UNIVERSITY OF SOUTHAMPTON

FACULTY OF SOCIAL, HUMAN AND MATHEMATICAL SCIENCES

Geography and Environment

The dynamics of socio-ecological systems in human-dominated landscapes: critical changes and continuing challenges in the Amazon estuary

by

Caio César de Araújo Barbosa

Thesis for the degree of Doctor of Philosophy

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THE DYNAMICS OF SOCIO-ECOLOGICAL SYSTEMS IN HUMAN-DOMINATED LANDSCAPES: CRITICAL CHANGES AND CONTINUING CHALLENGES IN THE AMAZON ESTUARY

Caio César de Araújo Barbosa

ABSTRACT

In this thesis I investigate the complex relationships between society, economy and nature taking place in the Amazon estuary, Northern Brazil. Throughout this work I use remote sensing data and techniques combined within the ecosystem services framework. Chapter 2 identifies the broad research gap concerning the effective assimilation of remote sensing into ecosystem services research. It provides a summary of what has been done, what can be done and what can be improved upon in the future to integrate remote sensing into ecosystem services research, laying out the main problems that are developed in the following chapters using the Amazon estuary as a case study. In chapter 3 I analyse the recent and increasing observed trends in forest cover change and ecosystem services as consequence of the interactions between political, economic and social factors. Through a cross-methodological approach, this section of the thesis exposes the political frontiers of forest cover change in the estuary and explores the spatially-explicit relationships linking the Green Vegetation Cover (GVC) to the availability of ecosystem services provided by forests. Such complex relationships are then captured by using an innovative approach used in economics to capturing the relationships between time-delayed variables. Chapter 4 presents the analysis of several time series in order to compare the case of the Amazon estuary to other similar relevant cases elsewhere in the world, providing important insight into the dynamics of social-ecological systems. This chapter, summarises and explore the state of slow and fast variables, observed drivers of change and recent trends in the Amazon, Mekong and Ganges Brahmaputra-Meghna deltas. The results show that there are various fundamental changes in many key ecosystem services, pointing to a changing dynamic state and increased probability of systemic threshold transformations in the near future. In chapter 5 I integrate a large and comrehensive set of social and economic context variables, aimed at understanding land use/cover transition processes in the Amazon estuary over the last three decades and how these might influence the estuarys landscape in the future. In this chapter I develop and apply an integrated modelling approach to capture intricate dynamics in the estuary. The results show that the modelling approach was able to identify and capture specific regional land/use cover dynamics in estuary, simulating the dynamic competition amongst different land use types under different scenarios.

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DECLARATION OF AUTHORSHIP

I, Caio César de Araújo Barbosa

declare that this thesis and the work presented in it are my own and has been generated by me as the result of my own original research.

The dynamics of socio-ecological systems in human-dominated landscapes in the Amazon estuary: critical changes and continuing challenges.

I confirm that:

1. This work was done wholly while in candidature for a research degree at this University;
2. Where I have consulted the published work of others, this is always clearly attributed;
3. Where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;
4. I have acknowledged all sources of help;
5. Where the thesis is based on work done by myself jointly with others, I have made clear exactly what was done by others and what I have contributed myself;
6. Part of this work has been published as:
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“Et voilà!”

Dedicated to the memory of Josefa Maria da Conceição

Definitions and Abbreviations

AFOLU	Agriculture, forestry and other land use
ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
AVHRR	Advanced Very High Resolution Radiometer
BBS	Bangladesh Bureau of Statistics
E.S	Ecosystem services
EDVI	Emissivity difference vegetation index
ESV	Ecosystem service values
FC	Forest cover
GBM	Ganges-Brahmaputra-Meghna
GDP	Gross domestic product
GIMMS	Global Inventory Modelling and Mapping Studies
GVC	Green Vegetation Cover
IAA	Integrated aquaculture-agriculture
IHOPE	Integrated History and future of People on Earth
INMET	Institute of Meteorology
LAI	Leaf area index
MA	Millennium Ecosystem Assessment
MODIS	Moderate Resolution Imaging Spectroradiometer
NEE	Net ecosystem exchange
NPV	Non-photosynthetic vegetation
NWDA	Natural wetlands distribution area

OG	Onset of greenness
PAR	Photosynthetically active radiation
PCA	Principal Component Analysis
PSI	Persistent Scatterer Interferometry
SAR	Synthetic aperture radar
SOM	Soil organic matter
SRTM	Shuttle Radar Topography Mission
TEV	Total Economic Value
TM	Thematic Mapper
TOPS	Terrestrial Observation and Prediction System
TSETSE	Two-storey equations of the transmission of solar energy

Chapter 1: Ecosystem services in large river deltas

1.1 Introduction

The concept of ecosystem services gained force in the scientific literature in the 1990s (Costanza et al., 1997; Costanza and Kubiszewski, 2012; de Groot, 2006; Egoh et al., 2007). This concept was made popular by the Millennium Ecosystem Assessment (MA) released in 2005 in which ecosystem services were defined as “benefits people obtain from ecosystems”(MA, 2005a). The conclusions from the global MA emphasized the importance of maintaining ecosystem services to human well-being, and the accomplishment of long-term development goals, including elements that can be used to support policy development and to identify research gaps (Fisher, 2012; Fisher and Brown, 2014; Kenter et al., 2015).

The MA came to four main conclusions on the present condition of ecosystem services, as follows: the world has been dramatically altered by human activity; ecosystem changes have led to considerable gains and equally considerable losses; continued damage caused to ecosystem services will make it harder to eradicate poverty; and ecosystem damage can be slowed and reversed, but it involves significant changes in policies, institutions, and practices (MA, 2005a; UNEP, 2014). At international level, evidence has demonstrated the economic benefits of sustainable ecosystem service management and that consumption of ecosystem services is not directly related to economic growth (de Araujo Barbosa et al., 2016b; Dearing et al., 2012b; Hossain et al., 2016; Zhang et al., 2015a). It is well known that the expansion of economic development into new areas can bring economic benefits but, in contraposition, economic growth without adequate attention to environmental and/or social impacts is unlikely to be converted into a higher state of human welfare (Cumming et al., 2014; de Araujo Barbosa et al., 2016e; Gauthier et al., 2015; Kirwan and Megonigal, 2013). Figure 1 represents how various categories of ecosystem services can be linked to human wellbeing.

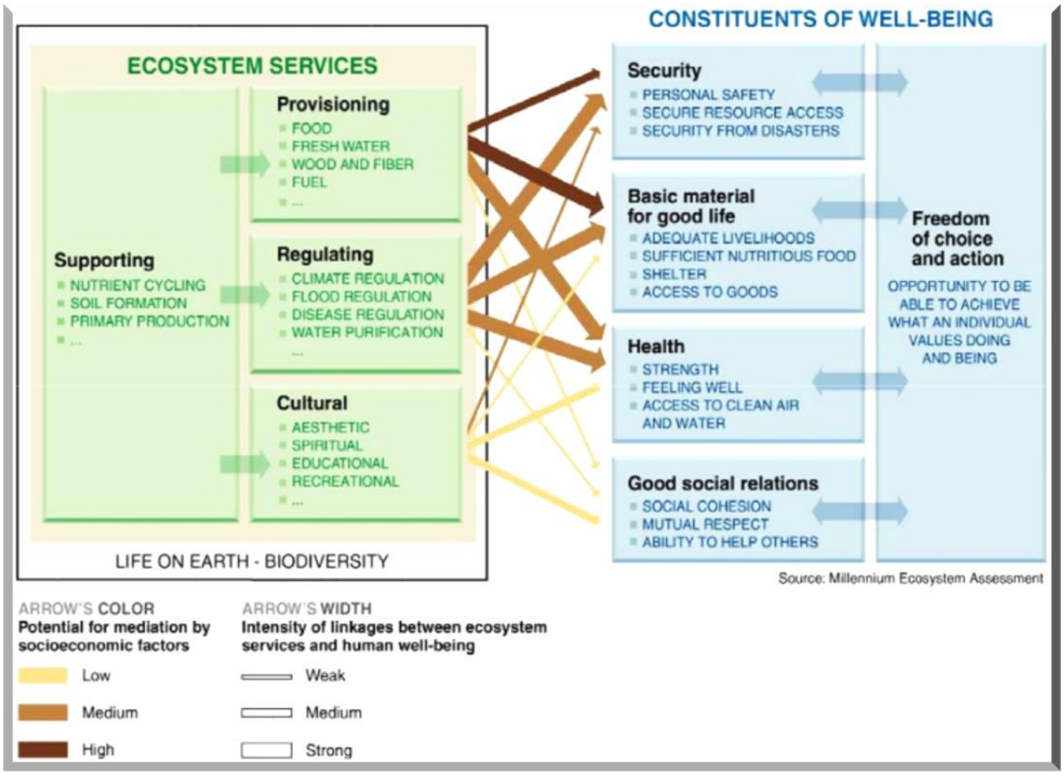


Figure 1.1 - Linkages between Ecosystem Services and Human Well-being (MA, 2005a)

While this categorization has been widely accepted, researchers have found it difficult to apply the conceptual framework in practice. In this sense, much research effort has been devoted to start conceptualising and identifying ecosystem services supplies, demands, values, components of ecosystem status, indicators, ecosystem processes, functions, and its direct and indirect relationships to human wellbeing (de Araujo Barbosa et al., 2015c, 2016b; de Araujo Barbosa et al., 2016e; Dearing et al., 2012b). Therefore, there is still a vast space for development of ecosystem services research, as to establish the basis for cross-sectoral, multi-scale and interdisciplinary approaches meant to build solid scientific foundations and practical frameworks (Fisher, 2012; Fisher and Brown, 2014; Kenter et al., 2015).

1.1.1 Valuation of ecosystem services

Numerous studies have shown the applicability and effectiveness of different methodologies in estimating the value of ecosystem services (Turner et al., 2016). In general, special emphasis has been given to the estimation of the monetary (and non-monetary) value of goods and services per unit area (Costanza et al., 1997; Turner et al., 2016; Turner et al., 2010). This has been aimed at helping to make the various ecosystem services more evident through their estimated economic values (Costanza et al., 1997;

Schaafsma, 2012 #2261). Even though it might not be possible to give monetary values to every single ecosystem good and service, an economic valuation is thought to help cover a wide range of services which are used by the different actors benefiting from the ecosystem as a whole (Costanza et al., 2014; Fisher and Turner, 2008(Schaafsma, 2012 #2261; Fisher et al., 2011; Fisher et al., 2009; Schaafsma et al., 2014b). However, the less linear and more robust quantitative relationships between biodiversity, ecosystem components, processes and services are still poorly understood (Costanza et al., 2014; Fisher and Turner, 2008(Schaafsma, 2012 #2261; Fisher et al., 2011; Fisher et al., 2009; Schaafsma et al., 2014b).

Criteria and indicators are essential to describe comprehensively the interaction between the ecological processes and components of an ecosystem and their services (Balmford et al., 2011; Fisher and Turner, 2008; Schaafsma et al., 2014a; Turner et al., 2016). The use of valuation methods for ecosystem services encompasses direct consumptive use values such as the value of timber, fish or other resources that ecosystems provide, and direct, non-consumptive use values such as those related to recreation and aesthetic appreciation (Camps-Calvet et al., 2016; D'Lima et al., 2016; Dennedy-Frank et al., 2016; Garcia et al., 2016; Walsh and Milon, 2016; Xu et al., 2016; Zoderer et al., 2016). The sum total of use and non-use values associated with a resource or an aspect of the environment is called Total Economic Value (TEV) (Carbone and Smith, 2013; Cui et al., 2012; Falk-Andersson et al., 2015; Finnoff and Tschirhart, 2008; Garcia-Llorente et al., 2012; Garcia-Llorente et al., 2011; Gurluk, 2006; Patterson and Cole, 2013; Tolunay and Bassullu, 2015; Werner et al., 2014). The estimated value of each ecosystem per hectare can be then multiplied by the area of each biome to find the Total Economic Value of ecosystem services provided (Figure 2) (Angelstam et al., 2013; Barrena et al., 2014; Boval et al., 2014; Brewer et al., 2016; Carbone and Smith, 2013; Cheng et al., 2012; Dutton et al., 2010; Faith et al., 2010; Gan et al., 2011; Ghosh and Mondal, 2013; Gurluk, 2006; Kontogianni et al., 2012; Nguyen et al., 2016; Povazan et al., 2014; Schaafsma et al., 2012; Tolunay and Bassullu, 2015).

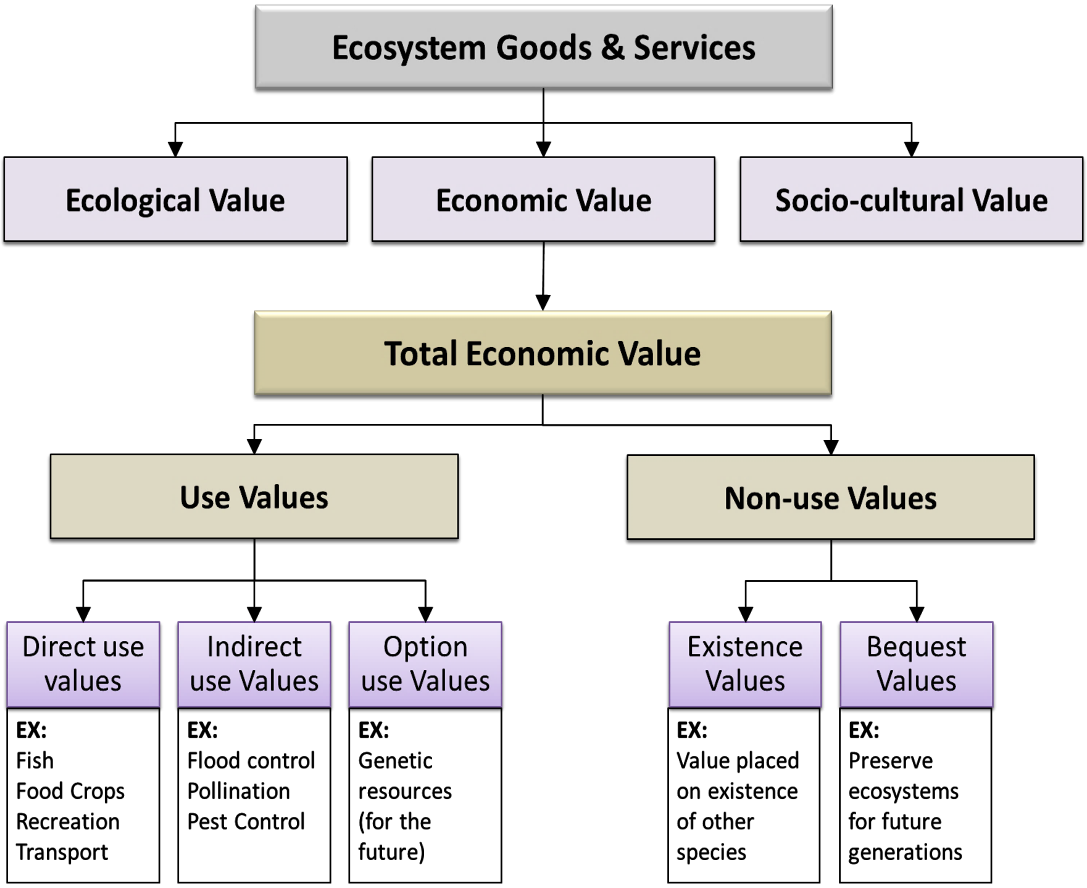


Figure 1.2– Framework of total economic value (modified from Peterson and Sorg (1987)).

Total economic valuation of ecosystem services is nevertheless a difficult and controversial task, and scientists have often been criticized for trying to "price tag" nature. However, economic valuation of ecosystem services can be useful, by providing a way to justify and set priorities for programmes, policies, or actions that protect or restore ecosystems and their services (Bieling et al., 2014; Mononen et al., 2016; Potschin and Haines-Young, 2011).

1.1.2 Ecosystem Services Supplies and Demands

The idea of ecosystem services supply refers to the capacity of a particular region to deliver a specific bundle of ecosystem goods and services, as well as the capacity to produce a set of natural resources realised and used at different spatial and time scales. This supply of multiple goods and services should be able to match the intrinsic demands of human societies, if self-sustaining human–environmental systems and a sustainable utilization of natural capital are to be achieved (MA, 2005a; UNEP, 2014). There is still much disagreement about the idea of ecosystem services supply and demand, and this is usually expressed using divergent definitions, for example: measuring of the equilibrium between demand and supply as habitually assumed in economics (Burkhard et al., 2012); others assume that the needs and preferences influence the behaviour in using, the experience, and consumption of a good or service and can, therefore also influence the willingness to protect certain services for the future (Schroter et al., 2014); another concept defines it as the amount of ecosystems goods desired or required per unit area over time (Villamagna et al., 2013).

Large scale changes in climate and land use might result in large changes in ecosystem service supply (Meyer et al., 2016). Some of these transformations may be positive (for example, increases in forest area and productivity) or offer opportunities (for example, surplus agricultural land intensification and production of Bioenergy) (Adesina et al., 2000; Angelsen, 2010; Bahadur, 2011; Bertolo et al., 2015). However, a constant rate of change in the natural environment will increase vulnerability as a result of a decreasing supply of ecosystem services, for example, declining soil fertility, declining water availability, increasing risk of forest fires and coastal erosion (de Araujo Barbosa et al., 2015c; Naidoo et al., 2008). Many authors agree that a more dynamic approach that accounts for ecosystem dynamics, including temporal change, climate and land use change, is still needed as it has the potential to be more effective and useful (Cumming et al., 2014; de Araujo Barbosa et al., 2016e; Gauthier et al., 2015; Kirwan and Megonigal, 2013). Therefore, being able to best describe respective states and dynamics, appropriate indicators and data, including quantitative and qualitative assessments of ecosystem services supplies and demands will become increasingly relevant in the near future (Boumans et al., 2002; de Araujo Barbosa et al., 2016e; Dearing, 2013; Du et al., 2015; Ghosh et al., 2016; Griffith and Fulton, 2014; Heimhuber et al., 2016; Levin et al., 2015; Ouyang et al., 2014; Thapa, 2012; Turner et al., 2015).

1.2 The dynamics of socioecological systems and ecosystem services in the Amazon estuary

Estuaries and coastal wetlands (e.g. Mangroves, salt marshes, mudflats, sand beaches and sea grass beds), are crucial transition zones between land, freshwater habitats, and the sea, capable of providing essential ecosystem services (Table 1.1) (de Araujo Barbosa et al., 2016b; Wang et al., 2012). Nevertheless, estuaries are among the most degraded ecosystems, for example, from pollution from fertilisers, run off from agricultural land, oil and gas production, mineral extraction, urban, industrial and port development, tourism industry and coastal aquaculture (Friess and Webb, 2014; Hoppe-Speer et al., 2015; Mukhopadhyay et al., 2015; Ren et al., 2011). Extensive economic activities exert high pressure on estuarine environments. Developmental activities such as mining and ore transportation can severely change metal levels in estuary waters and, together with the heavy human occupation, reduce nutrient productivity, which can lead to the disruption of the food chain. It is recognised that continued population growth and climate change will continue to put pressure on coastal ecosystems in the future, further reducing benefits and services (de Araujo Barbosa et al., 2016b; de Araujo Barbosa et al., 2016e; Hossain et al., 2016; Wang et al., 2012).

Table 1.1 – Ecosystem services provided by estuaries and deltas

Provisioning	Regulating	Habitat/Support	Cultural
Food (fish, fruit);	Good air quality (gas regulation and Carbon sequestration);		
Water availability	Storm protection;		Appreciated scenery;
Raw materials (Timber, fibres);	Flood prevention;	Nursery service;	Recreation and tourism;
Fuel and energy (fuel wood, organic matter, etc.);	Drainage and natural irrigation (drought prevention);	Maintain biodiversity;	Inspiration for art
	Clean water (waste treatment);	Photosynthesis, primary production,	Cultural heritage;
Genetic material;	Erosion prevention;	Water cycling;	Spiritual and religious use;
Drugs and pharmaceuticals;	Maintenance of productive and clean soils;	Maintaining; Biodiversity.	Use in science and education.
Resources for fashion; handicrafts, decoration, etc.	Pollination (biological control);		
	Pest and disease control.		

Adapted mainly from: Fisher et al. (2014) de Groot et al. (2012) Costanza et al. (1997); de Araujo Barbosa et al. (2015b); de Araujo Barbosa et al. (2016b).

The Amazon delta is located in the Amazon Basin, North Brazil, the largest river basin on Earth and also one of the least understood (Castello and Macedo, 2016; McGrath et al., 2001; Tian et al., 2000). The Amazon estuary in northern Brazil has an area of 294,000 km^2, it is located between the states of Para (PA) and Amapa (AP) and corresponds to no less than 4% of the entire Amazon Basin (de Araujo Barbosa et al., 2016b; Ericson et al., 2006b).

The estuary is one of the last frontiers and a land of opportunities for those seeking new enterprises in Brazil. Large fluxes of immigration to the Amazon estuary have been encouraged by Brazil's government in order to increase the population in the vast areas of wetlands and rainforests (Brondizio et al., 1994a; de Araujo Barbosa et al., 2016a; Padoch et al., 2008; Vogt et al., 2015). However, the problems associated with increasing population (e.g., increased poverty), represent now the largest challenge for the future of sustainable development and administration in the region (Assuncao et al., 2015; Brondizio et al., 1994a; de Araujo Barbosa et al., 2016c; Godar et al., 2012; McGrath et al., 2001; Padoch et al., 2008; Perz et al., 2009; Ramirez-Gomez et al., 2015; Salonen et al., 2014).

The Amazon estuary has high economic potential due its strategic location, optimally close to the sea and inland waterways, thereby, supporting intense economic activity (Fearnside, 2015c; Godar et al., 2012; Nepstad et al., 2014; Perz et al., 2009; Perz et al., 2015; Yamada and Gholz, 2002). However, the increasing pace of human development over the past several decades has strained environmental resources and produced extensive social and economic instability (Yamada and Gholz, 2002). Ecosystem services in this region are deteriorating rapidly due to human actions (e.g. deforestation, pollution of water and sediment flow reduction), climatic variability (e.g., more frequent and severe droughts), and resource exploitation (e.g., increased mining activity, increasing exports of forest products, agriculture and pasture expansion) (de Araujo Barbosa et al., 2016a; de Araujo Barbosa et al., 2016c; Fearnside, 2015c; Godar et al., 2012; Nepstad et al., 2014).

A considerable segment of the population living in the Amazon delta is directly dependent on the local extraction of natural resources for their livelihood (Brondizio et al., 1994a; McGrath et al., 2001). Areas sparsely inhabited may be exploited with few negative consequences for the environment (Aguiar et al., 2016a; Godar et al., 2012; Padoch et al., 2008; Soares et al., 2012; Tian et al., 2000). However, increasing pressure on ecosystem services is amplified by the large fluxes of immigrants from other parts of the country, especially from the semi-arid zone in Northeast Brazil to the lowland forests of the Amazon delta (Fearnside, 2015c; Padoch et al., 2008). Natural and anthropogenic perturbations in the Amazon basin will cause degradation of key forest ecosystem services, such as carbon storage in biomass and soils, the regulation of water balance and the modulation of local and regional climate patterns (de Araujo Barbosa et al., 2016c; do Vale et al., 2016; Langerwisch et al., 2016).

Is still poorly understood how deforestation, population growth and climate variability in the Amazon delta have changed the availability of ecosystem services (Grimaldi et al., 2014; Lima et al., 2014; Ramirez-Gomez et al., 2015). Therefore, it is still unclear how the current interactions between social and ecological systems influence the current and future availability of many important ecosystem services (Castello and Macedo, 2016; Coomes et al., 2016; Dias et al., 2016; Grima et al., 2016). Today, this represents a major challenge to environmental sustainability, in the sense that it plays a key role in reducing poverty and conserving the planet's life support system (Caviglia-Harris et al., 2016; Parry et al., 2010; Pinho et al., 2014; Satake and Rudel, 2007; Tallis and Polasky, 2009).

1.2.1 Mapping Ecosystem Services Dynamics in the Amazon estuary

Remote sensing has been used widely to quantify and map ecosystems processes and functions, at reasonably low costs, and with the capability of fast, frequent, and continuous observations, thus, providing suitable data for quantifying, monitoring and the modelling of complex environmental phenomena at varying spatial and temporal scales (Brown et al., 2014b, a; Grimaldi et al., 2014; Teferi et al., 2010). Satellite-based remote sensing has great potential coverage, repeatability and consistency through a combination of existing instruments and data, and has been successfully applied to measure and compare the cumulative effects of shifts in size and spatial configuration of coastal habitats, vegetation cover, surface temperatures, coastal water salinity, chlorophyll, biomass, total suspended matter over tropical regions (Brunier et al., 2016; Gavazzi et al., 2016; Hestir et al., 2016; Keith et al., 2016; Kim et al., 2016; Lee et al., 2016; Reid, 2016; Santos et al., 2016; Silva et al., 2016; Uhrin and Townsend, 2016).

Until now, the vast majority of studies applying remote sensing to map ecosystem services has focused on large spatial scale experiments, thus, providing important information and allowing quantitative estimation of the condition of ecological resources, the magnitude of stress, the exposure of a biological component to stress, and the extent of change in condition (Clarkson et al., 2013; Costanza et al., 1997; de Groot et al., 2012; Barbier, 2011). However, because the actual services and their values are site-specific, it is essential that the dynamics of ecosystem service change is measured across larger spatial areas and with sufficiently fine frequency of temporal coverage (Bagley et al., 2014; Dearing et al., 2014; Reyers et al., 2013; Tyukavina et al., 2015). Accurate mapping of ecosystem structure such as vegetation types, percentage of tree cover, erosion rates, stand age and density are essential not only to improve models, estimates, and predictions but also to gain insights into environmental management policies that consider ecological processes and ecosystem functioning (de Freitas et al., 2007; Feld et al., 2009; Mooney et al., 2013; Palm et al., 2014; Sullivan and Huntingford, 2009; Walker et al., 2008).

Quantifying and mapping ecosystem services provided by tropical estuaries is necessary to periodically determine their response to changes in socioecological determinants that respond differently to similar pressures and drivers (Bagley et al., 2014; Brown et al., 2014a; de Araujo Barbosa et al., 2014a; de Araujo Barbosa et al., 2016a; Dearing et al., 2014; Fezzi et al., 2015; Grimaldi et al., 2014; Palm et al., 2007; Teferi et al., 2010). Furthermore, it is recognised that continuous observations and a spatially explicit assessment of ecosystem services in estuaries and coastal regions is required, if quantifying and monitoring of indicators that can reflect fluctuations in flows of important services is to be successful (de Araujo Barbosa et al., 2016b; Dearing et al., 2014; Fezzi et al., 2015; Grimaldi et al., 2014). However, despite the necessity of such an approach in ecosystem services research, a lack of information and research gaps still exist, especially in the development of time series of ecosystem services indicators, and more complex time-dependent interactions between multiple drivers of change across different temporal and spatial scales (Barnett et al., 2009; de Araujo Barbosa et al., 2016c). Advancing our understanding requires research integrating multiple approaches and techniques, incorporating analytical and methodological procedures from physical and social sciences to measure, quantify, map and model ecosystem services in landscapes dominated by social and ecological systems.

1.3 Research aim and questions

This research is a contribution to the future of remote sensing and ecosystem services research, bringing structured scientific enquiry around debates that influence nature conservation, human and economic development. Therefore, the aim was to establish innovative strategies to measure and model social, economic and ecological determinants of ecosystem services availability in tropical estuaries and deltas. This is also a step forward in engaging scientifically with nature protection and social wellbeing in human-dominated landscapes.

1.3.1 Overarching Research Question

How do interactions between social and ecological systems affect ecosystem services flows in the Amazon estuary?

1.3.1.1 Sub-Research Questions

i. To what extent have ecosystem services in the Amazon delta changed over the last three decades?
ii. What are the main drivers of change in forest ecosystem services flows in the Amazon delta?
    a) How is it affected by local agents and decisions?
    b) How is it affected by global agents and decisions?
iii. How do the observed changes in the Amazon estuary compare to other cases?
    a) What are the overall trends in indicators of human wellbeing and ecosystem services?
    b) What are the environmental implications of specific ecological and social factors?
iv. What are the future implications for forest ecosystem services provision in the Amazon delta?

1.4 Chapter description and thesis structure

1.4.1 Chapter 2: systematic literature review

This chapter has been published as a peer reviewed scientific article in May 2015 in the journal Ecological Indicators, after thorough reviewing process and editing. My contributions to this paper constitutes its entirety, supported by supervisor's comments and feedback.

This chapter investigates the broad research gap on how integration of remote sensing technologies and ecosystem services concepts and practices can lead to potential practical benefits for the protection of biodiversity and the promotion of sustainable use of Earth's natural assets. Therefore, I review the last decades of research integrating concept/framework of ecosystem services into remote sensing science. Showing that such developments have led to a significant increase in the number of scientific publications, with the production of research papers referring directly to the topic of ecosystem services growing exponentially, following the publication of the Millennium Ecosystem Assessment.

This systematic review was aimed at identifying, evaluating and synthesising the evidence provided in published peer reviewed studies framing their work in the context of spatially explicit remote sensing assessment and valuation of ecosystem services. Initially, a search through indexed scientific databases found 5920 papers (between the years of 1960 and 2013) making direct and/or indirect reference to the topic of "ecosystem services" and only 211 papers with direct reference to the use of "remote sensing". This literature review does not include scientific material available in books, grey literature, extended abstracts and presentations.

In this chapter I quantitatively present the growth of remote sensing applications in ecosystem services' research, reviewing the literature to produce a summary of the state of available and feasible remote sensing variables used in the assessment and valuation of ecosystem services. The results provide valuable information on how remotely sensed Earth observation data are used currently to produce spatially-explicit assessments and valuation of ecosystem services.

Finally, using examples from the literature, we produce a concise summary of what has been done, what can be done and what can be improved upon in the future to integrate remote sensing into ecosystem services research. The reason for doing so is to motivate discussion about methodological challenges, solutions and to encourage an uptake of remote sensing technology and data where it has potential practical applications.

This chapter identifies the gaps and provides the basis on which the following chapter will explore and develop, using the amazon estuary as a case study.

1.4.2 Chapter3: The extent of changes in forest ecosystem services over the last three decades and the main drivers of change.

This chapter has been published as a peer reviewed scientific article in April 2016 in the journal Science of the Total Environment, after thorough reviewing process and editing. My contributions to this paper constitutes its entirety, supported by supervisor's comments and feedback.

This chapter presents a review and analysis of recent and increasing pressure on ecosystem services being maximised by a combination of environmental, economic and social elements. Therefore, presenting a cross methodological approach to identify the political frontiers of forest cover change in the estuary with consequences for ecosystem services loss.

To do so, I used a combination of data from earth observation satellites, ecosystem services literature, and official government statistics to produce spatially-explicit relationships linking the Green Vegetation Cover (GVC) to the availability of ecosystems services provided by forests in the estuary. More specifically, I used various time series of data on vegetation cover, governance indicators, currency exchange rates, exports of beef and forest products and enacted environmental laws integrated using a Pairwise Granger Causality (PGC) tests.

The results in this chapter show that that the continuous changes in land use/cover and in the economic state have contributed significantly to changes in key ecosystem services, such as carbon sequestration, climate regulation, and the availability of timber over the last thirty years.

1.4.3 Chapter 4: The Amazon estuary in comparison to other evolving deltaic systems

This chapter has been published as a peer reviewed scientific article in July 2016 on the Journal Sustainability Science, after thorough reviewing process and editing. My contributions to this paper constitutes its entirety, supported by co-authors and supervisor's comments and feedback.

This chapter shows how the analysis of available time-series in three deltas over past decades can provide important insight into the social-ecological system dynamics that result from complex interactions in these coastal regions.

This chapter uses time-series data from official statistics, monitoring programmes, and Earth observation data to summarise and explore recent trends, slow and fast variables, and observed drivers of change in the Amazon, Ganges-Brahmaputra-Meghna and Mekong deltas. This chapter also access and discuss what are the main positive and negative feedbacks coming from social and ecological factors, and their interactions over time, using an evolutionary approach that can address the inherent complexity of interactions between human wellbeing, provision of goods, and the maintenance of ecosystem services.

This chapter shows how the analysis of available time-series in tropical delta regions over past decades can provide important insight into the social-ecological system dynamics in deltaic regions. It also provides an exploratory analysis of the recent changes that have occurred in the major elements of the three tropical deltaic social-ecological systems, such as demography, economy, health, climate, food, and water.

The results show that there are various fundamental transitions in many key indicators of ecosystem services in these regions, pointing to a changing dynamic state and increased probability of systemic threshold transformations in the near future.

1.4.4 Chapter 5: The future of the Amazon estuary as a dynamic socioecological system

This chapter assimilates a large set of social and economic data variables to help understand the processes taking place in the Amazon estuary over the last three decades and how this might define the future of the region.

The work in this this chapter was intended to develop, apply and explore an spatially-explicit modelling approach aimed at capturing complex dynamics in the estuary. This modelling approach incorporates a diverse and carefully selected set of variables related to observed forest cover change, thus being able to identify and capture specific regional land/use cover dynamics over time.

The results show that the modelling approach was able to identify and simulate the dynamic competition between different land use types in two different scenarios (e.g. Business-as-usual and Alternative scenario). The business-as-usual (BAU) scenario shows that deforestation in the Amazon estuary will remain extensive and at a high rate, with municipalities prioritising reaching economic growth targets to the detriment of conserving forested land. The alternative scenario shows that at the current stage of development it is challenging but necessary to concentrate efforts in achieving alternative land use rationales (e.g. agriculture intensification rather than expansion, mixed agriculture/pasture) combined with better regulation and targeted policies.

1.4.5 Chapter 6: Reflections and conclusions

This chapter connects and discuss some of the key points presented in all research chapters. This chapter puts results, evidence and ideas in perspective, also pointing to the chapter's individual and combined contributions to the progress of scientific knowledge. This chapter reflects upon the knowledge and the important lessons drawn from the previous chapters, discussing the key points from chapters 2, 3, 4 and 5 that may prove useful in designing future integrative research initiatives. Here I concisely list and describe the main contributions to science as well as conclusions following from previous chapters.

Chapter 2: Remote sensing of ecosystem services: a systematic review

2.1 Introduction

The last decade has seen the rapid development of research on the topic of ecosystem services, and increasing awareness of the economic value of ecosystem goods and services among decision-makers and the general public (de Groot et al., 2010; Fish, 2011; Fisher et al., 2009; Morgan et al., 2008). Following the release of the Millennium Ecosystem Assessment in 2005, a significant increase in the number of scientific publications on the subject has been observed (Costanza and Kubiszewski, 2012; Egoh et al., 2007; Skourtos et al., 2010; White et al., 2010).

Preliminary work has made considerable advances in highlighting the links between social and environmental change influencing the capacity of ecosystems to maintain provisioning, regulating, supporting and cultural services (Costanza et al., 1997; de Groot, 2006; Deegan et al., 2012; Imhoff et al., 2004; Porras, 2012; Reid et al., 2006). Since then, spatially-explicit assessments have been used widely to map a multitude of ecosystems services, merging theory and practice to advance more effective sustainability and conservation actions (Alcaraz-Segura et al., 2013; Kienast et al., 2009; Liu et al., 2010; Murray et al., 2012; Sutton and Costanza, 2002; Xie and Ng, 2013).

Remote sensing plays an important role in the study of complex environmental interactions between natural and social systems, and has been used widely to quantify and map ecosystem properties and functions and infer ecosystem processes through a combination of existing instruments and data (Chambers et al., 2007; Chopra et al., 2001; Muraoka and Koizumi, 2009; Palacios-Orueta et al., 2012; Ustin et al., 2004). A key advantage of remote sensing is the capability to perform synoptic, spatially continuous and frequent observations resulting in large data volumes and multiple datasets (Atzberger and Rembold, 2013; Baraldi and Boschetti, 2012; Estreguil and Lambin, 1996; Knaeps et al., 2010; Lewis et al., 2013; Stolle et al., 2004). With the emergence of new and more sophisticated products, Earth observation data will continue to contribute extensively to research on modelling, mapping and valuation of ecosystem goods and services (Cabello et al., 2012; Marghany and Hashim, 2010; Nemani et al., 2009; Verburg et al., 2009; Zinnert et al., 2011).

There have been numerous efforts to classify the services provided by ecosystems (Costanza et al., 1997; Daily, 1997; de Groot et al., 2002; MA, 2005b), and more recent efforts on understanding the various contexts in which the ecosystem services concept are used have the potential to move a step closer to the establishment of a meaningful and clear classification system. Nevertheless, there has not yet been agreement on a single classification scheme that provides a meaningful and consistent definition for ecosystem services (Boyd and Banzhaf, 2007).

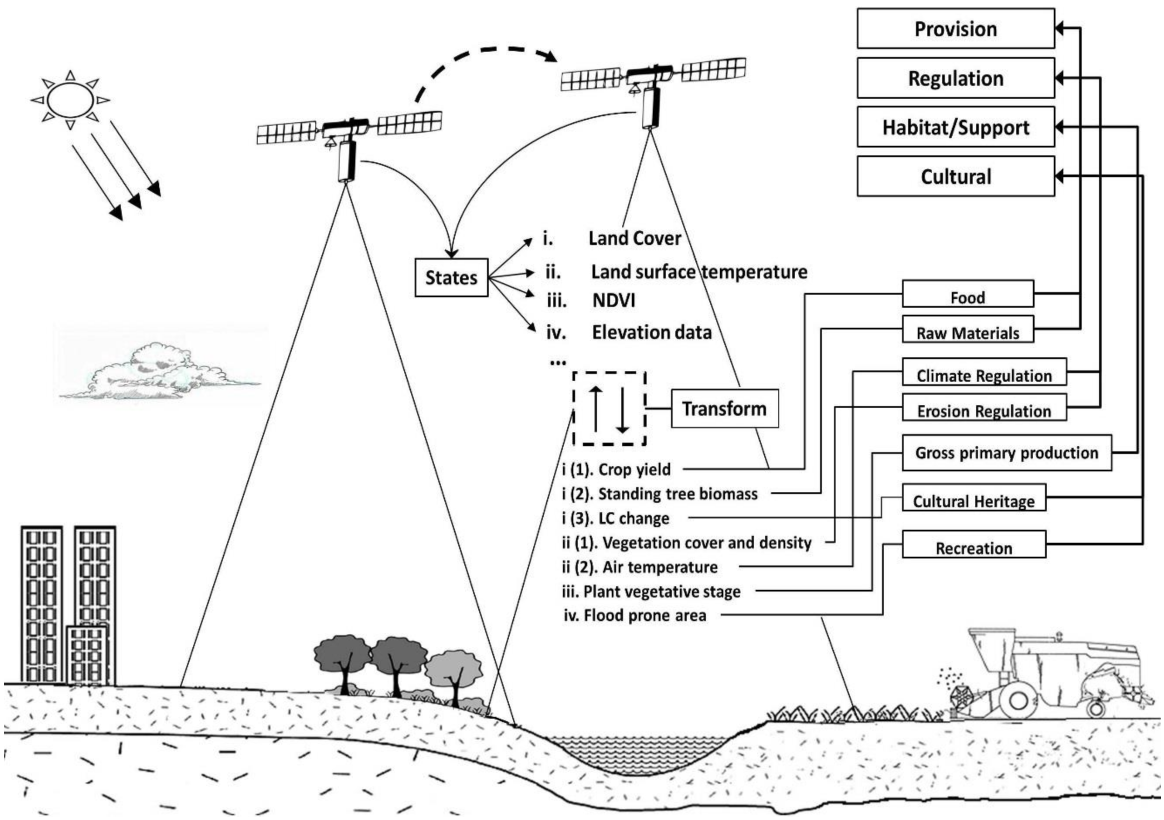


Figure 2.1 - The transform of remotely sensed data into ecosystem services values and flows (de Araujo Barbosa et al., 2015c).

