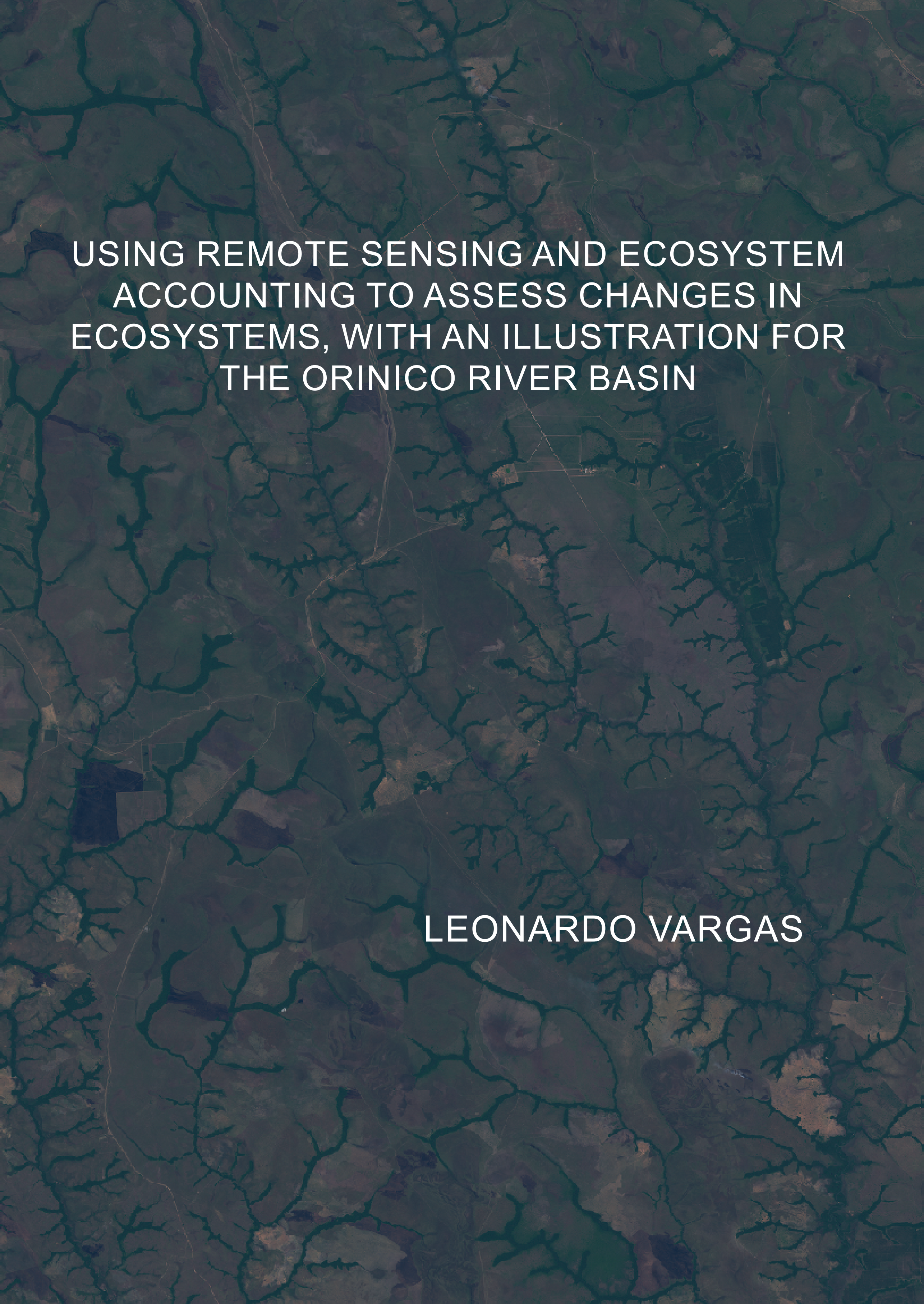

USING REMOTE SENSING AND ECOSYSTEM ACCOUNTING TO ASSESS CHANGES IN ECOSYSTEMS, WITH AN ILLUSTRATION FOR THE ORINICO RIVER BASIN

LEONARDO VARGAS

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Thesis committee

Promotor
Prof. Dr L. G. Hein
Professor Environmental System Analysis
Wageningen University & Research

Co-promotor
Dr L. Willemen
Faculty of Geo-Information Science and Earth Observation (ITC)
University of Twente

Other members
Dr Andrew Farrow, GeAgrofía, Wageningen, The Netherlands
Dr Leon Braat, Senior Researcher Ecosystem Services & Biodiversity, Wageningen University & Research
Prof. Dr Carolien Kroeze, Professor of Water Systems and Global Change, Wageningen University & Research
Prof. Dr Arnold K. Bregt, Professor of Geo-information Science Wageningen University & Research

This research was conducted under the auspices of the Graduate School for Socioeconomic and Natural Sciences of Environment (SENSE).

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LEONARDO VARGAS

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CONTENTS

Chapter 1

General introduction

1.1 BACKGROUND

1.1.1 ECOSYSTEMS CHANGE

There is a need to better understand the contribution of ecosystems to sustain human economic development. Global economic development and human population have been growing more rapidly during the last century than any other period of the human history (Lewis 2012; Steffen et al. 2015a). This achievement, however, could not be reached without the contribution of ecosystems through the supply of essential ecosystem services (see Box 1.1. for a definition of ecosystems and ecosystem services). Such contribution includes key ecosystem's processes that support the production of food products derived from animals, fungus and plants, the production of materials derived from wood and jute, and the production of energy derived from plants and microbes (Kumar 2010; MA 2003). Our increasing unsustainable production of food, materials, and energy to satisfy the needs of a growing population also accelerate the change and degradation of ecosystems (MA 2003; Steffen et al. 2015a). During the last fifty years, agricultural expansion has cleared or transformed 70% of the grasslands, 50% of the savannahs, 45% of the temperate deciduous forests and 27% of the tropical forests (Foley et al. 2011; WRI et al. 2015). In addition, 60% of the agricultural land has been degraded by erosion, nutrient depletion, salinization, and compaction (UNCCD 2017; WRI et al. 2005). Furthermore, agricultural and industrial activities have modified the cycles of water, carbon, nitrogen and phosphorous, reaching dangerous levels that can potentially trigger global regime shifts (MA 2003; Rockström et al. 2009; Steffen et al. 2015b). Nevertheless, the production of rice, wheat, and maize should increase by 40% and the production of livestock by more than 60% in order to feed a growing global population projected to reach 10 billion people by 2050 (Díaz et al. 2018; Searchinger et al. 2014; WRI et al. 2000). The required increase in food production may be unattainable if the current unsustainable economic development strategies that promote global economic growth but ignore the consequences of an accelerated change on ecosystems continue unabated.

The role of natural resources in the functioning of an economic system has increasingly received sporadic consideration from economists. A key contribution of natural resource economics has been to endorse the natural environment as a form of capital asset or natural capital (Dasgupta and Heal 1979; Haslinger 1984). Costanza and Daly (1992) defined natural capital as "a stock that yields a flow of goods and services into the future", highlighting the importance of the natural capital in the future supply of ecosystem services that fulfil and sustain human life. Ekins (1992) proposed functions for the natural capital including the provision of resources for production (e.g. food, fuel, and metals), absorption of waste from production, life-support, and pleasantness functions (e.g. beautifulness, inspiration). These critical functions attributed to the natural capital included the interaction of key ecological properties and processes in ecosystems, by which essential ecosystem services are supplied to sustain the economy, referred as the ecological capital (Atkinson et al. 2012; Barbier 2009; Barbier 2013).

BOX 1.1. ECOSYSTEMS AND ECOSYSTEM SERVICES

The MA defined ecosystem as "a dynamic complex of plant, animal and microorganisms communities and the non-living environment interacting as a functional unit" (MA 2003). The comprising types and abundance of the biotic and abiotic components of the ecosystem can be referred as ecosystems composition, and the distribution and arrangements of this components can be referred as ecosystems structure (Schmelzer 2015; Wallace 2007; WRI et al. 2005; WRI et al. 2015). The interaction between the biotic components, and between biotic and abiotic components of ecosystems, transfer energy and materials through the system. This transfers of energy and materials can be described as ecosystems processes, underpinned by ecosystems composition and structure (Chapin III et al. 2011; Lyons et al. 2005).

The term ecosystem service was coined by Ehrlich and Ehrlich (1981) as a bridging concept with notions from the social and the natural sciences (Braat and de Groot 2012). The concept of ecosystem services gained widespread attention with the publication of Daily (1997) and Costanza et al. (1998), and the establishment of ecological-economics as a bridging discipline between ecology and the economic sciences (Braat and de Groot 2012; Kumar 2010; WRI et al. 2005). Whereas the ecological roots of the concept highlighted the functions of nature to human society, its economic roots highlighted that the valuation of the contribution of ecosystems to human welfare was not adequately quantified in terms comparable to economic services and manufactured capital (Costanza et al. 1997; Foley et al. 2011). Under this perspective, non-marketed ecosystem services were overviewed as positive externalities that can be included in economic decision making as longs as they are valued in monetary terms. With the ambition to integrate measures of ecosystem services with the standard national accounts, the System of Environmental-Economic Accounting 2012-Experimental Ecosystem Accounting (SEEA-EEA) defined ecosystem services as the contribution of ecosystems to benefits used in economic and non-economic activities, making a clear distinction between ecosystem services and benefits. The SEEA-EEA rationale for measuring ecosystem services is that the inputs directly taken from ecosystems to economic activities such as agriculture and forestry are not recorded in the standard accounting framework and hence the contribution of ecosystems as suppliers is not recognized. The definition of ecosystem services used in the SEEA-EEA aligns with that of the Economics of Ecosystems and Biodiversity (TEEB). However, the MA and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) use slightly different definitions. The MA defined ecosystem services as "the benefits that people obtain from ecosystems", and the IPBES use the term nature's contribution to people (NCP) to extend the concept of ecosystem services to more broadly include interactions between people and ecosystems (Díaz et al. 2015; Díaz et al. 2018; Kumar 2010; MA 2003).

To avert decline in the stock of natural capital over time, it is essential to include ecosystems and ecosystem services, which sustain our economic development (Barbier 2016; Costanza et al. 2017; Dasgupta 2010; Ekins et al. 2003). The importance of ecosystems and ecosystems services – *natural capital* - in sustaining global economic development is also shared by international initiatives such as the Global Biodiversity Assessment (GBA), the Millennium Ecosystem Assessment (MA), The Economics of Ecosystems and Biodiversity (TEEB), and the Intergovernmental Platform for Biodiversity and Ecosystem Services (IPBES). The MA highlighted the importance of assessing changes; in the biophysical flow of ecosystem services, in the condition of ecosystems, and in well-being including changes in the total value of the benefits obtained from ecosystems. The MA recognized the importance of natural capital as a key determinant of human well-being, and as a component of the total wealth of each nation. The IPBES highlighted the importance of natural capital *–nature and nature's benefits to people-* as a key element that contributes to enhance our quality of life (Costanza et al. 2017; Díaz et al. 2015; Díaz et al. 2018). The MA and the IPBES has also documented the global importance of natural capital. Therefore, incorporating natural capital in economic accounts is important to inform policy-making at national and sub-national levels (Dalmazzone and La Notte 2013; Mäler et al. 2008). The existing System of National Accounts (SNA), which has over several decades been the main organizing principle for compiling economic accounts, is, however, not suitable for compiling complex environmental information and is limited in capturing the complexity of how the economy, society, and nature are related (Barbier 2016; Bartelmus 2009; Costanza et al. 2017; Kumar 2010).

1.1.2 ECOSYSTEM ACCOUNTING

The SNA is an international standard set of recommendations on how to compile measures of economic activity using a coherent, consistent and integrated set of economic accounts (United Nations et al. 2009). Incorporating environmental information into the SNA to allow for the assessment of changes in the natural capital of a country has been the interest of part of the scientific community (e.g. ecologists, economists) over the last three decades (Bartelmus 2014; Obst et al. 2016). A further step in achieving this vision was the development of the international statistic standard for environmental-economic accounting, the System of Environmental-Economic Accounting 2012-Central Framework (SEEA-CF), adopted in 2012 by the United Nations Statistical Commission (United Nations et al. 2014a). The SEEA-CF is a conceptual framework developed to understand the relation between the environment and the economy, describing stocks and changes in the stocks of environmental assets (United Nations et al. 2014a). In the SEEA-CF, natural capital refers to all environmental assets including biotic and abiotic natural resources (e.g. fish and timber stocks, mineral and energy resources) and ecosystems. However, the analysis on how different environmental assets interact as part of natural processes within a specific area to provide multiple services used in human economic and non-economic activities requires a complementary approach, which is synthesized in the System of Environmental-Economic Accounting 2012-Experimental Ecosystem Accounting (SEEA-EEA) (in short "ecosystem accounting") (United Nations et al. 2014b).

Ecosystem accounting incorporates a conceptual framework describing the relationship between stocks and flows to understand the connections between ecosystems and the economy (Fig 1.1). In this framework, stocks comprise spatially explicitly defined ecosystems -*ecosystem assets*-, and flows -*ecosystem services*- embrace the material (e.g. animals, plants and water) and non-material flows (e.g. recreation, and landscape contemplation) between ecosystems and from ecosystems to the economy (Fig 1.1). Measurements in ecosystem assets (in this thesis ecosystem assets are referred as ecosystems) entail the assessment of ecosystems in terms of (i) extent, (ii) condition, and (iii) an expected flow of ecosystem services. Whereas assessing the extent of an ecosystem reveals its size and location, assessing condition reveals the overall quality of an ecosystem in terms of its characteristics (e.g. water, soil, elevation, temperature, vegetation).

Assessing an expected flow of ecosystem services reveals the capacity of the ecosystem to supply ecosystem services given future changes in extent and condition. The United Nations et al. (2017) highlighted that the concept of capacity was mentioned in ecosystem accounting but was not clearly defined, calling for a clear definition of the concept regarding its importance to assess sustainable use of ecosystems. Hein et al. (2016) defined capacity for accounting purposes as “the ability of an ecosystem to supply an ecosystem service under current ecosystem condition and use, at the maximum yield and use, that does not negatively affect the future supply of the same ecosystem service or other ecosystem services”. Furthermore, Hein et al. (2016) brought the concept of potential to define the ability of an ecosystem to supply ecosystem services irrespective of the demand for ecosystem services. The concept of potential is required, because in the absence of a demand for ecosystem services there is no exchange value for the service, and capacity and supply will be zero. Measuring the extent, condition and expected supply of ecosystem services in terms of capacity and potential is of keen importance for a sustainable use of ecosystems.

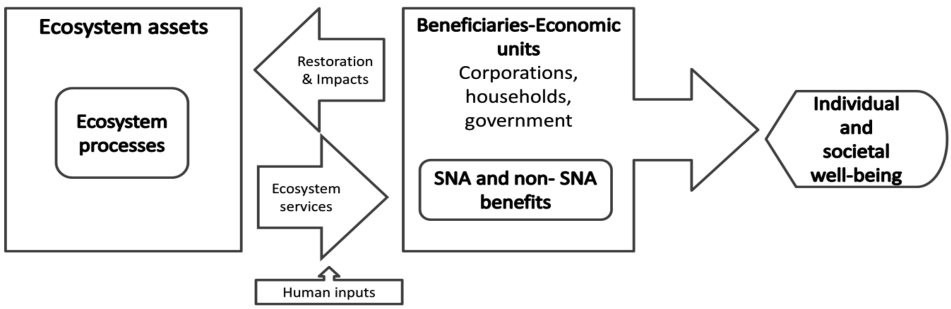

Fig. 1.1 In this basic model stocks are represented by the biotic and abiotic components of an ecosystem asset where ecosystem processes and its characteristics describe the functioning of an ecosystem. Flows are represented by the supply of a basket of ecosystem services generated by ecosystem assets, where the supply of ecosystem services combined with human inputs (e.g. labour) contribute to the production of benefits. These benefits can be included in the economic production boundary (SNA benefits) or can be received by individuals without being produced by economic units. Both SNA and non-SNA benefits contribute to individual and societal well-being. Adapted from United Nations et al. (2017)

1.1.3 ASSESSING ECOSYSTEMS CHANGE USING ECOSYSTEM ACCOUNTING

Assessing changes in ecosystems using ecosystem accounting entails a clear delineation of well-defined boundaries that allow the organization of information, and the presentation of accounts at a specific scale of analysis (e.g. ecosystem, river basin). To this end, cartographical and statistic information are required, including among others, land cover, meteorological, hydrological, soil, and population data. This information is collected from a broad variety of sources such as remote sensing, on-ground assessments, and surveys of landowners. Such information is organized using 3 statistic units; basic spatial unit (BSU), ecosystem assets (EA), and ecosystem accounting areas (EAA). The EAA defines areas for reporting purposes, including among others, administrative boundaries, management areas (e.g. national parks) and river basins. The BSU define small areas, typically overlapping a grid on a map, such as land parcels of a cadastre or remote sensing pixels. The EA defines contiguous areas formed by ecosystems with land cover as the starting point because land cover reflects the various natural and modified systems in a specific location at a certain point in time. Ecosystem accounting adopted, as a starting point, 15 land cover classes based on the Land Cover Classification System version 3 (LCCS 3) from the Food and Agriculture Organization of the United Nations (FAO) (FAO 2009). Additional information such as soil type, elevation, temperature, and precipitation, is also included to define the boundaries of each EA. EAs are aggregated in ecosystem types (ETs). ETs are classes of similar types of ecosystem assets (e.g. an ecosystem type may correspond to a land cover unit such as broad leaved forest). The EA and the ET are central for ecosystem accounting since all ecosystem accounts are connected to either or both of these units (e.g. supply, condition and capacity accounts). Monitoring changes in extent, condition and capacity accounts over different accounting periods is relevant to assess changes in the extent and the condition of ecosystems, and in the expected supply of ecosystem services. However, using ecosystem accounting to assess changes in ecosystems is often challenged by (i) the absence of spatially explicit information and statistical data, (ii) inadequate knowledge on how the complex non-linear dynamics (e.g. resilience, ecological thresholds) that characterize social-ecological systems can be integrated to ecosystem accounting (Hein et al. 2015; Weber 2014). Additionally, ecosystem accounting has been barely used to assess changes in human-managed ecosystems such as agricultural systems.

Detailed data is often non-existent or scarce, inaccessible and expensive, particularly in developing countries (Crossman et al. 2012a; Hein et al. 2015). Because detailed datasets are necessary to populate the different ecosystem accounts (e.g. supply, condition and capacity accounts), missing data constrain monitoring changes between accounting periods. This is a challenge in the assessment of ecosystem change which must be addressed. In practical terms, missing data has to be estimated with spatially explicit models. Remote sensing provides timely data over large coverages and can be a useful source of spatially explicit data at relatively low cost. In the past, remotely sensed data has been used for the assessment of changes in land cover, water availability, elevation, productivity, over large areas (Andrew et al. 2014; Ayanu et al. 2012b). Currently, there is a need to explore how remotely sensed data can be used following the ecosystem accounting guidelines, with a specific focus on analysing ecosystems. To date, majority of research in this context have focussed mainly on using remotely sensed data to assess changes in the supply of ecosystem services rather than changes in ecosystems (Burkhard et al. 2012; Crossman et al. 2012c; Egoh et al. 2012; Martínez-Harms and Balvanera 2012). Furthermore, there is a need to know if remotely sensed data can be used to assess changes in the capacity and potential of ecosystems to supply ecosystems services over large areas.

Social-ecological systems are integrated systems in which the interaction of people with the natural components of the system (e.g. ecosystems) is complex and subject to non-linear dynamics (e.g. ecological thresholds, resilience) (Liu et al. 2007). Gradual changes in ecosystems caused by human economic activities can shift these systems once ecological thresholds are crossed (Scheffer et al. 2001; Scheffer et al. 2012). Given that ecosystems are spatially connected, crossing ecological thresholds in these systems can propagate and cause a shift in the whole earth system (Barnosky et al. 2012; Peters et al. 2009). Assessing limits for human activities to avoid crossing ecological thresholds inspired the development of quantitative frameworks such as, planetary boundaries, safe minimum standards, and limits to grow (Crowards 1998; Meadows et al. 2004; Rockström et al. 2009). Although ecosystem accounting includes the concepts of resilience and ecological thresholds to highlight the importance of complex dynamics controlling ecosystems behaviour, a clear approach to define ecological thresholds was not mentioned (United Nations et al. 2014b). The lack of understanding on how ecological thresholds can be defined in the context of ecosystem accounting is a limiting factor for assessing the limits of undesirable and unsustainable changes in ecosystems, which is necessary to avoid critical transitions at such level.

Assessing changes in the natural capital underpinning agricultural production using an ecosystem accounting approach is challenging considering the type of benefits to which these systems contribute and the difficulties in determining a clear production boundary. Ecosystem services in the context of ecosystem accounting are perceived as the contributions of ecosystems used in human activities. “Contributions” is a keyword in the definition and imply that ecosystem services are combined with other inputs (e.g. labour) to produce benefits (SNA type). However, for some ecosystem services (e.g. air filtration, carbon sequestration) few human inputs are required for the generation of benefits (non-SNA type). Hence, the supply of such services are equivalent to their associated benefit. In addition, distinguishing the nature or type of benefits i.e. public or private benefits, is also important. Agricultural systems are privately owned assets that generate multiple ecosystem services (carbon sequestration, crop supply) which contribute to both public (e.g. carbon sequestration) and private benefits (e.g. crop supply). The ecosystem services generated by agricultural systems challenge the SNA because these private producers generate public benefits unintentionally (United Nations et al. 2014b; United Nations et al. 2009). Moreover, agriculture is a joint production process that involves ecosystem services (e.g. pollination), and human inputs (e.g. labour), hence, determining a production boundary that defines inputs from ecosystem services in the production of agricultural benefits is difficult. Furthermore, the various ecosystem accounting concepts applied to assess changes in ecosystems (e.g. extent, condition, capacity, ecosystem services flow and supply) have not been explored to assess changes in the natural capital of agricultural systems.

1.2 KNOWLEDGE GAPS

From the above, the following knowledge gaps have been recognized and will be addressed in this thesis:

1. Lack of evidence on how remote sensing information can be used in support of ecosystem accounting to assess changes in ecosystems.
2. Lack of evidence on how remote sensing information can be used in support of ecosystem accounting to assess changes in the capacity of ecosystems to supply ecosystem services
3. Lack of information on how the concepts used in the planetary boundaries framework to understand complex dynamics of socio-ecological systems can be used in ecosystem accounting.
4. Lack of information on how the natural capital underpinning agricultural production can be assessed using ecosystem accounting concepts.

These knowledge gaps exist because the spatially explicit approach inherent in ecosystem accounting in order to assess changes in ecosystems require new methods and concepts which are still emerging. The spatial perspective applied in ecosystem accounting depends on advances in geo-information science (GIS) methods (e.g. mapping, modelling), and instruments (e.g. satellites, drones), that enhance data availability, reliability, quality, and accuracy. New emerging concepts such as ecosystems capacity and potential to supply services, that support the assessment of an expected supply of services, need to be included in ecosystem accounting.

In this thesis, I use ecosystem accounting, and spatially explicit remote sensing data from the Moderate Resolution Imaging Spectroradiometer (MODIS) to assess changes in ecosystems in terms of extent, condition and capacity to supply ecosystem services, within the Orinoco river basin.

1.3 RESEARCH OBJECTIVES

This PhD aims to address the four afore-mentioned knowledge gaps. The overall research objective is to increase our knowledge on how remote sensing data can be used to support ecosystem accounting for the assessment of unsustainable changes in ecosystems in a large river basin. Four research sub-objectives are formulated to achieve the overall objective:

1. To examine if and how ecosystems can be analysed at large scale with the use of information provided by remote sensing. (Chapter 2);
2. To analyse how remote sensing spectral information can be used to support the assessment of the capacity of ecosystems to supply ecosystem services for large areas (Chapter 3);
3. To examine if and how planetary boundaries framework can be used in combination with ecosystem accounting for sustainable natural resource management at the level of a large river basin (Chapter 4);
4. To explore if and how concepts used in ecosystem accounting and yield gap analysis can be used to assess changes in the natural capital of agricultural systems (Chapter 5).

The four research sub-objectives are addressed in four scientific papers, presented in chapters 2-5 of this PhD thesis.

In chapter 2, I address the first research sub-objective (Table 1.1). The novelty of this chapter is the application of remote sensing indices derived from the MODIS products for the analysis of changes in ecosystems following the ecosystem accounting guidelines in a large river basin. I use the MODIS land cover product MCD12Q1 to analyse annual changes in the extent of six ecosystems. I use the Enhanced Vegetation Index (EVI) and the Normalized Difference Water Index (NWDI) from the MODIS vegetation indices product MOD13A3, and Land Surface Temperature (LST) from the MODIS product MOD11A2 to analyse changes in ecosystem condition. In addition, I use the Net Primary Productivity (NPP) from the MODIS NPP product MOD17A3 to analyse changes in the capacity of ecosystems to supply ecosystem services. The spatially explicit information that these MODIS products provide can be compiled using ecosystem accounting spatial units and ecosystem accounts enabling the assessment of changes in the condition of ecosystems over time.

In chapter 3, I address the second research sub-objective building upon the potential use of NPP derived from MODIS MOD17A3 to analyse changes in the capacity of ecosystems to supply ecosystem services addressed in chapter 2 (Table 1.1). In this chapter, I present new insights on the use of NPP data from the MODIS MOD17A3 combined with statistics to model changes in the capacity of ecosystems to supply ecosystem services following the ecosystem accounting guidelines. I use the annual accumulated NPP in six ecosystems for five years between 2001 to 2014 to quantify the amount of aboveground biomass allocated to supply four ecosystem services. I use agricultural statistics to model the supply of ecosystem services and compare the capacity of each ecosystem with the supply of ecosystem services. MODIS NPP is a useful source of spatially explicit information to model changes in the capacity of ecosystems to supply ecosystem services at a large scale in data poor contexts.

In chapter 4, I address the third research sub-objective (Table 1.1). The novelty of this chapter is the combination of integrated approaches to understand complex social-ecological systems at the level of river basin. I use two sets of criteria to compare and contrast the planetary boundaries and ecosystem accounting frameworks providing a general overview based on contextual criteria and an in depth comparison based on structural criteria. In addition, I assess the applicability of these frameworks for a sustainable natural resources management in the Colombian Orinoco river basin. The similarities and differences between planetary boundaries and ecosystem accounting are useful to overcome weaknesses and to reinforce strengths for natural resource management at the level of river basin. While the planetary boundaries framework provides a stronger interpretation of sustainability enabling the understanding of environmental risks, ecosystem accounting develops a comprehensive framework allowing the monitoring of environmental change.

In chapter 5, I address the fourth research sub-objective (Table 1.1). The innovation of this chapter is the assessment of the natural capital underpinning agricultural production using ecosystem accounting concepts, yield gap analysis, remotely sensed data and agricultural statistics. I use the ecosystem accounting concepts of extent, potential and capability, and the concepts of water-limited crop potential yield, the yield gap and water productivity as used for yield gap analysis. I use remotely sensed data from the MODIS land cover, the MODIS NPP and the MODIS evapotranspiration to estimate the extent of two agricultural systems, the capability of these two systems to produce six different crops and their water productivity. I combine MODIS NPP with agricultural statistics to link the capability to produce crops with the annual yield of each crop. Monitoring the extent of agricultural systems is key to stop their expansion over critical natural areas. Monitoring agricultural systems potential and capability is key to assess changes in their ability to sequester carbon and to produce crops. The yield gap is the basis to apply a capability analysis where a crop yield level can be linked with the agricultural system capability to produce crops. This analysis is important to establish tradeoffs between ecosystem services, as agricultural systems prioritize the production of food and biofuels over the supply of other ecosystem services. Monitoring water productivity is key to assess changes in the efficiency of producing food per cubic meter of water. Remotely sensed is a powerful tool to obtain information that supports ecosystem accounting in the assessment of changes in the natural capital underpinning agricultural production.

1.4 STUDY AREA

The Orinoco is a transboundary river basin, located in the north of South America in the countries of Colombia and Venezuela. The Colombian side of the river basin covers an area of 345,000 km^2 (Fig 1.2). It collects waters from the Andes mountains, the Guyana shield, and floodplains between the Orinoco and Amazon river basin (Barletta et al. 2016; León 2005). More than 245,000 km^2 of the study area are covered by natural forests, páramos, rivers, lakes, wetlands, woody grasslands and natural savannahs. Likewise, more than 90,000 km^2 are used to graze 5 million of cattle heads, 4,000 km^2 to harvest rice and oil palm, and 1,000 km^2 for other crops (e.g. soy, maize and cassava) (Dane and Ministry of Agriculture 2016; Fedegan 2014; Fedepalma 2015).

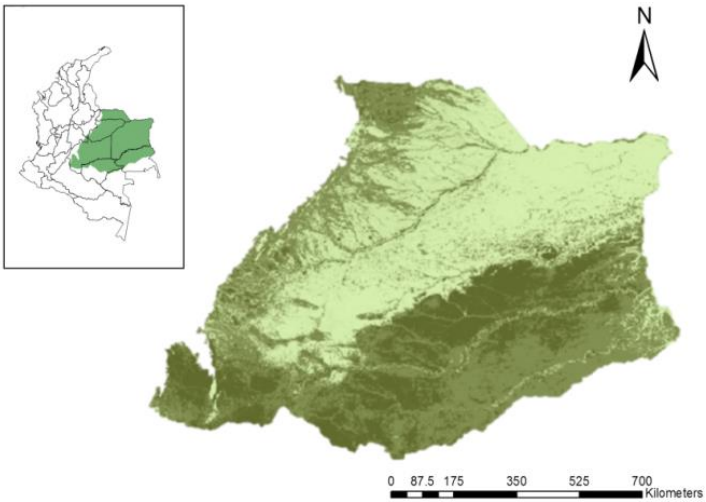

Fig 1.2 Map of the Colombian Orinoco river basin showing in dark green dense vegetation (e.g. forests) and in light green sparse vegetation (e.g. savannahs)

Agriculture is the most important economic activity in the river basin, supporting a growing population reaching 1.4 million inhabitants by 2014 (Benavides 2010; Dane and Ministry of Agriculture 2016). In 2014, livestock husbandry provided 160,000 jobs, rice and oil palm production provided 60,000 jobs , and other crops such as soy, maize, cassava and rubber provided 80,000 jobs (Benavides 2010; Dane and Ministry of Agriculture 2016). Even though large areas in the river basin are not used for agriculture, still 40,000 km^2 can be potentially transformed from natural ecosystems (e.g. wetlands and savannahs) to agriculture (Benavides 2010; CONPES 2014). According to Etter et al. (2010) between 1970-2000, more than 1,000 km^2 of natural savannahs in the river basin were annually transformed for pastures and 100 km^2 for crops (e.g. oil palm). Etter et al. (2010) projected that about 22,000 km^2 of natural savannah area will have been cleared by 2020. Reducing the rate of expansion of agriculture in the river basin is of keen importance, because the Orinoco savannahs contain 55% of all the wetlands, 40% of the subterraneous waters, 46% of birds, and 40% of the fish species of the country, and is an important biological corridor between the Amazon and the Andes (Romero-Ruiz et al. 2012b).

1.5 THESIS OUTLINE

This PhD thesis consists of six chapters (Table 1.1). After providing the background information and thesis' objectives, in chapters 2-5 I focus on research objectives. Table 1.1 gives an overview of the different inputs, methods and outputs used to address the research sub-objectives in chapters 2-5. Chapters 2 and 3 are linked, the MODIS land cover and NPP products downloaded and processed in chapter 2 are used as inputs for modelling and mapping ecosystems capacity to supply ecosystem services in chapter 3. The main outputs from chapter 3 are capacity maps for different ecosystems, these maps are used in chapter 4 to illustrate the application of planetary boundaries and ecosystem accounting for natural resource management in the Orinoco river basin. The MODIS land cover and NPP products downloaded and processed for chapter 1 are used as inputs for chapter 5, however, a new land cover map was used to adjust areas covering agricultural systems (e.g. cropland and pastures). In chapter 6, I reflect upon the methods and results of this thesis and I draw the main conclusions. From my research the reader can gain better understanding on the different aspects and applications of the ecosystem accounting approach, particularly focusing on assessing changes in ecosystems rather than changes in the supply of ecosystems services. This information is important when developing effective environmental policies. This work might positively contribute to decrease the rapid increase in ecosystems change, and hopefully can be used as an example for other river basins where data availability is a strong limiting factor to inform policy making.

Table 1.1. Overview of the research presented in this thesis. The table shows the research sub-objectives, corresponding chapters, main input, the method selected, and the main output

Research sub-objective	Chapter	Main input		Methods	Main output
Examine if and how ecosystems can be analysed at large scale with the use of information provided by remote sensing	2	**Maps** National land cover/land use map **Literature** Economic activities Ecosystem accounting	**Remote sensing** MODIS MCD12Q1 Land cover MODIS MOD13A3 Vegetation indices MODIS MOD11A2 Land surface Temperature MODIS MOD17A3 Net Primary Productivity	**Remote sensing** Time series analysis (bi-annual changes) **Ecosystem accounting** Concepts Spatial units	Maps and tables concerning changes in ecosystem extent, condition and capacity to supply ecosystem services
Analyse how remote sensing spectral information can be used to support the assessment of the capacity of ecosystems to supply ecosystem services for large areas	3	**Literature** Economic activities Ecosystem accounting NPP allocation models Agriculture	**Remote sensing** MODIS MCD12Q1 Land cover MODIS MOD17A3 Net Primary Productivity **Statistical data** Agricultural production Timber harvesting Forest products	**Remote sensing** Time series analysis (multiple years) **Modelling** Ecosystem services capacity and supply following ecosystem accounting guidelines	Maps and tables concerning modelled changes in ecosystem extent and capacity to supply ecosystem services. Table comprising modelled changes in the supply of ecosystem services
Examine if and how planetary boundaries can be used in combination with ecosystem accounting in proposing limits on human activities at the level of a large river basin	4	**Maps** Land cover and capacity from chapter 3 **Literature** Economic activities Ecosystem accounting Agriculture Planetary boundaries Socio-ecological systems Ecological thresholds Adaptive management	**Frameworks** Planetary boundaries for nitrogen, phosphorus and water cycles and land system change. Ecosystem accounting; ecosystems extent, condition, capacity to supply ecosystem services	**Comparing frameworks** Contextual and structure criteria **Applicability** Adaptive management criteria	Maps, figures and tables providing an overview on the complementary between both frameworks and the applicability for adaptive natural resource management
Explore if and how concepts used in ecosystem accounting and yield gap analysis can be used to assess changes in the natural capital of agricultural systems	5	**Literature** Economic activities Ecosystem accounting Yield gap analysis	**Remote sensing** MODIS MCD12Q1 Land cover MODIS MOD17A3 Net Primary Productivity MODIS MOD16A2-A3 Evapotranspiration **Statistical data** Agricultural production	**Remote sensing** Time series analysis (multiple years), **Modelling** Agricultural systems extent, capability and potential to supply ecosystem services following the ecosystem accounting guidelines	Maps, tables and figures about changes in agricultural systems extent, capability and potential to supply crops Yield gap figures and maps about changes in water productivity

Chapter 2

Accounting for Ecosystem Assets using Remote Sensing in the Colombian Orinoco River Basin lowlands

Abstract

In many parts of the world, ecosystems change compromises the supply of ecosystem services (ES). Better ecosystem management requires detailed and structured information. Ecosystem accounting has been developed as an information system for ecosystems, using concepts and valuation approaches that are aligned with the System of National Accounts (SNA). The SNA is used to store and analyse economic data, and the alignment of ecosystem accounts with the SNA facilitates the integrated analysis of economic and ecological aspects of ecosystem use. Ecosystem accounting requires detailed spatial information at aggregated scales. The objective of this paper is to explore how remote sensing images can be used to analyse ecosystems using an accounting approach in the Orinoco river basin. We assessed ecosystem assets in terms of extent, condition and capacity to supply ES. We focus on four specific ES: grasslands grazed by cattle, timber and oil palm harvest, and carbon sequestration. We link ES with six ecosystem assets; savannahs, woody grasslands, mixed agro-ecosystems, very dense forests, dense forest and oil palm plantations. We used remote sensing vegetation, surface temperature and productivity indexes to measure ecosystem assets. We found that remote sensing is a powerful tool to estimate ecosystem extent. The enhanced vegetation index can be used to assess ecosystems condition, and net primary productivity can be used for the assessment of ecosystem assets capacity to supply ES. Integrating remote sensing and ecological information facilitates efficient monitoring of ecosystem assets, in particular in data poor contexts.

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2.1 Introduction

Ecosystems supply a large variety of ecosystem services (ES) such as food, fibres, erosion control and climate regulation which sustain human life (Daily et al. 2009; Kumar 2010; MA 2005). The supply of ES depends on the dynamic interaction of the living (e.g. animal, plants, microorganisms) and non-living (e.g. soil, water) components of ecosystems (MA 2005). In many parts of the world, specifically in areas undergoing rapid land cover change, ecosystem change affects the supply of ES (MA 2005). Despite their importance, ecosystems, ES and the costs of ecosystem change are still not systematically considered in information sources for policy formulation such as the System of National Accounts (SNA) (Obst and Vardon 2014). The SNA is an information system that comprises physical and monetary information on economic production and consumption, and on produced capital assets in a set of consistent, structured accounts. However, it was not developed to compile complex non-monetary environmental information, and consequently the impacts of human economic activity on ecosystems are not reflected (Pedersen and Haan 2006).The increasing interest on how to incorporate complex non-monetary environmental information into the SNA promoted the development of the System of Environmental Economic Accounts (SEEA) (Bartelmus 2014; Obst and Vardon 2014; Pedersen and Haan 2006). The SEEA complements the SNA by integrating environmental (i.e. environment assets, natural resource stocks and flow accounts) and economic information (Bartelmus 2014; Pedersen and Haan 2006). An important step in the development of SEEA was the adoption of SEEA Central Framework (CF) by United Nations statistical commission as the international environmental accounting standard in 2012 (United Nations et al. 2014a). The SEEA-CF focuses on the assessment of individual environmental assets, defined as the natural living and non-living components of the Earth, constituting the biophysical environment that may supply benefits for humanity (United Nations et al. 2014a). In this definition the SEEA-CF included individual natural and biological resources such as land, timber, water, and ecosystems as individual components of the environment. The SEEA-CF did not included the non-material benefits supplied by the environment such as cultural and religious ES. A second perspective on environmental assets is described in the SEEA Experimental Ecosystem Accounting (EEA) which included the non-material benefits of the environment and encompasses interactions between the different individual environment assets within ecosystems (United Nations et al. 2014a).

The SEEA-EEA accounting approach represents a major step forward in environmental-economic accounting (Obst and Vardon 2014). The SEEA-EEA applies accounting concepts and rules in an integrated approach to assess the environment through the measurement of ecosystem assets and ES in line with the SNA standards (Hein et al. 2015; United Nations et al. 2014b). The ecosystem accounting framework included in SEEA ecosystem accounting allows the connection between stocks of ecosystems -*ecosystem assets*- and flows -*ecosystem services*- with other source of information concerning environment, economic and social aspects (United Nations et al. 2014b). Crucial in the SEEA-EEA approach, as in the SNA, is the distinction of ecosystem assets (related to the capacity of ecosystems to generate services as a function of ecosystem extent and condition, (e.g. Hein et al. (2015)) and ecosystem services, reflecting the use of ecosystems (United Nations et al. (2014b). Ecosystem assets in the context of ecosystem accounting are defined as spatial areas comprising biotic and abiotic components and other characteristics that function together, recognizing ecosystems as the underlying assets that supply ES (Costanza et al. 2014; Costanza et al. 2006; Daily 2000; Daily et al. 2009). Ecosystem assets in ecosystem accounting are measured from two perspectives: in terms of ecosystem condition and extent and in terms of ecosystems capacity to supply ES in the future. (United Nations et al. 2014a). Ecosystem extent refers to ecosystem asset location and size. Ecosystem condition refers to ecosystems overall quality expressed through key characteristics (e.g., soil type, climate, vegetation, water, elevation). Ecosystem capacity to supply ES refers to the future regeneration of ES after harvesting and extraction under current management, understood as a function of condition and extent (United Nations et al. 2014b). Ecosystem capacity to supply ES can also be defined as ecosystems long-time potential to deliver services under current sustainable management (Schröter et al. 2014), based on biophysical properties, social conditions, and ecological functions (Chan et al. 2006; Daily et al. 2009; Egoh et al. 2008). Difficulties in the compilation of complex spatial explicit biophysical information required for the measurement of ecosystem assets over large areas in data poor contexts represents a major constraint (Hein et al. 2015). Therefore, an innovative approach is required to assess ecosystem assets in ecosystem accounting, in particular in areas with poor data availability. The assessment of ecosystem assets in ecosystem accounting entails mapping their spatial distribution to allow the compilation of statistical information (United Nations et al. 2014b). Research on mapping ES has been rapidly increasing over the past decade (Crossman et al. 2012c; Egoh et al. 2012). A large number of ES mapping methodologies has been developed (Crossman et al. 2013; Eigenbrod et al. 2010; Martínez-Harms and Balvanera 2012), based on direct observations or proxies (e.g. estimated from land cover). Proxy based maps from remote sensing datasets are more common than maps based on primary data. While primary data is scarce, expensive and difficult to obtain, remote sensing provides relative low cost, available data commonly used to map ES (Ayanu et al. 2012a; Crossman et al. 2013). Remote sensing is important for assessing changes in ecosystem characteristics such as vegetation type, water availability, elevation, productivity, and to provide biophysical information over large areas in different time periods (Andrew et al. 2014). There is therefore a need to analyse how remote sensing can be used in support of ecosystem accounting, with a specific focus on analysing ecosystem assets given that most research on using remote sensing to date in this context has focussed on mapping ecosystem services rather than ecosystem assets (Burkhard et al. 2012; Crossman et al. 2012c; Egoh et al. 2012; Martínez-Harms and Balvanera 2012). For that reason remote sensing data and techniques are promising to compile statistic accounting information, combined with ground assessments, administrative data or land-owner surveys.

The aim of this paper is to examine if and how ecosystem assets can be analysed at large scale with the use of information provided by remote sensing. Specifically, we want to provide an innovative approach to map ecosystem assets in terms of ecosystem extent, condition and capacity to supply ES in line with the SEEA-EEA framework. For simplicity we will use ecosystem when we refer to ecosystem assets. In this section, first we present the Orinoco river basin and we define ecosystems based on two aspects; (i) a selection of relevant ES, and (ii) the main characteristics required to supply each ES. Second, we describe ecosystem accounting units. In the last section we introduce how we used remote sensing images to measure ecosystem assets, condition and capacity to supply ES.

2.2 Methods

2.2.1 The Colombian Orinoco river basin

The Orinoco is an international river basin located in South America, and extends over 958,500 km² between Colombia and Venezuela (Wolf et al. 1999). The study area covers the lowlands (all areas below 600 meters altitude above sea level) of the Colombian side of the Orinoco river basin (Fig 2.1), covering 311,100 km² with a population of 1.7 million inhabitants (Correa et al. 2005).

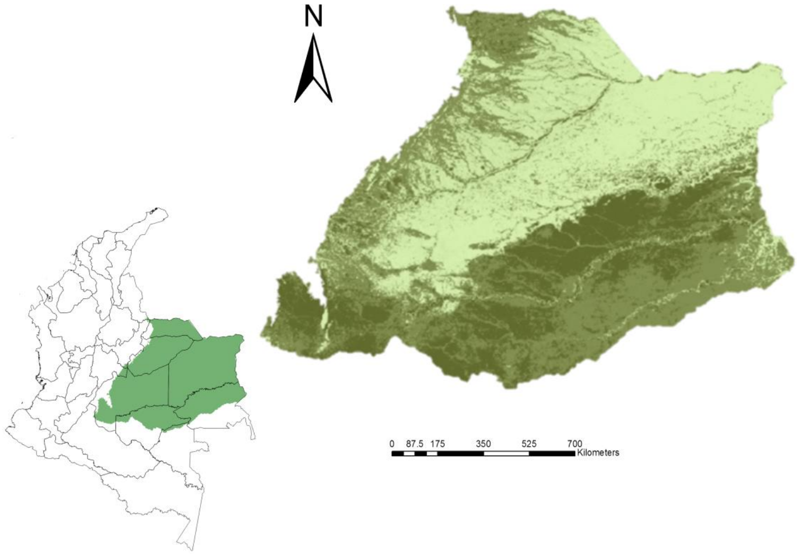

Fig 2.1 Vegetation land cover in the Orinoco river basin lowlands, showing in dark green areas with high vegetation density and in light green areas with low vegetation density.

We selected the lowlands because fast land use changes are occurring at this altitude with the introduction of crops such as palm oil and exotic grass species. The geographic boundaries of the study area are the Arauca river in the North, the Orinoco river in the East, the Inírida river in the south, and the foothills of the east Andes cordillera in the West (Ideam 2013; Romero et al. 2004; Rudas 2003). Natural forests covered more than 80,000 km^2 in 2010, from which the Amazon border is one of the main national deforestation hotspots (Ideam 2011a; Sanchez-Cuervo and Aide 2013). Current development strategies contribute to increase agriculture (i.e. oil palm, soy, sugar cane and rice), and livestock production (Viloria De la Hoz 2009). Oil palm species *Elaeis guineensis* and *Eleais oleífera* have been the crops with the highest expansion rate in the Orinoco river basin, increasing from 126 km^2 in year 2002 to 17,800 km^2 in 2013 (Benavides 2010; Fedepalma 2013; Ocampo 1996). Large areas have been transformed to grasslands to feed cattle and agricultural land. In the study area 97,500 km^2

were used to feed nearly 5 million cattle and 2,720 km^2 to harvest rice, oil palm and maize in 2008 (Benavides 2010).

