Evaluating ecosystem services in Tropical Dry Forests

by

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ABSTRACT

Ecosystems provide humankind with a wide range of services that are fundamental for our wellness, development, and survival. In recent decades, the loss and deterioration of ecosystems a result of anthropogenic pressures had increased global awareness. Tropical Dry Forests (TDFs) are ecosystems with considerable potential for the provision of essential Ecosystem Services (ES). As such, the main objective of this thesis was to determinate the potential of TDFs to supply two key ES: carbon sequestration and water provision. By doing so, first I estimated the biophysical capacity of TDFs for sequestering carbon and their associated economic value across the Americas. Overall results showed important contributions of the TDFs for the regulation of climate given their efficiency for the transfer of carbon dioxide from the atmosphere as terrestrial biomass. Main biophysical trends showed to be higher and consistent overtime at northern latitudes, and lower but increasing at southern latitudes. Furthermore, I estimated the cost of carbon sequestration for a hectare of TDFs in different Latin American countries.

I also established a baseline for the biophysical assessment of water provisioning services in a TDF, by using a hydrological process-based modeling approach. Simulation results showed appropriate provisioning of water over time. Moreover, important considerations regarding the spatial variability in the provisioning of water in the catchment area were identified and were also relevant to determine differences in the supply of water for human settlements. Since TDFs are ecosystems vulnerable to water scarcity, I also analyzed the potential effect of extreme climatic events on the reliability of the provision of water services. Scenario analysis suggests possible effects of El Niño in the supply of water yield at the study area.

Keywords: ecosystem services, tropical dry forests, quantification, carbon, water.

PREFACE

This thesis is an original work completed by Marissa Castro Magnani. No part of this thesis has been previously published.

DEDICATION

For Flora and Josefina.

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LIST OF SYMBOLS AND ABBREVIATIONS

ACG	Guanacaste Conservation Area
AET	Actual Evapotranspiration
ARIES	ARtificial Intelligence for Ecosystem Services
BLUT	Biome Property Look-Up Tables
BCCR	Banco Central de Costa Rica
CEOS	Centre for Earth Observation Sciences
CFSR	Climate Forecast System Reanalysis
CUE	Carbon Use Efficiency
CV	Coefficient of Variation
DICE	Dynamic Integrated Climate-Economy
ENSO	El Niño Southern Oscillation
ES	Ecosystem Service
FUND	Climate Framework for Uncertainty, Negotiation, and Distribution
FONAFIFO	Fondo Nacional de Financiamiento Forestal
GHG	Green House Gases
GPP	Gross Primary Productivity
IMN	Instituto Metereologico Nacional de Costa Rica
InVEST	Integrated Valuation for Ecosystem Services and Tradeoffs
TDF	Tropical Dry Forest
LAI	Leaf Area Index
LULC	Land Use Land Cover
MEA	Millennium Ecosystem Assessment

MODIS	MODerate- resolution Imaging Spectroradiometer
NCEP	National Center for Environmental Prediction
NGO	Non-Governmental Organization
NPP	Ney Primary productivity
NOAA	National Oceanic and Atmospheric Administration
PAGE	Policy Analysis of the Greenhouse Effect
PES	Payments for Ecosystem Services
PET	Potential Evapotranspiration
PTF	Pedo Transfer Function
REED	Reducing Emissions from Deforestation in Developing countries
RH	Autotrophic Respiration
SCC	Social Cost of Carbon
SWAT	Soil and Water Assessment Tool
TEEB	The Economics of Environment and Biodiversity
SEEA	System of Economics and Environmental Accounting
WAVES	Wealth Accounting and the Valuation of Ecosystem Services

Chapter 1: Introduction

1.1. Introduction

Earth's ecosystems provide a myriad of services that are fundamental for human health, livelihoods, development, and survival (Costanza et al., 1998; McNeely et al., 2009; Sukhdev, 2010). Over the last decades, and as a result of anthropogenic transformations, ecosystems have experienced dramatic changes that are compromising their future sustainability. The imbalance produced between the ever-growing demands of human settlements and the capability of ecosystems to provide a service has resulted in the loss and deterioration of all types of ecosystems around the world (Baskin, 1997). The decline of ecosystems and its consequences on the services they provide was reported in 2005 by the Millenium Ecosystem Assessment (MEA), an extensive study from the United Nations Environmental Programme (UNEP) on the state and relevance of ecosystem services to society. In recent years, the severe consequences of global climate change, and the maintenance of the basic functions of the planet have been addressed by important discussion panels of world leaders, scientists and local communities around the globe. On January of 2016, the United Nations approved the Sustainable Development Goals (SDGs) for 2030, an agenda that promotes actions against climate change, biodiversity loss and land degradation. As a result, the importance of protecting ecosystems and the services they provide has gained more recognition.

The concept of ecosystem services (ES) has been subject of much debate since the 1980´s when was first coined by Ehrilch and Ehrlich (1981) in a philosophical attempt of relating social and natural sciences for a study about species extinction. Since the 1990´s, the evolution of a discipline called ecological-economics (Costanza, 2017), lead the development of many ES case studies and positioned the concept as a novel way of reframing the link between nature and people. The MEA (2005), in accordance with authors such as Daily (1997) and Costanza (1998), defined ES as the "benefits human populations derive from ecosystems." Further definitions conceptualize ES as "the components of nature directly enjoyed, consumed or used to produce human well-being" (Boyd and Banzhaf, 2007) or the "direct and indirect flux of contributions of ecosystems to human welfare" (Farley, 2008; Sukhdev, 2010). Although the ES concept is still under discussion considering its ecological or economical roots (Braat and de Groot, 2012), it is undeniable that has emerged as an essential milestone in the road to preserve the environment as an asset rather than an impediment to human development (Costanza et al., 2017).

A significant contribution of the MEA (2005) was also the establishment of four major categories of ES: (1) provisioning services or the products directly obtained from ecosystems such as food, water and other materials, (2) regulating services or the benefits derived from the regulation of ecosystem processes and functions such as air quality maintenance, climate regulation or water purification, (3) cultural services or the nonmaterial benefits people obtain through spiritual enrichment, cognitive development, recreation or aesthetic experiences, and (4) supporting services that are necessary for the production of all other ecosystem services. These categories rapidly became a global standard and remain at the core of more detailed classification systems (Costanza et al., 2017).

In recent years, a plethora of valuation and quantification methods have been developed reframing the relationship between people and nature (Seppelt et al., 2011; Braat and de Groot, 2012), and eventually supporting policy and decision making at different scales (Daily et al., 2009). Global initiatives such as the post-Kyoto international negotiation process for the development of carbon credits for Reducing Emissions from Deforestation in Developing countries (REDD+), the United Nation's Framework convention on Climate Change (UNFCCC, 2016), The Economics of Environment and Biodiversity project (Sukhdev, 2010), the United Nations System of Environmental Accounting (SEEA, 2012) and the Wealth Accounting and the Valuation of Ecosystem Services (WAVES) of the World Bank-led global partnership, aim to the establishment of standard frameworks that consider the various components of ES assessments.

Assessing the contributions of ecosystems to society, in an ES context, involves a line of three components: ecological, economic and social (Braat and de Groot, 2012). The ecological component position the natural science as a base to understand the structure, processes, and functions of ecosystems. These analyses are usually related to the selection of biophysical indicators that reflect the quantitative aspects of biodiversity and ecosystem properties involved in the provision of a particular ES. The economic component relies on the biophysical estimates and converts them into economic terms to define the worth of an ES, usually in monetary values. This component constitutes the willingness to pay for the supply of an ES or to determine compensations for their loss. Economic values are often aimed to be included in the national economic accounts of a nation, representing their natural capital (Costanza et al., 1998) and the importance of its conservation. The social, and final component, aims to transform the recognition of the worth of an ES into concrete policy and management actions, besides the involvement of social actors in community-based projects that can be sustainable for the ecosystems in the long term. However, the complexity of the social component is deeply related to the different perceptions of well-being that social groups and decision makers in a territory can have, and to how these perceptions will influence the choices regarding the sustainable use and management of their natural capital (Hall, 2012).

Effective ES assessments frameworks, such as the TEEB and the SEEA, consider the spatial representation of ES as an essential stage. For Fisher et al. (2009), without the precise delineation of ES provision boundaries and spatial patterns, no assessment would be reliable since ES are spatially explicit. This concept means that ES can be directly represented in the geographical space as a set of discrete units such as polygonal or equal-area raster cells (Haslauer et al., 2015). In recent years, an increasing number of studies have improved the use of remotely sensed products, modeling and mapping tools for illustrating and quantifying the supply of different ES (Egoh et al., 2012; Martínez-Harms and Balvanera, 2012). In this sense, spatial ES quantification rises as a crucial method to synthesize the complex functions and dynamics of ecosystems (Crossman et al., 2013). Integrating these techniques with comprehensive sets of biophysical indicators and socioeconomic metrics allows a full understanding of the multiple types of services that an ecosystem can provide and its implications for human welfare and environmental policies (Clec'h et al., 2016).

Tropical dry forests (TDFs) are ecosystems that provide a wide range of goods and services (Balvanera et al., 2012), but are also subject to intense anthropogenic disturbances given their suitability for the development of human activities such as agricultural development, cattle ranching, and timber extraction. TDFs have been one of the most deforested and least protected forest ecosystems in America (Janzen, 1988). TDFs are defined as a type of forest with vegetation usually dominated by deciduous trees (at least 50% of drought-deciduous trees), mean annual temperature of 25 C°, and total annual precipitation that ranges between 700 and 2000 mm with three or more dry months every season (Sanchez-Azofeifa, 2005). According to Portillo-Quintero and Sanchez-Azofeifa (2011), the current extent of TDFs in America is approximately 500,000 km2, from which only 4.5% is under protection, and most representative areas are located in Mexico, Brazil, and Bolivia. Although the endangerment of TDF is undeniable, countries like Costa Rica have proven to have fragments under regeneration processes (Arroyo-Mora et al., 2005), an accomplishment related to the combined action of environmental policies, scientific support and stakeholders involvement.

From an ES perspective, TDFs contribute to the regulation of climate as they remove carbon dioxide (CO2) from the atmosphere and efficiently capture it away in their tissues as biomass with more efficient accumulation rates (Mora et al., 2017). According to Maass et al. (2005), TDFs are also the source of many provisioning ecosystem services such as food, firewood, hunting species, medicinal plants and fresh water for local human settlements. Moreover, since forests influence the quantity of water available locally, TDF ecosystems play a significant role in the local hydrological cycle, as they regulate the flow and purification of water (Maass et al., 2005). TDFs also hold considerable biodiversity rates, important not just for adaptations to the effects of climate change and pollination (Balvanera et al., 2012), but are also a source of scenic beauty that drives the development of recreational and spiritual activities for people. Although previous research establishes the importance of monitoring ES in TDFs (Maass et al., 2005; Balvanera et al., 2012), for Calvo-Rodriguez et al. (2016) a complete understanding of the supply of provisioning, regulating and cultural ES in TDFs remains poorly understood. Especially if it is compared to the extensive scientific efforts focused on tropical rainforests (Sanchez-Azofeifa, 2005).

Without having a profound understanding of the multiple ecosystem services that are provided in TDFs, its spatial organization or the relevance that these contributions can have on local and national well-fare accounts, it is unlikely that any conservation or sustainable management solution can be implemented adequately in the coming years. It is in this context that this study performs an analysis of the potential of TDFs for the provision of two key ES, carbon sequestration, and water provision by using a subset of spatial-based tools and models for highlighting their biophysical and economical contributions.

The main objective of Chapter 2 "Assessing terrestrial carbon sequestration in the tropical dry forests of America" is to evaluate the supply of carbon sequestration as a major ecosystem service for six TDF sites in America. Since carbon sequestration represents not just an important driver for climate regulation but also contributes to national accounts and promotes the development of financial incentives, both a biophysical and an economical component are quantified. By providing these estimates, this work aims to contribute to stakeholders, government agencies and NGO's with a baseline for future carbon ES assessments, the implementation and monitoring of incentive schemes and more informed decision-making processes among others.

The main objective of Chapter 3 "Quantifying water provision services in a tropical dry forest: the case of Guanacaste Conservation Area" is to assess the biophysical capacity of the Guanacaste Conservation Area (ACG) in Costa Rica for the supply of water provision ecosystem services. The role of hydrological process-based modeling and spatial explicitness are also explored to finally understand the accessibility of human settlements to water ecosystem services and how reliable the provisioning of water for human well-being is. This chapter aims to provide a basis for future scenario analysis of water resource management, synergies and trade-offs with other ecosystem services and to satisfy the needs of stakeholders for better decision making processes in the future.

A final closing of this thesis is given in Chapter 4 "Conclusions and future work" with a summary describing the future challenges regarding the assessment of ecosystem services in TDFs in the context of forest and water ecosystem services. Additionally, this chapter also addresses some recommendations for pursuing a deeper understanding of ecosystem services in TDF.

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Chapter 2: Assessing terrestrial carbon sequestration in the tropical dry forests of America

2.1. Introduction

Tropical forests sequester and store more carbon than any other terrestrial ecosystem in the biosphere (Gibbs et al., 2007). Carbon sequestration rates represent a direct contribution to the regulation of climate in our planet, and as such are considered as one of the major ecosystem services (ES) provided by forests worldwide (Bonan, 2008). The significance of the concept of ES, also defined for the Millenium Ecosystem Assessment (MEA, 2005) as the benefits people derive directly or indirectly from nature, has become an important milestone for linking the conditions and processes of natural ecosystems to human livelihoods and survival (Costanza, 1998; Fisher et al., 2008). Moreover, for the MEA (2005) the reduction of atmospheric carbon dioxide (CO_2) for terrestrial ecosystems is defined as a key regulating service that contributes to the regulation of climate on the planet. The processes and functions of the terrestrial ecosystems for regulating carbon, help to the reduction of the effects of climate change and the incidence of related phenomena, such as drought and extreme weather cycles. Valuable not only for the security of human settlements' but also for their national accounts (Runting et al., 2017).

Carbon dioxide along with other greenhouse gases (GHG's) are continuously exchanged between terrestrial and marine ecosystems and the atmosphere (Bonan, 2016). However, with the exponential increase of anthropogenic emissions due to industrialization, CO_2 has risen by approximately 40% according to the 5th report of the Intergovernmental Panel on Climate Change (IPCC, 2013), this represents an increase from 278 ppm in 1750 to 390.5 ppm in 2011 (IPCC, 2013). Furthermore, there is high confidence that climate change will induce major concentrations of global CO_2 in the future. Therefore, the maintenance of terrestrial carbon sinks is fundamental for mitigation in future years. According to the MEA (2005), the biosphere absorbs an equivalent of 20% of total fossil fuel emissions, from which forest ecosystems sequester approximately 335 to 365 billion tons of carbon per year, helping to decelerate the effects of global climate change and supporting human welfare (Hall, 2012).

Deforestation and forest degradation are elements of higher impact in the tropics (MEA, 2005) since they have a negative influence on the carbon cycle. This impact also affects the provision of the carbon sequestration as an ES. In this context, measuring current carbon sequestration rates, forecasting losses and estimating the expected economic damages that these fluxes can have in governmental accounts, has recently lead scientific efforts and global initiatives worldwide (Costanza et al., 2017). Initiatives such as the post-Kyoto international negotiation process for the development of carbon credits for Reducing Emissions from Deforestation in Developing countries (REDD+), the United Nation's Framework Convention on Climate Change (UNFCCC, 2016), the United Nations System of Environmental Accounting (SEEA, 2012) and the Wealth Accounting and the Valuation of Ecosystem Services (WAVES) of the World Bank global partnership, aim not just to include forest values into national accounts and efficient decision-making systems. But also, lead the involvement of social actors in community-based projects that can be sustainable in the long term. Furthermore, they also lead the implementation of institutional frameworks for Payments for Environmental Services (PES), incentive systems in which landowners are compensated for the environmental services they generate (Pagiola, 2008), to maintain forested ecosystems and reduce deforestation rates.

The importance of tropical dry forests for climate regulation

TDFs provide a wide range of goods and services (Balvanera et al., 2012) and given their suitability for the development of human activities are one of the most disturbed ecosystems in America (Janzen, 1988). Sanchez-Azofeifa (2005) defined TDF's as a type of forest with vegetation usually dominated by deciduous trees, mean annual temperature of 25 C°, and total annual precipitation that ranges between 700 and 2000 mm with a marked dry season. The current extent of TDFs in America is approximately 500,000 km^2 (Portillo-Quintero and Sanchez-Azofeifa, 2011), with most representative areas in Mexico, Brazil, and Bolivia. Since most continuous fragments are under governmental protection, reliability on the provision of the service is critical for climate regulation at regional scales. Furthermore, given their high biomass accumulation rates (Mora et al., 2017), TDFs possess a good capability for reducing CO_2 concentrations in shorter periods of time (Maass et al., 2005).

Although previous research establishes the importance of monitoring carbon sequestration in TDFs (Sanchez-Azofeifa et al., 2009; Mora et al., 2017), a complete understanding of the provision of the service remains poorly understood (Calvo-Rodrigues et al., 2016). Primarily because of the extensive scientific efforts usually localized in the tropical forests of the Amazon (Sanchez-Azofeifa, 2005).

A remote sensing based approach for ES quantification

The capability of terrestrial vegetation to fix carbon as organic compounds (biomass) through photosynthesis is usually known as primary production and relies almost exclusively on the sun's energy (Woodwell and Whittaker, 1968). A widely used metric to analyze the net photosynthetic accumulation of biomass is net primary production (NPP) and is given by the relation of total atmospheric carbon uptake or gross primary production (GPP) and the energy used to maintain the plant, i.e., dark respiration (Ruimy et al., 1994). Besides playing an essential role in the evaluation of ecological patterns, dynamics, and processes (Waring and Running, 2004), NPP is one of the metrics more frequently used as an ecosystem service indicator for climate regulation (Costanza et al., 2017).

Since NPP calculation involves the capture of solar energy by vegetation, remotely sensed products have proved to be successful alternatives to approximate forest carbon estimations (Gibbs et al., 2007). Current remote sensing procedures measure the radiative properties of vegetation with an emphasis in two regions of the electromagnetic spectrum, the chlorophyll-absorbing red spectral region (0.6 to 0.7 μm) and the non-absorbing near infrared region (0.7 to 1.1 μm) (Chuvieco and Huete, 2010). These regions are often used to develop predictive relations of biophysical variables of productivity usually based on empirical and semi-empirical productivity models (Grace et al., 2007, Turner et al., 2005). However, in-situ methods based on small-scale estimations of ecosystem productivity remain the most direct approach to measure carbon sequestration (Brown, 2002). A significant benefit of choosing a satellite-based approach is that information can be collected systematically, over large geographic areas and that remains a more cost-effective choice (Jones and Vaughan, 2010), not to mention that remote sensing approaches are under continuous evolution and development (Boyd and Danson, 2005).

Why is it important to value carbon sequestration?

TDFs perform a key social role in supporting the life of millions of people, directly or indirectly (Balvanera et al., 2012). Valuation is an inevitable process that is not always related to a monetary value (Costanza, 2017), but it is mostly associated with the recognition of the importance of certain goods and services that are provided by ecosystems. Initiatives like eco-labeling forest products, user fees for recreational activities and the implementation of payments for environmental services (PES) had successful results in some countries of Latin America (Hall, 2012). For example, Costa Rica has been a forerunner in the recognition of forest environmental services through the implementation of economic incentives paid to landowners for preserving lands with forest cover (Sanchez-Azofeifa et al., 2007). Furthermore, programs like REDD+ are also trying to engage developing countries such as Brazil, Mexico, Bolivia and others into the maintenance of forest land through the implementation of carbon credits (Hall, 2012). Incentives for carbon sequestration have reached its inclusion in global markets by usually quantifying how much carbon is sequestrated per unit of area and its equivalent in USD currency.

There are a considerable number of methods to determinate the value of carbon sequestration, most of which rely on econometric models (Tol, 2009; Hope, 2011; Nordhaus, 2017) that estimate the economic cost caused by an additional ton of CO_2 emitted to the atmosphere as an effect of global warming. Regardless of the difference between econometric models for the calculation of monetary values, their primary objective is to promote the recognition of forest ecosystems for the regulation of climate in our planet (Ninan, 2007). A recognition that becomes an important milestone for the development of proper public policies, more informed decision-making processes, and the consistent involvement of social actors for maintaining TDFs ecosystems.

Given this context, this study assesses the supply of carbon sequestration as a major ecosystem service of TDFs. Because carbon sequestration represents not just an important driver for climate regulation but also a base for the development of financial incentives, in this analysis the main questions to be answered are: 1) How much carbon dioxide is being sequestrated in TDFs ecosystems?, and 2) What is the associated economic value for climate regulation over time?

By providing a range of biophysical and economical estimates of carbon sequestration across the Americas, I hope that this work will contribute to stakeholders, government agencies and NGO's with a baseline for the development of forest conservation policies, PES schemes, and more informed decision-making processes. Helping to facilitate reductions in emissions from tropical deforestation and other sources.

2.2 Methods

2.2.1 Study areas

The study was conducted at tropical dry forests located across different spectrums of land use in the Americas (Figure 2.1): Chamela and Yucatan in Mexico, Santa Rosa in Costa Rica, Mata Seca in Brazil and San Matias and Tucavaca in Bolivia.

The most northerly sites are located in Mexico. The first one is in the Chamela-Cuixmala Biosphere reserve (127 km^2) on the west coast of Mexico (19°22′1.6″N–19°35′8.9″N, 104°56′15.3″W–105°3′24.04″W) (heretofore named CH). The mean annual temperature is 24.9 °C, and the mean annual precipitation is 748 mm, falling mainly between July and October (Balvanera and Aguirre, 2006). The area comprises a range of tropical deciduous forests (Lott et al., 1987; Sánchez-Azofeifa et al., 2009) that according to Kalácska et al. (2004) are under different successional stages (from early secondary growth to largely undisturbed forest). The second study site is in Mexico and is located in the Yucatan Peninsula (20°01′1.3″N–20°09′46.6″N and 89°23′24.2″W–89°35′59.8″W) (heretofore named YU). It is also characterized by a tropical dry semi-deciduous forest of 341 km^2 under current regrowth due to cropland abandonment (Dupuy et al., 2012). The mean annual temperature is 26.5 °C, and the mean annual precipitation is 1190 mm (Dai et al., 2015).

Located in the Pacific coast of northern Costa Rica with an area of 388 km^2 is the Santa Rosa National Park (10°44′7.2″N–10°57′16.4″N, 85°34′43.2″W–85°57′4.8″W), our third study site (heretofore named SR). The area is defined as a seasonally dry Neotropical forest (Sanchez-Azofeifa et al., 2005) covered by a mosaic of secondary forests under various stages of succession (early, intermediate and late) (Janzen, 2000; Kalácska et al., 2004; Arroyo-Mora et al., 2005). Although precipitation is yearly variable (Janzen, 1998), Sanchez-Azofeifa et al. (2014) describe a mean annual precipitation of 1390.8 mm with a dry season that extends from December to April and a mean annual temperature of 26.6 °C.