This paper provides a systematic review of the scientific literature related to the use of remotely sensed data within the explicit context of ecosystem services assessment. We present examples from the literature organised to summarise (i) what has been done so far using remote sensing, (ii) what can be done in the future, and (iii) what can be further improved. For this purpose, the selected literature was reviewed systematically, searching for specific information regarding the remote sensing platform used, remote sensing product used (e.g. land cover, biophysical indices, etc..), temporal coverage, and other relevant more technical aspects of remote sensing.

2.2 Methodology

2.2.1 Searching and selection strategy

This review focused on papers that used ecosystem services concepts, and made use of Earth observation data to estimate value flows, and past and current availability of provisioning, regulating, habitat/supporting and cultural services.

The approach taken to querying the literature consisted of a selective keyword search in specific scientific libraries. At each query, terms and keywords such as 'Ecosystem Services', 'Ecological Services', 'Environmental Service' 'Remote Sensing' and 'Earth Observation were used individually to produce an extensive list of articles. Using Scopus and Web of Knowledge the body of literature was searched based on a fixed set of inclusion criteria:

(i) the literature should address ecosystems services as either main or secondary subject;

(ii) the predefined keywords should exist as a whole in at least one of the fields: title, keywords or abstract;

(iii) the paper should be published in a scientific peer-reviewed journal;

(iv) the paper should be written in the English language.

Figure 2.2 represents the literature search and selection process.

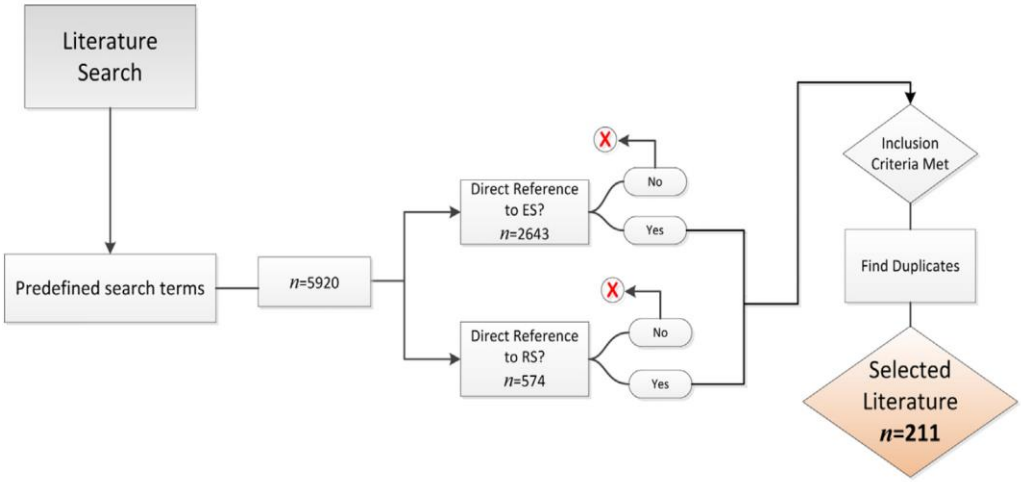


Figure 2.2 - Simplified literature search and selection flow process (de Araujo Barbosa et al., 2015c).

The initial steps of the search process in Figure 2.2 returned 5920 published articles on the topic of ecosystem services alone. The search was subsequently refined by querying through the first set of results. Searching for predefined keywords and terms related to the topic of remote sensing, the total number of entries was reduced to 211 scientific publications. During the data extraction process, the selected publications were scanned to identify and extract information in the form of descriptive text and identify labels for automated sorting using a custom bibliographic matrix. Ecosystem services found the publications were categorised according to the classification scheme proposed by the Millennium Ecosystem Assessment (MA, 2005b). which states that ecosystem services only exist when there are people benefiting from it, and therefore distinguishing ES from ecological processes and functions.

2.3 Results

2.3.1 Systematic review: what has been done

Total ecosystem services value depends not only on the natural phenomenon and location, but also on context and more subjective values related to societal wellbeing and cultural perceptions. In this context, current mapping methods based on remote sensing data are not sufficient alone for quantifying the full range of ecosystem services. At the same time, integration of ecosystem services valuation methods and remote sensing observation seems to be the only practical and realistic approaches to regionalise our understanding of complex interdependent socio-ecological systems. Where estimation is possible, translation from remote sensing data into estimates of certain ecosystem services values and flows may require data other than the spectral information itself. Therefore, if remotely sensed data are to be used to predict multiple ecosystem services it is required first to identify the nature of the relationship between the spectral information and the ecosystem service.

Integration of remote sensing data in the explicit context of ecosystem services assessment has enabled numerous studies on how different social and ecological systems are interacting over time. For example, using land-use/land-cover classifications from time-series of Landsat images many studies were able to measure deforestation rates and, thus, estimate changes in provisioning of raw materials and genetic materials, for example, in China, Argentina and the United Sates (Klepeis et al., 2013; Volante et al., 2012; Wang et al., 2006).

Similarly, remote sensing data classified into land cover class, combined with biophysical variables such as vegetation indices (VIs), were often used to quantify and map food production (e.g. Castella et al., 2013; Fegraus et al., 2012; Johnson et al., 2012). In this context, vegetation indices are used as a proxy for crop growth during the growing season, and then coupled with land cover and rainfall estimates to predict crop yield and enable forecasts of food production (e.g. Fuller, 1998; Gong et al., 2010; Ivits et al., 2012; Konarska et al., 2002; Palacios-Orueta et al., 2012).

Muukkonen and Heiskanen (2005) used reflectance data from the Advanced Spaceborne Thermal Emission and Reflection Radiometer (Dearing et al., 2014) together with a standwise forest inventory to estimate the biomass of boreal forest stands, using non-linear regression analysis and neural networks. Merging of VIs and land cover data may also be useful in creating models for prediction of the spatial distribution of species and biodiversity to estimate the availability of genetic materials (Barman et al., 2010; Dalezios et al., 2001; Moriondo et al., 2007; Nemani et al., 2009; Saatchi et al., 2007).

Using land cover classifications produced by applying a support vector machine classification to time-series of Landsat imagery, Brandt et al. (2012) investigated how changes in forest type, loss of old-growth forest and ecotourism development were affecting the availability of provisioning services (raw and genetic material), habitat services (biological refugia) and regulation services (pollination and biological control). They found that changes in management practices and politics had a great influence in increasing and decreasing of logging rates of high-diversity old-growth forest in the biodiversity hotspot of northwest Yunnan Province in southwest China.

Krishnaswamy et al. (2009) investigated the relationship between spectral distances and tree species diversity using a quantile–quantile plot method as an alternative to regression-based approaches. Spectral distances were used in modelling forest type variability assuming that the larger the spectral distance the greater the species diversity within sites. A scale-dependent relationship was found between spectral and the floristic distances between areas. This approach can be applied to quantify and map multiple ecosystems services related to habitat, biodiversity, water and carbon storage.

Combination of landscape metrics and time-series of land cover classifications from the Landsat Thematic Mapper (TM) and the Moderate Resolution Imaging Spectroradiometer (MODIS) matched with the proxy biomes from Costanza et al. (1997), has enabled numerous studies (e.g. Chen et al., 2007; Niu et al., 2012; Sutton and Costanza, 2002; Zhang et al., 2007; Zhao et al., 2004), with ecosystem service values (ESV) being associated to land cover classes assuming a linear relationship between land cover and market prices per unit area.

Other studies, aiming at quantifying and mapping the spatiotemporal variation in ecosystem services values, based their methodology on time-series of land cover data from satellite sensor imagery coupled with landscape fragmentation and urban sprawl analysis (Estoque and Murayama, 2013; Kreuter et al., 2001; Su et al., 2012), econometric calculations (Niu et al., 2012), historical data (Cai et al., 2013a) and meta-analysis of socio-economic variables (Camacho-Valdez et al., 2013).

Integration of suspended particulate matter concentration and aerosol optical thickness data (from MODIS-Aqua and MODIS-Terra), land cover (from Landsat and MODIS-Terra), nutrient cycling (from SeaWifs) and NDVI (From Landsat and MODIS-Terra) has been used to measure spatial and seasonal patterns of water provision and flood regulation services (Ghobadi et al., 2012; Muthuwatta et al., 2010), erosion regulation, climate regulation and maintenance of habitat services (Chopra et al., 2001; Gong et al., 2010; Ke et al., 2011) in India, China, Uganda, Iran, Mexico and the United States.

Furthermore, Nedkov and Burkhard (2012) used water retention functions (of the vegetation and soil cover), digital elevation models and land use information to assess the capacities of different ecosystems to regulate floods. The resulting maps represent estimates of regional supply-demand balances converted into the ecosystems' flood regulating service capacity for the Malki Iskar river basin in northern Bulgaria.

Multi-date imagery from the Sea-Viewing Wide Field-of-View Sensor (SeaWiFS), MODIS and Landsat sensors has been demonstrated to be effective in measuring and monitoring the cumulative effects of shifts in size and spatial configuration of coastal habitats, vegetation cover, surface temperatures, coastal water salinity, chlorophyll, biomass and total suspended matter of marine ecosystems (Cloern and Jassby, 2012; Doron et al., 2011; Paerl and Paul, 2012).

2.3.2 Numbers and spatial distribution of research

After the literature search, we found that the number of published research papers referring directly to the field of ecosystem services grew from 115 in 2005 to 5920 in 2013, denoting the rapid development of research within the field. The exponential growth of scientific publications in the field of ecosystem services research was followed similarly by an exponential increase in the number of published papers integrating Earth observation data in ecosystem services research (Figure 2.3).

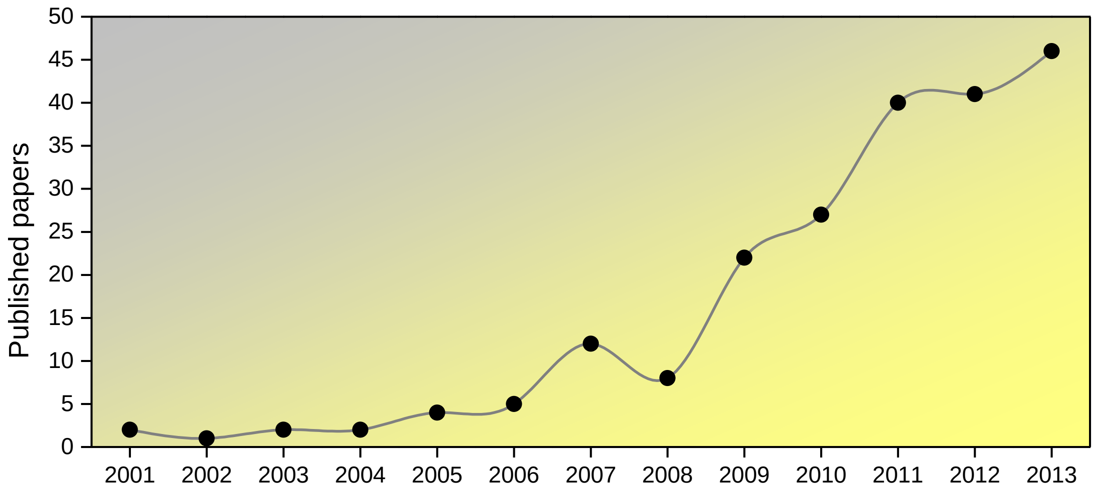


Figure 2.3 - Number of papers published annually between 2001 and 2013. This plot represents only those research articles which integrate Earth observation data into the explicit context of ecosystem services research.

The number of published research papers referring directly to ecosystem services and remote sensing has also increased significantly. The percentage increase in remote sensing and ecosystem services integrated studies relative to the total scientific production in the field of ecosystem services is presented in Figure 2.4.

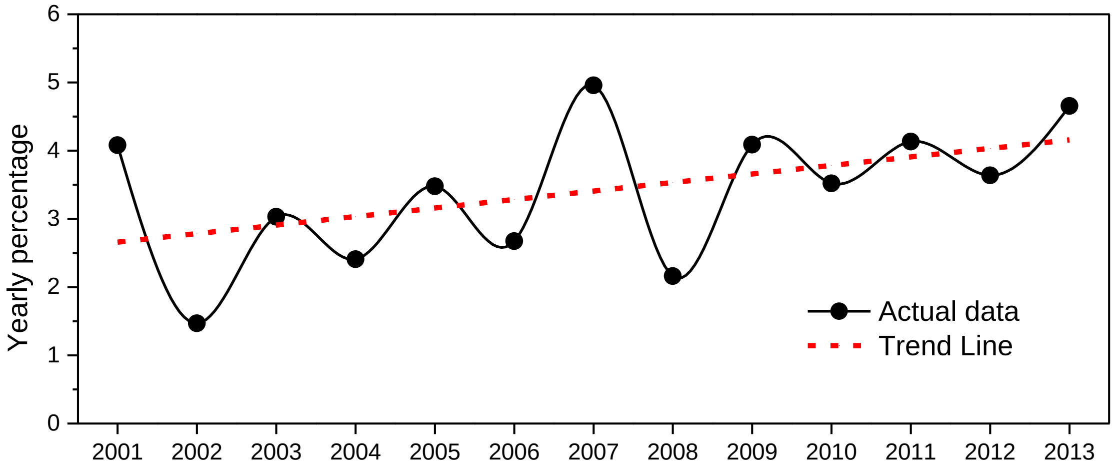


Figure 2.4 - Percentage of publications integrating remote sensing and ecosystem services relative to the total scientific production in the field of ecosystem services annually. The fitted regression line clearly indicates a positive trend.

To identify biophysical and socio-economic drivers of change in ecosystem service supply and demand, as well as long and short term trends, it is essential that the temporal coverage be of sufficient length and resolution. In particular, to monitor effectively the effects of time-sensitive socio-ecological interactions, it is necessary to use a remote sensing platform with a sufficiently frequent revisit period. Nevertheless, this is not an easy task since the availability of remote sensing data for long-term monitoring purposes is constrained by sensor characteristics (e.g. revisit time) and environmental factors (e.g. cloud cover). From Figure 2.5 it is clear that the great majority of published material has focused on a small temporal extent. Several articles analysed more than 20 years temporal extent, but the number still falls short of what is required to perform integrated analyses, aiming at continuous long-term monitoring of changes in ecosystem services supply and demand.

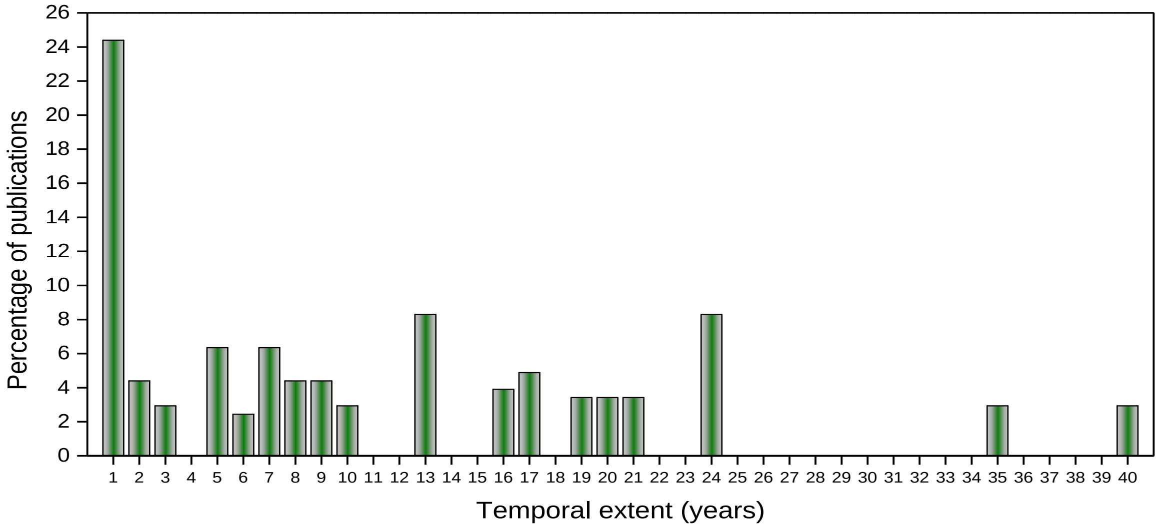


Figure 2.5 - Temporal extent of studies identified within the literature selected.

The ecosystem services (and respective categories of services as defined by the Millennium ecosystem assessment (MA, 2005b)) identified within the selected literature are presented in Figure 2.6 in such a way that it is possible to visualise which services had a greater presence in research integrating Earth observation data into ecosystem services research.

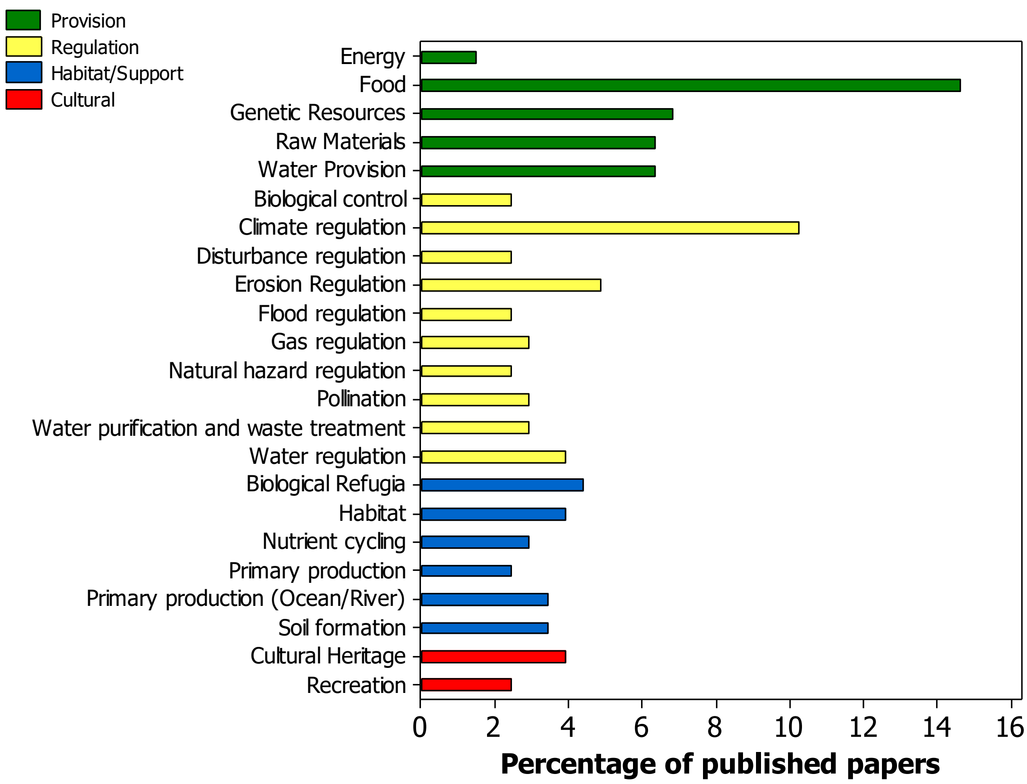


Figure 2.6 - Percentage of publications using remote sensing to assess provisioning, regulating, habitat/support and cultural ecosystem services.

From the map in Figure 2.7 it is evident that considerable gaps in geographical focus still exist around the world especially, rather surprisingly, in the African continent. Other areas lacking studies integrating remote sensing into ecosystem services assessment exist in Eastern Europe, Central America and the Middle East.

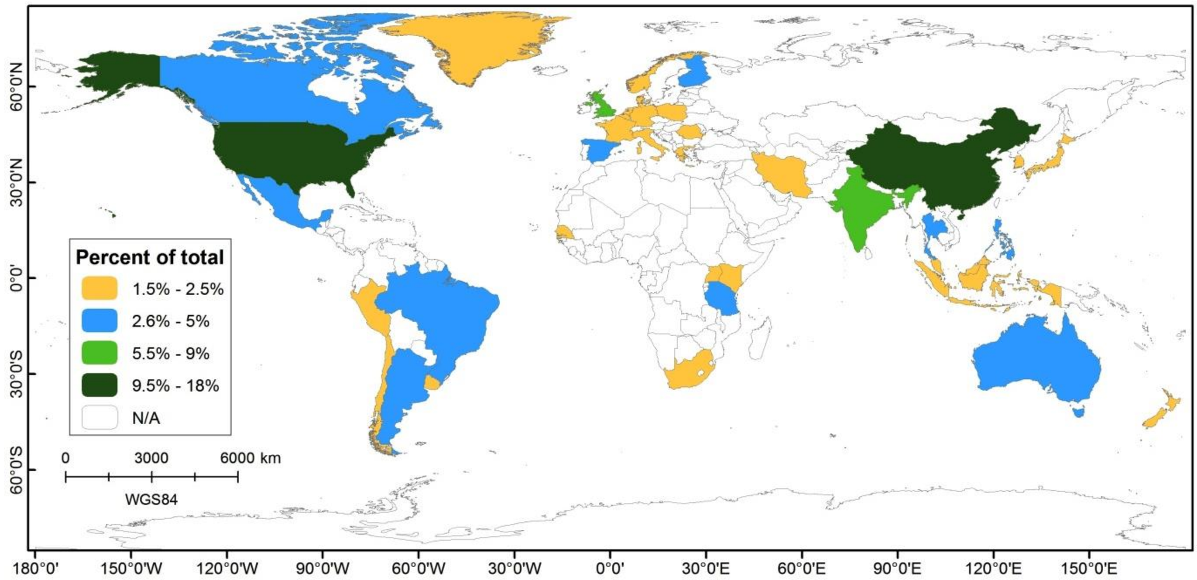


Figure 2.7 - Percentage of total published papers per study site integrating satellite remote sensing data within ecosystem services assessment.

2.3.3 Remote Sensing proxy data used in ecosystem services research

Figures 2.8, 2.9, 2.10 and 2.11 show how published works use remotely sensed biophysical properties to map and quantify provisioning (Figure 2.8), regulating (Figure 2.9), habitat/support (Figure 2.10) and cultural (Figure 2.11) ecosystem services. The figures indicate that the biophysical properties used generally differ between each category of ecosystem service.

For provisioning services (Figure 2.8), the categories most well covered are water provision and food provision where seven remote sensing /proxies were used, the least well represented categories are genetic resources and energy where only two proxies were used respectively. The dominant ecosystem services proxy variables are respectively land cover (≈58%) and NDVI (≈ 35%).

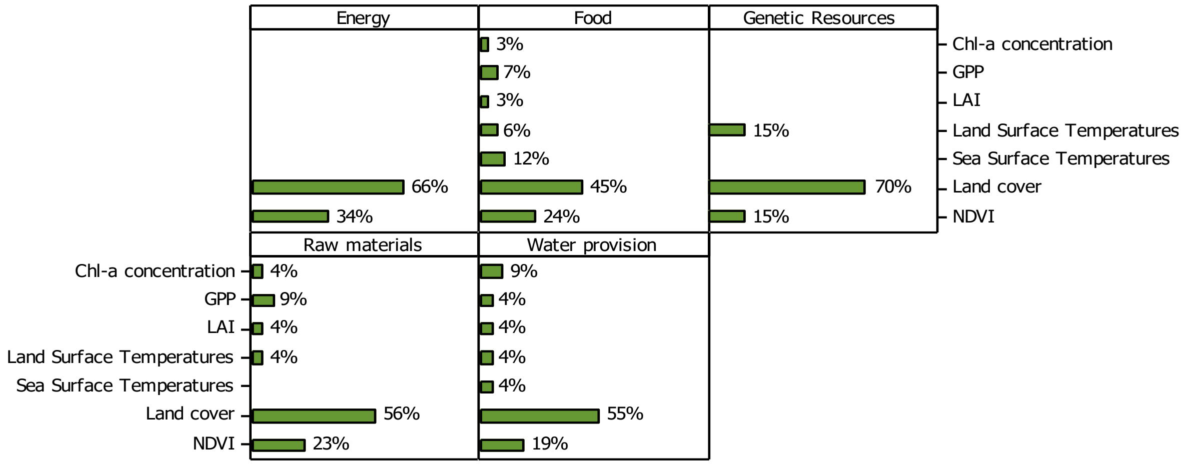


Figure 2.8 - Identified remote sensing data within the selected literature used to assess ecosystem provisioning services.

For regulating services (Figure 2.9), climate regulation and water purification and waste treatment are the most well covered categories where respectively eight and five remote sensing /proxies were used, important services such as biological control, disturbance regulation and gas regulation are the least well represented categories where only three remote sensing proxies were used. Land cover is the dominant proxy data used among all the categories, followed by NDVI.

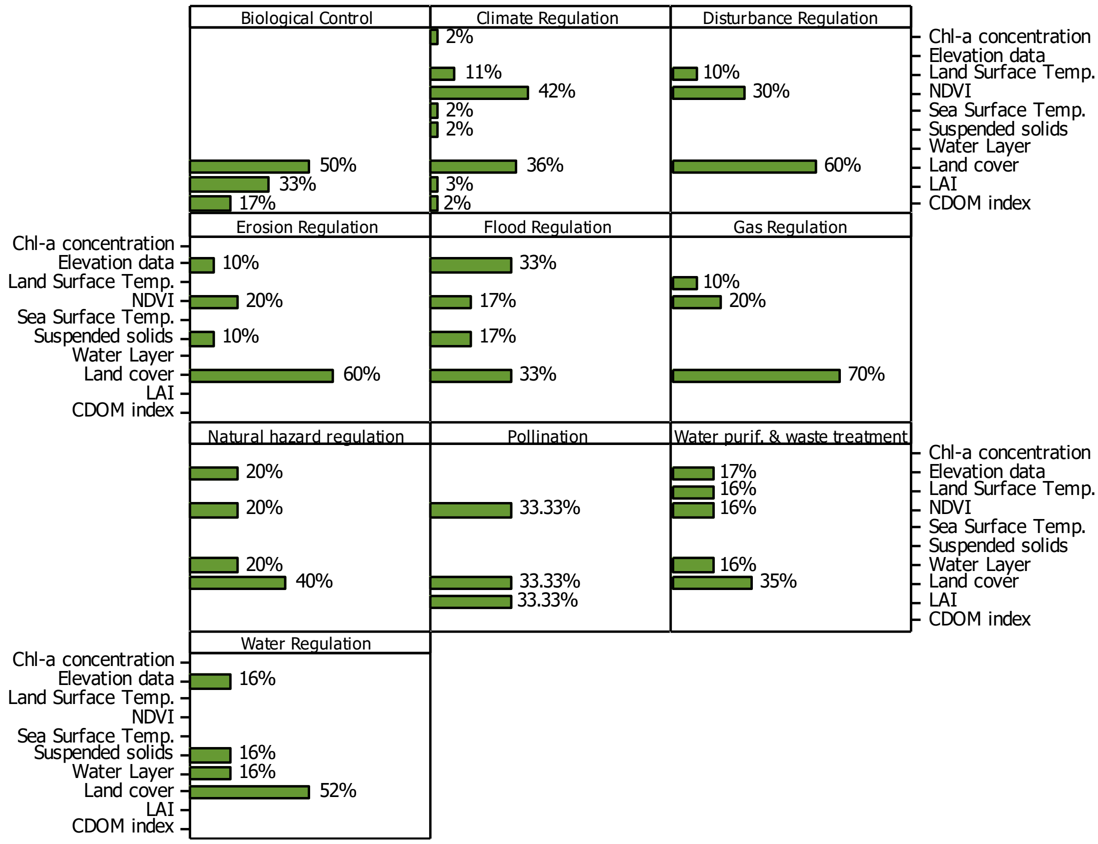


Figure 2.9 - Identified remote sensing data within the selected literature used to assess ecosystem regulating services.

For Habitat/Support services (Figure 2.10), primary production (for marine and riverine environments) and biological refugia are the most well covered categories where seven remote sensing /proxies were used, soil formation, primary production (land) and nutrient are the least well represented categories with only three remote sensing proxies used. Again, land cover is the dominant proxy data used among all the categories, now NDVI and LAI biophysical data have a significant presence.

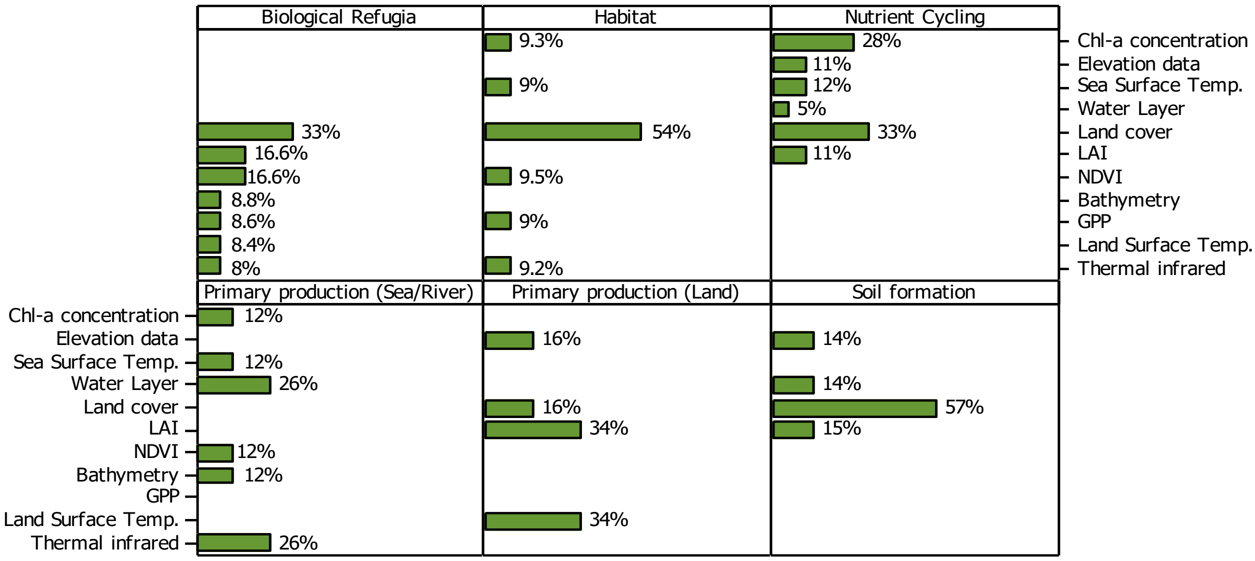


Figure 2.10 - Identified remote sensing data within the selected literature used to assess ecosystem Habitat/Support services.

Currently for cultural ecosystem services (Figure 2.11) there is still a lack of remote sensing proxy data in use, with recreation services being covered by three proxy data and recreation services being covered by only two remote sensing proxy data. Land cover appears as dominant among all the categories, followed by NDVI. In the case of cultural ecosystem services, it is in many ways challenging to produce satellite measurements of more subjective values related to societal wellbeing and cultural perceptions.

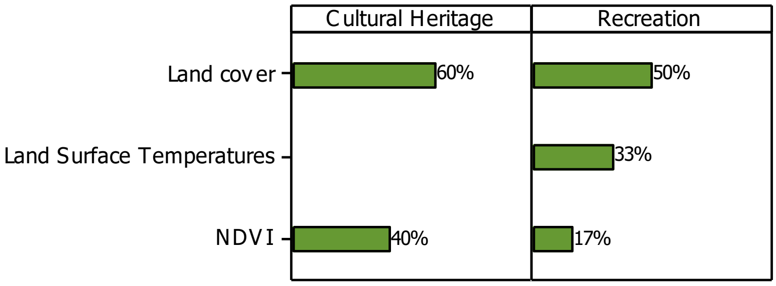


Figure 2.11 - Identified remote sensing data within the selected literature used to assess ecosystem cultural services.

2.3.4 Potential of remote sensing: what can be done

Climate regulating services (e.g. carbon storage and sequestration) can be estimated through quantification of net ecosystem exchange (NEE) of CO_2 flux, as NEE can be applied to determine the amount of atmospheric carbon stored in an ecosystem (Potter et al., 2008; Soojeong et al., 2006; Van Tuyl et al., 2005). Additionally, vegetation indices such as the NDVI, emissivity difference vegetation index (EDVI) and water band index, as well as the leaf area index (LAI), can be used as indicators of net CO_2 flux (Dagg and Lafleur, 2010; Hao et al., 2012; Kross et al., 2013; La Puma et al., 2007; Olivas et al., 2010).

Air quality can be measured using trace gases and aerosols as indicators, since aerosol data, coupled with simulation models, enable the source of polluting air to be identified. Additionally, a change in the amount of dust build-up in the air will change the spectral reflectance of objects. Thus, pollutants can be detected from variation in the spectral reflectance (Chu et al., 2003; Emeis and Schaefer, 2006; Mozumder et al., 2013; Roots et al., 2011; Yong et al., 2010).

Maintenance of soil fertility can be estimated from remotely sensed indices such as the non-photosynthetic vegetation (NPV), green vegetation (Muller et al.), NDVI and soil spectral mixture analysis, as an increase in NPV is suggested to be related to a decline in soil fertility (Numata et al., 2003). Moreover, reflectance data (within wavelengths 0.425 - 0.695 µm) can be related to soil organic matter (SOM) and used as a proxy for soil fertility (Ji et al., 2012; Liu et al., 2013a; Numata et al., 2003). Other mineral features present in soils and detectable by remote sensing (e.g. salinity, iron-oxide content, and heavy metal contamination) change the reflectance properties of soils on bare lands and the surrounding vegetation indicating the past and current status of soils (Kemper and Sommer, 2002; Melendez-Pastor et al., 2010; Taylor et al., 2001; Wu et al., 2005).

Erosion control regulation can be measured with moderate resolution optical imagery (e.g. Landsat Thematic Mapper and Landsat Enhanced Thematic Mapper) by identifying the reflectance properties of the constituents of sediments (e.g. lithological composition, grain size and moisture content) enabling detection of eroded land and material deposition (Small et al., 2009). Vegetation indices and elevation data are suitable for predicting soil erosion risk, as mapping variability in vegetation cover and plant residue can help in revealing areas more or less prone to erosion. Remote sensing-based soil erosion models integrate NDVI, vegetation fraction cover, slope gradient and land use (from SRTM data) to estimate annual soil erosion rates (Hochschild et al., 2003; Rahman et al., 2009; Wang et al., 2013; Wu et al., 2011).

The diffuse attenuation coefficient (Maghradze et al.) for photosynthetically active radiation (PAR) can be utilised to assess water purification capacity of ecosystems; since the availability of light is fundamental for phytoplankton and seagrass production, light attenuation can be used as a measure of turbidity or water clarity (Chen et al., 2009; Liu et al., 2013b). Thus, correlating satellite-derived reflectance data with *in situ* measured light attenuation enables assessment of water clarity and, hence, the water purification capacity of ecosystems (Behrenfeld and Boss, 2006; Zhao et al., 2011; Zhu et al., 2009).

Earth observation data can be used to quantify the production capacity of forests and agro-ecosystems using biomass as an indicator. Narrowband and broadband vegetation indices such as NDVI, EVI, fPAR and LAI can be used as indicators of productivity in a crop growing season, as they are able to show variation in phenology and photosynthetic potential of crops and help identify the cropping cycle and growth (Brown and de Beurs, 2008; Kastens et al., 2005; Ling et al., 2009; Muukkonen and Heiskanen, 2005; Prabakaran et al., 2013; Wall et al., 2008a; Wardlow and Egbert, 2008).

Potential sources of renewable energy such as solar power can be identified using daily surface insolation data and atmospheric transmittance extracted from optical and thermal infrared bands in addition to models of aerosol optical depth and columnar ozone (Bhattacharya et al., 2013; Morelli et al., 2012; Perez et al., 1994; Roujean, 1998). In coastal regions, wind speed retrieval from radar imagery such as synthetic aperture radar (SAR) provides spatial patterns of wind speed useful in mapping offshore wind resources, as backscatter coefficients are physically related to wind speed through the roughness of the open sea (Hasager et al., 2002; Johannessen et al., 1999; Kozai and Ohsawa, 2007; Xiulin et al., 2013).

Potential mass flow and rain-induced shallow landslide probability predicted using indices based on remote sensing images, such as vegetation indices, the soil brightness index, principal components transformation and terrain properties (Hyun-Joo et al., 2012; Shuwen et al., 2012; Yang et al., 2013) can be used to assess natural hazard regulation services. Furthermore, radar interferometry (using SAR images) can be applied with image correlation techniques to generate displacement maps (e.g. the Line of Sight direction and the Azimuth direction), resulting in information that can be combined routinely to undertake landslide ground surface deformation monitoring by measuring the three dimensional surface displacement field at different epochs (Colesanti and Wasowski, 2006; Noferini et al., 2007; Raucoules et al., 2013; Righini et al., 2012; Tantianuparp et al., 2013).

Remote sensing techniques applied to differentiate the composition of landscape/seascape can enable quantification of nutrient input and cycling (Asner et al., 2004; Spillman et al., 2007). Water quality parameters such as dissolved oxygen, chlorophyll-a and turbidity are closely linked with changes in absorbance of natural radiation and can be detected by passive remote sensors. Phosphorous concentration levels are also used to track spatiotemporal nutrient cycling patterns across seascapes. Other remote sensing data used to estimate nutrient cycling are related to measures of forest fragmentation and homogeneity, soil moisture and surface temperatures (Chang and Xuan, 2011; Karl, 2002).

Monitoring and predicting wind storms is of particular importance in assessing the capacity of ecosystems to mitigate the negative effects of storms on coastal areas, croplands and associated infrastructure. Using satellite sensor data it is possible to accomplish this task; for example, using the MODIS reflective and emissive bands and artificial neural networks (Brakenridge et al., 2013; El-ossta et al., 2013; Notaro et al., 2013). Moreover, satellite sensor observations of damage caused by storms can also be used to assess an ecosystem's ability to regulate wind storms, since the damage extent on infrastructure and vegetation will most likely vary with the wind speed and strength. Valuable information about storm events (e.g. the trends in moderate-duration of storms and mean sea level) can also be derived from TOPEX/Poseidon and Jason satellite altimetry data (Bhaskaran et al., 2013; Han et al., 2012; Sadhuram et al., 2006)

Combination of remote sensing data, stochastic and multi-agent models has clear advantages in modelling and simulating land cover change, soil erosion, food production, and vegetation dynamics (Aitkenhead and Aalders, 2011; Evans and Kelley, 2008; Linard et al., 2013). Agent-based and stochastic modelling are potentially suitable approaches for expanding our knowledge about landscape functioning, climatological variability, biophysical constraints, economic and social dynamics and, therefore, are capable of providing insights into how highly complex systems behave and interact, impacting the availability of ecosystem services (Brown et al., 2008; Evans and Kelley, 2008; Frank et al., 2012; Jepsen et al., 2006; Wada et al., 2007).

2.4 Discussion: suggestions for improvements

The use of remote sensing within ecosystem services research is still limited, and before 2009 this topic had a relatively small presence in the literature. Nevertheless, the general trend indicates that the number of scientific publications integrating remote sensing into ecosystem services studies is rapidly increasing. This is no surprise given the possibilities for integrating spatial data into ecosystem studies and establishing relationships between remote sensing data and socio-ecological phenomena. The increasing exponential rate of integrated scientific studies is expanding our knowledge and pointing to areas for future development.