2.2.2 The SEEA-EEA framework and remote sensing

ES and ecosystems

ES can be defined as the contributions of ecosystems to productive or consumptive activities and the last output before entering a production chain (Edens and Hein 2013; United Nations et al. 2014b). We selected four relevant ES based on land cover, economic importance, and data availability (Benavides 2010; Etter et al. 2006a; Romero-Ruiz et al. 2012b). The selected ES were: cattle grazing, timber harvesting, oil palm fresh fruit bunches (FFB) harvesting and carbon sequestration. We analysed six ecosystems: savannahs, mixed agroecosystem, woody grasslands, very dense forests, dense forests and oil palm plantations, by linking each selected ES with the ecosystem that supplies the service, using the SEEA-EEA guidelines (United Nations et al. 2014b) (Table 2.1).

Table 2.1 Link between ecosystem services, ecosystem characteristics to supply the service and the ecosystem that supply ES

Ecosystem Service	Ecosystem characteristics to supply ES	Ecosystem
Cattle grazing	Cattle graze the land dominated by herbaceous plants, with at most 40% cover by trees and/or shrubs (Suttie et al. 2005). Soils are acidic and compact (pH 3-5) with low organic matter (Lavelle et al. 2014). Average temperature 23ºC, precipitation 800 -2,500 mm/year, elevation from 100 to 300 meters above sea level (Rivera et al. 2013). Savannahs ecosystems were mostly covered by native grass species (e.g. Axonopus purpusii, Paspalum pectinatum) and less than 10% by shrubs and trees. Woody grasslands include exotic improved grass species (e.g. Braquiaria humidicola, Braquiaria decumbens) and 10-40% shrubs, woody vegetation and trees. Mixed agroecosystems contain improved grass species, crops (e.g. maize, rice, soy) shrubs, woody species and trees with a maximum of 40% tree and/or bush cover.	Savannahs Woody grasslands Mixed Agroecosystem
Oil palm fresh fruit bunches harvesting	Oil palm FFB are supplied by oil palm plantations land cover. They required a temperature higher than 24ºC, elevation between 200 and 600 meters above sea level, average precipitation between 2,000 and 4,000 mm per year, soils well drained, moderate organic content with pH between 4-7 (Corley and Tinker 2008; Owen and Eric 1995).	Oil palm plantations
Timber Harvesting	Forests are areas covered by trees which occupy more than 1 ha where trees grow more than 5 m tall and with canopy cover of at least 10% (UNFCCC 2002). Temperature between 23ºC and 30ºC, elevation from 100 until 1,100 meters above sea level and precipitation between 3,000 and 5,000 mm/year (Romero et al. 2004). Oils are tropical oxisoles, ultisoles forests acidic soils, well and poorly drained. Poor to moderate organic matter (Castro 2003).	Very dense forest Dense forest
Carbon sequestration	Carbon sequestration was defined as the uptake of carbon from atmospheric CO2 into a reservoir (e.g. trees, grasslands, soil biomass) in terrestrial ecosystems (Lal et al.	Savannahs, woody grasslands, mixed agroecosystem, oil palm

2013). Energy enters and flows through ecosystems when atmospheric CO2 is reduced to form organic carbon compounds during the process of photosynthesis and is lost when organic carbon is oxidized to produce CO2 after respiration (Chapin III et al. 2011).	plantations, very dense forest and dense forest

Accounting units

Ecosystem accounting units are spatial areas in which information is collected and assembled. Three different and related accounting units are used in the context of ecosystem accounting: basic spatial units (BSU), ecosystem units (EU) and ecosystem accounting units (EAU) consistent with the SEEA-EEA (United Nations et al. 2014b). EAU embrace large areas comprising administrative boundaries, natural managed areas and large scale natural features such as river basins. The scale at which EAU can be determined varies between national, sun-national, regional and local. EAU at national and subnational scale can contain different ecosystems reflected on different EU. The EU define spatial areas that satisfy certain characteristics such as land-cover type, water resources and altitude that can be considered as ecosystems, however it is recognized that ecosystems cannot be entirely spatially defined. BSU covers small areas, ideally formed by tessellations (e.g. 1 km^2) from overlaying a grid on a map of the territory, however it can be cadastral parcels and remote sensing pixels (United Nations et al. 2014b).

We used two EAU; (i) a large scale natural area embracing the geographical boundaries of the Orinoco river basin, (ii) protected areas for natural conservation including national parks and indigenous reserves. We used an ecosystems map from Instituto de Investigación de Recursos Naturales Alexander von Humboldt Colombia (IAvH) to define the geographical boundaries of Orinoco River basin EAU. We used national park and indigenous reserves maps from the Colombian National system of protected areas to define the boundaries of the protected areas EAU. Protected areas were defined as those areas where the main aim was natural resources conservation, regardless of their ownership (i.e. government, private) and management. Protected areas included national parks and indigenous reserves that are part of the Colombian National System of protected areas (SINAP) (Sandra 2005). Ownership and use are different for both areas, national parks are owned by the government and have strong restrictions in the use (i.e. extraction) of natural resources. Indigenous reserves are owned by indigenous groups, where the use of natural resources is limited but not restricted (Laborde 2007). Indigenous reserves have special regulations which allows the extraction of natural resources for subsistence, however large areas overlap with national parks where extraction and use are strongly restricted (Laborde 2007).

We used land cover categories from Moderate Resolution Imaging Spectroradiometer (MODIS) land cover type product MCD12Q1 from year 2003 and 2013 to define EU (Friedl et al. 2010). MODIS MCD12Q1 offers annually classified ready to use land cover information suitable for ecosystem accounting annual assessments. The MODIS land cover product include five land cover classification systems, we selected the International geosphere-Biosphere programme classification (IGBP) system which consists of 17 land cover types (Friedl et al. 2010; Loveland and Belward 1997). We reclassify the 17 land cover types into six EU (savannahs, woody grasslands, mixed agroecosystem, very dense, dense forests and oil palm plantations), each EU comprises one ecosystem and an aggregate amount of basic spatial units. The BSU was MCD12Q1 pixel size resampled at 1km^2.

Mapping and analysis

Satellite data

We used spectral information from the MODIS aqua and terra satellites for the measurement of ecosystem extent, condition and capacity to supply ES. We selected MODIS because it can be linked with the SEE-EEA measurement spatial structure which is based on accounting units. The large size of MODIS footprints facilitates the delineation of large EAU such as river basins and national administrative boundaries. MODIS pixel size at three spatial resolutions (i.e. 250m, 500m and 1000m) allows the collection and aggregation of information over large areas included in BSU. We downloaded 340 MODIS images from years 2003 and 2013 by the US Geological Survey website www.earthexplorer.usgs.gov. Images were geographically projected to Universal Transverse Mercator (UTM) WGS 19 and mosaicked using MRT software (DAAC 2011). Mosaicked images were clipped to match study area extent using ERDAS 2013 software and the ecosystems map from Instituto de Investigación de Recursos Naturales Alexander von Humboldt Colombia (IAvH) (Romero et al. 2004). Quality bands were used to check pixel quality over clipped mosaics.

MODIS products MOD13A3 and MOD17A3 were downloaded at high processing stages with high quality pixels (i.e., clear, low aerosol, cloud assessments). We used 48 images from MOD13A3 vegetation indexes monthly product (24 images per year, based on 2 footprints to cover the Orinoco river basin extent for 12 months). We selected the Enhanced Vegetation Index (EVI) from MOD13A3 because it provided information about vegetation canopy conditions, minimized canopy-soil variations and sensitive for canopy dense vegetation conditions (Huete et al. 2002). We used the normalized difference water index (NDWI) to estimate vegetation water content, it was derived from near infrared NIR (850nm) (band 2) and short wave infrared SWIR (2130nm) (band 7) according to Chen et al. (2005) algorithm and spectral information from MOD13A3. We excluded low quality pixels and we calculated the annual mean values for EVI and NDWI. We used 192 images from 8-days MOD11A2 land surface temperature product (LST) (96 images per year, based on 2 footprints to cover the Orinoco extent every 8-days for 12 months). We excluded low quality pixels and we calculate the annual mean LST. We used 4 images from MOD17A3 net primary production product yearly images (2 footprints per year). We used 96 images from 8-day MYD17A2 gross primary productivity product (2 footprints, 8-days, 12 months) for net photosynthesis in year 2013. We calculate monthly mean value of net photosynthesis.

Ecosystem extent

We assess decadal changes in EAU and EU extent by comparing two years, 2003 and 2013. Ecosystem extent was obtained from the number of pixels (BSU) in km² covering each EAU and EU. We used ArcGIS 10.1to map and measure the extent of both EAU, and the six EU. In the measurement of protected areas EAU, overlapping areas between indigenous reserves and national parks were treated as National parks and excluded from indigenous reserves.

Ecosystem condition

We considered relevant key characteristics of each ecosystem such as vegetation type, canopy water status, and day/night land surface temperature to assess ecosystem condition. We used the EVI to provide information about vegetation photosynthetic activity (PA) and vegetation type based on the relation between EVI and leaf area (Huete et al. 2002). We used NDWI to provide information about canopy water status (Chandrasekar et al. 2010; Gao 1996). We used LST to provide information about top canopy temperature (Wan 2008). Four multiband stacked images per year (2003 and 2013) were created containing each indicator (EVI, NDWI, day LST, night LST). Each image was overlapped with the EU map using ArcGIS 10.1to obtain multiband stacked images for 2003 and 2013 to monitor changes. The monthly variation was assessed by using EVI, NDWI monthly mean values of all EU in year 2013.

Ecosystems capacity to supply ES (grasslands grazed by cattle, harvested timber and oil palm bunches)

We examined if MODIS yearly accumulated NPP and aboveground NPP can be used to assess ecosystem capacity to supply ES. We used monthly accumulated NPP to provide information about the seasonal variation in NPP. Our study used NPP based on two principles. First, terrestrial primary productivity through plants photosynthesis provides energy and organic matter (carbon) to ecosystems. Plants respiration returns 50% (Waring and Running 2010) to 70% (Malhi et al. 2009) of the assimilated carbon. NPP is the remaining organic matter after plant respiration (Chapin III et al. 2011). Second NPP is distributed in the plant. NPP can be allocated to leaf, wood, fine root tissue and volatile organic compounds (Malhi et al. 2011a). NPP allocation can be grouped in aboveground (ABNPP) and belowground (BGNPP) (Clark et al. 2001a). We only considered ABNPP because all ES selected in our study were related to aboveground biomass harvesting. To calculate ABNPP we assumed that it was 50% of NPP for savannahs (Scurlock and Olson 2002) and 64% of NPP for tropical rainforest (Aragão et al. 2009b). For oil palm we assumed ABNPP was 96% of NPP (Corley and Tinker 2008), and 5% of the ABNPP was allocated to FFB biomass (Pulhin et al. 2015).

Ecosystems capacity to sequester carbon

We selected net ecosystem productivity (NEP) as an indicator for ecosystems capacity to sequester carbon (Schröter et al. 2014). While NPP accounts for the organic matter left after plant respiration once chemical and solar energy are converted to biomass, NEP includes both plant (autotroph) and heterotrophic respiration (Chapin III et al. 2011). To calculate NEP we use gross primary productivity GPP and ecosystem respiration (R_{eco}), where R_{eco} is a percentage of GPP (Luyssaert et al. 2007):

$$NEP = GPP - R_{eco} \tag{1}$$

For savannahs and woody grasslands the R_{eco} was 83% of GPP, where 53% is explained by plant ecosystem respiration (San José et al. 2014) and the additional 30% is cattle respiration (Lemaire et al. 2011). For mixed agroecosystem the R_{eco} was 63% of GPP, because 80% of the EU is covered by improved pastures (R_{eco} 53%) and 20% by crops (e.g. R_{eco} for rice 153-

43% (Saito et al. 2005) and R_{eco} for maize R_{eco} 73% (Verma et al. 2005). For forest and oil palm EU the R_{eco} was 87% of GPP (Malhi et al. 2011a).

2.3 Results

2.3.1 Ecosystem extent

Protected areas

The Orinoco river basin EAU extent covered 311,309 km^2 of which 61% were non-protected areas and 39% were covered by protected areas EAU (Fig 2.2).

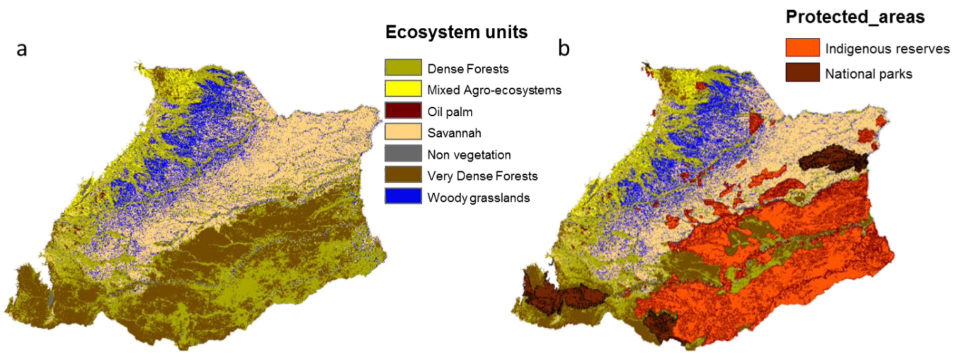

Fig 2.2 Maps (a) and (b) shows the extent of six ecosystem units in year 2003 (a) and 2013 (b), map (c) shows the extent of protected areas in 2013, and (d) shows changes per pixel (Basic Spatial Unit) between 2003-2013

We observed changes in EU extent between 2003 and 2013 in savannahs, woody grasslands, mixed agroecosystems and oil palm EU.

Table 2.2 Ecosystem extent in 2003 and 2013 in the Colombian Orinoco river basin EAU and protected areas Eau including national parks and indigenous reserves

EU	Orinoco river basin and protected areas EAU in 2003 and 2013							
	2003				2013			
	Orinoco river basin EAU			EU/Orinoco EAU (%)	Orinoco river basin EAU			EU/Orinoco EAU (%)
	Protected areas EAU		Non-protected(%)		Protected areas EAU		Non-protected (%)	
	National Parks (%)	Indigenous Reserves (%)			National Parks (%)	Indigenous Reserves (%)		
Savannahs	5	12	82	27	5	12	82	26
Woody grasslands	2	8	90	12	2	10	88	14
Mixed agroecosystem	7	14	79	2	5	12	83	3
Forest								
• Very dense	14	51	35	27 30	15	48	37	22 35
• Dense	8	40	52		9	44	47	
Oil Palm	0	0	100	0	0	0	100	0
Non-vegetation	0	10	90	1	0	9	90	1

We observed that in year 2003 41% of the Orinoco river basin EAU extent was covered by savannahs, woody grasslands, mixed agroecosystems and oil palm EU. The large part of savannahs (82%), woody grasslands (90%), mixed agroecosystems (72%) and all oil palm EU were located outside protected areas (Table 2.2). Most of the human economic activity is allowed and takes place in these four EU. We observed an increase in the extent covered by these four ecosystems in year 2013 in which they occupied 43% of the Orinoco river basin EAU. Woody savannahs increase 2% inside protected areas, however 4% of the mixed agroecosystem changed into non-protected areas. Most of the Orinoco river basin EAU (57%) was covered by forests, however 65% of the very dense forests and 48% of the dense forests were inside protected areas. Between 2003 and 2013 very dense forest extent decreased by 4,316 km^2 (5%) of which 796 km^2 were inside and 3,520 km^2 outside protected areas. Looking at differences among protected areas we found large differences between national parks and indigenous reserves. Very dense forest ecosystems inside national parks accounted for 27% of total protected areas, in contrast to 73% occupied by indigenous reserves (Fig 2.2). Indigenous reserves are key for the development of natural resources conservation strategies because they are strongly oriented on natural resources conservation covering a large part of EAU protected areas extent. Mapping specific areas of forests ecosystem reduction in non-protected areas can be used to decrease deforestation hotspots by focus natural resource extraction policies in specific areas.

Grasslands grazed by cattle from savannahs, mixed agroecosystem and woody grasslands ecosystems

Savannahs, woody grasslands and mixed agroecosystems together changed from 128,090 km^2 in 2003 to 131,309 km^2 in year 2013, increasing by 3,443 km^2.When looking at changes in extent, we observed that while savannahs EU decreased by (1%) 3,049 km^2, mixed agro-ecosystems and woody grasslands increased by (1%)1,190 km^2 and (2%) 5,302 km^2 respectively (Table 2.2). Human management and natural interventions (e.g. fires) could have transformed savannahs into woody grasslands (Table 2.3). Human management (e.g. fertilization, irrigation, ploughing genetic improved exotic varieties) have changed species composition in two ways, from native grasses to exotic improved varieties, and from grasslands to crops, shrubs, woody vegetation or trees (e.g. through abandoned land, agriculture frontier expansion). Woody grasslands and mixed agroecosystems increase their extent at expense of very dense and dense forests through deforestation to increase agriculture and livestock production (Table 3). Areas without vegetation increased at the expense of savannah EU, due to increased infrastructure, fallow land, bare soil, burned areas.

Fresh fruit bunches (FFB) harvested from Oil palm plantations

We found that oil palm EU extent doubled in 10 years, increasing from 393 km^2 in 2003 to 880 km^2 in 2013. Oil palm extent increased at the expense of very dense, dense forests and to lower extent of mixed agroecosystems and woody grasslands (Table 2.3).

Timber harvested from forests

Very dense and dense forest EU covered 56% of total EAU in 2013, together they were the largest EU in the Orinoco river basin (Table 2.2). We observed changes in extent. Very dense EU decreased, and dense forests EU increased. However, most of the changes were from very dense to dense forest and to lower extent to woody grasslands and mixed-agroecosystem (Table 2.3).

Table 2.3. Cross tabulate results showing changes in Ecosystem Units*

EU	Changes per EU in km² from 2003 to 2013							
				Forest				
	Savannahs	Woody grasslands	Mixed agroecosystem	Very dense	Dense	Oil palm	Non-vegetation	Total change
Savannahs*		-1,318	62	280	-786	0	-1,302	-3,064
Woody grasslands	1,318		577	9,791	-5,776	-8	149	6,051
Mixed agroecosystem	-62	-577		2,441	-664	46	5	1,189
Forest								
• Very	-280	-9,791	-2,441		-14,647	-258	461	-26,956
• Dense	786	5,776	664	14,647		-113	195	21,955
Oil Palm	0	8	-46	258	113		0	333
Non-vegetation	1,302	-149	-5	-461	-195	0		492

*note: the first row second column expresses that between 2003 and 2013 1,318 km² were converted from savannas to woody grasslands

Changes among both forests EU can be explained by changes in species composition and ecosystem conditions. A change in species composition can be the result of deforestation followed by new species transition with different canopy structure. The extent of both forests EU together in 2003 covered 179,307 km² and 174,991 km² in 2013 decreasing 4,316 km². According to Armenteras et al. (2013) 4,650 km² of forests were lost in Orinoco region between 2005-2010, which is slightly lower than our measurement. Changes from very dense forests to mixed agroecosystem can be seen as the result of human induced land use change (e.g. conversion of forests to pastures or croplands).

2.3.2 Ecosystem condition

Savannahs, mixed agroecosystem and woody grasslands ecosystems

We used NDWI to describe canopy water status. It is influenced by both natural conditions (e.g. vegetation type, vegetation species and climate) and management (e.g. irrigation). A low NDWI suggests longer dry periods (e.g. in savannah and woody grasslands) as well as an absence of irrigation (in mixed agroecosystems). We used LST to assess top canopy temperature. A low LST suggests a low non-vegetation background (i.e. soil) with higher surface temperature. A low LST in mixed agroecosystems suggests that the dominant vegetation has a lower canopy openness such as in the case of crops and improved grasslands. A high LST in savannahs and woody vegetation can be result of low ground cover (more soil exposed) and dominance of open canopy vegetation such as native grasslands in savannah EU.

Savannahs, woody grasslands and mixed agroecosystems share similar conditions such as land cover, vegetation type, canopy water status, and surface temperature. The provision of grass to graze cattle however, is limited by specific ecosystem conditions (e.g. soil type, grass species, elevation) (Table 2.1). Relevant factors are grass palatability, maturity and anti-nutritional factors (e.g. tannins). These conditions can constrain grazing even when the provision of grass is high. Remote sensing can be used to monitor biophysical ecosystem conditions such as vegetation type, photosynthetic activity (PA) and leaf area but is limited to monitor other conditions such as palatability. However, it could be examined if spectral information provided through remote sensing indexes (EVI, NDWI, LST) can be used to assess relevant ecosystem conditions such as species composition that would allow more in-depth mapping of the ecosystem's capacity to supply animal feed.

Oil palm plantations

Comparing 2003 and 2013 condition indicators (Table 2.4) it is noticeable that the EVI and NDWI differ strongly between the different ecosystems, but that there are broad similarities between the EVI and NDWI in both years. Only for oil palm there is a marked difference. Differences in EVI and NDWI may relate to climate conditions (drier years during El Niño), management (irrigation, fertilization) and age (young oil palms have a lower EVI and leaf area than mature palms). Given that these factors reflect the productivity of oil palm we postulate that remote sensing indicators such as EVI, NDWI, and LST can be used to assess leaf characteristics indicating the health of the plant and thereby the production of FFB. However, accurately measuring soil physical and chemical characteristics (e.g. nitrogen, phosphorus status) in dense oil palm plantations with remote sensing is not feasible (Mulla 2013).

Table 2.4. Ecosystem asset condition mean values in 2003 and 2013

EU	Remote sensing condition indicators*							
	2003				2013			
	EVI	NWDI	LST (in °C)		EVI	NDWI	LST (in °C)	
			day	Night			day	night
Savannahs	0.31	0.32	35	24	0.29	0.29	33	22
Woody grasslands	0.30	0.35	32	23	0.31	0.33	32	22
Mixed agroecosystem	0.47	0.56	30	23	0.47	0.55	30	22
Forest								
• Very dense	0.52	0.69	27	23	0.52	0.70	26	21
• Dense	0.44	0.57	29	22	0.44	0.59	27	21
Oil Palm	0.55	0.67	29	22	0.51	0.63	29	22

*Enhanced vegetation index (EVI), Normalized water index (NDWI) land surface temperature (LST)

Dense and very dense forests

EVI for very dense and dense forests EU did not change between 2003 and 2013. However, a higher NDWI and lower LST were observed in 2013. A higher EVI for very dense forests suggests differences in vegetation type (e.g. species, density), natural conditions (e.g. climate, slope, elevation, soil type) and human intervention (deforestation, management). A higher NDWI was observed in 2013 for both EU, which suggests higher water availability (e.g. El Niño phenomena). A low LST in very dense forests can be the result of larger canopy structure which decreased top canopy temperature. Timber production depends on forests ecosystem conditions such as vegetation type, species composition, age, precipitation, temperature, elevation, soil type, among others. remote sensing indicators (e.g., EVI, NDWI, LST) are suitable to assess forests conditions such as vegetation type, top canopy temperature, elevation and slope.

2.3.3 Monthly variation

High NDWI and low EVI were observed during the heavy raining period from April to August, when also floods are likely to occur (Lozano et al. 2013) (Fig 2.4). EVI minimum values were observed in the period between December and March for savannahs, and forests.

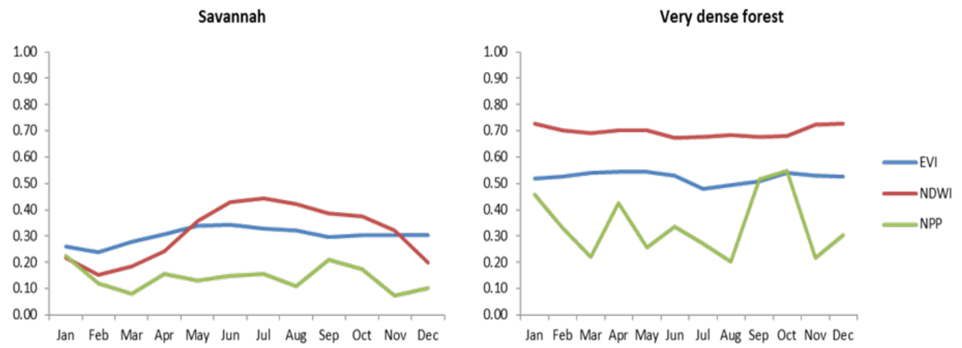

Fig 2.4 Monthly variation in ecosystem conditions such as water status and photosynthetic activity expressed by remote sensing indexes enhanced vegetation index (EVI) and normalized difference water index (NDWI), and ecosystem capacity by monthly NPP for two ecosystem units in year 2013

As soon as the rainy season starts around April-May, the growing season begins. EVI and NDWI increases, vegetation and water conditions change, changing canopy structure and PA, influencing vegetation regrowth especially for savannahs and to lesser extent in forests EU (Fig. 3). Vegetation and water conditions have a seasonal monthly variation. Remote sensing vegetation indexes such as EVI and NWDI are sensitive to spatial and temporal changes in photosynthetic activity, water status and canopy structure (Huete et al. 2002). These indexes are therefore suitable for the assessment of ecosystem conditions in terms of vegetation structure and composition among EU during the year. NWDI is sensitive to vegetation water content (Chen et al. 2005; Gao 1996), and suitable for monitoring changes in ecosystems water conditions (e.g. floods and droughts). Given that these indicators vary strongly throughout the year, this has important repercussions for measuring ecosystem condition with remote sensing in support of accounting. An option is to use remote sensing recordings of a specific month to estimate ecosystem condition for accounting. However, there is often some variability at the starting and ending of the rainy season, which means that there would be a significant difference in measured condition if in some years condition was measured just before and in other years just after the start of the rainy season. Therefore, when measuring condition with remote sensing images, it is recommendable to use annual time series of the various indicators (Reed et al. 2003).

2.3.4 Ecosystem assets capacity to supply ES

Savannahs, mixed agroecosystem and woody grasslands ecosystems capacity to supply grasslands grazed by cattle

Ecosystems capacity to supply ES is a function of ecosystems extent and condition. As explained in Section 3, we examine in this paper if and how NPP can be used as indicator for capacity. Table 2.5 presents the main findings in relation to NPP of the different EU. It is noticeable that the NPP of mixed agroecosystems (on a per km^2 basis) is 3 times larger than the NPP of savannahs and 2 times larger than that of woody grasslands. A higher NPP suggests a higher capacity to provide aboveground and belowground energy and organic matter. Ecosystems capacity to supply grass to cattle depends both on NPP and NPP allocation (low belowground NPP constrains the development of roots and the production of aboveground biomass). Hence, the annual total AGNPP provides an indication of the amount of grass that can be grazed on an annual basis by cattle. However, as discussed above, this does not consider effects such as palatability.

Table 2.5. Ecosystem capacity based on NPP

EU	Mean NPP and ABNPP			
	2003		2013	
	NPP (Gg C/km^2/year)	ABNPP (Gg C/year)	NPP	ABNPP
Savannahs	0.42	17,326	0.43	17,356
Woody grasslands	0.80	15,579	0.84	18,524
Mixed Agroecosystem	1.26	4,091	1.22	4,988
Forest				
• Very Dense	1.03	56,067	1.03	44,701
• Dense	0.98	45,368	1.01	68,208
Oil palm	1.43	17	1.47	62

Oil palm harvested bunches from Oil palms

Of all the EU, oil palm has the highest NPP (see Table 2.5). This is a function of, in particular, water status (e.g. irrigation and drainage) and nutrition (by fertilization). In young palms, NPP can be distributed belowground for root system development and aboveground to promote leaf area growth. The production of FFB in mature oil palms depends on NPP and NPP allocation, however most NPP is allocated to ABNPP. It is however not straightforward, in the case of oil palm, to relate NPP as measured with RS to the production of FFB. Total NPP allocated to the FFB can vary from 5% (Pulhin et al. 2015) to 58% (Lamade and Bouillet 2005; Melling et al. 2008) according to age, soil, nutrient availability, water and climate conditions. In addition, water stress or pests and diseases may disproportionally affect fruiting compared to leaf growth (Corley and Tinker 2008). Hence, further testing is required to find if a statistical significant correlation can be found between NPP and FFB production.

Timber harvested from forests

Very dense forest vegetation has a fuller canopy compared to dense forests. This is reflected in the NPP, which is higher in very dense forest (Table 2.5). The capacity of the forest to support timber harvesting depends on species composition, age distribution as well as NPP. The allocation of biomass to leaves, wood, steam and roots is also important. A large part of

NPP is generally allocated to stem biomass (Malhi et al. 2009). In addition, NPP also reflects forests regrowth after harvesting. Within a forest with a given age distribution (e.g. a natural forest or a plantation with a specific age distribution), NPP is therefore an adequate indicator of the mean annual increment (MAI), which is widely used as an indicator in forestry (Hasenauer et al. 2012). Changes in soil nutrient or water conditions are reflected in NPP and thereby in MAI. Different tree species will generally follow different strategies to allocate NPP (above/belowground, shoots/stem) in response to environmental conditions (Knapp et al. 2014b; Malhi et al. 2011a). Therefore the relation between NPP and capacity to generate timber is likely to vary spatially, as a function of spatial variability in species composition, as can be analysed using RS.

Carbon sequestration

For the period 2003 and 2013, information on soil respiration by EU was available from several studies (Malhi et al. 2009; Nakano et al. 2008; Wu et al. 2014). For Orinoco, we were therefore able to estimate the capacity of the ecosystem to sequester carbon. Note that in the case of this service, capacity and flow are usually assumed to be similar, because all carbon that the ecosystem is capable to sequester benefit people (independent of where sequestration takes place) (Bagstad et al. 2014; Schröter et al. 2014).

In total most carbon was sequestered in forests (Table 2.6). On per km^2 basis sequestration was higher in mixed agroecosystems and oil palm plantations which can be related to the use of fertilizers and irrigation systems. However, it needs to be considered that harvesting of crops and oil palm, and in particular replacement of oil palm after 20-30 years removes most of the carbon sequestered. This effect is not incorporated in table 6 (note that in our study area most oil palms are younger than 20 years old). Therefore carbon should be always analysed in terms of both sequestration and stocks, with forests having by far the highest stocks of carbon both in total and on per km^2 basis.

Table 2.6. Ecosystem capacity to sequester Carbon

EU	Gross primary productivity (GPP) and Net ecosystem productivity(NEP) (Gg C/yr)					
	2003			2013		
	GPP	NEP	NEP/km^2	GPP	NEP	NEP/km^2
Savannahs	77,474	13,171	0.16	75,279	12,797	0.16
Woody grasslands	57,929	7,531	0.19	67,272	8,745	0.20
Mixed agroecosystem	16,662	6,165	0.92	20,088	7,432	0.94
Forest						
• Very Dense	215,296	27,989	0.33	177,428	23,066	0.34
• Dense	205,254	26,683	0.28	247,046	32,116	0.30
Oil palm	1,416	184	0.38	2,630	342	0.39

2.4. Discussion

2.4.1 Remote sensing implications for ecosystem accounting

The measurement perspective in the standard system of environmental-economic accounting central framework (SEEA-CF) focuses on the physical measurement of individual environmental assets such as natural resources, cultivated biological resources and land compiled in terms of tons, hectares of land, and cubic meters of water (United Nations et al. 2014a). The measurement perspective in ecosystem accounting described in SEEA-EEA focus on ecosystems as functional systems that generate a set of ecosystem services (United Nations et al. 2014b). Ecosystem accounting moves from the physical measurement of individual environmental assets towards the spatial measurement of ecosystem assets through accounting units (United Nations et al. 2014b). However, moving from the physical to the spatial measurement of ecosystem assets requires a different spatial explicit approach and geo-referenced information which is not always available. Our study tested if remote sensing spatial information can be compiled in accounting units to measure (i) ecosystem extent, (ii) condition and (iii) ecosystem capacity to supply ES.

Ecosystem extent

The spatial measurement of ecosystem extent requires the spatial delineation of accounting units such as EAU, EU and BSU. Mapping EAU spatial extent is used to analyse interactions between human activities and ecosystems within large spatial areas such as countries, river basins and administrative boundaries. Our study included maps concerning two types of EAU, (i) a large scale natural area embracing the geographical boundaries of the Orinoco river basin, (ii) protected areas for natural conservation including national parks and indigenous reserves. We found that a large part of the Orinoco river basin extent is outside protected areas in which most economic activity takes place. Important human economic activities such as the supply of oil palm FFB and grass species to graze cattle are provided by ecosystems in non-protected areas. Large areas in the Orinoco river basin are used for natural resource conservation inside protected areas, mostly inside indigenous reserves. Our study included six different EU based on land cover ecosystem characteristics. Ecosystem accounting uses EU to determine spatial areas that share specific characteristics, however land cover can be seen as the dominant ecosystems characteristic used to determine EU. Remote sensing is a powerful tool to provide land cover information and therefore key for the spatial delineation of EU.

We found decadal changes in extent between different EU. Oil palm plantations increase their extent at expenses of other ecosystems, mainly forests. While savannahs extent decreased, mixed agro-ecosystems and woody grasslands increased. Large parts of the Orinoco river basin are covered by very dense and dense forests, however the extent of very dense forests decreased and the extent of dense forests increased. However, if we compared our results with other sources of information such national statistics we can find differences in extent. For instance, Fedepalma (2013) used net area by using plant density (number of plants per ha) to measure the area covered by oil palm plantations, which was 1,706 km^2 in 2012, our findings calculated 880 km^2 in 2013. Savannahs, woody grasslands and mixed agroecosystems together changed from 128,090 km^2 in 2003 to 131,309 km^2 in year 2013, increasing by 3,443 km^2. Other studies stated that 110,000 km^2 were covered by grasslands in year 2000 (Romero-Ruiz et al. 2012b) and 97,000 km^2 in 2008 (Benavides 2010). The extent of forests decreased between 2003 and 2013 by 4,316 km^2, however other studies found that 4,650 km^2 of forests were lost in Orinoco region between 2005-2010 (Armenteras et al. 2013). Differences in area extent can be caused by classification errors. In particular, oil palm can be classified as forest and young palm plantations can be classified as non-vegetation. Our results suggest that if only MODIS imagery is used spectral information needs to be combined with statistical, land cover and land use information to improve ecosystem extent assessments. An alternative is to use higher resolution images (potentially around 5 meter resolution is required to analyse oil palm plants with higher accuracy, given that mature oil palm crowns are generally around 8 to 9m in diameter), or to combine optical and radar images to enhance the classification. However, applying this at the national scale is challenging in terms of costs and handling data for countries the size of Colombia. Our findings complement those in Weber (2014) who mentions the potential use of earth observation systems as an essential tool to define land cover units for ecosystem accounting.

Ecosystem condition

The assessment of ecosystem condition is important in ecosystem accounting, it reflects the state of the ecosystem as well as its capacity to supply ecosystem services (United Nations et al. 2014a). The assessment of ecosystem condition entails the decomposition of ecosystems in relevant characteristics through indicators. Condition indicators should represent aspects relevant for ecosystem functioning and may reflect water, soil, vegetation, and biodiversity (DeFries et al. 2005; United Nations et al. 2014a). Indicators should be sensitive to changes in ecosystems integrity and functioning, and should provide structured quantifiable information to be compiled in national accounts. To date, the SEEA-EEA guidelines did not yet sufficiently consider data availability, how to deal with the different scales at which indicators may be appropriate, and the seasonal variation of ecosystem condition.

Several authors have highlighted that remote sensing can be used to analyse ecosystem condition (Ayanu et al. 2012a; Hein et al. 2015). Our study revealed two important insights. First, ecosystem conditions may vary strongly within the year (e.g. as a function of rainfall patterns and temperature variations). Ecosystem condition seasonal variation can be monitored by remote sensing using spectral information from different sensors, at different spatial, spectral and temporal resolutions (Andrew et al. 2014; DeFries et al. 2005; Kerr and Ostrovsky 2003). Our findings suggest that seasonal changes in condition indicators reflecting vegetation type, water status, canopy density, and surface temperature can be monitored with MODIS, for which images are available throughout the year. Remote sensing-derived indicators such as EVI, NDWI and LST are suitable to detect changes in the functioning and integrity of ecosystems. However, the assessment of ecosystem condition is not an easy task, and depends on the selection of relevant ecosystem characteristics linked to the supply of ES, the availability of remote sensing data with adequate spatial and temporal resolution, and insights in ecosystem functioning in order to translate remote sensing- derived indicators to ecosystem conditioning. In order to understand the effects of seasonal variation, images should be considered for each season, if feasible with a monthly or bi-monthly interval.

Second, if condition is key for the assessment of ecosystems capacity to supply ES then some indicators can be retrieved from remote sensing but others cannot. Remote sensing indicators such as EVI, NDWI, NPP use spectral information for the assessment of relevant ecosystem characteristics such as photosynthetic activity, vegetation type and structure and water status. However, many other ecosystems characteristics (e.g. soil fauna, nutrients availability, grasslands palatability) relevant for the supply of specific ES cannot be assessed by remote sensing. Palatability, for example, is influenced by among others the presence of anti-nutritional factors (e.g. tannins, alkaloids and poisonous compounds) in plant leaves, roots and steams (Campbell et al. 2014; Sollenberger and Burns 2001), which cannot easily be detected with remote sensing.

The integration of spectral information in structured national accounts can be challenging, nevertheless, remote sensing data can be combined with statistical information to assess ecosystems condition. Now that Sentinel satellite images will become available, there is a need to examine how these data can be used to support ecosystem accounting. A challenge is that not the whole globe (including part of the Orinoco basin) is as yet covered by Sentinel and that Sentinel needs to be combined with other images in order to analyse trends in ecosystems.

Ecosystem assets and capacity

Ecosystem assets capacity to supply ES relates to ecosystem's ability to generate multiple ES over time, as a function of ecosystem condition and extent (United Nations et al. 2014b). In turn, ecosystem condition and extent depend upon the naturally occurring vegetation in combination with how these ecosystems have been managed by people over time. Comparing ecosystem capacity and ecosystem service supply provides relevant insights in the sustainability of ecosystem use (Schröter et al. 2014). We assess ecosystem assets capacity to supply ES based on NPP as indicated by the remote sensing MODIS MOD17A3 product. Our study suggests that NPP is sensitive to changes in condition (e.g. canopy water status, vegetation type) and extent driven by land cover changes (e.g. from forests to mixed-agroecosystems).

NPP can be used to provide information about biomass re-generation patterns over time (e.g. grasslands accumulate biomass until they are grazed by cattle). Provisioning ES such as providing grass for cattle, providing timber that can be harvested and producing FFB therefore depend upon NPP, (as discussed in more detail in Section 2.3.4). NPP can be used as a proxy to assess the aggregation of aboveground NPP over time in grasslands, forests and oil palm plantations ecosystems, however more information is required in terms of the assessment of NPP allocation. Inter- and intra-annual variation in NPP allocation (ABNPP-BGNPP ratio) as a result of changes in ecosystem conditions adds complexity to NPP measurement (Knapp et al. 2014a). Furthermore, human management of ecosystems has a large influence on NPP allocation (e.g. irrigation and fertilization change NPP distribution from root systems towards leaf/fruit production).

New remote sensing developments increase the capability to spatially assess daily and yearly corrected NPP (Knapp et al. 2014b). Our study expands and complement recent ecosystem accounting studies (Remme et al. 2014a; Schröter et al. 2014) by introducing NPP as an indicator to assess ecosystem assets capacity to supply ES. Recognising the complexity of appropriately measuring NPP and dealing with variations in NPP, as well as the importance of a broad range of other ecosystem properties and processes that determine ecosystem capacity, our study illustrates that, in data poor contexts, remote-sensing-derived NPP can be used as a first proxy of capacity. Further work is required to validate NPP and capacity in a broad range of ecological contexts.

2.4.2 Uncertainty and accuracy

Two principal sources of uncertainty in ecosystem accounting relate to the physical measurement of ecosystem assets, and the assessment of ecosystem change (United Nations et al. 2014a). Uncertainty in the physical measurement of assets relates to data scarcity and scale (Schulp and Alkemade 2011). While data scarcity is related to information gaps in certain periods of time and lack of spatial data, scale is related to spatial resolution, aggregate levels, and extent. Moreover, combining data from different sources, at different scale and the spatial aggregation increases uncertainty and modelling errors (Remme et al. 2014a). Uncertainty in the assessment of ecosystem change relates to changes in ecosystem capacity to supply a future supply of ES. The occurrence of ecological thresholds, which may involve a sudden, fast, and sometimes irreversible change in ecosystem condition, adds to the uncertainties involved in measuring ecosystem capacity and ecosystem assets (Rockström et al. 2009; Scheffer et al. 2012). Remote sensing can be a valuable tool to reduce uncertainty in those two aspects by the provision of daily, monthly, yearly time series of spectral information to support the spatial measurement and the assessment of change in ecosystem assets (Hussain et al. 2013; Lunetta et al. 2006).

However, the use of remote sensing adds another source of uncertainty. Remote sensing is susceptible to errors associated to data acquisition, processing, analysis, conversion and final presentation, which decrease mapping accuracy and reliability (Congalton 1991; Shao and Wu 2008). During the last decade new remote sensing algorithms and techniques have been developed to reduce errors associated with data acquisition, processing and analysis. Up-to-date processed images can be obtained from MODIS and LANDSAT products which offers different levels of processing, from raw to very high processed products, such as vegetation indexes (Huete et al. 2002). Future and current missions, such as the Sentinel satellites, offer new opportunities (e.g. enhanced spatial and temporal resolution compared to MODIS or Landsat, and additional spectral resolution from radar). Misclassification is still one of the most common sources of error, however new classification techniques such as object-based (Blaschke 2010; Walter 2004) and texture-based classification (Liu and Fieguth 2012) have recently become available. In addition, data collection to validate accuracy measurement reports (Comber et al. 2012) can improve classification accuracy. Uncertainty in our study was related to EU classification and change assessment. To classify and distinguish between true changes (e.g. from savannahs to woody grasslands) and classification errors (e.g. woody grasslands were savannahs) more information (e.g. field observations, very high resolution images) are required to validate results.

2.4.3 Implications for natural resource management in Orinoco river basin

Ecosystem accounting provides a holistic overview of ecosystem assets (Latacz-Lohmann and Schilizzi 2014) and specific spatial explicit information to assess ecosystem assets and ecosystem uses (Edens and Hein 2013; Obst and Vardon 2014; United Nations et al. 2014a). Hence, ecosystem accounting can improve current land and ecosystems use strategies in the Orinoco river basin by providing essential information for policy making. In principle, the savannahs of the river basin are viewed as key for natural conservation strategies, but at the same time as an economic development opportunity related to use as pastures of, in the more humid northern part of the basin, for oil palm plantations. Current land and natural resources management policy identified three million ha which can potentially supply agricultural crops (rice, soy, oil palm, rubber and sugar cane), livestock and planted forests in savannahs ecosystems (CONPES 2014). Current policies recognize the importance of natural resource conservation, claiming that 54% of the Orinoco river basin is under special regulations concerning national parks and indigenous reserves (CONPES 2014). However, only 5% of the savannahs ecosystems are under conservation management as National Protected Areas (Armenteras et al. 2013). For very dense forests, a much larger part of the area is protected. Currently, around 15% of the very dense forest is protected as national parks and 48% as indigenous reserves. Nevertheless, as our study shows, the loss of very dense forests has been rapid, over 18% in the period 2003 to 2013. Expressed as percentage, deforestation in very dense forests has been approximately equal (i.e. around 18%) in national parks, indigenous reserves and non-protected areas (Table 2). This points to a lack of effectiveness of the protected area system that requires further validation and, if confirmed, correction.

2.5 Conclusion

Our study shows the suitability of remote sensing to support the spatial measurement of ecosystem assets for ecosystem accounting, specifically in data poor contexts where extensive field measurements would simply be too costly to populate the accounts. Our study examines how ecosystem assets can be measured with MODIS images in terms of extent, condition and capacity to supply ES, across the Colombian part of the Orinoco river basin. We found that ecosystem extent can be derived from MODIS (with 1km2 grid cells), and that several key condition and capacity related indicators can also be derived from MODIS. In particular, we used the enhanced vegetation index, normalized difference water index and land surface temperature that reflect key properties of the ecosystem including water availability and photosynthesis activity. We found that NPP is a key indicator for ecosystem capacity to supply ES, with relevance for the ecosystem services grass supply for grazing cattle, timber harvesting and, to a somewhat lower degree, FFB harvesting in oil palm plantations. However, in all these cases additional information is required to model ecosystem capacity, for instance grass palatability, species composition, and NPP allocation to fruits, respectively. The most promising application of remote sensing for accounting therefore, is to monitor changes in ecosystems over time, once the ecological properties such as palatability and species composition have been established for the different ecosystems. We expect that these ecosystem properties would change more slowly compared to the condition and capacity indicators that can be measured with remote sensing, and that they would need to be monitored at a much larger interval (say once in 10 years). Hence, an approach in which remote sensing is integrated in a data collection strategy is most likely to be optimal for ecosystem accounting.

Chapter 3

Assessing the capacity of ecosystems to supply ecosystem services using remote sensing and an ecosystem accounting approach

Abstract

Ecosystems contribute to economic development through the supply of ecosystem services such as food and fresh water. Information on ecosystems and their services is required to support policy making, but this information is not captured in economic statistics. Ecosystem accounting has been developed to integrate ecosystems and ecosystem services into national accounts. Ecosystem accounting includes the compilation of an ecosystem services supply and use account, which reflects actual flows of ecosystem services, and the ecosystem capacity account, which reflects the capacity of ecosystems to sustainably supply ecosystem services. A capacity assessment requires detailed data on ecosystem processes which are often not available over large scales. In this study, we examined how net primary productivity derived from remote sensing can be used as an indicator to assess changes in the capacity of ecosystems to supply services. We examine the spatial and temporal patterns in this capacity for the Orinoco river basin from 2001 to 2014. Specifically, we analyze the capacity of six types of ecosystems to supply timber, pastures for grazing cattle, oil palm fresh fruit bunches, and to sequester carbon. We compared ecosystem capacities with the level of ecosystem service supply to assess a sustainable use of ecosystems. Our study provides insights on how the capacity of ecosystems can be quantified using remote sensing data in the context of ecosystem accounting. Ecosystem capacity indicators indicate ecosystems change and harvesting-regeneration patterns which are important for the design and monitoring of sustainable management regimes for ecosystems.

