRef]
143. Steininger, K.; Lininger, C.; Droege, S.; Roser, D.; Tomlinson, L.; Meyer, L. Justice and cost effectiveness of consumption-based versus production-based approaches in the case of unilateral climate policies. *Glob. Environ. Chang.* **2014**, *24*, 75–87. [CrossRef]
144. Ferng, J.J. Allocating the responsibility of CO2over-emissions from the perspectives of benefit principle and ecological deficit. *Ecol. Econ.* **2003**, *46*, 121–141. [CrossRef]
145. Lancker, E.; Nijkamp, P. A policy scenario analysis of sustainable agricultural development options: A case study for Nepal. *Impact Assess. Proj. Apprais.* **2000**, *18*, 111–124. [CrossRef]
146. Heck, V.; Gerten, D.; Lucht, W.; Popp, A. Biomass-based negative emissions difficult to reconcile with planetary boundaries. *Nat. Clim. Chang.* **2018**, *8*, 151–155. [CrossRef]
147. Bjørn, A.; Hauschild, M.Z. Introducing carrying capacity-based normalisation in LCA: Framework and development of references at midpoint level. *Int. J. Life Cycle Assess.* **2015**, *20*, 1005–1018. [CrossRef]
148. Clift, R.; Sim, S.; King, H.; Chenoweth, J.L.; Christie, I.; Clavreul, J.; Mueller, C.; Posthuma, L.; Boulay, A.M.; Chaplin-Kramer, R.; et al. The challenges of applying planetary boundaries as a basis for strategic decision-making in companies with global supply chains. *Sustainability* **2017**, *9*, 279. [CrossRef]