2.4.1 Correcting data

A few points of connection were found relevant within the selected publications. For example, in 85% of the publications there was no clear indication of whether the effects of the atmosphere on the reflectance values was taken into account prior to the analysis of the remotely sensed data for ecosystem services assessment (e.g. land cover classification process). Within the 15% of papers that claim to have performed atmospheric correction, no reference was made to the algorithm applied. Many individual studies have focused on mapping changes in the delivery of ecosystem services through time. However, to maximize the benefits of doing so, change detection should be done automatically in an operational environment, and the resulting information should be sufficiently reliable that it can be related effectively to what is observed on site (Beresford, 1990; Coppin et al., 2004; Petit and Lambin, 2001; Sesnie et al., 2008; Song et al., 2001; Zeng et al., 2012). This implies an operational atmospheric correction procedure to correct for the effects of aerosols, Sun glint, thin clouds, cloud shadows and adjacency effects (Cairns et al., 2002; Elmahboub et al., 2006; Lambin, 1996; Vanonckelen et al., 2013).

Where correcting for atmospheric effects is mandatory, even a simple atmospheric correction algorithm such as the dark object subtraction can increase the accuracy of land cover classification and change detection results (Moran et al., 1992; Song et al., 2001). A sum of 28% of selected articles used single date imagery to assess ecosystem services; in these cases the application of an atmospheric correction algorithm may not be as essential as if image classification were to be performed for multiple dates (Kawata et al., 1990).

2.4.2 Land cover

Remote sensing for the assessment and valuation of ecosystem services is still highly dependent on land cover data used as proxies for the actual ecosystem service. Land cover has been used as the main source of RS data in the majority of publishing efforts, serving as proxy data to spatial value transfer-based modelling approaches, which have been demonstrated to be highly scale-dependent, producing inaccurate estimates if compared to primary data (Eigenbrod et al., 2010). By assuming an $ESV = Area * Market\ price$ relationship, the valuation exercise will produce ecosystem services values that rely largely on the area measurements extracted from the satellite imagery, therefore ecosystem services valuation efforts in the context of remote sensing will be highly dependent on spatial resolution and image classification accuracy (Cai et al., 2013b).

The option for appropriate measures to address the uncertainty of land cover classification is a prerequisite to any mapping exercise (Olofsson et al., 2013; Rocchini et al., 2013; Sexton et al., 2013). Through evaluation of the selected literature we identified the assessment of thematic accuracy of land cover classification to be another important issue, as only 17% of the studies selected appeared to have performed accuracy assessment on the land cover classification results. A variety of measures have been used to describe the accuracy of land cover classification. For example, to summarize the results of an accuracy assessment, authors have used the overall proportion of area and/or pixels correctly classified, the Kappa coefficient of agreement, and the user's and producer's accuracies. The applicability of land cover data relies on its quality, and in this context it is critical to assure the validity of the classification results and their suitability for any particular purpose. In addition, the error budget included in the classification will most likely propagate through the analyses linking the final land cover map to other datasets used for the ecosystem services valuation at any given spatial scale (Congalton and Green, 1993; De Clercq et al., 2009; Gahegan and Ehlers, 2000).

2.4.3 Vegetation

Vegetation indices derived from satellite sensor images have become a major data source for monitoring ecosystem services linked to vegetation condition. Nevertheless, none of the studies reviewed accounted for the influence of sensor characteristics in the calculation of NDVI values. For example, NDVI values derived from MODIS products were applied in capturing vegetation phenology, although the relatively coarse 500 m spatial resolution and smoothing forced by the 16 day BRDF model inversion reduces their ability to capture fine scale temporal variation in phenology (Hadjimitsis et al., 2010; Jing et al., 2004). Difficulties may, thus, emerge in detecting inter-annual variation in phenology, and estimating phenological parameters such as the onset of greenness (OG), length of season and end of senescence, affecting directly the robustness of relationships involved in estimating and forecasting crop cycles and crop yield (Basnyat et al., 2004; Drolet et al., 2008; Mkhabela et al., 2011; Wall et al., 2008b).

2.4.4 Scale

The spatial scale at which ecosystem services are assessed through remote sensing will greatly influence the resulting ES values, as changes in area within the land cover types will impact the amount of change in total ecosystem service value (Konarska et al., 2002). It is important to understand how these values vary with changes in spatial scale, especially with most ecosystem services valuation studies being supported by multi-sensor land cover data. For example, if using data from multiple sensors (assuming that they have different spatial resolutions), the resulting land cover classes will differ in area (Boumans et al., 2002), and consequently affect ecosystem service valuations. Depending on spatial resolution, data from different sensors will most likely produce different valuation estimates for the same area (Konarska et al., 2002).

Another issue will emerge if the image scene contains subdominant and rare classes. These exist largely at the sub-pixel scale and, thus, are difficult to resolve with thematic coarse resolution satellite sensor imagery (Aguirre-Gutierrez et al., 2012; Serra et al., 2003). Furthermore, due to the coarse spatial resolution (both spatial and spectral) of some satellite sensors, land cover classes that tend to appear in small patches are often omitted from the final classification product, and consequently replaced by other dominant classes surrounding them (e.g., using the maximum likelihood classification method). Depending on the classification method used, the chosen classifier might be overwhelmed by the prevalent class and fail to account for the rare examples. As a result, small and rare classes may be underestimated and the large classes overestimated (Congalton, 1991; Doan and Foody, 2007; Hammond and Verbyla, 1996).

2.4.5 Temporal Extent

Observation of patterns in human activities over time, and the capacities of different ecosystems to provide services under different scenarios of land use change is a key element in most social and ecological studies. In the literature selected 44% of studies have used Earth observation data corresponding to a temporal extent greater than ten years, and within this value 20% of studies concentrate on extending observations to more than twenty years.

Ecosystem services studies would benefit greatly from extensive repeat surveys, as they are extremely useful in detecting and analysing the temporal and spatial dynamics of ES and their likely beneficiaries. For example, if an area is converted from forests to agricultural land this inevitably causes loss of critical habitat, and any ES study that considers only a short period of time will fail to accurately assess the costs and benefits of that land use/cover conversion and the differences in the timing of subsequent impacts. Consequently, research efforts integrating Earth observation data into ES studies will need to assure that the temporal extent of the analysis is consistent with the temporal extent of the projected impacts.

2.5 Conclusions

This review summarises research progress towards the use and integration of remote sensing data within the context of ecosystem services assessment. Individual examples from the literature were reviewed to suggest what has been achieved so far, what can be done and what can still be improved when using remote sensing data to estimate ecosystem services.

Through a systematic review of the literature on remote sensing for ecosystem services assessment it was possible to quantitatively describe both the overall numbers of publications through time, and the relation between the range of remotely sensed biophysical properties predicted and the ecosystem services which they are used to estimate. The results show which ecosystem services are the most measured from space, what kind of remote sensing data has been applied more frequently in estimating these ecosystem services, the temporal resolution of these research efforts, and where in the world more research is still needed.

Several basic technical issues need to be addressed more frequently, as these issues will influence the overall uncertainty budget, as follows: the accuracies of classification results and biophysical data used as proxies, the different spatial, temporal and spectral resolutions of the satellite sensor system (in many cases this restricts the utilization of the remotely sensed data) and, when possible, the need to account for the dominant conditions at the time of image acquisition (e.g., atmospheric interference, viewing geometry, etc.). Future research needs to account for the fact that effective mapping of the full range of ecosystem services is currently beyond the capabilities of remote sensing alone and, therefore, more concerted research is needed to overcome this and other limitations, for example, through fusion of remotely sensed data with information from other sources.

Chapter 3: Extravagance in the commons: resource exploitation and the frontiers of ecosystem service depletion in the Amazon estuary.

3.1 Introduction

Estuaries and river deltas are hotspots of human development and have been under continuous pressure arising from anthropogenic development. For example, large river deltas such as the Ganges-Brahmaputra, the Mekong delta, Yangtze delta, the Niger delta and Nile delta are home to more than 800 million people, and an increasing proportion of this population is converging in urban areas (Anthony et al., 2014; Gorman, 2014; Liersch et al., 2013; Sarwar and Woodroffe, 2013; Tian et al., 2014). The problems arising from a growing population and resource exploitation, coupled with globalisation of markets, pose major challenges to the sustainability of deltas (Cumming et al., 2014; Hoekstra and Wiedmann, 2014; Nicholls and Cazenave, 2010; Ostrom, 2009).

The Amazon estuary has intrinsic similarities and yet is unique in comparison to other large river deltas and estuaries across the globe (da Costa et al., 2013; Parolin et al., 2004). It is one of Brazil's last frontiers of development supported by the Brazilian government, and is seen as a land of "vast" opportunity to those seeking new enterprises (Fortini and Carter, 2014; Metzger, 2002; Steward, 2013). A large segment of the population in the estuary region (≅50000 people), depends on local extraction of natural resources for livelihoods (Furtado, 2003; Sorj et al., 2007). Areas sparsely inhabited have been utilised with few negative consequences for the environment so far (Brondizio et al., 1994a; IBGE, 2012; Moran et al., 2000). However, intensive exploitation of natural resources now shows signs of an advanced state, triggered by long lasting negative interactions between the population and the environment (Ishimaru et al., 2014; Peres et al., 2003; Potter et al., 2001).

The increasing pressure on ecosystem services in the Amazon estuary, is amplified by the large influxes of immigrants from other parts of the country (IBGE, 1981, 1982). Immigration to the estuary has been encouraged since 1960 (on an initiative from Brazil's military regime at the time) to occupy Brazil's vast areas of rainforests (Loureiro and Pinto, 2005; Théry, 2005), thus, establishing Amazon's pioneer fronts on rubber extraction, mining and agriculture (Oliveira and Felix, 1980). Since then, the Brazilian government has acted systematically in offering subsidies to anyone seeking to progress and expand agricultural systems, industrial and urban development (Amaral et al., 2006; Gusso et al., 2014; Soares et al., 2006).

The Amazon estuary is naturally linked to the Amazon basin, as its shares the feedbacks from a multitude of natural processes taking place at the basin scale (Christoffersen et al., 2014; Tian et al., 1998). Here the major mechanisms of economic development have modified the landscape continuously, starting during the 1950s and leading to what we now see as widespread environmental degradation (Soares et al., 2006). Moreover, in the context of social and environmental change, economic and strategic interests make the estuary a distinguished case in the Amazon basin. Nevertheless, the Amazon estuary was developed relatively late, compared to other major world estuaries and deltas. As late as the 1970s the main cities in the estuary started growing in population, with major private and government investments flowing consistently into the area (Sorj et al., 2007). Markedly, after the implementation of the Plan for National Integration, where the main objective was to develop the country's far-flung territories by creating new opportunities for economic development (Albert, 1992; Kohlhepp, 2001; Zhouri, 2010).

Recent studies argue that deforestation in Amazonia decreased by 77% since 2004 and stabilised after 2009 (Godar et al., 2014a), as a consequence of forest policy interventions, private sector initiative and market conditions (Nepstad et al., 2014; Tollefson, 2008). In terms of historic and recent patterns of deforestation in the Amazon basin, the observed decline and stabilisation reflect adjustments in land use practices and policies as a consequence of the adverse impacts of deforestation (Lambin and Meyfroidt, 2011; Rudel et al., 2002). Long lasting deforestation in the region may have passed a threshold and is now moving towards recovery, which may be an indicative of forest transition (Lambin and Meyfroidt, 2010; Rudel, 2002; Rudel et al., 2005). However, this might not be exactly the case in the Amazon estuary, as some of the main factors influencing deforestation are still consistently challenging Brazil's efforts in preserving its tropical forests (van Vliet et al., 2013b).

Several major global economic crises have affected international agricultural commodity prices, technology and government subsidies and policies, and yet these are still driving land clearing as if there were no negative feedbacks emerging from it (Bustamante et al., 2014; Carmenta et al., 2013; Garrett et al., 2013a; Southworth et al., 2011). Continued natural and anthropogenic perturbations in the Amazon basin will cause degradation of key forest ecosystem services, such as carbon storage in biomass and soils, the availability of forest products, and the modulation of local and regional climate patterns (Boerner et al., 2007; Foley et al., 2007; Klemick, 2011). This generates a need for greater understanding of the driving forces and consequences of land use change and deforestation, and how it might be stopped and/or reversed (Laurance, 2015; Wiebelt, 1995; Wright, 2010).

This research provides an opportunity to advance our knowledge of the dynamic nature, and links between, aspects of politics, economy and society that have affected the availability of forest ecosystem services in the Amazon estuary over the last three decades. The objectives of this paper are:

1. What are the current status and recent trends over the last 32 years in the green vegetation cover in the estuary and what are the consequences to ecosystem services?
2. To what extent is development associated with such changes and, if so, what are the likely main drivers of frequent and unanticipated changes on forest cover in the Amazon estuary;
3. What are specific future threats to Amazon forest ecosystems arising from the complex interactions needed to sustain the on-going development of social-ecological systems in the region?

3.2 Methods

3.2.1 Study Area

The study site has an area of 294,000 km^2 and is located between the states of Para (PA) and Amapa (AP) Northern Brazil (Figure 3.1). The estuary is part of in the Amazon basin, and it is characterized by extensive areas covered by old growth forests, mature floodplain forests, grasslands and agricultural mosaics. The Amazon estuary extends from the mouths of the Amazon and Para rivers, merging at the eastern side of the Marajo Island (Brondizio et al., 1996). The area that delimits the Amazon estuary for the purposes of this study was based on (Ericson et al., 2006b), where a set of Global 30 Arc-Second Elevation grids (1 km x 1 km), soil maps, aerial photographs, satellite sensor imagery and river bifurcations data were used to create the dataset on the delta boundaries.

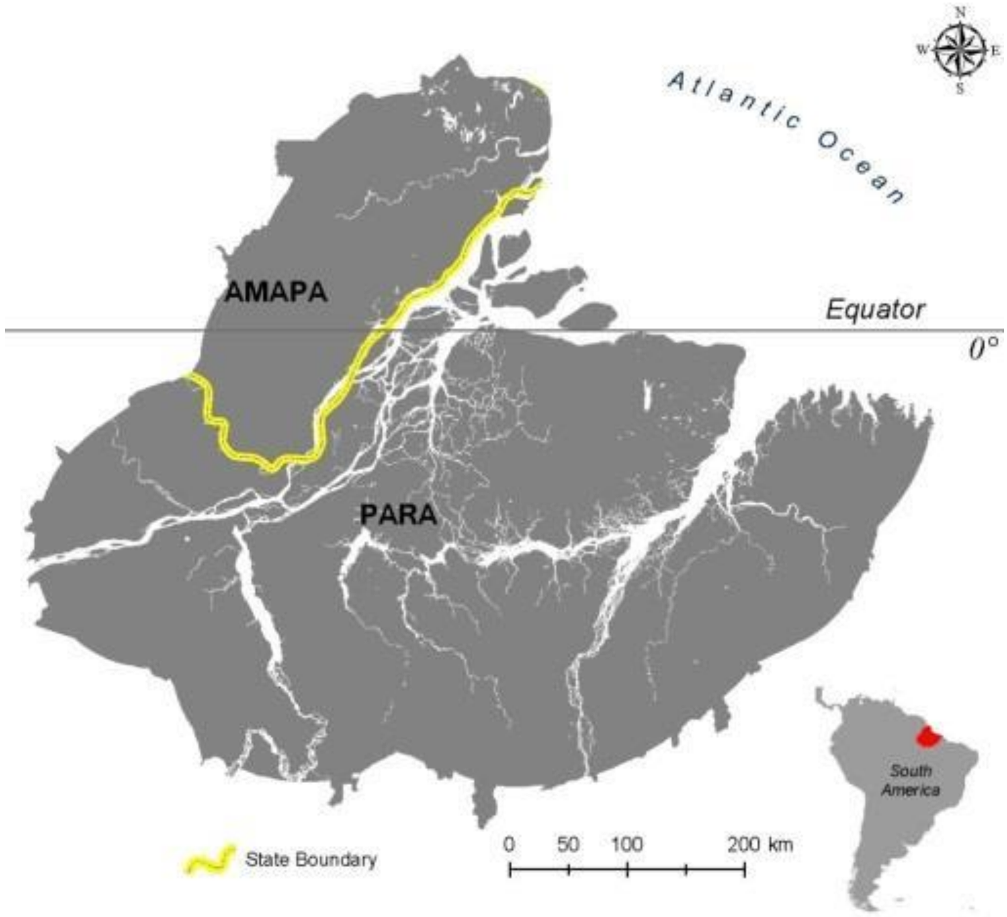


Figure 3.1– Amazon estuary in the north of the South American continent. The dashed yellow line represents the division or frontier between the states of Para and Amapa (de Araujo Barbosa et al., 2016b).

The managers of forests in the estuary have the difficult task of safeguarding the balance that allows ecological systems to thrive and, therefore, provide sustainable benefits to social systems at local, regional and global levels (Anderson et al., 1995; Fortini and Zarin, 2011; Hiraoka, 1995). The political boundaries that subdivide the study area create 17 regions, and 81 municipalities, all of which benefit directly from the resources and ecosystem services produced by forests. We applied a 40 km buffer around the defined region to include municipalities and regions directly and indirectly affected by social and ecological changes.

3.2.2 Data acquisition and Pre-processing

To characterise the changes in forest cover (FC) during the last three decades in the Amazon estuary we used datasets from the third generation of Global Inventory Modelling and Mapping Studies (GIMMS) (generated from Advanced Very High Resolution Radiometer sensors on board the NOAA 7, 9, 11, 14, 16, 17 and 18 satellites) covering a period of 33 years (Anyamba et al., 2014). The spatial resolution provided by the GIMMS data is 8 km, using the Albers Equal Area Conic projection on the Clarke 1866 ellipsoid (Zhu et al., 2013).

The GIMMS NDVI3g dataset provides monthly image composites over a 15-day period, resulting in two maximum-value NDVI composites per month. The first 15-day composite is the maximum value from the first 15 days, and the second corresponds to days 16 through to the end of the month (Zeng et al., 2013). Only positive NDVI values were used to eliminate (Norton et al.) pixels corresponding to water bodies.

Each set of two monthly sequential NDVI measurements produced by GIMMS were averaged to produce monthly NDVI images, which were later used to calculate the green vegetation cover. In this research, the time-series of GIMMS NDVI3g was built using measurements from 1982 to 2013 (32 years of GIMMS data). We chose to work with only full years covered by the data (the first year of GIMMS NDVI data covers only the time period from July to December 1981).

3.2.3 Ecosystem services data

To produce estimates of climate regulation, we used records from six meteorological stations in the estuary, made available by The Brazilian Meteorological Institute (INMET). This dataset is under constant collection, maintenance and storage by the Brazilian government, and it covers the period between 1960 to present. For the present purpose we matched this time scale to the temporal scale imposed by the satellite sensor data. Therefore, we used yearly averages for rainfall, relative air humidity, temperature, and number of days with precipitation. Additionally, annual ecosystem-atmosphere exchanges of CO_2, CH_4 and N_2O for tropical evergreen forests were utilized to account for greenhouse-gas emissions from natural forests due to natural processes (i.e., tree growth, ageing and mortality)(Lima et al., 2014; Tyukavina et al., 2015). Furthermore, data were obtained on the strongest El Nino events in the period 1982 - 2013. The climatological records were aggregated to compose a standardised climate regulation index.

Carbon sequestration estimates were produced using data from the published literature on average carbon sequestration per ha covered by forests (Mg C ha^{-1} yr^{-1}) (Keith et al., 2009; Luyssaert et al., 2007). To account for rapid changes in carbon storage we incorporated into our estimates data on average precipitation patterns and temperatures, as these can cause marked and relatively rapid shifts in carbon sequestration (Chambers and Silver, 2004). For estimates of timber availability we used timber availability per unit area estimates published by ecosystem services studies (Baraloto et al., 2014; Fearnside, 2008; Kirby and Potvin, 2007; Vilanova et al., 2012) along with statistics of industrial hardwood plantation.

3.2.4 Governance, forest exports and economic scenario

We used three indicators to assess whether Brazil has an environment that would favour governance, in the sense of being able to establish and monitor policy implementation, and control deforestation (Apaza, 2009). We combined a time series of the indicators of Control of Corruption, Rule of law and Regulatory Quality produced by the World Bank (Kaufmann et al., 2011; Merry et al., 2015). This dataset reports statistics compiled from the responses of a large number of enterprise, citizen and expert survey respondents in developing countries (Thomas, 2010).

To quantify and show the periods in time when a favourable scenario may have been in place for the export of forest products, we used historical currency exchange rates for both the USD and the Brazilian Real.

To show the effects of changes in governance and economy on the exports of forest products we used a time-series with yearly estimates on the export of forest products (raw wood, paper, rubber) measured in units of metric tonnes, from 1997 to 2013. The data were derived from official Brazilian agricultural statistics data. This is an up to date and rigorously collected set of data kept and maintained by the Brazilian government.

3.2.5 Extracting the green vegetation cover

The green vegetation cover derived from medium resolution satellite imagery has been used in many studies to estimate forest properties such as extent and biophysical properties (Boyd et al., 2006; Boyd et al., 2002; Ding et al., 2015; Foody and Curran, 1994; Mao et al., 2014). Here we draw upon methodological approaches used by various authors in estimating FC through measurements of fractional vegetation cover from medium spatial resolution satellite data (Foody et al., 1997; Guan et al., 2012; Ichii et al., 2003; Zeng et al., 2000). In this study Green Vegetation Cover (GVC) derived from GIMMS will be defined as the green vegetated area which is directly detectable by the sensors from any view direction (Purevdorj et al., 1998).

The mathematical approximation of the green vegetation cover is based on Deardorff (1978) who considered an area-averaged shielding factor to be feasible in accounting for land-surface heterogeneity, when using a surface-vegetation-atmosphere transfer model. The degree of which the foliage prevents shortwave radiation from reaching the ground can be considered as a shielding factor, and it will vary between complete shielding from the ground by vegetation and non-existent shielding.

Using equation 1 we are able to quantify the contributions of both vegetation cover and soil to the pixel value. A satellite measurement (λ) for each pixel denoting the totality of the contribution from vegetated (f_c), and non vegetated areas ($1 - f_c$), as a result the satellite data is a product of spectral linear mixing, as follows (λ_{veg} and λ_{soil} refer to the satellite observed values for vegetation and non-vegetation pixels) (Tateishi et al., 2004):

$$\lambda = \lambda_{veg}\, f_c + (1 - f_c)\, \lambda_{soil} \tag{1.3}$$

The values of NDVI are expected to vary throughout the year as a result of plant phenological cycles. To produce season-independent GVC estimates one needs to eliminate the potential for seasonal bias in the time-series. Consequently, the season-independent vegetation fractional cover f_c for each pixel needs to be estimated by applying the annual maximum NDVI value ($NDVI_{p,max}$) to equation 2.3 (Zeng et al., 2000):

$$f = {NDVI - NDVI_{soil}}\Big/{NDVI_{veg} - NDVI_{soil}} \quad , \tag{2.3}$$

taking the final form:

$$f_c = {NDVI_{p,max} - NDVI_{soil}}\Big/{NDVI_{c,v} - NDVI_{soil}} \tag{3.3}$$

Where, f refers to the estimate of GVC in each image pixel that is dependent on and varies according to seasonal factors affecting vegetation. The subscripts veg and $soil$ denote values over fully vegetated areas, and bare soil, respectively, $NDVI_{c,v}$ is the NDVI value for each land cover category that corresponds to 100% GVC and it is based on vegetation cover maps produced by The Brazilian National Institute For Space Research (INPE). The use of $NDVI_{p,max}$ implies that f_c represents the annual maximum green vegetation cover for a given pixel.

3.2.6 NDVI unmixing

A major problem associated with medium spatial resolution satellite sensor datasets is that the spatial resolution of the images makes it difficult to detect fine-scale variation (Boyd et al., 2003; Eckert et al., 2015). Since each pixel represents a large area on the ground, the image pixels are likely to contain more than one land cover class (Di Bella et al., 2009; Kumar et al., 2008; Quintano et al., 2012). Thus, pixel-wise classification approaches result in classified images where adjacent end-members jointly occupy the same pixel.

A spectral NDVI unmixing procedure is needed to account for variation in the spectral signal denoting targets of vegetation and non-vegetation, at the spatial and spectral resolution provided by the GIMMS imagery. The applied linear spectral unmixing procedure was used to decompose the mixed pixels into a collection of constituent spectra (endmembers) and a set of corresponding fractions, indicating the proportion of each endmember present in the pixel.

Since a quantity of land cover on the Earth’s surface is measured by the satellite sensor, and represented in the form of a pixel, each pixel represents the sum of the contributions of pure-element reflectance (endmembers), where the weights are the percentage of the pixel area occupied by each element. The NDVI values corresponding to the proportions of u_{soil} (soil) and u_{veg} (vegetation) on a pixel can be defined using the following:

$$NDVI_{soil} = \sum_{i=1}^{i=u_{soil}} \left(pi \ * NDVI_{soil\_i}\right)$$

(4.3)

$$NDVI_{veg} = \sum_{i=1}^{i=u_{veg}} \left(pi \ * NDVI_{veg\_i}\right)$$

(5.3)

Where the term pi refers to the proportions of $NDVI_{soil\_i}$ and $NDVI_{veg\_i}$ in the pixel, in equations 7 and 8, and $NDVI_{soil}$ and $NDVI_{veg}$ denote the values of soil and vegetation types in a pure pixel. Equations 4.3 and 5.3 can therefore be written as:

$$f = \frac{NDVI \ - \ \sum_{i=1}^{i=u_{soil}}\left(pj \ * NDVI_{soil\_i}\right)}{\sum_{i=1}^{i=u_{veg}}\left(pi \ * NDVI_{veg\_i}\right); \ - \ \sum_{i=1}^{i=u_{soil}}\left(pj \ * NDVI_{soil\_i}\right)}$$

(6.3)

3.2.7 Accuracy of the GVC estimates

The accuracy of the GVC estimation was evaluated based on the standard errors of the estimates. The relative error (σ) is then calculated using equation 7 (Zhang et al., 2006):

$$\sigma = \sqrt{\frac{\sum_{i=1}^{n}\{(A_i - \tilde{A}_i)/A_i\}^2}{n}}$$

(7.3)

where, A refers to the observed GVC value for a pixel, $\tilde{A}$ refers to the estimated value, and n corresponds to the number of observations. The relative error shows the accuracy of the estimate of GVC inside the specific range or the whole range of the values under analysis.

3.2.8 Time-series of GVC

The estimates of GVC produced by applying the algorithm described in section 3.2.5 were used to establish annual mean values corresponding to each municipality in the estuary. By averaging the twice per month GVC quantities a monthly time-series was created with clear information on inter-annual variability in the GVC on a pixel-by-pixel basis. Additionally, the averaged values were used in the transform from GVC into ecosystem service value flows for each pixel in the satellite sensor image over the 32-year period. We then used the GVC values combined with the ecosystem services values from the literature to produce estimates of important ecosystem services flows per hectare provided by forests in the estuary.

3.2.9 Quality of GIMMS data

This dataset was produced with better quality, using improved calibration procedures unlike its previous versions. This dataset is considered to be the most accurate, long-term AVHRR data record to date. A number of improvements have been made on the current version of GIMMS NDVI data, including corrections for: sensor degradation; inter-sensor differences; solar-illumination angle and sensor-view angle effects due to satellite drift (Pareeth et al., 2016; Pinzon et al., 2005).

3.2.10 Multivariate time series Analysis

Here we focus on defining the underlying relationships affecting FC in the Amazon estuary over time. Our task was to identify the dynamics and structure of multiple time series of economic data, and use this to verify the impacts that each of the economic variables had on vegetation cover.

We needed to investigate the correlations and dynamic structure of this datasets, modelling these relationships with an approach capable of quantifying the causal influence from one of the economically and market related variables into vegetation cover. Therefore, we used Granger Causality test to address the question of how changes in forest exports, governance, beef exports and currency exchange rate, would help explain the variations observed in GVC under different time lags (in years). Given a lag lg

we estimate the bivariate unrestricted Vector Autoregressive Equation for two variables (*X_t* and *Y_t)* as follows:

$$X_t = \beta_0 + \sum_{i=1}^{lg} \beta_i X_{t-i} + \sum_{i=1}^{lg} \alpha_i Y_{t-i} + \varepsilon_{1t}\,, \tag{8.3}$$

$$Y_t = \alpha_0 + \sum_{i=1}^{lg} \gamma_i X_{t-i} + \sum_{i=1}^{lg} \delta_i Y_{t-i} + \varepsilon_{2t}\,. \tag{9.3}$$

Where β, α, γ and δ are coefficients and ε_t is a residual term. The null hypothesis for the Granger causality ('Y does not Granger-cause X' and/or 'X does not Granger-cause Y') can be specified as follows:

$$H\_0\colon \alpha\_1 = \alpha\_2 = \ldots = \alpha\_m = 0\,, \qquad H\_0\colon \delta\_1 = \delta\_2 = \ldots = \delta\_m = 0 \tag{10.3}$$

The tested hypothesis can be implemented by an *F* test that can be implemented using a model composed by two regression steps:

$$X_{t\_u} = \beta_0 + \sum_{i=1}^{lg} \beta_i X_{t-i} + \sum_{i=1}^{lg} \alpha_i Y_{t-i} + ø_{1t} \tag{11.3}$$

$$X_{t\_r} = \beta_0 + \sum_{i=1}^{lg} \beta_i X_{t-i} + \mu_{1t} \tag{12.3}$$

Later computing the residuals sums of squares, using the equations:

$$RSS_u = \sum_{T=1}^{T} ø_{1t}^2 \tag{13.3}$$

$$RSS_r = \sum_{T=1}^{T} \mu_{1t}^2$$

(14.3)

Finally, to compare the sum of squared residuals with an *F* distribution and $(p, T - 2p - 1)$ degrees of freedom we use the following equation:

$$F(p, T - 2p - 1) \sim \frac{(RSS_r - RSS_u)/p}{RSS_u/(T - 2p - 1)}$$

(15.3)

If found that the value for the *F* statistic exceeds the critical value for a chosen level of significance, we inevitably reject the null hypothesis (significance level was set at $p \leq 0.05$).

3.3 Results

3.3.1 GVC losses at multiple scales

The changes in GVC over the last 32 years in 78 municipalities and 17 regions inside Para and Amapa states are presented on Table A1 in Appendices. The GVC on Table A1 through time is presented using a colour scheme set to represent low yearly mean GVC values (warm colours close to red) and high yearly mean GVC values (shades of green).

The way these GVC results are shown follows Brazil's administrative denominations (federative state, region, municipality). The layout enables the identification of local and regional patterns, reflected in the percentage of fractional vegetation cover from municipality to state level. The state of Para has the greatest number of municipalities with reduced GVC, and these municipalities represent in some cases entire regions, which shows that changes in GVC are fairly consistent at the regional scale.

These results reveal that the majority of municipalities, in both federative states (although more intensively in Para) show significantly less GVC per pixel throughout the period. Municipalities and regions have been continuously losing forest due to a combination of factors such as agricultural expansion, urbanization, selective logging, forest fires, and inundation of large areas for river dam construction (Cunha and Ferreira, 2012; Manyari and de Carvalho, 2007; Morton et al., 2008; Verburg et al., 2014).

The regional patterns observed in the GVC (Figure 3.2) illustrate how the regions are changing more or less rapidly over the period. Note that even the smallest variation (increase/decrease) in yearly mean GVC value represents huge losses of FC on the ground. From the series of line plots, we notice that the small peaks in average GVC values occur throughout the entire period, and these small peaks are particularly evident over the 1990s and early 2000s. Some regions in both states of Amapa (Macapa and Amapa) and Para (Belem, Salgado, Castanhal, Guama, Tome-Acu and Furos de Breves) have slowed down the decreasing trajectory in GVC, while the remaining regions reveal an opposite trajectory.

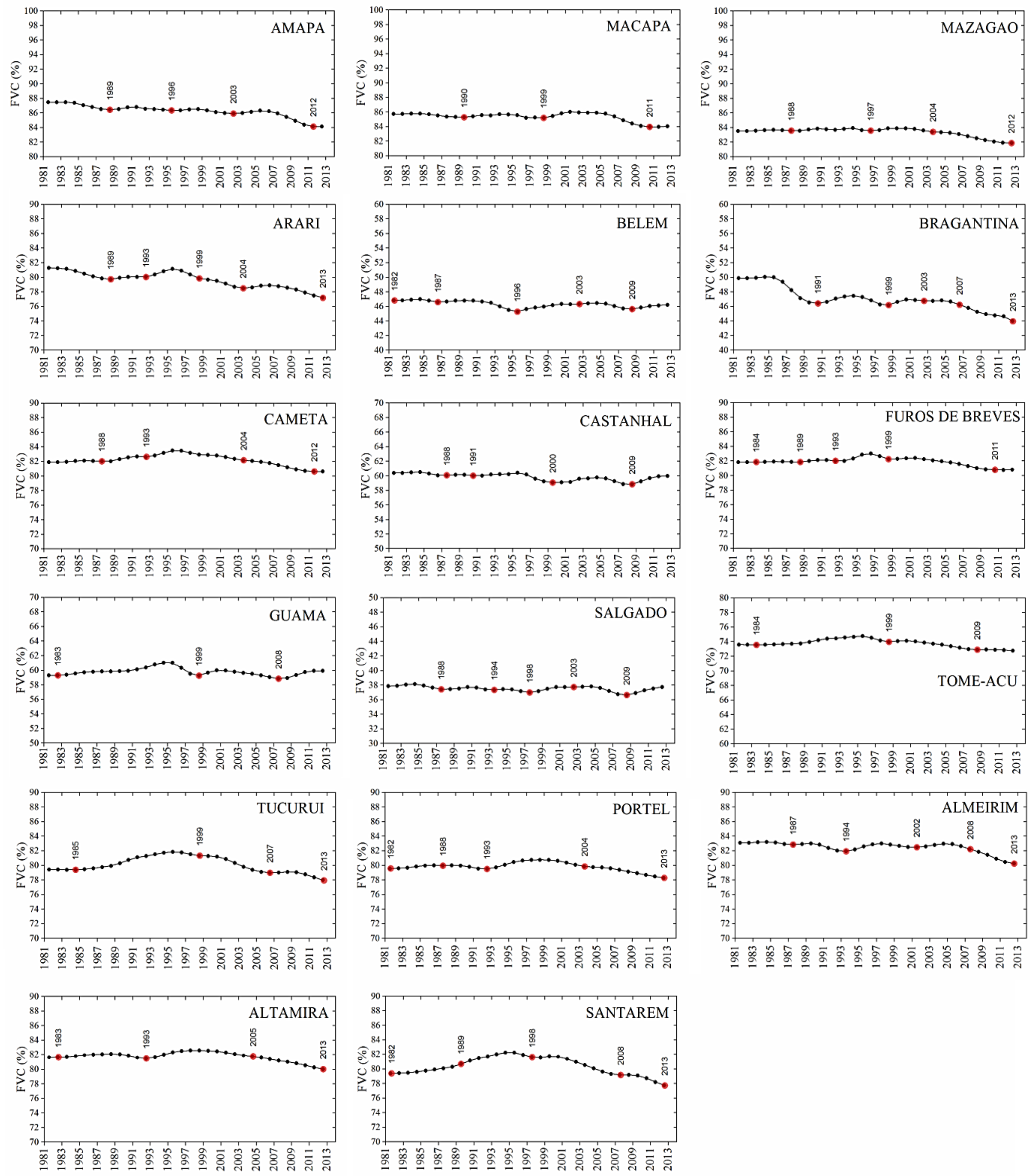


Figure 3.2– Regional means of GVC for Amapa and Para state. The red dots superimposed along the line on the times-series represent years with low peaks in GVC values (de Araujo Barbosa et al., 2016b).

The time-series of yearly mean GVC values is also shown as a sequence of maps (Figure 3.3), in order to visualise the spatiotemporal range of yearly mean GVC changes. Here, we should be able to see that the clearing of forests is a long-standing process, with a starting date that precedes the availability of satellite sensor data. Therefore, by the time these images started being available vast deforested areas were already in place. This is clearly visible in areas at the Northeast and South of the estuary. The sequence of satellite-derived maps indicate that the pace and extent of deforestation has taken place more rapidly during the late 1980s. Furthermore, the areas where deforestation has occurred more rapidly and monotonically are located in the state of Para. Thus, the evidence in the previous figures and tables (Figure 3.3 and Table A1 in Appendices) is here verified and spatially-explicit.

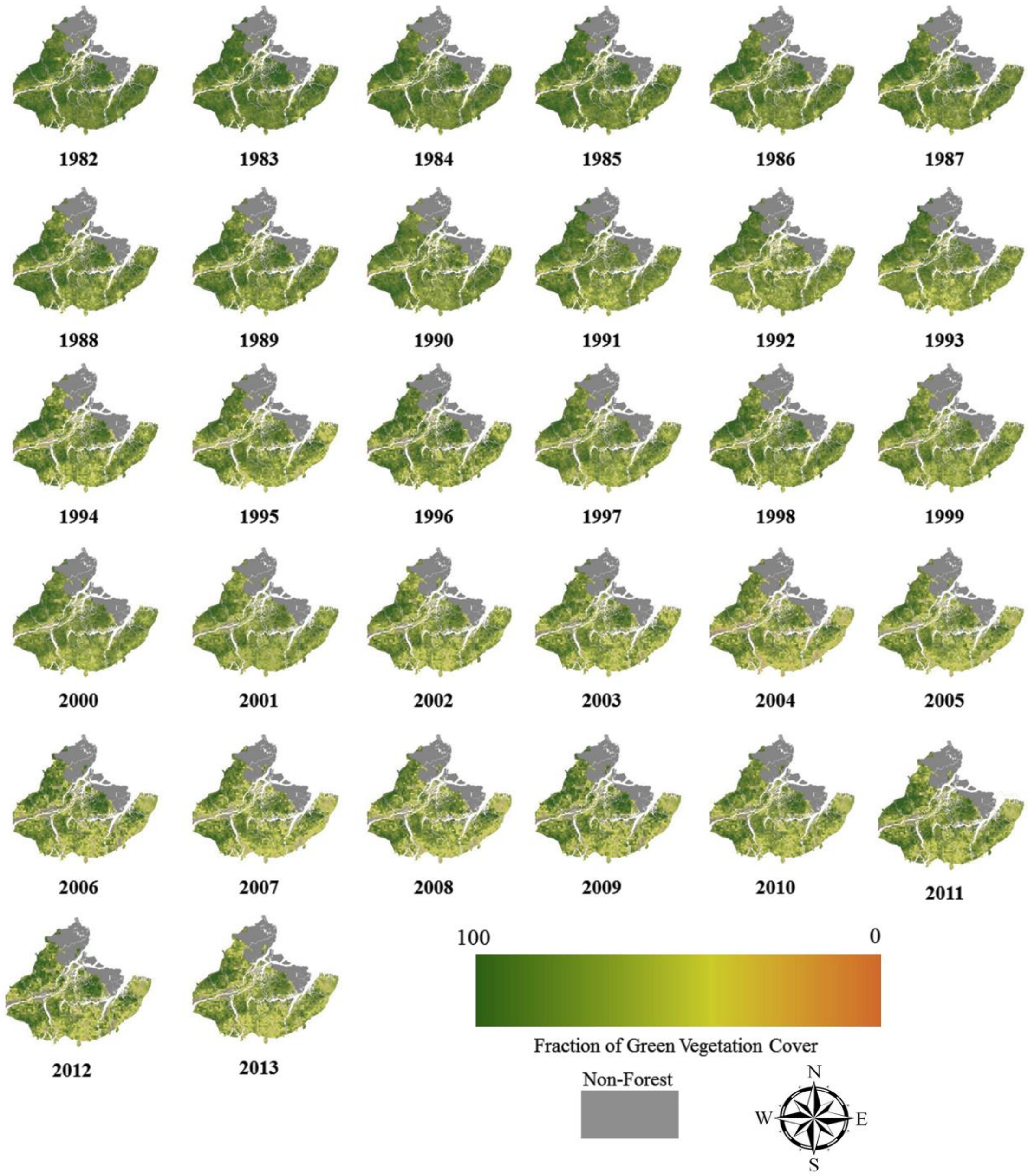


Figure 3.3 – Yearly mean GVC for the entire estuary from 1982 to 2013 (de Araujo Barbosa et al., 2016b).