This chapter is based on:

Vargas L, Willemen L, Hein L. Assessing the capacity of ecosystems to supply ecosystem services using remote sensing and an ecosystem accounting approach. (Accepted for publication in Environmental Management)

3.1 Introduction

Ecosystems provide a wide variety of ecosystem services essential for human survival, including the supply of food, the control of diseases, and the regulation of floods (Carpenter et al. 2009; De Groot et al. 2002). Nevertheless, ecosystems have been unsustainably changing for decades as a consequence of increasing economic activities such as agriculture and industry (Foley et al. 2005b; Steffen et al. 2015b).The design and implementation of policies aiming to decrease unsustainable changes of ecosystems are constrained by a lack of policy relevant information (Daily et al. 2009; Tallis and Polasky 2009). Particularly, because international economic monitoring systems that compile policy relevant economic information such as the System of National Accounts (SNA) do not include sufficient environmental information required to monitor changes in ecosystems (United Nations et al. 2009). International efforts to develop a monitoring system that integrates economic and environmental information led to the development of the System of Environmental Economic Accounting Central-Framework (SEEA-CF), as an international standard integrated monitoring system (United Nations et al. 2014a).

The SEEA-CF is complemented by the publication of the System of Environmental-Economic Accounting-Experimental Ecosystem Accounting (SEEA-EEA) to assess changes in ecosystems and the flow of ecosystem services, accounting for changes in stock and flows consistent with the SEEA-CF model (United Nations et al. 2015; United Nations et al. 2014a). A key innovation in ecosystem accounting is the inclusion and guidance on the spatially explicitly measurement of stocks by assessing changes ecosystems in terms of extent, condition and capacity to supply ecosystem services, and the measurement of flows of ecosystem services (United Nations et al. 2014b). Whereas extent reflects changes in ecosystem's size and location, condition reflects changes in its quality, and capacity reflect changes in the ability of an ecosystem to generate ecosystem services as a function of changes in extent and condition. In practical terms, two aspects can be distinguished from the occurrence of an ecosystem service; capacity and flow, where capacity is the long term ecosystem's potential to sustainably generate an ecosystem service, and flow is the actual use of the service (Schröter et al. 2014). A clear distinction between capacity and flow is important because the generation of some ecosystem services involves harvest-regeneration patterns and for some ecosystem the generation of ecosystem services can be above its capacity. This is important to assess the overall sustainability of the human activities in such ecosystem. The concept of capacity can be used to assess a sustainable use of ecosystems, as capacity reflects the ability of an ecosystem to sustainably supply a service under current ecosystem condition and uses at the highest yield or use level (Hein et al. 2016; United Nations et al. 2017). Here, the supply of ecosystem services is sustainable when the supply of an ecosystem service does not negatively affect the future supply of the same or other ecosystem services from that ecosystem. Current ecosystem condition means that the capacity is defined as it is now, neglecting alternative uses and independently from normative and historical baseline reference conditions (Hein et al. 2016). Therefore, by quantifying the capacity of an ecosystem to supply ecosystem services, the maximum amount of ecosystem services that can be supplied in a sustainable way is defined.

Different spatially explicit methods can be used to assess the capacity of an ecosystem to supply ecosystem services, including biophysical models, static land cover based look-up tables, remote sensing, and direct measurements (Bagstad et al. 2013; Schröter et al. 2015; Willemen et al. 2015). Direct measurements are desirable (e.g. by harvesting and measuring pasture biomass to assess grazing capacity) but unrealistic for large areas. Land cover, land use data, and ecosystem services biophysical models are combined using software modelling tools such as the Integrated Valuation of Ecosystem services and Trade-offs (InVEST)(Sharp et al. 2015) and the Artificial Intelligence for Ecosystem Services (ARIES)(Villa et al. 2014). In addition to these combination of methods, experts knowledge and statistic data are combined using a simple modelling tool as the matrix method (Burkhard et al. 2014; Burkhard et al. 2009). Because of requiring reliable and diverse input data, these modelling tools typically assess ecosystem services at one point in time. Satellite remote sensing has the ability to observe large areas offering an opportunity to overcome or complement extensive field surveys (Andrew et al. 2014; Crossman et al. 2012b). Remote sensing has increasingly been used to support ecosystem services assessments in the last decade, the repeating observations have been delivering often freely accessible spectral information to monitor key aspects of ecosystems including primary productivity, carbon, nitrogen, and water cycles (Andrew et al. 2014; Ayanu et al. 2012a). Remote sensing has the potential to cover large areas when direct measurements are not practically implementable and spatial and temporal data for biophysical models is lacking. Because ecosystem accounting requires environmental data, spatially explicit, repeatable and accessible, suitable to assess large areas in various accounting periods, this study explored the use of remote sensed data to assess the capacity of an area to sustainable deliver ecosystem services.

The objective of this study is to explore if and how remote sensing spectral information can be used to assess the capacity of ecosystems to supply ecosystem services following the ecosystem accounting guidelines. Specifically, we use Net Primary Productivity (NPP) information from Moderate Resolution Imaging Spectroradiometer (MODIS) combined with additional ecosystem services data to analyse changes in the capacity of ecosystems to supply ecosystem services over time and space in the Orinoco river basin. We selected six ecosystems; forest, oil palm plantations, grassland, savannah, woody savannah and mixed ecosystems that supply the following ecosystem services: oil palm fresh fruit bunches (FFB), timber, pastures for cattle grazing, and carbon sequestration. We analysed the spatial and temporal patterns in capacity, and we compared the capacity of these selected ecosystems with the supply of ecosystem services. We selected the Colombian side of the Orinoco River Basin as case study area because this area is one of the most pristine river basins of South America, while the river basin is at the same time witnessing fast degradation of ecosystems driven by economic development (Etter et al. 2010).

3.2 Methods

3.2.1 The Orinoco River Basin

The Orinoco is a transboundary river basin covering 655,000 km^2 in Venezuela and 345,000 km^2 in Colombia (Wolf et al. 1999), see Figure 3.1. Our study focuses on the Colombian part of the river basin covering the northern Andes mountains, the Guyana Shield, floodplains between the Orinoco and Amazon river basins, and high and low plains in the east. The basin is characterized by diverse ecosystems including páramos and cloud forests in the Andes mountains, natural savannah in the low plains, and Amazon tropical rainforests (Lasso et al. 2010). The average annual temperature varies from below 0ºC in the mountains to 38ºC in the eastern plains, while the annual precipitation varies from 4,000 mm on the eastern slopes of the cordillera to 1,500 mm in the eastern plains (Llanos) (Lasso et al. 2010; León 2005). The area is one of the most pristine river basins of South America but witnesses fast changes in many ecosystems, driven by economic development (Etter et al. 2010). Land cover and land use transformations occur with the introduction of crops (e.g. oil palm, rice, soy) and improved grass species allowing the intensification of livestock production (Benavides 2010).

3.2.2 Ecosystem accounting units

Ecosystem accounting uses three spatial units to organize information: ecosystem accounting areas (EAA), ecosystem assets (EA) and basic spatial units (BSU)(United Nations et al. 2017). EAA are large spatial areas defined by fixed and relative stable boundaries such as environmental management areas or administrative boundaries. In this study we define the EAA by the boundaries of the Colombian Orinoco River Basin as mapped by the Colombian Instituto de Investigación de Recursos Biológicos Alexander von Humboldt (Romero-Ruiz et al. 2012a). EA are spatial areas that form the conceptual base for accounting and where relevant statistics are integrated. The EA represent contiguous areas covering a specific type of ecosystem (e.g. forests, savannahs). Ecosystem accounting recommends the land cover classification presented in the SEEA-CF as a starting point to define EA (United Nations et al. 2014b).

In our study, we use the International Geosphere-Biosphere Programme classification (IGBP) land cover classification as a starting point to define EA. The IGBP classification is included in the MODIS MCD12Q1land cover product and describes 17 land cover types with an overall accuracy about 75% correctly classified (Friedl et al. 2010; Loveland and Belward 1997). We use the MODIS MCD12Q1 product because it provides annual land cover data that coincides with the standard length of the accounting period in ecosystem accounting: one year. To simplify the land cover map we merge the five IGBP forest classes (evergreen needle leaf, deciduous needle leaf, evergreen broadleaf, deciduous broadleaf, mixed forest) into one forest class, we merge closed and open shrubland into one class, and water and permanent wetlands into one class. Hence, we reduce the 17 IGBP land cover types to 11 merged land cover types (Fig 3a). The reclassification is needed as we are not able to differentiate ecosystem services supply between more than these 11 land cover classes, given the scale at which we work and the data availability for this area. Of the 11 grouped land cover types, only six land cover types (forest, grassland, savannah, woody savannah, natural mixed, and oil palm plantations) were relevant for the selected ecosystem services: oil palm FFB, grazing pastures for cattle, timber, and carbon sequestration. These ecosystem services are included in this study because of their importance for economic development as well as their implications for ecosystem change. We link the selected ecosystem services with the six EA: oil palm FFB are supplied by oil palm plantations, grazing pastures are supplied by four ecosystems: grassland, savannah, woody savannah and natural mixed ecosystem (which are pastures and trees), and timber is supplied by forest ecosystem (see Supplemental Materials for more details on ecosystem services and EA).

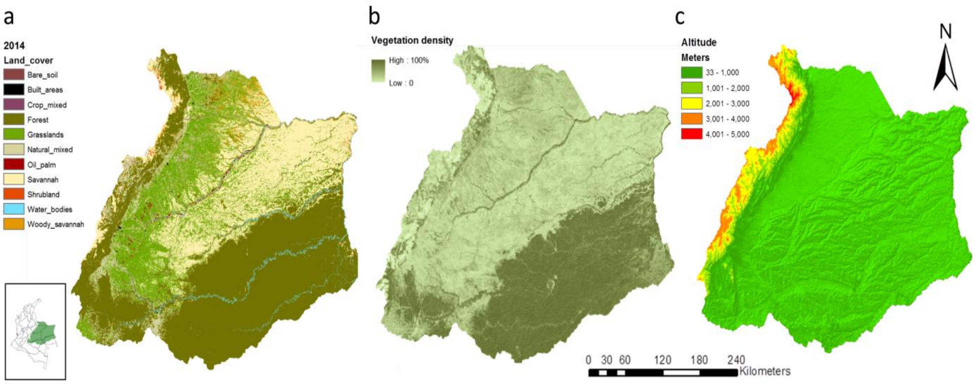

Fig. 3.1 Maps showing in a) the geographical boundaries of the Orinoco river basin in Colombia defining the EAA, and the EA based on the MODIS land cover product, b) Vegetation density based on MODIS (MOD44B), and c) Altitude based on digital elevation model (Global Multiresolution Terrain Elevation Data)

The accounting unit BSU is a small spatial area typically formed by grid tessellations (e.g. squares of 1 ha), like cadastral units or remote sensing pixels. In this study we define BSU by pixels from the MODIS land cover product which are 21.4 hectare in size (463.3 m by 463.3 m in the study area).

3.2.3 Assessing the capacity of ecosystems to supply ecosystem services

Net primary productivity as an indicator of ecosystem capacities

To assess the capacity of ecosystems to supply ecosystem services, appropriate indicators need to be selected and quantified. Such indicators should be sensitive to changes in ecosystem condition and extent, and reflect changes in the future generation of ecosystem services. Ecosystem functioning indicators such as Net Primary Productivity (NPP) have been used in earlier assessments of ecosystem change and ecosystem services supply (Costanza et al. 2007; van Oudenhoven et al. 2012). NPP is the net carbon gain by plants after respiration, including all new plants biomass, soluble organic compounds secreted into the environment, carbon transfers to microbes in the root systems, and volatile emissions from leaf tissues (Clark et al. 2001b). We selected NPP as an indicator of the capacity of ecosystems to supply ecosystem services because of two aspects. First, NPP is sensitive to changes in ecosystem condition, driven by abiotic (e.g. light, temperature, precipitation, evapotranspiration, nutrients) and biotic (vegetation structure, biodiversity, herbivorous consumption) factors (Knapp et al. 2014c). Second, all terrestrial ecosystems depends on NPP through plant photosynthesis to obtain energy and carbon, essential for the generation of ecosystem services (Chapin III et al. 2011).

MODIS as a data source to derive NPP

To assess the capacity of ecosystems to supply ecosystem services, spatially explicit information was needed. We used annual accumulated spatially explicit NPP derived from the

MODIS MOD17A3 for the time period between 2001 to 2014. MODIS (MOD17A3) provides high quality globally validated modelled NPP estimates based on the Monteith and Moss (1977) radiance use efficiency algorithm (Running and Zhao 2015). NPP depends on the amount of light reaching vegetation leaf tissue, called photosynthetically active radiation (PAR) and the capacity of vegetation to accumulate carbon to increase biomass (Knapp et al. 2014c). Hence, variations in NPP are the result of the PAR reaching the canopy, the amount that is absorbed (APAR), and conversion efficiency (Knapp et al. 2014c). Annual NPP in MODIS (MOD17A3) is calculated by subtracting maintenance and growth respiration costs for leaves, fine roots, and woody tissue from daily gross primary productivity (GPP), adjusted for different biomes (Running and Zhao 2015). Because NPP is the net carbon gain by plants stored in plant biomass tissue we refer to NPP as biomass accumulation in plants. The allocation of NPP between the different parts of the plant is not equal; NPP can be allocated aboveground or belowground.

The capacity of ecosystems to supply ecosystem services

The capacity of an ecosystem to supply an ecosystem service depends on the amount of aboveground biomass that is used to supply the ecosystem service (Fig 3.2).

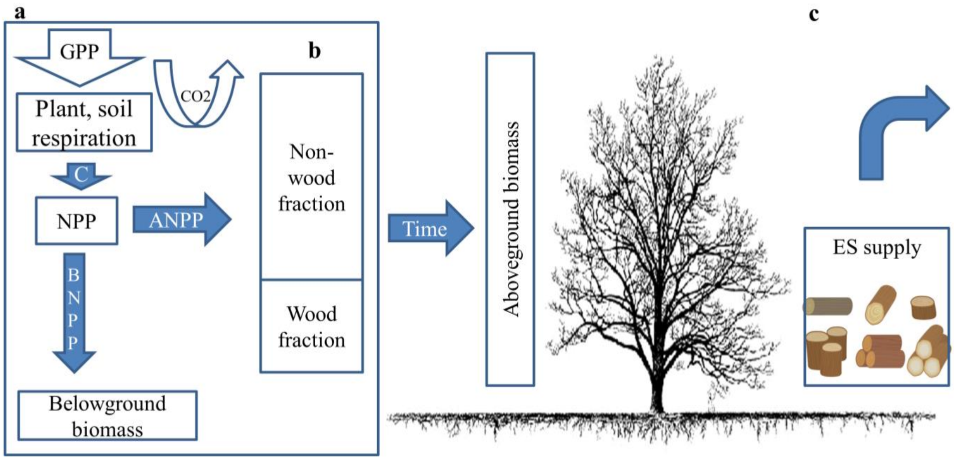

Fig 3.2 Schematic overview showing the capacity of ecosystems to supply ecosystem services, with timber as an example. a) indicates the gross primary productivity GPP is the source of the carbon available in forests ecosystems and NPP is the carbon available after plant and soil respiration which is allocated above (ANPP) and belowground (BNPP) as biomass,b) the amount of aboveground biomass that is used to supply timber including wood and non-wood fractions, andc) the accumulation of aboveground biomass of standing trees over time, and the supply of timber.

To assess the capacity of an ecosystem to supply ecosystem services *FANNP* we used equation 1. The parameters of the equation were based on NPP allocation models from different studies that simulate the distribution of NPP in above and belowground, and the distribution of aboveground NPP in different parts of plant tissues (Table 3.1). All models were applied for each EA per BSU, per year.

$$FANPPx_{i,y} = \beta \times ANPPx_{i,y} \quad (1)$$

$$ANPPx_{i,y} = \gamma \times NPPx_{i,y} \quad (2)$$

In equation 1, β is the part of the aboveground biomass that is used to supply each ecosystem service derived from literature (see Table 3.1), and $ANPPx_{i,y}$ is the annual supply of aboveground biomass at given ecosystem x, at location i, at year y. In equation 2, γ is the amount of aboveground biomass derived from literature (see Table 3.1), and $NPPx_{i,y}$ is MOD17A3 NPP at given ecosystem x, at location i, at year y. For example, the capacity of grassland ecosystem to supply pastures for grazing cattle in year 2014 at given BSU (with size of 21.4 ha) is the amount of aboveground biomass for grasslands (the NPP derived from MOD17A3 multiplied by 0.5), multiplied by 0.33, to specify the part of the aboveground biomass that is used to supply pastures for grazing. For each EA we calculated the arithmetic mean and the standard deviation of the capacity to supply ecosystem services, and we used time series and box plots to show annual fluctuations for 14 years between 2001 to 2014.

Table 3.1. Linking ecosystem services, the capacity of ecosystems to supply biomass and the fraction of biomass used to supply ecosystem services

Ecosystem service	The capacity of ecosystems to supply ecosystem services		
	Aboveground biomass (γ)	**Aboveground biomass that is used to supply ecosystem services (β)**	**References**
Pastures for grazing cattle	For grassland and savannah the γ is *0.5* For woody savannah the γ is *0.6* For natural mixed the γ is *0.7*	For grassland the β is 0.33 For savannah the β is 0.30 For woody savannah the β is 0.25 For natural mixed the β is 0.10	(Sarmiento and Pinillos 2001) (Hui and Jackson 2006) (Scurlock et al. 2002)
Timber	For tropical forests the γ is *0.8*	For tropical forests the β is 0.20	(Aragão et al. 2009a) (Malhi et al. 2011b)
Oil palm FFB	*F*or oil palm the γ is *0.96*	For oil palm FFB the β is 0.45	(Corley and Tinker 2008) (Kotowska et al. 2015)
Carbon sequestration	All biomass is relevant for carbon sequestration	$xNEP_{i,y} = xNPP_{i,y} - xHr_{i,y}$*	(Ott et al. 2015)

* To determine the capacity of ecosystems to sequester carbon and ecosystem services supply we use Net Ecosystem Production. Net Ecosystem Production (NEP) is defined as the net carbon gain after plant and heterotrophs respiration but excluding disturbances (e.g. fire) which are not common for the study area. $xNEP_{i,y}$ is net ecosystem production at each EA(x) (forest, grassland, savannah, woody savannah, natural mixed and oil palm), at year y. $xNPP_{i,y}$ is NPP from MODIS MOD17A3 at given ET x per BSU(i) in year(y).

2.3.4 Comparing ecosystem capacity and ecosystem services supply

To understand extraction-regeneration patters we compared the capacity of each EA to supply ecosystem services with the supply of ecosystem services between 2010 to 2014. While for regulating services (e.g. carbon sequestration) the capacity equals the supply of ecosystem services, for provisioning services, where biomass is extracted, the supply of ecosystem services can be lower, equal or higher than their capacity to supply biomass (Hein et al. 2016; Hein et al. 2015). To assess the balance between extraction of ecosystem services and regeneration of ecosystems we estimated the supply of ecosystem services based on equations and national statistics, and we compared this value with the capacity of each ecosystem to supply ecosystem services.

To estimate the supply of timber we used equation 3. We split forest ecosystems in upland and lowland forests because tree species harvested at mountain ecosystems are different than those harvested at low altitude. We used 1,500 meters above sea level as an altitude threshold to split upland and lowland forest, however, we recognize that tree species composition at this altitude is heterogeneous and this threshold simplifies the reality. In equation 3, St is timber supply in ecosystem f (upland or lowland) expressed in tons of timber at given year y. In equation 3, h is the amount of timber harvested (in m^3) per year y, multiplied by the average standing trees biomass (243 ton) of commonly harvested tree species in the study area, and the average wood density for tropical forest (0.6 ton/m^3) (FAO 2010; Ideam 2011a; Oliver 2013; Phillips et al. 2011).

$$St_{y,f} = h_y \times 243 \times 0.6 \tag{3}$$

To estimate the supply of pastures for grazing cattle we used equation 4. In equation 4, Sp is the supply of pastures to graze cattle at given grazing EA x at year y. Grazing ecosystems x are grassland, savannah, woody savannah and natural mixed ecosystem. In equation 4, φ is the annual cattle intake as estimated by Gaviria-Uribe et al. (2015), and c is the annual cattle stock per EA x based on statistics (Fedegan 2014) (see Supplementary Material).

$$Sp_{y,x} = \varphi \times c_x \tag{4}$$

To estimate the supply of oil palm FFB we used equation 5. In equation 5, So is the oil palm FFB supply, a is the annual FFB harvest multiplied by 0.56 which is the dry matter content in FFB (Contreras et al. 2012; Fedepalma 2013; Fedepalma 2015) (see Supplementary Materials for more details on the parameterization of the equations).

$$So = 0.56 \times a \tag{5}$$

For carbon sequestration, although capacity and supply are considered to be equal, removing biomass by harvesting timber, oil palm FFB and grazing pastures decrease the stock of carbon. We compared the estimated supply of timber, oil palm FFB, and pastures grazed by cattle (from equations 3,4,and 5), with the capacity to sequester carbon for each EA.

3.3 Results

3.3.1 Ecosystems supply of aboveground biomass

For each ecosystem, fluctuations in the supply of aboveground biomass fluctuated over time were assessed using the ANPP (Fig 3.3). The difference between the lowest ANPP in the year 2014 and the highest value in 2008 for grassland ecosystems was 1.9 ton of carbon per hectare on average. When looking at differences per hectare between grazing ecosystems we observed that the highest ANPP was in the natural mixed ecosystem and the lowest in savannahs (Fig 3.3a). For forests ecosystem, the annual ANPP increased 9.9 ton of carbon per hectare over the period from 2001 to 2014. Moreover, the annual ANPP in forest ecosystem fluctuated from 9.4(±2.6) ton of carbon per hectare in 2006 to 11(±2.6) ton of carbon per hectare in 2008 (Fig 3.3b).The annual ANPP in oil palm was higher in the year 2008 compared to 2010 (Fig 3.3b). Fluctuations in the ANPP reflect the sensitivity of NPP to changes in climatic conditions (e.g. rainfall pattern, light, and water availability). Moreover, changes in the ANPP can be related to climate phenomena such El Niño and La Niña which were particularly strong in 2008 and 2010, respectively (Ideam 2011b). These climatic phenomena increase water stress conditions stimulating the allocation of NPP to increase root system biomass in deep soil layers to overcome water shortages. The increase of root system biomass can decrease the availability of energy and carbon in other tissues such as leaves, diminishing photosynthetic activity.

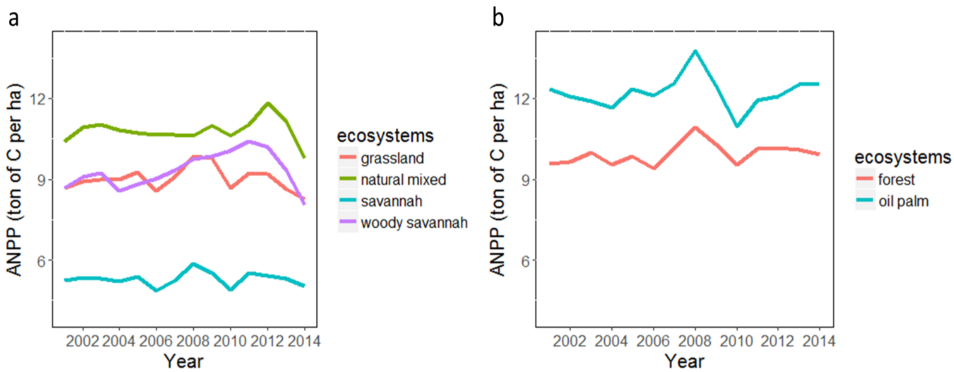

Fig 3.3 Time series showing annual fluctuations in the supply of aboveground biomass for each ecosystem. In a) the ANPP for grazing ecosystems; grassland, woody savannah, savannah and natural mixed ecosystem, and in b) ANPP for forest and oil palm plantations.

3.3.2 Ecosystems capacity to sequester carbon, pastures for grazing cattle, timber and oil palm FFB

Carbon sequestration

There is a clear variation in the capacity of ecosystems to sequester carbon between the ecosystems, defined by the land cover types. With an annual mean of 11 ton of carbon per hectare oil palm plantations had the highest capacity (compared to other ecosystems) to sequester carbon for the period 2001 to 2014. However carbon is released into the atmosphere at the end of the harvesting cycle of the plantation (usually after 25 to 30 years) when oil palms are cut and usually replanted. The biomass from felled oil palm trees will be mostly released into the atmosphere through the burning of felling residues to clear the fields for planting. The capacity of the forest to sequester carbon was 9 ton of carbon per hectare on average. This value was lower than the capacity to sequester carbon in the natural mixed ecosystem, equal to woody savannah and higher than grassland (8 ton of carbon per hectare) and savannah (Fig 3.4).

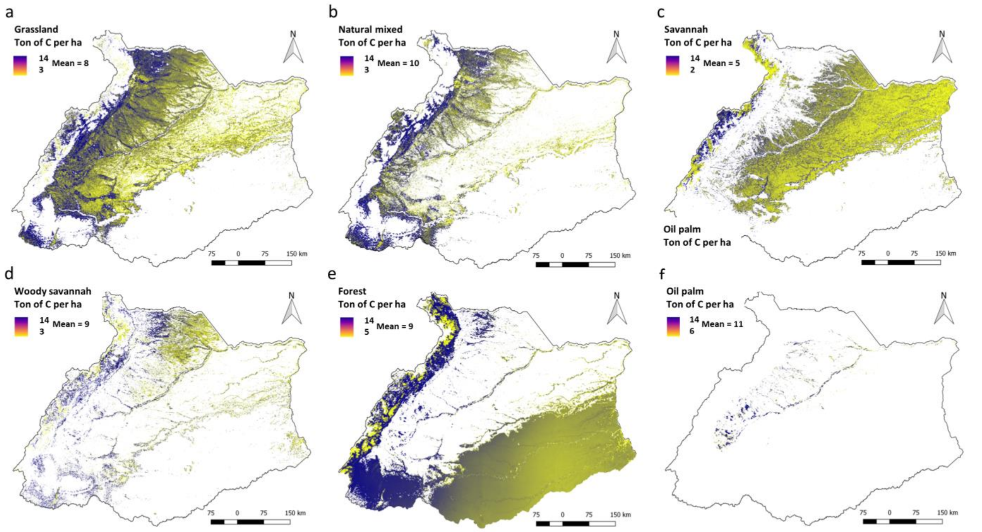

Fig 3.4 The capacity of ecosystems to sequester carbon based on NEP per year in a) grassland, b) natural mixed, c) savannah, d) woody savannah, e) forest, and f) oil palm, for the year 2014.

Adding more information about ecosystem's condition such as altitude and vegetation density provides insight into the spatial patterns in the capacity of ecosystems to sequester carbon. In Fig 3.1, we mapped the altitude and the vegetation density, and in figure 3.4 the spatial variation of the capacity to sequester carbon for forests, grasslands and natural mixed ecosystems. A high capacity was observed in the southwest of the river basin and at the eastern slopes of the Andes in locations with high tree vegetation density (Fig 3.1b). The capacity was low in the north-east plains in savannah ecosystem and in the high altitude forest ecosystem with low tree vegetation density (Fig 3.1).

Pastures for grazing cattle

The capacity to supply pastures for grazing cattle did not largely fluctuate over time among grazing ecosystems (Fig 3.4). Moreover, there were similarities between grasslands and woody savannah ecosystems, and between natural mixed and savannah ecosystems (Table 3.2). However, the NPP in woody savannah was higher compared to grassland, where most of the NPP was aboveground (Table 3.2). Because the non-grazed fractions (e.g. woody vegetation, trees) were higher in woody savannah ecosystem compared to grassland, the estimated biomass available to be grazed by cattle resulted in similar values for both ecosystems (Table 3.2). The capacity to supply pastures for grazing was similar for grasslands and woody savannah despite the NPP and the ANPP was higher for woody savannahs compared to grassland (Table 3.2). Likewise, the capacity to supply pastures for grazing was similar for natural mixed and savannah, despite that the NPP and the ANPP in natural mixed was more than twice the savannah ecosystem (Table 3.2).

Table 3.2 Net primary productivity, aboveground biomass and capacity to supply pastures for grazing cattle in ton per ha. (Mean and SD for the period 2001-2014)

Ecosystem	Primary productivity and aboveground biomass		Capacity
	NPP	ANPP	FANPP
Natural mixed	10.8±2.6	7.6±1.8	0.8±0.2
Grassland	9.0±2.5	4.5±1.3	1.4±0.4
Savannah	5.3±1.9	2.6±1.0	0.9±0.03
Woody savannah	9.3±3.0	5.6±1.8	1.4±0.05

We used a higher γ for natural mixed than for savannah ecosystem (0.7 vs 0.5 tons of carbon per ton of NPP), reflecting more trees in natural mixed compared to savannah ecosystem. However, in the natural mixed ecosystem, the capacity to supply pastures for grazing was lower compared to grassland and savannah (Fig 3.4), because a large part of the biomass in the ecosystem consists of trees not grazed by cattle. The combined effects of both aspects are shown in Fig 3.4.

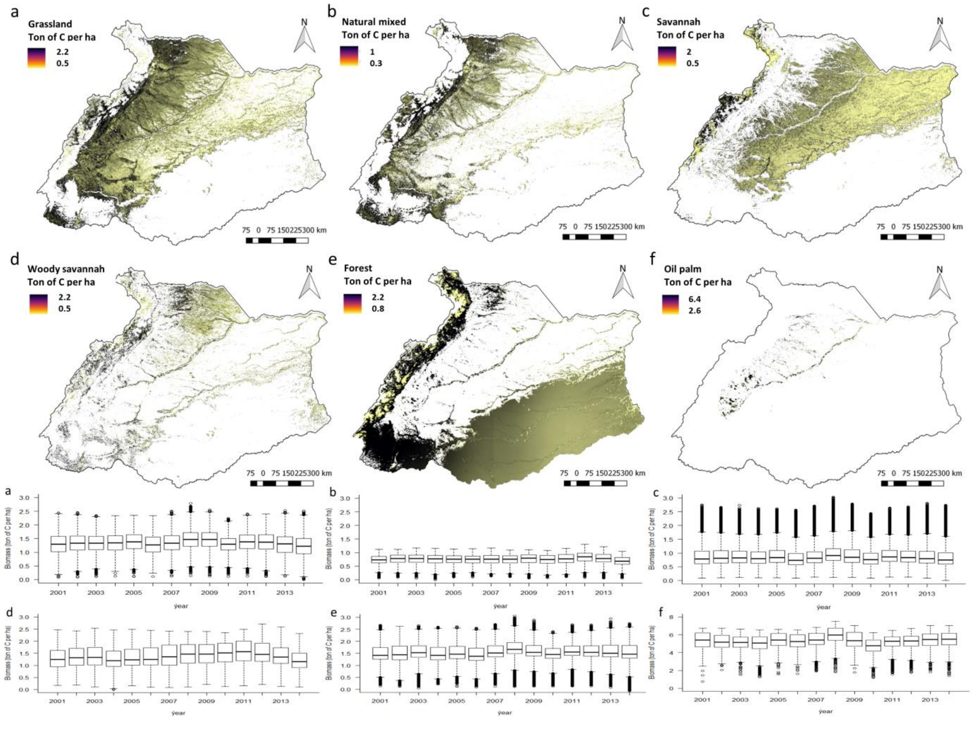

Fig 3.5 Maps showing the capacity to supply ecosystem services in a) grassland, b) natural mixed, c) savannah, d) woody savannah, e) forest and f) oil palm. Box plots showing capacity fluctuations between 2001 and 2014 for the same ecosystems.

The higher NPP in non-grazed fractions such as trees also indicates that the carbon stock is higher in natural mixed ecosystem compared to savannah and grassland (Table 3.2).

Timber

The forest ecosystem capacity to supply timber showed substantial spatial variation. Whereas in forests located on high altitudes the capacity to supply timber was below 0.8 ton of carbon per hectare, in forests located in the southwest corner of the river basin this capacity was generally above 2.2 ton per hectare (Fig 3.5). Moreover, in a large proportion of the forest ecosystem the average capacity to supply timber was 1.2 ton per hectare between 2010 and 2014, however, the annual capacity to supply timber was 1.5 ton of carbon per hectare in 2014 (Fig 3.5). Differences in ecosystem conditions (e.g. age, species, density, rainfall, altitude) influence forests photosynthetic activity, NPP allocation, and thereby the annual increment of harvestable timber. The spatial variation of these fraction followed different patterns related to differences in conditions such as altitude and tree density (Fig 3.1).

Oil Palm FFB

Oil palm plantations showed a non-homogeneous distribution of the capacity to supply FFB over time (Fig 3.5). On average, the capacity to supply FFB was 5 ton of carbon per hectare between 2001 and 2014, however, maximum values (7 ton of carbon per hectare) were observed in 2008 and minimum values (2 ton of carbon per hectare) in 2010 (Fig 3.5f). Differences in the capacity to supply FFB between years can be influenced by differences in the condition (e.g. rainfall conditions, age, disease occurrence) of the plantation.

3.3.3 Comparing ecosystems' capacities to ecosystem services supply

Carbon sequestration

In the case of carbon sequestration, the capacity to supply the service equals the supply of the service because in the conceptualisation of the SEEA EEA supply of regulating services does not involve an extraction or active use of the ecosystem and is therefore, in principle, always sustainable (Hein et al. 2016). Nevertheless, differences in the amount of biomass removed by harvesting influence the amount of carbon stored in each ecosystem over time. The amount of biomass removed from oil palm plantations was a bit higher (56%) compared with natural mixed (41%) and grassland (40%), higher compared with savannah (22%) and lowland forest (10%), and very high compared with woody savannah (6%) and upland forest (1%) (Table 3.3). Biomass in standing oil palms has a 25-30 years life span in which a substantial part of the NPP allocated aboveground is removed as FFB, and only a small part of the energy and carbon allocated belowground will remain for long periods of time. If carbon sequestration constitutes an avoided flow of carbon to the atmosphere and time is considered, then oil palm plantations will be the ecosystem with the lowest capacity. Forests, grassland and savannah ecosystem have a similar capacity to sequester carbon (in ton per hectare per year), however forests and woody savannah will provide more benefits as more carbon will be kept in the ecosystem for longer periods of time. Natural mixed, grassland and savannah ecosystem are intermediate as the biomass exported by these ecosystems depends on the amount of cattle raised by each ecosystem, the number of trees and the amount of vegetation not eaten by herbivorous.

Pastures for grazing cattle
Grassland and savannah ecosystem together provided 71% of the total capacity to graze cattle provided by all grazing ecosystems (Table 3.3), supporting 4.4 million cattle heads in 15 million hectares (for details on cattle stocks see Supplementary Material). However, grazing cattle removed 40% of the grassland capacity to supply pastures and 22% of the savannah (Table 3.3). Woody savannah provided 20% of the total capacity to graze cattle, supporting 243 thousand heads in 819 thousand hectares. Natural mixed systems provided 9% of the total capacity to graze cattle, supporting 660 thousand cattle heads in 2 million hectares (Table 3.3). However, grazing cattle removed 41% of the natural mixed capacity to supply pastures and 6% of the woody savannah. A substantial portion of the capacity to supply pastures is removed by grazing cattle in grassland (40%) and natural mixed (41%) ecosystems, because conditions (e.g. water availability, soil type, palatability) and management (e.g. infrastructure) in these two ecosystems are more favourable to graze cattle. In woody savannah most of its capacity is not removed by grazing cattle probably because the conditions in this ecosystem are less suitable for grazing (e.g. poor fertile soils, low nutritional quality grass species), and management is difficult (e.g. remote areas, flooded during raining season).

Timber
Whereas the capacity to supply timber was around 100 times the annual timber supplied in upland forests, this capacity was 10 times higher than the timber supplied in lowland forests (Table 3.3). Moreover, the forests (up and lowlands) capacity to supply timber was 10 times higher than the timber harvested. Yet, forest ecosystem covered 15 million hectares by 2014, but annually 38,000 hectares were deforested (Ideam 2015a). Accordingly, it can be considered that the annual capacity to supply timber was enough to cover the annual supply of timber. However, two additional location-specific issues need to be taken into account. First, most of the annual capacity to supply timber can be available but not suitable for harvest. Forests inside national parks (e.g. La Macarena, Tuparro) and indigenous reserves have a high capacity to supply timber, however, timber harvesting is forbidden by law in national parks and is controlled inside indigenous reserves. The area under national parks covers 1.5 million hectares, and inside indigenous reserves 8 million hectares (Correa et al. 2005). Moreover, this biomass can be inaccessible for timber harvesting, for example in forests located in pronounced slopes, flooded tropical rainforests, and remote forests with no access roads. Second, timber harvesting takes place in specific deforestation areas considered as hotspots, mostly in lowlands forests located in the southwest portion of the Orinoco river basin (Etter et al. 2006b). The capacity to supply timber may not be able to compensate the supply of timber in hotspot deforestation areas over long periods of time, putting the sustainability of these hotspot areas at risk. Our analysis shows that changes in land cover and land-use alter the future capacity of ecosystems to supply ecosystem services both in terms of types of ecosystem services and in a number of services that can be sustainably generated.

Oil palm FFB
Adult oil palms mobilize a large portion of their annual additions of carbon (>80% NPP) to produce FFB biomass. Although adult oil palms have a high capacity to supply FFB biomass (16 ton of carbon per hectare per year), the harvesting of FFB removes 56% of this capacity

(9 ton of carbon per hectare per year). However, in non-producing young oil palms (younger than 34 months) most of the biomass remains in the system as there is no biomass removal by FFB harvesting (not measured in this study). High rainfall leading to floods, long periods of drought, poor nutrient availability and diseases decrease oil palm photosynthetic activity and lead to lower allocation of NPP to fruit tissues, which reduces FFB supply.

Understanding the link between capacity and supply

In order to test the applicability of information on ecosystem capacity derived from remote sensing, a comparison has been made between capacity and flow of ecosystem services. In principle, provided the estimates are of sufficient accuracy, flows exceeding capacity indicate unsustainable use. Such an assessment cannot be made by comparing average values over ecosystem types but the comparison needs to be for individual BSU (on a pixel by pixel basis). We are currently testing the overall approach and cannot ensure sufficient accuracy on a BSU by BSU level. We therefore only compare spatial averages of flow and capacity, in order to get a first idea of the order of magnitude of the overall difference (Table 3.3). Hence, the information in Table 3.3 may conceal that overharvesting of ecosystem services is taking place at specific locations.

Table 3.3. Comparing average annual capacity of ecosystems to supply timber, oil palm FFB, and pastures with the annual ecosystem services use in 7 ecosystems

Ecosystem		Ecosystem service		Capacity		Ecosystem services use	
	Area (in ha)	Service	Unit	Ton/year	Ton/ha/year	Ton/year	Ton/ha/year*
Forest							
• Upland	2,124,561	Timber harvesting	Ton of timber	6,232,000	2.93	58,000	0.03
• Lowland	13,050,873			54,140,000	4.15	5,492,000	0.43
Natural mixed	2,319,961	Pasture grazing	Ton of pasture	5,598,000	2.41	2,289,000	1.01
Grassland	6,792,129		in dry matter	26,468,000	3.90	10,644,000	1.60
Savannah	8,654,258			19,842,000	2.29	4,422,000	0.52
Woody Savannah	835,576			13,234,000	15.84	829,000	1.01
Oil palm	107,154	FFB harvesting	Ton FFB in dry matter	1,704,000	15.90	954,000	9.07

*Deforestation takes place on 391 ha in upland and on 37,691 ha in lowlands forest each year (annual average between 2010 and 2014), however, to compare capacity with supply it was calculated from the area covered by each ecosystem in ton/ha/year.

For example, not all timber is available for logging such as timber in protected areas hence ideally harvest patterns and capacity should be compared at the level of the BSU (See also Schröter et al. (2014)). Yet, overall, it appears as if timber harvesting rates are well below the average annual increment of harvestable timber. At the other hand, it seems as if a significant portion (1.6 ton out of 4.0 ton) of grass in pastures is grazed by livestock. Note that in this case a 100% use rate of the capacity is not likely for instance because some of the palatable biomass is produced during periods of high supply (wet season) or in areas not accessible to cattle. In the case of oil palm, it is likely that most of the produced FFB biomass is harvested, although there may be losses of FFB due to for example diseases. In this case, the difference between crop and flow, therefore, may reflect inaccuracies in our capacity estimates, inaccuracies in harvest statistics as well as crop losses, or a combination thereof.

3.4. Discussion

3.4.1 Using remote sensed information in ecosystem accounting

Ecosystem accounting is an integrated framework developed to incorporate measures of ecosystems and ecosystem services into the structure of national accounts (Hein et al. 2015; Obst and Vardon 2014; United Nations et al. 2014b). Ecosystem accounting is spatially explicit approach and includes an assessment of the capacity of ecosystems to supply ecosystem services (United Nations et al. 2014b). However, such assessment requires spatial explicit information which is not always available. Accordingly, the SEEA-EEA noted that data scarcity especially at national and subnational levels is one of the main sources of uncertainty for the physical measurement of the capacity of ecosystems to supply ecosystem services (United Nations et al. 2014b). Our study explored the feasibility of compiling remote sensed spatially explicit information following ecosystem accounting guidelines to assess the capacity of ecosystems to supply ecosystem services at the level of a river basin. Our results show that remote sensed information can be used to determine accounting units following the ecosystem accounting guidelines. Ecosystem accounting needs land cover information as the starting point to determine the spatial distribution of ecosystems using spatial explicit accounting units (United Nations et al. 2014b). However, land cover maps at the national level are not regularly produced every year, hindering the assessment of annual changes in the spatial distribution of ecosystems by monitoring accounting units such as EA. The MODIS MCD12Q1 land cover product is annually classified based on training and test sites providing global cross-validated information with 76% overall accuracy among land cover classes (Friedl et al. 2010).We believe that this land cover product can be used to support the determination of ecosystem accounting units. Moreover, this product can be used as the starting point for the assessment of changes in ecosystems and ecosystem services. Other sources of spatially explicit information such as cadastral data, elevation, soil type and land cover maps, and aerial photography can be used to complement the MODIS land cover product to determine spatial units. Our results also show that remote sensed information can potentially be used to assess the capacity of ecosystems to supply ecosystem services following the ecosystem accounting guidelines. NPP is highly sensitive to changes in conditions such as rainfall pattern, water, and nutrients availability (Knapp et al. 2014c), making NPP a suitable indicator to assess the capacity of ecosystems to supply ecosystem services. The approach applied in this study was based on the dependence of biomass harvest on plant NPP and on NPP allocation. We found that each ecosystem has a different capacity to supply ecosystem services that varies in space and time. NPP can be linked with the supply of multiple ecosystem services, making the comparison of ecosystem services supply and capacity possible. The MODIS NPP product combines information from land cover, meteorology and vegetation index products, with their own uncertainty that can propagate and influence MODIS NPP information (Zhao et al. 2010). Uncertainties and validation of these products have been regularly assessed (Turner et al. 2006; Zhao et al. 2006b), and improvements on the MODIS NPP algorithm have been released (Running and Zhao 2015). However, NPP has a limited use for assessing ecosystem services not direct related with primary productivity such as hydrological and cultural services. Remote sensing products such as MODIS global evapotranspiration product MOD16, METEOSAT-8, NOAA, and Tropical Rainfall Measuring Mission (TRMM) can be used to support the assessment of hydrological ecosystem services (Carvalho-Santos et al. 2013). New missions such as SENTINEL, SAR and LiDAR sensors, and initiatives such as the Group on Earth Observations Biodiversity Observation Network (GEOBON) offer new opportunities to support the assessment of ecosystems and their services and to remotely monitor change in ecosystems, ecosystem services, and biodiversity (Tallis et al. 2012).

3.4.2 Can capacity-supply models be used to analyze sustainability?

Ecosystem accounting focuses on the assessment of ecosystems and their services providing integrated information required to assess environmental sustainability (United Nations et al. 2014b). The supply of ecosystem services involves the extraction and harvest of resources. Harvest and regrowth rates in ecosystems determine the sustainability of ecosystem use (United Nations et al. 2014b). A first step towards the analysis of sustainability in ecosystems has been the use of spatial models that integrate the capacity of ecosystems to supply services and the supply of ecosystem services (Burkhard et al. 2012; Schröter et al. 2014; Villamagna et al. 2013). These models showed that the harvest of ecosystem services can exceed the capacity of ecosystems to supply the service putting ecosystems sustainability at risk. However, the assessment of the capacity of ecosystems to supply ecosystem services is challenging because of the ecosystem's extent and condition (e.g. water, nutrients, temperature, rainfall) change in time and space. We use NPP to estimate the capacity of ecosystems to supply ecosystem services and compare this with average use of ecosystem services (note that in the conceptual framework of the SEEA EEA, the use of an ecosystem service equals, by definition, the supply of the ecosystem service). For provisioning ecosystem services, the supply can be lower, equal or higher than the capacity of the ecosystem to supply ecosystem services. The spatial variation is an important aspect to consider in the assessment of the sustainability of ecosystems at large scale. We showed that the forest capacity to supply biomass at the scale of the whole river basin exceeds the amount of timber biomass harvested. Timber harvests take place in dedicated forest patches, where in the year of harvest, extraction exceeds regrowth. Our study indicates that at the level of the basin there is no overharvesting of timber. However, not all forests are subject to timber harvesting, for example, because they are protected or inaccessible. In the future, our analysis can be refined by comparing regrowth and extraction rates in areas that are harvested (see e.g. Schröter et al. (2014)). The harvested area can be derived from forest concessions (as well as from remote sensing images if the annual imagery of sufficient resolution ≤30m for a time period of at least one logging cycle is available). An additional future refinement is that the assessment of the overall sustainability of ecosystems by capacity-supply mapping models should consider the inter-annual variation of the capacity of ecosystems to supply ecosystem services. As we show, there can be substantial variation in biomass regrowth between years, for instance, due to different weather patterns. Annual budgets simplify the capacity of ecosystems–supply dynamics, climate events such as droughts during El Niño influence ecosystem services supply by altering the capacity of ecosystems to supply ecosystem services at specific locations. Such refinements can enhance the accuracy and thereby the applicability of our approach. Potentially, this could lead to an efficient way of measuring the sustainability of ecosystem use by comparing local patterns in regrowth and extraction rates. Where this measurement system can be embedded in the SEEA Ecosystem accounts, extraction rates for provisioning services can be linked to effects on regulating and cultural services (see Hein et al. (2016) for a potential way forward on linking ecosystem use to capacity for different types of services).