The fourth study site is located in the Southern hemisphere, in the Parque Estadual da Mata Seca, State of Minas Gerais in Brazil (14°48′36″S–14°56′59″S and 44°04′12″W–43°55′23.9″W) and it covers an area of 116 km^2 (heretofore named MS). The forest is defined, according to the Instituto Estadual de Florestas (2000), as a dry season deciduous with a mean annual temperature of 24.9 °C and a mean annual precipitation of 871 mm with a dry season that ranges from May to October (Kalacska et al., 2005). Vegetation also incorporates various successional stages (early, intermediate and late) of natural regeneration (Sanchez-Azofeifa et al., 2009), given the abandonment of pasture and agricultural fields (Instituto Estadual de Florestas, 2000).

The southern study sites are located in the Chiquitano TDF of Bolivia, a transition zone between the humid evergreen forests of the Amazon and the deciduous thorn-scrub vegetation of the Gran Chaco. The fifth site of this study is located in the Area Natural de Manejo Integrado San Matias (heretofore SM), in the lowlands of Santa Cruz of east Bolivia (16°54′27.4″S–

18°06′28.4″S and 58°43′2.4″W–59°37′9.4″W). San Matias has a mean annual temperature of 24.9 °C and a mean annual precipitation of 1488 mm with a dry season from April to October. It is characterized by a high diversity of fauna and flora that spreads across two fragile interconnected ecosystems: the Chiquitano dry forest and the Pantanal. The region was designated as a wetland site of international importance for the Ramsar Convention on Wetlands of International Importance especially as Waterfowl Habitat in 2001 (Gobierno Autonomo Departamental de Santa Cruz-Bolivia, 2017). A sector of 5713 km^2 of the dry forest was selected for this study.

The sixth study site is located in the Tucavaca Municipal Wildlife Reserve (heretofore TU) (18°07′47.3″S–18°33′58.9″S and 58°57′33.2″W–59°32′43.8″W). The reserve protects a portion of the Chiquitano TDF and covers an area of 1741 km^2 (Gobierno Autonomo Departamental de Santa Cruz-Bolivia, 2017). The mean annual temperature is 25.19 °C, and the yearly precipitation is 1143.3 mm, presenting a wet season from November to March and a dry season from April to October (SEARPI, 2011). According to Navarro and Maldonado (2004), the area represents a transitional zone of the semi-deciduous forest towards the Chaco and Bolivian-Tucuman bio-geographical provinces.

All six study sites were selected based on Portillo-Quintero and Sanchez-Azofeifa (2010) identification of TDF extent across North and South America. Although the sites have differences in age, structure, and land use history at local scales, as pointed for Gibbs et al. (2007) these are not significant for the distinction of the ecosystem service in this study since the estimates rely on site averages that consequently are integrated into regional and national quantifications.

2.2.2 Biophysical quantification

To address the first question of this study, biophysical quantification of carbon sequestration in TDF's was conducted by using a remotely sensed spatially explicit approach. This was done by establishing a relationship between productivity and its equivalent to total CO_2 sequestration per spatial unit in each of the sites.

Net Primary Productivity as an ecosystem service indicator

Estimating aboveground living biomass of trees is a critical step in quantifying carbon sequestration in tropical forests (Gibbs et al., 2007). Because Net Primary Productivity (NPP) is usually quantified in terms of carbon stored in vegetation per unit of area and time (Bonan, 2016), I selected NPP as the ES indicator. To capture NPP values, a balance (equation 1) between total carbon uptakes during photosynthesis, also known as Gross Primary Productivity (GPP), and total carbon loss during growth and maintenance of vegetation, also known as autotrophic respiration (RA), must occur (Waring and Running, 2007).

$$NPP = GPP - RA \quad (1)$$

The main driver of NPP in terrestrial ecosystems is photosynthesis, a process by which plants absorb light energy and produce carbohydrates from atmospheric CO_2 (Bonan, 2016). Using remote sensing instruments for approximating NPP has proved to be a successful approach for evaluating carbon stocks proxies across larger areas (Turner et al., 2005; DeFries et al. 2005; Romero-Sanchez and Ponce-Hernandez, 2017).

MODIS satellite products processing

A well-known global vegetation productivity remotely sensed product is provided by the MODerate- resolution Imaging Spectroradiometer (MODIS) (Running, 2004). Data for MODIS GPP product (MOD17A2H) version 6 was collected in a cumulative 8-day composite at 500 meter pixel size resolution from the year 2001 to 2015 using the geographic boundary of each site. Since MODIS provides an 8-day composite for each scene, I aggregated the correspondent scenes on a monthly base for the studied time interval. Using the average GPP monthly value for all pixels at every site (kg C m^2) I calculated monthly NPP taking as a reference the MOD17 NPP algorithm (Running, 2004). I improve the input data by incorporating local meteorological values, and parameters from the MODIS Biome Property Look Up Tables (BPLUT) as detail in Table 2.1 (Running and Zhao, 2015).

The MOD17 algorithm estimates net photosynthesis (NPP) by subtracting growth respiration (Rg) and maintenance respiration (Rm) from GPP as shown in equation 2.

$$NPP = GPP - Rm - Rg \quad (2)$$

Since the algorithm estimates Rm as a function of Leaf Area Index (LAI), data for MODIS LAI (MOD15A2H version 6 at 500 meter pixel size) was collected on an 8-day composite that consequently I aggregated on a monthly base. There are six parameters within the BLUT (Table 2.1) needed to calculate Rm. LAI aggregates were used to calculate Leaf mass as shown in equation 3, Fine root mass is then estimated as shown in equation 4 to calculate the maintenance respiration of the fine root mass (Froot_MR) as is explained in equation 5.

$$Leaf\_Mass = LAI/SLA \quad (3)$$

$$Fine\_Root\_Mass = Leaf\_Mass * froot\_leaf\_ratio \quad (4)$$

$$Froot\_MR = Fine\_Root\_Mass * froot\_mr\_base * Q10\_mr^{(Tavg - 20)/10} \quad (5)$$

A leaf maintenance respiration (Leaf_MR) was calculated as equation 7. A constant value of 2.0 for Q_{10}, the factor by which the rate of a measurement increases for every 10 degree rise in the temperature (Bonan, 2016), was used for fine root and live wood. But an acclimated Q_{10} was used for Leaf_MR as proposed by Tjoelker et al. (2001) and shown in equation 6. A local site temperature average (Tavg) was also used in this calculation.

$$Q_{10} = 3.22 – 0.046 * Tavg \quad (6)$$

$$Leaf\_MR = Leaf\_Mass * leaf\_mr\_base * Q10\_mr^{(Tavg - 20)/10} \quad (7)$$

To determine live wood maintenance respiration (Livewood_MR), the live wood mass was calculated as shown in equation 8 and then used in equation 9.

$$Livewood\_mass = Leaf\_mass * livewood\_leaf\_ratio \quad (8)$$

$$Livewood\_MR = Livewood\_mass * livewood\_mr\_base * Q10\_mr^{(Tavg - 20)/10} \quad (9)$$

In accordance with Cannell and Thornley (2000), monthly growth respiration (Rg) is empirically parameterized as 25 % of NPP. Finally, all previous components were used in equation 10 to estimate NPP in (kg C m^2). A flowchart showing the steps behind the algorithm is presented in Figure 2.2.

$$NPP = 0.8 * (GPP – Leaf\_MR – Froot\_MR – Livewood\_MR) \quad (10)$$

I finally accumulated annually (from January to December) for each year of the time period selected. This represents the annual NPP accumulation during the growing season.

Total Carbon Dioxide (CO_2) calculation

The carbon footprint is expressed in tons of carbon dioxide per unit of area on a yearly base. In order to calculate total annual estimates of sequestered CO_2, I used a conversion based on the relationship between the molecular mass of an existing amount of carbon in one mole of CO_2 and, as defined by Clark in 1982, I multiplied each kg of C by a value of 3.667. Final values were expressed in tons of CO_2 per hectare per year (t CO_2 ha^{-1} $year^{-1}$).

To evaluate the biophysical capability of TDF's for the provision of the ES, I examined the capacity of forests to transfer carbon from the atmosphere to terrestrial biomass. This relationship is defined by Gifford (1994) as Carbon Use Efficiency (CUE) and relies on a fixed ratio of NPP to GPP as is shown in equation 11.

$$CUE = NPP/GPP \tag{11}$$

2.2.3 Economic valuation

Benefit transfer approach

To address the second question of this study, I selected a benefit transfer approach, this method relies on existing economic information derived from previous estimates at some place and time and makes inferences about the economic value of environmental services at another type of landscape or time (Wilson and Hoehn, 2008).

As recommended by Saklaurs et al. (2016), to determine the monetary value of sequestered CO_2 in TDF's I selected a range of economic values (in US $) by using the Social Cost of Carbon. I calculate the total monetary value of CO_2 sequestered in every TDF site (EV_{CSi}) as expressed in equation 12.

$$EV_{CSi} = CS_i \times V_c \tag{12}$$

Where *CSi* is the amount of carbon sequestered annually in a hectare of TDF (expressed in t CO_2 ha^{-1} $year^{-1}$), and V_c is the monetary value (US $) of one ton of CO_2. V_c reference values were selected based on previous "Social Cost of Carbon" (SCC) studies.

The Social Cost of Carbon (SCC)

The SCC is a monetary value of the damage costs associated with the emission of one additional ton of CO_2 to the atmosphere and its further effects on climate change in a given year (Keck, 2014). Since the SCC is a way of putting a price on CO_2 emissions, this value can also be used to weigh the benefits of sequestrating CO_2 against the costs of the negative impacts of climate change and global warming on national economies (Pizer et al., 2014). There are three primary econometric models currently being used, the Dynamic Integrated Climate-Economy - DICE (Nordhaus, 2017), the Climate Framework for Uncertainty, Negotiation, and Distribution - FUND (Tol, 2009) and the Policy Analysis of the Greenhouse Effect - PAGE (Hope, 2011). All of them developed to estimate reference SCC values. All three studies deal with different assumptions regarding the changes in socio-economic projections, net agricultural productivity, human health, property damages from increased climate risks, among other inputs.

For the determination of the monetary reference values (V_c) in TDFs, as pointed before, a range of three SCC estimated values were selected. Table 2.2 shows the estimates from highest to lowest. The SCC estimates are given in US $ of the year in which they were calculated. In order to perform the calculations, I updated the selected values considering the annual inflation rates at each site (Pizer et al., 2014) within the evaluation period (2001 to 2015). As a lower limit, I used the last update of the average SCC value estimated by Tol (2009) with the FUND model and Nordhaus (2017) with the DICE model, while in the upper limit I used the updated estimates of Hope (2011) for the PAGE model. Carbon prices were expressed in US $ per ton of CO_2 sequestrated.

2.3 Results

2.3.1 Estimation of carbon balance in TDFs

Annual GPP trends comparison among sites (Figure 2.3) indicate that for years 2001 to 2015 northern sites (SR, CHA, and YU) accumulated higher GPP than those compared at southern latitudes (MS and TU), except MA in Brazil. A database of all monthly GPP estimates from MODIS17A is shown in Appendix 1.

Results of the GPP partitioning into NPP and autotrophic respiration (RA) for every TDFs site are detailed in Table 2.3. Average monthly values for every place are also displayed in Figure 2.4. The site with most productivity is SR in Costa Rica, with an annual average GPP of 12.9 tC ha^{-1} $year^{-1}$ and an annual loss for respiration of 3.2 tC ha^{-1} $year^{-1}$ which generates an annual average NPP of 9.6 tC ha^{-1} $year^{-1}$, an equivalent of 74% of total GPP. Months with higher productivity rates range from November to March. The second site with high productivity records is MA in Brazil with an annual GPP average of 10.9 tC ha^{-1} $year^{-1}$ and a total NPP of 8.1 tC ha^{-1} $year^{-1}$, also 74 % of total GPP, respiration annual average accounts for 2.8 tC ha^{-1} $year^{-1}$. December to May are the months that present higher NPP values.

TDF sites in Mexico present similar values for productivity. Annual GPP averages for CH and YU are 9.9 tC ha^{-1} $year^{-1}$ and 9.0 tC ha^{-1} $year^{-1}$ respectively. However, CH shows a more efficient use of carbon (approximately 70.15% of total GPP), an equivalent to 7.0 tC ha^{-1} $year^{-1}$ of annual NPP and 2.9 tC ha^{-1} $year^{-1}$ annual respiration. For YU, an annual NPP average of 5.6 tC ha^{-1} $year^{-1}$ and 3.4 tC ha^{-1} $year^{-1}$ of annual respiration was calculated, just 61% of total GPP is converted to NPP. Season length for NPP accumulation ranges approximately from November to April in both sites.

Southern TDFs sites in Bolivia presented lowest productivity rates. SM showed an annual average GPP of 5.3 tC ha^{-1} $year^{-1}$ with a loss for respiration of 2.1 tC ha^{-1} $year^{-1}$ and a total annual NPP of 3.2 tC ha^{-1} year^{-1}, 58% of total GPP is accumulated as NPP. TU registered the lower values across all TDFs sites with an annual average GPP of 5.0 tC ha^{-1} year^{-1} and 2.9 tC ha^{-1} year^{-1} of annual NPP, approximately 57 % of all incoming GPP, annual average respiration losses summed a total of 2.0 tC ha^{-1} year^{-1}. Compared to northern sites, SM and TU present shorter length of the season for NPP accumulation, approximately from April to July.

Figure 2.5 presents results of annual NPP and respiration rates for all TDFs sites. Two main trends can be distinguished, an almost constant accumulation with slight increases for sites in CH and YU in Mexico, SR in Costa Rica and MA in Brazil; and an increase in carbon accumulation over time for SM and TU in Bolivia.

Furthermore, from the Carbon Use Efficiency analysis, we determined that TDFs have an average CUE of 0.66 across all sample sites (Table 2.4), this means an approximate transfer of 66% of the carbon from the atmosphere converted as terrestrial biomass in the ecosystem. The strength of the relationship for all years is presented in Figure 2.6. Where SR and MA show higher CUE values (0.74 for both sites), 0.71 and 0.61 for CH and YU in Mexico and 0.58 and 0.57 for SM and YU in Bolivia, respiration in these two last sites is the responsible factor for lower NPP transfer rates (DeLucia et al., 2007).

2.3.2 Carbon dioxide sequestration in TDFs

The mean annual value for the service of carbon sequestration in TDFs is 22.3 ± 3.2 tCO_2 ha^{-1} yr^{-1} (Figure 2.7), with values of 35.3 ± 2.9 tCO_2 ha^{-1} yr^{-1} in SR (Costa Rica), 29.7 ± 3.3 tCO_2 ha^{-1} yr^{-1} in MA (Brazil), 25.8 ± 2.5 tCO_2 ha^{-1} yr^{-1} in CH and 20.5 ± 1.5 tCO_2 ha^{-1} yr^{-1} in YU (both in Mexico). Carbon and productivity rates for Bolivia contrast with the rest of TDFs sites by having lower carbon sequestration rates per hectare with 11.8 ± 5.0 tCO_2 ha^{-1} yr^{-1} in SM and 10.8 ± 4.1 tCO2 ha^{-1} yr^{-1} in TU.

Regarding historical trends, as presented in Figure 2.8 and detailed in Table 2.5 from 2001 to 2015, similar patterns with no significant differences as seen in the productivity rates were revealed. A consistent annual behavior with lower variations at sites in Costa Rica, Mexico and Brazil and increasing trends in TDF sites at Bolivia.

2.3.3 Estimated economic values of TDFs

The overall mean annual value of carbon sequestration for a hectare of TDF across all sites ranges from $488,8 USD ha^{-1} yr^{-1} in the lowest estimated value for Tol (2009), $771,7 USD ha^{-1} yr^{-1} for Nordhaus (2017) and between $1414.2 USD and $2828.5 USD ha^{-1} yr^{-1} for highest estimated values according to Hope (2011). Same patterns (lowest boundary defined by Tol's reference value, the median for Nordhaus and highest for Hope) were revealed for each TDF site and are presented in Figure 2.9 and detailed in Table 2.6.

For Mexico, two ranges of values were calculated for CH and YU. Results for CH show the lowest estimate of \$564,9 USD ha^{-1} yr^{-1} according to Tol (2009), \$891,8 USD ha^{-1} yr^{-1} for Nordhaus (2017) and a range of \$1634.3 to \$3268.6 USD ha^{-1} yr^{-1} for Hope (2011). In contrast to CH, YU presents lower economic values for Mexico considering it sequestration rates. Estimates for YU for the lowest boundary were \$449,5 USD ha^{-1} yr^{-1} for Tol (2009), \$709,6 USD ha^{-1} yr^{-1} for Nordhaus (2017) and between \$1300.4 to \$2600.9 USD ha^{-1} yr^{-1} for Hope (2011).

SR in Costa Rica represents the maximum economic value for a hectare of TDF. Economic values for the lowest boundary are \$772,5 USD ha^{-1} yr^{-1} (Tol, 2009), \$1219,5 USD ha^{-1} yr^{-1} for Nordhaus (2017) and a range of \$2234.8 to 4469.7 USD ha^{-1} yr^{-1} for the highest boundary, according to Hope (2011).

In the TDF of Brazil, economic valuation was higher compared to all sites in the southern hemisphere (the second highest from all sites). Annual averages range from \$650,2 USD ha^{-1} yr^{-1} according to Tol (2009), \$1026,5 USD ha^{-1} yr^{-1} for Nordhaus (2017) and between \$1881.2 to \$3762.3 USD ha^{-1} yr^{-1} for Hope (2011).

Furthermore, compared to all values in northern sites, lowest economic values were identified for the sites at SM and TU in Bolivia. Results for SM show the lowest estimate of \$259.1 USD ha^{-1} yr^{-1} for Tol (2009), \$409.1 USD ha^{-1} yr^{-1} for Nordhaus (2017) and a range of \$749.7 to \$1499.3 USD ha^{-1} yr^{-1} for Hope (2011). Same trends were estimated for TU with a lower economic boundary of \$236.8 USD ha^{-1} yr^{-1} for Tol (2009), \$373.8 USD ha^{-1} yr^{-1} for Nordhaus (2017) and between \$685 and \$1370.1 USD ha^{-1} yr^{-1} for Hope (2011).

However, all previous values are those estimated for one single hectare of TDF at each site. Total estimates might differ by considering entire area of a site. For example, by assuming Nordhaus (2017) as the median and latest updated reference value of the SCC, and considering total area occupied by each TDF study site, greater variations were found. Regarding the value of carbon sequestration for protected sites in Bolivia, by conserving all 5713.2 km^2 of TDF in SM a total estimate of \$233.7 million USD was calculated. Closer estimates were found for TU with \$65.1 million USD for a total area of 1741.4 km^2. In contrast, SR with a smaller area of 388.11 km^2 presents a total value of \$47.3 million USD. The total value of YU (340.7 km^2) and CH

(127.08 km^2) in Mexico were of \$24.2 million USD and \$11.3 million USD respectively. Finally, MA in Brazil with an extension of 116.2 km^2 possess an estimated value of \$11.9 million USD.

2.4 Discussion

This study had the goal to present the overall contribution of TDFs for the regulation of climate through the carbon sequestration service in the Americas. Results revealed that TDF supply an annual average CO_2 sequestration rate of 22.34 ± 3.31 tCO_2 ha^{-1} yr^{-1}, which equals to an economic value that ranges from \$488,8 USD ha^{-1} yr^{-1}, for the lowest estimate, to \$2828.5 USD ha^{-1} yr^{-1} for the highest value. These values represent a sequestration of approximately 1.16 gigatons of CO_2 per year for all 519.597 km^2 of TDFs in the Americas (extent calculated by Portillo-Quintero and Sanchez-Azofeifa in 2010) with an equivalent economic value that ranges between USD 25 billion for the lowest estimate to USD 146 billion for the highest estimate.

Regarding comparisons, the CO_2 sequestration estimates of this study fall below the range of regional values calculated by Brown (1997), with 78 tCO_2 ha^{-1} and Achard et al. (2004) with 56 - 155 tCO_2 ha^{-1}. But above the values of specific TDF studies and growth stages, such as Kalacska et al. (2008) with values that range from 31,8 to 88.9 Mg CO_2 ha^{-1} and Cao et al. (2016) with estimates that range from 11,8 to 27.8 Mg CO_2 ha^{-1}. It is important to acknowledge that these studies present variations regarding the type of indicator and method used, as well as the scale of investigation. For example, some regional studies tend to not clearly differentiate between tropical humid and tropical dry or deciduous forest. A list of all comparative studies and details such as indicator used, methods and results are presented in Appendix 2.

For studies that include economical estimates of carbon sequestration or a specific focus in ES assessments, a comparison point regarding functionality, methods, and values were not comparable since current research efforts lack an integrated ES approach that incorporates biophysical and economic values. This was also emphasized by Calvo-Rodriguez et al. (2017) during an evaluation of current ES research trends in TDFs. Moreover, previous studies have focus on explore particular components of only the biophysical (Balvanera and Aguirre, 2006; Vargas et al., 2008; Dai et al, 2015; Mora et al., 2017) or economical dimensions (Pfaff et al., 2007; Pagiola, 2008) of the carbon sequestration service. For which this study represents an important integrated baseline for future research.

Significance and limitations of the biophysical quantification

Regarding the biophysical quantification, maximum values for carbon sequestration were found in northern sites, with SR in Costa Rica the forest site with a higher contribution for the delivery of the service, followed by MA in Brazil and both sites in Mexico (CH and YU). Estimates of the carbon use efficiency (CUE) at each site explain that the ratio of NPP to GPP for SR and MA is similar (0.74). This means that 74% of total GPP is converted to NPP. Against estimates of CUE for all forest types established by DeLucia et al. (2007) who defined a standard ratio of 0.53, CUE in our study proved to vary across TDFs sites explaining differences not just in NPP accumulation but also in respiration losses. An important remark for the high CO_2 rates in Costa Rica and Mexico is the spectrum of use, dry forests in Costa Rica are usually dominated by higher regrowth rates and biomass accumulation as a result of good conservancy policies (Sanchez-Azofeifa et al., 2009). Recovery of carbon pools in Mexico was also proved to be relatively efficient, along with the improvement of tree diversity and composition due to regrowth (Mora et al., 2017).

Unlike sites at northern latitudes, sites in Bolivia (SM and TU) presented the lowest values for CO2 sequestration but also lower CUE rates (just 58% is accumulated as NPP). Differences that according to Navarro and Maldonado (2004) could be explained by forest structure and composition since forests in SM and TU are more dispersed, and the dominant species have lower heights. TDFs in Bolivia, and also in Brazil, are also subject to many anthropogenic disturbances given their potential for timber and non-timber products (Pinto-Ledezma and Rivero-Mamani, 2014). Furthermore, in contrast to the rest of the sites, annual trends in Bolivia showed an increase in sequestration of approximately 50% from the year 2001 to 2015, trends that remain stable with slight increases for Costa Rica, Mexico, and Brazil.