The slow down on deforestation rates in the Amazon Basin means that the pressure for clearing of new forest areas has been reduced, although these maps reveal the consequence of years of intensive forest clearing. At the estuary scale, pressure for clearing of new forest land seem to have decreased but not stopped.

3.3.2 The relationships inside the multivariate system

Figure 3.4 shows a time-series of annual ecosystem service estimates for the Amazon estuary as a series of normalized curves for carbon sequestration, climate regulation and availability of timber. The plot also shows the time-series of GVC values, and the country's official statistics data on beef exports, exports of forest products (raw wood, rubber, cellulose and paper), currency exchange rates (USD vs Brazilian Real) and governance. The numbered dashed lines spread along the *X*-axis represent changes in environmental law in the country during the period. Comparing the trajectories in the plot, it can be seen that major changes have occurred in ecosystem services during this 32 year period.

Overall, the GVC has decreased since 1986, later moving upwards and decreasing again in the early 2000s. The curves for carbon sequestration and timber availability remained stable during the early 1980s, pointing towards recovery in the late 1980s, with a later significant reduction during the early 2000s. Currency exchange rates show a dramatic change between 2000 and 2003; soon after this the value of the USD begins a steady fall, and starts recovering again after 2011. With a time delay of four years, the time-series of exports of forest products reflects a similar pattern. Beef exports in the region show a steady increasing trajectory, with no significant decreases up to 2013. Finally, governance shows an inverse behaviour (decrease followed by recovery) compared to exports and currency exchange rates during the period between 1997 and 2013.

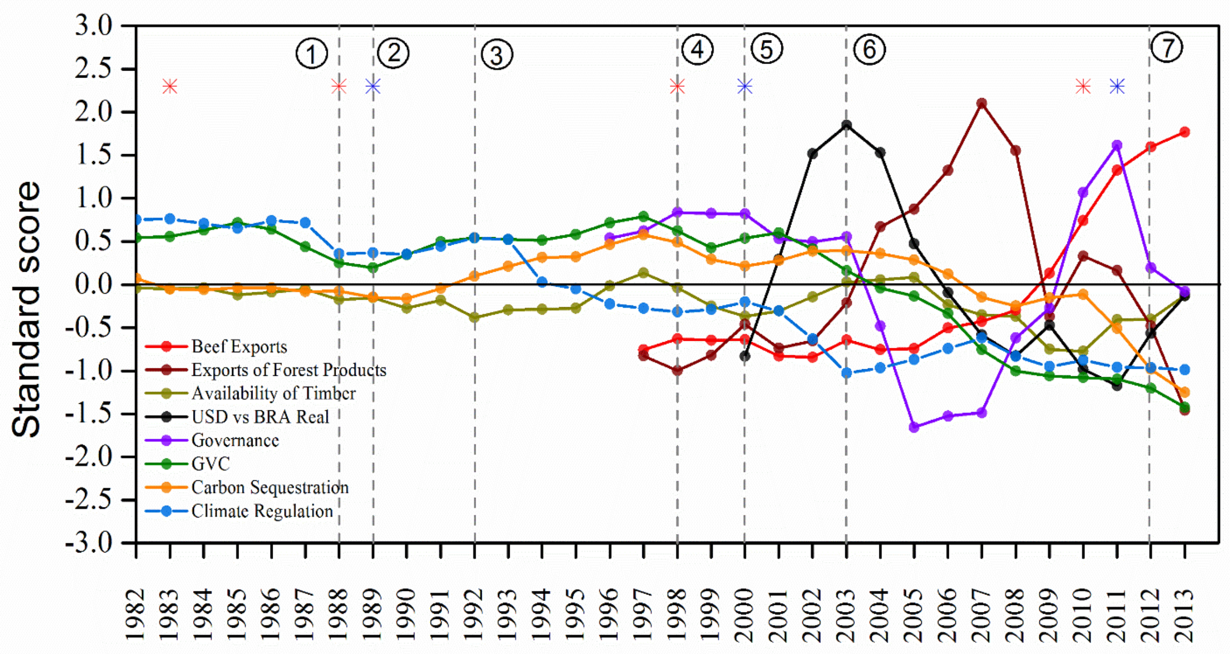


Figure 3.4 – Evolution of the availability of the forest ecosystem services of Climate regulation, Carbon Sequestration and Timber availability per hectare of forest (under sustainable management), against the observed values for USD currency *vs* Brazilian currency, Governance index, total exports of beef and forest products (raw wood, paper, rubber) reported for Para and Amapa state together. The symbol * represents years with strong ENSO anomaly occurrence (El Niño* and La Niña*). The ecosystem service line trajectories indicate whether the services are decreasing or increasing. The dashed lines and numbers refer to specific points in time where relevant forest regulatory laws were put in practice in Brazil: **1)** New Brazilian Constitution, that replaces the charter drawn up two decades ago by generals. Now states and municipalities can develop their own environmental laws; **2)** Brazilian Federal Law 7.735/1989 establishing the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA); **3)** Establishing of The Brazilian Ministry of the Environment (MMA) responsible for formulating environmental law in Brazil; **4)** Federal Law Nº 9.605/1998 enacted in Brazil to establish criminal penalties for individuals and corporations that commit environmental crimes, and it is currently the main legal instrument regarding environmental criminal and administrative liabilities; **5)** Law Nº 9.985/2000 which establishes the National Protected Areas System, and classifies the conservation units in: Areas of Integral Protection and Areas for Sustainable; **6)** Decree that establishes The National Biodiversity Commission (CONABIO) and The National Commission for Forests (CONAFLOR); **7)** Federal Law Nº 12.651/2012, the Brazilian New Forestry Code that revokes the Brazilian Forest Code of 1965 (the Federal Law Nº 4.771/1965) (de Araujo Barbosa et al., 2016b).

From the graph in Figure 3.4 one can see that, on paper, Brazil have a reasonable number of decrees, regulations and resolutions concerned with the protection of its forests and natural assets, working within different levels of policy jurisdiction. Nevertheless, In reality, integrated analysis reveal that proximate causes of forest clearing in Brazil are related to more complex interactions and emergent pressing forces arising in the context of its inevitable integration into the global economy. Consequently, the associated ecosystem services derived from forests are in constant struggle with the rapid transformations, and may be inevitably getting to a critical point.

In order to explore the apparent relationships in Figure 3.4, we decided to take a quantitative approach using multivariate time series statistical methods. The pairwise Granger causality tests (Table 3.1.) show the possible relationships across the set of variables being analysed. Here we can determine whether variables such as governance, exchange rate beef Exports and exports of forest products presented in Figure 3.4 can help explain the behaviour of GVC over time (the significance level was set at $p \leq 0.05$).

Table 3.1 – Pairwise Granger causality tests for the relationships between GVC, Governance, Exchange Rate (USD vs BRL) Beef Exports and Exports of Forest Products.

Pairwise Granger Causality

Variable Pairs*	Time Lag (years)	Type of Causality	Data Points (years)	Probability (≤ 0.05)
Exports of Forest Prod ↔ Governance	3	Bidirectional	14	0.013
Governance ↔ Exports of Forest Prod	5	Bidirectional	12	0.046
GVC ↔ Governance	2	Bidirectional	15	0.024
Governance ↔ GVC	5	Bidirectional	12	0.030
Currency exchange rate → GVC	4	Unidirectional	13	0.032
Beef Exports → GVC	2	Unidirectional	15	0.005
Exports of Forest Prod → GVC	2	Unidirectional	15	0.041

* The symbols (→ and ↔) represent the direction of the Granger causality.

Here we see that different variables respond more or less rapidly to inputs. Some of the variables that start exerting influence at shorter time lags are also influenced at longer time. The GVC seems to respond faster to changes in Exports of Forest Products and Beef Exports, and it tends to require more time steps before responding to inputs coming from variables such as Currency Exchange Rate, Governance. Interestingly, but not surprisingly, changes in GVC values do help explain Governance after a short time lag.

3.4 Discussion

3.4.1 GVC losses at the regional level

The broad scale patterns of GVC observed in the 17 regions confirm the general trend in the fraction of area covered by forests, that is consistent with findings from other studies (at the Amazon basin scale) (DeFries et al., 2013;). The regional patterns observed in GVC values reflect old and ongoing patterns of land use decisions and deforestation processes; recurrence of fires, increasing land allocation of pasture for cattle ranching, and failed resource management (Morton et al., 2008; Garrett et al., 2013b).

From 1982 to 1995 these seven regions: Amapa, Macapa, Mazagao, Almeirim, Cameta, Furos de Breves and Altamira showed the highest average GVC values per pixel (85% to 81%). The first three of these regions are located in the state of Amapa (in fact, these are all three regions under analysis for this state). In this case, the remaining four regions with the highest GVC values are from the state of Para, therefore representing ≅28% of all regions in our analysis for the state of Para.

The regions of Santarem, Tucurui, Portel, Arari, Tome-Acu, Castanhal and Guama in the state of Para have mean GVC values ranging from 80% to 59% during the period between 1982 – 2013. These regions represent a share of ≅50% of all regions analysed for the state of Para and 31% of a total of 22 regions in Para. The lowest average per pixel GVC values found in the estuary were for the regions of: Bragantina (≅47%), Belem (≅46%) and Salgado (≅37%) in the state of Para.

3.4.2 Last frontier of resource exploitation and loss of ecosystem services

During the last three decades the Amazon estuary has been transformed as a frontier of development, open to an unsustainable model of economic growth. The policies that underpin such changes are based on a well-established belief that multiple economic benefits will arise from the clearing of land, consequently generating economic growth in the Amazon, and improving human wellbeing as more land is cleared to provide space for agriculture or other exploitative activities (de Carvalho et al., 2015; Pacheco and Benatti, 2015). In reality, these benefits have been quickly reversed as cleared land is inevitably exhausted by intensive agricultural land use and expansion of industrial-scale agricultural systems.

The states of Amapa and Para (and its municipalities) have made large commitments for the creation of protected areas and parks, resulting from a growing awareness of the negative effects arising from conversion of natural ecosystems (Muller and Cloiseau, 2015; Nkonya and Anderson, 2015; Schneider et al., 2015). The connection between deforestation and anthropogenic climate change, and the view that the protection of forests will improve states' and municipalities' capa city to cope with future stresses, for example, due to climate change, has produced positive impacts resulting in increased attention to the state and management of forests at regional and local levels. However, for this to be materialised and translated into practice, there is still much more that needs to be done (Torres et al., 2012).

The results show how key forest ecosystem services have been affected by changes in environmental law and economy. It is possible to conclude that the availability of ecosystem services provided by forests in the estuary has been considerably affected, with major decreases in carbon sequestration followed by decreases in the immediate availability of timber since the 1980s. Carbon sequestration is an important ecosystem service provided by forests, and it is key for maintaining other ecosystem services used by individuals locally and globally (e.g. timber production, air quality regulation, primary production, biodiversity and nutrient cycling) (Archibald et al., 2009; Merbold et al., 2009; Scholes et al., 2009; Stursova and Sinsabaugh, 2008). Reduction of carbon sequestration has furthered the negative impacts of changes in climatic patterns through coupled ecosystem-atmosphere feedbacks (Obeng and Aguilar, 2015; Rice et al., 2004). Depletion of terrestrial carbon stores has potential effects on ecosystem functions and their ability to provide non-carbon ecosystem goods (Sommer et al., 2000).

3.4.3 Critical drivers of change

The Granger causality tests shows that changes in GVC can be explained by changes in governance, exports of beef and forest products, and currency exchange rates (USD/BRL).

The pairwise relationships suggest that the export of forest products help explain changes in governance after a 3-year time lag (p=0.013). The relationship between exports of forest products and governance seems to be bidirectional, meaning that, after a 5 year time lag, governance can also help explain changes the exports of forest products (p=0.046). This might provide some insight into how Brazil has been managing and regulating the exploitation of forest resources in the estuary.

There is a strong relationship linking changes in GVC to the state of governance at p= 0.024 (GVC → Governance) after a 2-year period. The relationship between governance and GVC exist in both directions, and there is also a strong relationship between the variables now in the direction of governance → GVC after a 5-year time lag with p= 0.030. Once GVC starts to decrease it may lead to changes in governance, these alterations may take place but will only start feeding back into GVC after some period of time (at least half a decade).

The pairwise Granger causality tests for currency exchange rates and GVC (Currency exchange rate → GVC) indicate a unidirectional relationship (with p=0.032), after a 4-year time lag. This provides some insight into the influence of market conditions into deforestation, which adds to the bulk of drivers of forest cover change.

The pairwise Granger causality test suggests that there is a unidirectional relationship between beef exports and GVC (p=0.005) at a time lag of 2 years. With the world economy still recovering from a period of crisis, demand for beef has increased (followed by its global market price), and this is a commodity that is in many cases produced on deforested land (Gollnow and Lakes, 2014; Hoelle, 2014; Merry et al., 2004). Beef exports have started its trajectory upwards at the time of critical reductions in GVC. Nevertheless, once deforestation has occurred, beef exports may fluctuate without sensitive effects on GVC.

The export of forest products can help explain the time-series trajectory observed for GVC (Exports of Forest Prod → GVC). The pairwise Granger causality test points to a unidirectional relationship at a p=0.041. This may be indicating that indicating that changes in the number of exports of forest products affects the GVC with a 2-year time lag.

The pairwise Granger causality tests show that there are short-term causality effects arising from the relationships between exports of forest products, GVC and governance. For these variables, causal relationships start to arise after periods of only 2 or 3 years. In the long run, governance appears to have a more direct impact on GVC and exports of forest products in the delta. This shows the input resulting from governance status in the country, consequently affecting exports of forest products and GVC. This indicates that the governance status in the country is key variable, influencing the exploitation of natural forest resources in the Amazon estuary.

Major changes in FC have been taking place in the region, especially in the Southeast and Northeast of the estuary. The areas where deforestation has been taking place more rapidly are also those with well-developed road infrastructure, consequently connecting the region to markets (Lambin et al., 2001; van Vliet et al., 2013b; Walker et al., 2013). After continuous years of environmental degradation, the capacity of Brazil to protect and regulate the use of its forests through improved governance may lead to a more environmentally sustainable future for its forests. Nevertheless, what has been revealed in this paper reflects insufficient environmental monitoring and law enforcement, combined with consecutive years of social and economic instability (with roots in a military dictatorship that ruled Brazil from 1964 to 1985) (Cole and Liverman, 2011; Lissovsky and Aguiar, 2015). This setting has created the development path for Brazil up the ladder of increased economic performance with no attention to the sustainable use of its forests (Lissovsky and Aguiar, 2015; Pinto, 2015) (e.g. Kuznets curve in Appendix B which shows the state of development in the Amazon in comparison with the Mekong and Ganges Brahmaputra-Meghna deltas).

3.5 Conclusions

Since 1982 the Amazon estuary has been passing through significant transitions, with massive implications for the landscape. Inevitably, the states of Para and Amapa now face major social and environmental challenges resulting from the loss of ecosystem services.

A multitude of drivers have contributed to the situation on the ground, and the results presented here reveal the temporal changes, how the major variables are potentially linked, and how this is feeding back into both natural and anthropogenic systems. It is clear that the current situation is worsened by the interdependencies existing between drivers of change linked to economic development that are often controlled by societal and market demands originating both internally and externally. Areas that once were far away from the economic mainstream, due to historical factors and lack of infrastructure, are now accessible and fully integrated into the global chain of production and trade through increased road infrastructure, urbanisation, and expansion of the agricultural frontier.

The way that the landscape has evolved in the Amazon estuary is comparable to the way that other large estuaries and river deltas have evolved. Irrespective of its pace and pattern, the combination of economic development, population growth and changes in climate poses huge challenges to local governments, and their actions now will determine how they are going to cope with these challenges in the near future. Patterns now observable over recent decades in the Amazon estuary are a consequence of uninterrupted periods of resource exploitation and degradation of natural ecosystems. Ultimately, this will have short- and long-term implications for livelihood vulnerability and resilience in the context of global environmental change.

Chapter 4: Evolutionary social and biogeophysical changes in the Amazon, Ganges-Brahmaputra-Meghna and Mekong deltas.

4.1 Introduction

Deltas are economic and environmental hot spots, home to a significant proportion (>500 million) of the world population (Ericson et al., 2006b) and sheltering rich and biodiverse ecosystems (Bianchi and Allison, 2009). The rivers that flow through the deltas are an important source of fresh water and nutrients and create ideal environmental conditions for food production (e.g. agriculture, fish farming and aquaculture production) and the support of biodiversity (Kuenzer and Knauer, 2013; Seck et al., 2012; Syvitski, 2008; Wong et al., 2014). The increasing pace of human development in coastal deltas over the past five decades has strained environmental resources and produced extensive economic and sociocultural changes in deltas (Duval-Diop and Grimes, 2005; Ernoul and Wardell-Johnson, 2013; Kuenzer and Renaud, 2012). Deltaic regions all over the world are changing rapidly due to human actions (e.g., pollution of water and sediment flow reduction/increase), climatic variability (e.g., droughts and sea level rise), and resource exploitation (e.g., mining, groundwater exploration and hydrocarbon extraction) (de Araujo Barbosa et al., 2016b; Holgate et al., 2013; Ibanez et al., 2014; Kuenzer and Renaud, 2012; Restrepo, 2013).

Changes in average climate conditions in deltaic regions strongly influence important physical and chemical properties within the ecosystems (Larsen et al., 2011; Morrison et al., 2006; Tsai et al., 2005). Extreme drought and rainfall peaks affect sediment shedding from the hinterland, water salinity, abundance of photosynthetic organisms, turbidity load, flood level, nutrient recycling and biological productivity, with strong potential effects on supporting, provisioning, cultural and regulatory ecosystem services (de Araujo Barbosa et al., 2015b; Dearing and Jones, 2003; Hinderer, 2012; Maya et al., 2011; Naidoo et al., 2008). Thus, deterioration of deltas will cause reduction of important ecosystem services and increase the susceptibility of these areas to extreme climatic events (Aerts et al., 2006; Frappart et al., 2006; McMullen et al., 2009; Nijssen et al., 2001; Szabo et al., 2015b; van Slobbe et al., 2013).

The majority of tropical deltas in developing economies may be defined as complex, coupled social-ecological systems supporting high population densities with particular vulnerabilities to sea level change and upstream river management (Llovel et al., 2010; Stanley and Hait, 2000; Walsh et al., 2014). Evaluating the specific nature of current vulnerabilities and how these may change in the near future requires consideration of the recent social-ecological dynamics (Dearing et al., 2012a; Duval-Diop and Grimes, 2005; Kuenzer and Renaud, 2012). Here we examine how social and ecological elements may have interacted over time to give rise to the complex dynamics that characterise the modern social-ecological systems present in the Amazon, Ganges-Brahmaputra-Meghna (GBM) and Mekong deltas (Figure 4.1).

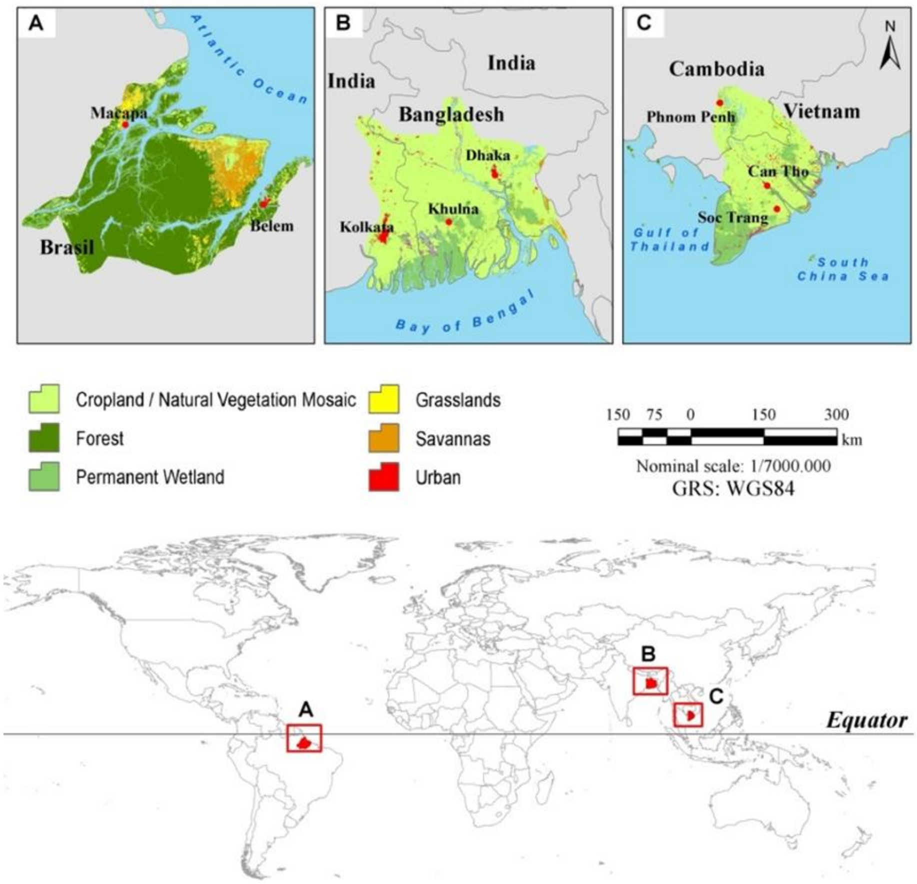


Figure 4.1 - Figures A, B and C represent the spatial extent of the three deltas, namely the Amazon, GBM and Mekong, defined as described in subsection 4.1.1. This figure contextualises the three deltas, geographic locations across the globe, and main land cover classes as observed from the Moderate Resolution Imaging Spectroradiometer (MODIS) in 2013, at 1km spatial resolution (de Araujo Barbosa et al., 2016e).

Multi-decadal trends for social, economic, ecological conditions and external drivers provide an evolutionary perspective that enable us to explore the recent changes in social-ecological dynamics, trade-offs, driver-responses, multivariate dynamics and regime shifts (Hossain et al. 2015; Zhang et al., 2015), also giving insight into what might constitute the conditions for safe and just operating spaces (Dearing et al., 2014). Ideally, an evolutionary perspective considers a large set of highly interconnected variables, with high spatial and temporal resolution, that allows detailed analysis of relationships over an extended multi-decadal period (Costanza et al., 2012; Dearing et al., 2014; Dippner and Kroncke, 2015). Unfortunately, many regions of the world, including tropical deltas, are deficient in key records for major social and ecological variables. Therefore, here we adopt an initial analysis of limited data in order to compare and contrast general properties of three major world deltas. We address the following general questions: (1) What are the observable dynamics, including key drivers and feedback loops, that are steering the system toward its current equilibrium or disequilibrium state? (2) How are these dynamics affected by human intervention and ongoing environmental change? (3) Are there any common human development trajectories, including population change, ecological deterioration, rates of poverty, and indicators of human wellbeing?

4.1.1 Study Sites

4.1.1.1 Amazon

The Amazon delta (Figure 4.1A) is one of the last frontiers for land development and agricultural production in Brazil (Garrett et al., 2013b; Lorena and Lambin, 2009; UNEP, 2004; Viers et al., 2005). The human populations living in the Amazon delta are highly dependent on local extraction of natural resources (Guedes et al., 2012; Ludewigs et al., 2009), and the densely inhabited areas now show declines in the abundance of fish and game, water quality and in the quality of soils for smallholding agricultural production (Almeida et al., 2003; Brabo et al., 2003). Natural and anthropogenic perturbations in the Amazon delta region are reported to be degrading its capacity to maintain carbon storage in biomass and soils, rainfall regimes, river flow, nutrient cycling and the modulation of regional climate patterns (Boerner et al., 2007; de Araujo Barbosa and Atkinson, 2013; Foley et al., 2007; Klemick, 2011). The delta is also predicted under climate change scenarios to experience a decrease in rainfall, and it is unclear how this will impact social and ecological systems developing in the delta (Nepstad et al., 2011; Tao et al., 2013; Vergara and Scholz, 2011).

In the Amazon River basin as a whole, the major mechanisms of economic development have modified the landscape continuously, starting in the 1950s, and leading to what we now see as widespread environmental degradation. Recent studies argue that deforestation in Amazonia decreased by 77% since 2004 and stabilised after 2009, as a consequence of forest policy interventions, private sector initiative and market conditions (Godar et al., 2014c). Therefore, the long lasting deforestation in the region may have passed a threshold and is now moving towards recovery, which may be an indicative of forest transition. However, this might not be the case in the Amazon estuary, as some of the main factors influencing deforestation there are still challenging Brazil’s efforts in preserving its tropical forests (de Araujo Barbosa et al., 2014a; de Araujo Barbosa et al., 2016d; Godar et al., 2012).

4.1.1.2 Ganges-Brahmaputra-Meghna

The GBM delta (Figure 4.1B) is notable for an extremely high population density, natural mangrove forest ecosystems (the Sundarbans), and a vast complex of intertidal and estuarine areas that provides nursery grounds for many species of fish and invertebrates across multiple political boundaries (Babel and Wahid, 2011; Hossain et al., 2012; Hossain et al., 2013; Siddique-E-Akbor et al., 2014). This delta has been experiencing a rapid urban growth, which has resulted in widespread social and economic inequality across different spatial scales (Babel and Wahid, 2011; Brichieri-Colombi, 2004; Sharma et al., 2010; Webster and Jian, 2011).

The recurrence of flood events in the region has long been viewed as a necessary trade-off against the relative beneficial conditions available for agriculture and food production (Asada and Matsumoto, 2009; Islam et al., 2010a; Ruane et al., 2013; Sharma et al., 2010). Nonetheless, during the dry season the GBM delta experiences tidal water movement of more than 100 km inland, and a relative sea-level rise exceeding the global average that reflects the impact of land subsidence, all of which contributes to an increasing salinity problem (Gupta et al., 2014; Hanebuth et al., 2013; Higgins et al., 2014; Pethick and Orford, 2013; Rogers et al., 2013; Shearman et al., 2013). Furthermore, infrastructure developments, such as the Farakka barrage on the main Ganges channel, have been influencing the regime of water flow in the region, with negative effects on water availability (Hossain et al., 2015a). In the long run, climate threats are likely to include changing distribution of river floods, warming temperatures, and changes in rainfall regime, which in turn will maximize current vulnerability to climate extremes, posing substantial challenges to sustainability in a region undergoing intense social changes (Asada and Matsumoto, 2009; Chakraborty, 2004; Cook and Lane, 2010; Gain and Giupponi, 2014; Islam et al., 2010b; Khan, 2012; Markandya and Murty, 2004; Varis et al., 2012).

4.1.1.3 Mekong

The Mekong river delta (Figure 4.1C) is the world's third largest delta, and it is formed by a large transboundary river system travelling through China, Myanmar, Lao People's Democratic Republic (Laos) Kingdom of Thailand, Cambodia and Vietnam (Armitage et al., 2015; Givental and Meredith, 2016; Kotera et al., 2008; Tin et al., 2001). It is a territory where regional meteorological and hydrological regimes support diverse social and economic activities, as well as a diverse and productive natural environment. The delta provides 50% of Vietnam's rice production and 80% of the aquaculture production (Gummert, 2013; Nguyen, 2011).

Rice cropping systems in the Mekong delta have shown signs of stress in response to an increasing number of severe floods, droughts, storms and tropical cyclones, followed by growing population demand for food production (Ahmed et al., 2014; Berg and Tam, 2012; Huysveld et al., 2013; Lusterio, 2009; Nguyen, 2011; Son et al., 2013). Food production systems are rapidly expanding into the flood and salinity-intrusion areas (Kotera et al., 2014; Tuong et al., 2003), as a result of engineering works aimed at protecting populations and infrastructure from storms, rice cropping systems and shrimp farms from saltwater intrusion (Berg et al., 2012; Nguyen et al., 2014).

The construction of several large-scale dams, and major channel-bed mining activities (Piman et al., 2013), have now been reported as the cause of an imbalance between flow and sediment entrainment conditions (Xue et al., 2011), affecting human livelihoods and the ecological equilibrium in the delta. Predictions of exposure to the effects of climate change for the Mekong delta include a rising sea level, rising temperatures, increased variability in rainfall regime and higher frequency of extreme events (Haruyama and Ito, 2009; Kotera et al., 2008; Nguyen et al., 2014). These changes are likely to have strong negative impacts on the production of food and human wellbeing (Berg et al., 2012; Berg and Tam, 2012; Kotera et al., 2014; Nguyen, 2011). The combined effect of economic development, agricultural production practices, and change in consumption patterns is likely to increase the relative vulnerability of this delta to social and ecological changes (Coclanis and Stewart, 2011; Dang et al., 2014; Dun, 2011; Few and Pham, 2010; Kuenzer and Renaud, 2012; Piman et al., 2013; Quyet et al., 2012; Smith et al., 2013; Vu et al., 2014).

4.2 Methods and techniques

4.2.1 Conceptual framework

The framework used in this study (Figure 4.2) serves to provide a rationale for the dynamic and integrated approach used in this work to investigate social and biogeophysical changes occurring in the three tropical deltas. This framework is based on a set of indicators reflecting the current and past state of provisioning and regulating ecosystem services. These two ecosystem service categories are considered to be of key importance, having major direct and indirect impacts on human wellbeing. For example, deforestation can affect people's livelihoods by eliminating access to provisioning services and leading to changes of landscape, which can in turn increase the risk of natural hazards (Alcamo et al., 2003). Indirect effects can operate through a number of biophysical, socio-economic and political processes. More specifically, soil salinity can negatively affect food security and health outcomes by impacting water quality (Alcamo et al., 2003; Szabo et al., 2015a). Lack of safe drinking water may lead to communal tensions and violence (Fjelde and von Uexkull, 2012; Meier et al., 2007), though the impact of increasing/decreasing ecosystem services on human wellbeing indicators seems to have produced conflicting findings over the decades (Millennium Ecosystem Assessment, 2005).

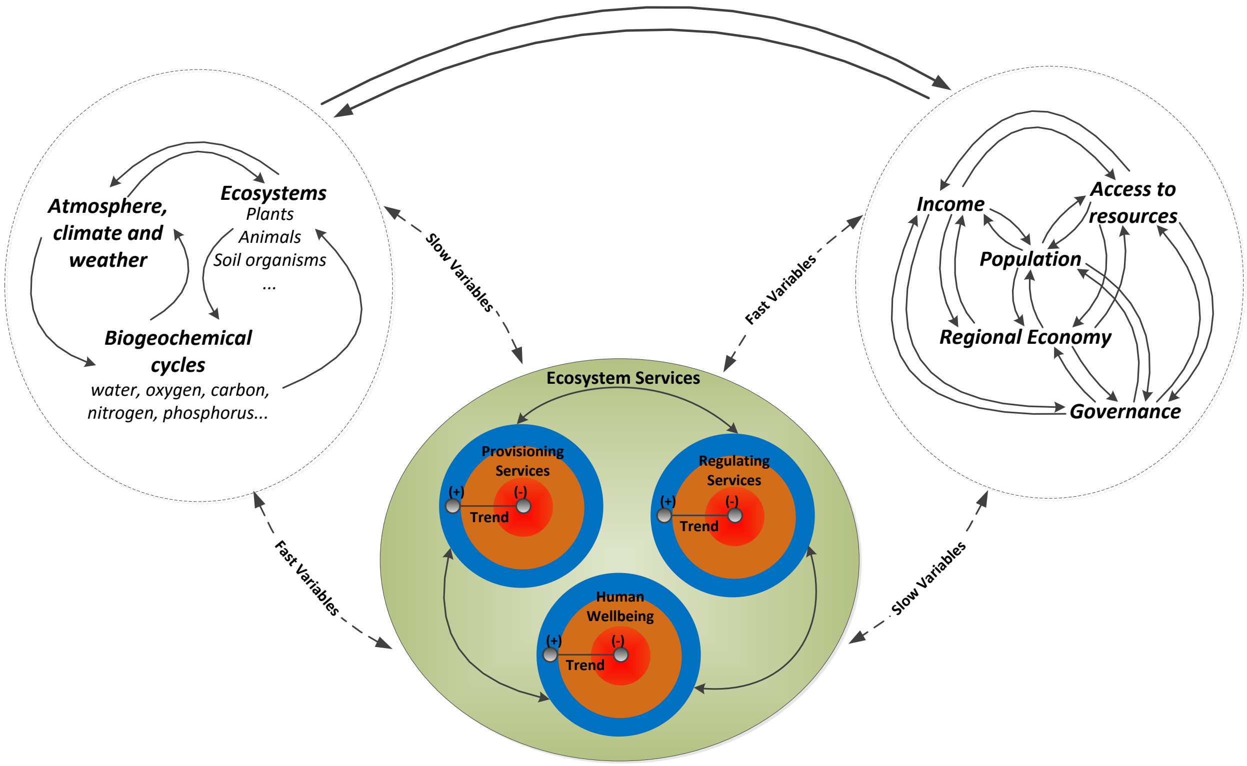


Figure 4.2 - Systemic dynamic relationships between environmental and human dimensions affecting the changes on ecosystem services and human wellbeing through time (de Araujo Barbosa et al., 2016e).

Provisioning services encompass crops, livestock, freshwater, fisheries and aquaculture (Carpenter et al., 2009) and are thus critical to reducing the risk of food insecurity and enhancing broader wellbeing. Regulating services, which in this study, mainly involve regulation of water and air quality can affect human wellbeing in a number of ways. For example, failure to prevent air pollution can lead to ill health, which is increasingly a challenge in the context of rapid urban growth in many delta regions. Other examples include development of adequate infrastructure, such as buffering zones along the coasts through plantations of mangroves and beach forests (Butler and Oluoch-Kosura, 2006). Regulating services are intrinsically linked to provisioning ecosystem services through a two-way relationship. For example, deteriorating quality of water and food may trigger new environmental policies with new regulations that constrain land use practices. Finally, external drivers, include, on the one hand, human drivers, (e.g. national governance and international trade) and, on the other hand, environmental factors (e.g. rising temperatures and sea levels). All these factors affect boundary conditions and the internal dynamics of ecosystem services and wellbeing interactions. Ultimately, all these dynamics have an impact on the wider socio-economic development of the delta regions and beyond. The variables chosen for our analysis were selected in order to provide clear insights into the hypothesised relationships and complex dynamics of society and nature under an ecosystem services framework (Costanza et al., 2012; Dearing et al., 2014).

4.2.2 Delta definitions

For the purposes of data collection, we have defined the spatial extent of each delta primarily by the area downstream of the first distributary as mapped by the Shuttle Radar Topography Mission (SRTM). For the GBM, the altitude of the first distributary (the Hoogli river) at the Farraka Barrage is about 18-20 m asl. Using an SRTM contour map, the 18-20 m asl contour includes the main areas of frequent flooding along the Brahmaputra river and in the Sylhet basin. This definition is represented by over 45 districts in whole division areas of Khulna, Barisal, Dhaka, Sylhet but also a large part of Chittagong division in Bangladesh, and the Districts of Nadia, South 24-Parganas and North 24-Parganas in India. For the Mekong, the first distributary point at Phnom Penh lies at about 7-9 m asl. The Mekong SRTM contour map shows that this would delineate a delta mainly in Vietnam with a small part in Cambodia, and excludes Ho Chi Min City (Vietnam) and the Phnom Penh Municipality (Cambodia). The Amazon is less a delta and more an estuary. Therefore, we follow Ericson et al. (2006b) use of a 5 km buffer from coastline that intersects roughly with the first distributary. This definition maps well to the distribution of municipal districts.

4.2.3 Data collection and Analysis

Data availability for deltas is a challenging issue because official statistics tend to map onto national or regional administrative areas rather than physiographically defined areas. Here, we have compiled data from official regional sources (with two variables at the country level feeding into food and nutrition) in order to create realistic regional time-series of major physical drivers, regulating and provisioning ecosystem services, and human wellbeing over multiple decades (Table 4.1). We use z-scores to standardize our variables, allowing us to position variables, relative to other variables, and therefore establish relevant comparisons between observed social and ecological states and transitions along the time series. For a total set of variable observations, dimensionless z-scores are obtained by subtracting the variable mean from an individual raw score, and dividing by the standard deviation (Eq. 1).

$$z = \frac{(x - \mu)}{\sigma} \tag{1.4}$$

In Eq. 1 z refers to z-score, x is the value of a given variable, μ is the mean of the total set of observations for a given variable, and σ is the standard deviation. Data for provisioning services and drivers are plotted with y axes showing positive z scores for high levels of each variable (e.g. high grain yields). Data for regulating services are plotted on reversed y axes showing positive z scores for desirable ecological quantities and negative z scores for undesirable (e.g. low salinity), as determined locally (Table 4.1).

Table 4.1 - Data collected for the three deltas grouped by data category (physical drivers, regulating services, provisioning services and human wellbeing), data temporal coverage for the Amazon (AM), Ganges-Brahmaputra-Meghna (GBM) and Mekong (MK), with respective data sources and geographical locations where the data was collected.

Category	Data	Temporal Coverage			Source	Geographical Location		
		AM	GBM	MK		AM	GBM	MK
Physical drivers	*Temperature*	1960-2013	1961-2013	1984-2013	Brazilian Meteorological Service (INMET, 2013); Bangladesh Meteorological Department (BMD, 2014); Vietnam Hydro Meteorological Service (NHMS, 2014); Mekong River Commission (MRC, 2014).	Belem, Macapa, Altamira, Soure, Cameta and Breves municipalities	Khulna, Barisal and Patukhali districts	Soc Trang, Can Tho, An Giang and Vinh Long provinces
	Rainfall	1960-2013	1950-2013	1984-2013				
	Relative air humidity	1960-2013	-	1984-2013				
	Water level	1968-2013	1979-2013	1988-2013				
Regulating services	*Water Level (-desirable / +undesirable)*	1964-2012	-	1988-2012	Brazilian National Water Agency (ANA, 2014); Brazil's National Institute For Space Research (INPE, 2014); de Araujo Barbosa et al. (2016a); Bangladesh Water Development Board; Bangladesh meteorological department; Islam et al. (2011); Vietnamese Hydro-Meteorological Service (NHMS, 2014); Mekong River Commission (MRC, 2014).	Amazon River (Macapa and Almeirim) Amazon estuary	Khulna, Barisal and Patukhali districts	Can Tho, An Giang and Vinh Long provinces
	Water discharge (-desirable / +undesirable)	1898-2013	1978-2008	-				
	Water Quality (Salinity) (high salinity undesirable)	-	1964-2008	1984-2013				
	Water Quality (Water pH) (low pH undesirable)	1952-2013	-	-				
	Forest cover (high forest cover desirable)	1982-2013	1985-2014	-				
	Sediment Concentration (-desirable / +undesirable)	-	-	1986-2010				
Provisioning services	*Shrimp Production*	-	1991-2007	-	Brazilian Institute of Geography and Statistics; Brazilian Institute for Applied Economic Research; Bangladesh Bureau of Statistics; General Statistics Office of Vietnam.	Amapa and Para states	Khulna, Barisal and Patukhali districts	Long An, Tien Giang, Ben Tre, Tra Vinh, Vinh Long, Dong Thap, An Giang, Kien Giang, Can Tho, Hau Giang, Soc Trang, Bac Lieu, and Ca Maun provinces
	Fish production	-	1950-2010	1989-2013				
	Aquaculture Production	-	-	1989-2013				
	Rice Production	1944-2013	1961-2013	1989-2013				
	Cassava production	1944-2013	-	-				
	Animal production	1990-2013	-	1989-2013				
	Agricultural production	1944-2013	-	-				
Human Wellbeing	*Total population*	1960-2013	1950-2014	1989-2013	Brazilian Institute of Geography and Statistics (IBGE, 2014); Bangladesh Bureau of Statistics (BBS, 2013); General Statistics Office of Vietnam (GSO, 2014); Food and Agriculture Organization of the United Nations. (2014)	Amapa and Para states, Brazil	Khulna, Barisal and Patukhali districts, Bangladesh	Can Tho, An Giang and Vinh Long provinces, Vietnam
	Real inflation-adjusted GDP	1960-2013	1960-2014	1989-2013				
	Employment	1990-2013	1990-2014	1989-2013				
	Food security	1990-2013	1990-2013	1990-2013				

4.2.4 Dynamic Principal Component Analysis

Connectivity is increasingly viewed as an important property of complex systems, especially with regards the higher levels of connectivity and homogenisation observed in unstable systems (Scheffer et al., 2012). Here we use dynamic or sequential Principal Component Analysis (PCA) (Billio and Caporin, 2010; Zhang et al., 2015b) to provide a crude measure of the multi-decadal changes in connectivity between provisioning and regulating ecosystem services in the three deltas. We applied this approach to ecosystem service time- series indices since the 1960-70s (1990s in the case of the Mekong delta), calculating covariance using a 10 year moving window. PCA axis 1 (first eigenvalue) captures ≥ 50% of the variation for the combined datasets in all three deltas.