3.4.3 Implications for SEEA-EEA

The SEEA-EEA considers the spatial assessment of the capacity of ecosystems to supply multiple ecosystem services as central to understand how human activities change ecosystems and how these changes are related to the future generation of ecosystem services. Specifically, this concept helps to define ecosystem use patterns, to develop and evaluate alternative use scenarios, and to assess ecosystem degradation (Edens and Hein 2013; United Nations et al. 2014b). However, the assessment of the capacity of ecosystems to supply services is challenging because ecosystems are complex dynamic systems influenced by many factors (e.g. changes in soil pH, water availability, climate, light). Changes in land use for instance, by switching from forest to agriculture modify ecosystem conditions (e.g. by polluting downstream waters), and the capacity of ecosystems to supply fresh water, compromising the supply of fresh water in the future. Ecosystem capacity indicators should be able to spatially reflect changes in ecosystem condition in space and time, and the implications in the future ecosystem services supply. In our study, we explored if NPP can be used as an indicator for the assessment of the capacity of ecosystems to supply ecosystem services (timber and FFB harvesting, carbon sequestration and pastures for grazing cattle).

Our study included MODIS NPP to assess capacity by assessing the amount of aboveground biomass that is used to supply an ecosystem service. Whereas assessing the supply of aboveground biomass give us insights about ecosystem regeneration patterns, NPP allocation is key to link the supply of aboveground biomass with a specific ecosystem service. However, the assessment of NPP allocation is challenging as ecosystems are dynamic systems where the amount of aboveground biomass allocated to supply an ecosystem services can change in time and space. We used different NPP allocation models to assess the amount of aboveground biomass that is used to supply ecosystem services. However, the information provided by these models was not specific for the Orinoco river basin, it was adjusted from different countries such as Malaysia for oil palm, and China, Uruguay, and Venezuela for grazing pastures. Ecosystem conditions are clearly different in the locations where these models were developed increasing uncertainty to our results. MODIS NPP is a powerful tool to assess the spatial variation of the capacity of ecosystems to supply ecosystem services in large areas, however, can be limited in contexts where a high level of detail is required such as municipality and local level, e.g. in Remme et al. (2015) and Villamagna et al. (2013). However, the moderate spatial resolution of this sensor can be compensated with the dimensions of its swath that covers 2.3 by 10 km per scene every day, the temporal resolution where products are available every 8-16 days, month and year processed from level 2 up to level 4 where products are modelled and produced at high quality. New developments in remote sensing can play an important role in the further development of SEEA-EEA ecosystem accounting by providing information to assess harvesting-regeneration patterns, monitor ecosystem change, and to assess the future generation of ecosystem services. New opportunities by combining remote sensing with economic and social information can be useful for the assessment of current and future ecosystems use, alternatives scenarios, and ecosystems degradation towards sustainable use of ecosystems.

3.5 Conclusion

There is a growing interest in ecosystem accounting to support the protection of ecosystems and the future supply of ecosystem services. Ecosystem accounting was developed as an experimental system towards the integration of environment and economic information into the system of national accounts. Ecosystem accounting includes the assessment of ecosystems' capacity to supply multiple ecosystem services (United Nations et al. 2014b). Our study provided insights on (i) how the capacity of ecosystems to supply ecosystem services can be assessed and (ii) how remote sensing can be used to support this assessment in large areas. In our study, we proposed NPP as a suitable indicator to assess the capacity of ecosystems to supply timber, pastures for grazing cattle, oil palm FFB, and to sequester carbon because NPP is sensitive to changes in ecosystem condition and changes in primary productivity that affect the supply of ecosystem services. However, more research on NPP allocation is required to improve current knowledge, on mapping the capacity of ecosystems to supply ecosystem services and ecosystem services supply. Annual land cover information from MODIS MCD12Q1 can be a potential source of information to assess land cover changes in line with the annual periodicity of ecosystem accounting, in particular for large, relatively homogeneous ecosystems as found in the Orinoco river basin. Our study explored MODIS primary productivity to provide spatial information for the assessment of the capacity of ecosystems. We found that MODIS NPP can be a powerful source of spatial information to assess the capacity of ecosystems at river basin scale such as the Colombian Orinoco. However, NPP is most relevant for provisioning and selected regulating services, much less so for cultural services. New developments in earth observation (higher spatial temporal resolution, new sensors) will complement currently available datasets for ecosystem assessments and for the integration of environment and economic information. The presented approach used for the assessment of the capacity of ecosystems to supply ecosystem services is a basis for further refinements that will allow developing capacity-supply models for ecosystem accounting and other applications.

Chapter 4

Linking planetary boundaries and ecosystem accounting, with an illustration for the Colombian Orinoco river basin

Abstract

Economic development has increased pressures on natural resources during the last decades. The concept of planetary boundaries has been developed to propose limits on human activities based on earth processes and ecological thresholds. However, this concept was not developed to downscale planetary boundaries to sub-global level. The absence of boundaries at sub-global levels constrains the use of the concept in environmental governance and natural resource management, because decisions are typically taken at these levels. Decisions at national level are currently supported, among others, by statistical frameworks in particular the System of National Accounts. However, statistical frameworks were not developed to compile environmental information, hindering environmental decision making. Our study examines if and how ecosystem accounting can be used in combination with the concept of planetary boundaries in guiding human activities at the level of a river basin. We assess the applicability of both frameworks for natural resource management in the Orinoco river basin, based on adaptive management components. Our analysis indicates that differences in the purpose of analysis, information provided, and methods constrain the potential integration of both frameworks. Nevertheless, the way both frameworks conceptualize the social system and the interactions between social and ecological systems can facilitate translating planetary boundaries into indicators considered in ecosystem accounting. The information recorded in national ecosystem accounts can support establishing ecological thresholds and, more fundamentally, to relate ecological thresholds to human impacts on ecosystem condition. Capitalizing on these synergies requires further exchange of experiences between the communities working on ecosystem accounting and planetary boundaries.

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4.1. Introduction

Economic growth has progressively increased human pressures on the earth system (Foley et al. 2005a; Folke 2010; Steffen et al. 2011). Human pressures on the earth system have led to, among others, the modification of nitrogen, phosphorus and water cycles, and changes in land cover and ecosystems (Carpenter 2005; Foley et al. 2005b; MA 2003). The discussion on how to best reconcile economic development with sustainable natural resource management is still ongoing, but is reinforced by the increasing pressure on relatively undisturbed ecosystems in developing countries. Sustainable development is challenged by the complexity of the environmental problems derived from human and nature interactions. Complex environmental problems such as climate change and ocean acidification, cannot be fully understood by separate disciplinary approaches, they demand integrative, multidisciplinary approaches (Liu et al. 2015; Ostrom 2009). Integrated approaches view human and nature as connected entities embedded in socio-ecological systems, interacting at multiple organizational (e.g. administrative arrangements), spatial (e.g. river basin) and temporal scales (e.g. years) (Berkes et al. 2008; Liu et al. 2007; Ostrom 2009). Liu et al. (2015) highlighted the development of quantitative frameworks as a significant contribution to better understand complex environmental problems in socio-ecological systems by assessing the connections between the socio-economic and the environmental components of the system. Integrated approaches such as used in the Intergovernmental Panel on Climate Change assessments and the Millennium Ecosystem Assessment, incorporate quantitative frameworks to explore the links between global environmental and social-economic changes (MA 2003). However, few approaches incorporate quantitative frameworks to propose limits on economic activities within earth system functioning to reconcile economic development with sustainable natural resource management. Two complimentary integrated approaches aiming to reconcile economic development with sustainable natural resource management through quantitative frameworks are the planetary boundaries framework and ecosystem accounting.

The planetary boundaries framework identifies nine priority earth system processes; the nitrogen, phosphorus, carbon and water cycles, climate, stratosphere, land and ocean systems, biodiversity, aerosol loading and chemical pollution (Rockström et al. 2009). The framework presents, for each of these earth system processes, quantified boundary levels that are associated to ecological thresholds. Crossing these thresholds would generate unacceptable environmental change (Rockström et al. 2009). The framework distinguishes between boundaries associated to continental or global thresholds, such as stratospheric ozone depletion, and boundaries based on processes with no evidence of planetary threshold behavior, such as water use. Boundaries associated to earth system processes with no evidence of planetary behavior are not quantified in the framework. However, because earth system comprises smaller scale, spatially connected, interacting systems, crossing ecological thresholds in these small scale systems can propagate and cause a shift in the whole system (Barnosky et al. 2012; Peters et al. 2009). The increasing awareness on irreversible changes in the earth functioning triggered by crossing ecological thresholds increase the influence of planetary boundaries in international discourses of global environmental governance and global sustainable development (e.g. United Nations 2030 Sustainable Development Agenda) (Griggs et al. 2013). However, the implications of using planetary boundaries in global environmental governance are challenged by uncertainties in the boundaries associated to unknown ecological thresholds arising at sub-global level, and by multilevel governance (Galaz et al. 2012). Decisions concerning environmental governance and natural resources management are mostly taken at national and sub-national level, requiring multilevel governance between institutions, policies and social organizations (Galaz et al. 2012; Häyhä et al. 2016; Nilsson and Persson 2012).

Ecosystem accounting has been developed under auspices of the United Nations Statistical Commission, synthetized in the System of Environmental Economic Accounting Experimental Ecosystem Accounting (SEEA-EEA) (United Nations et al. 2014b). Ecosystem accounting complements the international statistical standard for environmental economic accounting, the System of Environmental Economic accounting-2012 Central Framework (SEEA-Central Framework) (United Nations et al. 2014a). Ecosystem accounting organizes spatially explicit biophysical and monetary data in a set of tables and accounts in which different aspects of ecosystems, and flows of ecosystem services are quantified and linked to economic activities. Information compiled in tables and accounts can be reported at national and sub-national level, following the same concepts, definitions and accounting rules synthetized in the SEEA-CF and the System of National Accounts (United Nations et al. 2014b). Recent studies demonstrate the potential use of ecosystem accounting information to support decision and policy making on land and resource management (e.g. water purification in Europe, and monetary valuation of ecosystem services in The Netherlands (La Notte et al. 2017; Remme et al. 2015)).

The planetary boundaries and ecosystem accounting frameworks have a merit in supporting decision and policy making in natural resource management to achieve a more sustainable development, however both frameworks have their own limitations. The planetary boundaries framework was not designed to be applied at national and sub-national levels, hindering the ability to influence decision making at these levels. Ecosystem accounting can record a wide range of data sources (e.g. remote sensing and statistical information) at national and sub-national level including ecological thresholds as indicators, however there have as yet not been any ecosystem accounts that include such thresholds. Hence, there is a need to explore if both frameworks can be reconciled, and to assess if experiences from both frameworks can be used to mutually reinforce one another.

The aim of this paper is to examine if and how planetary boundaries can be used in combination with ecosystem accounting in proposing limits on human activities at the level of a large river basin. In particular, this paper will compare and contrast the planetary boundaries and ecosystem accounting frameworks, and illustrate if looking for similarities and differences provides complementary information for sustainable natural resource management in the Colombian Orinoco river basin. The Orinoco river basin is selected because of rapid ecosystem change ongoing in this area. Human activities such as oil palm and energy generation have been growing over the last two decades thereby increasing the pressure on large areas of relatively undisturbed tropical forests. The area is also subject to increasing withdrawal of water resources for irrigation and hydropower and the eutrophication of lakes and rivers.

4.2. Method

4.2.1 Comparing frameworks

We compare the planetary boundaries and ecosystem accounting frameworks using two sets of criteria based on Binder et al. (2013). First we provide a general overview and discuss the frameworks based on contextual criteria, and second we provide an in-depth comparison based on structural criteria. Contextual criteria provide information about the origin of the framework, the purpose, information provided, use in policy making, and spatial and temporal domains (Table 1). Structural criteria describe how the social and ecological systems are conceptualized, how they interact, and their analytical depth. The conceptualization of the social system includes the analysis of different hierarchical levels (e.g. individual, groups), how these levels interact, and how these levels are integrated. The conceptualization of the ecological system includes a (i) description on how the system is overviewed from an anthropocentric or eco-centric perspective, and (ii) how the ecological dynamics are described, (e.g. using natural language or specific equations). The conceptualization of the interaction between both systems is described using three forms of interaction; (i) if the ecological system influences the social system, (ii) if the social system influences the ecological system and (iii), if reciprocity between both systems is considered (Binder et al. 2013) (Table 4.1). And lastly we classify the frameworks as either analysis-oriented or action-oriented. Whereas the goal in analysis oriented frameworks is to provide a general language to formulate and approach different research questions, the goal in action oriented frameworks is to act or intervene.

Table 4.1. Criteria used to compare the PB and EA frameworks

Contextual criteria	**Question**
Disciplinary origin	What discipline provides the starting point of the framework?
Theoretical origin	Which theories are the foundations of the framework? What is the motivation for applying these theories?
Purpose of analysis	What is the aim of the framework? What type of information is provided? How is the framework used in policy making?
Temporal domain	At what temporal level is the framework applied (years, months, days) ?
Spatial domain	At what spatial level is the framework applied (global, national, sub-national)?
Structural criteria	**Question**
Social system	
• Name	How is the social system described in the framework?
• Level	What institutional level is included in the framework (individual, group, organizational, society) ?
• Conceptualization and dynamics	How is the social system conceptualized? How are the interactions between the levels incorporated? Macro: Depicts the social system only at macro level Micro: Depicts the social system only at micro level Macro → micro: The macro level influence the micro level Micro → macro: Focuses in the micro level and how this level impacts the macro level Macro↔ micro: Duality between macro and micro levels
Ecological system	
• Name	How is the ecological system included in the framework?
• Level	At which spatial level is the framework designed (global, national,

	sub-national)?
• Conceptualization and dynamics	How is the ecological system conceptualized?
Social-ecological system	
• Conceptualization of interactions	How are the interactions between the levels of the ecological system incorporated? How are the dynamics and interactions between social and ecological systems conceptualized? E→S: Ecological system influence the social system S→E: Social system influence the ecological system E↔S: Reciprocity between systems How are they measured?
• Depth of social and ecological system	Are the social and ecological systems treated as equal?
Orientation	Is the framework 'analysis' or 'action' oriented?
Adapted from Binder et al. (2013)	

4.2.2 Assessing the applicability of both frameworks for natural resource management in the Colombian Orinoco river basin

The Orinoco Basin

The Orinoco is a transboundary river basin between Colombia and Venezuela. The Colombian side of the river basin embraces 345,000 km^2, collecting waters from the Andes mountains, the Guyana shield, and the Amazon river basin (Barletta et al. 2015; León 2005). Water resources include six rivers with an annual average discharge higher than 1,000m^3/sec, 55% of the national wetlands, 33% of the national fresh water reserves, 40% underground waters, and 40% fish species (Correa et al. 2005; Romero-Ruiz et al. 2012b). The river basin is an important reserve of tropical forests covering more than 80,000 km^2 of different types of forest (Ideam 2011a). Fast changes are occurring in the river basin including deforestation, the introduction of exotic crops (e.g. oil palm, rice, soy), and the increase of improved grass species to raise cattle (Benavides 2010; Ideam 2011a; Sanchez-Cuervo and Aide 2013).

Adaptive management in the Orinoco River Basin

Natural resource management in socio-ecological systems is challenging because the dynamic underlying social and ecological systems with potential non-linear feedbacks are difficult to predict and control (Armitage et al. 2015; Folke et al. 2002; Levin et al. 2013). Adaptive management of natural resources reduces the uncertainty in the impacts of different policy choices by making decisions in an iterative way over time (Holling 1978; Rist et al. 2013b; Walters 1986). We use adaptive management components described by Rist et al. (2013a) as criteria to compare the applicability of the planetary boundaries and ecosystem accounting frameworks for natural resource management in the Orinoco river basin (Table 4.2).

Table 4.2. Adaptive management criteria

Components	Question	Description
Stakeholders participation	Is the information useful to define stakeholders? How are stakeholders defined in the framework?	In this criterion we report if the information provided by each framework can be used to define the people, organizations and institutions who use, influence and have an interest in the use of natural resources

Definition of the management problem	Is the information useful to define management objectives? How are management objectives defined and measured in the framework?	In this criterion we assess if the information provided by each framework can be used to define clear management objectives, how can be measured and if this information can be used to assess the impacts of management actions
Establishment of a baseline of understanding	What type information is supplied? How is the information collected and used? How are alternative management options defined?	In this criterion we identify if the models used by each framework can provide useful information to evaluate alternative actions by stakeholders
Implementation of actions/policies	How can actions/policies be implemented?	In this criterion we assess which actions/policies are defined from a set of possible alternatives, guided by management objectives adjusted according to possible changes.
Monitoring effect	What monitoring strategies can be supported? What institutional arrangements can be defined for monitoring progress?	In this criterion we assess the monitoring potential by evaluating if the information provided by each of the frameworks can be used to design an efficient monitoring system
Adapted from Rist et al. (2013a)		

Planetary boundaries and earth system functioning processes at sub-global level

To assess the applicability of planetary boundaries for adaptive management of natural resources in the river basin, we use three earth system processes with sub-global dynamics critical for earth system functioning; - *biogeochemical flows of nitrogen (N) and phosphorus(P), land system change, and fresh water use* - (Steffen et al. 2015b) (Table 4.3). These three processes are directly relevant for river basin management in the Orinoco.

Table 4.3. Earth system functioning processes of the PB approach that operate at sub-global level

Earth functioning processes	**Relevance of the Earth system functioning process and main human pressures**	**Reference**
Biogeochemical flows of nitrogen and phosphorous	Assimilable forms of nitrogen and phosphorous are currently included in the industrial production of synthetic fertilizer to increase the production of food, fibers and biofuels. The turnover rate of nitrogen have doubled, and the annual application of phosphorus to agricultural ecosystems is about a third of which naturally cycle through all terrestrial ecosystems .	(Carpenter 2005; Gruber and Galloway 2008; Steffen et al. 2015b)
Land system change	Tropical forests play a significant role in global biophysical climate regulation by modulating the exchanges of energy and water between land surface and atmosphere. Deforestation in regional tropical forests influence global climate regulation by altering evapotranspiration patterns	(Chapin et al. 2008; Foley et al. 2003; Steffen et al. 2015b)
Freshwater use	All terrestrial biomes depend on fresh water provided through land precipitation as part of the water cycle. Land precipitation returns water to the atmosphere via evapotranspiration (green water) without entering the terrestrial water cycle (blue water including stream flow and groundwater recharge and outflow). Human manipulation of the global water cycle affects ecosystem functioning, biodiversity, food, and human health, about 25% of the planetary river basins run dry before reaching the ocean because human water use (blue water)	(Bogardi et al. 2013; Molden et al. 2007; Steffen et al. 2015b; Trenberth et al. 2007)

Ecosystem accounting

To assess the applicability of ecosystem accounting for adaptive management of natural resources in the river basin we assess the information required to compile extent, condition, capacity and ecosystem services supply accounts following the structure and guidelines of the

SEEA-EEA (Table 4.4) (United Nations et al. 2014b). We provide an illustration for the river basin, assessing two ecosystems (oil palm plantations and tropical forest) in terms of extent, condition, capacity to supply ecosystem services, and the supply of ecosystem services, based on the methods described in Hein et al. (2015) and Vargas et al. (2017) to obtain the different values for the different accounts.

Table 4. Information compiled in the different ecosystem accounting accounts

Ecosystem accounting accounts*	Explanation
Extent	Defines ecosystem's size and location, typically delineated be on land cover type
Condition	Reflects key ecosystem's characteristics (e.g. hydrological and nutrient cycles, species composition and productivity) that influence ecosystem's extent, functioning and quality
Ecosystem capacity to supply ecosystem services	Defines ecosystem's ability to supply ecosystem services under current conditions without degrading the ecosystem. Ecosystem capacity to supply ecosystem services depends on ecosystem extent and condition, current and future ecosystem use patterns and involves resource harvesting and regeneration
Ecosystem services supply	Reflects the supply of ecosystem services such as food, fibers, medicines and fresh water from the different types of ecosystems. Ecosystem services are ecosystem's contributions to benefits used in economic and other economic activity.
*Information contained in the different accounts is based on Hein et al. (2015) and United Nations et al. (2014b)	

4.3. Results

4.3.1 Comparing frameworks

Description based on contextual criteria

Discipline origins

The origins of the planetary boundaries can be found in ecological economics, earth system science, and the literature on global change and on modelling complex ecological dynamics and ecological thresholds. Earth system science enables the identification of earth system functioning processes essential to maintain planet stability, and global change brings human activities as drivers of pressures in earth system. Ecological thresholds lead to abrupt irreversible transitions if they are crossed. They are crucial in setting limits on human activities. The complexity of the dynamics of ecological thresholds are summarized by splitting ecological thresholds in two categories, planetary thresholds driven by global processes, and sub-global thresholds that arise at regional and local level. The impacts of crossing planetary thresholds are palpable at sub-global level, e.g., coastal areas are vulnerable for a rise in the sea level as a consequence of melting polar ice. The impacts of crossing ecological thresholds that arise at sub-global level aggregate, increasing the risk of crossing thresholds in other earth system processes. For example, the use of synthetic fertilizer containing nitrogen and phosphorus in farming areas increases the risk of eutrophication in downstream water resources, gradually increasing the risk of large-scale anoxia in the oceans, with potential consequences for biodiversity and other earth system functions (Watson et al. 2017). Rockström et al. (2009) recognized the spatial heterogeneity of many earth system processes, especially in those where sub-global dynamics play an important role, such as in the nitrogen, phosphorus and water cycles. Of the various disciplinary bases of the planetary boundaries, economics is least visible. Economic information is not used in defining the boundaries, only to indicate economic development as a driver of the increasing pressures on earth system processes.

Ecosystem accounting is based on measurement concepts from different disciplines including statistics, ecology, spatial modeling, economics, and accounting. Ecology brings in concepts such as the ecosystem, ecosystem services, ecosystem processes and biodiversity. Ecosystems are viewed as functional units from which plant, animals and microorganisms interact with the non-living environment, generating goods and services for people (United Nations et al. 2014b). Ecosystem services are defined as the ecosystems' contributions to human activities. Concepts such as resilience, complex dynamics and ecological thresholds are included in the ecosystem accounting framework, however the practical use of these concepts is not clearly indicated. Economic concepts such as production, consumption, accumulation and ownership of assets are brought from economics. Economic concepts related to ecosystem assets and flows of ecosystem services underpin the accounting perspective of ecosystem accounting, enabling the establishment of trade-offs between generation and use of ecosystem services, and ecosystem's potential to supply services in the future. The weak economic background perceived in the planetary boundaries framework can be strengthened by the strong economic background of ecosystem accounting. Likewise, earth system functioning, global change and ecological thresholds are strongly embedded in the planetary boundaries, providing key information to be compiled in ecosystem accounting condition accounts.

Theoretical background

The development of the planetary boundaries framework was fueled by the new paradigm that states we have now entered a new era of global change known as the "Anthropocene", driven by the rapid increase in human activities (Crutzen 2006; Rockström et al. 2009). The planetary boundaries theoretical background stems from earlier approaches to set limits on human activities including limits to growth, safe minimum standards, and the tolerable windows approach (Crowards 1998; Meadows et al. 1972; Raffensperger 1999). These approaches attempt to analyze and quantify natural boundaries using future scenarios and the application of the precautionary principle to avoid critical transitions. The theory behind the planetary boundaries differs from these approaches by focusing on earth system processes, the incorporation of associated ecological thresholds irrespective of human preferences, values and compromises, and the identification of boundaries from which humanity can take actions towards sustainable development. The development of ecosystem accounting was motivated by the fact that separate analysis of ecosystems and the economy do not fully encompass the relationship between human activities and the environment (United Nations et al. 2014b). A main premise is that individual and societal decisions concerning the use of environmental resources will be better informed by using integrated information connecting ecosystems to economic activities. Differences in the theoretical background between both frameworks can be seen as complementary. Whereas the planetary boundaries incorporate earth system processes in the human responsibility of defining limits for social and economic growth, ecosystem accounting provides environment and economic spatially explicit integrated information to inform human society on how to improve the use of natural resources.

Purpose of analysis

The purpose of analysis in the planetary boundaries framework is to propose a safe space in which human activities can take place while avoiding the transgression of critical ecological thresholds. The framework is implemented through expert assessments and synthesis of scientific knowledge. The framework provides information to set boundary levels through control variables (e.g. km^3 of water use per year). Although the planetary boundaries concept is not meant for targeting a specific institution, different international scientific-policy initiatives have used the concept (e.g. the Global Environmental Outlook 5) (Galaz et al. 2012). The purpose of analysis in ecosystem accounting is to integrate environmental and economic information to support policy making and environmental management. Information is presented in physical and monetary terms, and organized following the same logic used in standard measurements of the economy (e.g. national accounts). From a measurement perspective ecosystem accounting focuses on the (i) flows of ecosystem services, and (ii) changes on the stock of ecosystems, i.e. -*ecosystem assets*-. Global initiatives such as the World Bank Wealth Accounting and the Valuation of Ecosystem Services (WAVES), and The Economics of Ecosystems and Biodiversity (TEEB) are users of the framework. Ecosystem accounting can provide broader measures of progress and sustainable development, and can be used for policy in public areas of concern such as land and resource management (United Nations et al. 2014b). Although both frameworks pursue different purposes, supporting the achievement of sustainable development can be seen as a common ground between the two frameworks. Whereas the planetary boundaries aim to influence current trends in social and economic development, bringing a new concept to achieve global sustainability, ecosystem accounting supports national economic decision and policy making to achieve a sustainable use of natural resources.

Description based on structural criteria

Social system

The social system in the planetary boundaries framework is conceptualized from an anthropocentric perspective. The institutional level addressed in the framework is the global society, referenced as “humanity” and “global community”. The hierarchical level at which the social system is conceptualized is the macro level using a global perspective, disregarding interactions with lower hierarchical levels such as groups, organizations and individual persons. Humanity is perceived as a dominant force shaping the planet, and as the main driver of global change. Humanity determines the level and values of the planetary boundaries to maintain a safe distance from critical ecological thresholds. The distance is determined by normative judgements based on risk and uncertainty measures. Likewise, the social system in ecosystem accounting framework is conceptualized from an anthropocentric perspective. The institutional levels addressed in the framework are the institutional units recording statistical information. These institutional units include establishments, enterprises, government entities and households. Institutional units can be grouped in industries (economic units with similar economic activities) and sectors (economic units with similar purposes, objectives and behaviors). The hierarchical level at which the social system is conceptualized is a duality between macro and micro levels. For example changes in the economic behavior of consumption of goods and services at household level influence economic activity at the level of industry, and likewise, government decisions influence the industry and households. The accounting structure applied in ecosystem accounting to conceptualize the social system through institutional units, can be used to support implementing planetary boundaries at a lower hierarchical levels than "humanity", including enterprises and government entities at national level.

Ecological system

It can be argued that the ecological system is conceptualized from an eco-centric perspective in the planetary boundaries framework because the biophysical processes controlling the earth self-regulating capacity, and the ecological thresholds associated to these processes occur irrespective of human activities. Human activities are perceived as *embedded* in the earth system, and strongly depend on critical earth system processes (Heikkurinen et al. 2016). Ecological dynamics in the framework are described by control variables which are quantifiable units used to estimate a boundary level for each earth system process, and the safest distance from an ecological threshold. Expert assessments and biophysical data determine the value of the control variables. The ecological system is conceptualized from an anthropocentric perspective in ecosystem accounting, regarding the central role given to ecosystem services, defined as the contributions of ecosystems to benefits used in economic and other human activities (United Nations et al. 2014b). This definition has a profound anthropocentric perspective, as without human beneficiaries the flow of ecosystem services will be zero. Human activities are perceived as embedded within ecosystems, recognizing that human activities influence ecosystems across the planet. The assessment of ecosystems includes key characteristics such as structure (e.g. food webs, habitats), composition (e.g. fauna, flora), processes (e.g. photosynthesis) and functions (e.g. nutrients recycling). Measurements in the ecological system include assessments of changes in the stock of ecosystem assets and flows between ecosystem assets in physical and monetary terms, recorded in ecosystem's extent, condition and capacity accounts.

Interactions between social and ecological systems

The interactions between the social and ecological systems are conceptualized by describing planetary boundaries for human activities that pressure key earth system functioning processes in the planetary boundaries framework. The dynamics of the interaction is reciprocal, human activities pressure earth system processes and feedbacks from earth system affect human society. The framework is action-oriented because the main goal is to influence current development strategies by proposing limits on human activities. Conversely, the interactions between the social and ecological systems are conceptualized through the lens of ecosystem services in ecosystem accounting. The dynamics of the interaction is conceptualized as the ecosystem system influencing the social system by providing flows of ecosystem services that benefit human society. Although human activities modify ecosystems, often to influence the supply of ecosystem services (e.g. irrigation systems support crops during the dry season), ecosystems influence human society by regulating the stream of ecosystem services that benefit human society. Changes in the stock of ecosystem assets, and flows between ecosystem assets conceptualize the ecological system, however flows from ecosystem assets to economic and non-economic activities conceptualize the dynamics of the interaction between the social and ecological system. Measurements include flows from ecosystem assets to economic and non-economic activities, measured in physical and monetary terms and recorded in ecosystem services supply accounts.

4.3.2 The applicability of both frameworks for adaptive natural resource management in the Orinoco River Basin

This section summarizes the role of the planetary boundaries and ecosystem accounting frameworks for each adaptive management component presented in Table 4.2.

Stakeholder participation

The information provided by the planetary boundaries framework is not meant to identify stakeholders, however, the participation of stakeholders is central for implementing policy actions in natural resource management. For the Orinoco example stakeholders are connected to economic activities that generate impacts on the nitrogen and phosphorus cycles, water use, and land system change. These economic activities include among others oil palm and rice production, cattle ranching, timber harvesting, and hydropower generation (Benavides 2010). Stakeholders involved in these activities include farmers, farmers associations (e.g. FEDEPALMA, FEDEGAN), government institutions (e.g. municipalities) and hydropower industry (e.g. Chivor S.A). Likewise, the spatially explicit biophysical data recorded in ecosystem accounting tables and accounts is not meant to directly identify stakeholders. However, the economic statistical information covering economic transactions between households, legal entities and government institutions compiled in institutional units can be used to identify stakeholders that impact ecosystems (and are affected by changes in ecosystems) in the river basin. Legal entities include the production oriented groups, such as farmers associations (e.g. FEDEARROZ, FEDEPALMA) and non-profit organizations. In the Orinoco basin relevant government institutions include environmental corporations such as Corpochivor and Cormacarena, and municipalities.

Definition of management problems

The planetary boundaries framework can support setting management objectives to steer economic activities such as agriculture and energy generation, to reduce pressures on the nitrogen and phosphorus cycles, water use, and land system in the river basin. For instance, poor management practices in agriculture are a main cause of the disturbed nitrogen and phosphorus cycles. These management practices include either an excess in the application, or a deficit in the use of both nutrients. Excessive fertilizer use is pushed by the increasing demand for food, biofuels, and improved grass species to feed cattle in the river basin. A deficit on nitrogen and phosphorus is the consequence of extractive agriculture and livestock production (e.g. overgrazing), practices that remove nutrients from soil, plants and animals. Poor agricultural management practices are also connected to pressures on water use and deforestation. However, the planetary boundaries framework provides little guidance on optimizing ecosystem management (e.g. fertilizer use) within the boundaries. In addition it is far from straightforward to translate coarse scale thresholds to management objectives for local scale natural resource managers. In the context of ecosystem accounting, management objectives can be defined on the basis of targets related to ecosystem extent (e.g. forest cover), condition, biodiversity (provided a spatial, comprehensive biodiversity account has been developed, carbon or potentially also services flow (United Nations et al. 2014b). Given that the accounts relate ecosystems and the economy, and that information can be expressed in both physical and monetary terms accounting information can be used as input into economic optimization models. However, the accounts, if used in this way, implicitly assume a high degree of substitutability between different ecosystem assets (a weak sustainability interpretation).

Nevertheless, ecosystem accounting can be used to document extent, condition, and the capacity of ecosystems to supply ES over time, and as such support monitoring. For example, ecosystem accounting specifies the growth in oil palm plantations in the Orinoco as an important driver for environmental change, increasing from 53,000 hectares in 2004 to 174,000 hectares in 2014 (Dane and Ministry of Agriculture 2016).

Establishment of a baseline for understanding and identification of alternatives for the Orinoco

The planetary boundaries framework uses control variables to quantify the value of each boundary. Steffen et al. (2015b) propose a set of control variables for sub-global processes based on expert assessment and current scientific knowledge. The level for four control variables of the planetary boundaries can help to explore the establishment of a baseline of understanding between the different stakeholders to evaluate alternative policy actions in the river basin (Fig 4.1).

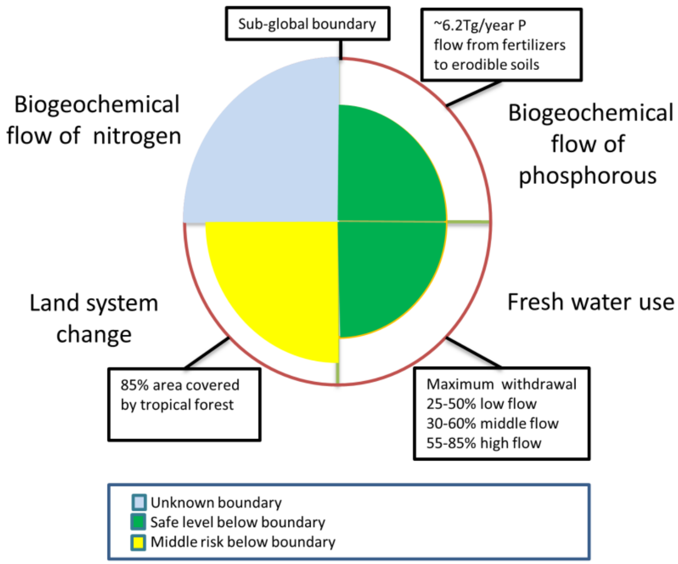

Fig 4.1 Boundary levels for earth system processes with sub-global dynamics. The size of the wedges show the position of the control variable for the Orinoco river basin.

Two control variables were introduced by Steffen et al. (2015b) to quantify the boundary associated to nitrogen and phosphorus cycles; (i) the industrial and biological fixation of nitrogen (mainly for agriculture uses), and (ii) the flow of total phosphorus from fertilizers applied on erodible soils. The first variable is estimated at a planetary level (62 Tg of nitrogen fixated per year (de Vries et al. 2013)), (Fig 4.1). The second variable is estimated to have a ~ 6.2 Tg of phosphorus per year as regional boundary to avoid eutrophication of fresh water systems, as the addition of phosphorus to river basins is almost entirely from fertilizers (Carpenter and Bennett 2011; Steffen et al. 2015b). The values for both variables are not known for the Orinoco. However, modelled estimates show a nitrogen fixation around 0.3 Tg of nitrogen per year, and a flow of phosphorus of around 0.02 Tg of phosphorus per year in year 2000, projected to reach 0.45 Tg of nitrogen per year, and 0.035 Tg of phosphorus per year by 2025 in the Orinoco basin (Camargo and Alonso 2006; van der Struijk and Kroeze 2010). To set the boundaries for land system change, Steffen et al. (2015b) proposes as a control variable the area of forested land as a percentage. The boundary proposed by Steffen et al. (2015b) is 85% of the remaining area of tropical forests, (Fig 1). Currently, 93% of the tropical forests in 2000 remained in 2015 in the river basin, (Ideam 2011a; Ideam 2015a). Although the area covered by tropical forests is below the boundary, the risk of reaching the boundary in the coming years is high, as deforestation continue in the river basin (Ideam 2015a). For water use Steffen et al. (2015b) proposes as the control variable the percentage of water withdrawal of monthly river flows (Gerten et al. 2013; Pastor et al. 2014). The water use boundary was estimated by Steffen et al. (2015b) as 25% (25-55%) for low flow months, 30 % (30-60%) for intermediate flow months and 55% (55-85%) for high flow months. Current estimations for the main rivers of the river basin (e.g. Casanare, Arauca and Meta) show water withdrawals of 50% for low flow months, 20% for intermediate flow months, and 50% for high flow months (Ideam 2011b; Ideam 2015b). Although water use is still below the basin boundary, the increasing demand for fresh water driven by increases on human activities (e.g. hydropower generation and irrigation) put a major pressure, especially in low flow months.

The biophysical information recorded in ecosystem accounting tables and accounts allows the assessment of changes in ecosystem's extent, condition and the future supply of ecosystem services. This information can be used to inform policy discussions, enabling the establishment of a baseline of understanding between different stakeholders to evaluate alternative policy actions in the river basin. For example, the ecosystem extent accounts for the Orinoco show the location of oil palm plantations and forests ecosystems, and the measurement of changes in extent over different accounting periods (Fig 4.2). Information recorded in condition accounts allows the assessment of changes in ecosystem characteristics, such as changes in land cover by switching from forests to oil palm plantations. Tradeoffs between the different ecosystem services supplied by alternative uses of ecosystems can be established, for example changes in carbon sequestration, timber harvesting, erosion control and flood regulation supplied by forests versus an increase in the supply of fresh fruit bunches from oil palm plantations (Vargas et al. 2017) (Fig 4.2). This is an example that shows the effect of combining both frameworks in order to highlight how short term, immediate resource gains are traded off against the long term stability and complexity of the interacting processes that regulate the earth system.

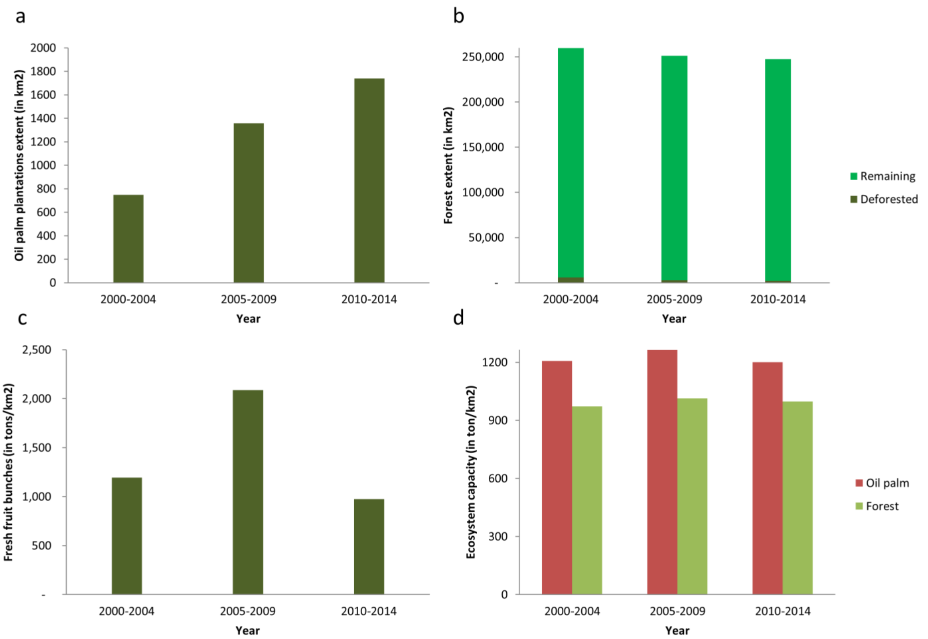

Fig 4.2. Time series between 2000 and 2014 for the Colombian Orinoco Basin showing a) changes in the extent of oil palm plantations (Fedepalma 2005; Fedepalma 2008; Fedepalma 2015) b) changes in the extent of forest and the extent of deforested areas (Ideam 2011a; Ideam 2015a), c) changes in the supply of oil palm fresh fruit bunches (Fedepalma 2005; Fedepalma 2008; Fedepalma 2015), and d) changes in the capacity of oil palm plantations and forest to supply ecosystem services based on net primary productivity (NPP) (Vargas et al. 2017).

Implementation of actions or policies

Information from the boundary levels can be used to target specific policies in Colombia, including water regulation (Ministry of environment housing and land 2010), forests and land use (Ministry of environment et al. 2000; Ministry of environment and sustainable development 2013) and the nitrogen and phosphorus cycles (Ministry of environment and sustainable development 2013). Different actions can be implemented to reduce the current rates of deforestation, and regulate water use and the application of fertilizers containing nitrogen and phosphorus. Current policies related to the biogeochemical cycle of nitrogen and phosphorus are still weak at national level. Information provided with ecosystem accounting can be used for decision and policy making including land use alternatives, alternative energy use and long term environmental trends awareness (Edens and Hein 2013; Obst and Vardon 2014; United Nations et al. 2014a). Tradeoffs between the supply of ecosystem services (e.g. flood control in savannahs ecosystems vs rice production), and changes in ecosystem's condition (e.g. nutrient depletion by poor agriculture practices), recorded in the different accounts can be used by governmental institutions such as autonomous regional environmental corporations (e.g. Corporinoquia, Cormacarena, Corpochivor) that regulate the use of natural resources in the Orinoco river basin.

Monitoring effects

The control variables proposed by Steffen et al. (2015b) such as phosphorus flows from fertilizers to erodible soils can be used to evaluate the impact of policy actions aiming to reduce the use of phosphorus in agriculture and eutrophication of lakes and rivers (Ministry of environment and sustainable development 2013; Ministry of environment housing and land 2010). Moreover, control variables such as area of forested land as a percentage of the original forest cover can be used to evaluate changes in the amount of tropical forest in the river basin, assessing the impact of forest conservation strategies. Ecosystem accounting provides consistent, spatially explicit, structured data over specific accounting periods useful to support monitoring strategies concerning ecosystems use and economic performance. Land use, land cover changes such as switching from forests to increase the extent of oil palm plantations can be monitored by crossing information recorded in extent and condition accounts every year. However, ecosystem accounting requires geo-referenced data and modelled outcomes which are not always available, especially in low-income countries. Ecosystem accounting provides biophysical information useful to measure the effectiveness of management options by looking resource impacts and benefits during each accounting period.

4.4 Discussion

4.4.1 Planetary boundaries and ecosystem accounting

Implementing the planetary boundaries framework for achieving sustainability is challenged by the absence of defined boundaries for earth system processes for sub-global dynamics. This is relevant because surpassing local ecological thresholds can lead to local transitions in ecosystems, as well as because decisions concerning the governance and management of natural resources are mostly taken at sub-global level. Hence, translating global planetary boundaries to boundaries relevant at national and sub-national levels is of utmost importance, and this has also been an active field of research (Cole et al. 2014; Dearing et al. 2014). Translating approaches either involves disaggregating global values by downscaling the control variables into (sub-)national targets, or aggregation by determining indicators at (sub-)national level (Häyhä et al. 2016). Our study shows that the aggregated approach applied by ecosystem accounting can support identifying thresholds, and monitoring progress towards maintaining ecological stability, in particular at national or sub-national level.

However, contextual and structural differences between both approaches, can either constrain or facilitate their potential integration. Contextual differences such as the purpose of analysis, methods, scale and orientation are a major constraint to their integration. This is apparent, in particular, in the conceptualization of the social and ecological systems. The social system in the planetary boundaries framework is, in general, somewhat poorly conceptualized, focusing on the pressures resulting from economic activities on earth system processes. In this way, the accounting structure applied in ecosystem accounting complements the planetary boundaries framework by providing detailed economic information derived from economic transactions recorded in institutional units at national level. The ecological system in the planetary boundaries framework focuses on key earth system processes. Biophysical information compiled in ecosystem accounting condition accounts can be relevant in assessing earth system processes with sub-global dynamics, by providing spatially explicit information concerning flows of nitrogen and phosphorus, evapotranspiration patterns and water availability. The condition account, in particular, can be developed in such a way that it comprises information on pressures exerted on ecosystems, changes in state indicators, and potentially (although this has never been tested yet): a comparison between the current ecosystem condition with the condition at which ecological thresholds are likely to be exceeded (using a relative metric, see UN, 2017).

Furthermore, According to Häyhä et al. (2016) spatially heterogeneous interconnected processes (e.g. biogeochemical cycle of nitrogen and phosphorus) are only recently seen as global problems, however they may not show up as national issues if only territorial approaches are applied at national level. The planetary boundaries framework can be a powerful tool to translate spatially heterogeneous inter-connected problems such as land system change and water use from global problems to national issues. Ecosystem accounting can be a powerful tool to support the planetary boundaries framework by incorporating global problems into information systems in support of national policies. Recent developments in earth observation systems including drones and satellite remote sensing (e.g. Landsat 8, Sentinel family missions) provide new data to populate condition and other accounts increasing the usefulness of ecosystem accounting also for monitoring ecosystem state in relation to planetary boundaries.