Carbon storage and sequestration are a subject of vital scientific research for ES quantification. The lack of a standard approach, not even for the SEEA framework, results in a range of studies that can have differences in scale, methodology, data collection, analysis, validation and interpretation among others (Sanchez-Azofeifa et al., 2005). Although the different methods are likely to be reasonably accurate, uncertainty due to various factors is of crucial concern in estimation.

A first uncertainty source in this type of study is the effect of human activities for the disturbance of forest ecosystems. TDFs is a very threatened ecosystem particularly in Bolivia and Brazil, where a large extension of TDF has been already converted to other land uses as a consequence of deforestation (Pinto-Ledezma and Rivero-Mamani, 2014). Miscalculations of deforestation rates and areas can lead to mistakes in carbon uptake and emissions balance (Calvo-Alvarado et al., 2009). The geographic location of this change, as well as the precision in the estimation of forested areas, might also affect CO_2 quantification. Kalácska et al. (2004) established these effects by analyzing different land use classification products for Costa Rica and Mexico, finding important discrepancies (27% for Costa Rica sites and 56% for Chamela in Mexico). Moreover, a proper carbon balance should also consider fire disturbances. The frequency of fires in countries like Costa Rica has always had an impact on biomass recovery and accumulation during certain seasons (Allen, 2001).

The second source of uncertainty is the differences in carbon accumulation that can be influenced by forest properties such as structure, stages of growth, age, site quality, species composition and others (Sanchez-Azofeifa et al., 2009). Although this study was conducted at a regional scale and forest properties were not taken into account, a local study by Mora et al. (2017) for a TDF in Mexico, found a significant relationship between observed aboveground biomass and forest age, the definition of methods and algorithms to accurately estimate forest age. Furthermore, Sanchez-Azofeifa et al. (2009) established that secondary regrowth possesses high potential for carbon sequestration, historical trends for Costa Rica have proven this statement. Recovery of carbon pools due to regrowth in Mexico can also occur relatively quickly, along with the improvement of tree diversity and composition (Mora et al., 2017). Transferring predictive relations over space to link biophysical variables such as LAI might also have an impact on the estimation of carbon sequestration (Nightingale et al., 2004).

A final source of uncertainty relies on the selection of the indicator and quantification method. A frequent and direct way to quantify carbon stocks is the use of allometric regression models to convert forest inventory data to estimates of aboveground biomass (Brown, 1997). However, this study demonstrates that using net primary production (NPP) as a proxy measure of carbon sequestration is a reasonably good alternative as it is the basis of several forest functions and processes. A good correlation between NPP and its agreement with regulating services was also established by Costanza et al. (2017). Since NPP is relatively easy to measure with remote sensing over space and time, this greatly simplified this assessment. The benefits of a remote sensing based approach are the access to systematic information in time, and its relatively friendly cost and applicability for more extensive areas in ES assessments (Gibbs et al., 2009; de Araujo et al., 2015).

Limitations regarding the selected method rely on the effects of cloud cover in tropical regions, in TDFs during the dry season, when chances for cloud-free images are better though when dry forests are predominantly leafless and difficult to distinguish (Kalacska et al., 2008). The quality of MODIS products has improved nowadays, and successful applications have been registered for characterizing spatial variation and monitor spatial changes of effective biomass (Running, 2004). However, because tropical ecosystems are particularly complex, discrepancies in the content should be considered (Kalacska et al., 2008). Recent studies have tried to improve the development of GPP and NPP algorithms by integrating MODIS with in-situ data and ecosystem modeling at finer scales (Wang et al., 2015; Cui et al., 2016).

In this scenario, remote sensing presents an effective alternative to improve the scientific understanding of terrestrial carbon fluxes but also presents some challenges since it cannot provide accurate information on what is happening below the canopy (Sanchez-Azofeifa et al., 2009). However, because no single method is considered as the most effective, a higher level of studies suggest the integration of different ecosystem modeling tools and field sampling methods to retrieve more accurate estimations (Grace et al., 2007). Variations regarding the combination of techniques will efficiently improve the biophysical estimation of the carbons sequestration service in forest ecosystems.

Main trends and gaps of the valuation method

TDFs ecosystems possess a value that goes beyond the extraction of raw materials, such as timber and other no forestal products. By slowing CO_2 emissions, it is more likely that the severity of climate change and its associated damages for human settlements also decrease (Dai et al., 2015). Since most of these avoided damages are implicit in society (Costanza et al., 2017), this study identified the monetary contributions of a hectare of TDF across all studied countries (in $USD), and using three reference values of the SCC (Tol, 2009; Hope, 2011; Nordhaus, 2017). In correspondence with the CO_2 sequestration rates from the biophysical component of this study, maximum values where found for Costa Rica (from $772.5 to $4469.7 USD ha^{-1}), followed by Brazil (from $650.2 to $3762.3 USD ha^{-1}), Mexico from ($507.2 to $2934.7 USD ha^{-1}) and lower estimates for Bolivia (from $259.1 to $1499.3 USD ha^{-1}).

Costa Rica is the only country that has developed a local carbon market with incentives for forests conservation, reforestation lands and agroforestry systems (Pagiola, 2008). Currently, the Fondo Nacional de Financiamiento Forestal (FONAFIFO) is the authorized entity for the trade of carbon compensation units, nowadays FONAFIFO handles a currency of $7.5 USD per ton of CO_2 (Sanchez-Chavez and Navarrete-Chacon, 2017) which given our biophysical estimates corresponds to a value of 264.5 USD ha^{-1}. This value falls under our estimates for Costa Rica but is higher than the ones expressed by Kalacska et al. (2008) who estimated values in a range of $14.6 to $43.9 USD ha^{-1}.

For comparisons in Mexico, Brazil and Bolivia, a review of REDD initiatives in Latin-America show costs in a range of $2 to $10 USD per ton of CO_2 (Bastos et al., 2017), which represent lower estimates than the ones estimated in this study by using the Social Cost of Carbon. REDD reference values are currently being debated given its low competitivity with other land uses, such as agriculture and cattle ranching, and the attractiveness of the benefits that these might have for local communities (Hall, 2012).

Results also proved that estimated unitary values could consequently be used to calculate the total revenue of a specific area, such as a protected area, reclamation lands, and others, which can change the perspective regarding overall values of a particular study site. This was proved by multiplying the unitary value for the total area of each site which surprisingly put Bolivia sites (SM and TU) as the sites with largest monetary values. According to Portillo-Quintero and Sanchez-Azofeifa (2010), Bolivia protects 10,609 km^2 of dry forests followed only by Brazil, which represents an essential contribution to global carbon sequestration. However, they also recognize the risk of anthropogenic pressures, which is higher compared to countries like Costa Rica or Mexico.

Regarding the methods, this assessment proved that the Benefit Transfer method is a useful approach for avoiding the costs and time of developing original econometric modeling, an important element usually considered by decision makers (Boyd and Banzhaf, 2007). A key challenge in any valuation process is imperfect information (Costanza et al., 2017), therefore considering the transfer of values from one ecological and social context to another one is likely to have an effect on uncertainty. Moreover, because the valuation relies on biophysical quantities, limitations of the biophysical method might also be transferred to the economic valuation. Sanchez-Azofeifa (2009) also established that trade-offs between carbon credits and debits must be analyzed to have better assessments. Furthermore, this study managed to offer to decision-makers, the scientific community, and society a set of estimates about the current value of TDFs.

Future work and challenges

The most significant challenge nowadays is to improve the efforts of researchers, governmental agencies, NGOs and stakeholders for a better understanding of the benefits of measuring, mapping and valuing the service of carbon regulation in TDFs. In agreement with Calvo-Rodriguez et al. (2016), this study also found that there is still a lack of deep research regarding all components of an ES assessment for carbon sequestration in TDFs, particularly for the economic valuation since the more comparative research was found for the biophysical component. Complementary efforts must also be linked to adequate funding levels that promote comprehensive in situ and comparative studies among different TDFs areas in countries of the Americas.

Since current carbon markets have lower reference costs, there is a clear need for increasing the appeal of incentives for local communities and stakeholders in all countries involved in this study, particularly in those that do not have a local carbon market established. This is also extremely important for countries such as Brazil and Bolivia since the largest areas of protected dry forest are located in these countries (Portillo-Quintero and Sanchez-Azofeifa, 2010), but unfortunately, these have experienced higher risks of deforestation and land use changes. For example, despite the increasing rates of carbon sequestration over time identified in Bolivia, the conservation of the Chiquitano TDF is very much in doubt (Pinto-Ledezma and Rivero-Mamani, 2014).

A key concern for implementing proper ES accounting for carbon sequestration is how precise and comparable the measurements are. In this sense improvements regarding the quantification methods, systematic collection of the data, the cost-effectiveness of the techniques, monitoring of results over time, and applicability at different scales among others must be done. All this to improve the understanding of carbon balance in TDFs.

Furthermore, for van Beukering et al. (2013) measuring ES supply is important but not sufficient to determine the benefits for society. To identify substantial achievements in social welfare from carbon markets, with exceptions such as Costa Rica, has been a difficult task mostly because of the low local involvement (Halls, 2012). In this context to include local and regional stakeholders into the development of conservancy measures, establishments of incentives, integrative land use/forest programs, and environmental education might have positive long-term benefits for the maintenance and restoration of TDF.

2.5 Conclusions

What this study makes abundantly clear is that TDFs are important ecosystems for the regulation of climate through the sequestration of CO_2, a benefit of great value for local and global welfare. This study also demonstrated fundamental contributions of the carbon sequestration service for national economic accounts by establishing the monetary value of TDFs in different sites of America. Moreover, this study provides a systematic methodology for a complete assessment of the CO_2 sequestration service. It demonstrated that the integration of a remote sensing based approach (for the biophysical component) and a benefit transfer method (for the economic component) are promising means to assess the supply of carbon sequestration as a vital ecosystem service in forests ecosystems. Although we may never have an exact estimate of the quantity or value of this service, this study provides an initial baseline for additional studies, not just for future ES assessments but also for analyzing changes through time and space and finding trade-offs and synergies with other services provided in TDFs.

Overall, the outcomes of this study highlight, not just the relative importance of maintaining and recovering TDFs areas in the Americas but also to reduce the risks to anthropogenic disturbances and to fill the gap between theory and policy. Expressing these contributions can have major impacts on the implementation of sustainable and inclusive use practices for local communities, better-informed decision-making processes for stakeholders, and more efficient conservation and restoration policies.

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2.7 Figures

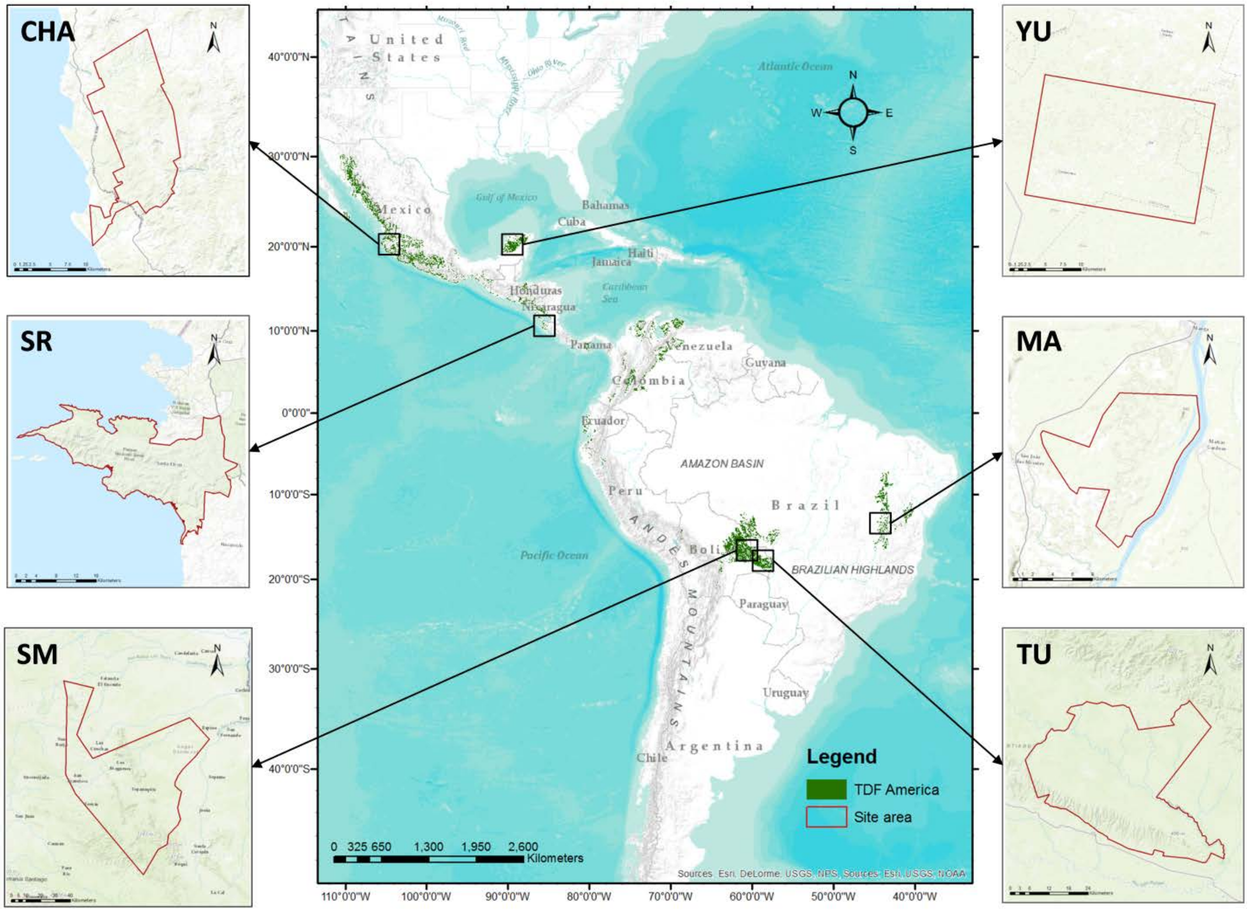


Figure 2.1 Location of the study sites across the Tropical Dry Forests in Central and South America. Chamela (CHA) and Yucatan (YU) in Mexico, Santa Rosa (SR) in Costa Rica, Mata Seca (MA) in Brazil and San Matias (SM) and Tucavaca (TU) in Bolivia.

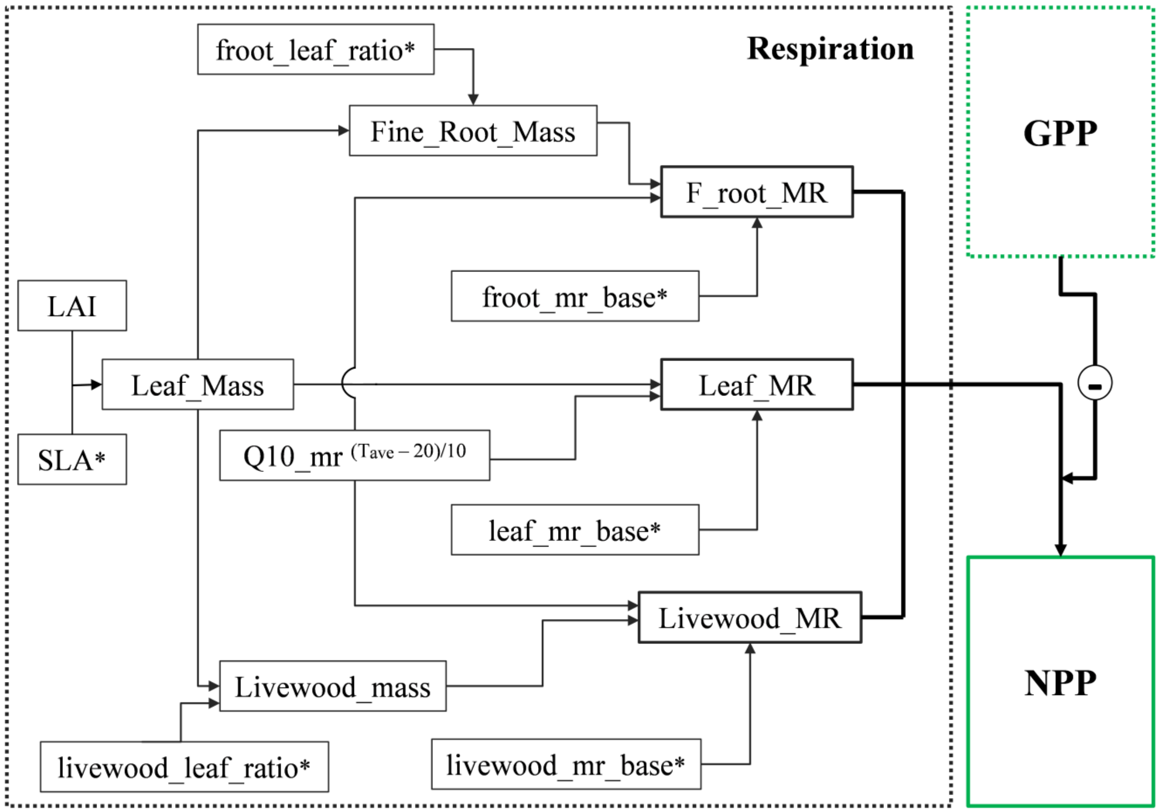
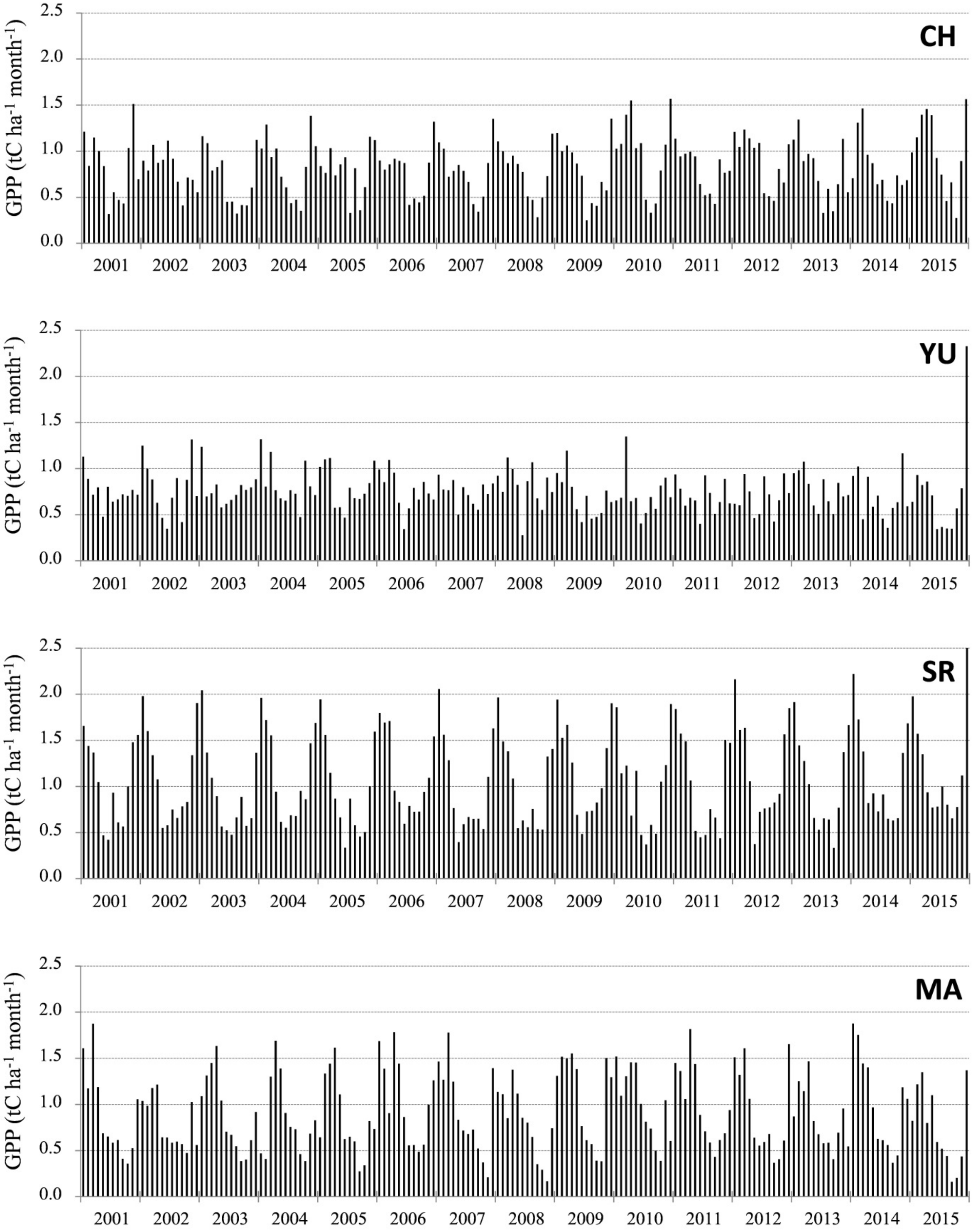
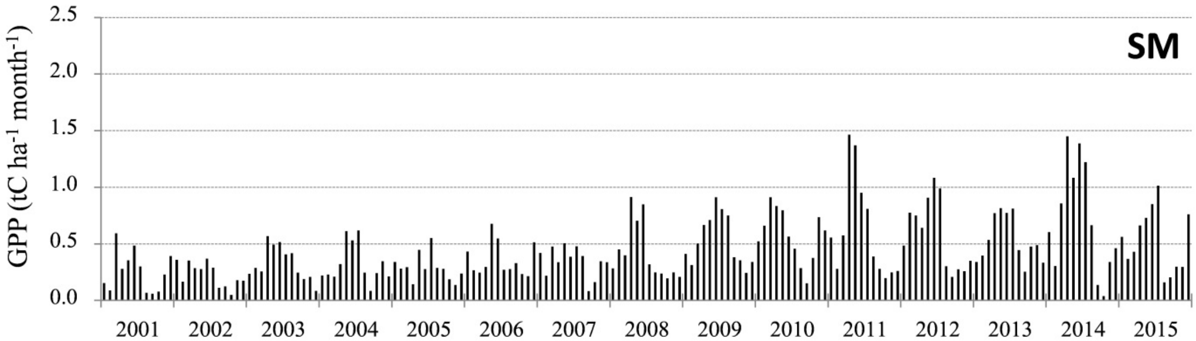


Figure 2.2 Flowchart showing the logic behind the MOD17 algorithm for calculating monthly and annual Net Primary Productivity (NPP). Elements designated with an asterisk correspond to biome constant values of the Biome-Property-Look-Up-Table (BLUT) of MOD17.