4.2.5 Key system variables

4.2.5.1 Physical Drivers

In each delta, the selected variables define the observed changes affecting the physical environment that will in turn feed back into key ecosystem services. (Carpenter et al., 2011; Nelson et al., 2006; Potschin and Haines-Young, 2011). For example, increasing variability in average values of temperature, rainfall and relative air humidity directly affect ecosystem conditions and services, and are key variables in sustaining people’s livelihoods across all levels of society (Harborne, 2013; O'reilly et al., 2003; Sara et al., 2014; Sullivan and Huntingford, 2009; Walker et al., 2008). In deltas and estuaries, water level plays an important role in preserving biodiversity, enabling agriculture, climate buffering and flood control. Additionally, the effective monitoring of water levels through time provides important insights into current and past management (e.g. prioritization of hydropower over other ecosystem services) (Day et al., 2008; Notter et al., 2012; Omer, 2009; Taguchi and Nakata, 2009). Nevertheless, the situation observed in these deltas over the decades is only partially driven by physical drivers, and human activities are of critical importance. Therefore, an integrated approach to the problem of environmental change, its short and long-term effects on ecosystem services, is key to achieving environmental sustainability in deltaic regions (Darby et al., 2015; Dearing et al., 2014; Mononen et al., 2016).

4.2.5.2 Regulating Services

The variables chosen here represent the multiple regulating ecosystem services found in deltaic regions (Barbier et al., 2011; de Araujo Barbosa et al., 2014b; Gonzalez-Esquivel et al., 2015; Iacob et al., 2014; Withers and Jarvie, 2008; Yoo et al., 2014). These variables provide valuable information about the major trends in important regulating services (e.g. climate regulation, water regulation, nutrient cycling, erosion control, flood control, and others) (Feld et al., 2009; King and Brown, 2010; Ma and Swinton, 2011; Nedkov and Burkhard, 2012; O'Leary and Wantzen, 2012) and provide the basis for exploring interactions with drivers and social conditions.

Over time, the interactions between salinity (water quality), water pH (water quality), sediment concentration (erosion control, water quality), and forest cover (climate regulation, erosion control) define the evolution of many common biophysical effects in the three deltas (Basher, 2013; Beier et al., 2015; Clarkson et al., 2013; de Araújo Barbosa et al., 2010; Fezzi et al., 2015; Grimaldi et al., 2014; Harmackova and Vackar, 2015; Sabater and Tockner, 2010; Sturck et al., 2014; Terrado et al., 2014; Turner et al., 2013). These effects influence ecosystem regulating services, with potential repercussions not only at the local, but also, at the regional and global scales (Bagley et al., 2014; Downing et al., 2014; Hicks et al., 2015; Laterra et al., 2012; Rodriguez et al., 2006; Teferi et al., 2010; Trumbore et al., 2015; Westphal et al., 2015).

4.2.5.3 Provisioning Services

In order to represent the trajectories of provisioning services in the three deltas over time, we use a set of key variables able to give important information about the major components of ecosystem services availability and flow at the delta scale (Gonzalez-Esquivel et al., 2015; Mononen et al., 2016; Ramirez-Gomez et al., 2015).

The variables in Table 4.1 serve as indicators not only of how much food has been produced in these deltas, but also the current level of natural resource exploitation and the intensification of various production systems in place (Gonzalez-Esquivel et al., 2015; Ramirez-Gomez et al., 2015; Suich et al., 2015). When linked to wellbeing indicators (Table 4.1) these variables give valuable information on how the trajectories in food production translate into the actual availability of food to the populations living in the delta. Furthermore, we can hypothesize how increases or decreases in production feed back into important regulating services and human wellbeing indicators (Albert et al., 2016; Essington and Munch, 2015; Macadam and Stockan, 2015; Wood et al., 2016).

4.2.5.4 Human Wellbeing

We selected variables that would provide conceptual and methodological confidence for plotting long term trajectories of human wellbeing (Bieling et al., 2014; Escobedo et al., 2015; Vidal-Abarca et al., 2014). Previous measures of socio-economic impacts of globalisation and large-scale development interventions have largely been restricted to indicators focused only on income, which do not necessarily reflect social needs and priorities (Dasgupta, 2010; Deutsch et al., 2003; Jordan et al., 2010; King et al., 2014; Malovics et al., 2009; Pretty, 2013; Reyers et al., 2013).

A set of indicators providing data on more explicit social facets, such as freedom and choice, human health and social relations, would be of great value but such data, even when available, does not provide sufficient temporal resolution in order to connect with the main objectives of this study. Therefore, we represent human wellbeing using simple and representative metrics, with time series of population growth, employment, gross domestic product (GDP), and food security providing the means to understand the multi-faceted impacts of biophysical change, globalisation and economic development on people's lives as the system co-evolves through time (Bieling et al., 2014; Chiesura and de Groot, 2003; de Freitas et al., 2007; Farley, 2010; Howe et al., 2014). GDP has been widely perceived as a measure of output rather than wellbeing but in cross-country data, GDP per capita, is positively correlated with life expectancy and negatively correlated with infant mortality, material living standards, health, education and political voice: all factors that are important for defining human wellbeing (Cohen et al., 2014; de Oliveira and Quintana-Domeque, 2014; Hou et al., 2015; Ngoo et al., 2015; Schmelzer, 2015; Schoenaker et al., 2015; Vecernik and Mysikova, 2015).

4.3 Results

4.3.1 Amazon

4.3.1.1 Physical drivers

The data for rainfall, temperature and relative air humidity were collected from six meteorological stations in the estuary and averaged to give insight into the climatologic variability operating at the delta scale (Figure 4.3c). We can see that there is a strong increasing trend in the mean temperatures starting in the early 1990s. Mean temperatures in the delta have significantly increased during the period 2000-2013 (from ≈31 ^{o}C during the 1960s to ≈32 ^{o}C). Annual mean rainfall values in the Amazon delta show a drying trend in the period 1960-2000 (from ≈2800 mm to ≈2200 mm since 2000). Average values corresponding to relative air humidity have responded promptly to changes in temperature and rainfall. The decadal relative air humidity has decreased from an average of ≈85% to 83%.

4.3.1.2 Regulating services

Figure 4.3d shows trends in water pH (ANA, 2014), water discharge and water level in the delta. Water acidity in the Amazon delta shows a relatively stable trend over the period, with mean pH values observed in the main river channel declining during 1960-1975, showing smallest variations after that (starting in 1984). The line data trajectory shows a rising trend on mean water discharge values, starting to pick up constant pace after 2005. The average water discharge in the Amazon delta shows an intermittent pattern, with major changes starting to take place after 1968. The mean water levels in the Amazon delta are increasing rapidly (starting in 2000), with negative impacts on related regulation services. The annual deforestation values here represent annual figures only for the Amazon delta region, comprising the federative states of Amapa and Para (de Araujo Barbosa et al., 2016d). The rise in mean water levels is associated with the increasing number of flooding events taking place in the delta (Bradshaw et al., 2007; Espinoza et al., 2012; Pinho et al., 2015). The peak in deforestation occurs during the late 1990s and persists during the early 2000s till 2005. There is a decreasing trajectory in the annual values of deforestation in the Amazon delta starting during 2006.

4.3.1.3 Provisioning services

For most of the period 1944-2013 the production of cassava and rice remained high in the region, with fluctuations in accordance with government subsidies, and in response to markets. In figure 3b we see a sharp decrease in the production of cassava (starting during mid 1990s) and rice (starting in early 2000s). These changes in agricultural production in the region are followed closely by significant increases in livestock production. More recently, production of other commodities such as corn, sugarcane and soybean has influenced the sharp rise in overall total agricultural production.

4.3.1.4 Human Wellbeing

Human wellbeing indicators, such as GDP, collected at local level show a general increase over the period (Figure 4.3a). Nevertheless, GDP in the Amazon delta has decreased during the early 1990s and early 2000s, returning to an upward trend after 2005, with occasional changes throughout the entire period (from 1960 to 2013). The total population of Para and Amapa (the two federative states in the Amazon delta) show a significant rise starting in 1960 (increasing from 1.2 million to nearly 9 million inhabitants), an increase of 660% over the last 50 years. This has been followed by an equally rapid increase in the percentage of population living in urban areas, a general trend observed in most of the developing and developed world. The positive trend in GDP is reflected in the general improvement of other social indicators such as school attainment and child mortality rate. However, the apparent improvement in human wellbeing indicators does not have the same positive impacts on the proportion of population employed. It is clear (Figure 4.3) that employment is still a major issue, especially for young male individuals. During this time, food security indicators have improved, although with slight changes in the early 2000s (for cereal import dependency ratio) and later on after 2008 (for the prevalence of undernourished children).

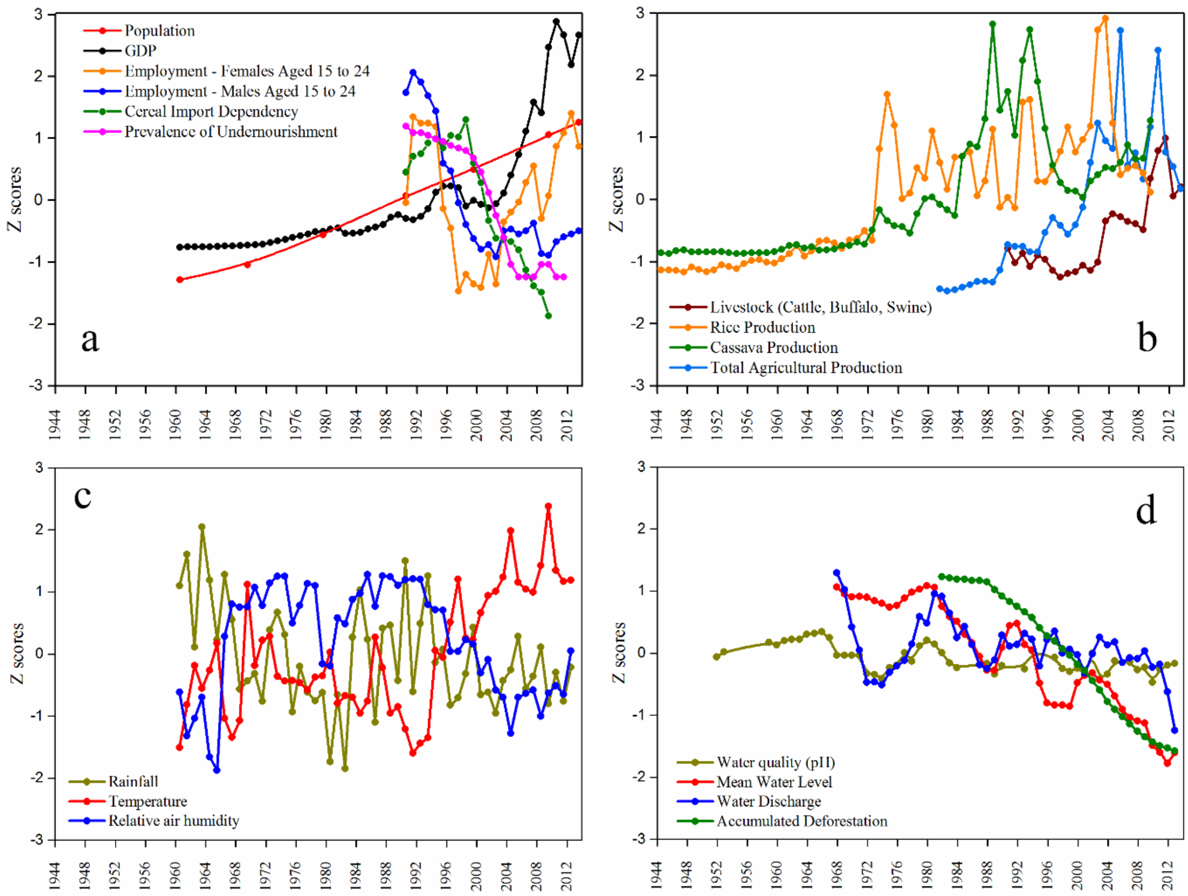


Figure 4.3 - Amazon delta 1944-2014: annual data for a) human wellbeing, b) provisioning services, c) physical drivers and d) regulating services. The plotted lines represent desirable (ascending progression) and undesirable (descending progression) Z-score values for regulating services (mean water level along the main river channel; river water discharge; water quality along the main river channel; accumulated deforestation in the delta). Ganges-Brahmaputra-Meghna

4.3.1.5 Physical drivers

Temperature figures show a step change at ≈1990 (when the mean temperature rose >26 °C whereas, mean temperature was <26 °C before 1990s) (Figure 4.4c) (Hossain et al., 2015a). Mean annual rainfall increased from ≈2500 mm before 1970 to 3000 mm after 1970. However, mean annual rainfall has decreased from ≈3000 mm to ≈2000 mm after 2007. From 1978 onwards sea level shows a rapid upward trend (confirming what has been pointed previously by Auerbach et al. (2015); Brown and Nicholls (2015); Kay et al. (2015)) this trend shows signs of slow down after 1990.

4.3.1.6 Regulating services

Salinity concentrations in the south west coastal area of GBM have increased from 5,000 Siemens (S) in 1970 to 50,000 S in 2005 suggesting that water quality in the region has degraded ≈10 fold within a 30-year period (Figure 4.4d). Although the mean (smoothed) annual water discharge shows a relatively stable trend, the original data shows fluctuation over the time period from 1977 to 2007. The mangrove forests in the region are undergoing accelerated deforestation, impacting the ability to provide important ecosystem services (e.g. erosion control) this has been developing much before 1982, with this trajectory becoming even more evident after 2004. Some of the peaks in the water discharge curve are linked to major flood events, especially 1988, 1995, 1998 and 2002 (Khan et al., 2015; Younus, 2014).

4.3.1.7 Provisioning services

Total rice production across the GBM delta rose four fold in the period 1961 - 2013 (Figure 4.4b). Although, the rice production has been rising steadily since 1961, with a rising trend that starts in 1995. Similarly, total inland fish catches, marine fish catches and shrimp production have increased between 1950 and 2010. Total fish catches increased between 1950 and 1970 (from 200,000 t to 600,000 t), whereas fish catches remained at a stable level between 1970 and 1990, before increasing sharply after 1990s. A similar sharp rising trend is observed for shrimp production with ≈3-fold rise between 1991 and 2008. Although the marine fish catch shows a steadily increasing trend since 1950, in 1974 it has significantly dropped, going through a period of recovery (1982-1997), followed by a sharp rising trajectory starting in 1998 (Hossain et al., 2015b).

4.3.1.8 Human Wellbeing

Total population in the GBM has been constantly increasing since 1950 (Figure 4.4a). Human wellbeing indicators such as GDP increased 35 fold (from ≈4,200 to ≈150,000 million USD in the period 1960-2012) with a rate of 1053 million USD yr^{-1} between 1960 and 1999 increasing sharply to 8300 million USD yr^{-1} since 2000. Consequently, the food security indicators show constant signs of improvement (with decreasing trend in cereal import dependency), although we can still observe deviations from the major trend in reducing the prevalence of undernourished children, which starts after 2005. While GDP and population in the region has been growing steadily since early 1950s, the situation seems very different when it comes to employment rates for both young men and women in the GBM delta, as it has been dropping constantly, with a significant increase from 1998 to 2005.

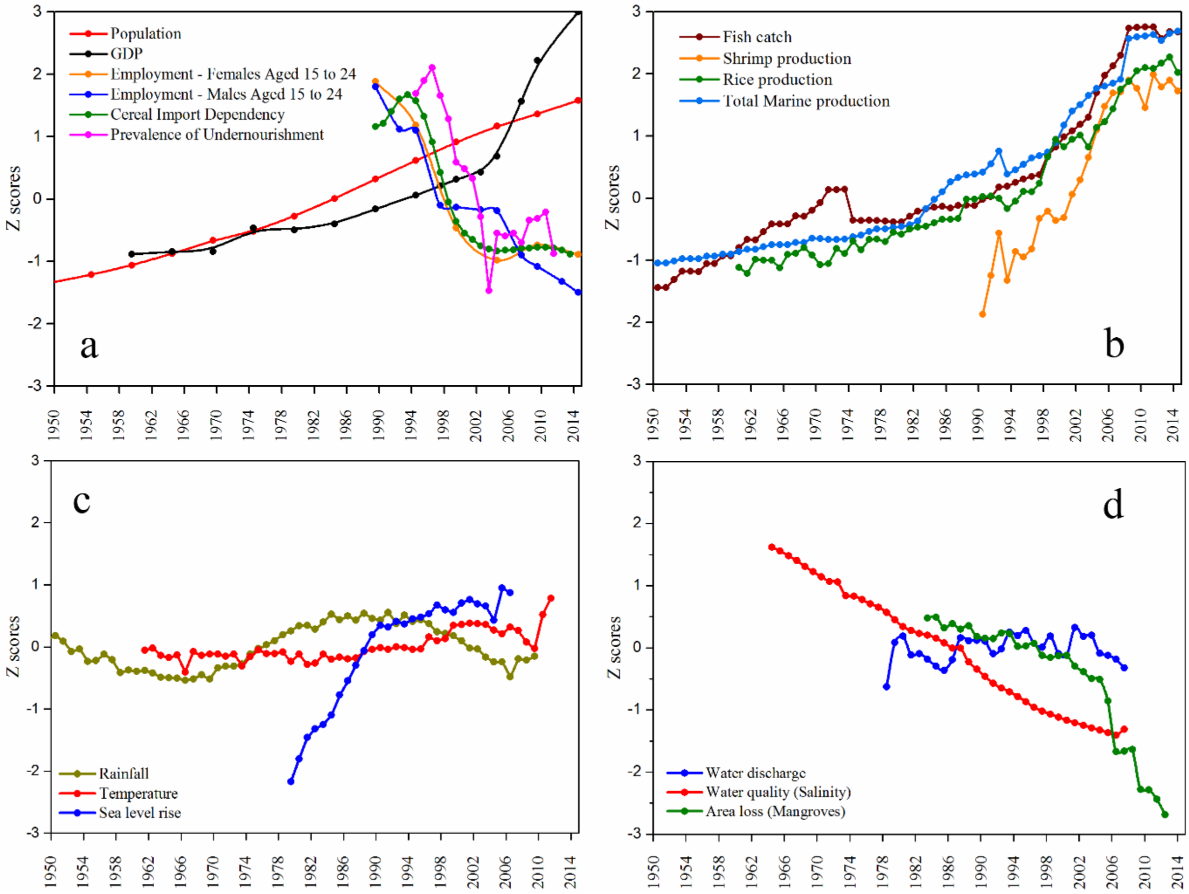


Figure 4.4 - Ganges-Brahmaputra-Meghna delta 1950-2015: annual data for a) human wellbeing ,b) provisioning services, c) physical drivers and d) regulating services. Sea level rise refers to changes in Mean Sea Level computed at tide stations. The plotted lines represent desirable (ascending progression) and undesirable (descending progression) Z-score values for regulating services (river water discharge; water quality at the coastal zone).

4.3.2 Mekong

4.3.2.1 Physical drivers

Mean temperature has tended to increase while relative air humidity and rainfall have been decreasing since 1990s (Figure 4.5c). Extremely hot years occurred in 1998, 2010, 2012 and 2013. As mean temperature increased, air relative humidity tended to decrease in the period 2000-2013. It is likely that saturation pressure of water vapour increased while vapour pressure has remained the same, causing the relative humidity to drop. Consequently, decreasing air humidity and rainfall can have negative impacts on crop production through higher crop irrigation requirement.

4.3.2.2 Regulating services

Levels of salinity, the concentration of suspended sediments, and average water levels have been changing considerably over the last few decades (Figure 4.5d). After 1989 water levels initiated a rising trajectory until 1996, when it is followed by a decreasing trend until 2001, where it seem to be following a more “stable” trajectory. This has been accompanied by a rising trend sediment concentration between 1986 and 1991, ending latter to give way to a more stable trajectory. The Mekong delta show a decreasing trend in salinity levels, between 1984 and 1986, when it starts a sharp decreasing trajectory until 2002, soon after changing its trajectory towards higher salinity values. Salinity intrusion in the Mekong is under constant change, continuing to cause negative impacts, as seen recently with 2016 being one of the worst years on record.

4.3.2.3 Provisioning Services

Positive trends can be observed for the production in all main crops and livestock (Figure 4.5b). Between 1990 and 2013 annual production of rice has generally increased, with only a few declines observed during the 2000s. The trajectory for total fish catch shows similar positive trend, closely followed by aquaculture production, with a few years where fish catch is positioned just below the aquaculture production. Nhan et al (2007) highlight that in the early 1990s less than 5% of the area suitable for aquaculture was used for that activity but this proportion increased to 22% by 2004 (Figure 4.5b). Livestock production rose significantly, with starting point during the 1990s, with its trajectory moving upwards during the period 1993 – 2006, and more recently (2008) it shows a descending trajectory.

4.3.2.4 Human Wellbeing

The region experienced increasing pressure from rising population numbers (Figure 4.5a). Population numbers in the Mekong delta have increased sharply with a change of 36% in the period 1990 – 2013. It has been followed (although not running as rapidly) by GDP, which has been following a major upward trend since 1990, a tendency that starts becoming more consistent during the early 2000s. During the same period, the employment ratio among the young population has been decreasing constantly, with a recent, although still modest, improvement starting in 2010. The food security indicators in the Mekong delta show signs of constant improvement in the form of a steadily decreasing trajectory in the prevalence of undernourished children. Deviations from this trend occurs during the period between 2009 but neverthless the situation seems very different from dependency on cereal imports. The sharp decrease during the early 1990s is replaced by a sharp rising trajectory after 1997.

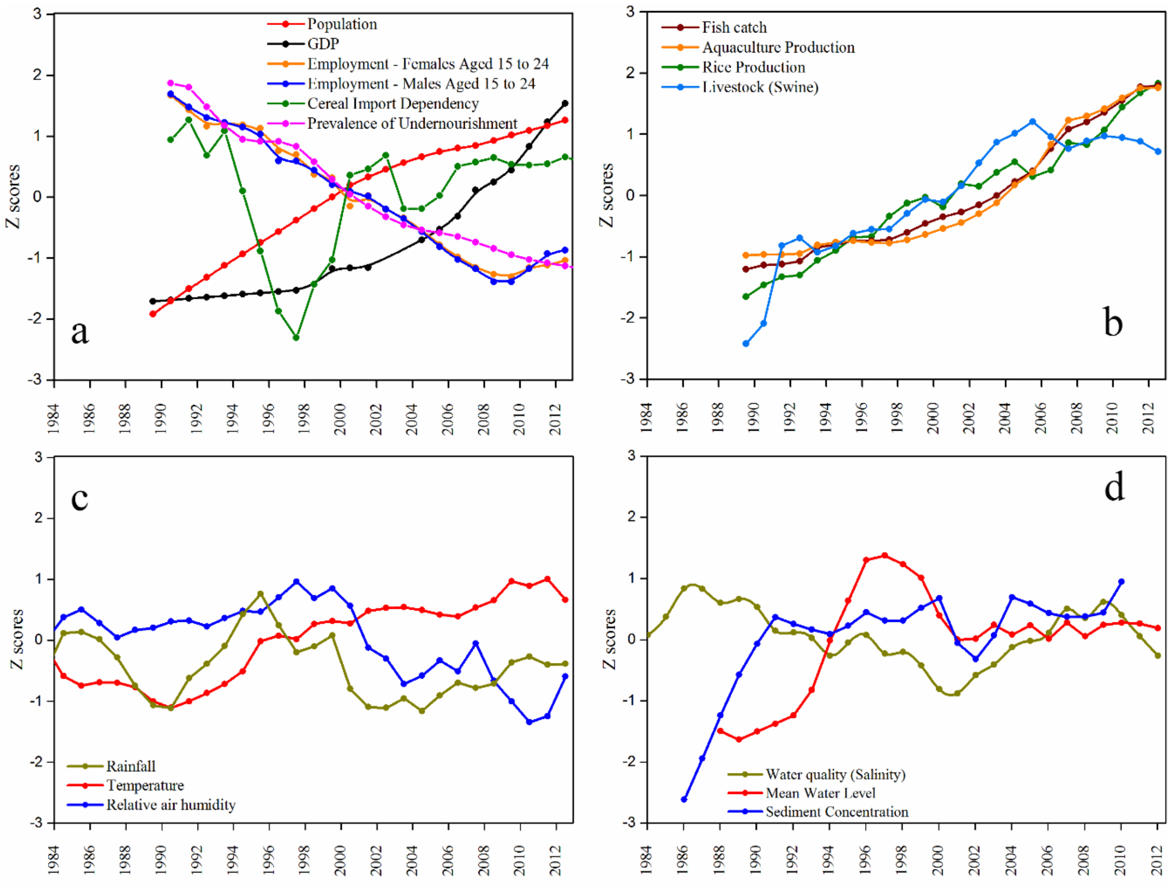


Figure 4.5- Mekong delta 1984-2013: annual data for a) human wellbeing, b) provisioning services, c) physical drivers and d) regulating services. The plotted lines represent desirable (ascending progression) and undesirable (descending progression) Z-score values for regulating services (mean water level along the main river channel; water quality along the main river channel; sediment concentration) (de Araujo Barbosa et al., 2016e).

4.3.3 Connectivity between provisioning and regulating services

The connectivity analyses (Figure 4.6) show remarkably similar results with the PCA 1 curves showing relatively high values before the 1990s with declining trends to the present day. The decline in PCA 1 values appears to start earliest (early 1990s) in the Amazon, later (late 1990s) in the GBM, and latest (early 2000s) in the Mekong. This common pattern may reflect the general effect of market economies and globalisation since the 1990s. The sequential PCA helped interpret the higher values before 1990 in terms of tightly coupled provisioning and regulating services at the regional scale, promoted by nationalisation and subsidies. For example, deteriorating water quality may have been more directly linked to standardised land use and agricultural practices that were often centrally controlled. The later effects of globalisation appear to have reduced the strength of this coupling, a finding that may be interpreted in alternative ways. The growth of provisioning services may have risen without producing proportional negative environmental impacts (e.g. Mekong: Figure 4.5), perhaps as the result of farming diversification and technological advances. But equally, the findings could be interpreted as an acceleration of environmental deterioration occurring despite a slowing down of agricultural production (e.g. Amazon, GBM: Figures 4.3 and 4.4).

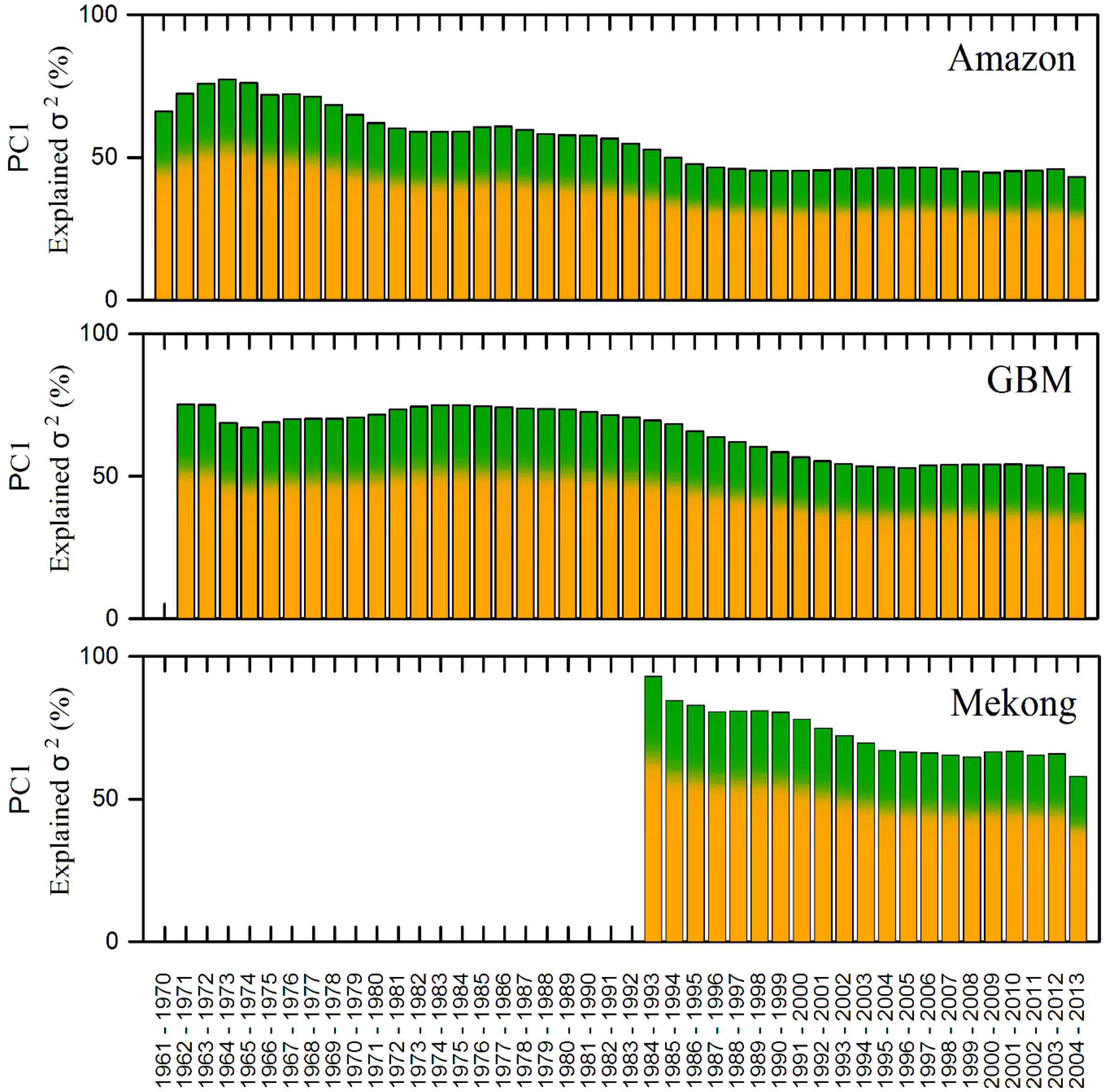


Figure 4.6 - Dynamic Principal Component Analysis using a 10 year moving window applied to time series indices for provisioning and regulating ecosystem services in each delta. The bars represent the proportion of variance (σ^2) explained by the first principal component during the moving time window.

4.4 Discussion

The general pattern of long-term trends from at least 1990 across the three deltas describes rising population and GDP, rising or fluctuating provisioning services, fluctuating climate drivers, and declining or fluctuating regulating services. Additionally, employment levels of young people have declined markedly. Major regional exceptions are found in the Amazon where there has been a rise in temperatures since the mid-late 1990s and continued growth in livestock production in the Amazon since the mid-2000s at the expense of rice and cassava production. Over a multidecadal timescale, none of the deltas currently show any variable running on with a quasi-stationary pattern. In system terms, all the deltas are in non-stationary or transient states with little evidence for stability or equilibrium.

The most rapid trends in each delta (relative to all others) are for GDP growth which has risen ≈3 standard deviation units since 2002 (Amazon), 1994 (GBM) and 1999 (Mekong). These levels of growth illustrate the successful economic development of the three regions associated with national schemes for poverty alleviation and the large-scale effects of globalization since the 1990s. Certainly, the rising trend of GDP in GBM has resulted in an improvement of other indicators such as school attainment and child mortality rate (Chowdhury et al., 2013; Dalal and Goulias, 2010). A major part of the economic growth has been based on intensification of agriculture and aquaculture as shown by the rapid trends in provisioning services reflecting the use of hybrid grains, fertilizers, pesticides and technological advances, such as irrigation techniques. The most rapid trends for these are livestock production since 2000 (Amazon), shrimp cultivation (GBM), and aquaculture production (Mekong).

Where the data exist, the evidence suggests that the rise in provisioning services has adversely affected the regulating services. In the Amazon delta, the rising trend for deforestation (≈3 standard deviation units between 2002 and 2010) seems to be a direct consequence of the expanding livestock grazing (de Araujo Barbosa et al., 2016d), and contrasts with claims that deforestation rates across the whole Amazon basin are stabilizing (Godar et al., 2014c). In the GBM and Mekong, the declining water quality (≈2 standard deviation units 1982-2008 and 1986-2000 respectively) is the result of higher salinity levels caused by shrimp ponds, inefficient irrigation, lower river discharges and marine intrusions (Abedin et al., 2014; Haider and Hossain, 2013; Kay et al., 2015; Pokrant, 2014). Local studies in the GBM coastal zone show that conversion of rice fields to shrimp farms is almost certainly a factor in increasing soil and surface water salinity (Hossain et al., 2015a).

Across the three deltas, the chains of causation that link governance and policy to farm decisions, through to production levels and environmental degradation are varied and complex. For example, deforestation in the Amazon represents a local and direct human activity, driven by a diverse set of demands generated across the globe, dependent upon complex interactions between domestic and trans-boundary drivers. Over time, transitions in global and regional temperatures and salinity values are driven by various distal (upstream), environmental, direct and indirect human actions that makes them more or less uncontrollable within the deltaic system (Ferguson et al., 2013; Folke et al., 2004; Janssen et al., 2004; Ullah et al., 2015). In the Mekong delta, the adverse effects arising from climate change, altered natural flow patterns in water and in sediments (as a consequence of hydropower development), have already created transboundary environmental tensions and reduced the capacity of people to maintain their livelihoods(Li and He, 2008; Lu and Siew, 2006; Manh et al., 2015; Xue et al., 2011).

In system terms, we can argue that for all three deltas the inherent dynamics of function, resilience, connectivity and feedback have significantly changed. The boundary conditions (e.g. climate, regional economies) that constrain the system functioning have significantly changed over the studied time periods. System resilience as defined by the condition of 'slow' variables (e.g. forest cover, water quality) shows long term decline. National policies and globalization have affected the coupling of ecosystem services. And, in each delta there are examples of rapidly rising or declining trends that may be viewed as the result of strengthening positive feedback mechanisms that link land use decisions to farm incomes and profit generation that in turn link to environmental degradation, often in the absence of robust environmental regulations. We may surmise that, in system terms, all three social-ecological systems may have moved outside safe operating spaces into unsustainable configurations.

Therefore, the overall changes in these deltaic social-ecological systems may be described in general terms as unsustainable trade-offs between rising food production and a deteriorating natural environment, though perhaps only to a limited extent in the Mekong. The leading question is what the long-term consequences might be. Lower rates of deterioration in recent years in the GBM and recent fluctuations of water quality in the Mekong suggest that the long term environmental decline may be stabilizing (Li and He, 2008; Renaud et al., 2015; Tho et al., 2012). But the most recent provisioning data indicate slowing or stabilizing trends for rice and livestock production (Amazon), rice production (GBM) and livestock production (Mekong). We may therefore ask whether these are the result of negative feedback loops driven by deteriorating regulating services, that are now constraining food production, or whether market conditions have changed, or more sustainable methods have been introduced?

Unfortunately, this is difficult to answer. The connectivity analyses underline the possible negative and positive effects of globalisation on the relationship between provisioning and regulating services but do not provide evidence for causation. Regulating services may have declined generally but the evidence for negatively impacting agricultural production levels is equivocal because the effects of ecological degradation, environmental regulation and land use selection on provisioning services are conflated. In our exploratory analysis we are constrained by the availability of data and a simple trend analysis of standardized data that cannot compare trends in terms of absolute effects without calibration. For example, one z-score unit of change in temperature may be more harmful to crop production than the same relative change in salinity. However, the general decline in regulating services has to be viewed as an unsustainable loss of natural capital with the possibility, as argued elsewhere (Raudsepp-Hearne et al., 2010), for time-lagged declines in crop and fish production as positive feedback mechanisms strengthen (Hossain et al., 2015a).

The possibility that rapidly declining regulating services in the Amazon and GBM are now increasingly decoupled from agricultural production levels (though not the agricultural practices themselves) is consistent with a heightened risk for rapid social-ecological change as unsustainable system dynamics play out through tipping points and regime shifts. It may be uncertain whether these mechanisms are happening already but they will certainly be accentuated by the observed rises in temperature which may be expected to adversely affect pollination, crop yields and surface water quality. From a systems perspective of sustainability, the current deltaic systems lie outside safe and just operating spaces in potentially dangerous zones (Dearing et al., 2014). Current work is aimed at developing simulation models that can both capture the complex dynamics revealed here and anticipate the effects of alternative governance measures on future social-ecological states.

4.5 Conclusions

This paper shows that the biophysical and socioeconomic changes in the three deltas have similar origins. Several well-established mechanisms exist to explain the overall variation we observe in the three deltas. These mechanisms are created and maintained by feedbacks originated from the interaction between society and the environment. Current and past observed condition were shown here using a series of records denoting changing patterns of rainfall, temperatures, sea level rise, water levels and discharge, sediment flow, forest cover, in the context of intensive social changes. The complex myriad of interactions currently in place in these deltaic systems will increasingly be affected by changes in average weather conditions, management practices and the fast pace of resource exploitation. The power of international and institutional arguments that emphasize the necessity for sustainable economic development, and adaptation to changes in climate, has yet to translate into effective safeguards for the sustainable use of natural resources in these deltas and the livelihoods and wellbeing that depend upon them. This finding suggests that the sustainability of tropical deltas urgently requires the decoupling of local economic growth from local resource use before irreversible ecological shifts develop.

Chapter 5: The crossroads between economic growth and environmental sustainability in the Amazon estuary

5.1 Introduction

New research shows that deforestation rates in the Brazilian Amazon have not stabilised since 2011 as previously argued. In fact, the clearance of forested land in the Brazilian Legal Amazon has increased since 2012 (de Araujo Barbosa et al., 2016b; Fearnside, 2015a; Godar et al., 2014c). In the States that house the estuary (Para and Amapa States) average deforestation rates have increased by 35% since 2012-13, rates that are high even when compared to the rest of the Brazilian Legal Amazon (INPE, 2015). Recent literature has not given sufficient attention to the causes of the recent deforestation trends (e.g. Aguiar et al. (2016a); Godar et al. (2014b)). It therefore remains unclear as to whether recent trajectories in deforestation are a result of policy interventions or well-established socioeconomic drivers, and also how these drive future land use/cover scenarios (Diniz et al., 2015; Kruger and Lakes, 2015; Schmitz et al., 2015; Swann et al., 2015).