4.4.2. Natural resource management in the Colombian Orinoco river basin

Given the rapid changes in land use in the area, management actions are needed to reduce human pressures in the biogeochemical cycle of nitrogen and phosphorus, land system change and water use in the Orinoco river basin. Our study includes adaptive management criteria as described by Rist et al. (2013a) to assess the applicability of the planetary boundaries and ecosystem accounting frameworks for natural resource management in the Orinoco river basin. A current national policy CONPES (2014), has identified 3 million hectares in the Orinoco river basin as potential land to be converted from forests and savannahs to agriculture fields, without assessing the environmental responses following these changes. Potential impacts derived from implementing this policy include: doubling the annual yield of nitrogen and phosphorus, reducing the total area covered by tropical forests, and reaching the maximum water withdrawal threshold in low flow months (Ideam 2011a; Ideam 2015a; van der Struijk and Kroeze 2010). Turning planetary boundaries into management actions in the Orinoco river basin is challenged by difficulties in identifying stakeholders, uncertainties in defining the level of the boundaries and associated thresholds, and difficulties in developing monitoring strategies based on uncertain thresholds. Ecosystem accounting is an analysis-oriented framework that can provide useful information to overcome these challenges. Economic information compiled in institutional units can be used to identify relevant stakeholders in the river basin by identifying groups of economic activity such as the oil palm industry and farmers associations (e.g. FEDEPALMA and FEDEGAN), as well as, among others, profits and employment generated by these economic activities. Moreover, the accounting structure in ecosystem accounting allows aggregating source information to derive indicators, enabling the use of control variables such as phosphorus flow from fertilizers to erodible soils and nitrogen fixation as indicators to monitor changes in condition in aquatic and soil ecosystems in the river basin.

Condition accounts can be used to monitor changes in the river basin ecosystems caused by external pressures in planetary processes, and shifts in ecosystems quality aggregating biophysical information in condition indicators (e.g. evapotranspiration, water stress, drought, water quality and biodiversity indicators). Setting limits on economic activities based on boundaries associated to nitrogen and phosphorus cycles, water use and land system change, supported by information compiled in an ecosystem accounting structure can be a promising approach for natural resource management at the level of river basin. However, in the case of the Orinoco basin, measurement of such flows is incomplete at the moment, and where measurements are available there is not structured and regular reporting on these data. Ecosystem accounting in combination with the planetary boundary approach can assist in identifying key data gaps. Ecosystem accounting can also support developing assessment and communication approaches for such information (building on experiences with the national accounts that in Colombia, as in most other countries, are reported on an annual basis following a specific set of guidelines including on data quality assurance).

4.5 Conclusions

Achieving global sustainable development requires humanity to manage ecosystems in such a way that critical thresholds are avoided. Challenges occur both in defining these thresholds, and in managing the trade-offs involved in resource management within the safe operating space. A solid monitoring system is required in order to assess how far the social-ecological system is removed from these thresholds, and to guide policy actions in the environmental space. Our study postulates that ecosystem accounting can be used to support the translation of planetary boundaries into indicators that can be monitored at the national level, including boundaries for the nitrogen and phosphorus cycles, water use, and land system change. The concisely organized structure of ecosystem accounting accounts provides consistent information necessary to support decisions concerning environmental management and environmental policy and can facilitate the potential use of planetary boundaries at the national level. As we have shown with an application in the Orinoco river basin, shifting the traditional approach to governance and management towards a sustainable use of natural resources requires a combination of analytical and action oriented frameworks to better inform decision makers. Although both the planetary boundaries and the ecosystem accounting frameworks pursue different purposes, supporting the achievement of sustainable development can be seen as a common ground between the two frameworks. Given strengths and weaknesses of both approaches, their combination is strongly recommended. Specifically, the planetary boundaries framework involves a stronger interpretation of sustainability compared to the ecosystem accounting framework, and allow understanding environmental risks, whereas the ecosystem accounting offers a comprehensive monitoring framework as well as an opportunity to balance trade-offs within humanity’s safe operating space.

Chapter 5

Using ecosystem accounting to assess the natural capital of agricultural systems, a case study for the Orinoco river basin

Abstract

To meet our future demand for food and biofuels current global crop production has to double by 2050. The increasing scarcity of land and water resources for agriculture and the increasing decline in the natural capital of agricultural systems undermine our ability to produce enough food and biofuels. Making use of the potential to produce crops on existing farming land is needed to increase the production of food and biofuels. The potential to produce crops is determined by the water-limited potential yield, the yield gap, and water productivity. The yield gap is the difference between the water-limited potential and what farmers' produce in the field, narrowing this gap is necessary to increase crop production. However, narrowing this gap using synthetic fertilizers and pesticides undermines the natural capital underpinning agricultural production (e.g. pollination and nutrients supply). Yield gap analysis do not include such decline in the natural capital, making the integration of this analysis with natural capital assessments is essential to avoid a further decline in the natural capital underpinning agricultural production. Ecosystem accounting has been developed to assess the natural capital of ecosystems but is barely applied to assess agricultural systems. In this study, we used remote sensing information, agricultural statistics, concepts used in ecosystem accounting and crop production potential analyses to assess the natural capital of two types of agricultural systems in two municipalities of the Orinoco river basin in Colombia. We assessed the extent of each agricultural system and their ability to produce rice, soy, sugar cane, oil palm FFB, grass to graze cattle and to sequester carbon. Ecosystem accounting allows for monitoring the expansion of agricultural areas and the analysis of changes in ecosystem services making the link between agriculture and the environment explicit and traceable. This is relevant to satisfy the needs for food and biofuels by a growing population without declining the supply of key ecosystem services such as carbon sequestration. Including a broader set of ecosystem services such as pollination, water supply and pest control is needed to make a more comprehensive analysis of changes in the natural capital underpinning agricultural production.

This chapter is based on:

Vargas L, Willemen L, van Bussel Lenny G.J. Using ecosystem accounting to assess the natural capital of agricultural systems, a case study for the Orinoco river basin. Agricultural Systems. (Submitted)

5.1. Introduction

The intensive use of fertilizers, machinery and improved genetic crop varieties has resulted in a doubling of the global production of cereals in the last 40 years (Godfray et al. 2010; Tilman et al. 2002). Agricultural activities have however transformed a large portion of the ice-free land, increased their use of fresh water resources and impacted water quality (Bouwman et al. 2013; Foley et al. 2011). For example, 38% of the land surface is currently used for agricultural activities, 80% of the world's agricultural land is occupied by rain-fed agricultural systems and 70% of the global fresh water withdrawals is used by irrigated systems (Falkenmark and Rockström 2013; FAO 2015; World Bank 2018). To cover our future demand for food and biofuels, crop production should double again by 2050 (Foley et al. 2011; Tilman et al. 2011).The best suitable land for farming is however already in use and a large portion of the remaining land is not suitable for agriculture (covered by forests, protected by law, or populated). Consequently, transforming additional land into agricultural fields will generate high environmental and social costs (Lambin et al. 2013; Lambin and Meyfroidt 2011). Furthermore, climate change and human population growth will increase the human use of water and thus intensify the scarcity of water resources for agriculture (Pimentel et al. 2004). These limitations in land and water resources for agriculture will hamper our ability to produce sufficient food and biofuels. To minimalize the pressure on land resources, it is essential to make use of the potential to produce crops on existing faming land, particularly in developing countries (Tilman et al. 2011; van Ittersum et al. 2013).

The water limited potential yield, the yield gap and water productivity determine together the crop production potential of an agricultural system (van Ittersum et al. 2013). The water-limited potential yield is the yield of an adapted crop cultivar for which the growth rate is only controlled by solar radiation, temperature, atmospheric CO_2, genetic traits governing the length of the growing period, light interception by crop canopy, water supply, soil type and field topography, and not by nutrients and biotic stresses (Evans 1996; van Ittersum et al. 2013). Different methods can be applied to estimate the water-limited potential yield: field experiments, yield contests, primary productivity measurements, and crop modelling simulations (Lobell et al. 2009). Crop modelling simulations are the most robust approach to estimate water-limited potential yield as they evaluate the interactive effects of genotype, climate and management over time (Grassini et al. 2015). However, for crop modelling simulations consistent, high quality and spatially explicit information about weather conditions and crop management is required, information which is lacking in most parts of the world (Grassini et al. 2015; van Bussel et al. 2015). Lobell et al. (2009) suggested the use of natural ecosystem Net Primary Productivity (NPP) to estimate the water-limited potential yield. The NPP of natural ecosystems can be used as an alternative to simulation models in situations where data is poorly available for large areas, particularly because NPP data are globally available from remote sensing.

The difference between the water-limited potential and the average farmers' yield is the yield gap (Lobell et al. 2009). Estimating the yield gap is important to analyze which factors are limiting farmers yields including a lack of water, nutrients and/or pollinators, pest damage and diseases (Cassman et al. 2003). Narrowing the yield gap has been considered necessary to increase the production of food and biofuels on underperforming farming land and to stop the expansion of agriculture into natural ecosystems (Foley et al. 2011). In addition to the yield gap, water productivity of a crop system is key to determine because of the increasing scarcity of water resources for agriculture (Passioura 2006). Water productivity is the efficiency with which water is converted to food, and can be quantified as the ratio between grain yield and evapotranspiration (Grassini et al. 2011; van Ittersum et al. 2013). Estimating water productivity is relevant to know where water productivity can be improved. Water productivity enhancements by improved management of water, land and crop resources, can save an estimated of 5,600 km^3 of fresh water withdrawals per year to feed our growing population by 2050 (Rockström and Barron 2007).

Narrowing the yield gap by intensifying the use of external inputs such as pesticides and synthetic fertilizer will decline the natural capital that underpins agricultural production (Bommarco et al. 2013; Pretty and Bharucha 2014; Rasmussen et al. 2018). Natural capital refers to the living and non-living components of ecosystems including plants, animals, soils, and natural processes that contribute to the generation of goods and services of value to people -*ecosystem services*- (Guerry et al. 2015; Mace et al. 2015). The analysis of the yield gap can help to identify regions with high potential to produce more food and biofuels, however, such an analysis does not address changes in natural capital because of intensified use of external inputs. Changes in natural capital, such as the quality and amount of water or the decreased supply of pollination, might have however negative implications for narrowing the yield gap. Moreover, the overemphasis on enhancing agricultural productivity narrowing the yield gap make the contribution of the natural capital underpinning agriculture production invisible and unaccounted (TEEB 2018). For example, nitrogen and phosphorus are removed from the soil to support crop production, however, if these nutrients are not replenished soils become depleted, representing a "hidden cost" since nutrients exported from the soil are not evident nor accounted (Díaz de Astarloa and Pengue 2018).

The integration of natural capital assessments and yield gap analysis can avoid a further decline in the natural capital underpinning agricultural production. This integration is essential because natural capital assessments make the contribution of the natural capital –*ecosystems and ecosystem services*- sustaining agricultural production visible, they help to integrate this information into decision and policy contexts, and they are important tools to inform sustainable development (Guerry et al. 2015; TEEB 2018). An important step in the assessment of natural capital through the analysis of changes in the condition and use of ecosystems and in the supply of ecosystem services has been the development of the System of Experimental Ecosystem Accounting (SEEA-EEA) (in short "ecosystem accounting") (United Nations et al. 2014b). An ecosystem in this context is a functional unit where a community of plants, animals, microorganisms and non-living components interact, and an ecosystem service is the contribution of an ecosystem in the production of benefits to people (e.g. water supply, climate regulation) (United Nations et al. 2014b). Most of these benefits, however, require for their provision a combination of inputs, for example the production of food and biofuels depends on ecosystem processes such as nutrient cycling along with farm labor, knowledge, and transport (Belinda et al. 2013; Díaz et al. 2015). In practice, decoupling ecosystem processes and services from human inputs is difficult and therefore for the measurement of cultivated crops in ecosystem accounting it is assumed that the inputs of the natural capital such as pollination and nutrients from the soil are in a fixed proportion to the quantities of the harvested crop (United Nations et al. 2014b). Recent studies demonstrated the use of spatially explicit information compiled in ecosystem accounts to assess the contribution of ecosystems to economic activities in among others, The Netherlands, Norway and Indonesia (Remme et al. 2015; Schröter et al. 2014; Suwarno et al. 2016). However, ecosystem accounting concepts have not been explicitly explored for its use to assess the natural capital of agricultural systems, and moreover, neither integrated with concepts related to yield gaps analysis.

The aim of this paper is to explore if and how ecosystem accounting and yield gap analysis can be used to assess changes in the natural capital influencing agricultural production. In particular, this paper uses agricultural statistics and remote sensing information following the ecosystem accounting guidelines to analyze the potential of agricultural systems to produce food, fodder and biofuels in two municipalities of the Altillanura region in the Colombian Orinoco river basin. A large portion of this region is still covered by tropical forests, natural savannahs and unpolluted rivers. The current national policy CONPES (2014) encourages the expansion of agricultural areas to start new plantations of rice, soy, grass to feed cattle, sugar cane and oil palm, among others. At the same time this policy aims to protect the biodiversity and natural resources in the Altillanura. The recently signed peace agreement, can potentially increase clearing natural ecosystems to allow the expansion of agriculture fields, as for several decades the violence in this part of the country was a barrier for agricultural investments (Sánchez-Cuervo and Aide 2013; Sierra et al. 2017).

5.2. Methods

5.2.1 Study area

The Orinoco river basin is located in the north of South America, between Colombia and Venezuela, covering 350 thousand km^2 on the Colombian side. The Colombian Altillanura is a sub-region in the Orinoco river basin covering 130 thousand km^2. In Altillanura, 28 thousand km^2 are earmarked as suitable for agricultural use: 12 thousand km^2 for cropping and 16 thousand km^2 for grazing cattle (CONPES 2014). The Altillanura is also an important reserve for biodiversity and water resources, however, the protection of these resources through national parks and policies is not enough (Lasso et al. 2010; Sanchez-Cuervo and Aide 2013). We selected two municipalities in the Altillanura (Fig 5.1), where agriculture activities are currently being developed, where a large portion of the land is still available for agriculture, and where agricultural statistics for the period 2010-2014 were available. The larger municipality is Puerto Gaitán, covering 18,000 km^2 with a population of 22,000 inhabitants, an average temperature of 28 ºC, an altitude of 140 meters above sea level, and an average rainfall of 1,500 mm per year. The municipality of Puerto Lopez covers 6,740 km^2 with a population of 33,500 inhabitants, with a similar average temperature and altitude, but with a higher average rainfall of 2,300 mm per year. The main agricultural activities in both municipalities include croplands covered with rain-fed and irrigated rice (*Oryza sativa*), sugar cane (*Saccharum officinarum*), soy (*Glycine max*), oil palm (*Elaeis guineensis*), fruit trees, and pastures covered with improved exotic grass species (*Braquiaria spp*) to graze cattle.

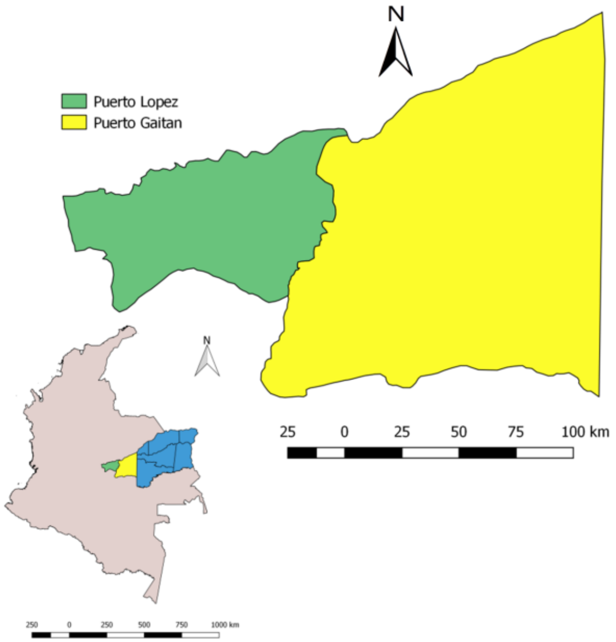

Fig 5.1. Maps showing the location of the Altillanura within the Orinoco river basin, and the municipalities of Puerto Lopez and Puerto Gaitán

In 2014 the Orinoco river basin supplied 58% of the rice, 80% of the soy, 36% of the oil palm FFB, and 21% of the cattle to fulfill the national demand for these crops (Dane and Ministry of Agriculture 2016).

5.2.2 Using ecosystem accounting concepts to assess the natural capital of agricultural systems

Ecosystem accounting concepts and yield gap analysis

The concepts used for ecosystem accounting and yield gap analysis, and the variables we used to assess the natural capital influencing agricultural production are shown in Table 5.1. The rows in Table 5.1 are used to align concepts, however only one of the two aligned concepts is used in the analyses (in bold). Following the structure of accounting systems, ecosystem accounting is founded on the relation between stocks and flows, where the stocks are spatial areas comprising an ecosystem, including agricultural systems, and the flows are flows of ecosystem services. In this study an agricultural system is defined as a human managed ecosystem intended to produce a single or a variety of crops. We focus on the assessment of changes in the stock of two types of agricultural systems: cropland and pastures. Whereas cropland is intended to produce rice, soy, sugar cane and oil palm, pastures are intended to produce grass to be grazed by cattle.

Table 5.1. Ecosystem accounting concepts, crop production potential analysis and the variables used in this study to analyse agricultural systems

Ecosystem accounting		Crop production potential		Variable	Unit
Concept	Characteristics	Concept	Characteristics		
Ecosystem	• Complex dynamic system • Plants, animals, microorganisms and their non-living environment interact as a functional unit	**Agricultural system**	• Human managed complex dynamic system • Agricultural and non-agricultural plants, animals, microorganisms and their non-living environment interact as a functional unit	Extent: Area per agricultural system and per crop	Km^2
Potential to supply ecosystem services	• Ability to generate an ecosystem service • Current condition and management or use • Supply is independent of the demand for the service			NEP: amount of carbon sequestration (per agricultural system and per crop)	ton of C per ha per year
Capability to supply ecosystem services	• Ability to supply an ecosystem service • Maximum sustainable yield • Optimal management • A demand for the service is required	Water limited potential yield	• The hypothetical highest yield of a rain-fed crop cultivar • Nutrients are non-limiting and (a)biotic stresses are effectively controlled • Water supply might be a limiting factor	NPPC: crop production (per crop)	ton of C per ha per year
Ecosystem service Ecosystem service Flow	• Contribution of ecosystems to benefits used in economic and non-economic activities • Reflect the amount of an ecosystem service extracted or received by a beneficiary (e.g. government, companies and households)	**Annual actual yield**	• The annual yield of each crop obtained by an average farmer • Reflect the supply and use of crops between agricultural systems and beneficiaries • Crop yields are traded outside farms	Y: actual yield of each crop	ton of C per ha per year
Capability analysis	Difference between the capability and the flow of an ecosystem service	The yield gap	• Difference between the water limited potential yield and the annual actual yield	Yg: NPPC-Y for each crop	ton of C per ha per year
		Water productivity	• Efficiency with which water is converted to food	WP:NPP/ET (per agricultural system)	kg of C per m^3

Four ecosystem services were defined based on Haines-Young and Potschin (2018) classification: standing rice, soy and sugar cane that can be harvested and used for the production of food (i), standing oil palm fresh fruit bunches (FFB) that can be harvested and used for the production of biofuel (ii), standing grass that can be used to feed cattle (iii), and carbon sequestration to regulates the climate (iv). The flow of these ecosystem services are the annual harvest of each crop and the amount of carbon sequestered by each agricultural system. Carbon sequestration was selected because of its importance in mitigating the global effects of climate change, the production of rice, sugar cane, soy, FFB and grass grazed by cattle were selected because of their economic importance in the Altillanura (Benavides 2010). We assessed changes in the stock of these two agricultural systems by estimating their extent, their potential to sequester carbon and their capability to produce rice, soy, sugar cane, oil palm and grass. We applied a capability analysis to compare the annual yield of each crop with each agricultural system *potential* to sequester carbon and *capability* to produce crops. Potential and capability reflect changes in the stock of natural capital of ecosystems represented by changes in their ability to sustainable supply ecosystem services as a flow over time (Hein et al. 2016). In this study potential reflect an agricultural system's ability to sequester carbon irrespective of its demand and capability prioritize agricultural system's ability to produce crops from the supply of other ecosystem services to satisfy the demand for raw materials used to produce food and biofuels under optimal management conditions. We assessed water productivity for each agricultural system to estimate the efficiency with which water is used to produce crops. These assessments were applied for the two municipalities for the time period between 2010 and 2014.

The extent of cropland and pastures

To estimate the extent of cropland and pastures we used remote sensed data from the Moderate Resolution Spectroradiometer (MODIS) land cover product MCD12Q1 (see supplemental material for more details) and agricultural statistics. We first multiplied the number of pixels within the cropland and grassland land cover class with the pixel size of the MODIS land cover product (21.4 ha). To assess the area harvested rice, soy, sugar cane and oil palm we used agricultural statistics data compiled by the Colombian Department of Statistics, as the land cover data did not specify these (Dane and Ministry of Agriculture 2016). The extent of the annual area grazed by cattle, i.e. pastures, was estimated using livestock data from the Colombian federation of cattle producers (Fedegan 2014). To calculate this number, we assumed that one hectare of pasture was required per year to feed one head of cattle with an life weight of above 450 kg (slaughter life weight) (Benavides 2010).

The potential to sequester carbon

The potential to sequester carbon is the annual net carbon accumulation of standing crops in cropland and pastures. We used the Net Ecosystem Production (NEP) to estimate the annual net carbon accumulation (Luyssaert et al. 2007; Randerson et al. 2002). To assess the NEP we subtracted the annual accumulated heterotrophic respiration (Hr) from the annual accumulated NPP (Equations 1 and 2) (Luyssaert et al. 2007). We used MODIS MOD17A3 data to derive the annual accumulated NPP and NASA's carbon monitoring flux pilot project data to derive the annual Hr (Ott et al. 2015; Running and Zhao 2015). All the calculations are spatially explicit, estimated for each pixel $_i$, in cropland $_c$ and pastures $_p$, per year $_y$:

$$NEP_{c,i,y} = NPP_{c,i,y} - Hr_{c,i,y} \qquad (1)$$

$$NEP_{p,i,y} = NPP_{p,i,y} - Hr_{p,i,y} \qquad (2)$$

The capability to produce crops

The capability of cropland to produce rice, soy, oil palm FFB and sugar cane ($NPPC_{1\text{-}4}$) and the capability of pastures to produce grass ($NPPC_5$) is the annual NPP of potential vegetation by specific crops ($NPPo_{1\text{-}5}$) divided by a fraction that represents the NPP allocated to the plant organs that produce yield ($NPPa_{1\text{-}5}$)(Equations 3 and 4). The $NPPo_{1\text{-}5}$ is the NPP value of the natural ecosystem that would prevail in the absence of a specific crop (Table 5.2), for example, if a savannah ecosystem is cleared to plant a rice field the *NPPo* estimates the NPP of the savannah in the absence of the rice field derived from the area used to plant rice every year and the amount of rice harvested (Medková et al. 2017). The $NPPa_{1\text{-}5}$ is the percentage of the total NPP allocated to the plant organs that produce yield (Table 5.2). All the calculations are spatially explicit, estimated for each pixel $_i$, for each crop $_{1\text{-}5}$, per year $_y$.

$$NPPC_{1-4,i,y} = NPPo_{1-4} \times NPPa_{1-4,c,i,y} \tag{3}$$

$$NPPC_{5,i,y} = NPPo_{5} \times NPPa_{5,p,i,y} \tag{4}$$

Table 5.2. Variables used to estimate the NPP of potential vegetation (*NPPo*) and the NPP allocated to the plant organs (*NPPa*)

Variable	Unit	Crop					Reference
		Rice	Soy	Oil palm FFB	Sugar cane	Grass	
NPPo=NPP×α	Ton of C per ha	α=2.13	α=6.6	α=0.76	α=0.15	α=1.1	(Haberl et al. 2007; Medková et al. 2017)
NPPa=NPP×β	Ton of C per ha	β=55%	β=45%	β=17%	β=75%	β=35%	(Emmanuelle et al. 2016; Inman-Bamber et al. 2002; Kumar et al. 2014; Sarmiento and Pinillos 2001; Silva-Olaya et al. 2017; Song et al. 2013)

Capability analysis

Capability in combination with harvest data can be used -as a proxy- for determining the yield gap of a certain crop. In this study we compare the annual yield for each crop with the cropland and the pastures capability to produce each crop. The annual yield level can be below, equal or higher than the agricultural system capability to produce each crop and can negatively affect agricultural system's potential to sequester carbon. To assess this level ($Yg_{1\text{-}5}$) we subtracted the annual yield ($Y_{1\text{-}5}$) of each crop from the cropland and pastures capability to produce each crop ($NPPC_{1\text{-}5}$) (Equations 3 and 4). The annual yield is the annual harvest of each crop per unit of area (in tons per ha) obtained from agricultural statistics (Dane and Ministry of Agriculture 2016) and converted to tons of carbon per hectare using the dry matter and carbon content of each crop. The dry matter content in rice, soy, grass, and oil palm FFB is 80% the harvested product, and in sugar cane is 20% (Ivanov et al. 2011). The carbon content in plants and animals is 50% of the total biomass (Chapin III et al. 2011). The annual yield of grass (Y_5) was derived from the annual supply of cattle (in tons of live weight per hectare of grazed grass) obtained from livestock statistics (Fedegan 2014) and converted to tons of carbon per hectare. The annual supply of cattle reflect the annual yield of grass because grazing on native or improved grass varieties provide most of the nutrients required to rear cattle in the Altillanura. All the calculations are spatially explicit, estimated for each pixel $_i$, for each crop $_{1\text{-}5}$, per year $_y$.

$$Yg_{1-4,i,y} = NPPO_{c,i,y} - Y_{1-4,i,y} \tag{5}$$

$$Yg_{5,i,y} = NPPO_{p,i,t} - Y_{5,i,y} \tag{6}$$

Water productivity

The water productivity (WP) describes the efficiency with which water is used to grow crops in each agricultural system. To estimate WP we used remote sensed data derived from MODIS NPP MOD17A3 and evapotranspiration (ET) MODIS MOD16A3 globally validated with an accuracy of 76% (Courault et al. 2005; Kalma et al. 2008). The WP was not assessed at per crop basis because spatially explicit data to identify the location of each crop and to discriminate between rain-fed and irrigated crops is lacking. We assumed that most of the crops are rain-fed, however, we recognized that irrigation systems are used in the area. The WP for each agricultural system was estimated calculating the amount of water used relative to their net carbon gain (in ton of C per m^3) (Steduto and Albrizio 2005; Tian et al. 2010). The WP links NPP with evapotranspiration (ET), where ET is the total vapor flux between the canopy and the atmosphere, consisting of evaporation from the soil, plant transpiration and evaporation of the rain water intercepted by the canopy before it reaches the ground (Running et al. 2017). The WP in cropland $_c$ and pastures $_p$ is the ratio between the $NPP_{c,i,y}$ and $NPP_{p,i,y}$ (from Equations 1 and 2), and the annual ET in cropland $_c$ and pastures $_p$. To assess the relation between NPP and ET, we used a simple linear regression model and a scatter plot using the dplyr and ggplot2 packages in R software (Team RStudio 2015). All the calculations are spatially explicit, estimated for each pixel $_i$, for cropland $_c$ and pastures $_{p,}$ per year $_y$.

$$WP_{c,i,y} = {NPP_{c,i,y}}/{ET_{c,i,y}} \tag{7}$$

$$WP_{p,i,y} = {NPP_{p,i,y}}/{ET_{p,i,y}} \tag{8}$$

In addition to water productivity the seasonal distibution of water is a key limiting factor that influence cropland and pastures capability to produce crops (*NPPC*). Altought ET and precipitation are two main important components of the water cycle regulating the availabilty of water for agricultural systems, precipitation was not part of this study as ET integrates the different sources of water that can be attribuited to the cultuvation of a crop. To assess the annual and monthly ET for the time period 2010 to 2014, we used remote sensing images from the MODIS MOD16A3, and monthly (averaged 16-day) remote sensing images from the MODIS MOD16A2 ET product. We used the seasonal decomposition algorithm from the ggseas package in R (Team RStudio 2015), to analyse the seasonal distributiton of ET.

5.3. Results

5.3.1 The extent of cropland and pastures

Assessing changes in the size and location of cropland and pastures provide insights about the level of expansion of these agricultural systems in the Altillanura. In 2014, the pastures covered almost half of the total area in both municipalities (41% of the total land in Puerto Lopez and 52% in Puerto Gaitán), and the cropland covered less than 20% of the total area (5% of the total land in Puerto Lopez and 18% in Puerto Gaitán) (Fig 5.2).

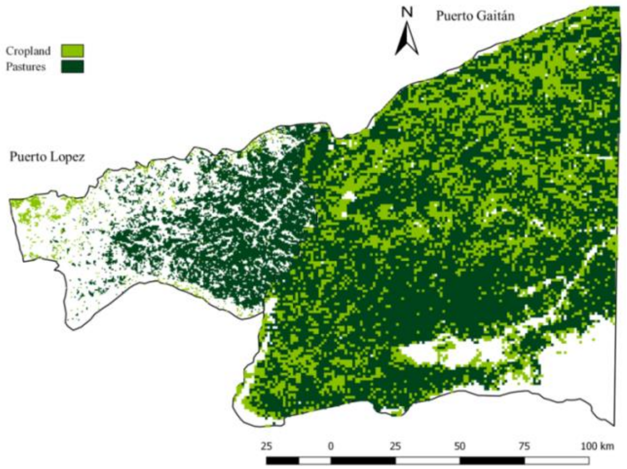

Fig 5.2. Map showing the location of cropland and pastures in the muncipalty of Puerto lopez and Puerto Gaitan in 2014. White areas indicate forest and other land cover types not included in this study.

Our results in Table 5.3 indicate changes in the size of the cropland and the pastures (based on remote sensing), and changes in the area annually harvested for each crop (based on agricultural statistics).

Table 5.3. The extent of the cropland and the pastures (in hectares) based on remote sensing land cover data (*), the total area annually harvested per agricultural system and per crop based on agricultural statistics (Dane and Ministry of Agriculture 2016)(in hectares)

Municipality	Year	Land cover*		Harvested area		Harvested area per crop			
		Cropland	Pastures	Cropland	Pastures	Rice	Soy	Oil palm	Sugar cane
Puerto Lopez	2010	26,114	272,245	22,931	129,947	17,916	2,914	1,500	601
	2011	25,278	237,491	30,316	137,044	19,039	8,477	1,500	1,300
	2012	15,608	231,831	22,785	133,106	17,574	1,861	2,050	1,300
	2013	22,126	229,151	36,061	136,306	17,910	11,690	2,050	4,411
	2014	22,126	229,151	34,124	133,961	14,522	11,832	3,050	4,720
Puerto Gaitán	2010	26,690	129,990	16,563	99,631	4,337	8,201	4,000	25
	2011	25,110	120,890	21,181	101,769	4,298	3,653	13,200	30
	2012	33,890	118,710	18,024	98,954	3,698	8,291	6,000	35
	2013	34,580	97,250	26,339	103,365	4,031	9,073	13,200	35
	2014	34,580	97,250	26,589	103,501	2,574	5,980	18,000	35

Whereas the size of the pastures decreased between 2010 and 2014 in both municipalities, the size of the cropland increased in Puerto Gaitán but decreased in Puerto Lopez. Moreover, for Puerto Lopez whereas estimates based on remote sensed data showed an increase in the area for both agricultural systems, estimates based on agricultural statistic showed a decrease in the area of these systems. These differences can be explained as remote sensed data was annual land cover where several factors such as the length of the growing period, flood, drought, pests and diseases that influence the size of the area annually harvested were not revealed by the annual measurement of land cover. When comparing changes in the area annually harvested per crop we found differences among the two municipalities. Whereas for soy the area annually harvested increased more than 300% in Puerto Lopez, the area harvested in Puerto Gaitán decreased 27%. The importance of oil palm, sugar cane and cattle production increased, as the area annually harvested and grazed increased in the two municipalities. However, there is a particular interest to increase oil palm production in Puerto Gaitán as the area annually harvested increased 350%. Likewise, the importance of sugar cane production has been growing in Puerto Lopez, as the area annually harvested increased 685%. The importance of rice production decreased in the two municipalities as the area annually harvested decreased 19% in Puerto Lopez and 41% in Puerto Gaitán. The area grazed by cattle increased 3% in Puerto Lopez and 4% in Puerto Gaitán (Table 5.3). Monitoring changes in the extent of cropland and pastures, and changes in the area annually harvested per crop over a period of five years can be used to highlight which of the crop is determining changes in land cover and land use. Moreover, the expansion of the agricultural area decline the natural capital of the river basin directly if such expansion is the result of clearing natural systems, and indirectly by increasing the environmental impacts associated to agriculture (e.g. biodiversity decline, eutrophication).

5.3.2 The potential to sequester carbon

To assess the ability of croplands and pastures to sequester carbon we calculated the NEP (Fig 5.3).

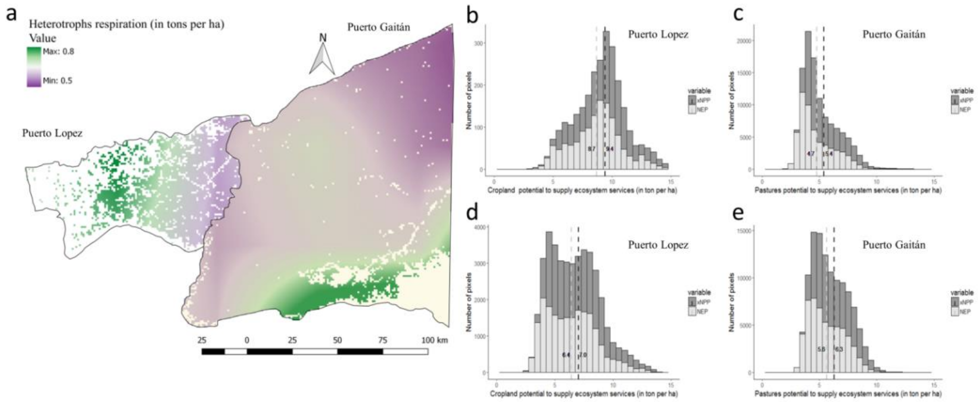

Fig 5.3. In a) the spatial distribution of heterotrophs respiration in 2014. Histograms showing the distribution of NEP and NPP (in tons of carbon per ha) between 2010 and 2014, in b) Puerto Lopez cropland, c) Puerto Lopez pastures, d) Puerto Gaitán cropland, and e) Puerto Gaitán pastures. The dashed lines are the accumulated mean NPP and the accumulated mean NEP between 2010 and 2014

Our results show that cropland sequester more carbon than pastures. Cropland in Puerto Lopez sequester more carbon than cropland in Puerto Gaitán, and there were no large differences between pastures in the two municipalities (Fig 5.3). However, the amount of carbon sequestered by cropland largely varies, showing a tendency for low-medium NEP and NPP values. Moreover, the histogram in Fig 5.3 shows a bimodal distribution in Puerto Gaitán with a tendency for low and middle NEP and NPP values, and a more symmetrical distribution in Puerto Lopez with a tendency for middle values. These symmetrical differences suggests that cropping conditions (e.g. crop varieties, nutrients, (a)biotic stresses, management) among the municipalities differ, influencing the amount of carbon sequestered. Although, there is a tendency for low NEP and NPP values in the pastures, the variation was lower compared to the cropland. This tendency was similar for the pastures in both municipalities, suggesting similar grazing conditions (e.g. grass species, management, nutrients, soil type). To better understand the differences between NEP and NPP, we assessed and mapped the -*heterotrophs respiration*- (Fig 5.3). The NEP was 93% the NPP, meaning that 7% of the NPP is lost by heterotrophs respiration. However, heterotrophs respiration was not homogenously distributed among our study area. Heterotrophs respiration was higher in the center of Puerto Lopez and in the south of Puerto Gaitán compared to low values in the right corners of both municipalities where most pastures are located (Fig 5.3). Biophysical differences such as the photosynthetic capability between food and fodder crops, and other factors such as human management and water stress influence the cropland and the pastures potential to sequester carbon. The use of external inputs such as synthetic fertilizer and high yielding crop varieties that increase agricultural systems ability to sequester carbon might contribute to decline their natural capital as these inputs modify key ecosystem processes such as the water, nitrogen and phosphorus cycles, the supply of ecosystem services such as soil fertility and erosion control, and the availability of energy for other species (Bommarco et al. 2013; Vitousek et al. 1986).

5.3.3 The capability to produce crops

To assess cropland and pastures annual capability of to produce crops we estimated the NPPC for each crop. Our results show changes in the NPPC over time and by location (Fig 5.4).

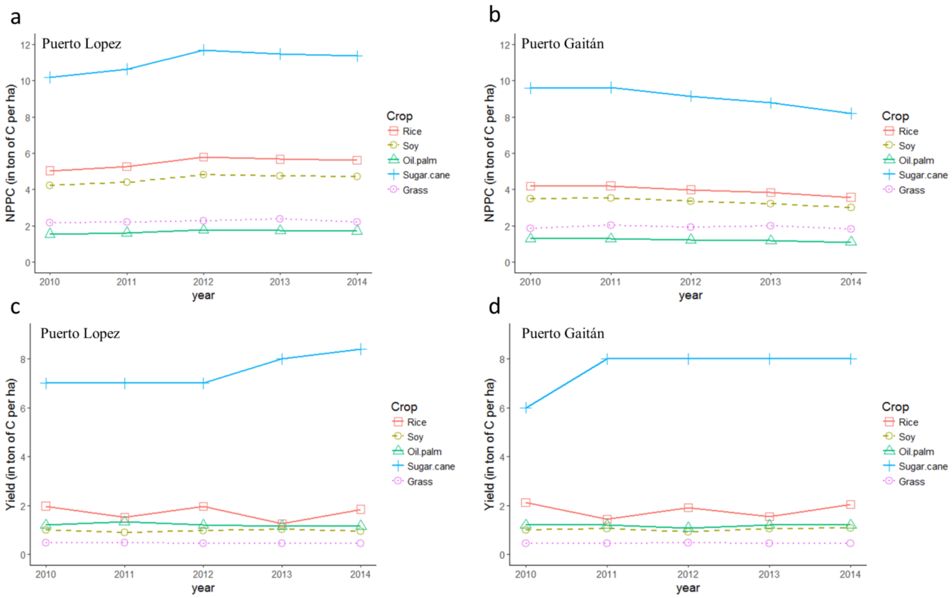

Fig 5.4. Line plots showing changes in the capability to supply crops (NPPC) over time in a) Puerto Lopez and b) Puerto Gaitán and changes in the annual yield for each crop in c) Puerto Lopez and d) Puerto Gaitán

The capability of the pastures to grow grass for grazing cattle remained stable for the five years in the two municipalities. The capability of the cropland to produce rice, soy, oil palm

FFB and sugar cane for harvesting showed an increasing trend in Puerto Lopez (e.g. the NPPC for sugar cane increased 0.8 ton of carbon per hectare in five years). This increase was particularly strong by 2012 compared to 2010, as the NPPC increased on average 0.7 ton of carbon per hectare for all crops. The capability of cropland to produce most of the crops showed a decreasing trend in Puerto Gaitán, (e.g. the NPPC for rice decreased 0.6 ton of carbon per hectare in five years). Our results show that an increase in the NPPC was not reflected in a change in the annual yield. In particular, because the annual yield of all the crops decreased in Puerto Lopes with the exception of sugar cane where an increase of 1.4. ton of carbon per hectare was found. Moreover, the annual yield for all crops remained stable (except for sugar cane that increased in 2011), regardless of the decreasing trend in the NPPC for Puerto Gaitán. The ability of each agricultural system to produce crops vary by location and over time. This variation can be the result of human actions such as among others the introduction of genetic improved crop varieties, irrigation systems, but can also be the result of changes in environmental conditions such as rainfall, light, soil type and temperature that influence the allocation of biomass to plant organs that produce the yield.

5.3.4 Capability analysis

We compare the annual yield for each crop with the cropland and the pastures capability to produce each crop to assess an annual yield level that does not negatively affect agricultural system's capability to produce crops and agricultural system's potential to sequester carbon. The annual yield for each crop cannot exceed the annual agricultural system's capability to produce crops (NPPC) as the total regrowth was annually estimated. However, the difference between NPPC and yield for all the crops was higher in Puerto Lopez than in Puerto Gaitán. This difference was on average 2.2 ton of carbon per hectare for rice, 1.8 for oil palm FFB, 0.6 for sugar cane and 2.1 for grazing grass (Fig 5.5). The annual yield of a certain crop influence agricultural system's potential to sequester carbon by removing photosynthetic organs, and influence climate regulation by harvesting biomass that decrease the amount of carbon accumulated as a carbon stock (NEP). Although both municipalities share characteristics such as altitude, temperature, and soil type, there are differences among crops and by location (Fig 5.5).

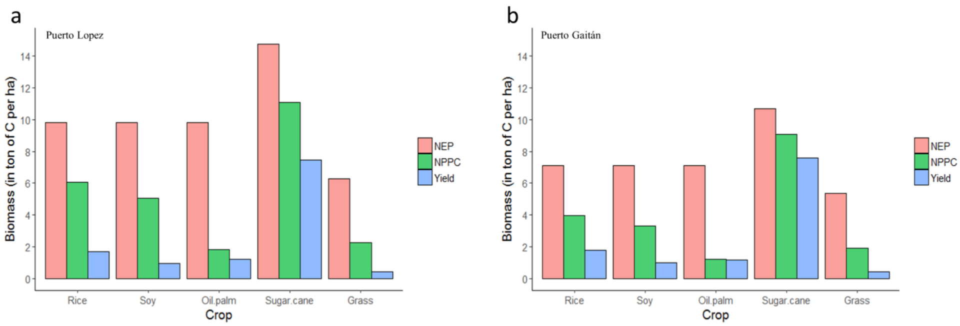

Fig 5.5. Bar plots showing the average yield, NPPC and NEP for different crops between 2010-2014 in a) Puerto Lopez and b) Puerto Gaitán.

The influence on carbon sequestration potential by removing photosynthetic organs is lower for oil palm FFB compared with the other crops as there are differences in the photosynthetic organs removed by harvesting. Whereas harvesting rice, soy, sugar cane and grazing grass removes photosynthetic organs in plant leaf tissues, harvesting oil palm FFB do not directly remove these organs. The influence of annual yield on climate regulation (NEP) by harvesting biomass for rice, soy, oil palm FFB and grazing grass is lower compared to harvesting sugar cane biomass. Whereas harvesting sugar cane decreases the cropland NEP more than 60%, harvesting the other crops decreases the cropland and pastures NEP less than 25%. For example, rice lands accumulated on average 9.5 tons of carbon per ha (NEP) and harvesting rice removed on average 1.8 tons of carbon per ha from this system (Fig 5.5). However, other factors such as ploughing, weeding and pruning might also contribute to decrease the NEP of these systems. Harvesting crops in Puerto Gaitán decreases the NEP on average 70% less than harvesting crops in Puerto Lopez. For example, whereas harvesting sugar cane in Puerto Lopez released on average 7.4 tons of carbon per hectare harvesting this crop in Puerto Gaitán released 5.3 tons of carbon per hectare (Fig 5.5). The length of the growing period of each crop can also influence climate regulation as photosynthetic organs and biomass are removed at different period of time. Whereas for rice and soy both the photosynthetic organs and the biomass are removed twice a year, they are removed by grazing grass every two months, for sugar cane they are removed every 18 months, and for oil palm photosynthetic organs are not removed by harvesting and FFB are harvested once a year. The use of the raw material can also influence climate regulation, as for example sugar cane in Puerto Gaitán is harvested to produce food (e.g. panela which is a Colombian food) and in Puerto Lopez sugar cane is harvested to produce bio-ethanol for the fuel industry (Bioenergy 2017). There are important trade-offs between climate change mitigation and the production of food and biofuel. An increase in the production of food and biofuel will decrease agriculture systems ability to sequester carbon by removing photosynthetic organs and will remove more biomass already stored in these systems. However, if more food should be produced and this food comes from new agricultural land the consequences for climate change will be even worse.

5.3.5 Water productivity

To assess the efficiency with which water is used to produce food or biofuel, we estimated and map the WP. Our results show a positive linear relation between the NPP and ET, where ET explains 55% and 69% of the variation of NPP in cropland (R^2 0.55-0.69), and 50% and 59% in pastures (R^2 0.50-0.59) (Fig 5.6). Areas with a high NPP corresponded to areas with a high ET, such as in oil palm in the north east of Puerto Gaitan (Fig 5.6). Conversely, areas with low NPP corresponded to areas with a low ET, particularly in pastures covering a large portion of both municipalities (Fig 5.6). Our results show that the WP was higher in cropland than in pastures, and higher for Puerto Lopez compared to Puerto Gaitán (Fig 5.6c). On average the cropland gained 0.8 kg of carbon per m^3 of water lost to the atmosphere and the pastures 0.6 kg in Puerto Lopez. Likewise, the cropland gained 0.6 kg of carbon was per m^3 of water lost to the atmosphere and the pastures 0.5 kg in Puerto Gaitán. The seasonal distributiton of the ET can be an influecning factor modulating the NPPC in rainf-fed crops. Annually, two seasons can be distinguished in the Altillanura, a rainy season, and a dry season where rains are scarce and periods of drought are common (Amézquita et al. 2013). Our results show differences in the intensity and in the seasonal distribution of ET.