CH
YU
SR
MA
GPP (tC ha-1 month-1)
0.0
0.5
1.0
1.5
2.0
2.5
2001 2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015



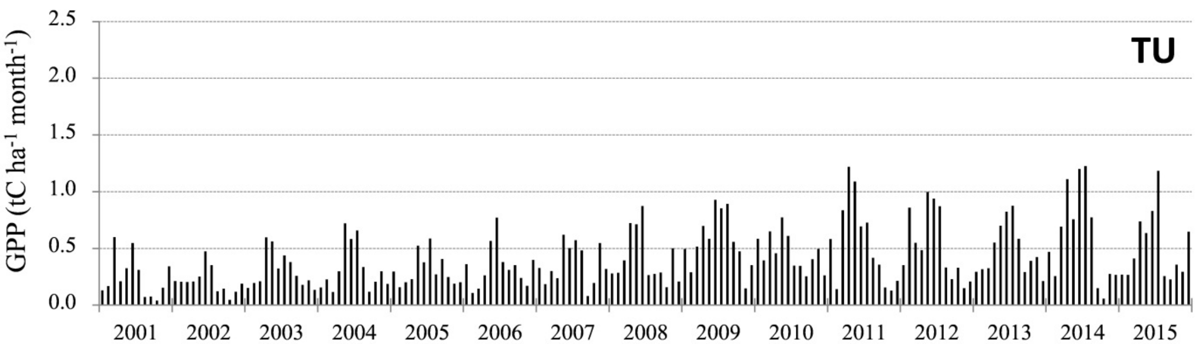


Figure 2.3 Total monthly values of MODIS GPP for all TDFs sites from years 2001 to 2015. Units are expressed in tC ha^{-1} $month^{-1}$ and site information is displayed from North to South as follows: Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU).

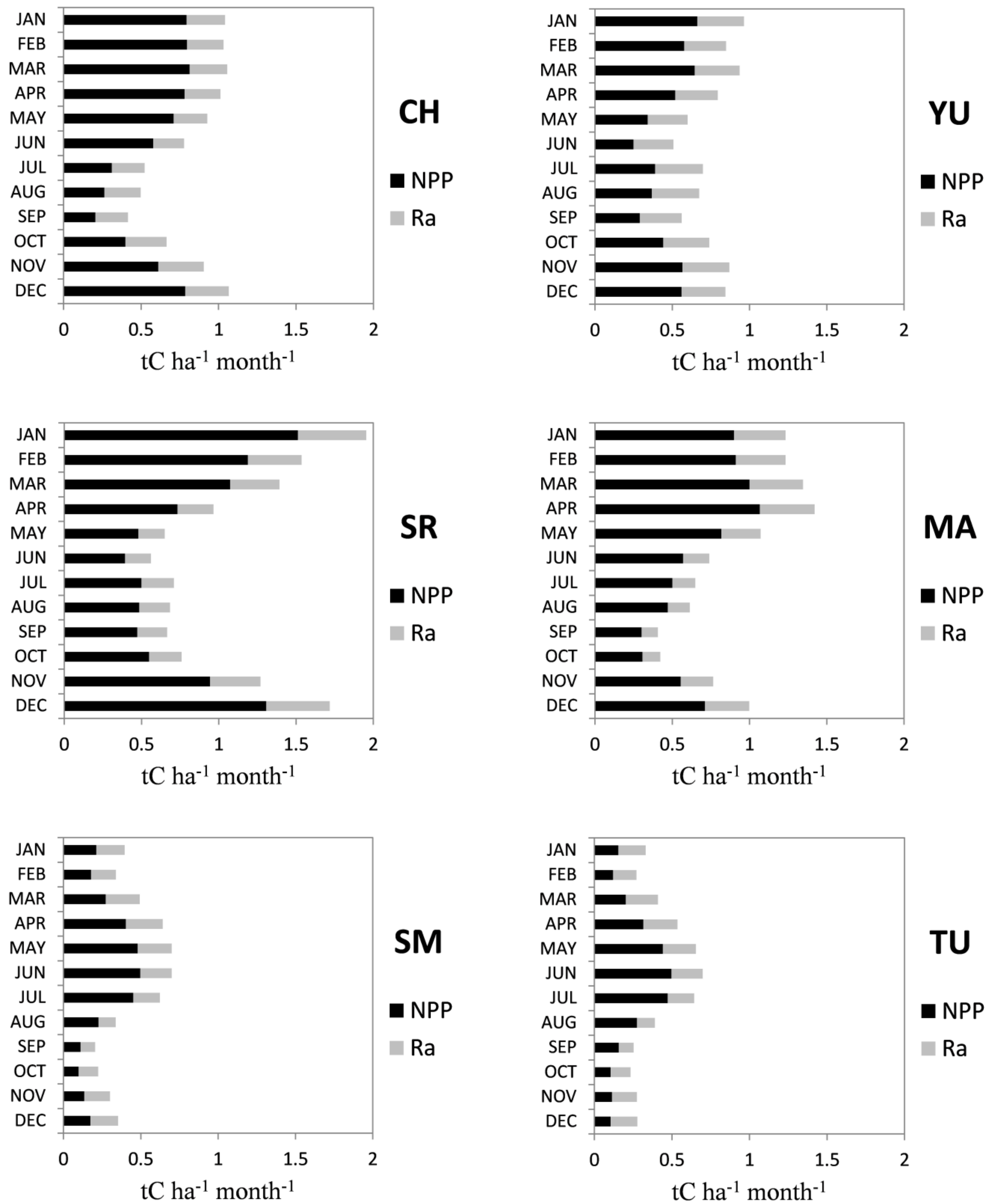


Figure 2.4 Mean monthly values of GPP partitioned into Autotrophic Respiration (Ra) and Net Primary Productivity (NPP) for years 2001 to 2015, values are expressed in tC ha^{-1} $month^{-1}$ for all TDFs sites including Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU).

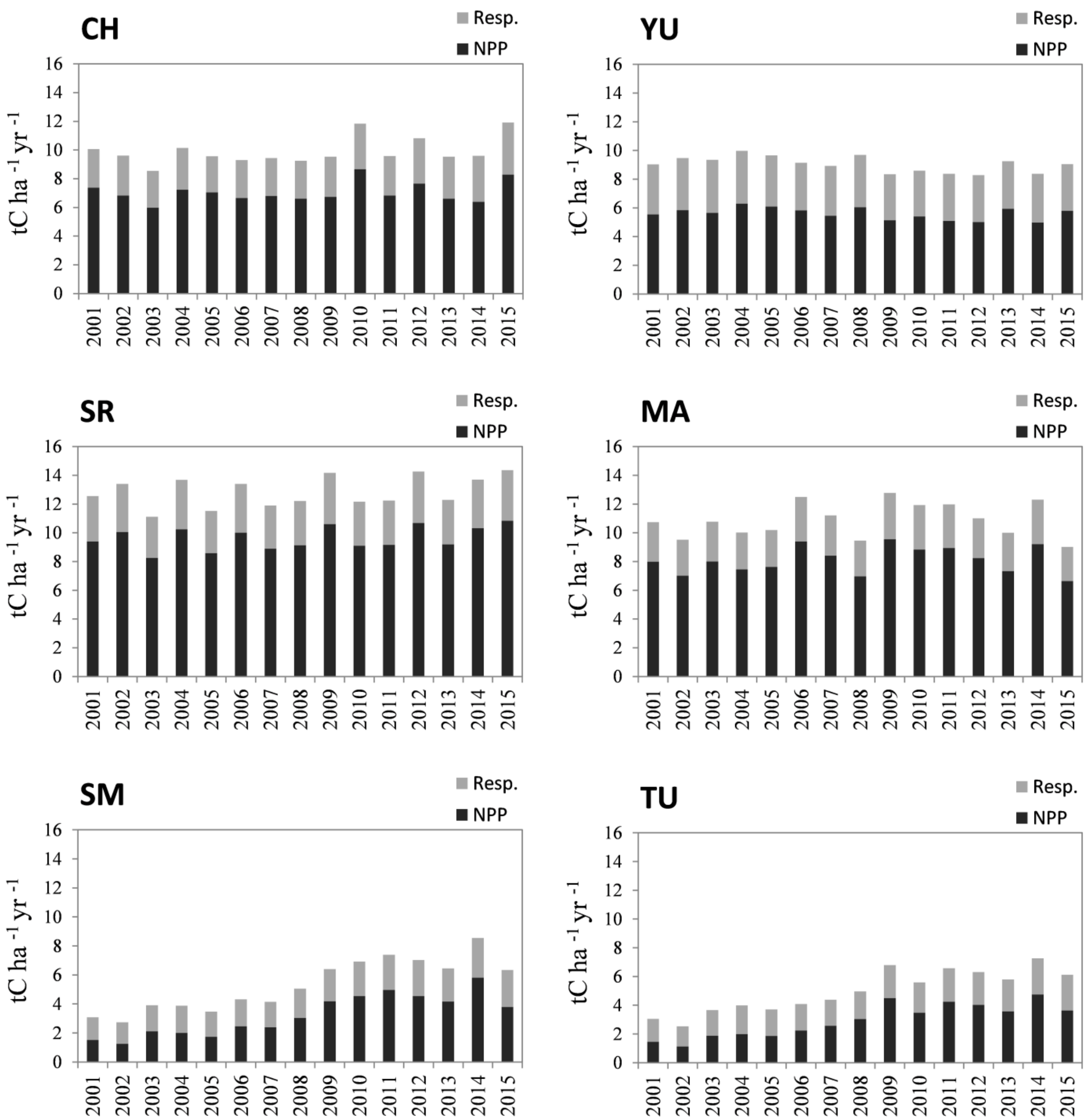
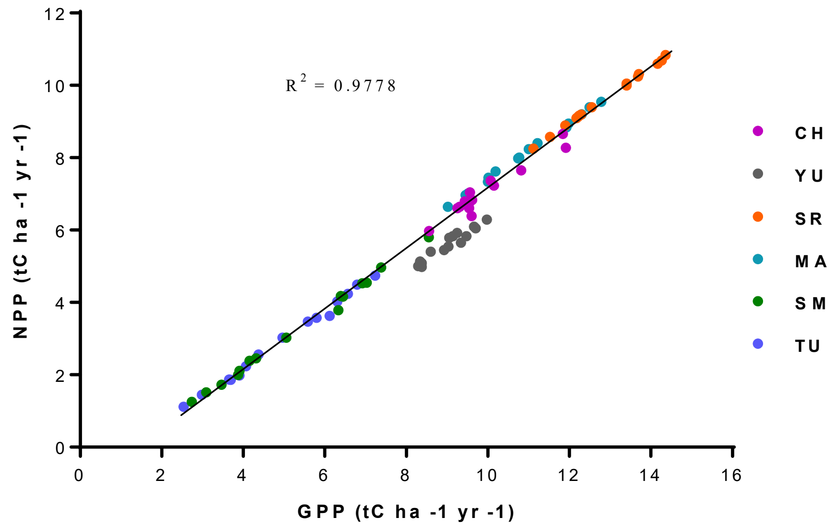


Figure 2.5 Total annual Net Primary Productivity (NPP) and respiration (Ra) rates from years 2001 to 2015. Values are expressed in tC ha^{-1} yr^{-1} for all TDFs sites, including Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU).

a)



b)

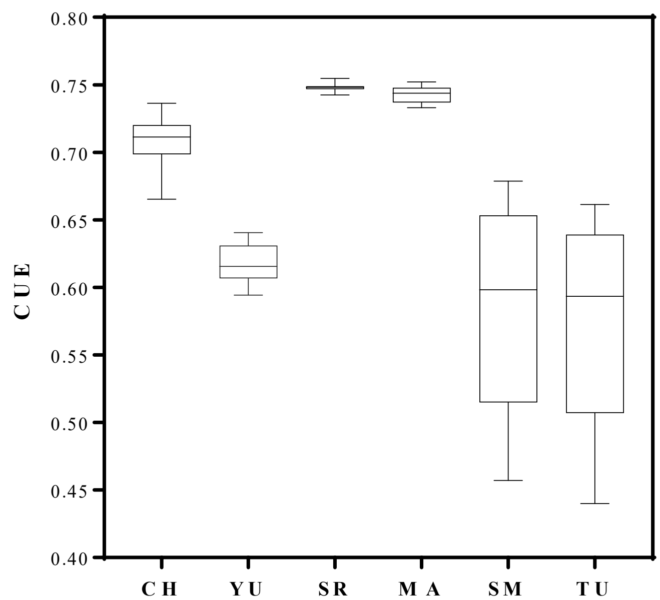


Figure 2.6 a) Ratio of NPP vs. GPP for all TDFs sites including Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU). **b)** Distribution of CUE values for all TDFs sites. Higher values indicate more efficiency and NPP accumulation. The boundaries of the boxes indicate the 25th and 75th percentiles and the median is indicated by a solid line inside the box. Whiskers above and below the boxes indicate the 90th and 10th percentiles.

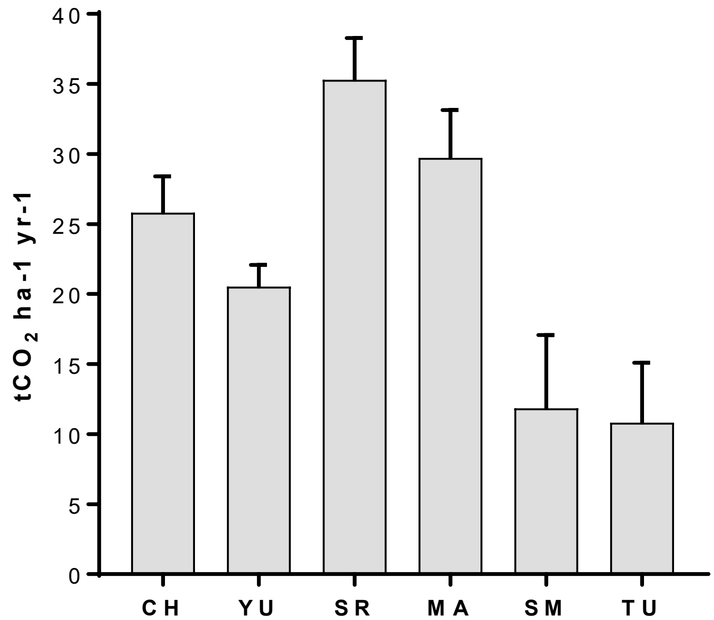


Figure 2.7 Total mean annual estimates of CO_2 sequestration from years 2001 to 2015 for all TDFs sites including Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU). Values are expressed in tCO_2 ha^{-1} yr^{-1}. Bar size represents the mean across all years and whiskers above represent the standard deviation.

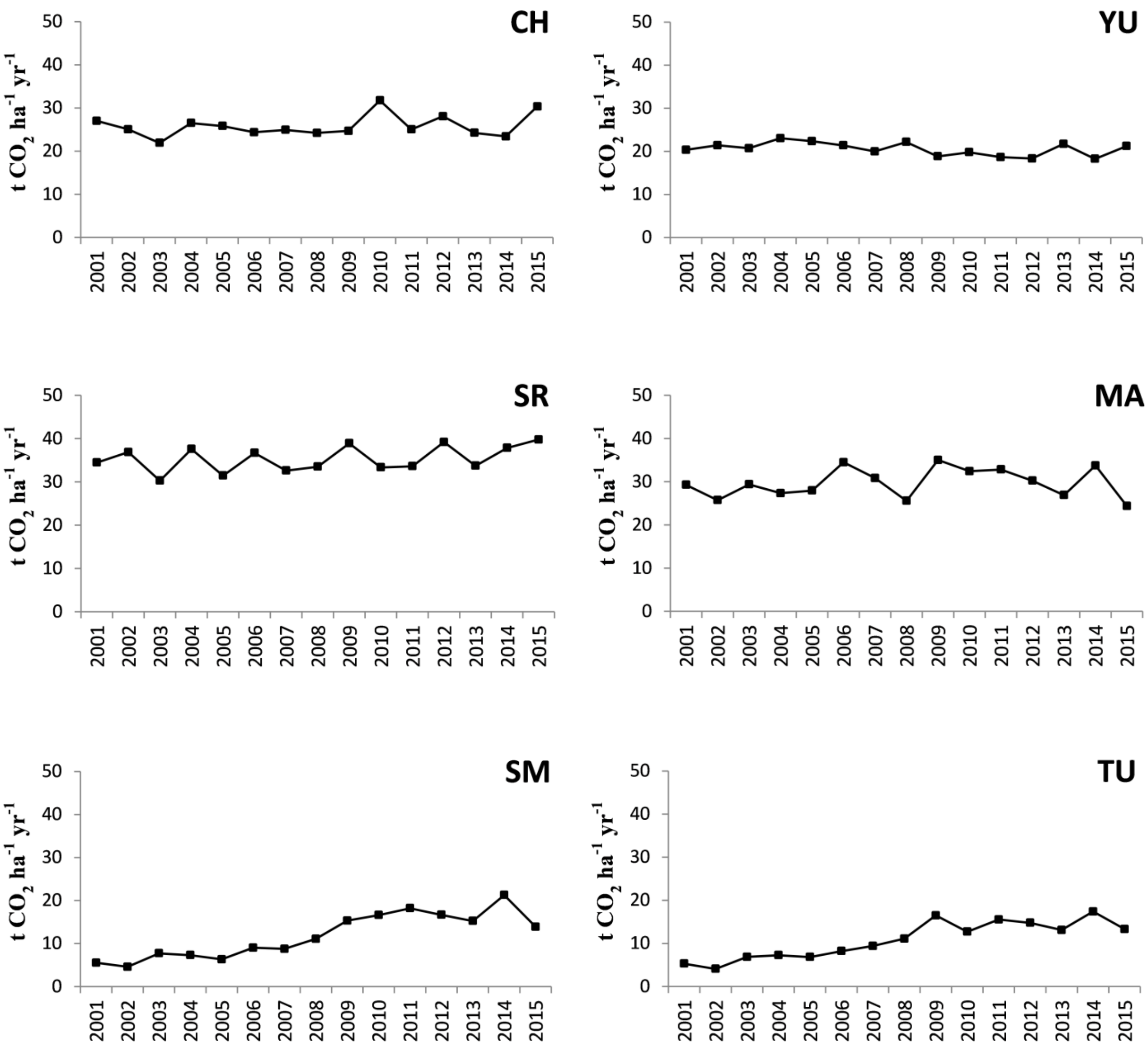


Figure 2.8 Annual CO_2 sequestration rates from years 2001 to 2015 for all TDFs sites including Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU). Values are expressed in tCO_2 ha^{-1} yr^{-1}.

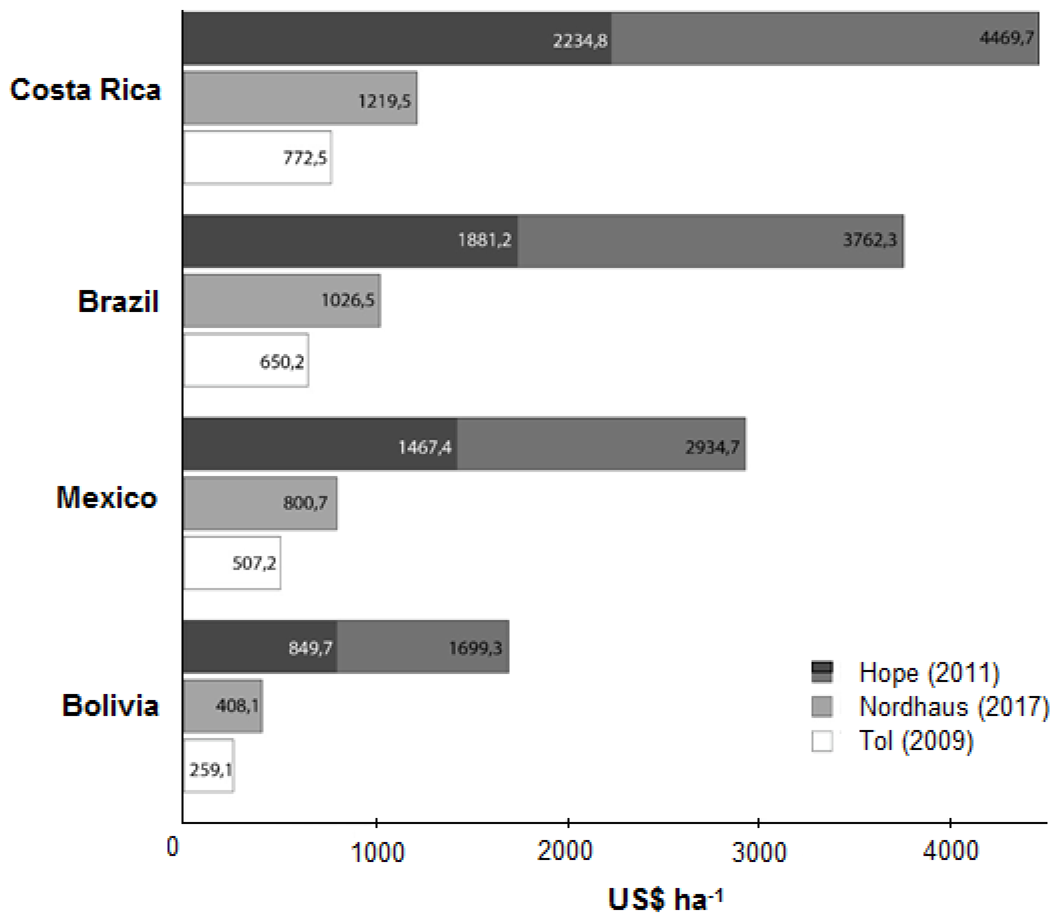


Figure 2.9 Estimated mean values (in $USD) of the carbon sequestration service for one hectare of TDFs in Costa Rica, Brazil, Mexico, and Bolivia. Three references of the Social Costs of Carbon (SCS) are displayed from highest for Hope (2011), medium for Nordhaus (2017) to lowest for Tol (2009).

2.8 Tables

Table 2.1 Values of Biome-Property-Look-Up-Table (BLUT) for MODIS NPP algorithm calculation (Running and Zhao, 2015).

Parameter	Value	Units	Description
SLA	21.8	(m^2 kg C^{-1})	Projected leaf area per unit mass of leaf carbon.
froot_leaf_ratio	1.1	None	Ratio of fine root carbon to leaf carbon.
leaf_mr_base	0.00778	(kg C kg C^{-1} day^{-1})	Maintenance respiration per unit leaf carbon per day at 20 °C.
froot_mr_base	0.00519	(kg C kg C^{-1} day^{-1})	Maintenance respiration per unit root carbon per day at 20 °C.
livewood_leaf_ratio	0.203	None	Ratio of live wood carbon to annual maximum leaf carbon.
livewood_mr_base	0.00371	(kg C kg C^{-1} day^{-1})	Maintenance respiration per unit live wood carbon per day at 20 °C.

Table 2.2 Reference values of the Social Cost of Carbon (SCC) used for the economic assessment of CO_2 sequestration in TDF's.