The near-term prospects do not point to a reduction in the current rate of deforestation in the Amazon basin as well as in its estuary (de Araujo Barbosa et al., 2016a; Fearnside, 2015b; Lu et al., 2016). Studies show that during recent decades, the Amazon estuary has been transformed, becoming one of the main frontiers of development in Brazil, open to an unsustainable model of economic growth. The policies that have supported such land changes were based on a well-established belief that various economic benefits will arise from the clearing of forest land for industrial agriculture and/or other exploitative activities that can possibly generate income and economic growth, and consequently improve human wellbeing (Arrow et al., 2012; Dasgupta, 2010; Richards, 2015). In practice, these benefits have been quickly reversed as cleared land has been exhausted by intensive agricultural land use and expansion of industrial-scale agricultural systems (Casanovas Mdel et al., 2013; Ceddia et al., 2013; Pretty, 2013; Schmelzer, 2015). There are indeed benefits resulting from targeted policies and coordinated actions to halt deforestation (de Araujo Barbosa et al., 2016b; Nepstad et al., 2014). Nevertheless, prosperous but volatile commodity prices and currency exchange rates continue to influence the business of forest clearing and land allocation (Barretto et al., 2013; Byerlee et al., 2014; Oliveira, 2016; Richards et al., 2012).

The creation of new protected areas has stalled with existing ecological reserves having their official status removed, and government expenditure on enforcing environmental law cut drastically (Fearnside, 2015a). Recent political appointments followed by an unstable political context in Brazil are sending a message of anti-commitment to environmental sustainability. Furthermore, large infrastructural investment in roads cutting across the Amazonian forests progress firmly (Anwar and Stein, 2015; Assuncao et al., 2015; Bicknell et al., 2015; Fearnside, 2015c; InfoAmazônia, 2015; Perz et al., 2015; Rosa et al., 2015). Increasing political turmoil has destabilized the institutions in charge of regulating natural resource exploitation and the protection of natural resources (Tollefson, 2015a, b). What we see now is the threat of political instability and economic uncertainty, combined with unstable global markets (Arestis et al., 2016; Ribeiro et al., 2016).

We understand that the processes taking place in the Amazon estuary reflect complex and close relationships, with feedbacks and loops steering the system to its current state (Acevedo et al., 2008; Rammer and Seidl, 2015; Rounsevell et al., 2014). In a globalised and interconnected landscape such as the Amazon estuary, change is likely to have differential impacts on biophysical, social and economic determinants (Badger and Dirmeyer, 2015; Holdo et al., 2010; Mayer et al., 2014). Such problems require that empirical and theoretical knowledge are used with integrative and dynamic approaches, capable not only of identifying negative and/or positive interactions but also capable of casting light upon alternative pathways (Collins et al., 2011; Uriarte et al., 2010a). Therefore, the task of capturing such complexity and accepting these as arising from multiple dynamic interactions between ecological and social determinants is not trivial and cannot only be defined simply as being positive/negative (An et al., 2014; de Araujo Barbosa et al., 2015c; de Araujo Barbosa et al., 2016e; Uriarte et al., 2010b).

In this paper, our work was to develop and apply a modelling approach to capture a multitude of complex relationships operating at different scales and time lag intervals in the Amazon estuary in northern Brazil. With this in mind we, therefore, created an integrated and carefully parametrised model capable of updating its input variables after each interaction, while also returning a set of projected system states over time. More specifically, we fitted cellular automata (CA) model that would provide us with a practical approach to capture and describe complex spatial phenomena by assimilating different spatial datasets. This modelling approach was well suited to the purposes of this study, being able to incorporate a diverse set of context specific determinants directly or indirectly related to forest cover change, capturing specific regional land/use cover dynamics (Blesh and Wittman, 2015; Lopez et al., 2013; Vogt et al., 2015).

5.1.1 Research objectives and Questions

The overall goal of this research was to model a series of system states over time, by linking land use/cover change, economic and social determinants in the Amazon estuary in Northern Brazil. Therefore, this paper seeks to address the following research questions:

i. What are the general observed changes in social-ecological dynamics, driver-responses and the resulting multivariate dynamics in the estuary?
ii. How would the current landscape evolve under two different scenarios: one that prioritises intensive resource exploitation as has been the case in recent decades; and another that is framed on stricter regulation of resource use and economic development?
iii. What impacts would these scenarios have on the estuary's landscape over time?
iv. Where might incremental change be desirable, and where would a more strategic shift in regulation be needed to achieve a more sustainable scenario?

This chapter combines remote sensing, social, ecological and economic variables using a spatially-explicit dynamic modelling approach to shed light into the empirical links, short- and long- term impacts on future land use/cover trajectories in the estuary? The modelling approach used here is aimed not only at forecasting changes in land use/cover, but more so to offer insights and support to the design of policies, to target areas, to consider alternative pathways and to highlight opportunities to act in reducing deforestation and land degradation in the Amazon estuary.

5.2 Results

5.2.1 Business as usual scenario

After 50 annual time steps the Business as Usual (BAU) scenario produced the spatial patterns represented in Figure 5.1. Here, the land use/cover transition patterns remain the same, with expansion of agricultural land, urban, pasture, developing in detriment of forested land. The Figure 5.1 shows four simulated forward snapshots for 2017, 2035, 2050 and 2065. Spatially, it remains evident that areas of forest land in the state of Para will continue in constant transition, from forest to other concurrent land cover types, continuing to reproduce the same spatial and temporal relationships leading to clearing of forest land in the estuary. In contrast, the land use/cover transitions in the federative state of Amapa progress at a slower rate, with comparatively small scale changes happening over time. The current dynamics point to few previously deforested areas being reclaimed by other major non-forest land use types (e.g. agriculture and pasture land), although the model was able to account for other land cover types to reclaim non-forest areas (previously deforested land).

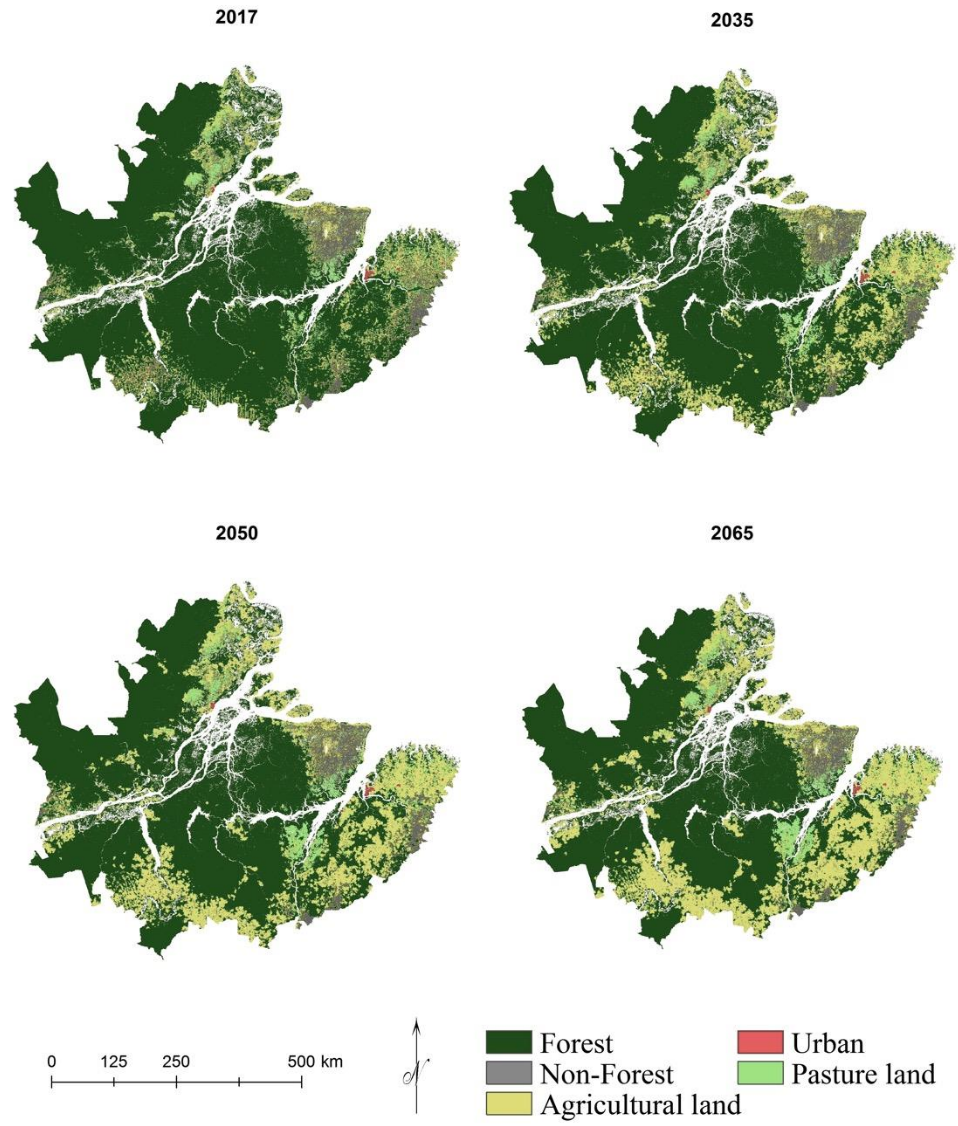


Figure 5.1 - Business-as-usual simulated scenario. Maps are representing snapshots of the continuous times series of simulated data and refer to the years of 2017, 2035, 2015, 2065.

In Figure 5.2 the land use/cover evolution is represented along the entire 50 year period, these trajectories represent the transitions at the estuary level. Again in the BAU scenario the major competing land use types continue developing rapidly over the years with forest land declining constantly and at a faster rate after 2025. It becomes evident that there will be increased competition between pasture and agriculture land over time, with areas designated for cattle ranching overpassing agricultural land after 2049. Urban spread in the region will grow at a fast rate, later stabilising and being reduced after 2036, suggesting that urban areas might become denser instead of expanding.

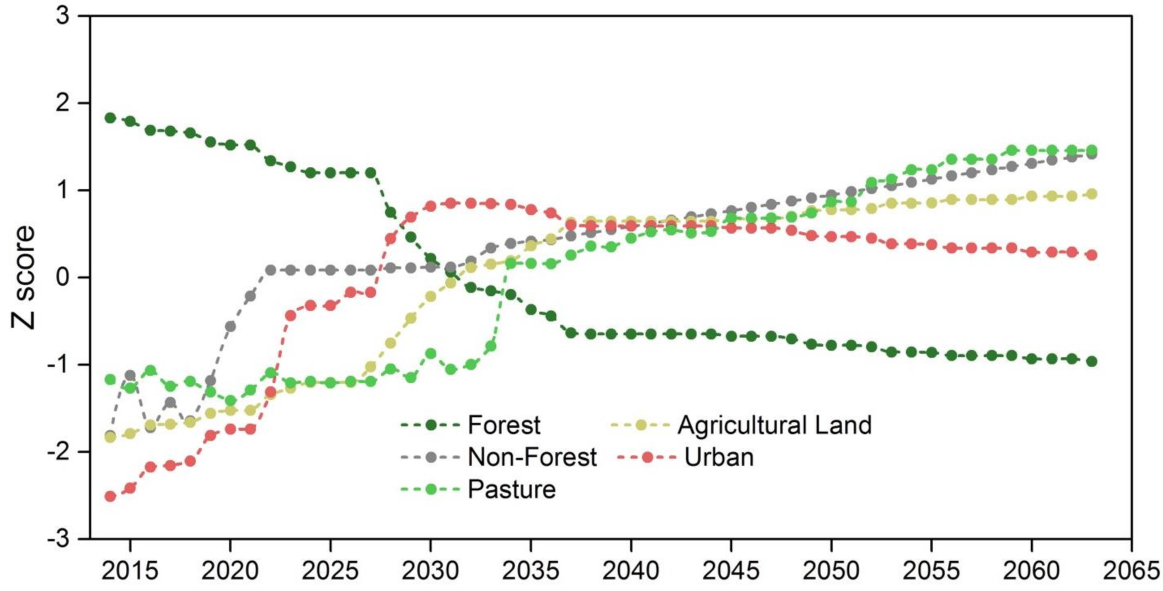


Figure 5.2 – Business as Usual Scenario (BAU) during the period between 2017-2066 for the whole estuary (states of Para and Amapa). The figure represents a forward modelled trajectory for land use/cover types: Forest, Non-forest area, Agricultural Land, Urban/Built-up and Pasture land.

The simulated changes show a non-linear pattern over time with a significant transition period between 2025 and 2035 where predicted forest cover dips considerably, with subsequent stabilisation. This suggests a major shift between two alternate states as a result of changes in key system variables. The decline in forest cover in Figure 5.2 is similar to what has happened between 1997-2004, also not entirely different from recent reports on deforestation increasing year-on-year. This suggests that the model has been capable of capturing and simulating very complex dynamics in the system.

Figure 5.3 represents the predicted changes in forest cover per different categories of GDP, cattle density, population density and crop area categories averaged for all municipalities in the Amazon estuary using the BAU scenario. Here, over time there will be similar patterns of forest cover change, although with some particular trajectories observed for those municipalities with lower GDP, cattle density, population density and crop area. Municipalities with high and medium GDP, crop area, population and cattle density will drive the forest cover change trajectories down sharing a similar pattern throughout the period.

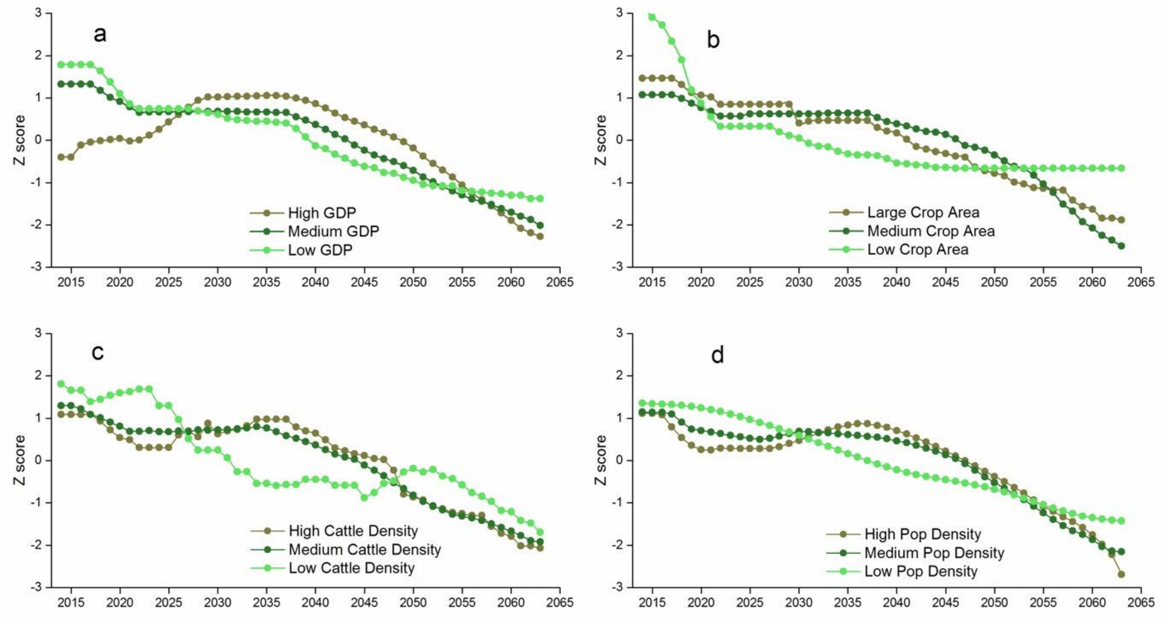


Figure 5.3 – BAU scenarios outputs 2015 to 2060. a) Projected available forest land per GDP, b), Permanent Crop Area, c) Cattle Density and d), Population Density.

At a larger scale, Brazil has achieved good results in reducing deforestation due to a combination of government policy and issuance of land rights over large tracts of forest to indigenous people. However, the BAU scenario reveals the stark reality that waits recent commitments in reducing deforestation (including achieving zero deforestation commitments). Therefore, such scenario ahead clearly has to do with states and municipalities (supported by national and international companies) who have never fully addressed the issue of deforestation that only continues. This should be more than enough to continue repeating a model of land use/cover positioned in line with commodity oriented agribusinesses that expands into new forest areas.

5.2.2 Alternative scenario

The Alternative simulated scenario using yearly time steps (summing up to 50 years simulated land use/cover change) reflects less destructive spatial patterns (Figure 5.4). Here, the land use/cover transition patterns show a slower (more conservative) expansion of agriculture, deforested land, urban and pasture land over the years. After 50 time steps the land use/cover change do not evolve at the expense of forested land. In Figure 5.4 there are four snapshots of the 50 year period set apart by +/- 15 years. In the Alternative scenario, the change from forest to other concurrent land use/cover types is reduced considerably. Consequently, most of the expansion in agriculture and pasture land has occurred in regions of degraded and deforested land (in both the federative states of Para and Amapa).

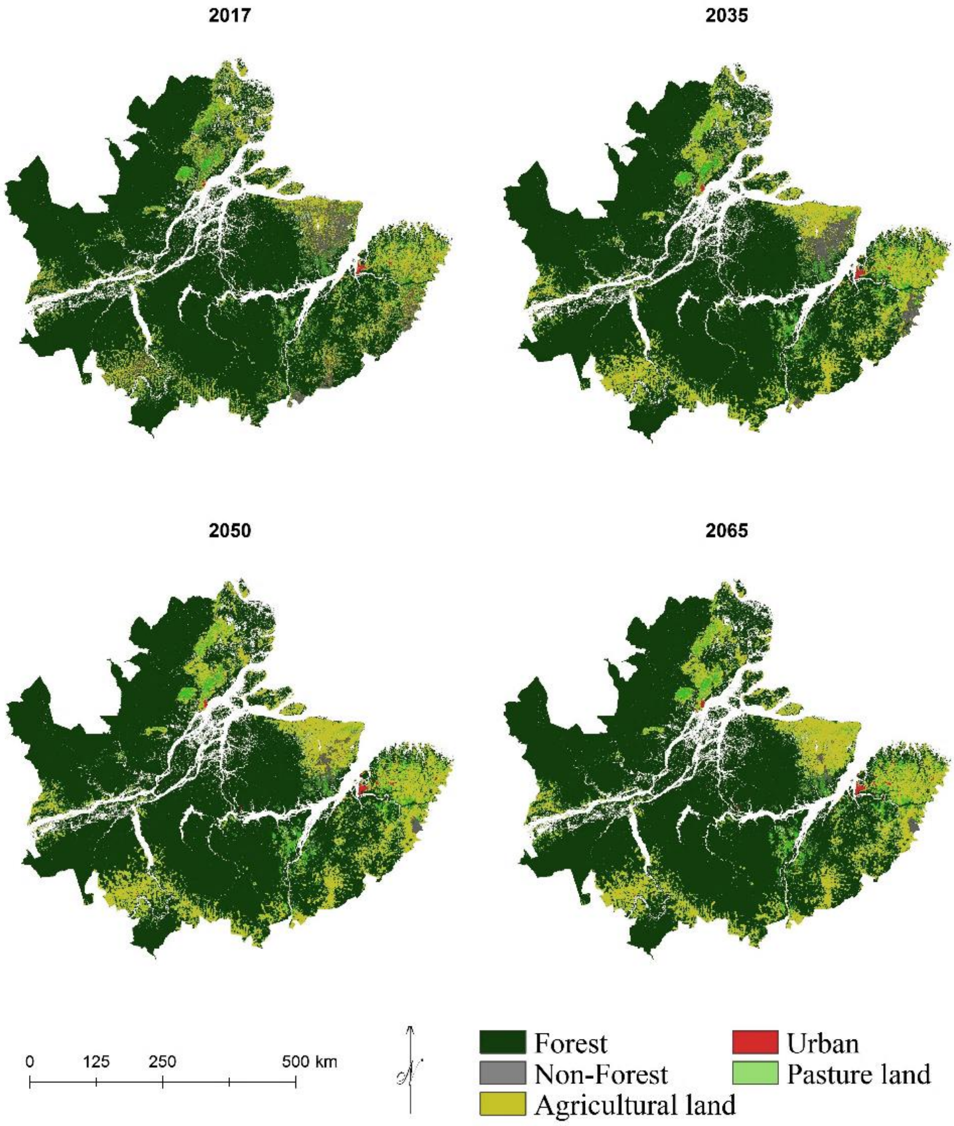


Figure 5.4 - Improved regulation simulated scenario. Maps are snapshots of the continuous times series of simulated data and refer to the years of 2017, 2035, 2015, 2065.

In Figure 5.5, the land use/cover evolution is represented along the entire 50 year period, these trajectories represent changes at the estuary level. In the Alternative scenario the competition between the land use types continues over time, but now allowing for forested land to recover. At first (2016-2035), competition between land uses is predicted to continue, with the observed dynamics in past years only being reversed or stopped after several time steps. Nevertheless, areas designated for agriculture and cattle ranching start to decrease and/or stabilise after 2040. The internal dynamics reveals that crucial system variables drag forest cover down considerably during the period between 2023-2035, with a later stabilisation and recovery.

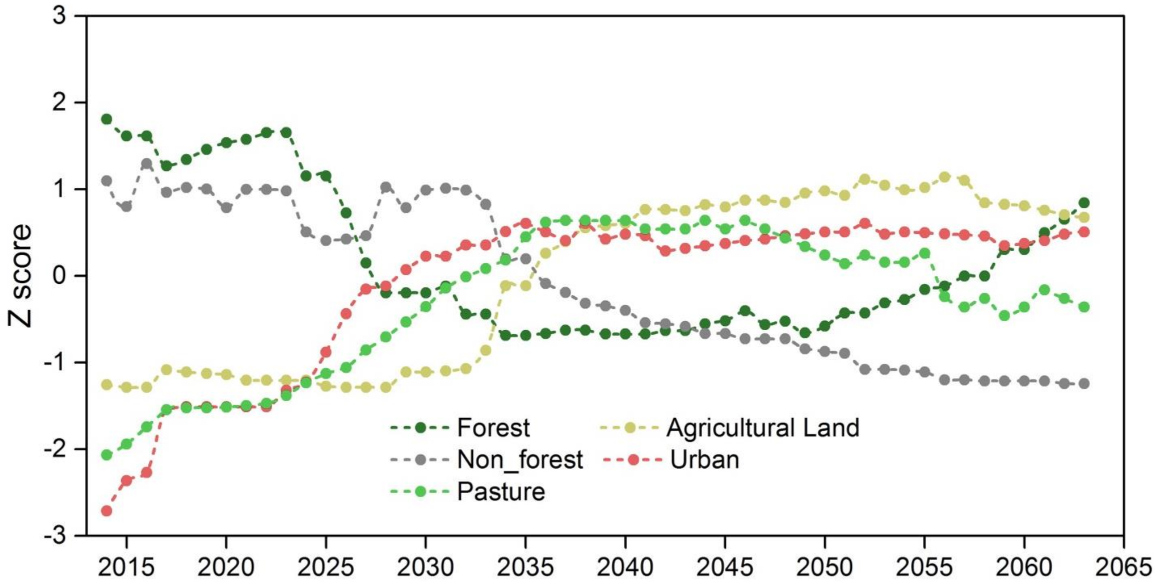


Figure 5.5 **–** Alternative scenario (AS) for the period between 2017-2066 comprising the whole estuary (states of Para and Amapa). The figure represents a forward modelled trajectory for land use/cover types: Forest, Non-forest area, Agricultural Land, Urban/Built-up and Pasture land.

In Figure 5.6 the forest cover observed for the different categories of GDP, cattle density, population density and crop area categories averaged for all municipalities in the Amazon estuary using the Alternative scenario show distinct trajectories. Along the 50 time steps, the patterns of forest cover change for the selected socioeconomic categories evolve in a similar fashion with recovery of forested land expected quickly after 2020. Nevertheless, municipalities with low cattle density show a considerable increase followed by a sharp fall between 2027-2036, and in the case of municipalities with medium and high population density the areas of forest require more time for the negative trend to be reversed.

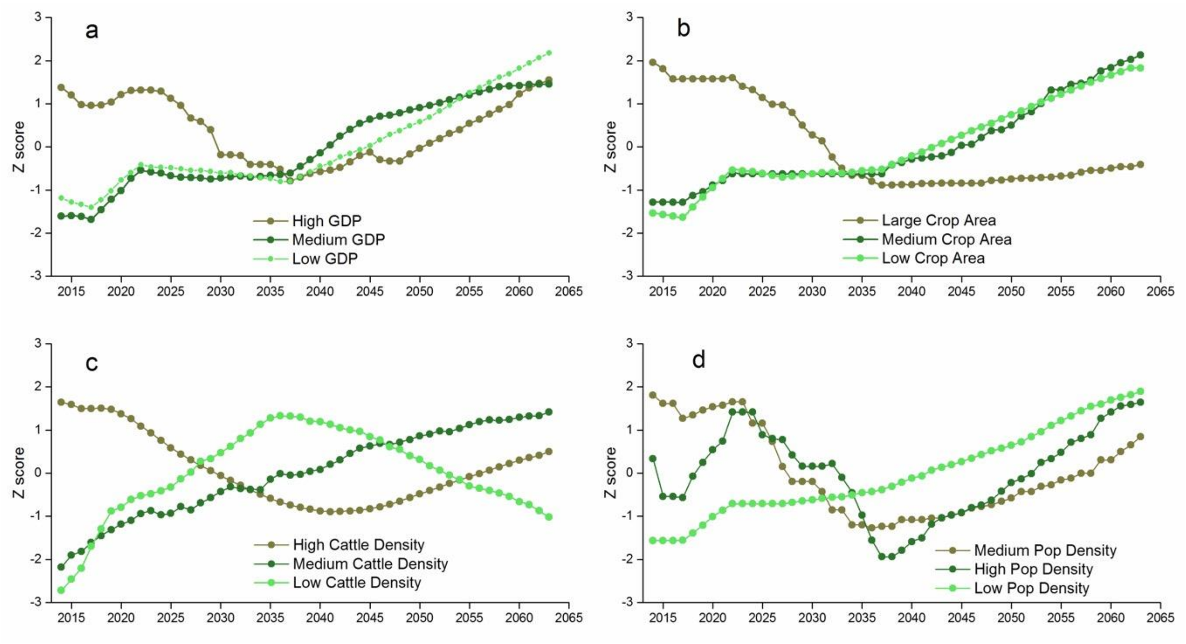


Figure 5.6 – Projected available forest land per GDP (a), Permanent Crop Area(b), Cattle Density (c), Population Density (d) in the amazon estuary (Alternative scenario setup).

What we see though the alternative scenario might be explained by the land use competition that is exacerbated by commodity driven agribusiness continuing to have a great impact on defining the course ahead. Furthermore, we will see that companies committed with better practices in their supply chain (e.g. zero deforestation) will tend to focus their investments into buying land already developed, and on the opposite side the deforestation frontier will move forward helped by new players and companies which historically have shown insufficient concern for sustainability.

5.3 Discussion

5.3.1 Assessing the Impacts

The simulated land use/cover patterns shown through the BAU scenario are likely to continue placing agriculture and pasture land conversion amongst the dominant drivers of forest change in the region. Furthermore, over the years increased integration of Brazil into international markets, several periods of high international agricultural commodity prices, and a weak national currency, have favoured rapid expansion of pasture and agricultural land in the estuary and across the Amazon basin (IBGE, 2013b; IPEA, 2013; Nepstad et al., 2014; Richards et al., 2012). Interestingly, the BAU scenario also reflects the fact that returns derived from production per hectare of crop land farmed have always been more favourable than derived from cattle ranching (Cohn et al., 2016; Latawiec et al., 2014; Richards et al., 2012). The land use/cover simulated results suggest that the share of cropland and cattle pastures in the Amazon estuary will continue to increase, probably as it adapts to meet various demands (some or most of it generated outside its borders) (Dumortier et al., 2012; Nepstad et al., 2014; Ruviaro et al., 2014).

What has been captured on the BAU simulated results shows that agricultural expansion will likely continue into forested and deforested land, following the current logic. Nevertheless, this means business, and it aligns with Brazil's interests in adding more and more to its slowing down GDP growth (which has been declining at 5.4% year-on-year) (de Paula et al., 2015). Such dynamics will have major impacts on the landscape over time, which does not necessarily ensure economic development, as large-scale crop farming and cattle ranching in the estuary might result in more environmental degradation, social and economic instability (Cardoso et al., 2015; Chen et al., 2015; Coleman and Feler, 2015; Verdu and Ferraz, 2015).

The conversion of forests to other purposes has been associated with negative environmental externalities at multiple scales (Aguiar et al., 2016b; Barlow et al., 2016; Brondizio et al., 2009). If the observed trajectories persist, with agriculture and pasture expansion, mining, forest fires, logging and infrastructure development facilitating access to new areas for deforestation, a range of important ecosystem services and functions can be damaged irreversibly (da Conceicao et al., 2015; Maciel and Santos, 2010; Rammer and Seidl, 2015). Moreover, there will be increased risk of reaching critical thresholds for the functioning of the ecological setup needed to regulate the frequency of storms, droughts and other extreme weather events (IPCC, 2007; Mann et al., 2015). Thus, the emphasis on the sustainable use and conservation of forest resources in the estuary will continue being increasingly important, not only for meeting increasing local and global demands for forest products, food and export commodities, but also for providing ecosystem services and sustaining local livelihoods in the future (Dias et al., 2016; Lewis et al., 2015).

Both BAU and Alternative scenarios show that while socioeconomic status and mediating factors (i.e., education, health, employment, integration to markets) play important roles in establishing certain relationships with the environment. However, additional research is still needed to quantify the trade-offs resulting from deforestation and to understand how patterns are modified in response to changes in different socioeconomic factors across local and regional scales. Furthermore, continued efforts are needed to link the modelling of changes in land use/cover in the Amazon estuary to policy, as well as to specific political and/or cultural contexts. Regional and local markets in the region will continue responding quickly to changes in international markets, leading to rapid expansions (or contractions) in certain land use/cover types. This reflects the faster and more complex externally driven changes, with the actors profiting from globalization continuing as major driving forces of land use/cover conversion in the estuary.

5.3.2 The dynamics of land-use/cover change (crossroads of sustainability vs growth)

As progress is made in understanding the dynamics of land use/cover change, arguments that view social and economic pressure as one of the major factors driving clearing of forested land continue to gain support (Mendonca et al., 2012; Nepstad et al., 2013; Verburg et al., 2002; Vogt et al., 2015). Given the complex web of drivers involved in driving land use/cover change and consequently deforestation in the Amazon estuary the results show that not only one but several inter-connected social and economic factors combined are the major drivers of deforestation in the region.

For example, projected available forest land per class of GDP shows that regions will switch into a more competitive mode. The simulated BAU scenario shows that municipalities will increasingly allocate more land to other uses rather than maintain forested land. Therefore, there will be higher contributions to GDP coming from the region, potentially incentivising more infrastructure development (e.g. roads) and connectivity to markets inside and outside the country. The prospect of a growing GDP in the region, for all its prominence, is no indicator of social progress and it takes no account of how income is shared, or of how it is generated. The Alternative scenario shows that a significant and steady increase will be experienced by municipalities in the medium and low GDP brackets (starting as soon as 2018), while high GDP areas will continue the downwards trajectory until 2036. This scenario confirms that GDP will continue playing a vital role in shaping the state of forest resources in the future. Municipalities with medium and large areas of permanent croplands are projected to see a steady decline in forest area, whereas areas with a smaller area of permanent cropland will follow a similar trajectory towards less forest, but stabilising later on (after 20140). The BAU scenario shows that the negative interactions will continue, with the possibility of being exacerbated by inefficient and hazardous economic and social policies in Brazil.

The BAU scenario shows a pattern crafted over several decades of an economic model highly dependent on exports of agricultural commodities (De Castro et al., 2012; Guidolin and La Ferrara, 2010; Povarova, 2016; Rossi, 2012; Voituriez, 2001). The Alternative scenario reveals that the municipalities with large cropland area will continue removing forested land, but stabilising after 2036. In contrast, although not surprisingly, areas with low-to-medium cropland area will follow a less destructive pathway after 2017. This indicates that some municipalities in the estuary will be able to use forests as part of the production system rather than see them as an obstacle.

The municipalities with lower densities of cattle will be responsible for the highest rates of forest clearing over the next few decades. This might be due to such areas already being highly integrated and specialised into other activities (e.g. agriculture, services,). By becoming more specialized, the allocation of land and resources becomes more and more efficient, encouraging over-specialization of activities. The Alternative scenario shows municipalities with medium cattle density on a steady and continuous path of forest land increase, while municipalities with low and high cattle densities move in the opposite direction throughout the period. This might be due to the fact that municipalities with high and low cattle densities will evolve shifting forest-to-pasture conversion between them, or that large-scale cattle ranching will expand continuously to nearby pristine territories as international demand for beef increases.

Areas with higher and medium population density will produce a downwards trajectory with a slight improvement after 2030 until 2037, where areas with lower population density have forest area continuously decreasing. This suggests that population pressure alone is insufficient to explain the elevated rates of forest cover change, and that the means and demand for deforestation are generated elsewhere. Furthermore, the wealth generated by these resource- rich but sparsely populated regions will never be effectively captured and spread, geographically and socially inside their borders. The Alternative scenario shows comparable results, as the municipalities with low population density exhibit a positive trajectory in forest cover, with an early improvement but a later decrease in forestland (shifting towards a positive trend only after 2037). Not surprisingly, these findings reveal a complex arrangement of relationships reflecting regional and cultural differences, as well as specific local socioeconomic conditions.

In both scenarios (BAU and Alternative) the model shows that forest cover loss will continue in the estuary. This merely confirms what we have seen over the past decades; with periods when deforestation is reduced due to changes in law enforcement, policy and international markets, followed by periods of increased deforestation triggered by the same combination of factors that once led to reduction in deforestation. At present, Brazil's currency exchange rate against the US dollar is highly favourable to an increase in the exports of several commodities produced using deforested land, such that it is increasingly profitable to continue the clearing of forest land (de Araujo Barbosa et al., 2016b; Godar et al., 2012; Latawiec et al., 2014; Nepstad et al., 2014; Perz et al., 2009; Vogt et al., 2015). This shows that land allocation in this region is highly dependent on the current order of increased interaction and integration between the private and public sector, that is also driven by international trade, cross-border investment and the rapid pace at which information is disseminated. This gives powerful insight into how the social-ecological dynamics, driver-responses and multivariate dynamics and multiple trade-offs will evolve. The results show that Alternative scenarios require the intensification of current agricultural land, mixed farming systems combined with better regulatory power, to produce a more positive horizon of opportunities for society and nature in the estuary (Brondizio et al., 2009; Fortini et al., 2015; Medina et al., 2009).

Moving towards alternative and more sustainable pathways seems to be the logical approach to deal with the set of complex social and ecological challenges in the region, as discussed above. Furthermore, only then we might be able to ensure that multiple sectors of society can realise (sooner rather than later) the importance and benefits of maintaining scarce and finite ecological resources. In the future, the need for public policies and private decisions that steer global production and consumption systems towards a more sustainable economy path will be key for enabling socioecological systems in the estuary to continue. Moving towards this path will be pressured by the rising economic and environmental costs of changing the natural ecological balance at multiple scales.

5.4 Methods

Data: the data used in this paper comprises a large number of distinct variables that have been observed to influence land use/cover dynamics in the estuary through time (Bustamante et al., 2014; de Araujo Barbosa et al., 2016b; de Araujo Barbosa et al., 2016e; Gardner et al., 2010). This is not an exhaustive list of relevant variables, although it is a large set that adds specific context and helps determine the observed changes in land use/cover, also connecting regional and global drivers to the dynamics at the estuary level (de Sherbinin et al., 2008; McCauley et al., 2015). To define different periods of land use/cover in the estuary we used data from the Moderate Resolution Imaging Spectroradiometer (MODIS) land cover type product (MCD12Q1 and MCD12Q2) (Abercrombie and Friedl, 2016; Zeng et al., 2015) and a combination of several social and economic variables at the municipality level (Appendix D, Table D.1). The selected years were those representing periods of higher (2002-2005) and lower average rates of deforestation (2009 - 2011). This allowed definition of a series of changing parameters defining different periods of dynamic change in land use/cover.

Modelling: We opted for a modelling approach capable of capturing complex spatial phenomena by assimilating different spatial datasets, such as in a cellular automata (Cemin and Ducati, 2015; Perz et al., 2009; Salonen et al., 2014; Sohl and Claggett, 2013; Troupin and Carmel, 2016). Several studies exist on complex spatial phenomena using CA, for example, to simulate the growth of agricultural land in Northeast Thailand (Walsh et al., 2006), to simulate the actions of agents responsible for forest clearing in Venezuela (with different projected scenarios) (Moreno et al., 2007), in measuring and modelling land cover dynamics in the Philippines and Malaysia (Beltran and David, 2014; Chen et al., 2011; Verburg et al., 2002), and to simulate deforestation processes in Southern Cameroon (Bonne et al., 2008; Mertens and Lambin, 2000). The challenge is always to obtain a sufficient set of data to develop a well-parameterized and validated model able to capture the main relationships driving land use /cover change without ignoring the key (often unclear) drivers. Therefore, the approach to modelling such systems needs to capture relationships occurring between a comprehensive set of context-specific variables, and these are likely to affect each other at different time lags (D'Acci, 2008; Irwin and Geoghegan, 2001; Lewis and Plantinga, 2007; Perz et al., 2009). Such a structured and dynamic approach needs to account for the fact that each observed variable can be a linear function of past lags of itself and past lags of other spatial variables (de Araujo Barbosa et al., 2016b; Irwin and Geoghegan, 2001; Perz et al., 2009; Ruiz-Medina, 2012). Only then, our model would be able to capture the underlying spatial and temporal relationships between land use types and their driving forces, finally representing not only current but also future complex dynamics (An et al., 2015; Irvine et al., 2016; Irwin et al., 2009; Kim et al., 2015; Perz et al., 2009) (Further details in Appendix D – SI).

Scenarios: the scenarios explored in this article include qualitative and quantitative components. The qualitative component consists of narratives describing future developments as envisaged by national and international institutions (e.g. population growth, social and economic development). The quantitative components involve time-series of data on commodity production and exports (e.g. beef and forest products such as cellulose, raw wood, paper and rubber), socioeconomic change (health, education and formal employment evolution over time), satellite remote sensing data on previous land use/cover change and economic development (urban development, agricultural expansion characteristics, infrastructure development, GDP) (Brondizio et al., 2009; Cohn et al., 2016; de Araujo Barbosa et al., 2016b; de Araujo Barbosa et al., 2016e; IBGE, 2013a, c; Nepstad et al., 2014; Ribeiro et al., 2016; Tollefson, 2015a) (Further details in Appendix D – SI).

Chapter 6: Reflections and Conclusions

6.1 Introduction

Chapter 2 focused on finding the broad research gap on how integration of remote sensing technologies and ecosystem services concepts and practices can lead to practical benefits for the protection of biodiversity and the promotion of sustainable use of resources. By reviewing various consecutive decades of peer reviewed scientific articles, it was shown that there is increasing interest in, and the need for, concerted application of methods and techniques of remote sensing into ecosystem services research. Chapter 2 also explores, evaluates and synthesizes using a quantitative approach the evidence provided in published peer-reviewed studies framing their work in the context of spatially explicit remote sensing assessment and valuation of ecosystem services. The results in this chapter provide state-of-the-art information on how remotely sensed Earth observation data have been used currently into ecosystem services research. Furthermore, the chapter is a concise summary of what has been done, what can be done and what can be improved upon in the future to integrate remote sensing into ecosystem services research. This is also a valuable contribution to future informed discussions about methodological issues, challenges and solutions and to encourage not only practical but effective application of remote sensing into ecosystem services research.