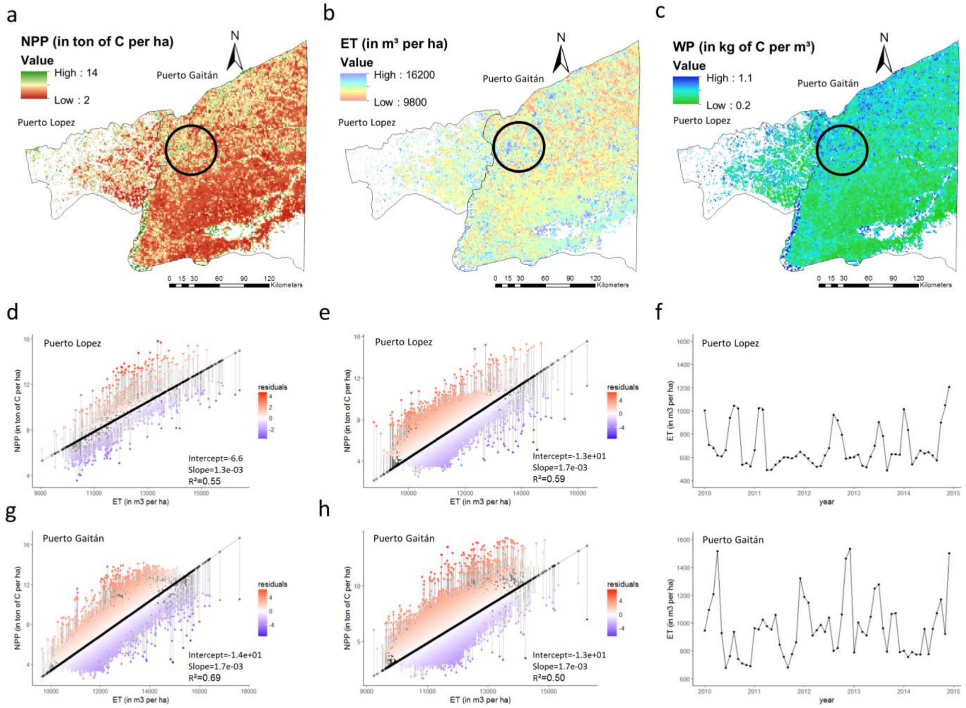

Fig 5.6. Maps showing the spatial distribution of a) NPP, b) ET and c) WP for both municipalities in 2014. The circles indicate oil palm plantations. Scatter plots showing the linear relation between NPP and ET, including the observed, predicted and residual values in d) the cropland, in e) the pastures in Puerto Lopez, and in g) the cropland, in h) the pastures in Puerto Gaitán. In f) the seasonal decomposition of ET showing the monthly distribution of WP for Puerto Lopez and Puerto Gaitán.

Altought, ET shows a bi-modal seasonal pattern in both muncipalities with two non-consecutive periods of high, and two periods of low ET, the intensity of the rainy and the dry season was more severe in Puerto Lopez (Fig. 5.6). Morevoer, the rain is particularly scarce between the last part and the beginning of each year (November-February) intensifying the severity of water stress conditions for both muncipalities. Differences in the intensity and in the monthly variation of the rainy and the dry season, suggests a non-homogeneous distributiton of the ET.

5.4. Discussion

Using ecosystem accounting to assess the contribution of ecosystems to the economy has been demonstrated by several studies (e.g. Remme et al. (2014b), Schröter et al. (2015)), however, using ecosystem accounting concepts to assess the natural capital influencing agricultural production has been rarely explored. Applying an ecosystem approach to agricultural assessment and management is relevant given the importance of ecosystems and their services in supporting agricultural production and the environmental impacts derived from our increasing demand for food and biofuels (DeClerck et al. 2016; Wood et al. 2018). Here we reflect upon i) lessons learned from using ecosystem accounting concepts and remote sensing information the assess agricultutal systems, ii) the applicabiliy of ecosystem accounting to assess agricultural systems

5.4.1 Crop production potential and ecosystem accounting

There is a need to produce sufficient food and biofuels to cover our increasing demand for these products. This endeavor requires to make use of the potential to produce crops on existing faming where crop yield potential, the yield gap and water productivity together determined such potential. Estimating crop yield potential is the basis for yield gap analyses, and yield gap analyses are an entry point to explore factors limiting farm yields, (e.g. soil and management), to select crop varieties and best agricultural practices to narrow yield gaps (van Ittersum et al. 2013). Moreover, yield gap analyses are indispensable to intensify agricultural production, to evaluate the impact of future scenarios of climate change in agriculture, and as inputs to economic models and policy making at local and global levels (Lobell et al. 2009; van Ittersum et al. 2013). In this study we used the yield gap as a conceptual basis to apply a capability analysis linking a crop yield level to the capability of an agricultural system to produce crops and the potential to sequester carbon. The importance of such analysis is to make these links explicit and traceable, even though in the assessment the supply of one ecosystem service is prioritized from the supply of other ecosystem services. This is essential when agricultural systems are analyzed using ecosystem accounting, as these systems prioritize the production of food or biofuels from the supply of other ecosystem services such as flood regulation and carbon sequestration. Moreover, a key opportunity of using ecosystem accounting for the analysis of agricultural systems is emphasizing the importance of those non-prioritized services. Non-prioritized ecosystem services include for example carbon sequestration and the contemplation of agricultural landscapes supplied together with the supply of food and biofuels, and ecosystem services such as pollination, soil formation and pests control that influence agricultural production. Making the supply of these ecosystem services as relevant as the supply of food is essential for the sustainability of agricultural production.

Our study included water productivity as an indicator of water use efficiency for agriculture within a context of water as a limited resource. Water productivity includes plant respiration in a carbon gain assessment measured in terms of NPP, and expressed in tons of carbon per m3 of water lost to the atmosphere. In our study, we did not discriminate the water productivity for specific crops, instead, we assessed the water productivity per land use type, cropland and pastures . Our results show that the average water productivity varies between 0.8 kg of carbon per m3 for the cropland and 0.5 kg of carbon per m3 for pastures. These results are similar to other studies that reported 0.7 kg per m3 for the cropland in China (Cao et al. 2015) and 0.7-1 kg per m3 for pastures in Brazil (Fernandes et al. 2018), assessed as water use efficiency. Although the annual water productivity is a good indicator to monitor the amount of water used to produce food, the spatial distribution of the water productivity is important to identify areas with water stress conditions. Moreover, water stress conditions are more evident during the dry season particularly on February and March, as water productivity is influenced by the seasonal distribution of water. Management is also an important factor influencing water productivity, as suggested by Wang et al. (2018), who found differences in the water productivity for winter wheat and summer maize under conventional irrigation and reduced irrigation. Water productivity is a useful indicator that can be used in ecosystem accounting to reflect changes in the condition of an ecosystem. Particularly because changes in the of availability water influence ecosystem's productivity, soil characteristics, nutrient cycling, biodiversity and vegetation type, moreover water stress conditions such as floods and droughts highly influence the future supply of ecosystem services (e.g. food supply, carbon sequestration). Compiling water productivity information in ecosystem condition accounts can be useful to guide agricultural production in making an efficient use of water and land resources. Recent initiatives such as the global yield gap atlas (van Bussel et al. 2015), and methods such as AquaCrop-OS (Foster et al. 2017), WATPRO (Zwart et al. 2010), and the water productivity score proposed by Bastiaanssen and Steduto (2017), provide valuable spatially explicit information concerning water productivity for different crops and agro-climatic zones contributing to assess a more efficient use of water resources in agriculture. A new approach that focus on resource use efficiency rather than on enhancing crop production, that overview agricultural systems as part of a multifunctional landscape that produce more than food and fibers is essential for a sustainable use of those systems (van Noordwijk and Brussaard 2014). The ecosystem approach used by ecosystem accounting to analyze changes in ecosystems and changes the supply of multiple ecosystem services at multiple scales ranging from multifunctional agricultural landscapes to national and subnational levels is useful to support this shift. Agricultural intensification strategies such as ecological intensification that optimize the use of the natural capital by integrating ecosystem processes and ecosystem services in multifunctional agroecosystems can play an important role in making agricultural production more sustainable (Bommarco et al. 2013; Tittonell 2014).

5.4.2 Applicability of ecosystem accounting for agricultural systems

In this study, we used ecosystem accounting concepts to assess two agricultural systems: cropland and pastures for two municipalities in Colombia. The spatially explicit approach applied in ecosystem accounting enables the spatial analysis of changes in the natural capital of agricultural systems over time, however, three important aspects regarding its applicability should be considered. The applicability of a method is determined by how well the accuracy and feasibility are aligned with the purpose of the method (Bagstad et al. 2018; Schröter et al. 2015). The first aspect is the accuracy of the spatial information used to support ecosystem accounting for the assessment of agricultural systems. All remote sensing products and models include a level of generalization of the reality which influences our system description. Especially relevant in this study are the spatial data and models used to quantify extent and capability of cropland and pastures of the two Colombian municipalities. The assessment of extent of the system is the starting point in ecosystem accounting for which a land cover map is typically used based on the combination of, remote sensed, cadastral, agricultural, transport and environmental data and maps (United Nations et al. 2014b; Weber 2014). This land cover map need to be aligned with the length of the accounting period of ecosystem accounting, which is typically one year. However, national level land cover maps are normally updated every 5 years making a reliable annual land cover map difficult to obtain (Weber 2014). In this study we used remote sensed data from the MODIS MOD12Q1 which is an annual land cover modelled product with an accuracy of 75% on the overall classification (Friedl et al. 2010). The accuracy of this data can be enough for the measurement of extent as a high level of accuracy is not required to cover large homogeneous areas. However, if a high level of accuracy is required (e.g. for local planning) the use of this product is constrained. For the assessment of capability we estimated the water-limited potential yield for rice, soy, oil palm, sugar cane and grass to graze cattle, using the NPP of potential vegetation, NPP allocation models and data derived from the MODIS MOD17A3 product. This product has a reported accuracy of around 80% (Cohen et al. 2003; Pan et al. 2014a; Turner et al. 2006). Using these MODIS NPP values in natural and agricultural areas to calculate water-limited yield resulted in an average 13 ton per hectare for rice, 10 ton per hectare for soy, 4 ton per hectare for oil palm FFB, 101 ton per hectare for sugar cane and 5 ton per hectare for grass to feed cattle. These results can be similar to the results obtained using other methods such as simulation models and field experiments. Dingkuhn et al. (2015)) reported a potential yield for irrigated rice between 5 and 12 ton per hectare in Colombia using a radiation use efficiency simulation model. Valencia and Ligarreto (2010)) reported for soy a potential yield between 1.5 and 4 ton per hectare based on field experiments. Woittiez et al. (2017)) reported for oil palm FFB a water-limited potential yield of 5 ton per hectare using simulation models. Monteiro and Sentelhas (2017)) reported a potential yield between 63 and 237 ton per hectare for irrigated and rain-fed sugar cane in Brazil using simulation models. These differences can be explained by the use of NPPC to estimate the water limited potential yield in rain-fed crops, which is less accurate than crop simulation models, particularly because this method do not assess the interaction of multiple factors such as the age of the cultivar, crop variety, breeding status and plant density among others that could influence the potential yield estimates (Lobell et al. 2009). NPPC derived from MODIS NPP can be an alternative to estimate the potential yield in rain-fed and irrigated crops in data poor contexts as MODIS NPP is globally available, even though NPP is not very highly correlated to yield potential (Lobell et al. 2009).

A second aspect is the availability of agricultural statistics to compare crop yields with the capability of agricultural systems to produce crops. Location-specific agricultural statistics are often non-available, particularly in developing countries. Annual agricultural statistics for time series longer than 5 years are still missing for many crops in the Orinoco river basin. In our study, only the two selected municipalities had agricultural statistics for the period 2010-2014, hindering the assessment for more municipalities in Altillanura. Time series of agricultural statistics over at least 10 years would enable a better understanding of the link between crop yield and changes in the capability of agricultural systems to produce crops, help to assess data quality and the completeness of other variables (Weber 2014). This means that our presented approach, though developed for data-scarce areas, does need agricultural statistics for time series longer than 10 years for more municipalities if a more complete analysis reflecting agricultural system's capability to produce crops covering the entire Altillanura is needed.

A third aspect are the availability of feasible options to include a broader set of regulating ecosystem services such as pollination, erosion control, water regulation and processes such as the carbon, water and nutrient cycling along with current set of provisioning ecosystem services. A broader selection of ecosystem services will make the natural capital and trade-offs influencing agricultural production more visible (TEEB 2018). Including a wide range of ecosystem services from agricultural systems allows for assessments of limits on agricultural economic activities associated to pressures on key earth system processes such as the nitrogen and phosphorous cycles using a quantitative framework as proposed by Rockström et al. (2017). Multiple assessment frameworks can be used to set limits on human economic activities based on earth system processes (Raworth 2012; Vargas et al. 2018). In principle, the criterion for deciding if a spatial model is accurate enough depends on the policy purpose, and consequently spatial models that are too inaccurate or have too low feasibility are not suitable to support ecosystem accounting (Schröter et al. 2015). The above mentioned limitations indicate that the data and models used in this study can be considered a first test case to apply ecosystem accounting to inform policy and decision making aiming to include the contribution of the natural capital in the production of food, fibers and biofuels. Current scientific development in remote sensing and agricultural statistics will reduce these three feasibility challenges. Remote sensing data are a useful alternative when up-to-date spatially explicit crop yield information is not available. New improvements in remote sensing classifications, algorithms, and products based on Landsat 8 and the Sentinel family will provide land cover products with higher spatial and temporal resolution (Forkuor et al. 2018; Pettorelli et al. 2014). The availability of agricultural statistics such as the national agriculture census (Dane and Ministry of Agriculture 2016) and global initiatives such as the World Program for the Census of Agriculture 2020 (FAO 2017) will be of benefit for the feasibility of agricultural systems analysis using ecosystem accounting.

5.5. Conclusion

Concepts from the fields of ecosystem accounting and yield gap analysis, agricultural statistics, and remote sensing were used to assess the natural capital influencing agricultural production. Spatial explicit information derived from the MODIS land cover and net primary productivity products can be used to populate ecosystem accounts. Compiling such information in ecosystem accounting accounts enable the assessment of changes in agricultural systems extent and changes in their ability to supply ecosystem services. The concept of *capability* is not yet used in the SEEA-EEA, however, this concept is key to assess changes in the ability agricultural systems to produce crops following the ecosystem accounting guidelines. Particularly because agricultural systems are amended ecosystems where the production of one ecosystem service such as the supply of food, fodder, biofuels, medicinal plants or fibers is prioritized from the supply of other ecosystem services. Our agricultural system capability analysis describes this prioritized supply using remote sensing and statistic data. The concept of *potential,* although is not yet used in the SEEA-EEA can be used to assess the ability of an agricultural system to supply those non-prioritized ecosystem services such as flood regulation, pollination and carbon sequestration that underpin agricultural production. Water productivity can be used in ecosystem accounting to assess changes in the condition of agricultural systems, as water productivity reflect changes in net primary productivity, soil characteristics, nutrient cycling and the biodiversity of these systems. The accuracy and the spatial resolution of MODIS products are important factors to consider when policy purposes such as local land use planning require detailed information with a high level of accuracy and a fine spatial resolution. Assessing changes in the natural capital of agricultural systems using ecosystem accounting is an opportunity to increase the production of food and biofuels without declining the supply of key ecosystem services such as carbon sequestration and to maintain agricultural systems in a healthy state .

Chapter 6

Synthesis

6.1 Objectives and main findings

The objective of this thesis was to increase our knowledge of how remotely sensed data can be used to support ecosystem accounting in the assessment of changes in ecosystems in a large river basin. To achieve this objective, four research sub-objectives were formulated:

1. To examine if and how ecosystems can be analysed at a large scale with the use of information provided by remote sensing. (Chapter 2);
2. To analyse how remote sensing spectral information can be used to support the assessment of the capacity of ecosystems to supply ecosystem services for large areas (Chapter 3);
3. To examine if and how the planetary boundaries framework can be used in combination with ecosystem accounting for sustainable natural resource management at the level of a large river basin (Chapter 4);
4. To explore if and how concepts used in ecosystem accounting and yield gap analysis can be used to assess changes in the natural capital of agricultural systems (Chapter 5).

These sub-objectives were addressed in 4 scientific papers presented in Chapters 2-5. An overview of the research presented in this thesis is shown in Table 6.1, including the main findings from the different chapters.

Table 6.1 Overview of the research presented in this thesis. The table shows the corresponding chapters, the main output and the main findings

Chapter	Main output	Main findings
2	Maps and tables concerning changes in ecosystem extent, condition and capacity to supply ecosystem services	• MODIS land cover, vegetation indices and land surface temperature products provide useful spatially explicit information to support ecosystem accounting in the assessment of changes in the extent, condition and capacity to supply ecosystem services
3	Modelled changes in ecosystem extent, and capacity to supply ecosystem services. Modelled changes in the supply of ecosystem services	• MODIS NPP spatially explicit information support ecosystem accounting in the assessment of changes in the capacity of ecosystems to supply ecosystem services • Using a NPP allocation approach to model the capacity of ecosystems to supply ecosystem services is useful to establish trade-offs between the supply of rival ecosystem services • Using a spatially explicit approach in capacity-supply models highlights specific locations where ecosystem services are unsustainably supplied above its capacity
4	Overview on the complementary between both frameworks and the applicability for adaptive natural resource management	• Ecosystem accounting can be used to monitor changes in the nitrogen, phosphorus, carbon and water cycles at national level • Improving natural resource management to avoid critical transitions in ecosystems embedded in complex social ecological systems requires a combination of integrated approaches and quantitative frameworks

5	Maps, tables and figures about changes in agricultural systems extent, potential and capability to supply ecosystem services Yield gap figures Maps showing changes in water productivity	• The ecosystem accounting concepts of potential, and capability to supply ecosystem services are useful to monitor changes in the natural capital of agricultural systems • Ecosystem accounting and yield gap analysis provide spatially explicit information key to guide a sustainable use of agricultural systems

This final chapter consists of five sections. Section 6.2 reflects on the implications of using the MODIS land surface products and other remote sensing sensors to assess changes in extent and condition of ecosystems following ecosystem accounting concepts and guidelines. Section 6.3 analyses the implications of using the MODIS NPP product to assess changes in the capacity of ecosystems to supply ecosystem services. Section 6.4 describes the implications of combining integrated approaches to assess changes in ecosystems to avoid critical transitions in complex social-ecological systems, focussing on the planetary boundaries and ecosystem accounting frameworks. Section 6.5, assesses the implications of using the ecosystem accounting and yield gap analysis to assess the natural capital of agricultural systems. In Section 6.6, I present overall conclusions and recommendations arisen from this thesis.

6.2 Using remote sensing for the analysis of ecosystem change

Assessing and mapping the supply of ecosystem services has been an active field of research during the last two decades (Burkhard and Maes 2017; Willemen et al. 2015). However, mapping changes in ecosystems (e.g. ecological processes, structure, extent) and the consequences of these changes in the supply of ecosystem services is challenging (Lavorel et al. 2017; Maes et al. 2012), particularly, given the scale of data gathering and analysis required for a complete and systemic assessment of changes in ecosystems and supply in ecosystem services (Mace et al. 2015). This thesis followed the ecosystem accounting guidelines to assess changes in ecosystems at the scale of river basin. Remote sensing indices such as the Enhanced Vegetation Index (EVI), Normalized Difference Water Index (NDWI), and Land Surface Temperature (LST), were used to assess changes in the condition of ecosystems, particularly, changes in photosynthetic activity, canopy water status and top canopy temperature. Net Primary Productivity (NPP) was used to assess changes in ecosystems capacity to supply ecosystem services which will be discussed in section 6.3. These indices were derived from MODIS land surface products (e.g. land cover, vegetation index and surface reflectance). The MODIS land surface products are the outcome of modelling the results of lower-level data and they are produced by the NASA EOSDIS Land Processes Distributed Active Archive Centre (LP DAAC). This thesis shows that the MODIS land surface products compiled following the ecosystem accounting guidelines are useful to analyse changes in ecosystems, in terms of extent, condition and capacity to supply ecosystem services (discussed in section 6.3). However, several factors should be considered.

First, land cover is key for detailed monitoring of changes in ecosystems, as changes in land cover influence the ecological properties (e.g. ecological processes, structure, biodiversity) of ecosystems that underpin the supply of multiple ecosystem services (Burkhard et al. 2012; De Groot et al. 2010). Land cover maps at national, continental and global level are derived from remote sensing, because a direct field inventory of the spatial organization of the natural elements at such levels is not feasible (Gong et al. 2013; Schulp and Alkemade 2011). There are, however, uncertainties and inherent error propagation in using remotely sensed land cover products. Remotely sensed land cover maps are uncertain for outlining the shape and location of objects, and for translating the reflectance signature of remote sensing images into a classified land cover class (Giri et al. 2005; Herold et al. 2008). Dong et al. (2015) pointed out that these uncertainties propagate mapping errors, and consequently, the accuracy and reliability of the spatial models that estimate changes in ecosystems and in the supply of ecosystem services decrease. Accordingly, international initiatives such as the Group of Earth Observation (GEO), and the Global Climate Observation System (GCOS) developed high quality land cover maps such as the GLC2000 (Bartholomé and Belward 2005), the GlovCover (Arino et al. 2007), and the UMD Land Cover to reduce these uncertainties increasing the quality of global land cover datasets. These global land cover maps provided land cover information every 5 years from 2000 until 2010, using different satellites (e.g. MERIS, SPOT4 and NOAA), reporting an accuracy of around 70-75% correctly classified (Herold et al. 2016). The MODIS land cover product used in this thesis provided annual land cover data from 2001 until 2014 with a reported overall accuracy of around 75% correctly classified (Friedl et al. 2010). The annual periodicity of the MODIS land cover product is key for ecosystem accounting given that the length of the accounting period in ecosystem accounting is often one year. However, the spatial resolution of the MODIS land cover product ($500m^2$) is relatively coarse for a detailed monitoring of changes in small-sized ecosystems. Recent land cover initiatives such as the global forest change database and the Copernicus Global Land Operations reported overall an accuracy of around 82% and 74% correctly classified at a 30m and a 100m resolution respectively (Hansen et al. 2013; Smets et al. 2017). These land cover products at a finer spatial resolution are an opportunity to enhance the assessment of changes in ecosystems increasing the level of detail.

Second, remote sensing is a key primary source of spatially explicit information to assess changes in ecosystem condition, where condition reflects the overall quality of an ecosystem in terms of its characteristics (e.g. biodiversity, water, nutrients, vegetation type) (Pettorelli et al. 2014; Smith et al. 2014; United Nations et al. 2014b). Vegetation indices such as the EVI and the Normalized Vegetation difference index (NDVI) are useful to monitor changes in ecosystem condition, in particular, changes in photosynthetic activity, and seasonality (Ivits et al. 2013; Ma et al. 2014; Rasmus et al. 2015). LST and NDWI can be used to monitor water stress conditions in ecosystems, complementing vegetation indices that saturate at high values particularly in forested ecosystems (Pérez-Hoyos et al. 2014). These indices, however, are insufficient to assess the overall condition of ecosystems which requires the assessment of other relevant characteristics such as, species diversity, structure, and nutrients availability. High spectral resolution and hyperspectral sensors such as Airborne Visible Infrared Imaging Spectrometer (AVIRIS) are useful in detecting biochemical and structural changes in specific vegetation types, enabling the assessment of physiological states, species diversity, nitrogen content, health status and diseases. High spatial resolution sensors on board satellites such as IKONOS and Quickbird are useful to map vegetation at species level and tree canopies. Vegetation structure, height and functional attributes of the understory species can be obtained with active remote sensing such as the Light Detection And Ranging of Laser Imaging Detection (LiDAR) and Radio Detection And Ranging (RADAR). Nevertheless, the dimensions of each scene and the temporal resolution in this type of sensors are low compared to moderate resolution sensors limiting their use in ecosystem accounting if a large area such as a river basin is annually monitored.

The development of moderate resolution image spectroradiometers from MODIS and MERIS, increased the potential to globally map weekly, monthly and annual changes in ecosystem condition with narrow bands in specific segments of the electromagnetic spectrum that allow the assessment of variations in vegetation biophysical, physiological and structural quantities. With MERIS no longer operational and MODIS getting closer to the end of its lifespan these developments will continue with the launching of Sentinel 3 and the Visible Infrared Imaging Radiometer Suite (VIIRS) currently on the Suomi National Polar-orbiting Partnership (NPP), as one of several instruments on board the Joint Polar Satellite System (JPSS) (Houborg et al. 2015). The land surface products derived from the MODIS and the forthcoming products derived from the VIIRS are appropriate to assess changes in ecosystems following the ecosystem accounting guidelines. Particularly, because policy and decision making informed by ecosystem accounting require spatially explicit information without a high level of detail, aggregated at national and sub-national levels, and covering accounting periods not longer than one year.

6.3 Assessing the capacity of ecosystems to supply ecosystem services

Assessing the capacity of ecosystems to supply multiple ecosystem services using a spatially explicit approach is important to understand how and where changes in ecosystems influence the supply of ecosystem services. Understanding the link between capacity and supply of ecosystem services is fundamental to monitor the management and use of ecosystems, to develop and evaluate alternative uses of ecosystems, and to assess ecosystem degradation. In this thesis, remotely sensed data from the MODIS NPP product was used to model the capacity of six different ecosystems (e.g. forests and savannahs) to supply four selected ecosystem services (e.g. carbon sequestration and timber). National statistics were used to model the supply of the selected ecosystem services (e.g. pastures to graze cattle and oil palm fresh fruit bunches). This thesis shows that MODIS NPP spatially explicit information can be compiled following the ecosystem accounting guidelines and NPP can be used to monitor annual changes in the capacity of ecosystems to supply ecosystem services at the level of river basin. Particularly because changes in the productivity of ecosystems reflect changes in relevant characteristics (e.g. water and nutrients) that influence the amount of aboveground biomass available to supply provisioning ecosystem services. Using a spatially explicit approach in capacity-supply models is useful to highlight large areas or specific locations where ecosystem services are unsustainably supplied above their capacity. Capacity-supply models can be applied at a large scale, as for example to assess if forest ecosystem capacity to supply timber measured at the scale of the whole river basin exceed the harvest of timber biomass. However, an unsustainable use of forest ecosystem can take place when the harvest of timber in dedicated forest occur in patches where in the harvest year, extraction exceeds regrowth. There are important limitations in using remote sensing information from MODIS NPP to monitor changes in the capacity of ecosystems to supply ecosystem services.

First, although MODIS is the only source of NPP data publicly available at a global level with accuracy and quality assessed (Zhao et al. 2006a), the spatial resolution of this product can be relatively coarse for local detailed assessments. Moreover, the spatial resolution of MODIS NPP attenuates the spatial variation of NPP at such level, as NPP tend to saturate at very high and very low values (Turner et al. 2004). In addition, uncertainties in the estimation of autotrophs respiration can represent a limiting factor for its accurate representation in accurate carbon-use efficiency models (He et al. 2018; Robinson et al. 2018). In spite of these limitations and uncertainties, MODIS NPP data has been used for capacity-supply, energy-based sustainability assessments. For example, Smith et al. (2012) assessed the capacity of different biomes to sustainably support the supply of bioenergy. Furthermore, MODIS NPP data has been used to assess the capacity of agricultural systems to supply enough food to meet potential future demand (Sallaba et al. 2017), and to quantify the amount of earth system productive capacity derived to support human activities (Haberl et al. 2014).

Second, although the concept of capacity links ecosystems and the supply of ecosystem services, the concept of capacity was not included in the SEEA-EEA framework (United Nations et al. 2017). The main reason was that measuring the link between ecosystems and the supply of ecosystem services is challenging, as changes in the overall condition of ecosystems and the individual supply of ecosystems include notions of resilience, non-linear reactions and ecological thresholds (United Nations et al. 2017). Nevertheless, the high sensitivity of NPP to reflect changes in environmental conditions such as rainfall patterns, water, light and nutrients availability (Knapp et al. 2014c), make NPP a useful indicator to assess changes in the capacity of ecosystems to supply ecosystem services. Costanza et al. (1997) suggested that NPP can be used as an indicator to monitor changes in ecosystem condition, in the supply of ecosystem services, and as a proxy to estimate the value of ecosystem services based on the strong dependence of animal food webs on plant productivity (Costanza et al. 2006). Pan et al. (2014b) used NPP to establish trade-offs between ecosystem services, and Zhang et al. (2017) as a surrogate indicator to map ecosystem services. Egoh et al. (2008) highlighted the high correlation between NPP and the supply of ecosystem services, but, suggesting caution in using NPP as an ecosystem services surrogate. Although NPP is sensitive to changes in ecosystems condition and is highly correlated with the supply of provisioning ecosystem services, NPP is limited in assessing ecosystem services non-directly related to primary productivity particularly for regulating and cultural ecosystem services. Moreover, NPP is limited in assessing the overall capacity of ecosystems to supply ecosystem services, which entail notions of non-linear relations, resilience, and ecological thresholds.

Third, spatial heterogeneity is an important factor to consider in the assessment of ecosystems capacity to supply ecosystem services. Particularly because land cover heterogeneity directly affects the capacity to supply ecosystem services through ecosystem functions (e.g. nutrient retention), and indirectly through biodiversity (Emilie et al. 2015; Lovett et al. 2005; Mitchell et al. 2015). According to Schröter et al. (2015) the accuracy of spatially explicit capacity-supply models decreases with spatial heterogeneity. Moreover, the influence of spatial heterogeneity in mapping the capacity of ecosystems to supply ecosystem services is stronger at landscape compared to the national level (Verhagen et al. 2016). Landscape configuration is particularly important for mapping capacity, as configuration change with the spatial resolution of the analysis, and local effects of configuration largely average out at large national, sub-national level (Verhagen et al. 2016). Moreover, landscape configuration is not considered when the MODIS NPP product is globally validated, instead, land cover type -*landscape composition*- is used to match the 1 km resolution of this product with plot scale measurements on the ground (e.g. eddy covariance towers)(Turner et al. 2006). Using MODIS NPP to assess ecosystems capacity to supply ecosystem services at a local level can be constrained by spatial heterogeneity, particularly by the spatial arrangement of land cover types -*landscape configuration*-.

6.4 Socio-ecological frameworks to assess ecosystems change

Social-ecological systems are complex adaptive systems where the interaction between its components is often unplanned and unpredictable, with a potential for non-linear feedbacks, chaotic dynamics and irreversible shifts (Lenton et al. 2008; Levin et al. 2013; Scheffer et al. 2001). Multidisciplinary integrated approaches that use quantitative frameworks to dissect the complexity of these systems are necessary to analyze their sustainability (Levin et al. 2013; Liu et al. 2015; Ostrom 2009). Planetary boundaries and ecosystem accounting are among the few integrated approaches that use quantitative frameworks to compile environmental information that can be used to reconcile economic development with sustainable natural resource management. In this thesis, two sets of criteria based on Binder et al. (2013) were used to compare planetary boundaries and ecosystem accounting quantitative frameworks. The comparison evaluated if similarities and differences between these two frameworks provide complementary information for sustainable natural resource management in the Orinoco river basin. Additionally, adaptive management components described by Rist et al. (2013a) were used to compare the applicability of both frameworks. This thesis shows that whereas the planetary boundaries framework facilitates the display of spatially heterogeneous interconnected processes (e.g. the carbon, nitrogen and water cycles) as national issues, ecosystem accounting allows the incorporation of these issues into information systems that support national policies. Moreover, setting limits on economic activities based on boundaries associated with the nitrogen and phosphorus cycles, water use and land system change, and supported by information compiled in ecosystem accounting can be a promising approach for sustainable natural resource management at the level of river basin. However, shifting the traditional approach of ecosystems management into a new approach that includes the assessment of boundaries associated with ecological thresholds aiming to avoid critical transitions in ecosystems will require a greater degree of integration between these integrated approaches. According to Liu et al. (2015) because integrated approaches have been studied in isolation even though they are interconnected through human activities, a greater degree of integration will provide broad implications on management, policy and sustainability. For example, the planetary boundaries approach provide an overview of the social-ecological system as a complex dynamic system that includes life-supporting systems and the biosphere as the foundations of the economy, society, and the human dimension as a whole, defining the stability and resilience of the biosphere based on limits for human activities (Folke et al. 2016; Rockström et al. 2009; Steffen et al. 2015b). However, the achievement of the environmental goals proposed by the planetary boundaries approach will require the integration of this approach in a larger set of sustainable development objectives (Raworth 2012). A single framework cannot be used to address all issues of complex socio-ecological systems and hence the right framework has to be chosen based on the problem to be studied and on how the socio-ecological system is conceptualized (Binder et al. 2013). A greater degree of integration between approaches is not only a further step to better understand the complexity of socio-ecological systems, but it is also essential to inform policy and decision making. This thesis shows that combining the planetary boundaries and ecosystem accounting frameworks is useful to inform natural resource management, however, there are many integrated frameworks that can be a useful complement. Social foundations such as food security, gender equality and healthcare can be used to complement planetary boundaries. Moreover, combining the millennium development goals with planetary boundaries can be useful to assure the stability of our earth systems (Griggs et al. 2013; Raworth 2012).

6.5 Using ecosystem accounting concepts to assess changes in the natural capital of agricultural systems

Global crop production should double by 2050 to meet our future demand for food and biofuels (Foley et al. 2011; Tilman et al. 2011). To achieve this, it is necessary to produce more crops in existing farmlands, particularly in under yielding farm lands in developing countries. Narrowing the gap *–the yield gap-* between the hypothetical maximum yield achieved by a crop under optimal conditions and what farmers' obtain in the field is necessary to increase crop production. However, intensifying the use of pesticides and synthetic fertilizer to narrow the yield gap may degrade the natural capital on which agricultural production depends (Bommarco et al. 2013; Pretty and Bharucha 2014). Using natural capital assessments to guide a sustainable agricultural production is essential because such assessments make the natural capital influencing agricultural production more visible, avoiding its further decline (TEEB 2018; Tittonell 2014). In this thesis, I explored if and how ecosystem accounting and yield gap analysis can be used to assess changes in the natural capital influencing agricultural production. I used remote sensing, agricultural statistics, the ecosystem accounting concepts of extent, potential and capability, and the concepts of yield gap and water productivity used in yield gap analysis. Whereas the concept of extent reflects the expansion of agricultural areas, water productivity reveals areas with water stress conditions. Monitoring the expansion of agricultural areas and areas with water stress conditions can support the achievement of both food security and environmental sustainability. Particularly because achieving food security and environmental sustainability depends on stopping the expansion of agricultural areas into natural ecosystems and reducing water withdrawals in areas where water has competing demands (Foley et al. 2011). The expansion of agricultural areas is particularly sensitive in the tropics where it is estimated that 80% of new croplands replaced tropical forests (Foley et al. 2011; Laurance et al. 2014).

Without increasing productivity gains per cubic meter of water used for agriculture, the additional fresh water withdrawals will account for 5,600 km^3 per year by 2050 which is three times the global use of water for irrigation (FAO 2015; Rockström and Barron 2007).

The concept of capability reflects the ability of an agricultural system to produce crops. This concept allows the assessment of agricultural systems where the production of crops and biofuels is prioritized from the supply of other ecosystem services. The ecosystem services not prioritized by these systems include the supply of nutrients from the soil, soil formation, pollination, flood regulation, and other ecosystem services that highly influence crop production but are not as evident as the supply of food and biofuels. The yield gap was the basis to analyze capability, linking the annual harvest of crops with the capability of an agricultural system to produce crops. Capability analysis is essential to establish trade-offs between the supply of food, biofuels and fibers and those ecosystem services not prioritized such as carbon sequestration and pest regulation. This thesis shows the relevance of these concepts to assess changes in the natural capital influencing agricultural production. However, two aspects should be considered regarding the accuracy of the spatial models used to support ecosystem accounting. First, the accuracy of the MODIS land cover data (75% on the overall classification (Friedl et al. 2010)) can be enough for the measurement of extent as a high level of accuracy is not required to cover large homogeneous areas. However, if a high level of accuracy is required the use of this product can be constrained. Second, the use of remotely sensed data and a simplified spatial model to estimate capability is less accurate than crop simulation models, because crop models assess the interaction of multiple factors (e.g. the age of the cultivar, crop variety, crop density). Although using remotely sensed data and simplified models can be a feasible alternative to estimate capability in data-poor contexts, accurate assessments require process-based models which simulate multiple processes that influence crop production.

Ecosystem accounting is a powerful tool to monitor changes in natural capital and to inform policy and decision making, supporting the transition of agricultural systems towards sustainability. The distinction between stocks and flows described in ecosystem accounting reflect the status of the maintenance and degradation of agricultural systems on one side, and the supply of ecosystem services on the other. Condition accounts can be useful to assess the health of agricultural systems by monitoring changes in among others, soil and water quality, vegetation type (e.g. exotic plants), photosynthetic activity and nutritional status. Ecosystem services supply accounts are useful to assess changes in the flow of ecosystem services from agricultural systems, recording not only the production of food and biofuels, but the supply of ecosystem services underpinning agricultural production such as carbon sequestration, erosion control, water supply and pollination. Capacity accounts reflect the ability of agricultural systems to supply ecosystem services as a function of their size and condition, where the supply of ecosystem services can be lower, equal to or higher than their capacity. These three accounts can be used to support the transition of agricultural systems towards sustainability, establishing sustainable levels of supply where any decline in the natural capital of agriculture systems should be considered as unsustainable.

6.6 Conclusions

This thesis provides original and detailed insights for the assessment of changes in ecosystems compiling remotely sensed data and statistics following the ecosystem accounting guidelines. This thesis also enhances the understanding of the integration of quantitative frameworks developed to understand complex social-ecological systems. In addition, this thesis demonstrates the advantages of using an ecosystem accounting approach to analyse the natural capital underpinning agricultural production. In particular, this thesis shows that:

1. Remotely sensed data from MODIS is a useful source of information to support ecosystem accounting in the spatial measurement of changes in the extent and condition of ecosystems. Particularly at the level of river basin where extensive field measurements would simply be too costly to populate the accounts. The MODIS land cover product is particularly useful to measure changes in the extent of an ecosystem in line with the annual periodicity of ecosystem accounting. Remote sensing indicators (e.g. EVI, NDWI and LST) derived from MODIS land surface products are suitable to detect changes in the condition of ecosystems, in particular, photosynthetic activity, vegetation type and water status. These MODIS products constitute a consistent source of spatially explicit information ready to be used in ecosystem accounting. However, policy purposes that require a high level of detail and accuracy should use these products in combination with high spatial resolution remote sensed data (e.g. Landsat, sentinel, quick bird) and ground-truthing to validate their accuracy.
2. MODIS NPP is a powerful source of information to assess the capacity of ecosystems to supply ecosystem services, particularly the supply of provisioning ecosystem services and carbon sequestration. NPP is as a suitable indicator to assess capacity, as NPP is sensitive to changes in ecosystem conditions such as rainfall patterns, vegetation type and photosynthetic activity that affects the supply of provisioning ecosystem services and carbon sequestration. Assessing the supply of aboveground biomass provide insights about ecosystem regeneration patterns, where NPP allocation is key to link the supply of aboveground biomass with a specific ecosystem service. Using NPP to assess capacity depends on the accuracy and availability of NPP allocation models, as not all biomass is used to provide ecosystem services.
3. Ecosystem accounting can be used to support the translation of planetary boundaries into indicators that can be used to monitor complex spatially interconnected processes such as the nitrogen and phosphorus cycles, water use, and land system change at national level. Although the planetary boundaries and the ecosystem accounting frameworks pursue different purposes, supporting the achievement of sustainable development can be seen as a common ground between the two frameworks.
4. The spatially explicit approach used in ecosystem accounting is useful to identify areas where the potential to produce crops can be realized given the increasing scarcity of land and water resources for agriculture, and the increasing environmental impacts derived from this activity. Ecosystem accounting enables monitoring

changes in the extension of agricultural systems and in the use of water by these systems, supporting food security and environmental sustainability assessments. The concept of capability is useful to monitor changes in the natural capital underpinning agricultural production. Monitoring changes in the natural capital underpinning agricultural production using an ecosystem accounting approach is key to inform policy and decision making in supporting the transition of agricultural systems towards sustainability.

Based on the results of this thesis, four main recommendations are made to support the assessment of changes in ecosystems using the ecosystem accounting approach.

(I) PROMOTE THE USE OF REMOTE SENSED DATA TO DELINEATE SPATIAL AREAS FOR ECOSYSTEM ACCOUNTING

In this thesis, I have shown how remotely sensed data can be used to delineate spatial units, particularly the MODIS land cover products, vegetation indices, evapotranspiration and net primary productivity. These products are free and globally available. Meteorological and high-resolution data from the Landsat, Sentinel, and the Tropical Rainfall Measuring Mission (TRMM) is also globally available and can be obtained for free. However, uncertainty on the spatial interpretation and the need for validation and ground-truthing challenge the use of remotely sensed data for ecosystem accounting (United Nations et al. 2017). Because the uncertainty on the delineation of spatial units can be propagated throughout all the assessments that depend on these units, information about the spatial interpretation, validation and ground truthing of remote sensed data should be reliable and always available. Remotely sensed data obtained with a high level of processing where missing data is interpolated and variables are derived from the instrument, and where information about pixel quality, validation and ground-truthing is available is highly desirable. Besides the MODIS products used in this thesis, there are other MODIS products such as the atmosphere (e.g. aerosol product, atmospheric profiles), cryosphere (e.g. sea Ice and Ice surface temperature) and ocean products (e.g. sea surface temperature) that can be useful to incorporate other ecological areas for accounting such as the atmosphere and the sea. Land cover initiatives such as the global forest change database and the Copernicus Global Land Operations provide high-quality data at a 30m and a 100m resolution and are an opportunity to enhance the definition of spatial units for ecosystem accounting. Additionally, the development of a National Spatial Data Infrastructure (NSDI) where a common spatial projection, coordinate system, reference grid, minimum size of contiguous areas, and layers integrating spatial information (e.g. official boundaries, topography, land cover data), and information about spatial interpretation, validation and ground-truthing can be stored, will improve the quality of the spatial units for ecosystem accounting.

(II) PROMOTE RESEARCH ON NPP ALLOCATION MODELS TO ENABLE THE USE OF REMOTELY SENSED NPP IN ECOSYSTEM SERVICES CAPACITY-SUPPLY MODELS

This thesis has shown that the NPP derived from MODIS can be used in ecosystem accounting to assess the capacity of an ecosystem to supply provisioning ecosystem services at large scale such as the Orinoco river basin. In this thesis, I used NPP allocation models to evaluate the amount of carbon allocated to specific plant organs that produce the biomass to be harvested as raw material utilized to produce food, fibres or biofuels. These models are important as not all the carbon sequestered by the plant is harvested or extracted. There is a need to increase our understanding of how NPP is allocated above and below ground at the level of ecosystem, and how NPP is partitioned in the different organs of the plant. This knowledge is required for modelling the distribution of NPP in the different organs of the plant, this is relevant to improve the assessment of the capacity of an ecosystem to supply ecosystem services using NPP. Another important aspect that should be considered when downscaling MODIS NPP to assess capacity at lower levels of aggregation such as landscape level is the spatial heterogeneity. Particularly because the influence of landscape configuration averaged out at large aggregated scale such as national and sub-national level. Further research is needed to assess the influence of landscape configuration in the supply and capacity of provisioning ecosystem services. Nevertheless, the approach used in this thesis for the assessment of the capacity of ecosystems to supply ecosystem services can be used as a basis for further development of capacity-supply models for ecosystem accounting and other applications.

(III) PROMOTE ECOSYSTEM ACCOUNTING AS A SOURCE OF INFORMATION FOR NATURAL RESOURCE MANAGEMENT IN THE ORINOCO RIVER BASIN

The traditional dominance and control perspective of natural resources management where ecosystems responses to human use are assumed to be linear, controllable and predictable, where the human and the natural system are assessed individually has been applied in the Orinoco river basin since a long time. This perspective is embodied in the current social and economic development policy CONPES (2014). The CONPES (2014) asserts, on the one hand, the suitability of 3 millions of hectares to expand agricultural activities, and on the other proclaim the importance of the ecosystem services, biodiversity and water resources supplied and regulated by the natural ecosystems (e.g. savannahs and forests) in the area. However, human and the natural systems are assessed individually and natural ecosystems responses to agricultural activities are ignored. This is an opportunity to bring a new dynamic and adaptive perspective for natural resource and ecosystem management. This new perspective overview the human and the natural system as a coupled social-ecological system subject to complex non-linear dynamics where the notions of resilience and ecological thresholds are fundamental. The development of quantitative frameworks used by integrated approaches such as the planetary boundaries and the ecosystem accounting are indispensable to understand the complexity of these systems. These frameworks provide essential information to monitor changes in ecosystems and in the spatial interconnected processes such the nitrogen, phosphorus, carbon and water cycles, associated with human activities. Using these frameworks together is an opportunity to translate complex spatially interconnected processes into indicators that can be monitored at the national level. Monitoring changes in these processes are key to anticipate critical transitions in the natural ecosystems of the Orinoco river basin. However, more information on the measurement of flows of nitrogen, phosphorus, annual deforestation and water withdrawals for agriculture is needed to define ecological thresholds associated with the cycles of nitrogen, phosphorus, water and land system change in the Orinoco river basin.

(IV) PROMOTE THE USE OF ECOSYSTEM ACCOUNTING TO ASSESS CHANGES IN THE NATURAL CAPITAL OF AGRICULTURAL SYSTEMS

The increasing use of external inputs to boost agricultural production leads, in some cases, to the degradation of the natural capital on which agricultural systems depend and increases the environmental impacts associated with this activity. Yet, maintaining the natural capital underpinning agricultural production while providing more food, biofuels and fibers to meet the needs of an increasing global population is a clear challenge for policy makers (Godfray et al. 2010). The development of ecosystem accounting is an opportunity to organize, monitor and disseminate environmental and economic information required to assess changes in the natural capital of agricultural systems. However, incorporating the concept of capability in ecosystem accounting is necessary to assess agricultural systems, as these systems prioritize the supply of food, medicinal plants and biofuels irrespective of the supply of other ecosystem services such as pollination and pests control that underpin agricultural production. Further research is needed to assess the ability of agricultural systems to supply among others, erosion control, water supply, carbon sequestration, nutrient cycling and soil formation, key ecosystem services that underpin agricultural production. Assessing changes in the supply of these ecosystem services is essential to avoid a further decline in the natural capital underpinning agricultural production. The analysis of trade-offs between these ecosystem services and the supply of provisioning ecosystem services is key to support the transition of agricultural systems towards sustainability.