SCC Initial value $ USD	SCC Updated value $ USD *	Author	Model
$13,62 USD (1995)/ t CO_2	$21.88 USD / t CO_2	Tol (2009)	FUND Meta-Analysis
$31 USD (2010)/ t$CO_2$	$34,54 USD / t CO_2	Nordhaus (2017)	DICE – 2016R
$50 - 100 USD (2005)/ t$CO_2$	$63,3 – 126,6 USD / t CO_2	Hope (2011)	PAGE09

* Values were updated to December 2017.

All values are express in US American Dollars ($ USD)

Table 2.3 Total estimated values of Gross Primary Productivity (GPP) and the partitioning into respiration (RA) and Net Primary Productivity (NPP) from years 2001 to 2015, values are expressed in tC ha^{-1} yr^{-1} for all TDFs sites, including Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU).

	CHA			YU			SR			MA			SM			TU		
YEAR	**GPP**	**RA**	**NPP**	**GPP**	**RA**	**NPP**	**GPP**	**RA**	**NPP**	**GPP**	**RA**	**NPP**	**GPP**	**RA**	**NPP**	**GPP**	**RA**	**NPP**
2001	10.1	2.7	7.4	9.0	3.5	5.5	12.6	3.2	9.4	10.7	2.8	8.0	3.1	1.6	1.5	3.0	1.6	1.4
2002	9.6	2.8	6.8	9.5	3.6	5.8	13.4	3.4	10.1	9.5	2.5	7.0	2.7	1.5	1.3	2.5	1.4	1.1
2003	8.6	2.6	6.0	9.3	3.7	5.7	11.1	2.9	8.3	10.8	2.8	8.0	3.9	1.8	2.1	3.7	1.8	1.9
2004	10.1	2.9	7.2	10.0	3.7	6.3	13.7	3.4	10.2	10.0	2.6	7.5	3.9	1.9	2.0	3.9	2.0	2.0
2005	9.6	2.5	7.0	9.7	3.6	6.1	11.5	3.0	8.6	10.2	2.6	7.6	3.5	1.7	1.7	3.7	1.8	1.9
2006	9.3	2.7	6.6	9.1	3.3	5.8	13.4	3.4	10.0	12.5	3.1	9.4	4.3	1.9	2.5	4.1	1.8	2.2
2007	9.4	2.6	6.8	8.9	3.5	5.4	11.9	3.0	8.9	11.2	2.8	8.4	4.2	1.8	2.4	4.4	1.8	2.6
2008	9.3	2.6	6.6	9.7	3.6	6.0	12.2	3.1	9.1	9.5	2.5	7.0	5.1	2.0	3.0	5.0	1.9	3.0
2009	9.5	2.8	6.7	8.3	3.2	5.1	14.2	3.6	10.6	12.8	3.2	9.5	6.4	2.2	4.2	6.8	2.3	4.5
2010	11.8	3.2	8.7	8.6	3.2	5.4	12.2	3.1	9.1	11.9	3.1	8.8	6.9	2.4	4.5	5.6	2.1	3.5
2011	9.6	2.8	6.8	8.4	3.3	5.1	12.2	3.1	9.2	12.0	3.0	8.9	7.4	2.4	5.0	6.6	2.3	4.2
2012	10.8	3.2	7.7	8.3	3.3	5.0	14.3	3.6	10.7	11.0	2.8	8.2	7.0	2.5	4.5	6.3	2.3	4.0
2013	9.5	2.9	6.6	9.2	3.3	5.9	12.3	3.1	9.2	10.0	2.7	7.3	6.4	2.3	4.2	5.8	2.2	3.6
2014	9.6	3.2	6.4	8.4	3.4	5.0	13.7	3.4	10.3	12.3	3.1	9.2	8.6	2.7	5.8	7.2	2.5	4.7
2015	11.9	3.6	8.3	9.1	3.3	5.8	14.4	3.5	10.8	9.0	2.4	6.6	6.3	2.5	3.8	6.1	2.5	3.6
Total Average	9.9	2.9	7.0	9.0	3.4	5.6	12.9	3.2	9.6	10.9	2.8	8.1	5.3	2.1	3.2	5.0	2.0	3.0
Total Standar Dev.	0.9	0.3	0.7	0.5	0.2	0.4	1.0	0.2	0.8	1.2	0.3	0.9	1.7	0.4	1.4	1.4	0.3	1.1

Table 2.4 Carbon Use Efficiency (CUE) values, or the rate of NPP to GPP (unitless), for all TDFs sites. Sites include Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU) from years 2001 to 2015. Total annual averages and standard deviations across all TDFs are also included.

Year	CH	YU	SR	MA	SM	TU	Annual Average
2001	0.73	0.61	0.75	0.74	0.49	0.49	0.64
2002	0.71	0.62	0.75	0.74	0.46	0.44	0.62
2003	0.70	0.61	0.74	0.74	0.54	0.51	0.64
2004	0.71	0.63	0.75	0.74	0.52	0.51	0.64
2005	0.74	0.63	0.74	0.75	0.50	0.50	0.64
2006	0.71	0.64	0.75	0.75	0.57	0.55	0.66
2007	0.72	0.61	0.75	0.75	0.58	0.59	0.66
2008	0.71	0.62	0.75	0.74	0.60	0.61	0.67
2009	0.71	0.62	0.75	0.75	0.65	0.66	0.69
2010	0.73	0.63	0.75	0.74	0.66	0.62	0.69
2011	0.71	0.61	0.75	0.75	0.67	0.65	0.69
2012	0.71	0.60	0.75	0.75	0.65	0.64	0.68
2013	0.69	0.64	0.75	0.73	0.65	0.62	0.68
2014	0.67	0.59	0.75	0.75	0.68	0.65	0.68
2015	0.69	0.64	0.75	0.74	0.60	0.59	0.67
Total Average	0.71	0.62	0.75	0.74	0.59	0.58	0.66
Total Standar Dev.	0.02	0.01	0.003	0.01	0.07	0.07	0.03

Table 2.5 Total annual CO_2 sequestration rates from years 2001 to 2015, values are expressed in t CO_2 ha^{-1} yr^{-1} for all TDFs sites, including Chamela (CH), Yucatan (YU), Santa Rosa (SR), Mata Seca (MA), San Matias (SM) and Tucavaca (TU). Total annual averages and standard deviations across all TDFs are also included.

Year	CH	YU	SR	MA	SM	TU	Annual Average
2001	27.0	20.3	34.4	29.3	5.5	5.3	20.3
2002	25.0	21.4	36.9	25.7	4.6	4.1	19.6
2003	21.9	20.7	30.3	29.4	7.7	6.8	19.5
2004	26.5	23.1	37.6	27.3	7.3	7.3	21.5
2005	25.8	22.3	31.4	27.9	6.3	6.8	20.1
2006	24.4	21.4	36.7	34.5	9.0	8.2	22.3
2007	24.9	20.0	32.6	30.8	8.8	9.4	21.1
2008	24.2	22.2	33.5	25.6	11.1	11.1	21.3
2009	24.7	18.8	38.9	35.0	15.3	16.5	24.9
2010	31.7	19.8	33.3	32.4	16.6	12.7	24.4
2011	25.0	18.6	33.6	32.8	18.2	15.5	24.0
2012	28.1	18.3	39.2	30.2	16.7	14.8	24.5
2013	24.2	21.7	33.7	26.9	15.3	13.1	22.5
2014	23.4	18.3	37.8	33.7	21.3	17.4	25.3
2015	30.3	21.2	39.7	24.3	13.9	13.3	23.8
Total Average	25.8	20.5	35.3	29.7	11.8	10.8	22.3
Total Standar Dev.	2.5	1.5	2.9	3.3	5.0	4.1	3.2

Table 2.6 Estimated monetary values of CO_2 sequestration (in $ USD per Ha) for all TDFs sites in this study, results consider three reference SCC for Tol, Nordhaus, and Hope.

		Tol (2009)	**Nordhaus (2017)**	**Hope (2011)**
Country	**TDF SITE**	**$ USD Ha^{-1}**	**$ USD Ha^{-1}**	**$ USD Ha^{-1}**
Costa Rica	Santa Rosa (SR)	772,5	1219,5	2234.8 - 4469.7
Mexico	Chamela (CH)	564,9	891,8	1634.3 - 3268.6
Mexico	Yucatan (YU)	449,5	709,6	1300.4 - 2600.9
Brazil	Mataseca (MA)	650,2	1026,5	1881.2 - 3762.3
Bolivia	San Matias (SM)	259,1	409,1	749.7 - 1499.3
Bolivia	Tucavaca (TU)	236,8	373,8	685.0 - 1370.1

* Values consider the mean of all years from 2001 to 2015.

All values are express in US American Dollars ($ USD)

Chapter three: Quantifying water provision services in a tropical dry forest: the case of the Guanacaste Conservation Area

3.1 Introduction

Water is fundamental for the functioning of all terrestrial ecosystems on our planet, but also to support human survival and livelihood (Martin-Ortega, 2015). Humanity relies on water resources not only for domestic consumption but also for the production of food, removal of wastes, production of energy, industry, and transportation among other uses (Haddeland et al., 2013). This wide range of benefits derived directly and indirectly from the structure and functioning of different ecosystems are defined as ecosystem services (ES) (Costanza et al., 1997). A concept that over the last decades has tried not only to influence public policy by providing evidence about the importance of natural ecosystems for human well-being (Costanza et al., 2015). But also to support sustainable management decisions through regulations and investments oriented to ensure the prevalence of this services in the future (Braat and de Groot, 2012). The importance of ecosystems to society was globally reported for the Millennium Ecosystem Assessment (MEA) in 2005, an extensive study from the United Nations Environmental Program (UNEP). A further accomplishment of the MEA was also the classification of the ecosystem services into three major categories, (1) provisioning or the products directly obtained from ecosystems such as food, water and other materials; (2) regulating services or the benefits derived from the regulation of ecosystem processes and functions such as air quality maintenance, climate regulation or water purification; and (3) cultural services or the nonmaterial benefits people obtain through spiritual enrichment, cognitive development, recreation or aesthetic experiences.

Although fresh water for human consumption is a fundamental service provided by ecosystems, a human being consumes between 1000 and 1300 m^3 of fresh water on a yearly base (Hoekstra, 2015), the most considerable benefit is the one related to the agriculture sector and the production of food. In this context, irrigation represents the primary use of fresh water worldwide (Bates et al., 2008). According to the Assessment of Water Management in Agriculture (2007), up to 85% of total freshwater is used in agriculture in developing countries. The proportion of fresh water used is likely to increase in future years given the overgrowing population demands. Moreover, the MEA (2005) also reported that approximately 5% to 25% of global freshwater use exceeds long-term accessible water supplies, and as a consequence, ecosystems have been dramatically modified in order to meet these demands. The development of infrastructure, agricultural expansion, water withdrawal, pollution, the introduction of invasive alien species, overharvesting and pesticide leaking, are just some of the primary drivers for the degradation of water-related ecosystems (Grizzetti et al., 2016). In a global scale, impacts on water resources will more likely be exacerbated with increasing climate change effects (Haddeland et al., 2014) and as a consequence water stress and reliability over the next decades are becoming a matter of growing concern (Bates et al., 2008). Against this background, ecosystem service-based approaches emerge as a novel way of understanding the complex interactions between ecosystems and water cycling. But also, to address the intrinsic relation of water services and human well-being for the support of more informed policy and decision making processes (Burkhard et al., 2013).

The increasing interest of decision makers and stakeholders for more reliable and convenient water assessment and management approaches has lead the scientific community to discuss how to link the complex functions of ecosystems and water cycling at a catchment level to the final quantification of goods and services (Fisher et al., 2009), but also to be able of integrate ecosystem capital into national accounts (SEEA, 2012). Although previous research has focused in the hydrological (Thompson, 2017) or the ecological (Ritcher et al., 2003) domain of water resources, there is an emerging group of quantitative assessment tools for measuring ES related to water resources (Grizzetti et al., 2016). In a revision of the applicability of thirteen tools for multiple ecosystem services assessments, Bagstad et al. (2003) found that the selection of the tool is hugely related to the decision making context for which an assessment is needed. Additional elements were also the economic feasibility of its implementation, the flexibility for running a model and accessing to documentation, the accessibility to information and parameterization inputs, and the scalability for multiple spatial requirements.

The potential of an ecosystem service-based approach for assessing water provision

According to Martin-Ortega (2015), an ecosystem service-based approach is a way of understanding the complex relationships between ecosystems and humans to support decision-making and to reverse the declining status of ecosystems. The central cores of this approach are the consideration of the effects on human wellbeing, the biophysical underpinning of service delivery and the transdisciplinarity between both perceptions for policy making (Ninan, 2014). In a water context, an ES approach must recognize the influence of forest structure and soil-vegetation processes on water availability, as well as the impact of anthropogenic activities and potential uses (Grizzetti et al., 2016). Linking these components represents a fundamental challenge and may provide uncertain results depending on the tool to be used (Bagstad et al., 2013).

A critical component in an ES based approach is the spatial explicitness of the services that ecosystems provide (Fisher et al., 2009). This means that ES has the capability to be directly represented in a geographical space as a set of discrete units (Haslauer et al., 2015). Furthermore, the framework of the System for Environmental Economic Accounts (SEEA) determinates that ES are spatially heterogeneous and that its spatial variability must be capture for final accounting. In this context, most spatially explicit approaches represent different ES variables as polygonal or equal-area raster cells. The main strength of models that include this type of approach is that results allow users to identify spatial patterns and processes. A valuable advantage for the comprehension of the complex dynamics of ecosystems and the services they provide.

Spatial modeling of ES rises as a crucial method to synthesize the complex functions and dynamics of ecosystems (Burkhard et al., 2012). Integrating spatial analysis techniques with comprehensive sets of indicators and metrics allows a full understanding of the multiple types of ES that an ecosystem can provide (Egoh et al., 2012). Moreover, essential distinctions regarding the complexity of the models, number of variables, interacting processes, resolution, and computational requirements must be taken into account (Seppelt and Voinov, 2002). Considering all these elements will improve the comprehension of the uncertainties involved in the modeling of ES (Costanza and Voinov, 2004).

Several models and indicators were designed for assessing the provision of water services. Most of these models usually select water yield as the main ES indicator and also use water basins as the minimal spatial units for spatial representation (Crossman et al., 2013). There are three types of models used to assess the provisioning of water services and are classified based on their complexity. The most straightforward models, also defined as Tier 1 models, rely on basic basin-scale water balance functions (Zhang et al., 2001). Although these type of models are more accessible to apply in a restricted data environment, they present some constraints regarding the certainty of the analysis. Following in order, and defined as Tier 2 models, are the models that simulate hydrological variables by linking additional processes and land use and land cover (LULC) information. Examples of these type of models are usually included in larger multiple ES modelling suites, such as the Integrated Valuation for Ecosystem Services and Tradeoffs (InVEST) of the Natural Capital Project (Tallis and Polasky, 2009), the ARtificial Intelligence for Ecosystem Services (ARIES) project (Villa et al., 2009) or Polyscape (Jackson et al., 2013). Finally, Tier 3 models represent the highest level of complexity for water ecosystem services modeling. These type of models are process-based since they estimate the quantity of locally available water by explicitly connecting major hydrological and biophysical variables (such as relative humidity, wind, solar radiation, soil and others). Anthropogenic influences (such as land use, harvest times, pesticide management and others) are also considered in this type of models. Simulation results are also provided with a wealth of spatial and long-term resolutions.

The Soil and Water Assessment Tool (SWAT) is a Tier 3 model developed and implemented by Arnold et al. (1998) and used for some applications since then. Although hydrological modeling is the original function of SWAT, according to Francesconi et al. (2015) on a systematic review of SWAT studies for ES quantification, an increase in the use of the model for this purpose has been registered in the last years. Essential advantages of SWAT are also related to its capability to simulate direct values of provisioning and regulating services, or proxies of cultural services in the ES water context. However, they also emphasize that the high requirements for data inputs and time processing, as well as the impossibility to include a monetary valuation, are some of the challenges that users of the model might take into account.

Ecosystem services in the Guanacaste Conservation Area

Costa Rica has been a forerunner in recognizing the importance of ecosystem services among other notable conservation policies. In 1996 the country enacted the Forestal Law No.7575, which recognized four primary environmental services provided by forest ecosystems: (i) mitigation of greenhouse gas emissions; (ii) protection of hydrological services for different purposes, including drinking water, irrigation, and energy production; (iii) biodiversity conservation and sustainable use; and (iv) provision of scenic beauty for recreation and tourism. Consequently, the Fondo Nacional de Financiamiento Forestal (FONAFIFO) and the Payments for Environmental Services (PES) Program were created to implement the administration of economic incentives given to landowners who own forestall lands for the protection of all four ecosystem services detailed in the Forestal Law. Though this program did not have a direct impact on the reduction of deforestation rates (Sanchez-Azofeifa et al., 2007) or the reduction of poverty (Pagiola, 2007), it became a promising initiative. Most studies related to ecosystem services in Costa Rica have focused on assessing its impacts and development (Sanchez-Azofeifa et al., 2007; Pagiola, 2007; Kalacska et al., 2008; Locatelli et al., 2014).

The Area of Conservation Guanacaste (ACG) is one of eleven conservation areas in the lowlands of Costa Rica's Pacific coast and embraces one of the most successful initiatives of forest restoration in the last decades (Allen, 2001). Given its important biodiversity and different ecosystem types, the ACG provides a wide range of ecosystem services (Figure 3.1). According to the MEA classification (2005), these can range from provisioning services such as timber, biomass-based energy, water provision and food production; regulating services such as carbon sequestration, control of pollution, pollination, soil fertility cycling, flood and sediment control; and cultural services such as scenic beauty, recreation and the basis for scientific research among others. In Guanacaste, for a number of years, academics have focused in understanding the complex dynamics of forest ecosystems (Kalacska et al., 2004; Sanchez-Azofeifa and Portillo, 2011), ecological functions (Quesada et al., 2004, Sanchez-Azofeifa et al., 2009) and processes (Leiva et al., 2009). However, until today there has not been a profound understanding of the key hydrological processes that involve the provision of water ecosystem services (Calvo-Rodriguez et al., 2016), its spatial organization or the relevance that these can have on local and national well-fare.

Given this context, this study aims to assess the biophysical capacity of the Guanacaste Conservation Area (ACG) in Costa Rica for the supply of water provision ecosystem services. In this analysis the main questions to be answered are: (1) how can we quantify the provision of water ES? (2) what is the access of stakeholders to water provision services?, and (3) how reliable is the provision of water for human well-being?

By providing a range of biophysical estimates of water supply, I hope to provide a basis for future scenario analysis of water resource management, synergies and trade-offs with other ecosystem services and to satisfy the needs of stakeholders for better decision making processes.

3.2 Methods

3.2.1 Study area

The study was conducted at the Guanacaste Conservation Area (heretofore ACG for its Spanish abbreviation). ACG is a conservation area located in the Pacific coast of northern Costa

Rica between the administrative areas of La Cruz and Liberia in the province of Guanacaste, and Upala in the province of Alajuela (10°44′7.2″N–10°57′16.4″N, 85°34′43.2″W–85°57′4.8″W). The site is one of eleven conservation areas in Costa Rica and has emerged as a backbone in forest restoration history (Janzen, 1988; Allen, 2001). It is also integrated by a number of protected areas that represent approximately 2.4% of the world's terrestrial biodiversity and also possess 60% of all the biodiversity in Costa Rica (Janzen, 2000). The essence of this biogeographical region (Figure 3.2) is a continuous area of 3451.8 km^2 that extends from the marine pacific coast integrating three major tropical forest ecosystems: dry forest, rainforest and cloud forest (ACG, 2016).

Mean annual temperatures range from 26.6 °C to 27.5°C. Although precipitation is yearly variable among ecosystems (Janzen and Hallwachs, 2016), mean annual precipitation ranges from 1390 mm to 1800 mm. Water becomes a limiting factor during the dry season (from December to April) mainly at the seasonally neotropical dry forest (Sanchez-Azofeifa et al., 2005). Vegetation is composed by a mosaic of mature and secondary forest under various stages of succession (early, intermediate and late) (Janzen, 2000; Kalácska et al., 2004; Arroyo-Mora et al., 2005). Forest areas integrate south of the conservation area with different land use areas for agriculture and cattle ranching, giving birth to what is defined as the agro-landscape (Cao and Sanchez-Azofeifa, 2017). No significant land cover changes have been reported since the 1980's (Calvo-Alvarado et al., 2009).

According to the Direction of Water of the Ministry of Environment and Energy (MINAE, 2017), the Tempisque river constitutes the major basin in the area since this basin covers approximately 53% of Guanacaste Province and drains 10.6% of the national territory, hence its importance for the sustaining of major anthropogenic activities, such as agriculture, fishing, and cattle ranching. Following in importance are the Tempisquisto, Blanco, Liberia, and El Salto basins. Topographic origin of all streamflows in the area relates to the Volcanic Cordillera of Guanacaste with elevations that range between 500 y 1916 meter height (Janzen and Hallwachs, 2016).

3.2.2 Hydrological modeling

To address question one of this study, I selected the Soil and Water Assessment Tool (SWAT), a computationally efficient and spatially explicit hydrologic model (Arnold et al., 1998) developed to simulate hydrology at different scales. SWAT is also a physically based model that takes into account meteorological parameters, topography, vegetation and the impact of land management practices and soil characteristics over long and continuous periods of time in a watershed (Douglas-Mankin et al., 2010). Main components of SWAT for ecosystem services assessments are hydrology, climate, nutrient cycling, sediment movement, crop growth, water quality and pesticide dynamics among others (Jayakrishnan et al., 2005). For this study, I selected water yield as the ecosystem service indicator. Water yield is defined as the net amount of water that leaves the basin and contributes to streamflow in the reach during a time interval (Ullrich and Volk, 2009) and is calculated in the model as expressed in equation (1).

$$\mathrm{WYLD} = \mathrm{PREC} - \mathrm{SURQ} - \mathrm{GW} - \mathrm{PET} - \mathrm{ET} - \mathrm{SW} \tag{1}$$

Where;

WYLD: Water yield (mm of H_2O)

PREC: Amount of Precipitation in mm

SURQ: Amount of Surface runoff in mm

GW: Groundwater Contribution in mm

PET: Potential Evapotranspiration in mm

ET: Actual Evapotranspiration in mm

SW: Soil water content (mm)

The model simulates potential evapotranspiration (PET) using the Hargreaves method (Hargreaves and Samani, 1982). Actual evapotranspiration (AET) is predicted based on the methodology developed by Ritchie (1972). The daily value of the leaf area index (LAI) was used to partition the PET into potential soil evaporation and potential plant transpiration. A more detailed description of the model is given by Neitsch et al. (2011). For computational running of the hydrological model, I used QSWAT (Bansode and Patil, 2016), where Quantum GIS (QGIS version 2.6) and SWAT Editor (version 2012) were used to generate the estimations.