Chapter 3 analyses the recent and increasing stress on ecosystem services being maximised by a combination of political, economic and social factors. Through a cross methodological approach, this work reveald the political frontiers of forest cover change in the estuary followed by its inevitable consequences to key ecosystem services. Therefore, by using datasets form Earth observation satellites, the ecosystem service literature, and official government statistics this chapter explores and shows the spatially-explicit relationships linking the Green Vegetation Cover (GVC) to the availability of ecosystem services provided by forests in the estuary. Furthermore, after gathering several time-series of data on vegetation cover, governance indicators, currency exchange rates, exports of beef and forest products and enacted environmental laws the challenge was to try to effectively measure the relationships between these variables. This was made possible by using a new innovative approach (e.g. Granger causality) to shed new light into the drivers of land use/cover change, and social and economic instability leading to reduction of key ecosystem services.

Chapter 4 presents the analysis of several time-series, including the Amazon estuary alongside two other major deltas, providing important insights into their social-ecological system dynamics. By using such an approach, this work summarises and explore slow and fast variables, observed drivers of change and recent trends in these regions. The chapter, therefore, focuses on accessing and discussing what are the main positive and negative feedbacks coming from social and ecological factors, their interactions over time, producing knowledge regarding the inherent complexity of connections between human wellbeing, provision of goods, and the maintenance of ecosystem services. Through the results we see that there are various fundamental modifications in several key indicators of ecosystem services, pointing to a changing dynamic state and increased probability of systemic threshold transformations in the near future.

Chapter 5 is an attempt to assimilate a large and comprehensive set of social and economic context variables, aimed at understanding land use/cover transition processes in the Amazon estuary over the last three decades. This chapter develops and applies an integrated modelling approach to capture intricate dynamics in the estuary (updating the system of variables, while returning a set of projected envisaged scenarios). In this chapeter a modelling approach is used which is able to incorporate a distinct and carefully selected set of variables that relate directly to forest cover change. The results show that the modelling approach was able to identify and capture specific regional land/use cover dynamics in the estuary, simulating the dynamic competition amongst different land use types under different scenarios. This chapter contributes to our understanding of the processes and rationale behind business-as-usual (BAU) practices in the Amazon estuary, leading to extensive environmental degradation as a consequence of failed measures and targets set for economic growth. In contrast, the alternative scenario shows that alternative pathways to current development and economic growth strategies are challenging but necessary to achieve environmental sustainability in the region. Combined with better regulation and targeted policies, this will need to be highly adaptable such as to remain relevant in tackling future challenges.

6.2 The Amazon estuary: a multivariate and adaptable system

In this research new modelling and statistical techniques played an important role in the development of thinking of the Amazon estuary as a complex and adaptive system. Chapters 3, 4 and 5 shows, through rigorous investigation, the nature of relationships in the estuary, and how close relationships between drivers of change can be established and evolve with time.

This brings us closer to improving key practical aspects related to how long-term relationships, lagged variables, simple rules and the addition of spatial dimension to a modelling exercise will lead to the identification of complex, and adaptive behaviours. In the Amazon estuary, we observe that there are is a large collection of diverse parts interconnected in a hierarchical manner, such that organization will continue to persist and/or grow over time following a set of controls triggered by local and/or global stimuli.

It is observed that through a series of dynamical, continuously unfolding processes, individual units within this system have actively responded to stimuli from not only internal (or local) but also external (or global) circumstances. Exploring such relationships and information sheds light on the empirical relationships arising from a complex web of highly dependent and adaptable components in the Amazon estuary.

Over time it was observed that competition (e.g. land uses, ecosystem services demands, commodity prices, etc..) has been operating to maintain and/or strengthen certain patterns and properties while at the same time constraining or eliminating others (e.g. human wellbeing, forests, wetlands and climate regulation).

In the way it is now organised, the complex socioecological and adaptive systems in the Amazon estuary are naturally (and/or irreversibly) set for constant potential change and adaptation to variations of internal and external rules, connections, and responses, reflected in the continuous change in the immediate environment. When compared to other complex and adaptive systems one sees that the resulting coevolution of key social and ecological determinants is in itself an extremely complicated process, which offers insight into the future sustainability of such regions of the world.

While powerful forces like population growth, resource scarcity, and economic instability have created the need for transformative changes in the political and economic arenas, the ruling ideas still constitute the very foundation on which progress continues towards intensive resource exploitation to achieve maximum profit.

6.3 System components respond to changes in temporal and spatial patterns and processes differentially: the empirical links inside a noisy multivariate system

It can be seen that different system components respond at different time intervals and show distinct spatial patterns as they evolve. In terms of spatial and temporal patterns the Pairwise Granger Causality tests have shown that some of the changes in land use can be explained empirically by changes in exports of beef and forest products, governance and currency exchange rates.

Similar ideas have been explored in the literature before, but such studies have never looked into trying to establish relationships between one variable (social, economic, physical) by using prior values of another variable (social, economic, physical) in the context of rapid socioecological change in the Amazon (nor in the estuary). Furthermore, here it is verified empirically whether some of these relationships are bidirectional, meaning that there might be important feedbacks between lagged variables over time that would not otherwise be identified, therefore, providing new insight into how Brazil might engages in the future with regulation and the management of natural resources in the Amazon estuary.

The methods applied here help show that during the last three decades there were strong relationships linking changes in forest cover to the current state of politics in Brazil, as well as the condition of markets outside its borders. Interestingly (but not surprisingly), once environmental degradation progresses it will lead to adjustments in governance, and once such adaptation occurs the effects will be felt in the region afterwards. This is evidence showing how coordinated action from the Brazilian government can change the deforestation trajectories in the country and in the Amazon estuary. Such findings give further insight into how to evaluate policy and government action, identifying what is likely to and what is not likely to work in reducing deforestation. Such approaches can and should be applied to other case studies elsewhere in world.

Building upon this study, future research can apply and expand this approach to shed light on issues such as: verifying to what extent government and business can benefit from applying new regulation and resource exploitation methods; if resources that are closely monitored or used by external stakeholders or the public are exploited and/or evaluated in a more positive/negative manner over time; how the wellbeing of human populations evolve and is affected by the current *status quo* in economic growth and development; what can and cannot be expected as a consequence of average conditions observed in the economy, society and nature; are there different (or more influential) common causes and underlying driving forces of deforestation in different regions of the world?

One of the main challenges during this research relates to approaching the various factors contributing to environmental degradation accounting for the fact that they exist inside a multivariate system of relationships that behave differently and at different time intervals. For example, varying price signals across national and international borders, policy, land use change, physical drivers, socioeconomic factors and production, have very distinct ways of responding to change, and yet the interaction between them is key to understanding the system and its behaviour. It is, therefore, challenging to identify the pattern and nature of relationships (what drives what, and how?), making it difficult to accurately capture the essence of the system's behaviour. That is to say, how to avoid neglecting important relationships; avoid spurious associations; discern between relationships that are strong "in the short run", from strong causal relationships in the long run? Overcoming these challenges might be the turning point between prevention and remediation, between informed decision-making and guesswork.

On a more positive note, this research also shows that after several decades of environmental degradation, Brazil has shown that it has the ability to work towards better protection and regulation on the use of its forests through improved governance and law enforcement.

6.4 Evolution of socioecological systems in the Amazon estuary

The socioecological systems in the Amazon estuary are inherently linked to the chain of events that takes place at the basin level, naturally absorbing some of the positive/negative feedbacks arising as a consequence of multiple natural and human induced processes. This research has shown that the major mechanisms of economic growth and development have been continuously modifying the landscape in the estuary, as a series of processes starting after 1950 and continuing until now. The results in

Chapters 3, 4 and 5 show that this has happens as a consequence of conflicting economic and strategic interests taking place locally and elsewhere.

In comparison with other major deltaic systems (such as the Mekong and the Ganges Brahmaputra-Meghna) we see that the Amazon estuary is in a different stage in its progress, and has started its development relatively late (Appendix B). The main cities in the estuary started expanding and growing in population after the 1970s (due to massive private and government investments in the region). More precisely, this happens after the implementation of the Plan for National Integration, where the main objective was to develop the country's far-flung territories by creating new opportunities for the development of new production units, cities and consumption markets.

It is evident that there are long- and short-term effects that arise from the relationships between society and the natural environment. These are triggered by subsequent interactions between effective/ineffective policy, social and economic transformation, changes in internal and external markets and political instability. This situation explains the rapid pace of forest cover clearing and ecosystem services depletion in the estuary (with particular focus on regions in the Southeast and Northeast of the estuary, with vast areas of cleared forests and agricultural expansion).

The areas where deforestation has been taking place more rapidly are also those where the biogeophysical units and associated social actors and institutions are better coordinated towards economic growth. This is evident though observing that most of the forest change has occurred in areas close to urban centres and with well-developed road infrastructure, which are able to effectively connect the region to markets. After several years, such a configuration has produced widespread environmental degradation, leading to a less likely environmentally sustainable future.

Over the years, we observed that intrinsically, the various resource units in the Amazon estuary are constantly interacting with other broader resource systems that are characterized by particular ecosystem types and biophysical processes, at different geographic scales. Interactions among these units and dimensions are mediated by the broader social, economic, and political settings and related ecosystems within which the region's socioecological systems are embedded. It is possible to conclude that the availability of ecosystem services provided by forests in the estuary has been considerably affected by the current dynamics arising from increasingly negative interactions between society, economy and nature. In the future, several key ecosystem services provided by forests in the estuary, which are key also for sustaining individuals locally and globally (e.g. timber production, air quality regulation, primary production, biodiversity and nutrient cycling), are going to become increasingly scarce.

6.5 Ecological, social and economic Interactions: old and new dilemmas in prospect

The estuary is representative of a landscape that is intrinsically multifunctional through its simultaneous support of habitat, productivity, regulatory, social, and economic functions. Over the years, heterogeneity has been the main force supporting all the interactions and interdependencies in the landscape, and this heterogeneity is fundamental for this landscape to support various, sometimes contradictory, functions simultaneously.

As progress is made in understanding the drivers and dynamics of land-use/cover change (LUCC), arguments that view social and economic pressure as one of the major factors driving clearing of forested land continue to gain support. Given the complex web of drivers involved in land use/cover changes in the Amazon estuary, this research confirms that not only one, but several social and economic factors combined will continue contributing to land use/cover change and deforestation.

This research has explored how many elements (originating though social, economic and biogeophysical interactions) in the Amazon estuary region have a multifunctional character. The estuary controls various fluxes of products, people, energy and materials though its territory, serving as a protective environment for biodiversity, food production, regulation of biological and physical processes, and providing recreational opportunities for people. Through time, we see that land use has been the key determinant in establishing the performance of the estuary with respect to socio-economic functions such as land-based production, infrastructure and housing. Therefore, the degree of integration between these social, economic and environmental functions (including natural resources protection) will increasingly become dependent on the patterns and intensities linked to transitions in land use.

The various complex interactions taking place in the Amazon estuary seem to repeat a very well established model of development that is increasingly dependent on consecutive changes in technology, trade, increased productivity, and increasing exploitation of natural resources. Such an attitude towards a seemingly highly connected and sensitive system will make it difficult to safeguard the balance that allows ecological systems to thrive, threatening the provision of key ecological benefits to human societies at the local, regional and global levels (de Freitas et al., 2007; Caviglia-Harris, 2016; Deutsch et al., 2003; Jonas et al., 2014).

Recent history has shown that the Brazilian government has yet to produce and present long-term strategies and action plans to address the widespread environmental changes in the Amazon estuary. Therefore, broad trends in the economy, demographics and social change will increasingly have major impacts on changing the landscape in the estuary, reflecting faster and more complex externally-driven changes. Furthermore, the observed negative developments are likely to continue and be worsened by ineffective and harmful policies, keeping the economy locked in strict dependency on commodity production and exports.

The Amazon estuary supports distinct landscape types where human influence plays an important role. Such landscapes are characterised by high dynamism, complexity and multifunctionality. Over time, we observe that the main interests surrounding economic, political and societal transformation is for the landscape to support continuous production. The trajectories in land use/cover show that high intensity agricultural and forestry production has been aimed at homogenising the heterogeneous landscape features to facilitate production, fluxes and exploitation of resources. This processes often comes coupled with environmental degradation including soil erosion, nutrient losses, groundwater pollution, and decreases in biodiversity and landscape scenic values.

6.6 Conclusions

This thesis, represented in four main chapters, bridges knowledge from physical and social sciences, connecting directly with a globally relevant and interdisciplinary research agenda. This research uses methods and techniques from social, mathematical and geographical sciences to address emerging issues affecting society and nature. The following is a generalised summary with some of the main contributions to the progress of scientific knowledge:

i. Systematic review of the state-of-the-art in integrated ecosystem services and remote sensing research, identifying gaps, methodological challenges, issues and pathways for improved research and practice;
ii. Identification of the major relationships driving forest cover change in the Amazon estuary over the last three decades, revealing that policy, markets, nature and society change and respond to stimuli differently and at different time intervals;
iii. Inter-comparison including the Amazon estuary case to provide perspective showing how it compares to other equally large and globally important estuarine systems, revealing the nature of environmental changes throughout the past decades, and the risk to systemic change and threshold transgression;
iv. Use of the knowledge produced to establish a forecasting framework integrating a comprehensive set of social, economic, and ecological variables to provide insight into the current and alternative pathways for the future coevolution of socioecological systems in the Amazon estuary.

Hopefully in the future, Brazil's economy, society and the various institutions responsible for promoting better environmental practices, will start and continue to pay increased attention to the estuary's natural capital and other non-productivity issues, to decouple economic growth from environmental degradation. Now, more than ever, local and global actors and institutions realise that during the last several decades the well stablish on-size-fits-all economic growth and development models have systematically exploited natural resources witout necessarily improving the safeguards for human well-being.

Very soon, public policies and private decisions that steer global production and consumption systems towards a more sustainable economic path will play a key role in determining the future sustainability of socioecological systems. Transitioning towards this rationale should be a priority, given the rising economic and environmental costs that come as a consequence of changing the natural ecological balance.

This thesis is a contribution to a more detailed understanding of the workings of the Amazon estuary's landscape, socioecological systems, with commensurate thresholds and strengths, and further research is required to provide greater understanding to support sustainable development of this unique landscape. This will allow for better socioecological resilience and improved service provision under scenarios of change. Finally, long-term commitment inside the political, social and economic arenas is the only way to try to slow, stop and/or reverse some of the negative interactions between nature, society and the economy. This has to be supported by constant innovation and research such as to build effective policies to deal with the challenges ahead.

Appendix A Municipality, micro and macro region's Mean Fraction of Green Vegetation (%)

Federative state	Region	Municipality	Mean Fraction of Green Vegetation Cover (%)																															
			1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013
AMAPA	AMAPA	Pracuúba	77.36	77.36	78.20	78.80	78.78	78.93	78.86	77.62	76.15	74.52	74.22	74.52	76.41	77.61	78.28	77.80	79.91	80.95	81.11	79.00	78.85	79.85	80.47	81.12	81.01	81.45	81.24	81.35	82.14	82.24	80.98	78.24
	MACAPA	Itaubal	72.00	72.00	72.42	72.53	73.08	73.23	73.53	72.59	72.02	71.26	71.44	71.21	72.79	73.51	74.25	73.51	74.11	74.25	75.48	75.66	76.07	75.75	75.70	75.44	75.60	75.79	75.87	75.81	75.49	75.22	74.08	72.40
		Pedra Branca do Amapari	83.79	83.79	82.76	82.92	84.29	84.84	84.87	84.23	84.22	84.16	83.62	83.01	83.79	85.29	87.46	87.91	88.21	87.18	85.76	84.85	85.38	86.51	86.95	86.89	87.10	86.89	86.39	86.50	87.37	87.59	87.43	86.08
		Porto Grande	71.65	71.65	72.46	74.23	77.11	78.52	78.90	78.57	79.47	79.58	79.92	79.63	81.77	81.40	81.12	79.34	80.01	80.63	81.35	81.34	81.21	80.36	79.30	78.43	78.89	79.98	80.39	80.91	81.54	82.27	81.31	79.08
		Serra do Navio	84.15	84.15	84.17	84.58	85.39	85.71	85.49	85.47	86.24	86.26	85.76	84.40	85.32	86.08	87.85	87.72	88.34	87.87	86.70	86.18	86.50	87.89	87.52	87.45	87.30	87.48	86.80	86.86	87.11	86.74	86.50	85.13
	MAZAGAO	Laranjal do Jari	82.77	82.77	82.75	83.35	84.23	85.30	85.26	85.20	85.78	85.30	84.72	83.48	85.16	85.99	86.98	86.63	86.91	86.25	85.23	84.83	85.48	86.45	86.04	85.50	85.08	85.13	85.03	85.53	86.72	86.98	86.83	85.28
		Mazagão	78.34	78.34	78.93	78.94	79.32	79.27	79.46	79.92	80.95	80.73	79.92	78.92	80.77	81.69	82.51	81.40	81.29	80.37	80.27	79.92	80.66	81.26	80.97	81.03	80.73	81.55	81.35	81.66	82.08	82.49	81.86	79.81
		Vitória do Jari	79.92	79.92	81.00	81.66	82.34	82.67	82.47	81.77	81.76	80.86	80.73	80.41	82.87	82.99	83.36	82.61	83.64	84.02	83.77	83.31	83.54	83.99	84.14	84.03	83.91	84.03	84.20	84.59	85.68	85.48	84.89	82.61
PARA	ALMEIRIM	Almeirim	82.69	82.69	82.83	83.18	83.47	83.44	83.33	83.35	83.43	83.18	82.74	82.49	82.69	83.15	83.57	83.85	83.84	83.79	83.69	83.64	83.53	83.36	83.32	83.42	83.53	83.42	83.07	82.65	82.23	81.84	81.53	81.42
		Porto de Moz	83.45	83.45	83.45	83.35	82.93	82.37	82.16	82.48	82.86	82.67	82.00	81.25	80.92	81.11	81.62	82.15	82.25	82.01	81.56	81.27	81.36	81.75	82.30	82.65	82.52	81.92	81.33	81.09	80.75	79.94	79.13	79.06
	ALTAMIRA	Altamira	80.86	80.86	80.94	81.15	81.41	81.46	81.40	81.41	81.44	81.23	80.84	80.69	81.04	81.57	81.98	82.20	82.16	82.18	82.18	82.19	81.99	81.56	81.19	80.96	80.78	80.54	80.28	80.17	80.08	79.78	79.39	79.17
		Anapu	82.93	82.93	82.90	82.90	82.94	82.93	82.99	83.14	83.23	82.85	82.21	81.76	81.81	82.24	82.72	83.04	82.91	82.67	82.54	82.57	82.53	82.37	82.36	82.46	82.44	82.12	81.73	81.54	81.40	81.12	80.75	80.62
		Brasil Novo	70.51	70.51	70.61	70.87	71.21	71.39	71.48	71.54	67.55	67.30	66.94	66.75	66.93	67.25	67.55	67.81	67.88	67.93	67.79	66.61	66.37	66.10	65.92	65.81	65.73	65.59	65.45	65.41	65.26	64.85	64.35	64.09
		Medicilândia	79.37	79.37	79.38	79.54	79.88	80.15	80.27	80.24	80.14	79.85	79.58	79.66	80.20	80.74	80.95	80.97	80.83	80.87	80.95	81.02	80.92	80.61	80.31	80.04	79.80	79.53	79.30	79.25	79.21	79.03	78.75	78.59
		Pacajá	71.07	71.07	71.05	71.10	71.22	71.25	71.26	71.33	71.43	71.34	67.17	67.17	67.51	67.90	68.13	68.27	68.27	68.37	68.43	67.38	67.21	66.90	66.62	66.34	66.10	65.84	65.58	65.37	65.16	64.99	64.85	64.84
		Senador José Porfírio	81.50	81.50	81.52	81.56	81.36	80.97	80.72	80.82	81.04	80.90	80.46	80.03	79.99	80.35	80.85	81.25	81.32	81.21	80.98	80.81	80.75	80.81	81.02	81.27	81.36	81.11	80.73	80.42	80.05	79.58	79.16	79.08
		Vitória do Xingu	80.58	80.58	80.62	80.76	80.98	81.14	81.27	81.33	81.31	81.15	80.98	80.91	81.11	81.41	81.64	81.84	81.93	82.09	82.13	82.00	81.69	81.24	80.95	80.86	80.78	80.55	80.30	80.28	80.23	79.88	79.35	78.98
	ARARI	Muaná	81.28	81.28	81.18	80.93	80.53	80.08	79.77	79.70	79.72	79.45	79.08	78.98	79.39	80.01	80.37	80.40	80.06	79.79	79.70	79.57	79.18	78.58	78.35	78.58	78.94	78.99	78.75	78.57	78.35	77.93	77.43	77.18
	BELEM	Barcarena	46.54	46.54	46.32	45.61	45.47	45.75	45.39	45.26	45.39	45.28	46.04	47.33	48.56	48.81	47.54	46.42	45.84	46.32	47.32	47.07	46.19	45.73	45.79	46.00	45.39	44.88	45.06	45.38	45.76	45.61	45.16	45.10
		Belém	35.36	35.36	35.79	37.19	37.60	36.72	37.45	38.01	37.07	36.48	36.60	37.07	37.87	37.79	37.39	37.31	36.64	36.68	37.22	37.03	36.51	36.37	36.10	35.68	36.81	37.53	36.67	36.16	36.09	36.04	35.78	35.87
		Benevides	46.36	46.36	46.47	46.56	46.26	45.67	45.83	46.40	45.95	45.07	44.73	44.22	43.85	44.54	45.59	45.61	44.76	44.07	44.21	44.74	45.13	45.71	46.01	46.07	45.91	45.22	44.69	44.66	45.16	45.71	45.75	46.02
		Marituba	46.36	46.36	46.47	46.56	46.26	45.67	45.83	46.40	45.95	45.07	44.73	44.22	43.85	44.54	45.59	45.61	44.76	44.07	44.21	44.74	45.13	45.71	46.01	46.07	45.91	45.22	44.69	44.66	45.16	45.71	45.75	46.02
		Santa Bárbara do Pará	59.46	59.46	59.46	59.48	58.94	58.04	58.33	59.13	58.70	57.68	57.79	58.47	59.30	59.93	60.02	59.33	57.87	57.60	58.38	58.55	58.12	58.08	58.34	58.58	58.42	57.65	56.87	56.56	57.07	57.80	57.91	58.08
	BRAGANTINA	Bonito	59.73	59.73	59.53	59.76	59.77	60.50	60.52	61.32	62.46	61.82	61.00	59.13	60.59	61.58	63.29	63.42	63.25	62.89	61.65	61.73	62.14	63.40	63.05	62.60	62.44	62.29	62.80	62.91	63.98	63.60	63.50	62.02
		Capanema	59.02	59.02	58.40	58.93	59.11	60.83	61.25	61.99	61.79	60.64	60.29	59.42	61.00	61.36	62.64	62.30	62.55	62.16	61.80	61.49	61.87	62.31	62.61	62.26	62.38	61.84	62.07	61.91	62.70	62.99	62.83	61.23
		Igarapé-Açu	57.51	57.51	57.25	58.18	58.88	60.02	59.49	59.76	60.49	60.61	60.36	58.95	60.18	60.86	62.05	62.26	62.46	62.16	60.33	60.38	60.85	62.28	61.96	61.75	61.74	61.58	61.72	61.18	61.87	62.42	63.25	62.38
		Nova Timboteua	60.26	60.26	60.11	60.21	60.74	62.03	62.68	63.34	63.11	62.66	62.12	61.53	62.97	62.83	63.38	62.59	63.27	63.37	62.65	62.42	62.63	63.43	63.35	62.88	62.40	61.66	62.11	62.55	63.92	63.64	63.47	61.84
		Peixe-Boi	61.82	61.82	60.27	59.81	60.07	60.85	61.21	62.39	62.92	62.37	61.46	60.64	62.23	62.64	63.48	63.04	63.48	63.16	62.38	61.79	62.33	63.06	63.27	62.95	63.12	62.51	62.93	63.04	64.58	64.89	64.55	62.61
		Primavera	62.69	62.69	61.80	60.93	61.38	62.16	62.55	62.91	62.63	62.53	62.15	61.93	63.86	64.10	64.52	63.59	64.25	64.07	63.27	62.79	63.22	63.89	63.57	63.26	63.62	63.37	63.50	63.06	64.08	64.80	64.88	63.40
		Santa Maria do Pará	44.06	44.06	43.46	43.57	43.43	43.98	43.47	43.61	43.77	43.50	43.37	42.46	43.77	44.17	45.59	45.48	45.25	44.58	44.46	45.14	45.61	45.73	45.47	45.62	45.19	44.80	44.79	45.57	45.91	45.74	45.26	44.25
		Santarém Novo	58.62	58.62	58.04	58.45	59.06	60.44	60.56	61.46	62.17	61.55	60.43	58.86	60.30	61.16	62.52	62.64	62.73	62.67	61.67	61.88	61.94	62.64	62.39	62.26	62.36	61.98	62.27	61.94	62.72	62.70	63.11	61.76
		São Francisco do Pará	61.85	61.85	60.50	60.33	61.25	62.31	62.14	62.48	62.13	62.03	61.52	61.06	62.51	63.26	64.41	64.28	64.38	64.23	63.36	62.76	62.89	63.64	63.92	63.53	63.34	63.08	63.25	63.36	64.01	64.65	64.72	63.55
		Tracuateua	57.47	57.47	56.94	57.55	57.91	59.60	59.25	60.02	60.37	58.78	57.76	55.69	57.93	59.27	61.33	62.07	62.04	61.77	60.62	60.36	60.92	61.57	61.85	61.58	61.69	61.27	61.34	61.58	62.40	62.98	63.02	61.75
	CAMETA	Baião	84.16	84.16	84.20	84.32	84.38	84.29	84.20	84.28	84.41	84.14	83.58	83.15	83.20	83.62	84.11	84.46	84.42	84.30	84.31	84.53	84.69	84.64	84.54	84.47	84.39	84.12	83.75	83.48	83.32	83.19	83.11	83.16
		Cametá	85.65	85.65	85.59	85.58	85.59	85.52	85.48	85.53	85.59	85.30	84.86	84.64	84.88	85.38	85.81	86.06	85.92	85.77	85.79	85.90	85.83	85.43	85.09	84.92	84.94	84.90	84.70	84.44	84.21	84.18	84.27	84.32
		Igarapé-Miri	77.90	77.90	77.90	77.93	77.92	77.75	77.58	77.65	77.86	78.09	78.26	78.44	78.70	78.89	79.02	79.28	79.68	80.14	80.21	79.80	79.26	78.79	78.47	78.16	77.79	77.52	77.33	77.07	76.72	76.39	76.20	76.21
		Limoeiro do Ajuru	82.01	82.01	82.05	82.14	82.18	82.06	81.98	82.09	82.16	81.91	81.57	81.60	82.11	82.68	83.02	83.15	83.00	82.84	82.69	82.56	82.36	82.08	81.97	81.99	81.95	81.63	81.18	80.87	80.68	80.43	80.14	80.07
		Mocajuba	84.08	84.08	84.14	84.29	84.40	84.28	84.11	84.12	84.17	83.90	83.54	83.53	84.04	84.68	85.04	85.12	84.83	84.58	84.53	84.62	84.58	84.32	84.16	84.17	84.23	84.06	83.60	83.06	82.63	82.41	82.35	82.42
		Oeiras do Pará	77.59	77.59	77.69	78.04	78.40	78.44	78.33	78.38	78.55	78.57	78.41	78.29	78.31	78.44	78.72	79.18	79.66	80.00	79.93	79.50	78.99	78.62	78.49	78.53	78.62	78.62	78.43	78.06	77.65	77.40	77.34	77.44
	CASTANHAL	Bujaru	64.15	64.15	63.98	63.89	63.96	63.46	63.30	63.79	63.38	62.77	63.10	63.39	63.71	64.12	64.19	63.91	63.02	62.77	63.59	63.76	63.27	63.29	63.43	63.35	63.07	62.35	61.87	62.03	62.79	63.50	63.43	63.47
		Castanhal	60.80	60.80	60.81	61.00	60.85	60.18	60.25	60.81	60.31	59.02	58.66	58.70	58.84	59.58	60.32	59.87	58.52	58.39	59.35	60.00	59.89	59.72	59.87	60.21	60.27	59.58	58.83	58.66	59.35	60.19	60.46	60.50
		Inhangapi	62.89	62.89	62.73	62.67	62.69	62.28	62.24	62.74	62.48	61.39	61.26	61.75	62.33	62.90	63.01	62.26	61.11	61.32	62.41	62.66	62.11	61.92	61.98	62.06	61.88	61.41	61.10	61.10	61.71	62.36	62.38	62.44
		Santa Isabel do Pará	60.40	60.40	60.43	60.63	60.63	59.92	59.94	60.43	59.97	59.62	59.66	59.21	59.12	59.64	60.15	60.35	59.98	59.74	60.18	60.19	59.48	59.12	59.27	59.58	59.63	59.38	58.79	58.38	59.06	59.87	59.82	59.67
		Santo Antônio do Tauá	53.73	53.73	54.18	54.74	54.18	53.50	53.96	54.36	53.93	52.94	52.55	52.98	53.84	54.33	54.52	54.00	52.59	52.63	53.62	53.93	53.64	53.43	53.61	54.11	54.22	53.56	53.08	52.96	53.18	53.56	53.70	53.79
	FUROS DE BREVES	Anajás	85.66	85.66	85.61	85.54	85.42	85.14	84.89	84.88	84.92	84.49	83.78	83.39	83.70	84.38	85.00	85.27	84.98	84.60	84.48	84.65	84.80	84.77	84.75	84.81	84.92	84.77	84.34	83.95	83.73	83.74	83.81	83.96
		Breves	80.19	80.19	80.20	80.28	80.43	80.51	80.57	80.69	80.70	80.06	79.16	78.75	79.17	79.94	80.48	80.57	79.99	79.39	79.29	79.68	79.99	80.04	80.05	80.06	80.03	79.79	79.43	79.19	79.10	79.11	79.08	79.08
		Curralinho	77.32	77.32	77.39	77.71	78.20	78.51	78.60	78.54	78.45	78.25	78.23	78.72	79.64	80.29	80.23	79.85	79.58	79.85	80.29	80.53	80.32	79.64	79.00	78.63	78.55	78.54	78.41	78.20	77.97	77.82	77.74	77.65
		São Sebastião da Boa Vista	84.18	84.18	84.11	83.94	83.75	83.54	83.35	83.29	83.27	83.22	83.26	83.60	84.28	84.85	84.98	84.91	84.74	84.77	84.88	84.89	84.76	84.52	84.36	84.21	83.93	83.43	82.90	82.59	82.45	82.41	82.38	82.48
	GUAMA	Aurora do Pará	59.20	59.20	59.24	59.42	59.57	59.48	59.62	59.86	59.74	59.39	59.40	59.55	59.96	60.26	60.13	59.84	59.14	59.10	59.84	59.96	59.49	59.31	59.28	59.36	59.19	58.76	58.61	58.65	59.07	59.45	59.36	59.43
		Ipixuna do Pará	59.72	59.72	59.85	60.21	60.28	59.92	60.02	60.29	59.92	59.38	59.66	60.05	60.34	60.45	60.43	60.25	59.43	59.36	60.25	60.57	60.20	60.04	59.86	59.76	59.73	59.33	58.89	58.72	59.40	60.30	60.44	60.43
		Irituia	59.64	59.64	59.72	60.01	60.39	60.48	60.52	60.58	60.28	59.63	59.68	60.17	60.65	60.73	60.44	60.01	59.01	59.00	59.98	60.41	60.21	60.14	60.00	59.75	59.55	59.14	59.06	59.42	59.98	60.40	60.41	60.42
		Mãe do Rio	60.98	60.98	60.79	60.92	61.23	60.93	60.89	61.25	60.89	60.11	60.24	60.63	61.11	61.53	61.47	60.98	60.05	60.09	61.09	61.44	61.06	60.90	60.73	60.48	60.52	60.34	59.87	59.80	60.62	61.40	61.47	61.44
		Ourém	61.33	61.33	61.30	61.55	62.11	62.17	62.12	62.17	61.77	61.38	61.70	62.07	62.55	62.71	62.40	62.16	61.46	61.44	62.31	62.56	62.20	61.99	61.74	61.46	61.31	60.96	60.80	61.13	61.79	62.32	62.41	62.40
		São Domingos do Capim	61.57	61.57	61.54	61.80	62.00	61.47	61.38	61.82	61.39	60.07	59.72	60.04	60.42	61.10	61.62	60.94	59.58	59.68	60.75	61.31	61.25	61.36	61.38	61.35	61.40	60.92	60.30	60.22	60.97	61.78	61.92	61.98
		São Miguel do Guamá	52.58	52.58	52.67	52.92	53.28	53.64	53.90	54.02	54.01	53.74	53.49	53.60	54.25	54.92	55.04	54.75	54.31	54.25	54.55	54.90	54.96	54.72	54.50	54.29	54.03	53.74	53.57	53.60	53.66	53.63	53.45	53.28
	PORTEL	Bagre	76.91	76.91	77.01	77.25	77.62	77.98	78.23	78.35	78.35	78.08	77.83	77.93	78.58	79.25	79.37	79.08	78.64	78.59	78.88	79.23	79.29	79.06	78.83	78.62	78.36	78.07	77.91	77.93	78.00	77.97	77.79	77.62
		Gurupá	78.99	78.99	79.09	79.32	79.48	79.35	79.15	79.15	79.25	79.15	78.88	78.69	78.81	79.11	79.47	79.82	80.04	80.16	80.00	79.69	79.36	79.10	79.03	79.11	79.23	79.15	78.85	78.50	78.08	77.62	77.27	77.17
		Melgaço	81.15	81.15	81.26	81.55	81.70	81.46	81.25	81.35	81.51	81.36	81.01	80.76	80.73	80.92	81.38	82.00	82.46	82.69	82.50	82.06	81.48	80.82	80.42	80.43	80.66	80.68	80.43	80.16	79.82	79.34	78.93	78.78
		Portel	81.22	81.22	81.23	81.31	81.33	81.18	81.02	81.05	81.09	80.75	80.25	80.08	80.51	81.18	81.72	81.97	81.79	81.63	81.66	81.77	81.61	81.15	80.85	80.80	80.80	80.59	80.23	79.97	79.85	79.80	79.68	79.59
	SALGADO	Colares	54.99	54.99	55.14	55.38	55.43	55.44	55.29	55.12	54.82	54.66	54.61	54.63	56.02	56.11	56.65	55.81	55.86	55.88	55.78	56.72	56.86	56.74	55.41	55.56	55.74	56.72	56.41	57.31	57.44	58.22	57.24	55.91
		Curuçá	65.54	65.54	64.52	64.64	65.67	66.63	65.71	65.48	64.54	64.62	64.38	64.79	66.75	66.40	67.58	67.09	68.32	68.26	68.28	68.36	68.58	68.78	67.55	67.18	66.54	66.21	66.17	67.29	68.82	68.66	67.83	66.35
		Magalhães Barata	51.18	51.18	50.38	51.55	52.89	55.64	54.98	55.10	55.57	55.87	55.54	54.02	55.39	55.67	56.79	56.99	57.75	57.66	56.31	55.86	56.31	57.26	57.33	57.26	57.66	57.65	57.68	57.89	58.56	59.35	59.02	57.65
		Maracanã	55.41	55.41	55.33	55.39	55.61	55.77	56.00	56.31	56.44	56.47	56.25	56.26	57.59	57.56	57.62	55.93	55.72	55.47	55.88	56.40	57.00	57.01	56.49	56.25	56.23	56.58	56.61	57.28	57.82	58.31	57.13	55.02
		Marapanim	64.75	64.75	64.73	64.93	65.08	65.34	64.87	65.06	66.41	66.98	66.89	66.06	67.61	66.88	68.04	67.88	69.37	68.94	67.67	67.76	68.63	69.66	69.99	69.31	69.88	69.65	69.86	69.31	69.38	70.11	70.21	69.18
		São Caetano de Odivelas	53.68	53.68	53.46	53.38	53.26	53.28	53.21	53.21	53.22	53.26	53.23	53.19	54.57	54.54	54.74	53.52	53.49	53.54	53.45	54.10	54.43	54.91	54.45	54.16	53.83	53.95	53.91	54.67	55.19	55.90	54.97	53.31
		São João da Ponta	64.94	64.94	64.57	64.20	64.00	64.47	64.65	66.23	66.16	65.83	64.48	62.74	63.23	63.64	65.41	65.80	66.40	66.08	65.59	65.37	65.93	66.56	66.72	66.86	67.29	66.80	67.09	67.65	68.52	69.54	68.29	66.48
		São João de Pirabas	65.45	65.45	65.22	65.68	66.89	67.92	68.46	69.00	69.46	69.46	69.08	68.29	69.42	69.37	70.39	70.03	70.75	70.29	69.61	69.28	69.74	69.91	69.62	69.08	69.40	69.01	69.42	69.60	70.77	71.58	71.46	70.12
		Terra Alta	45.19	45.19	44.97	45.43	46.64	47.67	48.21	48.75	49.21	49.21	48.82	48.04	49.16	49.11	50.13	49.77	50.49	50.03	49.36	49.02	49.48	49.66	49.37	48.82	49.15	48.76	49.17	49.34	50.52	51.33	51.21	49.87
		Vigia	53.21	53.21	53.75	53.83	53.85	53.41	53.72	53.83	53.61	53.25	52.93	53.10	54.52	54.58	54.42	53.09	53.20	53.40	53.49	54.53	55.31	55.99	55.02	54.54	53.84	53.95	53.77	57.75	58.23	57.95	53.78	52.09
	SANTAREM	Monte Alegre	78.77	78.77	78.81	78.72	78.46	78.21	78.10	78.18	78.33	78.45	78.52	78.55	78.79	79.18	79.40	79.49	79.45	79.57	79.64	79.49	79.17	78.74	78.47	78.41	78.24	77.82	77.38	77.23	77.11	76.54	75.79	75.37
		Prainha	80.01	80.01	80.10	80.45	81.01	81.56	82.07	82.41	82.59	82.57	82.60	82.91	83.47	83.69	83.36	83.00	83.05	83.62	84.03	84.04	83.70	83.22	82.64	81.82	80.96	80.58	80.79	81.19	81.30	80.99	80.46	80.10
	TOME-ACU	Acará	86.45	86.45	86.37	86.16	85.94	85.79	85.68	85.62	85.57	85.48	85.41	85.42	85.28	85.05	85.19	85.48	85.60	85.50	85.52	85.64	85.43	85.53	85.79	85.79	85.80	85.58	85.22	84.97	84.84	84.74	84.65	84.66
		Concórdia do Pará	29.97	29.97	29.87	29.97	30.31	30.03	30.04	30.46	29.97	29.35	29.62	30.00	30.45	30.85	30.90	30.58	29.53	29.33	30.29	30.60	30.24	30.19	30.03	29.70	29.42	28.83	28.42	28.66	29.46	30.15	30.18	30.25
		Moju	84.04	84.04	84.03	84.00	83.97	83.96	83.98	84.06	84.13	84.10	84.08	84.12	84.34	84.56	84.60	84.65	84.69	84.83	84.87	84.76	84.58	84.33	84.18	84.13	84.04	83.82	83.53	83.33	83.13	82.86	82.65	82.63
		Tailândia	83.03	83.03	83.02	83.04	83.17	83.40	83.59	83.63	83.59	83.52	83.48	83.40	83.39	83.46	83.48	83.61	83.82	84.18	84.42	84.43	84.31	84.10	83.91	83.71	83.45	83.22	83.11	83.14	83.10	82.89	82.60	82.43
		Tomé-Açu	84.43	84.43	84.44	84.56	84.80	85.00	85.16	85.29	85.36	85.20	84.92	84.75	84.82	85.02	85.20	85.44	85.57	85.68	85.69	85.61	85.42	85.10	84.81	84.58	84.41	84.26	84.18	84.21	84.19	84.06	83.89	83.81
	TUCURUI	Breu Branco	77.40	77.40	77.34	77.22	77.13	77.21	77.42	77.57	77.70	77.88	78.17	78.46	78.87	79.23	79.19	79.00	78.93	79.26	79.56	79.51	79.10	78.34	77.59	77.05	76.72	76.57	76.66	76.91	76.95	76.55	76.05	75.69
		Tucuruí	81.43	81.43	81.45	81.53	81.70	81.93	82.16	82.29	82.36	82.31	82.21	82.15	82.33	82.63	82.77	82.82	82.82	83.00	83.12	83.08	82.82	82.35	81.92	81.64	81.45	81.28	81.24	81.40	81.42	81.03	80.52	80.24

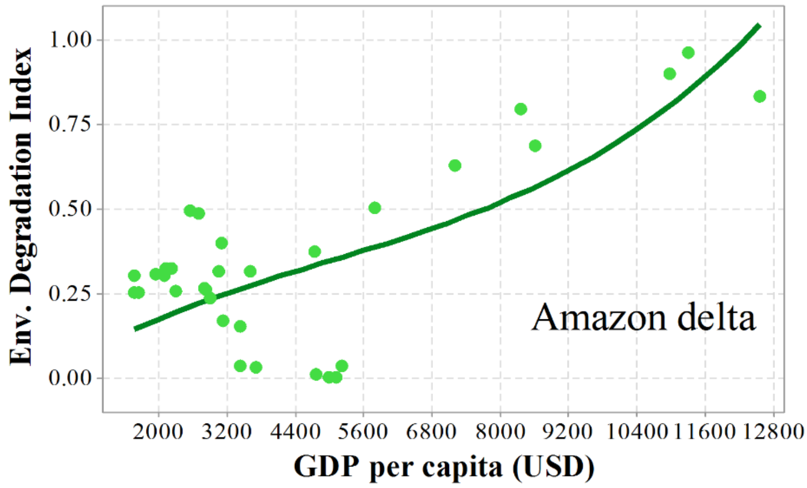
Some municipalities in both states exhibited small decreases throughout the period. For example, the municipalities of Serra do Navio, Pedra Branca do Amapari, Acara, Laranjal do Jari, Cameta, Tome-Açu, Anajas, Baiao, Moju, Mocajuba, Sao Sebastiao da Boa Vista, Tailandia and Almeirim kept per pixel average GVC values between 86% to 83% during the entire period.