Bibliography

Amézquita E, Rao IM, Rivera M, Corrales II, Bernal JH (2013) Sistemas Agropastoriles: Un Enfoque Integrado para el Manejo Sostenible de Oxisoles de los Llanos Orientales de Colombia. Centro Internacional de Agricultura Tropical (CIAT), Cali

Andrew ME, Wulder MA, Nelson TA (2014) Potential contributions of remote sensing to ecosystem service assessments. Progress in Physical Geography 38:328-353 doi:10.1177/0309133314528942

Aragão L, Malhi Y, Metcalfe D, Silva-Espejo J, Jiménez E, Navarrete D, Almeida S, Costa A, Salinas N, Phillips O (2009a) Above- and below-ground net primary productivity across ten Amazonian forests on contrasting soils. Biogeosciences 6:2759-2778

Aragão L, Malhi Y, Metcalfe D, Silva-Espejo J, Jiménez E, Navarrete D, Almeida S, Costa A, Salinas N, Phillips O (2009b) Above-and below-ground net primary productivity across ten Amazonian forests on contrasting soils. Biogeosciences 6:2759-2778

Arino O, Gross D, Ranera F, Leroy M, Bicheron P, Brockman C, Defourny P, Vancutsem C, Achard F, Durieux L, Bourg L, Latham J, Gregorio AD, Witt R, Herold M, Sambale J, Plummer S, Weber JL GlobCover: ESA service for global land cover from MERIS. In: 2007 IEEE International Geoscience and Remote Sensing Symposium, 23-28 July 2007 2007. pp 2412-2415. doi:10.1109/igarss.2007.4423328

Armenteras D, Cabrera E, Rodríguez N, Retana J (2013) National and regional determinants of tropical deforestation in Colombia.1-13 doi:10.1007/s10113-013-0433-7

Armitage D, Alexander S, Andrachuk M, Berdej S, Dyck T, Nayak PK, Pittman J, Rathwell K (2015) Emerging concepts in adaptive management. In: Allen CR, Garmestani AS (eds) Adaptive Management of Social-Ecological Systems. Springer, Dordrecht, pp 235-254. doi:10.1007/978-94-017-9682-8

Atkinson G, Bateman I, Mourato S (2012) Recent advances in the valuation of ecosystem services and biodiversity. Oxford Rev Econ Pol 28:22-47 doi:10.1093/oxrep/grs007

Ayanu YZ, Conrad C, Nauss T, Wegmann M, Koellner T (2012a) Quantifying and mapping ecosystem services supplies and demands: a review of remote sensing applications. Environ Sci Technol 46:8529-8541

Ayanu YZ, Conrad C, Nauss T, Wegmann M, Koellner T (2012b) Quantifying and Mapping Ecosystem Services Supplies and Demands: A Review of Remote Sensing Applications. Environ Sci Technol 46:8529-8541 doi:10.1021/es300157u

Bagstad KJ, Cohen E, Ancona ZH, McNulty SG, Sun G (2018) The sensitivity of ecosystem service models to choices of input data and spatial resolution. Appl Geogr 93:25-36 doi:https://doi.org/10.1016/j.apgeog.2018.02.005

Bagstad KJ, Semmens DJ, Waage S, Winthrop R (2013) A comparative assessment of decision-support tools for ecosystem services quantification and valuation. Ecosyst Serv 5:27-39 doi:https://doi.org/10.1016/j.ecoser.2013.07.004

Bagstad KJ, Villa F, Batker D, Harrison-Cox J, Voigt B, Johnson GW (2014) From theoretical to actual ecosystem services: mapping beneficiaries and spatial flows in ecosystem service assessments. Ecol Soc 19:64

Barbier EB (2009) Ecosystems as natural assets. Now Publishers Inc,

Barbier EB (2013) Wealth accounting, ecological capital and ecosystem services. Environ Devel Econ 18:133-161 doi:doi:10.1017/S1355770X12000551

Barbier EB (2016) Sustainability and development. Annual Review of Resource Economics 8:261-280 doi:10.1146/annurev-resource-100815-095227

Barletta M, Cussac VE, Agostinho AA, Baigún C, Okada EK, Carlos Catella A, Fontoura NF, Pompeu PS, Jiménez-Segura LF, Batista VS, Lasso CA, Taphorn D, Fabré NN (2015) Fisheries ecology in South American river basins. In: Freshwater Fisheries Ecology. John Wiley & Sons, Ltd, pp 311-348. doi:10.1002/9781118394380.ch27

Barletta M, Cussac VE, Agostinho AA, Baigún C, Okada EK, Catella AC, Fontoura NF, Pompeu PS, Jiménez-Segura LF, Batista VS (2016) Fisheries ecology in South American river basins. Freshwater Fisheries Ecology:311-348

Barnosky AD, Hadly EA, Bascompte J, Berlow EL, Brown JH, Fortelius M, Getz WM, Harte J, Hastings A, Marquet PA (2012) Approaching a state shift in Earth/'s biosphere. Nature 486:52-58 doi:10.1038/nature11018

Bartelmus P (2009) The cost of natural capital consumption: Accounting for a sustainable world economy. Ecolog Econ 68:1850-1857 doi:https://doi.org/10.1016/j.ecolecon.2008.12.011

Bartelmus P (2014) Environmental–Economic Accounting: Progress and Digression in the SEEA Revisions. Rev Income Wealth 60:887-904

Bartholomé E, Belward AS (2005) GLC2000: a new approach to global land cover mapping from Earth observation data. Int J Remote Sens 26:1959-1977 doi:10.1080/01431160412331291297

Bastiaanssen WGM, Steduto P (2017) The water productivity score (WPS) at global and regional level: Methodology and first results from remote sensing measurements of wheat, rice and maize. ScTEn 575:595-611 doi:http://dx.doi.org/10.1016/j.scitotenv.2016.09.032

Belinda R, Reinette B, S CG, Thomas E, P HA, Stephen P (2013) Getting the measure of ecosystem services: a social–ecological approach. Front Ecol Environ 11:268-273 doi:doi:10.1890/120144

Benavides J (2010) El desarrollo económico de la Orinoquia como aprendizaje y construcción de instituciones. Corporacion Andina de Fomento, CAF, Bogota, Colombia

Berkes F, Colding J, Folke C (2008) Navigating social-ecological systems: building resilience for complexity and change. Cambridge University Press, New York

Binder C, Hinkel J, Bots P, Pahl-Wostl C (2013) Comparison of frameworks for analyzing social-ecological systems. Ecol Soc 18:26 doi:http://dx.doi.org/10.5751/ES-05551-180426

Bioenergy (2017) Bioenergy. Bioenergy. http://www.bioenergy.com.co/SitePages/Home1111.aspx. Accessed 27092017 2017

Blaschke T (2010) Object based image analysis for remote sensing. ISPRS Journal of Photogrammetry and Remote Sensing 65:2-16 doi:http://dx.doi.org/10.1016/j.isprsjprs.2009.06.004

Bogardi JJ, Fekete BM, Vörösmarty CJ (2013) Planetary boundaries revisited: a view through the 'water lens'. Curr Opin Env Sust 5:581-589

Bommarco R, Kleijn D, Potts SG (2013) Ecological intensification: harnessing ecosystem services for food security. Trends Ecol Evol 28:230-238 doi:http://dx.doi.org/10.1016/j.tree.2012.10.012

Bouwman L, Goldewijk KK, Van Der Hoek KW, Beusen AH, Van Vuuren DP, Willems J, Rufino MC, Stehfest E (2013) Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900–2050 period. PNAS 110:20882-20887

Braat LC, de Groot R (2012) The ecosystem services agenda:bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosyst Serv 1:4-15 doi:http://dx.doi.org/10.1016/j.ecoser.2012.07.011

Burkhard B, Kandziora M, Hou Y, Müller F (2014) Ecosystem Service Potentials, Flows and Demands – Concepts for Spatial Localisation, Indication and Quantification. Landscape Online 34:1-32 doi: 10.3097/LO.201434

Burkhard B, Kroll F, Müller F, Windhorst W (2009) Landscapes' capacities to provide ecosystem services–a concept for land-cover based assessments. Landscape online 15:22

Burkhard B, Kroll F, Nedkov S, Müller F (2012) Mapping ecosystem service supply, demand and budgets. Ecol Indic 21:17-29 doi:http://dx.doi.org/10.1016/j.ecolind.2011.06.019

Burkhard B, Maes J (2017) Mapping Ecosystem Services. Advanced Books 1 doi:10.3897/ab.e12837

Camargo JA, Alonso Á (2006) Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. Environ Int 32:831-849 doi:http://dx.doi.org/10.1016/j.envint.2006.05.002

Campbell WB, López-Ortíz S, Villalba J, Provenza F, Gibson N, López-Ortíz S (2014) Veterinary Medicine: The Value of Plant Secondary Compounds and Diversity in Balancing Consumer and Ecological Health. In: Sustainable Food Production Includes Human and Environmental Health, vol 3. Issues in Agroecology – Present Status and Future Prospectus. Springer Netherlands, pp 165-190. doi:10.1007/978-94-007-7454-4_4

Cao X, Wang Y, Wu P, Zhao X, Wang J (2015) An evaluation of the water utilization and grain production of irrigated and rain-fed croplands in China. ScTEn 529:10-20 doi:https://doi.org/10.1016/j.scitotenv.2015.05.050

Carpenter SR (2005) Eutrophication of aquatic ecosystems: bistability and soil phosphorus. Proc Natl Acad Sci USA 102:10002-10005

Carpenter SR, Bennett EM (2011) Reconsideration of the planetary boundary for phosphorus. Environmental Research Letters 6:4009

Carpenter SR, Mooney HA, Agard J, Capistrano D, Defries RS, Diaz S, Dietz T, Duraiappah AK, Oteng-Yeboah A, Pereira HM, Perrings C, Reid WV, Sarukhan J, Scholes RJ, Whyte A (2009) Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proc Natl Acad Sci USA 106:1305-1312 doi:10.1073/pnas.0808772106

Carvalho-Santos C, Marcos B, Marques JE, Alcaraz-Segura D, Hein L, Honrado J (2013) Evaluation of Hydrological Ecosystem Services through Remote Sensing. In: Alcaraz-Segura D, Bella CMD, Straschnoy JV (eds) Earth Observation of Ecosystem Services. CRC press, Boca Raton, p 229

Cassman KG, Dobermann A, Walters DT, Yang H (2003) Meeting cereal demand while protecting natural resources and improving environmental quality. Annual Review of Environment and Resources 28:315-358

Castro DM (2003) Ensayo sobre tipología de suelos colombianos-Énfasis en génesis y aspectos ambientales. Rev Acad Colomb Cienc 27:319-341

Chan KM, Shaw MR, Cameron DR, Underwood EC, Daily GC (2006) Conservation planning for ecosystem services. PLoS Biol 4:e379

Chandrasekar K, Sesha Sai M, Roy P, Dwevedi R (2010) Land Surface Water Index (LSWI) response to rainfall and NDVI using the MODIS Vegetation Index product. Int J Remote Sens 31:3987-4005

Chapin FS, Randerson JT, McGuire AD, Foley JA, Field CB (2008) Changing feedbacks in the climate–biosphere system. Front Ecol Environ 6:313-320

Chapin III FS, Chapin MC, Matson PA, Vitousek P (2011) Principles of terrestrial ecosystem ecology. Springer, New York. doi:10.1007/978-1-4419-9504-9

Chen D, Huang J, Jackson TJ (2005) Vegetation water content estimation for corn and soybeans using spectral indices derived from MODIS near-and short-wave infrared bands. Remote Sens Environ 98:225-236

Clark DA, Brown S, Kicklighter DW, Chambers JQ, Thomlinson JR, Ni J (2001a) Measuring net primary production in forests: concepts and field methods. Ecological Applications 11:356-370

Clark DA, Brown S, Kicklighter DW, Chambers JQ, Thomlinson JR, Ni J (2001b) Measuring net primary production in forests: concepts and field methods. Ecological Applications 11:356-370

Cohen WB, Maiersperger TK, Yang Z, Gower ST, Turner DP, Ritts WD, Berterretche M, Running SW (2003) Comparisons of land cover and LAI estimates derived from ETM+ and MODIS for four sites in North America: a quality assessment of 2000/2001 provisional MODIS products. Remote Sens Environ 88:233-255

Cole MJ, Bailey RM, New MG (2014) Tracking sustainable development with a national barometer for South Africa using a downscaled "safe and just space" framework. PNAS 111:E4399-E4408 doi:10.1073/pnas.1400985111

Comber A, Fisher P, Brunsdon C, Khmag A (2012) Spatial analysis of remote sensing image classification accuracy. Remote Sens Environ 127:237-246

Congalton RG (1991) Remote Sensing and Geographic Information System Data Integration: Error Sources and. Photogrammetric Engineering & Remote Sensing 57:677-687

CONPES (2014) Política para el desarrollo integral de la orinoquia: Altillanura - fase I Bogota, D.C. Colombia

Contreras BÁP, Cayón SG, Corchuelo RG (2012) Models to estimate the bunch dry weight in African oil palm (Elaeis guineensis Jacq.), American oil palm (Elaeis

oleifera H.B.K. Cortes) and the interspecific hybrid (E. oleifera x E. guineensis). Agronomía Colombiana 30:46-51
Corley RHV, Tinker P (2008) The oil palm. John Wiley & Sons, Oxford
Correa HD, Ruiz SL, Arévalo LM (2005) Plan de acción en biodiversidad de la cuenca del Orinoco – Colombia / 2005 - 2015 – Propuesta Técnica. Corporinoquia, Bogotá D.C, Colombia
Costanza R, d'Arge R, De Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'neill RV, Paruelo J (1997) The value of the world's ecosystem services and natural capital. Nature 387:253-260
Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, van den Belt M (1998) The value of ecosystem services: putting the issues in perspective. Ecolog Econ 25:67-72 doi:http://dx.doi.org/10.1016/S0921-8009(98)00019-6
Costanza R, Daly HE (1992) Natural capital and sustainable development. Conserv Biol 6:37-46
Costanza R, de Groot R, Braat L, Kubiszewski I, Fioramonti L, Sutton P, Farber S, Grasso M (2017) Twenty years of ecosystem services: How far have we come and how far do we still need to go? Ecosyst Serv 28:1-16 doi:https://doi.org/10.1016/j.ecoser.2017.09.008
Costanza R, de Groot R, Sutton P, van der Ploeg S, Anderson SJ, Kubiszewski I, Farber S, Turner RK (2014) Changes in the global value of ecosystem services. Global Environ Change 26:152-158 doi:http://dx.doi.org/10.1016/j.gloenvcha.2014.04.002
Costanza R, Fisher B, Mulder K, Liu S, Christopher T (2007) Biodiversity and ecosystem services: A multi-scale empirical study of the relationship between species richness and net primary production. Ecolog Econ 61:478-491
Costanza R, Wilson M, Troy A, Voinov A, Liu S, D'Agostino J (2006) The value of New Jersey's ecosystem services and natural capital vol 13. Gund Institute for Ecological Economics, Burlington
Courault D, Seguin B, Olioso A (2005) Review on estimation of evapotranspiration from remote sensing data: From empirical to numerical modeling approaches. Irrigation and Drainage Systems 19:223-249 doi:10.1007/s10795-005-5186-0
Crossman ND, Burkhard B, Nedkov S (2012a) Quantifying and mapping ecosystem services. Int J Biodivers Sci Ecosyst Serv Manage 8:1-4 doi:10.1080/21513732.2012.695229
Crossman ND, Burkhard B, Nedkov S (2012b) Quantifying and mapping ecosystem services. International Journal of Biodiversity Science, Ecosystem Services & Management 8:1-4 doi:http://dx.doi.org/10.1080/21513732.2012.695229
Crossman ND, Burkhard B, Nedkov S (2012c) Quantifying and mapping ecosystem services. International Journal of Biodiversity Science, Ecosystem Services & Management 8:1-4
Crossman ND, Burkhard B, Nedkov S, Willemen L, Petz K, Palomo I, Drakou EG, Martín-Lopez B, McPhearson T, Boyanova K, Alkemade R, Egoh B, Dunbar MB, Maes J (2013) A blueprint for mapping and modelling ecosystem services. Ecosyst Serv 4:4-14

Crowards TM (1998) Safe Minimum Standards: costs and opportunities. Ecolog Econ 25:303-314 doi:http://doi.org/10.1016/S0921-8009(97)00041-4

Crutzen PJ (2006) The "anthropocene". In: Earth system science in the anthropocene. Springer, pp 13-18

DAAC ORNLDAACO (2011) MODIS subsetted land products, Collection 5. ORNL DAAC, Oak Ridge. Accessed 07-07 2014

Daily GC (1997) Nature's services: societal dependence on natural ecosystems. Island Pr,

Daily GC (2000) Management objectives for the protection of ecosystem services. Environmental Science & Policy 3:333-339 doi:http://dx.doi.org/10.1016/S1462-9011(00)00102-7

Daily GC, Polasky S, Goldstein J, Kareiva PM, Mooney HA, Pejchar L, Ricketts TH, Salzman J, Shallenberger R (2009) Ecosystem services in decision making: time to deliver. Front Ecol Environ 7:21-28 doi:10.1890/080025

Dalmazzone S, La Notte A (2013) Multi-scale environmental accounting: Methodological lessons from the application of NAMEA at sub-national levels. J Environ Manage 130:405-416 doi:https://doi.org/10.1016/j.jenvman.2013.09.008

Dane, Ministry of Agriculture (2016) Tercer censo nacional agropecuario vol 2. Departamento Nacional de Estadistica, Dane, Bogota

Dasgupta P (2010) Nature's role in sustaining economic development. Philosophical Transactions of the Royal Society B: Biological Sciences 365:5-11 doi:10.1098/rstb.2009.0231

Dasgupta P, Heal Ge (1979) Economic theory and exhaustible resources. Cambridge economic handbooks. Nisbet ;

Cambridge University Press, Welwyn [England]

[Cambridge, Eng.]

De Groot RS, Alkemade R, Braat L, Hein L, Willemen L (2010) Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol Complex 7:260-272 doi:http://dx.doi.org/10.1016/j.ecocom.2009.10.006

De Groot RS, Wilson MA, Boumans RM (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecolog Econ 41:393-408

de Vries W, Kros J, Kroeze C, Seitzinger SP (2013) Assessing planetary and regional nitrogen boundaries related to food security and adverse environmental impacts. Curr Opin Env Sust 5:392-402 doi:http://dx.doi.org/10.1016/j.cosust.2013.07.004

Dearing JA, Wang R, Zhang K, Dyke JG, Haberl H, Hossain MS, Langdon PG, Lenton TM, Raworth K, Brown S, Carstensen J, Cole MJ, Cornell SE, Dawson TP, Doncaster CP, Eigenbrod F, Flörke M, Jeffers E, Mackay AW, Nykvist B, Poppy GM (2014) Safe and just operating spaces for regional social-ecological systems. Global Environ Change 28:227-238 doi:http://dx.doi.org/10.1016/j.gloenvcha.2014.06.012

DeClerck FAJ, Jones SK, Attwood S, Bossio D, Girvetz E, Chaplin-Kramer B, Enfors E, Fremier AK, Gordon LJ, Kizito F, Lopez Noriega I, Matthews N, McCartney M, Meacham M, Noble A, Quintero M, Remans R, Soppe R, Willemen L, Wood SLR, Zhang W (2016) Agricultural ecosystems and their services: the vanguard of sustainability? Curr Opin Env Sust 23:92-99 doi:https://doi.org/10.1016/j.cosust.2016.11.016

DeFries RS, Pagiola S, Adamowicz WL, Akcakaya HR, Arcenas A, Babu S, Balk D, Confalonieri U, Cramer W, Falconi F (2005) Analytical approaches for assessing ecosystem condition and human well-being. Ecosytems and Human Well-being: Current state and trends, by Millenium Ecosystem Assessment Washington: World Resources Institute

Díaz de Astarloa DA, Pengue WA (2018) Nutrients Metabolism of Agricultural Production in Argentina: NPK Input and Output Flows from 1961 to 2015. Ecolog Econ 147:74-83 doi:https://doi.org/10.1016/j.ecolecon.2018.01.001

Díaz S, Demissew S, Carabias J, Joly C, Lonsdale M, Ash N, Larigauderie A, Adhikari JR, Arico S, Báldi As, Bartuska A, Baste IA, Bilgin A, Brondizio E, Chan KMA, Figueroa VE, Duraiappah A, Fischer M, Hill R, Koetz T, Leadley P, Lyver P, Mace GM, Martin-Lopez B, Okumura M, Pacheco D, Pascual U, Pérez ES, Reyers B, Roth E, Saito O, Scholes RJ, Sharma N, Tallis H, Thaman R, Watson R, Yahara T, Hamid ZA, Akosim C, Al-Hafedh Y, Allahverdiyev R, Amankwah E, Asah ST, Asfaw Z, Bartus G, Brooks LA, Caillaux J, Dalle G, Darnaedi D, Driver A, Erpul G, Escobar-Eyzaguirre P, Failler P, Fouda AMM, Fu B, Gundimeda H, Hashimoto S, Homer F, Lavorel S, Lichtenstein G, Mala WA, Mandivenyi W, Matczak P, Mbizvo C, Mehrdadi M, Metzger JP, Mikissa JB, Moller H, Mooney HA, Mumby P, Nagendra H, Nesshover C, Oteng-Yeboah AA, Pataki Gr, Roué M, Rubis J, Schultz M, Smith P, Sumaila R, Takeuchi K, Thomas S, Verma M, Yeo-Chang Y, Zlatanova D (2015) The IPBES Conceptual Framework — connecting nature and people. Curr Opin Env Sust 14:1-16

Díaz S, Pascual U, Stenseke M, Martín-López B, Watson RT, Molnár Z, Hill R, Chan KMA, Baste IA, Brauman KA, Polasky S, Church A, Lonsdale M, Larigauderie A, Leadley PW, van Oudenhoven APE, van der Plaat F, Schröter M, Lavorel S, Aumeeruddy-Thomas Y, Bukvareva E, Davies K, Demissew S, Erpul G, Failler P, Guerra CA, Hewitt CL, Keune H, Lindley S, Shirayama Y (2018) Assessing nature's contributions to people. Science 359:270-272 doi:10.1126/science.aap8826

Dingkuhn M, Laza MRC, Kumar U, Mendez KS, Collard B, Jagadish K, Singh RK, Padolina T, Malabayabas M, Torres E, Rebolledo MC, Manneh B, Sow A (2015) Improving yield potential of tropical rice: Achieved levels and perspectives through improved ideotypes. Field Crops Res 182:43-59 doi:https://doi.org/10.1016/j.fcr.2015.05.025

Dong M, Bryan BA, Connor JD, Nolan M, Gao L (2015) Land use mapping error introduces strongly-localised, scale-dependent uncertainty into land use and ecosystem services modelling. Ecosyst Serv 15:63-74 doi:https://doi.org/10.1016/j.ecoser.2015.07.006

Edens B, Hein L (2013) Towards a consistent approach for ecosystem accounting. Ecolog Econ 90:41-52 doi:https://doi.org/10.1016/j.ecolecon.2013.03.003
Egoh B, Drakou EG, Dunbar MB, Maes J, Willemen L (2012) Indicators for mapping ecosystem services: a review. European Commision, Luxembourg. doi:10.2788/41823
Egoh B, Reyers B, Rouget M, Richardson DM, Le Maitre DC, van Jaarsveld AS (2008) Mapping ecosystem services for planning and management. Agric, Ecosyst Environ 127:135-140
Ehrlich PR, Ehrlich AH (1981) Extinction: the causes and consequences of the disappearance

of species. Random House, New York
Eigenbrod F, Armsworth PR, Anderson BJ, Heinemeyer A, Gillings S, Roy DB, Thomas CD, Gaston KJ (2010) The impact of proxy-based methods on mapping the distribution of ecosystem services. J Appl Ecol 47:377-385 doi:10.1111/j.1365-2664.2010.01777.x
Ekins P (1992) A four-capital model of wealth creation. In: Ekins P, Max-Neef M (eds) Real-Life Economics: Understanding Wealth Creation. Routledge, London/New York, pp 147-155
Ekins P, Simon S, Deutsch L, Folke C, De Groot R (2003) A framework for the practical application of the concepts of critical natural capital and strong sustainability. Ecolog Econ 44:165-185 doi:http://dx.doi.org/10.1016/S0921-8009(02)00272-0
Emilie C, Maud M, Francis T, Coline B, Jeroen M, Frederic B, Johannes VP, Sandra L (2015) Assessing bundles of ecosystem services from regional to landscape scale: insights from the French Alps. J Appl Ecol 52:1145-1155 doi:doi:10.1111/1365-2664.12502
Emmanuelle L, Guillaume T, Hijri DN, Lobato RR, Chantal F, Caroline M, Marlène LS, Diana S, Jaleh G (2016) Natural 13C distribution in oil palm (Elaeis guineensis Jacq.) and consequences for allocation pattern. Plant, Cell Environ 39:199-212 doi:doi:10.1111/pce.12606
Etter A, McAlpine C, Wilson K, Phinn S, Possingham H (2006a) Regional patterns of agricultural land use and deforestation in Colombia. Agriculture, Ecosystems & Environment 114:369-386 doi:http://dx.doi.org/10.1016/j.agee.2005.11.013
Etter A, McAlpine C, Wilson K, Phinn S, Possingham H (2006b) Regional patterns of agricultural land use and deforestation in Colombia. Agric, Ecosyst Environ 114:369-386 doi:10.1016/j.agee.2005.11.013
Etter A, Sarmiento A, Romero MH (2010) Land Use Changes (1970–2020) and Carbon Emissions in the Colombian Llanos. In: Hill MJ, Hanan NP (eds) Ecosystem Function in Savannas: Measurement and Modeling at Landscape to Global Scales. Taylor & Francis CRC Press, Boca Raton, p 383. doi:10.1201/b10275-26
Evans LT (1996) Crop evolution, adaptation and yield. Cambridge University Press,
Falkenmark M, Rockström J (2013) Balancing water for humans and nature: The new approach in ecohydrology. Earthscan. doi:10.4324/9781849770521
FAO (2009) Land Cover Classification System v.3 (or Land Cover Meta Language): Design criteria. FAO, Rome

FAO (2010) Global Forest Resources Assessment 2010: Main Report. Food and agriculture organization of the United Nations (FAO), Rome

FAO (2015) Statistical pocketbook, world food and agriculture 2015. Food and Agriculture Organization of the United Nations

Rome

FAO (2017) World programme for the census of agriculture 2020 vol 1. Food and Agriculture Organization of the United Nations (FAO), Rome

Fedegan (2014) Analisis del inventario ganadero Colombiano 2014. Federacion Colombiana de Ganaderos (Fedegan), Colombia

Fedepalma (2005) Anuario estadistico 2005; la agroindustria de la palma en Colombia y el mundo. Federación Nacional de Cultivadores de Palma de Aceite (Fedepalma), Colombia

Fedepalma (2008) Anuario estadistico 2008; la agroindustria de la palma en Colombia y el mundo. Federación Nacional de Cultivadores de Palma de Aceite (Fedepalma), Colombia

Fedepalma (2013) Minianuario estadistico 2013; principales cifras de la agroindustria de la palma de aceite en Colombia. Federación Nacional de Cultivadores de Palma de Aceite (Fedepalma), Colombia

Fedepalma (2015) Mini anuario estadistico 2015; principales cifras de la agroindustria de la palma de aceite en Colombia Federación Nacional de Cultivadores de Palma de Aceite (Fedepalma), Colombia

Fernandes FHS, Sano EE, Ferreira LG, de Mello Baptista GM, Victoria DdC, Fassoni-Andrade AC (2018) Degradation trends based on MODIS-derived estimates of productivity and water use efficiency: A case study for the cultivated pastures in the Brazilian Cerrado. Remote Sensing Applications: Society and Environment 11:30-40 doi:https://doi.org/10.1016/j.rsase.2018.04.014

Foley JA, Costa MH, Delire C, Ramankutty N, Snyder P (2003) Green surprise? How terrestrial ecosystems could affect earth’s climate. Front Ecol Environ 1:38-44

Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, Chapin FS, Coe MT, Daily GC, Gibbs HK (2005a) Global consequences of land use. Science 309:570-574 doi:http://dx.doi.org/10.1126/science.1111772

Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, Chapin FS, Coe MT, Daily GC, Gibbs HK, Helkowski JH, Holloway T, Howard EA, Kucharik CJ, Monfreda C, Patz JA, Prentice IC, Ramankutty N, Snyder PK (2005b) Global consequences of land use. Science 309:570-574 doi:10.1126/science.1111772

Foley JA, Ramankutty N, Brauman KA, Cassidy ES, Gerber JS, Johnston M, Mueller ND, O/'Connell C, Ray DK, West PC, Balzer C, Bennett EM, Carpenter SR, Hill J, Monfreda C, Polasky S, Rockstrom J, Sheehan J, Siebert S, Tilman D, Zaks DPM (2011) Solutions for a cultivated planet. 478:337-342

Folke C (2010) How resilient are ecosystems to global environmental change? Sustainability Science 5:151-154 doi:http://dx.doi.org/10.1007/s11625-010-0109-x

Folke C, Biggs R, Norström AV, Reyers B, Rockström J (2016) Social-ecological resilience and biosphere-based sustainability science. Ecol Soc 21 doi:10.5751/es-08748-210341

Folke C, Carpenter S, Elmqvist T, Gunderson L, Holling CS, Walker B (2002) Resilience and sustainable development: building adaptive capacity in a world of transformations. AMBIO: A journal of the human environment 31:437-440 doi:https://doi.org/10.1579/0044-7447-31.5.437

Forkuor G, Dimobe K, Serme I, Tondoh JE (2018) Landsat-8 vs. Sentinel-2: examining the added value of sentinel-2's red-edge bands to land-use and land-cover mapping in Burkina Faso. GIScience & Remote Sensing 55:331-354 doi:10.1080/15481603.2017.1370169

Foster T, Brozović N, Butler AP, Neale CMU, Raes D, Steduto P, Fereres E, Hsiao TC (2017) AquaCrop-OS: An open source version of FAO's crop water productivity model. Agric Water Manage 181:18-22 doi:https://doi.org/10.1016/j.agwat.2016.11.015

Friedl MA, Sulla-Menashe D, Tan B, Schneider A, Ramankutty N, Sibley A, Huang X (2010) MODIS Collection 5 global land cover: Algorithm refinements and characterization of new datasets. Remote Sens Environ 114:168-182 doi:10.1016/j.rse.2009.08.016

Galaz V, Biermann F, Folke C, Nilsson M, Olsson P (2012) Global environmental governance and planetary boundaries: An introduction. Ecolog Econ 81:1-3 doi:https://doi.org/10.1016/j.ecolecon.2012.02.023

Gao B-C (1996) NDWI—a normalized difference water index for remote sensing of vegetation liquid water from space. Remote Sens Environ 58:257-266

Gaviria-Uribe X, Naranjo-Ramírez JF, Bolívar-Vergara DM, Barahona-Rosales R (2015) Intake and digestibility of nutrients in zebu steers grazing in intensive silvopastoral system. Arch Zootec 64:21-27

Gerten D, Hoff H, Rockström J, Jägermeyr J, Kummu M, Pastor AV (2013) Towards a revised planetary boundary for consumptive freshwater use: role of environmental flow requirements. Curr Opin Env Sust 5:551-558 doi:http://dx.doi.org/10.1016/j.cosust.2013.11.001

Giri C, Zhu Z, Reed B (2005) A comparative analysis of the Global Land Cover 2000 and MODIS land cover data sets. Remote Sens Environ 94:123-132 doi:https://doi.org/10.1016/j.rse.2004.09.005

Godfray HCJ, Beddington JR, Crute IR, Haddad L, Lawrence D, Muir JF, Pretty J, Robinson S, Thomas SM, Toulmin C (2010) Food Security: The Challenge of Feeding 9 Billion People. Science 327:812-818 doi:10.1126/science.1185383

Gong P, Wang J, Yu L, Zhao Y, Zhao Y, Liang L, Niu Z, Huang X, Fu H, Liu S, Li C, Li X, Fu W, Liu C, Xu Y, Wang X, Cheng Q, Hu L, Yao W, Zhang H, Zhu P, Zhao Z, Zhang H, Zheng Y, Ji L, Zhang Y, Chen H, Yan A, Guo J, Yu L, Wang L, Liu X, Shi T, Zhu M, Chen Y, Yang G, Tang P, Xu B, Giri C, Clinton N, Zhu Z, Chen J, Chen J (2013) Finer resolution observation and monitoring of global land cover: first mapping results with Landsat TM and ETM+ data. Int J Remote Sens 34:2607-2654 doi:10.1080/01431161.2012.748992

Grassini P, van Bussel LG, Van Wart J, Wolf J, Claessens L, Yang H, Boogaard H, de Groot H, van Ittersum MK, Cassman KG (2015) How good is good enough? Data requirements for reliable crop yield simulations and yield-gap analysis. Field Crops Res 177:49-63

Grassini P, Yang H, Irmak S, Thorburn J, Burr C, Cassman KG (2011) High-yield irrigated maize in the Western US Corn Belt: II. Irrigation management and crop water productivity. Field Crops Res 120:133-141

Griggs D, Stafford-Smith M, Gaffney O, Rockström J, Öhman MC, Shyamsundar P, Steffen W, Glaser G, Kanie N, Noble I (2013) Policy: Sustainable development goals for people and planet. Nature 495:305-307 doi:10.1038/495305a

Gruber N, Galloway JN (2008) An Earth-system perspective of the global nitrogen cycle. Nature 451:293-296 doi:10.1038/nature06592

Guerry AD, Polasky S, Lubchenco J, Chaplin-Kramer R, Daily GC, Griffin R, Ruckelshaus M, Bateman IJ, Duraiappah A, Elmqvist T, Feldman MW, Folke C, Hoekstra J, Kareiva PM, Keeler BL, Li S, McKenzie E, Ouyang Z, Reyers B, Ricketts TH, Rockström J, Tallis H, Vira B (2015) Natural capital and ecosystem services informing decisions: From promise to practice. PNAS 112:7348-7355 doi:10.1073/pnas.1503751112

Haberl H, Erb K-H, Krausmann F (2014) Human appropriation of net primary production: patterns, trends, and planetary boundaries. Annual Review of Environment and Resources 39:363-391

Haberl H, Erb KH, Krausmann F, Gaube V, Bondeau A, Plutzar C, Gingrich S, Lucht W, Fischer-Kowalski M (2007) Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. PNAS 104:12942-12947

Haines-Young R, Potschin M (2018) Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure. Fabis Consulting Ltd, Nottingham

Hansen MC, Potapov PV, Moore R, Hancher M, Turubanova SA, Tyukavina A, Thau D, Stehman SV, Goetz SJ, Loveland TR, Kommareddy A, Egorov A, Chini L, Justice CO, Townshend JRG (2013) High-Resolution Global Maps of 21st-Century Forest Cover Change. Science 342:850-853 doi:10.1126/science.1244693

Hasenauer H, Petritsch R, Zhao M, Boisvenue C, Running SW (2012) Reconciling satellite with ground data to estimate forest productivity at national scales. For Ecol Manage 276:196-208

Haslinger F (1984) Economic theory of natural resources. Metrika 31:114 doi:10.1007/bf01915192

Häyhä T, Lucas PL, van Vuuren DP, Cornell SE, Hoff H (2016) From Planetary Boundaries to national fair shares of the global safe operating space — How can the scales be bridged? Global Environ Change 40:60-72 doi:http://dx.doi.org/10.1016/j.gloenvcha.2016.06.008

He Y, Piao S, Li X, Chen A, Qin D (2018) Global patterns of vegetation carbon use efficiency and their climate drivers deduced from MODIS satellite data and process-based models. Agricultural and Forest Meteorology 256-257:150-158 doi:https://doi.org/10.1016/j.agrformet.2018.03.009

Heikkurinen P, Rinkinen J, Järvensivu T, Wilén K, Ruuska T (2016) Organising in the Anthropocene: an ontological outline for ecocentric theorising. J Clean Prod 113:705-714 doi:http://dx.doi.org/10.1016/j.jclepro.2015.12.016

Hein L, Bagstad K, Edens B, Obst C, de Jong R, Lesschen JP (2016) Defining Ecosystem Assets for Natural Capital Accounting. PLoS ONE 11:e0164460 doi:http://dx.doi.org/10.1371/journal.pone.0164460

Hein L, Obst C, Edens B, Remme RP (2015) Progress and challenges in the development of ecosystem accounting as a tool to analyse ecosystem capital. Current Opinion in Environmental Sustainability 14:86-92

Herold M, Mayaux P, Woodcock CE, Baccini A, Schmullius C (2008) Some challenges in global land cover mapping: An assessment of agreement and accuracy in existing 1 km datasets. Remote Sens Environ 112:2538-2556 doi:https://doi.org/10.1016/j.rse.2007.11.013

Herold M, See L, Tsendbazar N-E, Fritz S (2016) Towards an Integrated Global Land Cover Monitoring and Mapping System. Remote Sensing 8:1036

Holling CS (1978) Adaptive environmental assessment and management. John Wiley & Sons, Oxford

Houborg R, Fisher JB, Skidmore AK (2015) Advances in remote sensing of vegetation function and traits. IJAEO 43:1-6 doi:https://doi.org/10.1016/j.jag.2015.06.001

Huete A, Didan K, Miura T, Rodriguez EP, Gao X, Ferreira LG (2002) Overview of the radiometric and biophysical performance of the MODIS vegetation indices. Remote Sens Environ 83:195-213

Hui D, Jackson RB (2006) Geographical and interannual variability in biomass partitioning in grassland ecosystems: a synthesis of field data. New Phytol 169:85-93

Hussain M, Chen D, Cheng A, Wei H, Stanley D (2013) Change detection from remotely sensed images: From pixel-based to object-based approaches. ISPRS Journal of Photogrammetry and Remote Sensing 80:91-106 doi:http://dx.doi.org/10.1016/j.isprsjprs.2013.03.006

Ideam (2011a) Boletin forestal 2008-2010. Instituto de Hidrología, Meteorología y Estudios Ambientales de Colombia (Ideam), Bogota

Ideam (2011b) Estudio Nacional del Agua 2010. Instituto de Hidrología, Meteorología y Estudios Ambientales (Ideam), Bogota

Ideam (2013) Zonificación y codificación de uniades hidrográficas e hidrogeológicas de Colombia. Instituto de Hidrología, Meteorología y Estudios Ambientales de Colombia (Ideam), Bogotá

Ideam (2015a) Deforestación y afectación de los ecosistemas por ocupación del territorio y actividades económicas. Instituto de Hidrología, Meteorología y Estudios Ambientales de Colombia (Ideam), Bogotá

Ideam (2015b) Estudio nacional del agua 2014. Instituto de Hidrología, Meteorología y Estudios Ambientales (Ideam), Bogota

Inman-Bamber NG, Muchow RC, Robertson MJ (2002) Dry matter partitioning of sugarcane in Australia and South Africa. Field Crops Res 76:71-84

Ivanov ED, Young RH, Weber J-L (2011) Constructing the Net Primary Production and Biomass accounts and Ecosystem Carbon Balance Index: A Methodological Note. Paper presented at the Expert meeting on ecosystem accounting, London, 5-7 December, 2011

Ivits E, Cherlet M, Horion S, Fensholt R (2013) Global Biogeographical Pattern of Ecosystem Functional Types Derived From Earth Observation Data. Remote Sensing 5:3305

Kalma JD, McVicar TR, McCabe MF (2008) Estimating Land Surface Evaporation: A Review of Methods Using Remotely Sensed Surface Temperature Data. SGeo 29:421-469 doi:10.1007/s10712-008-9037-z

Kerr JT, Ostrovsky M (2003) From space to species: ecological applications for remote sensing. Trends Ecol Evol 18:299-305 doi:http://dx.doi.org/10.1016/S0169-5347(03)00071-5

Knapp AK, Carroll CJ, Fahey TJ (2014a) Patterns and Controls of Terrestrial Primary Production in a Changing World. Ecology and the Environment:205-246

Knapp AK, Carroll CJ, Fahey TJ (2014b) Primary Production in Terrestrial Ecosystems: Patterns and Controls in a Changing World. In: Ecology and the Environment. Springer, pp 1-36

Knapp AK, Carroll CJW, Fahey TJ (2014c) Patterns and Controls of Terrestrial Primary Production in a Changing World. In: Monson RK (ed) Ecology and the Environment. Springer New York, pp 205-246. doi:10.1007/978-1-4614-7501-9_2

Kotowska MM, Leuschner C, Triadiati T, Meriem S, Hertel D (2015) Quantifying above-and belowground biomass carbon loss with forest conversion in tropical lowlands of Sumatra (Indonesia). Glob Chang Biol 21:3620-3634

Kumar M, Singh RP, Panigrahy S, Raghubanshi AS (2014) Carbon density and accumulation in agroecosystem of Indo-Gangetic Plains and Vindhyan highlands, India. Environ Monit Assess 186:4971-4985 doi:10.1007/s10661-014-3752-3

Kumar P (2010) The Economics of Ecosystems and Biodiversity: ecological and economic foundations. UNEP/Earthprint, London and Washington

La Notte A, Maes J, Dalmazzone S, Crossman ND, Grizzetti B, Bidoglio G (2017) Physical and monetary ecosystem service accounts for Europe: A case study for in-stream nitrogen retention. Ecosyst Serv 23:18-29 doi:http://doi.org/10.1016/j.ecoser.2016.11.002

Laborde R (2007) Los territorios indígenas traslapados con áreas del Sistema de Parques Nacionales Naturales en la Amazonia colombiana: situación actual y perspectivas vol 23. Foro nacional Ambiental, Bogota

Lal R, Lorenz K, Hüttl RF, Schneider BU, von Braun J (2013) Ecosystem services and carbon sequestration in the biosphere. Springer,

Lamade E, Bouillet J-P (2005) Carbon storage and global change: the role of oil palm. Oléagineux, corps gras, lipides 12:154-160

Lambin E, Gibbs H, Ferreira L, Grau R, Mayaux P, Meyfroidt P, Morton D, Rudel T, Gasparri I, Munger J (2013) Estimating the world's potentially available cropland using a bottom-up approach. Global Environ Change 23:892-901

Lambin EF, Meyfroidt P (2011) Global land use change, economic globalization, and the looming land scarcity. PNAS 108:3465-3472

Lasso CA, Usma JS, Trujillo F, Rial B (eds) (2010) Biodiversidad de la cuenca del orinoco: bases científicas para la identificación de áreas prioritarias para la conversación y uso sostenible de la biodiversidad. Instituto de Investigacion de Recursos Biologicos Alexander von Humboldt, Bogotá, D. C., Colombia

Latacz-Lohmann U, Schilizzi S (2014) Creating physical environmental asset accounts from markets for ecosystem conservation: A comment. Ecolog Econ 108:266-268

Laurance WF, Sayer J, Cassman KG (2014) Agricultural expansion and its impacts on tropical nature. Trends Ecol Evol 29:107-116 doi:https://doi.org/10.1016/j.tree.2013.12.001

Lavelle P, Rodríguez N, Arguello O, Bernal J, Botero C, Chaparro P, Gómez Y, Gutiérrez A, Hurtado MdP, Loaiza S, Pullido SX, Rodríguez E, Sanabria C, Velásquez E, Fonte SJ (2014) Soil ecosystem services and land use in the rapidly changing Orinoco River Basin of Colombia. Agric, Ecosyst Environ 185:106-117 doi:http://dx.doi.org/10.1016/j.agee.2013.12.020

Lavorel S, Bayer A, Bondeau A, Lautenbach S, Ruiz-Frau A, Schulp N, Seppelt R, Verburg P, Teeffelen Av, Vannier C, Arneth A, Cramer W, Marba N (2017) Pathways to bridge the biophysical realism gap in ecosystem services mapping approaches. Ecol Indic 74:241-260 doi:https://doi.org/10.1016/j.ecolind.2016.11.015

Lemaire G, Hodgson J, Chabbi A (2011) Grassland productivity and ecosystem services. CABI, Wallingford. doi:10.1079/9781845938093.0000

Lenton TM, Held H, Kriegler E, Hall JW, Lucht W, Rahmstorf S, Schellnhuber HJ (2008) Tipping elements in the Earth's climate system. PNAS 105:1786-1793 doi:10.1073/pnas.0705414105

León GS (2005) La cuenca del río Orinoco: visión hidrográfica y balance hídrico. Revista Geográfica Venezolana 46:75-108

Levin S, Xepapadeas T, Crépin A-S, Norberg J, De Zeeuw A, Folke C, Hughes T, Arrow K, Barrett S, Daily G (2013) Social-ecological systems as complex adaptive systems: modeling and policy implications. Environ Devel Econ 18:111-132 doi: https://doi.org/10.1017/S1355770X12000460

Lewis SL (2012) We must set planetary boundaries wisely. Nature 485:417

Liu J, Dietz T, Carpenter SR, Folke C, Alberti M, Redman CL, Schneider SH, Ostrom E, Pell AN, Lubchenco J (2007) Coupled human and natural systems. AMBIO: a journal of the human environment 36:639-649 doi:https://doi.org/10.1579/0044-7447(2007)36[639:CHANS]2.0.CO;2

Liu J, Mooney H, Hull V, Davis SJ, Gaskell J, Hertel T, Lubchenco J, Seto KC, Gleick P, Kremen C (2015) Systems integration for global sustainability. Science 347:1258832 doi:10.1126/science.1258832

Liu L, Fieguth PW (2012) Texture classification from random features. Pattern Analysis and Machine Intelligence, IEEE Transactions on 34:574-586

Lobell DB, Cassman KG, Field CB (2009) Crop yield gaps: their importance, magnitudes, and causes. Annual review of environment and resources 34:179-204

Loveland T, Belward A (1997) The IGBP-DIS global 1km land cover data set, DISCover: first results. Int J Remote Sens 18:3289-3295

Lovett GM, Jones CG, Turner MG, Weathers KC (2005) Ecosystem Function in Heterogeneous Landscapes. In: Lovett GM, Turner MG, Jones CG, Weathers