Model inputs for parameterization (Figure 3.3) were compiled from different sources. Topographic data were obtained from a digital elevation model generated at 10 meters of resolution by the Fondo de Financiamiento Forestal de Costa Rica (FONAFIFO). A land use map of 30 meters resolution was created at the Centre for Earth Observation Sciences (CEOS) of the University of Alberta by classifying a LANDSAT satellite image for the year 2015, six land use classes were identified and are described in Table 3.1. Training sets included the selection of 300 training areas for all land covers that were used for image classification. Classification accuracy assessment was then conducted by comparing a set of 69 ground GPS reference points with final classification outputs. Soil information was obtained from the digital soil map of Costa Rica created in 2013 for the Centro de Investigaciones Agronomicas of the University of Costa Rica at a scale of 1:200.000 and based on the US Soil Taxonomy classification, from which 12 soil types were identified for the study area (Table 3.2). Using the percentages of sand, lime, and clay of the soil profiles, I calculate the SWAT soil parameters utilizing the Pedo Transfer Function (PTF) developed by Saxton and Rawls (2006). A time series of daily weather data from 1985 to 2013 for precipitation, temperature, relative humidity, wind speed and solar radiation for six stations (Table 3.3) were obtained from the Climate Forecast System Reanalysis (CFSR) (https://globalweather.tamu.edu), monthly distribution for precipitation and temperature are shown in Figure 3.5. The CFSR was designed and executed as a global, high resolution, coupled atmosphere-ocean-land surface-sea ice system for the National Center for Environmental Prediction (NCEP). Successful applicability of the CFSR data for SWAT was also explored in previous studies (Fuka et al., 2013; Dile and Srinivasan, 2014).

The SWAT Model was setup for a drainage area of 3090 Km^2. I set up a warm-up time of three years with a threshold equivalent to 2% of the total area according to Bansode and Patil (2016). This resulted in a subdivision of the watershed into 105 sub-basins which were characterized equally by dominant soil, land use and slope for a simulation period of 29 years (from 1988 to 2013). The detailed model framework is presented in Figure 3.4. Spatial distribution and hydrological balance values for each sub-basin were plotted with the use of Geographic Information Systems. Sub-basins are the central geographic unit used in the analysis.

3.2.3 Double-mass curves

The lack of validation data (e.g., river streamflows) to compare with simulation outputs is the main limitation of this study. However, Srinivasan et al. (2010) proved that running SWAT without calibration is feasible for exploratory studies. An alternative to overcome these limitations is the use of statistical tools to correlate input data sets of the model. In this context, I selected a Double Mass Curve analysis (Kohler, 1949) to compare the rainfall records of the meteorological stations (global) used as an input in SWAT (Table 3.3 and Figure 3.5) against the rainfall records of three stations (local) in the area. Three pairs of stations, each pair composed by one global and one local station, were selected for the correlation analysis based on the geographical proximity at the study area. The stations chosen for the Double Mass Curve analysis are described in Table 3.4, and their geographic location is presented in Figure 3.6.

The Double Mass Curve is a statistical analysis that can be performed by using a graphical method (Buishand, 1982). The analysis relies on the principle that the accumulation of rainfall at one station against the accumulation of rainfall at other station over the same time period will plot as a straight line as long as the data is proportional. To generate the graphic analysis for each pair of stations (105-856/SR, 111-856/LCC, and 111-853/FBDO), I first exclude all years that were not available at both stations to be compared, and then I accumulate the annual rainfall data for each pair of compared stations and finally plotted as a statistical linear regression in a graph. A straight line represents an extensive relation and more similarities between the compared data (Gao et al. 2017). Poor correlation and jumps in the statistical trends are evidence of inconsistencies in the records and can frequently be due to factors as the type, height and exposure of the instrument used to measure the precipitation. Furthermore, an increase in the length of a trend despite some discrepancies tends to offset the effect of a bad correlation (Müller and Thompson, 2016).

3.2.4 Water per capita calculation

Question two of this study was answered by integrating population metrics and water provision for each sub-basin in the watershed. For this, I use a 2.5 arcmin gridded map of the Population of the World from the Center for International Earth Science Information Network (CIESIN) in its fourth version (GPWv4) with information updated to July 2016 (http://sedac.ciesin.columbia.edu/data/collection/gpw-v4). In order to approximate population for each sub-basin, I used a spatial allocation approach in a GIS interface. Finally, estimates of water per capita were calculated, and expressed in m^3 hab^{-1} yr^{-1}, by dividing total annual water yield simulated at each sub-basin between the number of allocated habitants (Faramarzi et al., 2009). The number of people living within a sub-basin provides a first-order estimate of the water demand of the ecosystem service (Parish et al., 2012). Furthermore, this study does not attempt to account for uncertainties in population metrics.

3.2.5 Coefficient of variation

Finally, to address question three of this study, I estimate the coefficient of variation (CV) of simulated water yield from 1988 to 2013 for each sub-basin in the watershed. The CV is an indicator that expresses how reliable is the provision of water resources from year to year given its historical records (Faramarzi et al., 2009) and is given by equation 2.

$$CV = \frac{\sigma}{\mu} \times 100 \tag{2}$$

Where;

CV: Coefficient of variation in %

σ: Standard deviation of annual water yields

μ: Mean of annual water yields

As explained by Faramarzi et al. (2009), large CV values indicate areas that are experiencing extreme weather conditions and consequentially less reliability for the provision of water.

Considering the effect of extreme climatic events in the reliability of the provision of water, I estimate the CV for six different scenarios: (a) all years from 19888 to 2013, (b) only normal years excluding El Niño and La Niña years, (c) all normal years including El Niño years, (d) all normal years including La Niña years, (e) only El Niño years and (f) only La Niña years.

The selection of El Niño, La Niña and normal years was made by using The Oceanic Niño Index (ONI), a 3-month mean of the surface sea temperature (SST) anomaly for the Niño 3.4 region (NOAA, 2017), a standard commonly used by the National Oceanic and Atmospheric Administration (NOAA) for identifying El Niño (warm) and La Niña (cold) events in the tropical Pacific.

3.3 Results

Validation of weather stations

Results from the Double Mass Curve analysis for the three coupled sets of stations are displayed in Figure 3.7. The selection was carried out by geographical proximity and restricted only to the time interval available in both of the stations to be compared. Overall results showed good consistency between the stations used as an input for the simulation of the model (111-856, 111-853 and 105-856) and the records at the local stations used for validation (LCC, FBDO, and SR). Slight scattering of points at sides of the slopes was found not to affect the overall consistency given the R^2 coefficient results.

Individual correlations for stations 111-856 and LCC, located in the NW of the catchment area, for the years 1990 to 2008 presented the lower consistency (R^2=0.97) with mean annual precipitations of 1532.7 mm for 111-853 and 1367.1 mm for LCC. Followed in order, the comparison between stations 111-853 and FBDO, located at NE of the catchment area, was fair for the years 2003 to 2014 (R^2=0.98) presenting mean annual precipitations of 3012.8 mm in FBDO and 2393.3 mm in 111-853. Finally, the stronger consistency of validation was found for stations 105-856 and SR (R^2=0.99) at the center part of the study area with mean annual precipitations of 1782.5 mm for 105-856 and 1193.6 mm for SR. Furthermore, monthly precipitation patterns doesn't differ greatly from one to another during graphic comparisons (Figures 3.5 and 3.6) which supports the consistency between stations and their further use in the modelling stage.

Quantification of water provision

Results of the simulation of annual water yield for 105 sub-basins included in the model were estimated from years 1988 to 2013. The total catchment area was divided into four major watersheds (Figure 3.8) based on drainage orientation; these are Nicaragua Lake, Santa Elena Bay, Papagayo Gulf and Tempisque River.

To understand the link and/or influence of human activities in the provision of water yield, results of an overlapping spatial analysis of land use areas for all major watersheds are described in Table 3.5 and Figure 3.9. Results show that forest use is predominant in the Nicaragua Lake (73.9%), Santa Elena Bay (72.9%) and Papagayo Gulf (74.8%) watersheds and although forest also represent the principal use (55.9%) in the Tempisque River watershed, there is a higher diversification of other land uses such as agriculture (17.2%), pasture (24.9%) and essential urban areas (1.9%).

For a general overview of the hydrological components of the water quantification, results are presented as annual average values from 1988 to 2013 at the sub-basin level, which was used to characterize the spatial distribution of various components of the water balance, such as precipitation, soil water content and final water yield. Estimation of the water balance at general level for the ACG catchment area defines an annual average precipitation of 2502 mm yr^{-1}. Results can be partitioned into a mean annual evapotranspiration of 795.5 mm yr^{-1}, a mean annual water content in the soil of 218.6 mm yr^{-1} and a final mean annual water yield of 1487.9 mm of water. This represents a 59% of the total incoming precipitation running as water yield for provisioning services. The spatial distribution of the annual average precipitation is shown in Figure 3.10; higher precipitation rates are localized in the east side of the study area decreasing toward the west, the NW region is the one with lower precipitation rates. Historical annual variation of simulated precipitation from 1988 to 2013 shows a decrease in incoming precipitation for the last years starting from 2010. Monthly average estimates showed two marked seasons a wet season (May to October) and a dry season (November to April). Annual and monthly precipitation averages are presented in Figure 3.11.

Historical averages of annual soil water content along with the spatial distribution are also presented in Figure 3.12. In contrast to precipitation trends, soil water content presents more regular annual rates that fluctuate between 170 - 270 mm yr^{-1} with major spatial concentrations towards the east part of the study area, these also represent areas with more abundant soil moisture. Spatial distribution of simulated water yield is also presented in Figure 3.12, sub-basins with larger annual water yield are located on the east side of the ACG, overlapping with areas of more humidity and incoming precipitation. Sub-basins situated in western regions of the ACG are the ones who provide lower annual water yield rates which overlap with dry forest areas. Furthermore, sub-basins at the northwest of the study area presented the lowest simulated water yield. The historical annual water yield trend in Figure 3.12 shows a variable behavior in the provision of the service, higher water yield peaks over the 90's contrast to decreasing quantities towards the years 2012 and 2013. To further illustrate the efficiency of the provision of the water service, a ratio of annual water yield to total annual precipitation was estimated (Figure 3.13). The ratio shows the proportion of incoming precipitation that remains as water yield after losses through ETP and retention in the soils. The slope of the relationship was 0.59 (R^2=0.94), this means that 59% of all incoming precipitation flows as water yield, results also showed variations from 0.48 to 0.68 among all sub-basins.

To present results at a secondary level of aggregation, Table 3.7 describes all components of water balance for the four major watersheds previously identified. By contrasting all major watersheds, the Tempisque river watershed stands as the area with most substantial annual water yield simulated (1534.3 mm yr^{-1}), followed in order of importance by Papagayo Gulf (1477.6 mm yr^{-1}), Nicaragua Lake (1435.7 mm yr^{-1}), and Santa Elena Bay (1405.2 mm yr^{-1}).

Estimation of water per capita at the sub-basin level

To highlight the potential of the ACG for the provision of fresh water for human consumption, this study presents estimates of the annual per capita freshwater availability in each sub-basin. The spatial allocation of the human population at the study area was expressed in number of habitants per sub-basin and shown in Figure 3.14, where sub-basins in the southern region were identified as the ones with higher population density in contrast to lower densities in the rest of the catchment area. As calculated for the entire region and integrating total annual water yield in m^3, (calculated using the average water yield of all years from 1986 to 2013), a mean estimate of 11357.6 m^3 hab^{-1} yr^{-1} for the ACG region was calculated. Estimated values range from 696 m^3 hab^{-1} yr^{-1} to 50866 m^3 hab^{-1} yr^{-1} among all sub-basins. Results of water per capita at the sub-basin level are presented in Figure 3.15. The spatial distribution of the results identified sub-basins with more water per capita in the west and central regions of the study area, areas that have relatively lower population density. In contrast, sub-basins with lower water per capita are concentrated in the southern and upper north part of the study area as they keep the higher population densities in the ACG.

Estimation of the reliability in the provision of the service

To further illustrate annual variations and reliability in the provision of water yield from years 1986 to 2013, the coefficient of variation (CV) was calculated by using six historical scenarios, selected by inclusion or exclusion of years with El Niño and La Niña events. Estimates at the sub-basin level and spatial distribution are presented in Figure 3.16. Lower percentages of CV represent more reliability, and higher percentages of CV represent less reliability in the provision of the service.

The first scenario was estimated by including all years in the historical trend (from 1986 to 2013). Overall CV values range from 32 to 70%; results show that regions at the north-west of the study area have less reliability for the provision of the service. The second scenario was estimated by excluding all El Niño and La Niña years. In normal conditions, the CV values range from 18 to 44% of reliability where areas with less reliability are located on the northeast side of the ACG. The third and fourth scenarios were calculated by a combination of all normal years plus el El Niño years and all normal years plus La Niña years respectively. Results for both scenarios are spatially similar identifying the western sub-basins as the ones with less water reliability but with CV's that range from 27 to 72% for the third scenario and 29 to 57% for the fourth scenario. Fifth and fourth scenarios were calculated by selecting extreme years for each event. The fifth scenario estimated that the reliability for a series of just El Niño years ranges from 34 to 97 % having major impacts in the western region of the study area. The sixth scenario estimated reliability for a series of just La Niña years, in this sense CV values, range from 28 to 52% with higher impacts in the northern and north-west regions of the study area.

To highlight the variability of the results, a statistical distribution of the CV values for each scenario is presented in Figure 3.17. Median values and variations show more reliability in a series of just normal years (second scenario with 28%) and less reliability in scenarios which include effects of El Niño years (fifth scenario with 61% and third scenario with 41%). Reliability with the incidence of La Niña years has relatively low effects on the provision of the service (38% for fourth and sixth scenarios). The final results showed that the provision of water is relatively less reliable by taking in normal conditions, which means by taking into account the incidence of El Niño and La Niña years (47% for the first scenario).

3.4 Discussion

This study illustrates the use of an ecosystem service-based approach for the quantification of the water provision service at the ACG. The proposed approach emphasizes the integration of hydrological modeling tools (using SWAT) and the spatial explicitness of ecosystem services to simulate the supply of water for human welfare. Main results suggest that the ACG has an important potential for the provision of water for human populations, an approximated water yield of 1487.9 mm yr^{-1} with an annual water per capita of 11357.6 m^3 hab^{-1} yr^{-1}, and although the estimated reliability in the provision of the service seems to be good this might be affected by extreme climatic events such as El Niño. While this study is not meant to forecast an accurate estimation of the hydrologic components of the water balance, it did intend to provide a baseline for future research in water ecosystems services for the region and to support more informed decision-making processes.

Hydrological modeling of water provisioning services

Although there is a rapid evolution of tools developed to map and model ecosystem services in general (Crossman et al., 2013), few of them are oriented to incorporate the complexity of the hydrological components and its relation with biophysical functions or its spatial distribution, since most are too general, lack spatial representation or are very ES-specific. For example, in a previous study of thirteen ES tools, Burkhard et al. (2013) showed that depending on the tool selected hydrological estimations can vary broadly. Moreover, ES models like InVEST (Tallis et al., 2013) and ARIES (Villa et al., 2009), focus specifically on the assessment of particular ES such as wave energy, coastal vulnerability, erosion protection, marine fish aquaculture, esthetic quality, fisheries and recreation overlap, marine water quality or hydroelectric supply. Aditional service models, like water provision, are still in active development (Crossman et al., 2013). Other tools like Co$ting Nature (Mulligan et al., 2015) that use pre-loaded global datasets at 1 km^2 or 1 ha of resolution to quantify water, also proved to generalize the water balance and focus more on the econometric component of the provision. Hence, interest has focused on expanding the utility of hydrological modeling tools, such as SWAT, to conduct ES assessments (Vigerstol and Aukema, 2011). Although SWAT might be considered a traditional hydrologic model not explicitly linked to ecosystem services, previous research has demonstrated its utility to estimate several variables at different spatial scales and for long time intervals. SWAT outputs can also be used as a base for economic valuation (Arias et al., 2011), the very next step of ES assessment (Swallow et al., 2009; Francesconi et al., 2016; Karabutul et al., 2016).

Regarding the selected tool for this study, the simulated outputs were satisfactory to described water movement at all locations (105 sub-basins), temporal scale (from years 1986-2013) and for all main variables of the water cycle (from precipitation, evapotranspiration, soil water content and water yield for human uses). Results were consequently used to integrate with population metrics. The main limitation regarding the application of the model was the lack of local data for calibration and model testing since, to improve model performance, SWAT requires hydrologic flow data and significant calibration to ensure that the hydrologic processes simulated are accurate (Arnold et al., 2012). However, in a previous study for the Mississippi river basin, Srinivasan et al. (2010) proved that running SWAT without calibration is feasible to perform at the annual scale and by validating input or output variables with local data. In consequence, validation constraints were addressed by testing the validity of input rainfall data through the Double mass curve analysis, and as expected the three validation procedures produced good correlations. Regarding these results, the comparison showed to be more consistent at 105-856/SR then for the other two sets (111-856/LCC and 111-853/FBDO) which presented some scattered points at sides of the trend lines. According to Kohler (1949) breaks in slope could be given by changes in the constant proportionality of the data. However, for Gao et al. (2017) these breaks do not necessarily indicate inconsistency in the comparisons, but might be related to meteorological causes, gage location, observational methods, changes in exposure and others. SR has a manual reading of the daily records whereas LCC and FBDO use an automated method in which records sometimes presented sudden underestimations, these differences in the observational methods explain the scattering of some points in the analysis. This study aimed to estimate a proxy water yield for the assessment of water provision and not an accurate estimation of the hydrological water balance. Future challenges for improving the model performance of this work, and by managing to collect hydrologic flow data, are the testing, calibration, and validation of SWAT results by using some friendly tools developed to make this process easier (Arnold et al., 2012).

From the spatial outputs of the model, results showed the importance of quantifying ecosystem services in spatially explicit units (Crossman et al., 2015). As pointed by Fisher et al. (2009) supply and demand of ES may differ geographically. This principle is critical for the analysis of water ecosystem services, where for example the provision of water is usually given in the higher parts of a basin, and final beneficiaries might be located at different stages of the flow. Therefore, the chance of identifying vital geographical regions might be used for ensuring the supply of water through various planning strategies (Karabulut et al., 2016). But also to preserve those areas identified as fundamental for the provisioning of water (hotspots) (Schröter et al., 2016.) or to determine trade-offs and synergies in multiple ES assessments (Haines-Young et al., 2013). Distinguishing between mapped and tabulated results for a stakeholder might be more comfortable with illustrated communication tools. In this context, mapping delivery and demand of ES can then be used to communicate and support decision making at different stages.

Water provisioning in the Guanacaste Conservation Area

For Guanacaste, previous reports have examined different estimates of the water balance. In a statement of Centro del Agua para América Latina y el Caribe (Morales, 2010) annual precipitation of 2721 mm yr^{-1} and annual ETP of 983 mm yr^{-1} were estimated for the north and central Pacific region of Costa Rica. According to the last meteorological report of the Instituto Metereologico Nacional de Costa Rica (IMN, 2009) an annual precipitation that ranges between 1500 to 2000 mm yr^{-1} and annual ETP between 1300 and 1800 mm yr^{-1} were also defined for the lowest and drier western regions of Guanacaste. The same report describes an annual precipitation between 2000 and 3000 mm yr^{-1} and a yearly ETP of 1200 mm yr^{-1} for the humid areas in Guanacaste, close to volcanoes like Orosi and Rincon la Vieja. No water yield calculations were found at any of the reports. The simulated results of this study fall between these references, with predicted annual precipitations of 1766 to 2570 mm yr^{-1} at west and 2575 to 2947 at east. The predicted ETP showed to be underestimated (634 to 939 mm yr^{-1}) regarding national reports. Reasons for underestimation of ETP may be related to simulation of the water cycle in the soil profile and parameters such as soil conductivity (Arnold et al., 2012), elements that can be corrected with proper local data calibration.

There is strong spatial variability in the provisioning of water yield across the ACG. The sub-basins with larger water yield are located at the East side of the region. Two major watersheds receive higher water world contributions, the Tempisque River watershed at the south-east and the Nicaragua Lake watershed in the northeast. In contrast, the Santa Elena Bay watershed corresponds to the driest area, which overlaps with the most extensive tropical dry forest extension at the ACG. The importance of the tropical dry forests was highlighted by Sanchez-Azofeifa et al. (2005). Mainly because of their capability to encompass high biological diversity (Kalacska et al., 2004), regrowth potential (Janzen, 1988) and supporting of other major ecosystem services (Maass et al., 2005; Calvo-Rodriguez et al., 2016).

A high proportion of water yield is provided by the Nicaragua Lake watershed (approx. 1435 mm yr^{-1}). However, particular attention must be taken to the provisioning of water at the sub-basins that are part of this watershed since are all part of a transboundary catchment area. This means that all the rivers that flow this watershed will finally contribute to Nicaragua Lake, and do not only contribute to the welfare of populations within the boundaries of the ACG but also to other populations in the neighboring country Nicaragua. For Vogtmann and Dobretsov (2005) governance is a complex issue concerning transboundary waters since it requires the implementation of international agreements and integrated management strategies for sustainable use and development. The Nicaragua Lake transboundary basin constitutes the most significant freshwater reserve in Central America (Huete-Perez et al., 2015) and is fundamental not just for the prevalence of human populations but also for related terrestrial ecosystems and biodiversity. In recent years the possible construction of a channel that links the Pacific and Atlantic oceans through the Nicaragua Lake raised alerts in the scientific community, given the negative repercussions that this infrastructure might have on terrestrial ecosystems which rely on this source of water. Economic and social impacts are also likely to occur as a consequence of ecosystem service degradation (Huete-Perez et al., 2015).

This study also revealed the importance of the Tempisque River watershed for the provision of water, mainly because it presents the higher potential for water yield and efficiency in the ACG region but also for the close link with human beneficiaries and potential uses. Arias and Calvo-Alvarado (2012) also determined the importance of the Tempisque river watershed when they identified 164 granted water concessions, the declared purpose of these concessions corresponds in its majority to irrigation (70.4%), followed by agroindustry (28.6%), municipal and other human consumption systems (1.1%). The close link to human benefits of this watershed was also validated when analyzing current land use at each basin where large proportions of agriculture, pasture, and urban uses were found. As pointed by Cao and Sanchez-Azofeifa (2017) the southern part of the ACG, where the Tempisque river watershed is, is considered the birthplace of the "agro-landscape." An area associated with a history of deforestation and restoration of TDFs areas (Calvo-Alvarado et al., 2009), cattle ranching and agricultural activities and also the location of more important populated towns and touristic activities (Sanchez-Azofeifa and Portillo-Quintero, 2010). Since water provision is a core component of both human well-being and the thriving economy of the Tempisque area, establishing proper conservation and regulation policies is needed. Payments for environmental services (PES) are one of the strategies that Costa Rica has implemented to create incentives for the conservation of forested areas (Sanchez-Chavez and Navarrete-Chacon, 20017). From years 2006 to 2015 some 738 PES contracts for a value of USD 14.830.040 were signed by FONAFIFO to protect 38871.6 hectares of areas considered to be hydrologically important. Future challenges of this study are the use of its outputs to provide spatially explicit information about the identification of important provisioning areas for hydrological PES payments.