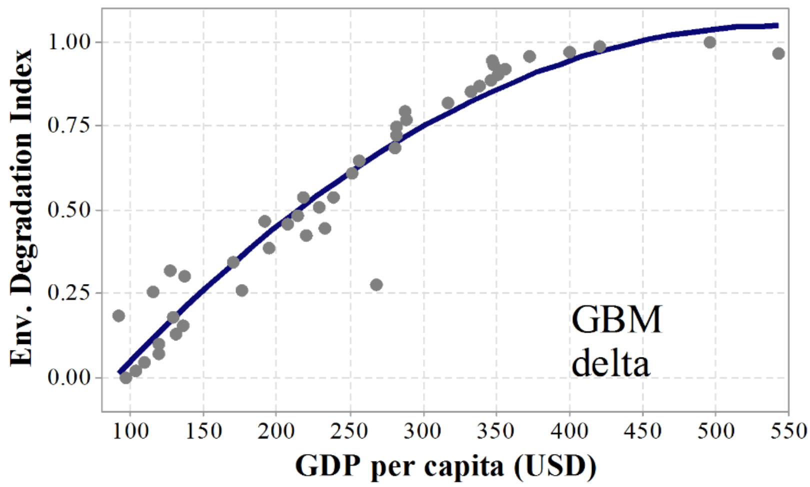
The municipalities of Vitória do Jari, Anapu, Prainha, Tucuruí, Limoeiro do Ajuru, Porto de Moz, Altamira, Melgaço, Vitória do Xingu, Portel, Senador Jose Porfirio, Mazagao, Medicilandia, Breves, Muana, Porto Grande, Gurupa, Pracuúba, Curralinho, Oeiras do Para, Monte Alegre, Bagre, Igarape-Miri and Breu Branco have consistently maintained per pixel average GVC values ranging from 82% to 77%, with minimum per pixel GVC values ranging from 80% to 71%.

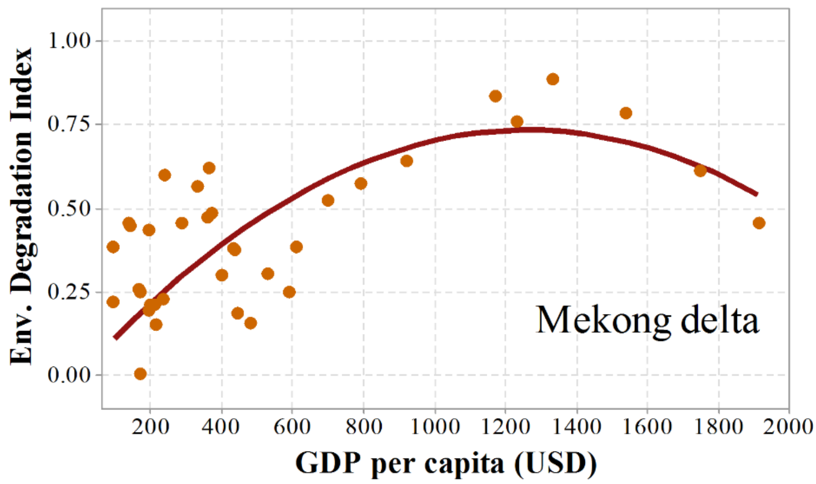
Low per pixel average GVC values were observed in the municipalities of Itaubal, Sao Joao de Pirabas, Pacaja, Marapanim, Brasil Novo, Curuça, Sao Joao da Ponta, Bujaru, Primavera, Sao Francisco do Para, Peixe-Boi, Nova Timboteua, Inhangapi, Ourem, Bonito, Capanema, Santarem Novo, Sao Domingos do Capim, Mae do Rio, Igarape-Açu, Tracuateua, Irituia, Ipixuna do Para, Castanhal, Santa Isabel do Para, Aurora do Para and Santa Barbara do Para, Maracana. Average GVC values for these municipalities were between 73% to 56%, with minimum values ranging from 71% to 54%.

The lowest per pixel GVC values were observed in the municipalities of Magalhaes Barata, Colares, Vigia, Sao Caetano de Odivelas, Sao Miguel do Guama, Santo Antônio do Taua, Terra Alta, Barcarena, Benevides, Marituba, Santa Maria do Para, Belem and Concórdia do Para. Per pixel GVC values for these particular municipalities were as low as 29% over the time period under analysis (ranging from 55% to 29%).

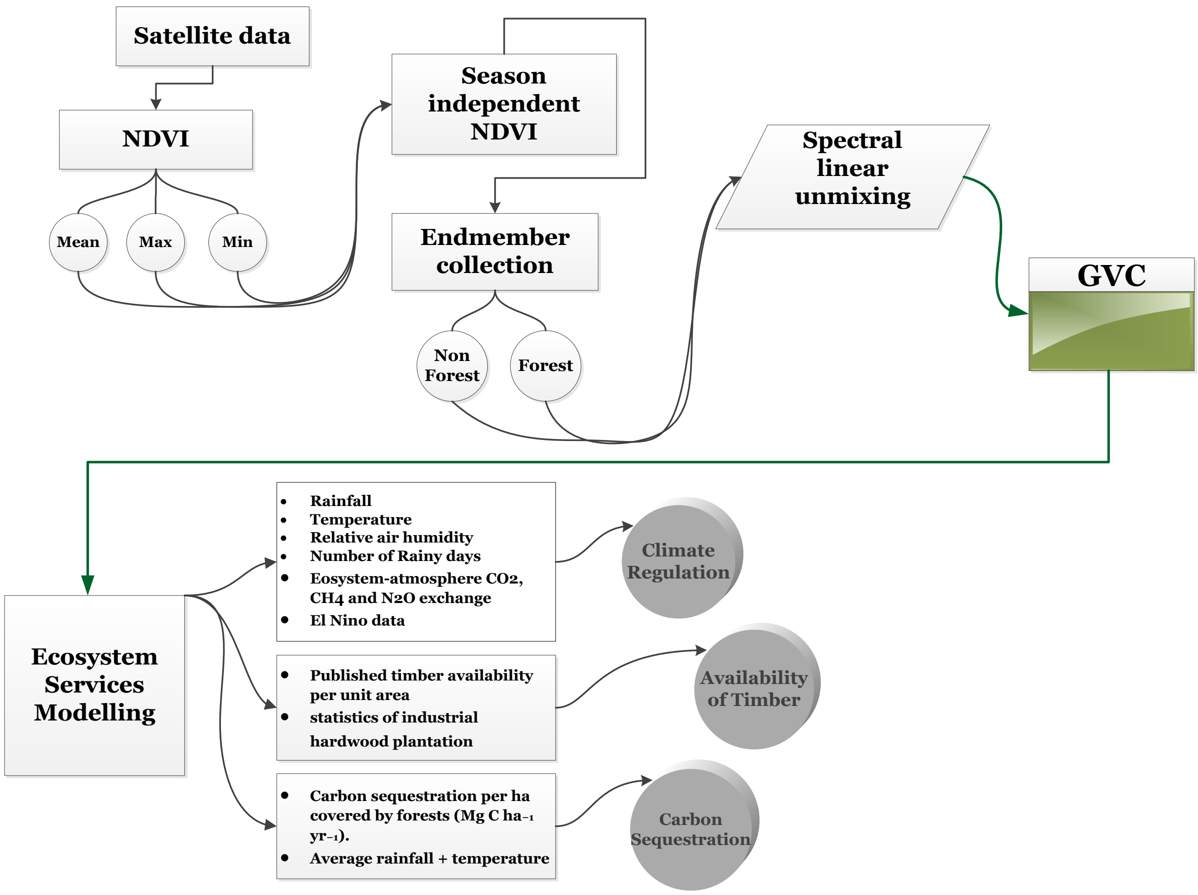
Appendix B Environmental Kuznets curve comparing the state of development in the Amazon, Mekong and Ganges Brahmaputra-Meghna deltas.







Appendix C Simplified schematic diagram representing the workflow process to retrieve the Fraction of Green Vegetation Cover and Ecosystem services estimates.



Appendix D Supplementary Information for Chapter 5

Study area

The study site has an area of 294,000 km^2 and is located between the states of Para (PA) and Amapa (AP) Northern Brazil (Figure D.1). The estuary is part of in the Amazon basin, and corresponds to an area equivalent to 5% of the AM Basin (de Araujo Barbosa et al., 2016b). The region is characterized by extensive areas covered by old growth forests, grasslands, mature floodplain forests and agricultural mosaics. The Amazon estuary extends from the mouths of the Amazon and Para rivers, merging at the eastern side of the Marajo Island (Brondizio et al., 1996). The area that delimits the Amazon estuary for the purposes of this study was based on Ericson et al. (2006a), where a set of Global 30 Arc-Second Elevation grids (1 km x 1 km), soil maps, aerial photographs, satellite sensor images and river bifurcations data were used to define the delta boundaries.

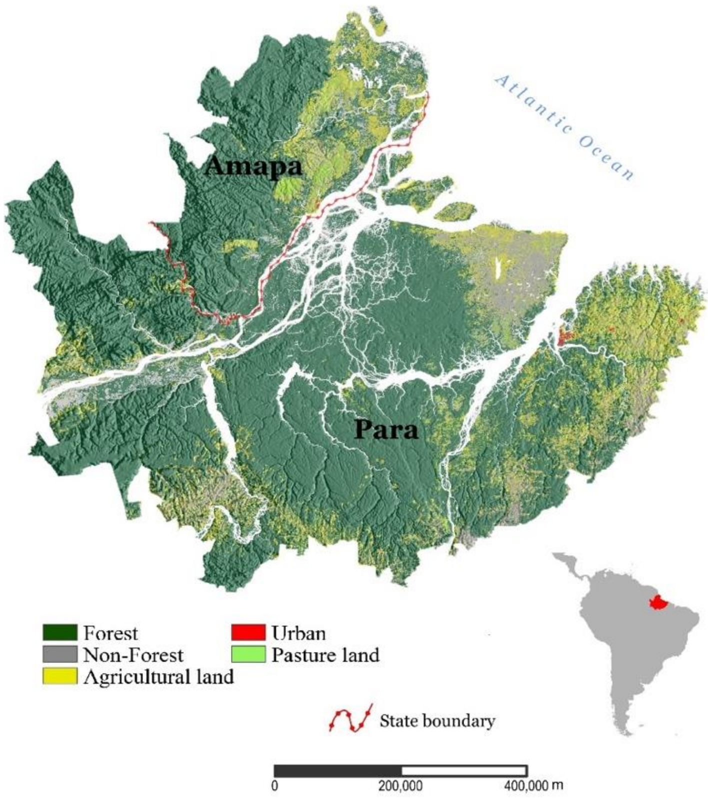


Figure D.1 - Land use/cover map (2015) for the Amazon estuary located in the North of the South American continent (MODIS Land Cover Type product at 500 m spatial resolution). The dash-doted red line represents the boundary between the federative states of Para and Amapa. The Land use/cover map has been overlaid on a shaded relief map produced using Advanced Land Observing Satellite (ALOS) Global Digital Surface Model at 30 m spatial resolution (http://www.eorc.jaxa.jp/ALOS/en/aw3d30/index.htm).

The managers of forests in the estuary have the difficult task of safeguarding the balance that allows ecological systems to thrive and, therefore, provide sustainable benefits to social systems at local, regional and global levels (Anderson et al., 1995; Fortini and Zarin, 2011; Hiraoka, 1995). The political boundaries that subdivide the study area create 17 regions, and 81 municipalities, all of which benefit directly from the resources and ecosystem services produced by the region's forests. We applied a 40 km buffer around the defined estuary area to include the majority of municipalities immediately affected by changes in average conditions.

Overall modelling approach

Our modelling approach had to be able to produce information about the nonlinear behaviour of system states and interactions over time. Such modelling approach was required to provide insight into real-world system behaviour. Therefore, dealing with the temporal evolution of spatial structures and the processes triggering this progression across space and over time. Thus, we opted for a modelling approach that can capture intricate interaction patterns, nonlinear processes and delays, allowing future behaviour to arise as consequence of system structure. Furthermore, allowing modelled natural and anthropogenic processes to modify the arrangement of spatial attributes over time, feeding back into systems behaviour.

Previous scientific work has investigated deforestation, trying to understand the drivers and dynamics of processes of forest conversion into other land uses (e.g. agriculture, pasture) (Lambin et al., 2003; Lambin et al., 2000; Lambin et al., 2001; Verburg and Overmars, 2009), and their short-/long-term effects on the environment (Macedo et al., 2013; Meyfroidt et al., 2014; Soares et al., 2013; Soares et al., 2012; Sohl and Claggett, 2013). Often these scientific studies use dynamic spatial models to simulate key driving processes and also to provide a picture of the changes in the landscape (Soares et al., 2013; Soares et al., 2012; Soares et al., 2006). This has created opportunities and raised questions about how to make such approaches more comprehensive, integrative, and consequently able to inform deforestation control measures in the tropics and around the globe (Azizi et al., 2016; Lambin and Meyfroidt, 2011; Lindquist et al., 2012; Liu et al., 2016; Troupin and Carmel, 2016). The approach taken in this study aims to model and evaluate changes in land use/cover in the Amazon estuary dynamically, arising as a result of the interaction between several context-specific variables over multiple time steps

(Brondizio et al., 1994a; Brondizio et al., 1994b; de Araujo Barbosa et al., 2015a; de Araujo Barbosa et al., 2016b; Padoch et al., 2008; Vogt et al., 2015).

List of Datasets used in the model:

Table D1 - Social, economic and physical spatial data layers.

Variable	Reference year	Measurement unit	Data source
Percentage of the Pop. Employed Formally	2013	% employed by municipality	IBGE/IPEA
Percentage of the Pop. Employed informally	2013	% employed by municipality	IBGE/IPEA
Percentage of the Pop. Employed in Agriculture	2013	% employed by municipality	IBGE/IPEA
Education	2013	% of the pop. who completed secondary school	IBGE/IPEA
Health	2012	% Number of hospitals per municipality	IBGE/IPEA
GDP	2014	USD	IBGE/IPEA
Favelas	2012	Area/location	IBGE
Indigenous Land	2013	Area of Indigenous land	IBGE
Permanent Crops	2012	Hectare	IBGE/IPEA
Crop rotation	2012	Hectare	IBGE/IPEA
Potential for agriculture	2012	Agriculture suitability areas	EMBRAPA/IBGE
Mining Concessions	2015	Area and location of mining concession	DNPM
Road Infrastructure	2016	Roads	DNIT
Road Density	2016	Number of Roads per municipality area	DNIT
Exports of forest products	2015	Exports of forest products per municipality (represents the degree of connectivity to international markets)	Brazilian Ministry of Agriculture
Beef exports	2015	Beef exports per municipality (represents the degree of connectivity to international markets)	Brazilian Ministry of Agriculture
Population density	2012	Population density per census track	IBGE
Nature protected areas	2014	State/federal natural reserves and protected areas	Brazilian Ministry of Environment
Number of fire events	2016	Average number of fires per municipality per year	MODIS (MCD64A1)
Livestock density	2015	Cattle numbers municipality	IBGE
Elevation	2016	Meters	ALOS (AW3D30)
Land cover type	2013	LUC data: 2002, 2003, 2005, 2009, 2011, 2013	MCD12Q1; MCD12Q2
Slope	2016	Degrees	ALOS (AW3D30)
Soils	2012	Soil type	EMBRAPA/IBGE
Water bodies	2010	Major lakes, reservoirs	IBGE
Flooded areas	2012	Areas under water most of the year	IBGE
Salt marshes	2012	Area of salt marshes	IBGE

Socioeconomic context data

The social and economic datasets gathered here serve to provide context-specific indicators, pointing to the likely factors that influence land cover transition processes in the Amazon estuary (Brondizio et al., 1994a; Brondizio et al., 1994b; Padoch et al., 2008; Vogt et al., 2015). Together, data from the Brazilian Institute of Geography and Statistics (IBGE) and from the Institute for Applied Economic Research (IPEA) constitute more than 66% of all data that were treated, spatialized and used as input. These variables (Table D1) add to the model's predictive capability, and help identify different characteristics at the municipality level that are key in identifying spatial patterns associated with land cover transitions in the region. Therefore, we selected carefully a set of important social and economic variables, which are representative of individual realities in such a vast region. Consequently, this gives insight into the complex nature of spatial patterns arising as a result of a well-established set of rules governing land use/cover transitions over the last few decades not only in the in the Amazon estuary but also globally (de Araujo Barbosa et al., 2016e; Kim et al., 2015; Lambin and Meyfroidt, 2011; Perz et al., 2009; Soares et al., 2006).

Variables such as: percentage of the population employed formally/informally, percentage the population employed by the agricultural sector, education (as in the number of people educated to the at least secondary level), health (as the number of hospitals per capita), GDP, permanent/rotation crop area, potential for agriculture, soils, road infrastructure, road density and livestock density helped define, develop and refine the modelling approach, while revealing the major relationships on the ground (e.g. in terms of agriculture and pasture expansion). In a similar fashion, we use beef exports and forest exports to help establish the connection of the estuary to international markets, since not only local but also distal drivers have important parts to play in changing land use/cover (de Araujo Barbosa et al., 2016b; Duncan et al., 2015; Gibbs et al., 2015; Godar et al., 2014b; Nepstad et al., 2014). Other variables such as elevation, slope gradient, water bodies, flooded areas, mining concessions, salt marshes, served to establish logical relationships of incentives or impediments to urban, agricultural, pasture and deforestation expansion/contraction (Lambin and Meyfroidt, 2011; Lambin et al., 2001; Lindquist et al., 2012; Troupin and Carmel, 2016). And all these variables provide the model with key information and metrics to try and predict land use/cover transitions in the Amazon estuary for the next 50 years (Aguiar et al., 2016a; Brondizio et al., 1994a; Brondizio et al., 1994b; de Araujo Barbosa et al., 2016b; Gong et al., 2015; Kim, 2013; Lima et al., 2014; Lu et al., 2016; Menard and Marceau, 2005; Perz et al., 2009; Soares et al., 2012; Soares et al., 2006). All these datasets were spatialized and converted to raster format and resampled to match the MODIS cell size (500 meters) for model development and processing.

Land use/cover modelling

The approach most often used in land use change modelling to define the model structure is regression on a previous land use/cover map (Al-sharif and Pradhan, 2016; Cohn et al., 2016; Millington et al., 2007; Molowny-Horas et al., 2015; Okubo et al., 2010; Tayyebi et al., 2014; Verburg et al., 1999; Verburg et al., 2002). This method frequently results in only one deterministic model structure, without uncertainty in either the observations used to fit the regression model or in the model itself (Macedo et al., 2013; Soares et al., 2013; Soares et al., 2012; Sohl and Claggett, 2013). Using a statistical approach also does not help in capturing spatial patterns of dependence or heterogeneity across space and/or time, and it only assesses the scale-dependent association between variables. It cannot characterise the dynamic environment on which to identify and model multiple complex relationships that vary across spatial and temporal scales.

The CA model used here was particularly useful in operationalising both theoretical and practical situations, capturing emerging spatial patterns (Soares et al., 2013; Soares et al., 2006). Consequently, allowing for the land use/cover transition model to effectively capture time-dependent spatial phenomena arising as consequence of complex relationships over time. Our CA modelling approach applied a set of transition parameters representing the processes that lead to changes in state over time. Therefore, it incorporated these transition parameters as a function that determines the transition suitability of each cell to a particular land use type (Basse et al., 2016; Blecic et al., 2012; Buizer et al., 2011; Feng and Liu, 2015; Irwin et al., 2009; Van Delden, 2009; Wang et al., 2010). This was determined with respect to a set of spatial variables (context-specific spatial data) that can potentially influence (dynamically) the transition from one particular land use type into another (e.g. distance to roads, elevation, population density, education, health, etc.) (Fontaine et al., 2014; Omar et al., 2014a; Omar et al., 2014b; Wang et al., 2011; Yang et al., 2014).

Model Calibration and Parametrization

Our model contains a comprehensive set of context-specific variables (Table D1) as indicative of the precise background that sets the dynamic environment in which land use/cover is realised in the Amazon estuary. Therefore, it incorporates a modelling approach in which land use/cover change transition rules can be updated and modified in accordance with constant and/or modified change scenarios associated with sustainable/unsustainable exploitation of resources and land allocation practices. In more technical terms, as represented in Figure D.2 the MODIS land cover is used as an input consisting of a bi-dimensional array of land cover types and it is used to estimate transition rates defining how the observed rate of change within the land use/cover system of classes over time. The land use/cover has the following classification types: non-forest (consisting of previously deforested areas, areas with no green vegetation);

agricultural land (areas covered mainly by croplands); urban areas (areas with dense urban settlements); pasture (areas predominantly used occupied by pasture and used for raising cattle); and forest (areas covered by dense tree canopy).

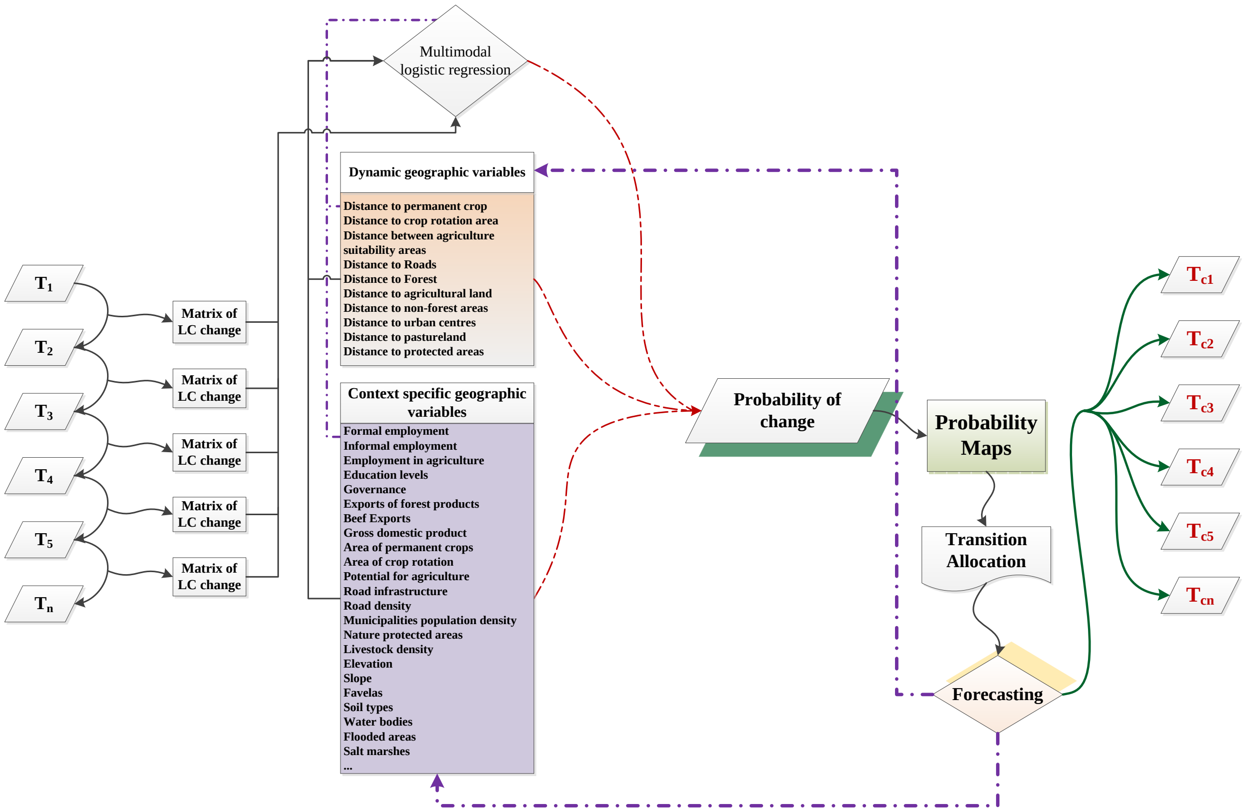


Figure D.2 - Simplified schematic diagram representing the land use/cover change modelling approach and steps. The purple dotted-dashed lines represent the feedbacks after a simulated time-step has been completed.

To quantify the most favourable areas for transition, accounting for the multiple context-specific determinants of physical, social and economic change, we estimate the effect these variables will have on determining areas for each type of a transition by calculating their relationships using a multinomial logistic regression approach. This establishes the probability of transition to a particular land use/cover type given that certain observed conditions exist inside the multivariate set of context-specific geographical variables. After that, the spatial transition probability maps represent the probability of a pixel in the space of coordinates to change from its initial state (*i*) to a subsequent one (*j*). The probabilities are calculated individually for each pixel on the MODIS data, taking into account multiple possible transitions between the land cover types. Furthermore, the transitions are allocated as a function of the spatial probabilities derived as a result of previous steps during the modelling process, resulting in a simulated land use/cover map per time step in the model. Furthermore, the simulation of the dynamic nature of changing land use/cover patterns is updated after each time step and occurs as a function of observed and modelled dynamic spatial variability and probability.

Scenario definition

The scenarios we developed were based on an interpretation of time-series data presented in Table D1 (Appendix D) and the literature. The rationale behind the scenarios was consistent with peer-reviewed storylines reporting the driving forces and relationships that are key to the co-evolution of socioecological systems (Albers et al., 1998; Colson et al., 2009; Eitzel et al., 2016; Liu and Weng, 2013; Maertens and Zeller, 2009).

Our scenarios were conducted over a 50 year time period, incorporating a range of stakeholder-driven perspectives (scenarios), with the purpose of creating a shared vision for both desirable and sustainable future outcomes (da Conceicao et al., 2015; Garrett et al., 2016; Meijer, 2015; Newton et al., 2013; Nolte et al., 2013; Pokorny et al., 2004). Furthermore, different scenario visualisation is key in examining future trajectories in land use/cover change, allowing to explore a broad array of possible future conditions resulting from a defined set of input variables (Dale et al., 1994; Fortini et al., 2015; Frohn, 2001). The scenarios investigated aimed to explore the likelihood of simple outcomes related to better environmental sustainability taking into account a set of key global assumptions for future simulations. Therefore, the model provides plausible forecasts from which to understand the impact of land use/cover change in the estuary at the landscape level (Camacho-Sanabria et al., 2015; Choudhury and DiGirolamo, 1998; Iwamura et al., 2014; Maeda et al., 2011; Mas et al., 2014; Pinel et al., 2015; Thapa et al., 2013; Yu et al., 2016). Thus, the model should be able to quantify and represent complex driving forces interacting over space and time, while at the same time integrate a comprehensive set of data that captures the development and evolution of social and ecological systems in the estuary (Canadas et al., 2016; Hanspach et al., 2016; Mohammed et al., 2016).

Business as usual scenario

The Business as usual (BAU) scenario reproduces the current observable dynamics taking place in the Amazon estuary, it assumes the long standing *status-quo* applied to land use/cover transition and management in the region. Thus, the main assumption is that stakeholders and institutions will continue the same practices and routines. We assumed that the rates of pasture and agricultural expansion, resource exploitation, urban growth, and other social and economic determinants will remain as observed during recent decades in the estuary. Furthermore, the BAU scenario assumptions applied were based on current local, regional and global arguments such as: ineffective or delayed implementation of the Brazilian Rural Environmental Cadastre and the obligation to restore or compensate illegally deforested areas; further reductions of conservation units in the Amazon (facilitating licensing and construction of hydroelectric power plants and validating illegal land grabbing); changing the forest code (to put in place the amnesty of illegal deforestation); implementation of major infrastructure projects without full understanding of environmental impacts.

The BAU scenario includes the following basic framework assumptions: (a) demand for beef will continue to increase around the globe triggering more demand for pasture land; (b) agricultural expansion will continue heading towards forested land (and only sporadically expanding into degraded land) to meet increasing production demand; (c) the Brazilian government will continue making large investments in the construction of roads cutting through the Amazonian rainforest; (d) there is continuous growth in exports of cellulose, raw wood, rubber and paper; (e) GDP, health, education, employment (informal, formal and in agriculture) and urbanisation per municipality continue at the present growth rate per year; (f) the yearly number of fire events per municipality and the number of mining concessions given will remain the same; (g) the proportion of permanent *vs* seasonal crop area will remain at the same progression rate; (h) the government of Brazil will continue expansion of infrastructure inside official indigenous land; and (i) favourable exchange rate towards exports of beef, agricultural and forest products. The rate of change in the socio and economic variables was defined using the observed percentage of increase/decrease year-by-year over the last 10 years.

Alternative (sustainable) land use/cover scenario

Recently published studies suggest that (at the Amazon basin scale), Brazil has already experienced positive feedbacks arising from successful technical and agricultural policies, leading to a reduction in deforestation. The sustainable land use/cover scenario is, therefore, intended to be seen as part of informed public and private interventions, to promote land use/cover policies and develop less destructive practices, so that business can continue “as-usual-as-possible” but promoting socioeconomic development under a more environmentally sustainable design.

The alternative (sustainable land use/cover) scenario is informed by national and international arguments, advocating that there needs to be a more robust understanding about the need to protect forests in the Amazon estuary. Ideally, this would come through increased environmental regulation, adoption of less destructive means of exploiting natural resources, and extensive enforcement of the national Rural Environmental Registry of private properties. Furthermore, our rationale assumes that rather than clearing new forested areas for pasture and/or cropland expansion, production needs to be boosted by increasing productivity on the land that is already in use, or on the millions of hectares of degraded land. This improved alternative to current resource exploitation, has also to be reconciled with the need to cut carbon emissions by investing in adaption and mitigation policies, better regulating expansion of agriculture, industry, exports of forest products. In more practical terms, this scenario is implemented as follows: (a) allocation of pasture land away from forested land and closer to other farm land uses and already degraded land; (b) agricultural expansion largely stopping its move towards forested land (with more intensification and expansion into already degraded land) in order to meet increasing production demand; (c) the Brazilian government continue investment on road infrastructure but less of it cutting through extensive areas of forest; (d) the rate of increase in exports of cellulose, raw wood, rubber and paper will be cut to 70% of its current yearly growth rate; (e) GDP, health, education, employment (informal, formal and in agriculture) and urbanisation per municipality continue at the current observed yearly growth rate; (f) the yearly number of fire events per municipality is cut down by two thirds (assuming better coordinated fire combat action and regulation); g) enactment of the new Mining Code (with stronger regulation on exploration rights, which is predicted to reduce the number of mining concessions by 25% after its implementation); (h) balancing the rate of growth of permanent *vs* seasonal crop area; (i) the government of Brazil will try delimit more protected areas rather than remove the status of the ones already in place; (j) favourable exchange rate towards exports of beef, agricultural and forest products.

Sensitivity analysis and scenario validation

The simulated scenarios were validated against known land use/cover years as measured by MODIS. The scenario validation consisted of applying a measure of spatial fit, comparing the discrepancies between the actual versus the simulated/modelled data for the same year (Carrero et al., 2014; Janalipour and Mohammadzadeh, 2016; Phillips et al., 2011). We aimed to capture a measure of spatial fit between the observed and simulated land cover changes within each cell's neighbourhood (Figure D.3). The resulting map with the estimated spatial fitness represents a fuzzy comparison map, specifying the degree of similarity on a scale of 0 to 1 for each pixel (Aithal et al., 2014; Bolliger, 2005; Hewitt et al., 2014; Nejad et al., 2014; Riveira and Maseda, 2006; Shahverdi et al., 2016; van Vliet et al., 2013a). Thus, it evaluates the spatial fit between the simulated and observed changes in the land use/cover image pixels by applying an exponential decay function with distance to weight the cell state distribution around a central cell (Andjel et al., 1995; Coletti and Tisseur, 2010; de Maere and Ponselet, 2012; Liao et al., 2014; Louis, 2004; Richter and Werner, 1996).

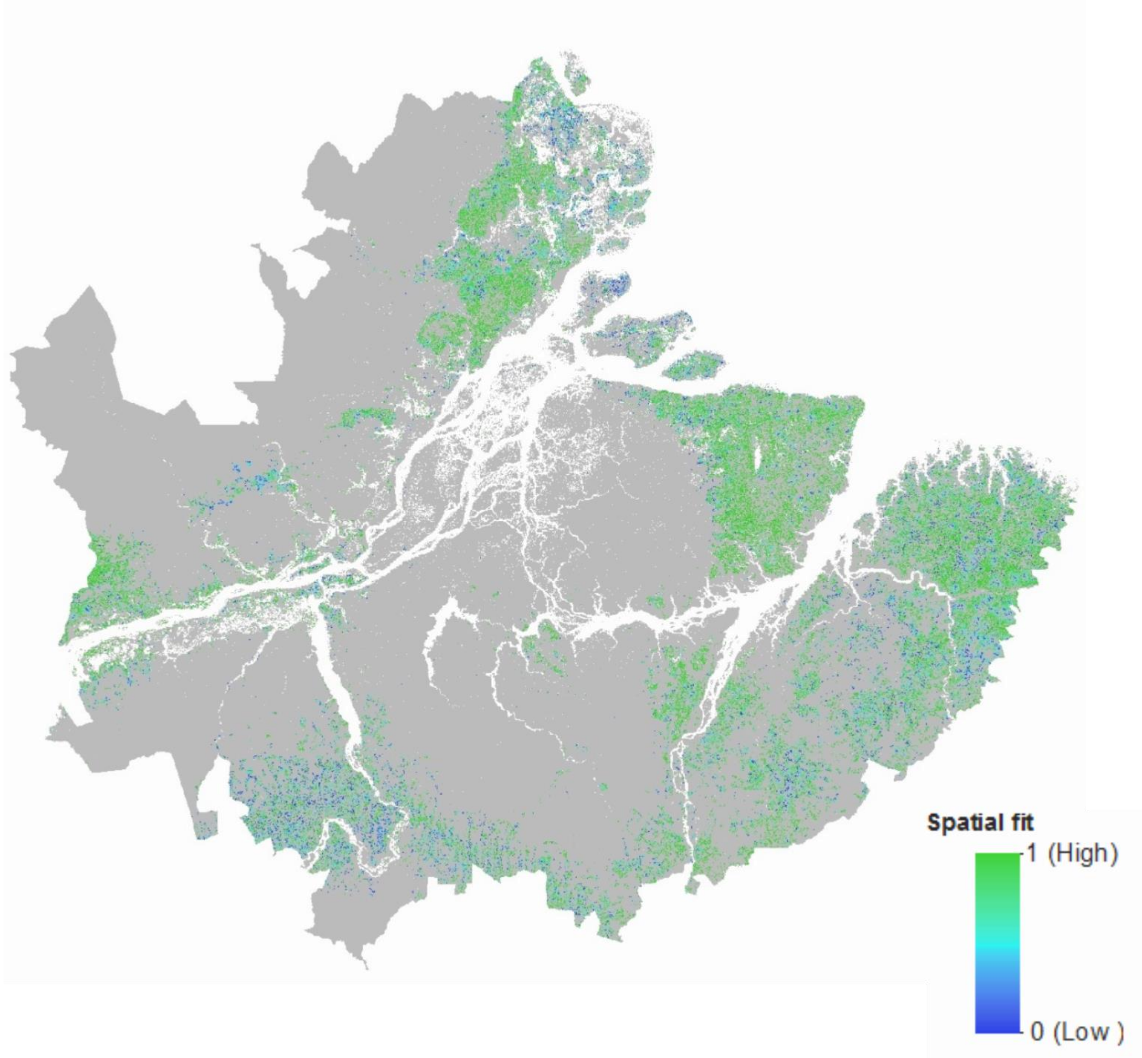


Figure D3 - Measure of Spatial fit: actual vs modelled Land use/cover for the year of 2006.

This validation method has been used in the calibration and validation of cellular automata models for land-use dynamics (Feng and Liu, 2013; Puertas et al., 2014; Whitsed and Smallbone, 2014). The model validation was used a MODIS satellite image that represents a key changing step in evolution of land use/cover in the Amazon estuary, this particular scene was therefore not part of the model parametrization process. This was aimed at eliminating any circular bias that could otherwise permeate modelled outputs. The chart in Figure D3 refers to the year of 2004 (as measured by MODIS) against the modelled output for the same period in time. The year of 2004 represents a period in time were the annual forest loss (in square km) was at its highest.

In order to try determine the change in model output values that results from changes in model input values, it was performed a model sensitivity analysis using a Global Sensitivity Analysis (GSA). Therefore, measuring the change in the model output in every region of the space of inputs, allowing to investigate further the importance of any given change in the model set up. Sensitivity of results to some input parameters shown in Table D1 was tested by running 10 simulations of 50 Monte Carlo (MC) realizations each, while allowing at each simulation step one parameter to vary randomly and keeping the other input parameters constant. Parameters analysed were: Percentage of the Pop. Employed in Agriculture, Education, GDP, Permanent Crops, Crop rotation, Mining Concessions, Road Density, Exports of forest products, Number of fire events, Livestock density. Together with the evaluation of the spatial fit of the model predictions (see Scenario validation), the exact character of the sensitivity analysis further engages the particular context and the questions of concern during the modelling process and adds much to the assessment of model precision and system performance for alternative scenarios.

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