KC (eds) Ecosystem Function in Heterogeneous Landscapes. Springer New York, New York, NY, pp 1-4. doi:10.1007/0-387-24091-8_1
Lozano ÁMB, Martínez AP, Barrera JEE, Castro EC (2013) Análisis de datos meteorológicos para identificar y definir el clima en Yopal, Casanare. Revista de Medicina Veterinaria:85-92
Lunetta RS, Knight JF, Ediriwickrema J, Lyon JG, Worthy LD (2006) Land-cover change detection using multi-temporal MODIS NDVI data. Remote Sens Environ 105:142-154 doi:http://dx.doi.org/10.1016/j.rse.2006.06.018
Luyssaert S, Inglima I, Jung M, Richardson A, Reichstein M, Papale D, Piao S, SCHULZE ED, Wingate L, Matteucci G (2007) CO2 balance of boreal, temperate, and tropical forests derived from a global database. Glob Chang Biol 13:2509-2537
Lyons KG, Brigham CA, Traut BH, Schwartz MW (2005) Rare species and ecosystem functioning. Conserv Biol 19:1019-1024 doi:10.1111/j.1523-1739.2005.00106.x
MA (2003) Millenium Ecosystem Assessment: Ecosystems and human well-being A framework for assessment. Island Press, Washington, DC
MA (2005) Millennium Ecosystem Assessment: Ecosystems and human well-being vol 5. Island Press Washington, DC,
Ma X, Huete A, Yu Q, Restrepo-Coupe N, Beringer J, Hutley LB, Kanniah KD, Cleverly J, Eamus D (2014) Parameterization of an ecosystem light-use-efficiency model for predicting savanna GPP using MODIS EVI. Remote Sens Environ 154:253-271 doi:https://doi.org/10.1016/j.rse.2014.08.025
Mace GM, Hails RS, Cryle P, Harlow J, Clarke SJ (2015) Review: towards a risk register for natural capital. J Appl Ecol 52:641-653
Maes J, Egoh B, Willemen L, Liquete C, Vihervaara P, Schägner JP, Grizzetti B, Drakou EG, Notte AL, Zulian G, Bouraoui F, Luisa Paracchini M, Braat L, Bidoglio G (2012) Mapping ecosystem services for policy support and decision making in the European Union. Ecosyst Serv 1:31-39 doi:http://dx.doi.org/10.1016/j.ecoser.2012.06.004
Mäler KG, Aniyar S, Jansson Å (2008) Accounting for ecosystem services as a way to understand the requirements for sustainable development. Proc Natl Acad Sci USA 105:9501-9506 doi:10.1073/pnas.0708856105
Malhi Y, Aragão LEOC, Metcalfe DB, Paiva R, Quesada CA, Almeida S, Anderson L, Brando P, Chambers JQ, da Costa ACL, Hutyra LR, Oliveira P, Patiño S, Pyle EH, Robertson AL, Teixeira LM (2009) Comprehensive assessment of carbon productivity, allocation and storage in three Amazonian forests. Glob Chang Biol 15:1255-1274
Malhi Y, Doughty C, Galbraith D (2011a) The allocation of ecosystem net primary productivity in tropical forests. Philosophical Transactions of the Royal Society B: Biological Sciences 366:3225-3245
Malhi Y, Doughty C, Galbraith D (2011b) The allocation of ecosystem net primary productivity in tropical forests. Philosophical Transactions of the Royal Society B: Biological Sciences 366:3225-3245

Martínez-Harms MJ, Balvanera P (2012) Methods for mapping ecosystem service supply: a review. International Journal of Biodiversity Science, Ecosystem Services & Management 8:17-25 doi:10.1080/21513732.2012.663792
Meadows DH, Meadows DL, Randers J, Bahrens III WW (1972) Club of Rome. The Limits to growth; a report for the Club of Rome's project on the predicament of mankind. A report for the club of rome's project on the predicament of mankind. Universe Books, New York
Meadows DH, Randers J, Meadows DL (2004) The limits to growth : the 30-year update. Chelsea Green Publishing Company, White River Junction, Vt :
Medková H, Vačkář D, Weinzettel J (2017) Appropriation of potential net primary production by cropland in terrestrial ecoregions. J Clean Prod 150:294-300 doi:http://dx.doi.org/10.1016/j.jclepro.2017.03.002
Melling L, Goh KJ, Beauvais C, Hatano R (2008) Carbon flow and budget in young mature oil palm agroecosystem on deep tropical peat. The Planter 84:21
Ministry of environment, ministry of trade, ministry of economic affairs, national department of planning, development moaar (2000) Plan Nacional De Desarrollo Forestal Bogota
Ministry of environment and sustainable development (2013) Política Nacional Para La Gestión Integral Ambiental Del Suelo (Gias). Ministerio de ambiente y desarrollo sostenible, Bogota
Ministry of environment housing and land (2010) Politica Nacional para la Gestion Integral del Recurso Hidrico. Ministerio de ambiente, vivienda y desarrollo territorial, Bogota
Mitchell MGE, Suarez-Castro AF, Martinez-Harms M, Maron M, McAlpine C, Gaston KJ, Johansen K, Rhodes JR (2015) Reframing landscape fragmentation's effects on ecosystem services. Trends Ecol Evol 30:190-198 doi:https://doi.org/10.1016/j.tree.2015.01.011
Molden D, Frenken K, Barker R, Fraiture C d, Mati, B, Svendsen M, Sadoff C, Finlayson C, M (2007) Trends in water and agricultural development. Water for food, water for life : a comprehensive assessment of water management in agriculture. Earthscan, Londres
Monteiro LA, Sentelhas PC (2017) Sugarcane yield gap: can it be determined at national level with a simple agrometeorological model? Crop and Pasture Science 68:272-284 doi:https://doi.org/10.1071/CP16334
Monteith J, Moss C (1977) Climate and the efficiency of crop production in Britain. Philos Trans R Soc Lond B Biol Sci 281:277-294
Mulla DJ (2013) Twenty five years of remote sensing in precision agriculture: Key advances and remaining knowledge gaps. Biosys Eng 114:358-371 doi:http://dx.doi.org/10.1016/j.biosystemseng.2012.08.009
Nakano T, Nemoto M, Shinoda M (2008) Environmental controls on photosynthetic production and ecosystem respiration in semi-arid grasslands of Mongolia. Agricultural and Forest Meteorology 148:1456-1466
Nilsson M, Persson Å (2012) Reprint of "Can Earth system interactions be governed? Governance functions for linking climate change mitigation with land use, freshwater and biodiversity protection". Ecolog Econ 81:10-20 doi:https://doi.org/10.1016/j.ecolecon.2012.06.020

Obst C, Hein L, Edens B (2016) National Accounting and the Valuation of Ecosystem Assets and Their Services. Environ Resource Econ 64 doi:10.1007/s10640-015-9921-1

Obst C, Vardon M (2014) Recording environmental assets in the national accounts. Oxford Review of Economic Policy 30:126-144 doi:10.1093/oxrep/gru003

Ocampo Á (1996) La palma de aceite, recurso estratégico en sistemas integrados de producción tropical. Revista Palmas 17:23-42

Oliver R (2013) Evaluation and scoping of EU timber importers and imports from South America. TRAFFIC International, Cambridge

Ostrom E (2009) A general framework for analyzing sustainability of social-ecological systems. Science 325:419-422 doi:10.1126/science.1172133

Ott LE, Pawson S, Collatz GJ, Gregg WW, Menemenlis D, Brix H, Rousseaux CS, Bowman KW, Liu J, Eldering A (2015) Assessing the magnitude of CO2 flux uncertainty in atmospheric CO2 records using products from NASA's Carbon Monitoring Flux Pilot Project. JGeophysRes-Atmos 120:734-765

Owen B, Eric J (1995) Estado de las principales características físico-químicas de los suelos palmeros de la región de la Orinoquia. Revista Palmas 16:25-30

Pan S, Tian H, Dangal SRS, Ouyang Z, Tao B, Ren W, Lu C, Running S (2014a) Modeling and Monitoring Terrestrial Primary Production in a Changing Global Environment: Toward a Multiscale Synthesis of Observation and Simulation. Advances in Meteorology 2014:17 doi:10.1155/2014/965936

Pan Y, Wu J, Xu Z (2014b) Analysis of the tradeoffs between provisioning and regulating services from the perspective of varied share of net primary production in an alpine grassland ecosystem. Ecol Complex 17:79-86 doi:https://doi.org/10.1016/j.ecocom.2013.11.001

Passioura J (2006) Increasing crop productivity when water is scarce—from breeding to field management. Agric Water Manage 80:176-196 doi:https://doi.org/10.1016/j.agwat.2005.07.012

Pastor AV, Ludwig F, Biemans H, Hoff H, Kabat P (2014) Accounting for environmental flow requirements in global water assessments. Hydrol Earth Syst Sci 18:5041-5059 doi:10.5194/hess-18-5041-2014

Pedersen OG, Haan M (2006) The system of environmental and economic accounts—2003 and the economic relevance of physical flow accounting. Journal of Industrial Ecology 10:19-42

Pérez-Hoyos A, Martínez B, García-Haro F, Moreno Á, Gilabert M (2014) Identification of Ecosystem Functional Types from Coarse Resolution Imagery Using a Self-Organizing Map Approach: A Case Study for Spain. Remote Sensing 6:11391

Peters DPC, Bestelmeyer BT, Knapp AK, Herrick JE, Monger HC, Havstad KM (2009) Approaches to Predicting Broad-Scale Regime Shifts Using Changing Pattern-Process Relationships Across Scales. In: Miao S, Carstenn S, Nungesser M (eds) Real World Ecology: Large-Scale and Long-Term Case Studies and Methods. Springer New York, New York, NY, pp 47-72. doi:10.1007/978-0-387-77942-3_3

Pettorelli N, Laurance WF, O'Brien TG, Wegmann M, Nagendra H, Turner W (2014) Satellite remote sensing for applied ecologists: opportunities and challenges. J Appl Ecol 51:839-848

Phillips J, Duque A, Yepes A, Cabrera K, García M, Navarrete D, Álvarez E, Cárdenas D (2011) Estimación de las reservas actuales (2010) de carbono almacenadas en la biomasa aérea en bosques naturales de Colombia. Estratificación, alometría y métodos análiticos. Instituto de Hidrología, Meteorología, y Estudios Ambientales (IDEAM), Bogotá

Pimentel D, Berger B, Filiberto D, Newton M, Wolfe B, Karabinakis E, Clark S, Poon E, Abbett E, Nandagopal S (2004) Water resources: agricultural and environmental issues. Bioscience 54:909-918

Pretty J, Bharucha ZP (2014) Sustainable intensification in agricultural systems. Ann Bot 114:1571-1596

Pulhin FB, Lasco RD, Urquiola JP (2015) Carbon Sequestration Potential of Oil Palm in Bohol, Philippines. Ecosystems & Development Journal 4:14-19

Raffensperger C (1999) Protecting public health and the environment: implementing the precautionary principle. Island Press, Washington

Randerson JT, Iii FSC, Harden JW, Neff JC, Harmon ME (2002) Net Ecosystem Production: A Comprehensive Measure of Net Carbon Accumulation by Ecosystems. Ecol Appl 12

Rasmus F, Stéphanie H, Torbern T, Andrea E, Eva I, Kjeld R (2015) Global-scale mapping of changes in ecosystem functioning from earth observation-based trends in total and recurrent vegetation. Glob Ecol Biogeogr 24:1003-1017 doi:doi:10.1111/geb.12338

Rasmussen LV, Coolsaet B, Martin A, Mertz O, Pascual U, Corbera E, Dawson N, Fisher JA, Franks P, Ryan CM (2018) Social-ecological outcomes of agricultural intensification. Nature Sustainability 1:275-282

Raworth K (2012) A safe and just space for humanity: can we live within the doughnut. Oxfam Policy and Practice: Climate Change and Resilience 8:1-26

Reed BC, White M, Brown JF (2003) Remote sensing phenology. In: Phenology: An Integrative Environmental Science. Springer, Dordrecht, pp 365-381

Remme RP, Edens B, Schröter M, Hein L (2015) Monetary accounting of ecosystem services: A test case for Limburg province, the Netherlands. Ecolog Econ 112:116-128 doi:https://doi.org/10.1016/j.ecolecon.2015.02.015

Remme RP, Schröter M, Hein L (2014a) Developing spatial biophysical accounting for multiple ecosystem services. Ecosystem Services 10:6-18 doi:http://dx.doi.org/10.1016/j.ecoser.2014.07.006

Remme RP, Schröter M, Hein L (2014b) Developing spatial biophysical accounting for multiple ecosystem services. Ecosyst Serv 10:6-18

Rist L, Campbell BM, Frost P (2013a) Adaptive management: where are we now? Environ Conserv 40:5-18 doi: https://doi.org/10.1017/S0376892912000240

Rist L, Felton A, Samuelsson L, Sandström C, Rosvall O (2013b) A new paradigm for adaptive management. Ecol Soc 18 doi:http://dx.doi.org/10.5751/ES-06183-180463

Rivera M, Amézquita E, Bernal J, Rao I (2013) Las Sabanas de los Llanos Orientales de Colombia: Caracterización Biofísica e Importancia para la Producción

Agropecuaria. Sistemas Agropastoriles:Un enfoque integrado para el manejo sostenible de Oxisoles de los llanos orientales de Colombia. Centro Internacional de Agricultura Tropical (CIAT), Cali

Robinson NP, Allred BW, Smith WK, Jones MO, Moreno A, Erickson TA, Naugle DE, Running SW, Pettorelli N, Paruelo J (2018) Terrestrial primary production for the conterminous United States derived from Landsat 30 m and MODIS 250 m. Remote Sensing in Ecology and Conservation 0 doi:doi:10.1002/rse2.74

Rockström J, Barron J (2007) Water productivity in rainfed systems: overview of challenges and analysis of opportunities in water scarcity prone savannahs. Irrig Sci 25:299-311

Rockström J, Steffen W, Noone K, Persson Å, Chapin FS, Lambin EF, Lenton TM, Scheffer M, Folke C, Schellnhuber HJ (2009) A safe operating space for humanity. Nature 461:472-475 doi:10.1038/461472a

Rockström J, Williams J, Daily G, Noble A, Matthews N, Gordon L, Wetterstrand H, DeClerck F, Shah M, Steduto P (2017) Sustainable intensification of agriculture for human prosperity and global sustainability. Ambio 46:4-17

Romero-Ruiz MH, Flantua SGA, Tansey K, Berrio JC (2012a) Landscape transformations in savannas of northern South America: Land use/cover changes since 1987 in the Llanos Orientales of Colombia. Applied Geography 32:766-776 doi:http://dx.doi.org/10.1016/j.apgeog.2011.08.010

Romero-Ruiz MH, Flantua SGA, Tansey K, Berrio JC (2012b) Landscape transformations in savannas of northern South America: Land use/cover changes since 1987 in the Llanos Orientales of Colombia. Appl Geogr 32:766-776 doi:10.1016/j.apgeog.2011.08.010

Romero M, Galindo G, Otero J, Armenteras D (2004) Ecosistemas de la cuenca del Orinoco colombiano. Instituto de Investigaciones de Recursos Biológicos Alexander von Humboldt. Bogotá, Colombia. IGAC, Bogota, Colombia

Rudas G (2003) Propuesta de un Sistema de Indicadores de Seguimiento del Convenio sobre la Diversidad Biológica de la Cuenca del Orinoco. Informe Final Instituto de investigación de recursos biológicos Alexander von Humboldt, Fondo Mundial para la Naturaleza Bogotá, Diciembre

Running SW, Mu Q, Zhao M, Moreno A (2017) User’s Guide MODIS Global Terrestrial Evapotranspiration (ET) Product (NASA MOD16A2/A3) NASA Earth Observing System MODIS Land Algorithm.

Running SW, Zhao M (2015) User's guide. Daily GPP and Annual NPP (MOD17A2/A3) Products NASA Earth Observing System MODIS Land Algorithm. University of Montana, Missoula, USA

Saito M, Miyata A, Nagai H, Yamada T (2005) Seasonal variation of carbon dioxide exchange in rice paddy field in Japan. Agricultural and Forest Meteorology 135:93-109

Sallaba F, Olin S, Engstrom K, Abdi AM, Boke-Olen N, Lehsten V, Ardo J, Seaquist JW (2017) Future supply and demand of net primary production in the Sahel. Earth System Dynamics 8:1191-1221

San José J, Montes R, Nikonova N, Grace J, Buendía C (2014) Effect of the replacement of a native savanna by an African Brachiaria decumbens pasture on the CO2 exchange in the Orinoco lowlands, Venezuela. Psyn 52:358-370

Sanchez-Cuervo AM, Aide TM (2013) Identifying hotspots of deforestation and reforestation in Colombia (2001–2010): implications for protected areas. Ecosphere 4:143 doi:10.1890/es13-00207.1
Sánchez-Cuervo AM, Aide TM (2013) Consequences of the Armed Conflict, Forced Human Displacement, and Land Abandonment on Forest Cover Change in Colombia: A Multi-scaled Analysis. Ecosystems 16:1052-1070 doi:10.1007/s10021-013-9667-y
Sandra G (2005) El Sistema Nacional de Áreas Naturales Protegidas de Colombia-SINAP. Parque Nacionales Naturales de Colombia, Bogota
Sarmiento G, Pinillos M (2001) Patterns and processes in a seasonally flooded tropical plain: the Apure Llanos, Venezuela. J Biogeogr 28:985-996
Scheffer M, Carpenter S, Foley JA, Folke C, Walker B (2001) Catastrophic shifts in ecosystems. Nature 413:591-596 doi:10.1038/35098000
Scheffer M, Carpenter SR, Lenton TM, Bascompte J, Brock W, Dakos V, Van De Koppel J, Van De Leemput IA, Levin SA, Van Nes EH (2012) Anticipating critical transitions. Science 338:344-348 doi:10.1126/science.1225244
Schmelzer M (2015) The growth paradigm: History, hegemony, and the contested making of economic growthmanship. Ecolog Econ 118:262-271 doi:https://doi.org/10.1016/j.ecolecon.2015.07.029
Schröter M, Barton DN, Remme RP, Hein L (2014) Accounting for capacity and flow of ecosystem services: A conceptual model and a case study for Telemark, Norway. Ecol Indic 36:539-551 doi:http://dx.doi.org/10.1016/j.ecolind.2013.09.018
Schröter M, Remme RP, Sumarga E, Barton DN, Hein L (2015) Lessons learned for spatial modelling of ecosystem services in support of ecosystem accounting. Ecosyst Serv 13:64-69 doi:http://dx.doi.org/10.1016/j.ecoser.2014.07.003
Schulp CJE, Alkemade R (2011) Consequences of Uncertainty in Global-Scale Land Cover Maps for Mapping Ecosystem Functions: An Analysis of Pollination Efficiency. Remote Sensing 3:2057-2075
Scurlock JMO, Johnson K, Olson RJ (2002) Estimating net primary productivity from grassland biomass dynamics measurements. Glob Chang Biol 8:736-753 doi:10.1046/j.1365-2486.2002.00512.x
Scurlock JMO, Olson R (2002) Terrestrial net primary productivity A brief history and a new worldwide database. Environ Rev 10:91-109
Searchinger T, Hanson C, Ranganathan J, Lipinski B, Waite R, Winterbottom R, Dinshaw A, Heimlich R, Boval M, Chemineau P, Dumas P, Guyomard H, Kaushik S, Markovsky D, Manceron S, Ben Ari T (2014) Creating a sustainable food future. A menu of solutions to sustainably feed more than 9 billion people by 2050. World resources report 2013-14 : interim findings. World Resources Institute,
Shao G, Wu J (2008) On the accuracy of landscape pattern analysis using remote sensing data. Landscape Ecol 23:505-511
Sharp R, Tallis, H.T., Ricketts, T., Guerry, A.D., Wood, S.A., Chaplin-Kramer, R., Nelson, E.,, Ennaanay D, Wolny, S., Olwero, N., Vigerstol, K., Pennington, D., Mendoza, G.,, Aukema J, Foster, J., Forrest, J., Cameron, D., Arkema, K., Lonsdorf, E., Kennedy, C.,, Verutes G, Kim, C.K., Guannel, G., Papenfus, M.,

Toft, J., Marsik, M., Bernhardt, J.,, Griffin R, Glowinski, K., Chaumont, N., Perelman, A., Lacayo, M., Mandle, L., Hamel,, P. V, A.L., Rogers, L., Bierbower, W.,. (2015) InVEST Version 3.2.0 User's Guide.
Sierra CA, Mahecha M, Poveda G, Álvarez-Dávila E, Gutierrez-Velez VH, Reu B, Feilhauer H, Anáya J, Armenteras D, Benavides AM, Buendia C, Duque Á, Estupiñan-Suarez LM, González C, Gonzalez-Caro S, Jimenez R, Kraemer G, Londoño MC, Orrego SA, Posada JM, Ruiz-Carrascal D, Skowronek S (2017) Monitoring ecological change during rapid socio-economic and political transitions: Colombian ecosystems in the post-conflict era. Environmental Science & Policy 76:40-49 doi:https://doi.org/10.1016/j.envsci.2017.06.011
Silva-Olaya AM, Davies CA, Cerri CEP, Allen DJ, Mello FFC, Cerri CC (2017) Quantifying above and belowground biomass carbon inputs for sugar-cane production in Brazil. Soil Research 55:640-648 doi:https://doi.org/10.1071/SR16090
Smets B, Buchhorn M, aLesiv M, Tsendbazar N-E (2017) Copernicus Global Land Operations “Vegetation and Energy”: User Manual.
Smith AMS, Falkowski MJ, Greenberg JA, Tinkham WT (2014) Remote sensing of vegetation structure, function, and condition: Special issue. Remote Sens Environ 154:319-321 doi:https://doi.org/10.1016/j.rse.2014.05.002
Smith WK, Zhao M, Running SW (2012) Global Bioenergy Capacity as Constrained by Observed Biospheric Productivity Rates. Bioscience 62:911-922
Sollenberger L, Burns J Canopy characteristics, ingestive behaviour and herbage intake in cultivated tropical grasslands. In: International Grassland Congress, 2001. Fealq Piracicaba, pp 321-327
Song Y, Jain AK, McIsaac GF (2013) Implementation of dynamic crop growth processes into a land surface model: evaluation of energy, water and carbon fluxes under corn and soybean rotation. Biogeosciences 10:8039-8066
Steduto P, Albrizio R (2005) Resource use efficiency of field-grown sunflower, sorghum, wheat and chickpea: II. Water use efficiency and comparison with radiation use efficiency. Agricultural and Forest Meteorology 130:269-281 doi:https://doi.org/10.1016/j.agrformet.2005.04.003
Steffen W, Broadgate W, Deutsch L, Gaffney O, Ludwig C (2015a) The trajectory of the Anthropocene: the great acceleration. The Anthropocene Review 2:81-98 doi:http://dx.doi.org/10.1177/2053019614564785
Steffen W, Persson Å, Deutsch L, Zalasiewicz J, Williams M, Richardson K, Crumley C, Crutzen P, Folke C, Gordon L, Molina M, Ramanathan V, Rockström J, Scheffer M, Schellnhuber HJ, Svedin U (2011) The Anthropocene: From Global Change to Planetary Stewardship. AMBIO: A Journal of the Human Environment 40:739-761 doi:10.1007/s13280-011-0185-x
Steffen W, Richardson K, Rockström J, Cornell SE, Fetzer I, Bennett EM, Biggs R, Carpenter SR, de Vries W, de Wit CA (2015b) Planetary boundaries: Guiding human development on a changing planet. Science 347:1259855 doi:10.1126/science.1259855
Suttie JM, Reynolds SG, Batello C (2005) Grasslands of the World. Food and Agriculture Organization of the United Nations (FAO), Rome

Suwarno A, Hein L, Sumarga E (2016) Who Benefits from Ecosystem Services? A Case Study for Central Kalimantan, Indonesia. Environ Manage 57:331-344 doi:10.1007/s00267-015-0623-9

Tallis H, Mooney H, Andelman S, Balvanera P, Cramer W, Karp D, Polasky S, Reyers B, Ricketts T, Running S, Thonicke K, Tietjen B, Walz A (2012) A Global System for Monitoring Ecosystem Service Change. Bioscience 62:977-986 doi:10.1525/bio.2012.62.11.7

Tallis H, Polasky S (2009) Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. ANYAS 1162:265-283

Team RStudio (2015) RStudio: Integrated Development for R. RStudio, Inc, Boston

TEEB (2018) TEEB for Agriculture & Food: Scientific and Economic Foundations. UN Environment, Geneva

Tian H, Chen G, Liu M, Zhang C, Sun G, Lu C, Xu X, Ren W, Pan S, Chappelka A (2010) Model estimates of net primary productivity, evapotranspiration, and water use efficiency in the terrestrial ecosystems of the southern United States during 1895–2007. For Ecol Manage 259:1311-1327

Tilman D, Balzer C, Hill J, Befort BL (2011) Global food demand and the sustainable intensification of agriculture. PNAS 108:20260-20264

Tilman D, Cassman KG, Matson PA, Naylor R, Polasky S (2002) Agricultural sustainability and intensive production practices. Nature 418:671-677

Tittonell P (2014) Ecological intensification of agriculture—sustainable by nature. Curr Opin Env Sust 8:53-61

Trenberth KE, Smith L, Qian T, Dai A, Fasullo J (2007) Estimates of the global water budget and its annual cycle using observational and model data. Journal of Hydrometeorology 8:758-769

Turner DP, Guzy M, Lefsky MA, Ritts WD, Van Tuyl S, Law BE (2004) Monitoring Forest Carbon Sequestration with Remote Sensing and Carbon Cycle Modeling. 33:457-466 doi:10.1007/s00267-003-9103-8

Turner DP, Ritts WD, Cohen WB, Gower ST, Running SW, Zhao M, Costa MH, Kirschbaum AA, Ham JM, Saleska SR (2006) Evaluation of MODIS NPP and GPP products across multiple biomes. Remote Sens Environ 102:282-292

UNCCD (2017) The Global Land Outlook, First edition edn. United Nations Convention to Combat Desertification, Bonn, Germany

UNFCCC (2002) Report of the Conference of the Parties on its seventh session, held at Marrakesh from 29 October to 10 November 2001 vol FCCC/CP/2001/13/Add.1. UNFCCC, 2001, Marrakesh, Morocco

United Nations, Convention on Biological Diversity, Norwegian Agency for Development Cooperation (2017) SEEA Experimental Ecosystem Accounting: Technical Recommendations Consultation Draft. United Nations New York

United Nations, Department of Economic and Social Affairs, Division P (2015) World Population Prospects: The 2015 Revision, Key Findings and Advance Tables.

United Nations, European Commission, Food and Agriculture Organization of the United Nations, International Monetary Fund, Organisation for Economic Co-operation and Development, World Bank (2014a) System of Environmental-Economic Accounting 2012: Central Framework vol 109. Studies in Methods. United Nations, New York

United Nations, European Commission, Food and Agriculture Organization of the United Nations, International Monetary Fund, Organisation for Economic Co-operation and Development, World Bank (2014b) System of Environmental Economic Accounting 2012: Experimental ecosystem accounting. United Nations, New York

United Nations, European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, World Bank (2009) System of National Accounts 2008. United Nations, New York

Valencia R, Ligarreto GA (2010) Mejoramiento genético de la soya (Glycine max [L.] Merril) para su cultivo en la altillanura colombiana: una visión conceptual prospectiva. Agronomía Colombiana 28:155-163

van Bussel LGJ, Grassini P, Van Wart J, Wolf J, Claessens L, Yang H, Boogaard H, de Groot H, Saito K, Cassman KG, van Ittersum MK (2015) From field to atlas: Upscaling of location-specific yield gap estimates. Field Crops Res 177:98-108 doi:http://dx.doi.org/10.1016/j.fcr.2015.03.005

van der Struijk LF, Kroeze C (2010) Future trends in nutrient export to the coastal waters of South America: implications for occurrence of eutrophication. Global Biogeochem Cycles 24 doi:10.1029/2009GB003572

van Ittersum MK, Cassman KG, Grassini P, Wolf J, Tittonell P, Hochman Z (2013) Yield gap analysis with local to global relevance—a review. Field Crops Res 143:4-17

van Noordwijk M, Brussaard L (2014) Minimizing the ecological footprint of food: closing yield and efficiency gaps simultaneously? Curr Opin Env Sust 8:62-70 doi:https://doi.org/10.1016/j.cosust.2014.08.008

van Oudenhoven AP, Petz K, Alkemade R, Hein L, de Groot RS (2012) Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecol Indic 21:110-122

Vargas L, Hein L, Remme RP (2017) Accounting for ecosystem assets using remote sensing in the Colombian Orinoco River Basin lowlands. Journal of Applied Remote Sensing 11:026008-026008 doi:http://dx.doi.org/10.1117/1.JRS.11.026008

Vargas L, Willemen L, Hein L (2018) Linking planetary boundaries and ecosystem accounting, with an illustration for the Colombian Orinoco river basin. Regional Environmental Change doi:10.1007/s10113-018-1282-1

Verhagen W, Van Teeffelen AJA, Baggio Compagnucci A, Poggio L, Gimona A, Verburg PH (2016) Effects of landscape configuration on mapping ecosystem service capacity: a review of evidence and a case study in Scotland. Landscape Ecol 31:1457-1479 doi:10.1007/s10980-016-0345-2

Verma SB, Dobermann A, Cassman KG, Walters DT, Knops JM, Arkebauer TJ, Suyker AE, Burba GG, Amos B, Yang H, Ginting D, Hubbard KG, Gitelson AA, Walter-Shea EA (2005) Annual carbon dioxide exchange in irrigated and rainfed maize-based agroecosystems. Agricultural and Forest Meteorology 131:77-96 doi:http://dx.doi.org/10.1016/j.agrformet.2005.05.003

Villa F, Bagstad KJ, Voigt B, Johnson GW, Portela R, Honzák M, Batker D (2014) A Methodology for Adaptable and Robust Ecosystem Services Assessment. PLoS ONE 9:e91001

Villamagna AM, Angermeier PL, Bennett EM (2013) Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecol Complex 15:114-121 doi:10.1016/j.ecocom.2013.07.004

Viloria De la Hoz J (2009) Geografía económica de la Orinoquia. Banco de la Republica, Cartagena de Indias

Vitousek PM, Ehrlich PR, Ehrlich AH, Matson PA (1986) Human appropriation of the products of photosynthesis. Bioscience 36:368-373

Wallace KJ (2007) Classification of ecosystem services: problems and solutions. Biol Conserv 139:235-246

Walter V (2004) Object-based classification of remote sensing data for change detection. ISPRS Journal of photogrammetry and remote sensing 58:225-238

Walters CJ (1986) Adaptive management of renewable resources. Macmillan Publishers Ltd, Basingstoke, UK

Wan Z (2008) New refinements and validation of the MODIS land-surface temperature/emissivity products. Remote Sens Environ 112:59-74

Wang Y, Zhang Y, Zhang R, Li J, Zhang M, Zhou S, Wang Z (2018) Reduced irrigation increases the water use efficiency and productivity of winter wheat-summer maize rotation on the North China Plain. ScTEn 618:112-120 doi:https://doi.org/10.1016/j.scitotenv.2017.10.284

Waring RH, Running SW (2010) Forest ecosystems: analysis at multiple scales. Elsevier, London

Watson AJ, Lenton TM, Mills BJW (2017) Ocean deoxygenation, the global phosphorus cycle and the possibility of human-caused large-scale ocean anoxia. Philos T Roy Soc A 375 doi:10.1098/rsta.2016.0318

Ecosystem Natural Capital Accounts: A Quick Start Package (2014) Secretariat of the Convention on Biological Diversity.

Willemen L, Burkhard B, Crossman N, Drakou EG, Palomo I (2015) Editorial: Best practices for mapping ecosystem services. Ecosyst Serv 13:1-5 doi:https://doi.org/10.1016/j.ecoser.2015.05.008

Woittiez LS, van Wijk MT, Slingerland M, van Noordwijk M, Giller KE (2017) Yield gaps in oil palm: A quantitative review of contributing factors. Eur J Agron 83:57-77

Wolf AT, Natharius JA, Danielson JJ, Ward BS, Pender JK (1999) International River Basins of the World. International Journal of Water Resources Development 15:387-427 doi:10.1080/07900629948682

Wood SLR, Jones SK, Johnson JA, Brauman KA, Chaplin-Kramer R, Fremier A, Girvetz E, Gordon LJ, Kappel CV, Mandle L, Mulligan M, O'Farrell P, Smith WK, Willemen L, Zhang W, DeClerck FA (2018) Distilling the role of ecosystem services in the Sustainable Development Goals. Ecosystem Services: Part A 29:70-82

World bank open data (2018) World Bank. https://data.worldbank.org/indicator/AG.LND.AGRI.ZS?end=2015&start=1961&view=chart. Accessed 12.03.2018

WRI, UNDP, UNEP, Bank W (2000) World Resources 2000–2001: People and Ecosystems: The Fraying Web of Life. World Resources Institute, Washington, DC

WRI, UNDP, UNEP, Bank W (2005) World Resources 2005: The Wealth of The Poor-Managing Ecosystems to Fight Poverty. World Resources Institute, Washington, DC

WRI, UNDP, UNEP, Bank W, cirad, INRA (2015) Creating a sustainable food future: A menu of solutions to sustainably feed more than 9 billion people by 2015. World Resources Institute, Washington D.C

Wu C, Gaumont-Guay D, Andrew Black T, Jassal RS, Xu S, Chen JM, Gonsamo A (2014) Soil respiration mapped by exclusively use of MODIS data for forest landscapes of Saskatchewan, Canada. ISPRS Journal of Photogrammetry and Remote Sensing 94:80-90 doi:http://dx.doi.org/10.1016/j.isprsjprs.2014.04.018

Zhang L, Lü Y, Fu B, Dong Z, Zeng Y, Wu B (2017) Mapping ecosystem services for China's ecoregions with a biophysical surrogate approach. Landsc Urban Plan 161:22-31 doi:https://doi.org/10.1016/j.landurbplan.2016.12.015

Zhao M, Running S, Heinsch FA, Nemani R (2010) MODIS-derived terrestrial primary production. In: Ramachandran B, Justice CO, Abrams MJ (eds) Land Remote Sensing and Global Environmental Change. Springer, Dordrecht, pp 635-660

Zhao M, Running SW, Nemani RR (2006a) Sensitivity of Moderate Resolution Imaging Spectroradiometer (MODIS) terrestrial primary production to the accuracy of meteorological reanalyses. J geophys res-biogeo 111 doi:10.1029/2004jg000004

Zhao M, Running SW, Nemani RR (2006b) Sensitivity of Moderate Resolution Imaging Spectroradiometer (MODIS) terrestrial primary production to the accuracy of meteorological reanalyses. Journal of Geophysical Research: Biogeosciences 111

Zwart SJ, Bastiaanssen WGM, de Fraiture C, Molden DJ (2010) WATPRO: A remote sensing based model for mapping water productivity of wheat. Agric Water Manage 97:1628-1636 doi:https://doi.org/10.1016/j.agwat.2010.05.017

Summary

To satisfy the needs of a growing global population the production of food, materials and energy have been unsustainably increasing during the last century. Such increase also accelerates the transformation of forests, savannahs and grasslands into agricultural fields, and the modification of the water, carbon, nitrogen and phosphorous cycles. Global changes on ecosystems have been ignored by current economic development strategies. However, the importance of ecosystems and ecosystem services as key determinants of human well-being and as components of the total wealth of each nation has been increasingly recognized by international initiatives such as the Millennium Ecosystem Assessment and the Intergovernmental Platform for Biodiversity and Ecosystem Services. A further step in understanding the connections between ecosystems and the economy has been the development of ecosystem accounting. Ecosystem accounting incorporates a conceptual framework describing changes on stocks and flows, where stocks comprise spatially explicitly defined ecosystems -*ecosystem assets*-, and flows -*ecosystem services*- embrace the material and non-material flows between ecosystems and from ecosystems to the economy. Assessing changes in ecosystems using ecosystem accounting entails a clear delineation of well-defined

boundaries that allow the organization of information and the presentation of accounts at a specific scale of analysis. To this end, cartographical and statistic information are required, including among others, land cover, meteorological, hydrological, soil, and population data. However, detailed data is often non-existent or scarce, inaccessible and expensive. Remote sensing provides timely data over large coverages and can be a useful source of spatially explicit data at relatively low cost. The objective of this thesis is to increase our knowledge on how remote sensing data can be used to support ecosystem accounting for the assessment of unsustainable changes in ecosystems in a large river basin.

In this thesis I use the enhanced vegetation index (EVI), the normalized difference water index (NWDI), land surface temperature (LST), and Net Primary Productivity (NPP), derived from the MODIS land surface products for the analysis of changes in ecosystems following the ecosystem accounting guidelines in a large river basin (Chapter 2). I use the MODIS land surface products as they are the outcome of modelling the results of lower-level data produced by the NASA EOSDIS Land Processes Distributed Active Archive Centre (LP DAAC). I find that the MODIS land surface products compiled following the ecosystem accounting guidelines are useful to analyse changes in ecosystems, in terms of extent, condition and capacity to supply ecosystem services. The MODIS land cover is a key product to define the boundaries of ecosystems and for the measurement of ecosystems extent. The annual periodicity of the MODIS land cover product is suitable for ecosystem accounting given that the length of the accounting period in ecosystem accounting is often one year with a reported overall accuracy of around 75% correctly classified. However, the spatial resolution of the MODIS land cover product ($500m^2$) is relatively coarse for a detailed monitoring of changes in small-sized ecosystems. Whereas the EVI and the NDVI are useful to monitor changes in ecosystem condition, the LST and the NDWI can be used to monitor water stress conditions, complementing vegetation indices that saturate at high values particularly in forested ecosystems. However, these indices are insufficient to assess the overall condition of ecosystems which requires the assessment of other relevant characteristics such as species diversity, structure, and nutrients availability.

I use the NPP derived from the MODIS NPP product and statistics to model changes in the capacity of ecosystems to supply ecosystem services following the ecosystem accounting guidelines (Chapter 3). I find that NPP derived from the MODIS can be compiled following the ecosystem accounting guidelines. Moreover, NPP can be used to monitor annual changes in the capacity of ecosystems to supply ecosystem services at the level of river basin in data-poor contexts. Using a spatially explicit approach in capacity-supply models is useful to highlight areas where ecosystem services are unsustainably supplied above their capacity. However, even though MODIS is the only source of NPP data publicly available at a global level, the spatial resolution of this product can be relatively coarse for local detailed assessments. Nevertheless, the high sensitivity of NPP to reflect changes in rainfall patterns, water, light and nutrients availability make NPP a useful indicator to assess changes in the capacity of ecosystems to supply ecosystem services. Although NPP is sensitive to changes in ecosystems condition and is highly correlated with the supply of provisioning ecosystem services, NPP is limited in assessing ecosystem services non-directly related to primary productivity particularly for regulating and cultural ecosystem services. In addition, the spatial arrangement of land cover types can constrain the use of MODIS NPP to assess ecosystems capacity to supply ecosystem services at a local level.

In Chapter 4, planetary boundaries and ecosystem accounting are presented as multidisciplinary integrated approaches that use quantitative frameworks to understand the complex dynamics of socio-ecological systems. The focus of this chapter is to explore if and how these two multidisciplinary approaches can be used in combination for sustainable natural resource management at the level of a large river basin. I use two sets of criteria to compare and contrast the planetary boundaries and ecosystem accounting frameworks providing a general overview based on contextual criteria and an in-depth comparison based on structural criteria. In addition, I assess the applicability of these frameworks for a sustainable natural resources management in the Colombian Orinoco river basin. A single framework cannot be used to address all issues of complex socio-ecological systems and hence the right framework has to be chosen based on the problem to be studied and on how the socio-ecological system is conceptualized. I find that the integration of multidisciplinary approaches is not only a further step to better understand the complexity of socio-ecological systems, such integration is also essential to inform policy and decision making. In particular, whereas the planetary boundaries framework facilitates the display of spatially heterogeneous interconnected processes as national issues, ecosystem accounting allows the incorporation of these issues into national policies.

Assessing the natural capital underpinning agricultural production is the focus of Chapter 5. In this chapter, I use concepts used in ecosystem accounting and yield gap analysis, remotely sensed data and agricultural statistics to assess changes in the natural capital of agricultural systems. In particular, the ecosystem accounting concepts of extent, potential and capability, and the concepts of water-limited crop potential yield, the yield gap and water productivity as used for yield gap analysis. I use remotely sensed data from the MODIS to estimate the extent of two agricultural systems, the capability of these two systems to produce six different crops and their water productivity. I use agricultural statistics to link the capability to produce crops with the annual yield of each crop. I find that whereas assessing changes in agricultural systems extent is useful to monitor the expansion of agricultural areas, assessing changes in water productivity enables monitoring areas with water stress conditions. Monitoring the expansion of agricultural areas and areas with water stress conditions is important to support the achievement of both food security and environmental sustainability. I find that the concept of capability is suitable for the assessment of agricultural systems because in these systems the production of crops and biofuels are prioritized from the supply of other ecosystem services. The ecosystem services not prioritized by agricultural systems include the supply of nutrients from the soil, soil formation, pollination and flood regulation. Although these ecosystem services highly influence crop production they are not as evident as the supply of food and biofuels. Linking capability and supply is essential to establish trade-offs between the supply of food, biofuels and fibers and to make those ecosystem services not prioritized such as carbon sequestration and pest regulation visible. Ecosystem accounting is a powerful tool to monitor changes in natural capital and to inform policy and decision making, supporting the transition of agricultural systems towards sustainability.

In summary, the MODIS land surface products used in this thesis are an important source of spatially explicit information to support ecosystem accounting in the assessment of unsustainable changes in ecosystems. Examples of how the MODIS products can be used to populate the extent, condition and capacity accounts have been demonstrated in the chapters of this thesis. Moreover, examples of how ecosystem accounting can be combined with other multidisciplinary quantitative frameworks and on how ecosystem accounting can be applied in the assessment of human-managed ecosystems have been also provided. The potential use of the moderate resolution sensor VIIRS and the high-resolution sensors on board the Landsat 8 and Sentinel satellites as a source of spatially explicit information to populate accounts was recognized in the synthesis chapter. Moreover, the potential use of other MODIS products such as the atmosphere, cryosphere and ocean products to expand the assessment of other ecological areas such as the atmosphere and the sea were identified in the synthesis chapter.

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About the author

Leonardo Vargas was born on 12 January 1976 in Bogota, Colombia. He finished his primary school in the capital city and the high school at the Colegio Departamental Honorio Angel y Olarte in the municipality of Pachavita (Boyacá), a small village in the countryside of Colombia. Living at the countryside was a life experience for him where the direct contact with nature, agriculture, and the simplicity of a peasant life shaped his lifestyle. After finish his high school Leonardo travelled to the Llanos, a region in the east side of the country where wild animals, natural savannahs, flooded forests, and cattle ranches coexist in a beautiful landscape. This stunning region motivated Leonardo to study veterinary medicine and zootechny at the University of the Llanos in Villavicencio where he finished his studies in 2001 and finally graduated in 2004. After 4 years of working as a veterinarian with cattle ranches, in 2006 Leonardo travelled to the north of The Netherlands to attain a short course in dairy farm management at the Practical Training Centre (PTC). This short course gave him the opportunity to experience the Dutch culture and lifestyle, and two years later, in 2008 Leonardo started a MSc in animal science at Wageningen University. During his MSc Leonardo wrote two major thesis. His MSc thesis at the Animal Nutrition group focused on the assessment of the nutritional quality of *erithrina edulis* a native bean from Colombia, and his MSc thesis at the Animal Production Systems group focussed on the assessment of ecosystem services and biodiversity of cattle farms in the Andes of Colombia. In May 2013, Leonardo started his PhD at the Environmental System Analysis where he focused on the use of remote sensing and ecosystem accounting for the assessment of changes in ecosystems in the Orinoco River Basin. This research assessed the use of remote sensed images from the MODIS as an alternative to populate ecosystem accounts in contexts where spatially explicit data is difficult to obtain. During his PhD Leonardo learned about Geographical information systems, remote sensing, cost benefit analysis, geo-statistics and ecosystem accounting, presented the results of his research on international conferences in Belgium and Scotland.

Awards

- In 2006, Leonardo received a NUFFIC fellowship for a short course at PTC in The Netherlands
- In 2008, Leonardo received a NUFFIC fellowship for a MSc at Wageningen University and Research
- In 2012, Leonardo received a COLCIENCIAS grant for a PhD at Wageningen University and Research

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Wageningen, 23 January 2019

The Chairman of the SENSE board	the SENSE Director of Education
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A K A D E M I E V A N W E T E N S C H A P P E N

The SENSE Research School declares that **Leonardo Vargas Barbosa** has successfully fulfilled all requirements of the Educational PhD Programme of SENSE with a work load of 45.9 EC, including the following activities:

SENSE PhD Courses

- Environmental research in context (2013)
- Techniques for Writing and Presenting a Scientific Paper (2014)
- High-impact writing in science (2015)
- Geostatistics (2015)
- Research in context activity: 'Creating clarifying and inspiring presentation on the working and importance of Ecosystems Services – in order to demonstrate the importance of understanding and involvement' (2018)

Other PhD and Advanced MSc Courses

- Cost-Benefit Analysis and Environmental Valuation, Wageningen University (2013)
- Introduction Geo-information Science, Wageningen University (2013)
- Remote Sensing, Wageningen University (2014)

Management and Didactic Skills Training

- Teaching in the MSc course 'Introduction to global change' (2015)
- Co-organization of PhD lunch meetings of ESA chair group (2016)

Poster Presentation

- *Accounting for ecosystem assets using remote sensing in the Colombian Orinoco River basin lowlands.* European Ecosystem Service Conference, 19-23 September 2016, Antwerp, Belgium

Oral Presentation

- *Accounting for ecosystem assets using remote sensing in the Colombian Orinoco River basin lowlands.* The international society for optics and photonics (SPIE) Remote Sensing + Security and Defence symposia, 26-29 September 2016, Edinburgh, Scotland

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Dr. Peter Vermeulen