Furthermore, by comparing estimates of water per capita at the ACG (approximately 11 357 m^3 hab^{-1} yr^{-1}) with the recently compiled Environmental Accounts for Water, Forestry and Energy of Costa Rica (BCCR, 2017), outputs suggest an underestimation regarding their last report (BCCR, 2017). They estimated an annual water per capita of 22 883 m^3 hab^{-1} yr^{-1}. Since 2013, the Costa Rica Central Bank (BCCR) in coordination with the World Bank through the initiative Wealth Accounting and Valuation of Ecosystem Services (WAVES) has tried to quantify the physical and economic value of water resources and their relevance to the national accounts. Differences in estimation are given by differences in the selected approaches, while the BCCR approach combines statistical metrics at the national level and a numeric water balance (Perales et al., 2014), this study selected a more accurate hydrological process-based water model. Moreover, the BCCR approach lacks a spatial approach from which spatial comparisons could be conducted. In this sense, the capability to extrapolate the proposed approach to larger scales, such as national, would benefit future integration with environmental accounts.

Water provisioning is not only significant to natural resource management, but water stress, scarcity, and reduced accessibility could cause future social conflicts. In this context, the reliance on the provision of water services at the ACG was tested based on historical water yield simulations from years 1986 to 2013. Given the historical trend of water provision, results suggested that there is good reliability in the provisioning of water yield at the region (CV less than 50%). This means that the provision of water yield from one year to another is less likely to be affected by water stress. However, complementary scenarios that included the identification of El Niño and La Niña years, climatic extreme events that over the last decades have become as important standards to identify the effects of climate change (IPCC, 2013), suggest an evident influence of El Niño on the reliability of the water provision service. According to Maass et al. (2017), El Niño Southern Oscillation phenomenon (ENSO) is characterized by a lower annual rainfall during its hot phase but higher annual rainfall during its cold phase (La Niña). Results of scenarios 3 (41% less reliability) and 5 (61.7% less reliability) suggest adverse effects on the provision of water by having the occurrence of El Niño years. Moreover, the relevance of these findings for human well-being will rely on understanding how reliable the delivery of provisioning water can be (Runting et al., 2017). Furthermore, the supply of water is essential for the subsistence of other ES, such as the seasonal and inter-annual variability of productivity and litter decomposition (the basis for carbon sequestration services), nutrient cycling, sedimentation and other services (Holmgreen et al., 2001). Moreover, the inclusion of climate change effects in the evaluation of ecosystems services is becoming an undoubtedly challenging task. Ignoring these complexities can result in inaccurate assessments that can mislead the implementation of policies and final decision-making processes (Runting et al., 2017).

3.5 Conclusion

This study provides an ecosystem services based methodology for the biophysical quantification of water provisioning services at the Guanacaste Conservation Area in Costa Rica. I demonstrate the importance of using a hydrological process-based model (SWAT) to provide better estimates of the hydrological variables needed for modeling the provision of water services in a specific area. Improvements on the application of the hydrological model used can be made by incorporating validation and calibration data from local river stream flows. Moreover, I also demonstrate the benefits of mapping spatial explicit ES, for identifying major provisioning areas as well as water stressed areas.

Considering the outputs of SWAT model, this study also proved that biophysical estimates can be integrated with population metrics for obtaining a better picture of the benefits human populations derive from water ecosystems. Furthermore, historical scenario analysis showed the potential effect of extreme climatic events on the reliability of the provision of water services. All these elements can aid in the next stages of ecosystem services assessments, which are the economic valuation and policymaking. The assessment of water ecosystem services offers a promising way to communicate the importance of water for sustaining life on our planet, analyze the tradeoffs and synergies among other ES, to avoid degradation and mainly to support the development of more sustainable water management practices and better-informed decision making processes for stakeholders.

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3.7 Figures

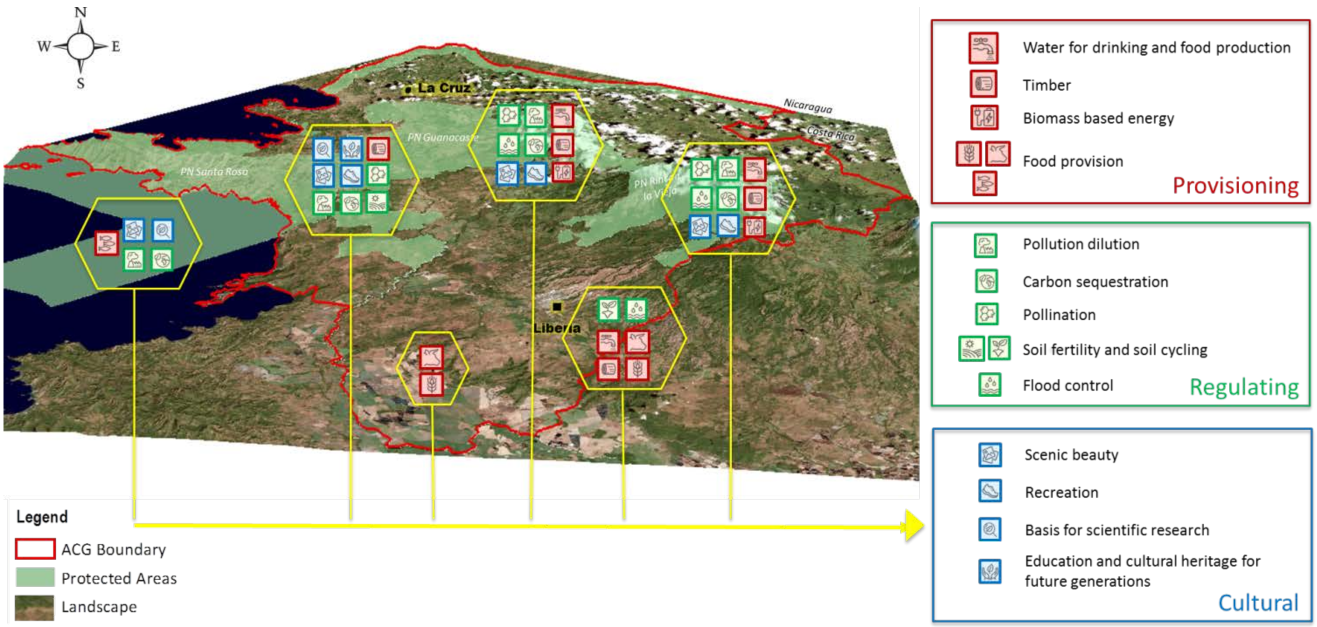


Figure 3.1 Spatial distribution of provisioning (red icons), regulating (green icons) and cultural (blue icons) ecosystem services at the landscape of the Guanacaste Conservation Area in northwest Costa Rica.

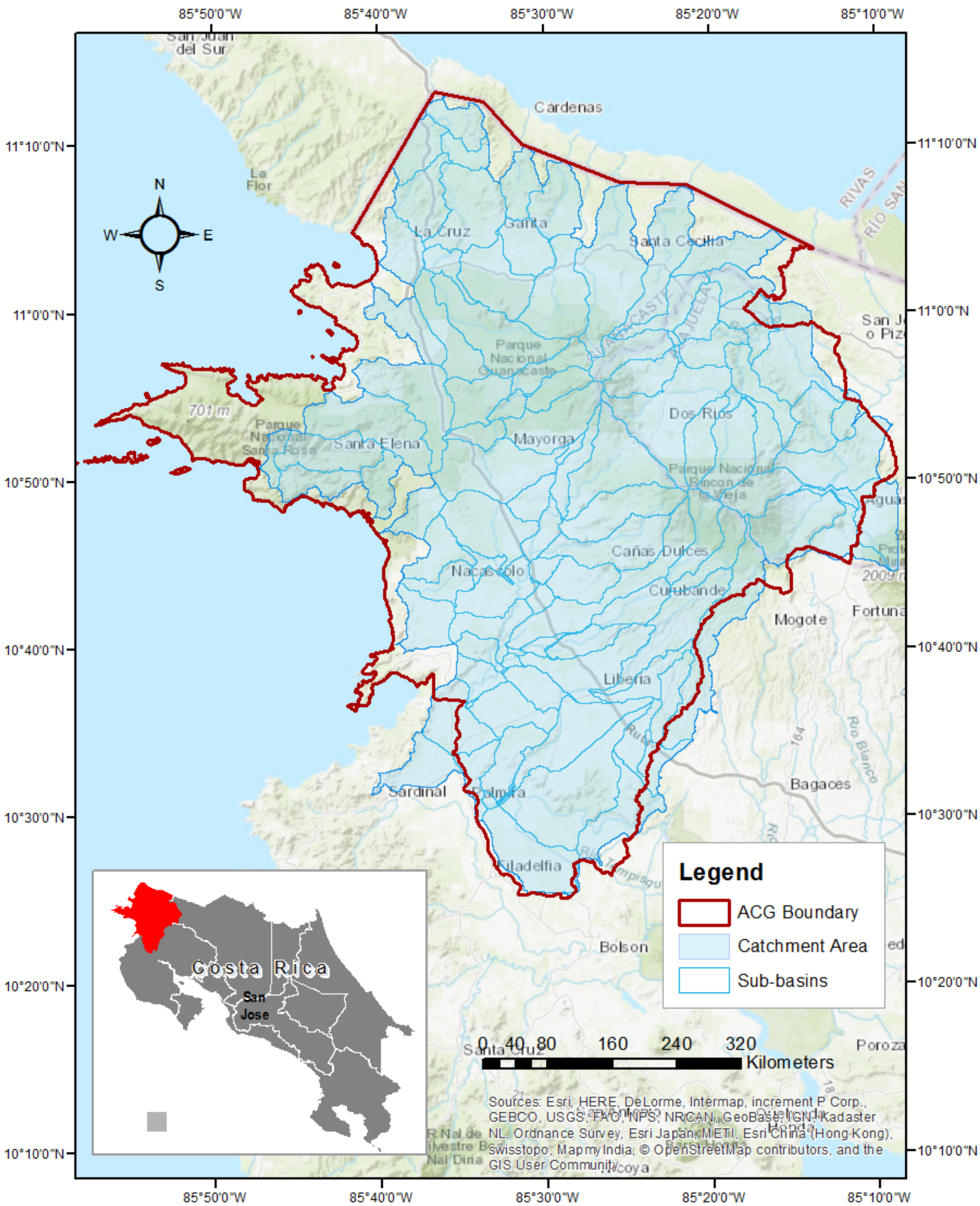


Figure 3.2 Location of the study area at the Guanacaste Conservation Area (ACG) in northwest Costa Rica, incorporating the total catchment area used for hydrological simulation and sub-basin subdivision within this area.

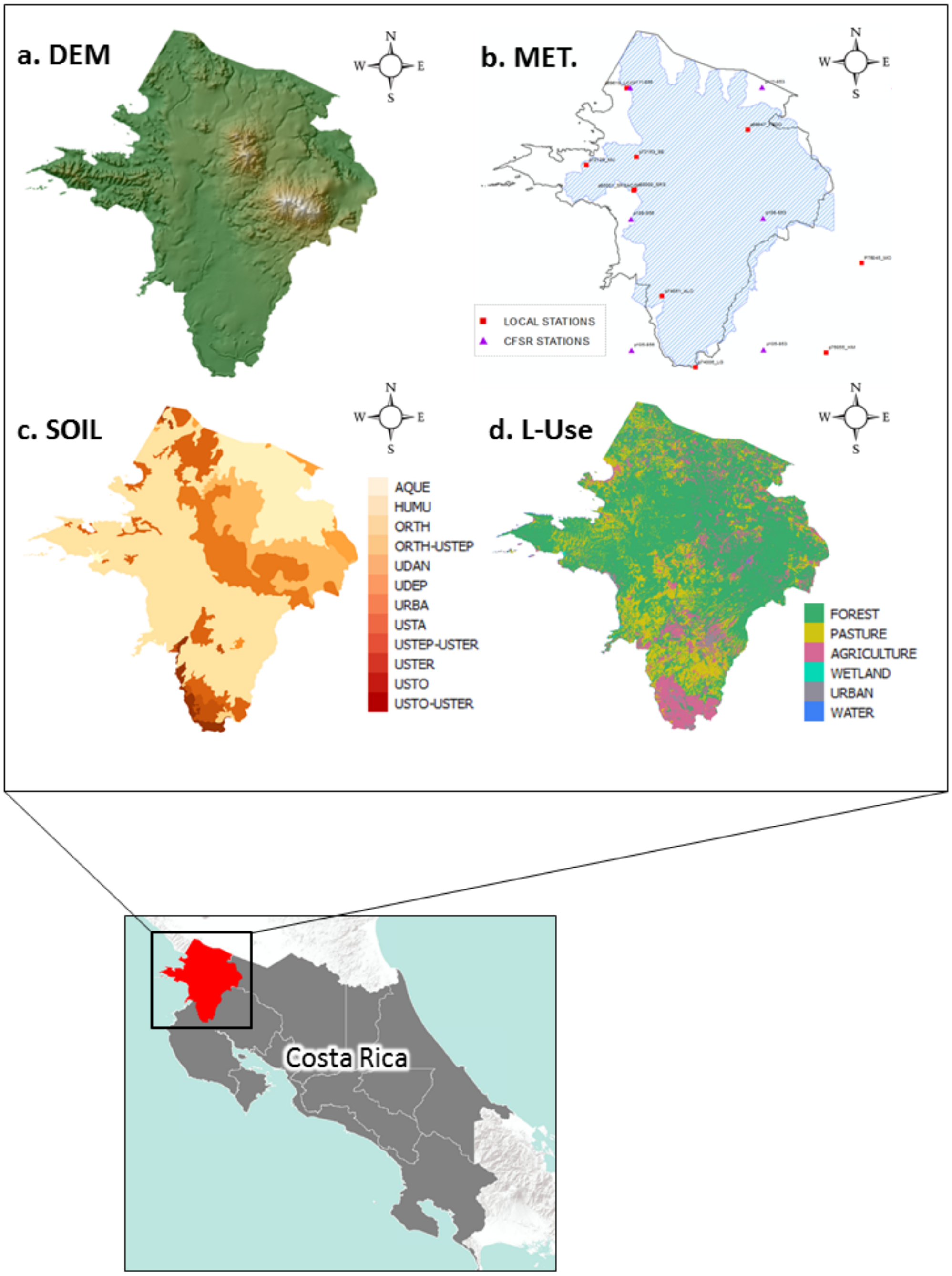


Figure 3.3 Input datasets used for the computational running of the SWAT hydrological model. (a) Digital Elevation Model (DEM) of 10 meters resolution; (b) Spatial distribution of global and local weather stations for precipitation, temperature, wind speed, solar radiation and relative humidity data; (c) Digital soil map according to the US Soil Taxonomy classification; (d) Land use map at 30 meters resolution for year 2015. Classes used are as follows: forest, pasture, agriculture, wetland, urban and water.

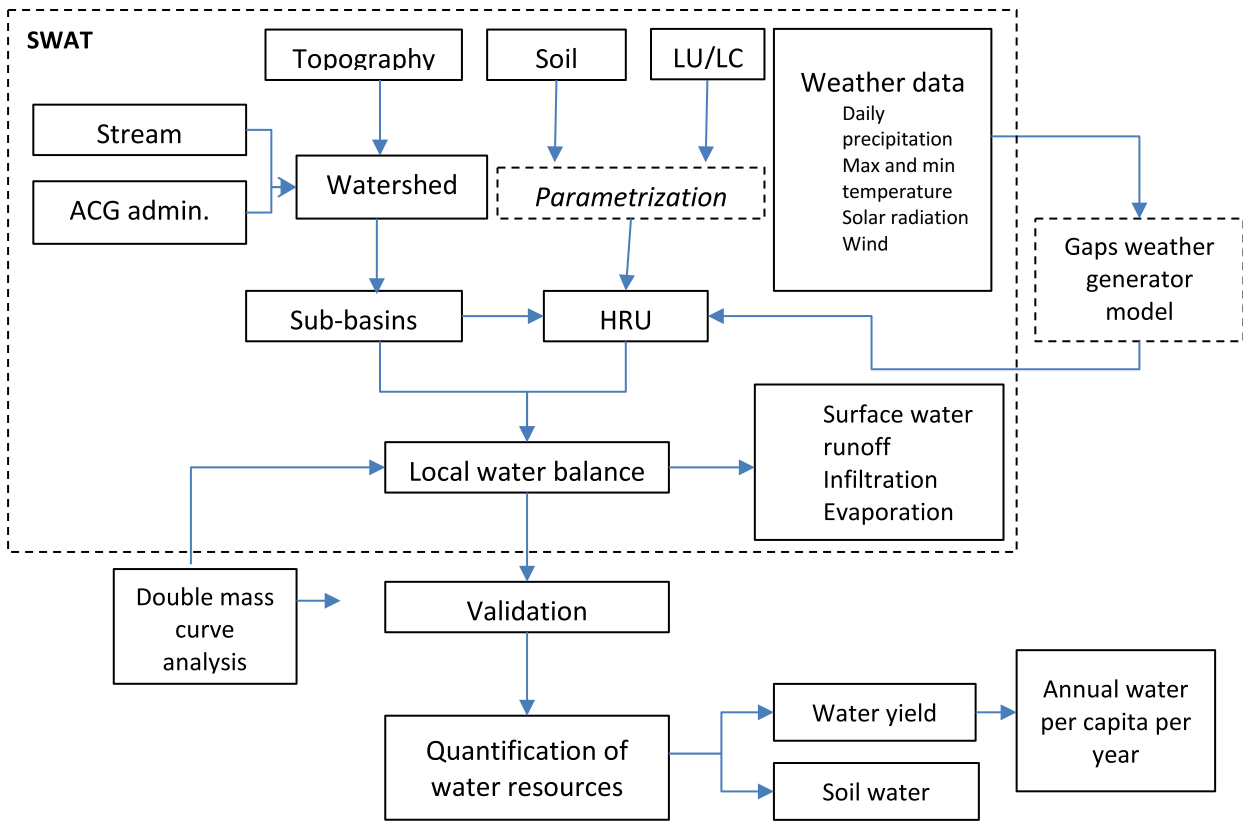


Figure 3.4 Methodological framework used for SWAT hydrological modeling and water provisioning services quantification at the ACG.

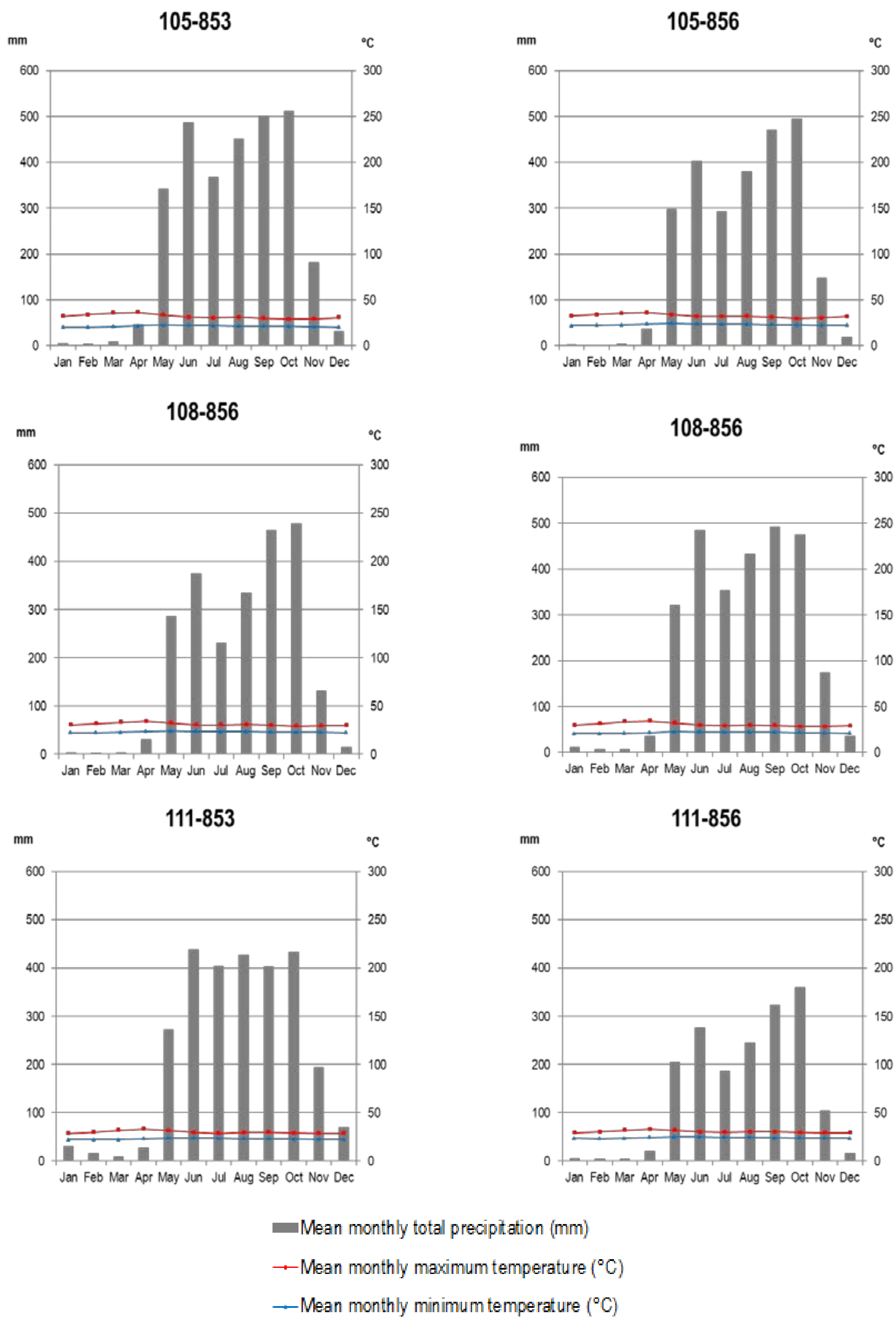


Figure 3.5 Climate diagrams for all input weather stations used in SWAT model. Grey bars represent the mean monthly precipitation at an individual station, and red and blue lines represent mean monthly maximum and minimum temperatures respectively.

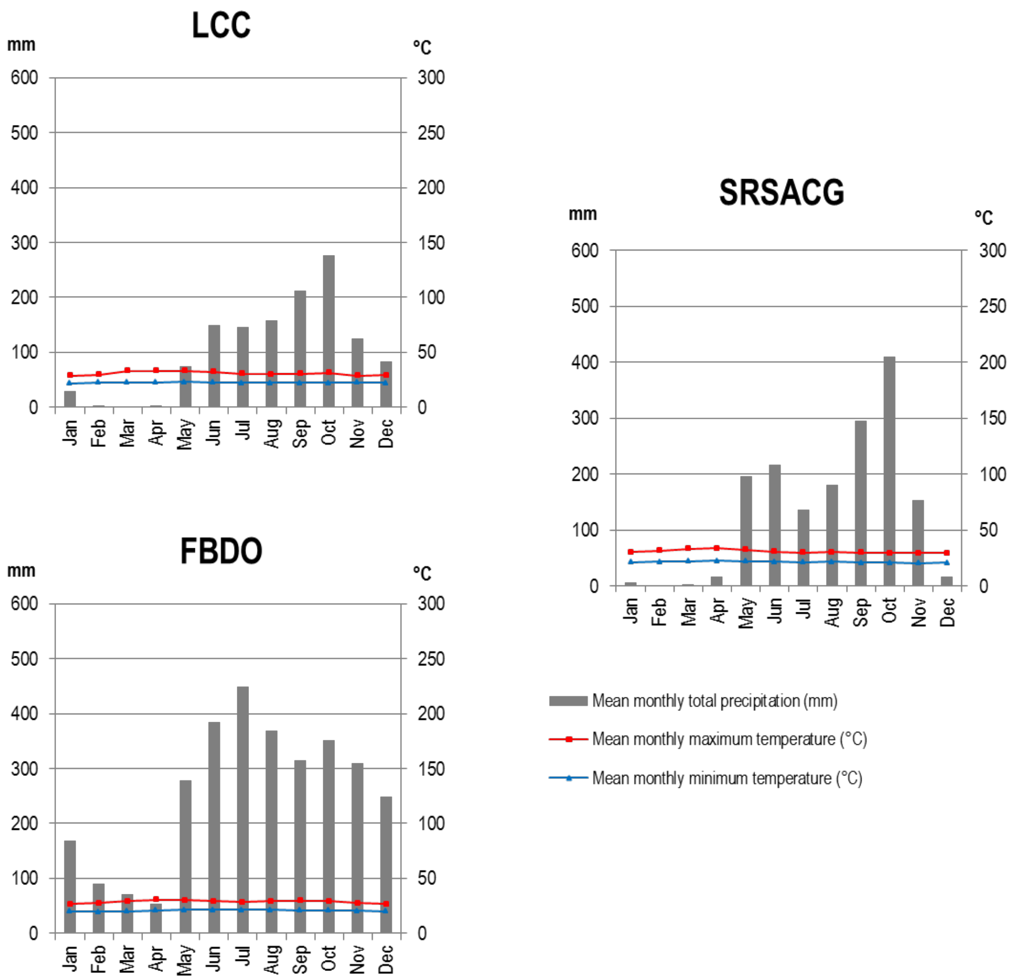


Figure 3.6 Climate diagrams for the three weather stations used for rainfall verification in the Double mass curve analysis. LCC: La Cruz Centro, FBDO: Finca Brasil del Oro and SR: Santa Rosa. Grey bars represent the mean monthly precipitation at an individual station, and red and blue lines represent mean monthly maximum and minimum temperatures respectively.

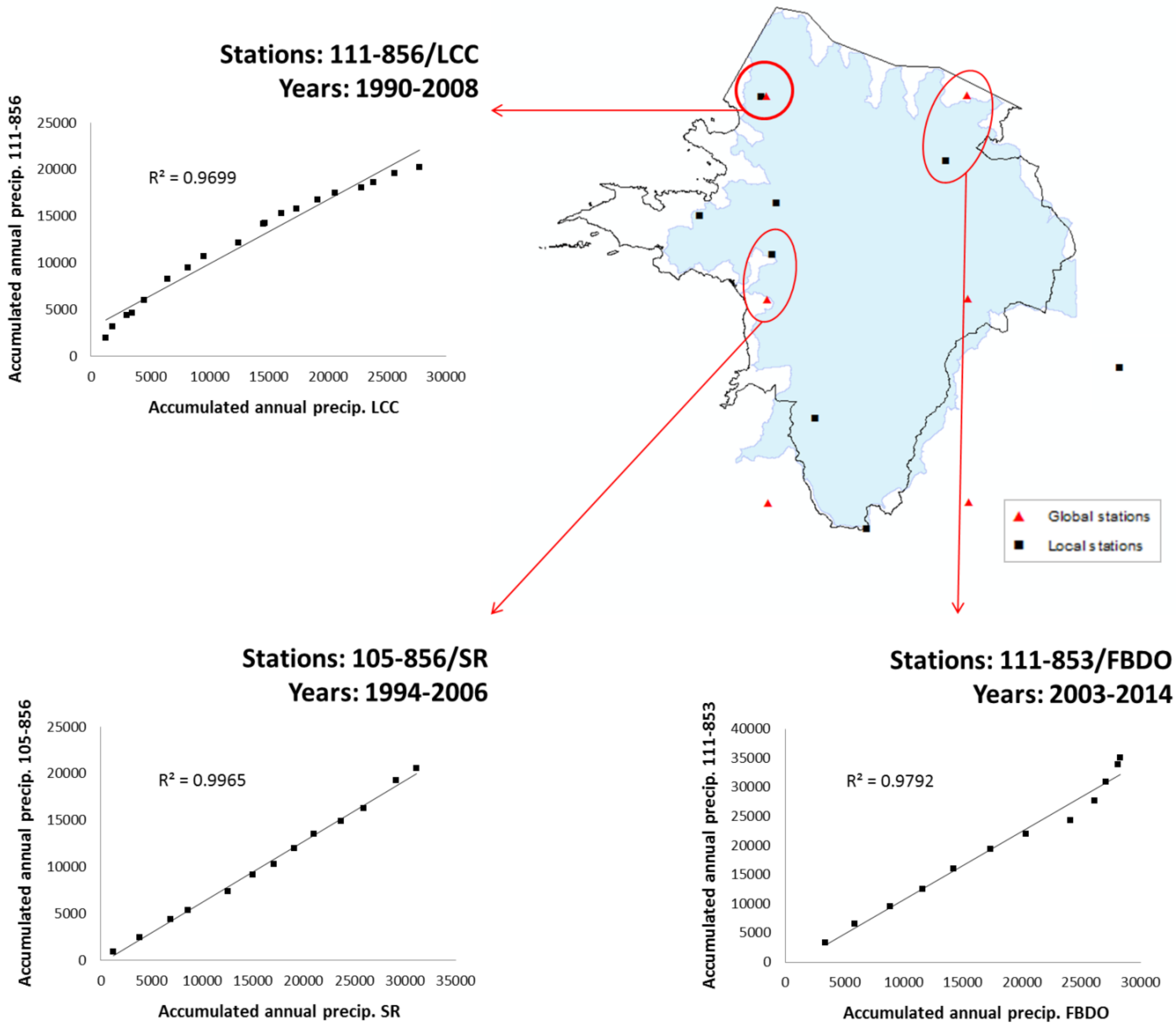


Figure 3.7 Double Mass Curves analysis between global weather stations used for simulation (red triangles) and local stations used for validation (black squares) at different time intervals. Statistical linear regressions are displayed for the three pairs of stations. Global stations used for correlation are 111-856, 105-856 and 111-853. Local stations used for correlation are La Cruz Centro (LCC), Santa Rosa (SR) and Finca Brasil del Oro (FBDO). Straight lines represent extensive relations between the compared stations and scattering is related to differences in the rainfall measurement methods. Selection of the stations for analysis was based on spatial proximity.

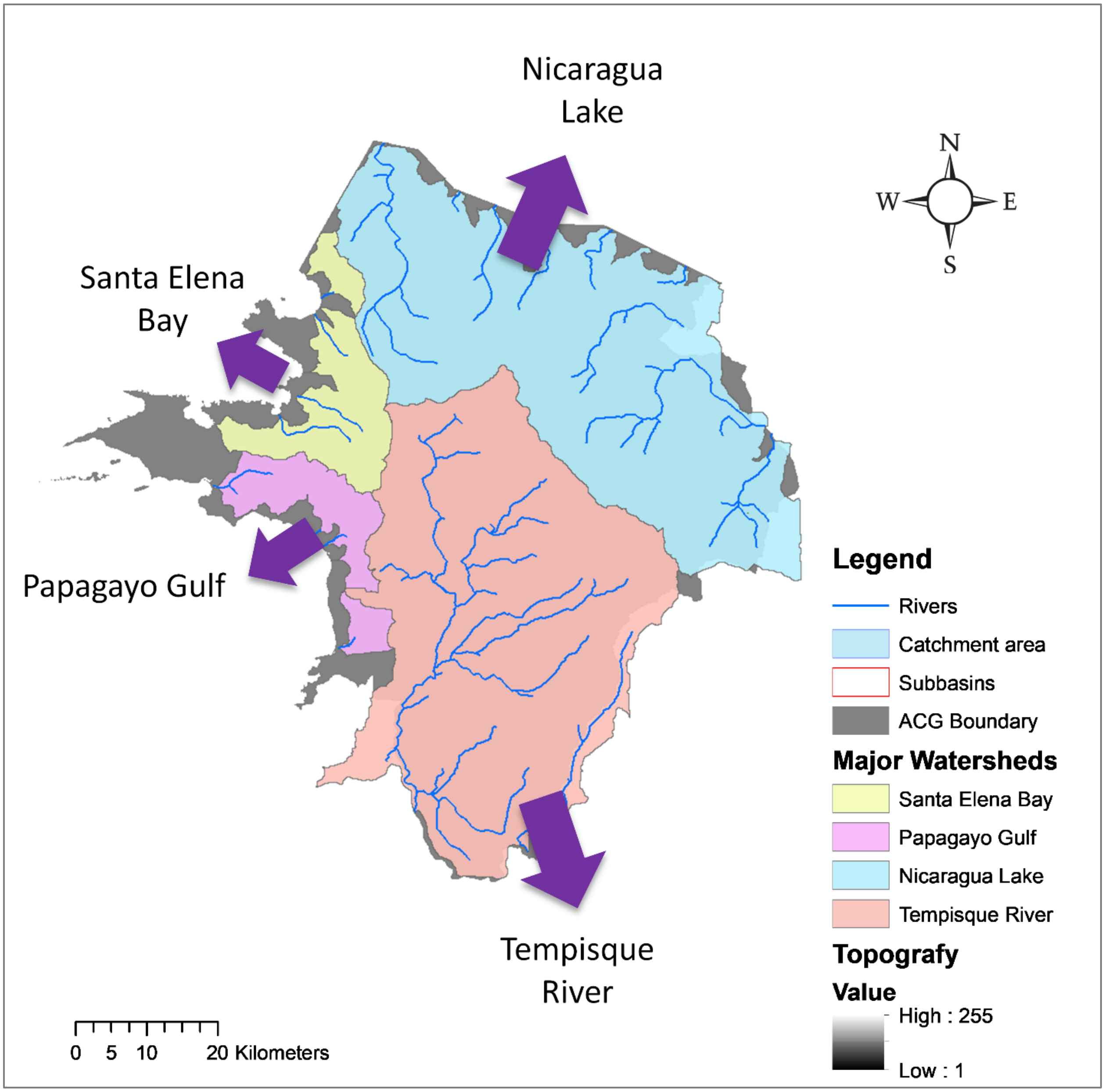


Figure 3.8 Map of major watersheds distribution and principal water flow directions for the study area.

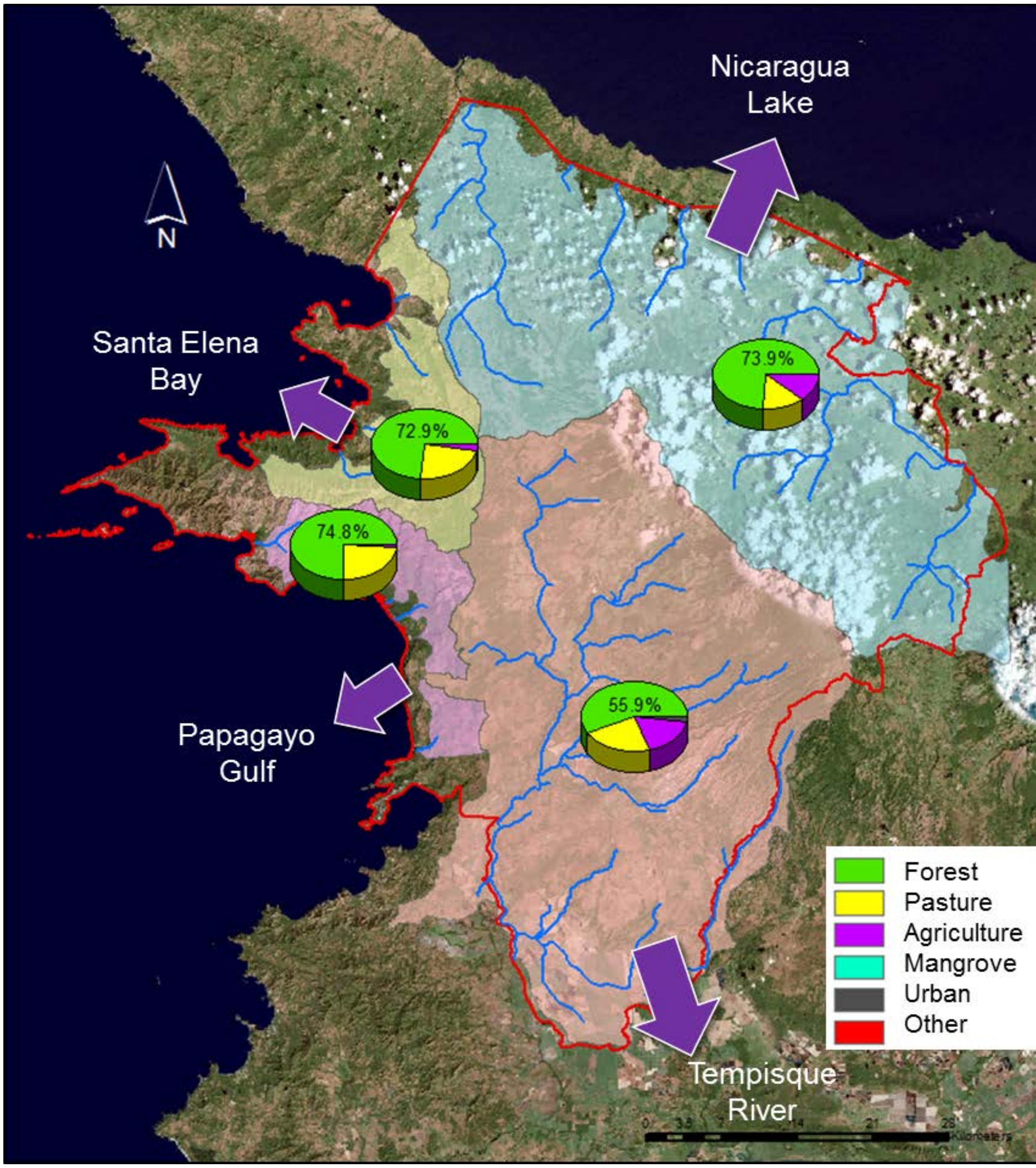


Figure 3.9 Map of the principal land uses expressed in (%) for all four major watersheds (Nicaragua Lake, Santa Elena Bay, Papagayo Gulf and Tempisque River) at the ACG.

Imagery source: Mapbox

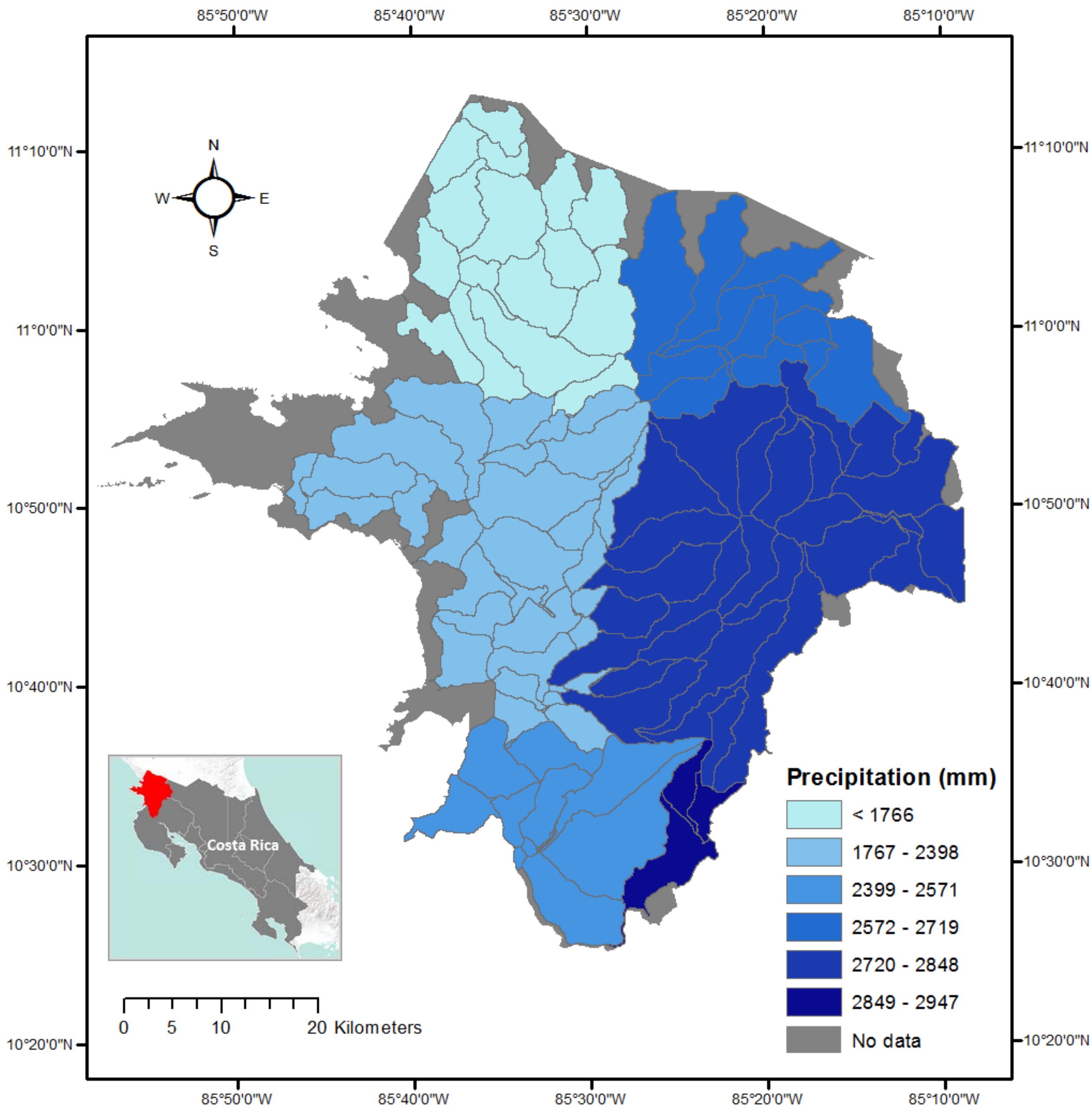
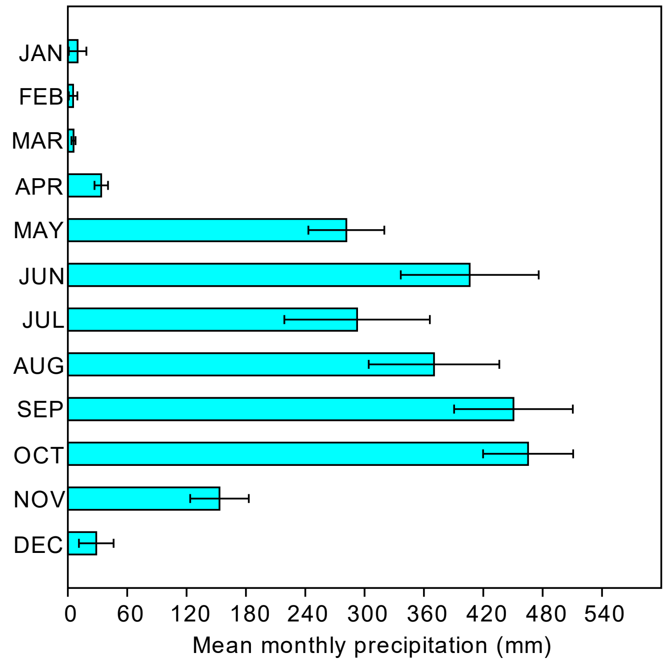


Figure 3.10 Estimates of mean annual precipitation in (mm) for the years 1988 to 2013 at sub-basin level simulated for SWAT.

a.



b.

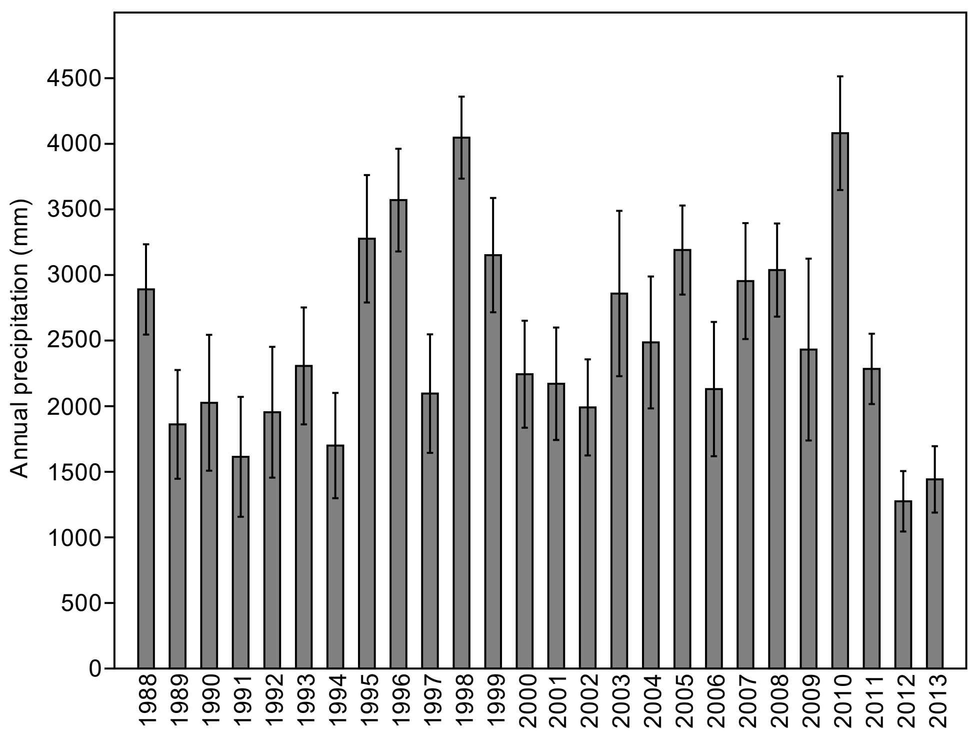


Figure 3.11 (a) Mean monthly precipitation from 1988 to 2013 simulated in SWAT (b) Mean annual precipitation trends across all sub-basins in the study area. Values are expressed in (mm). Bars represent mean rainfall values and whiskers represent the standard deviation of the values.

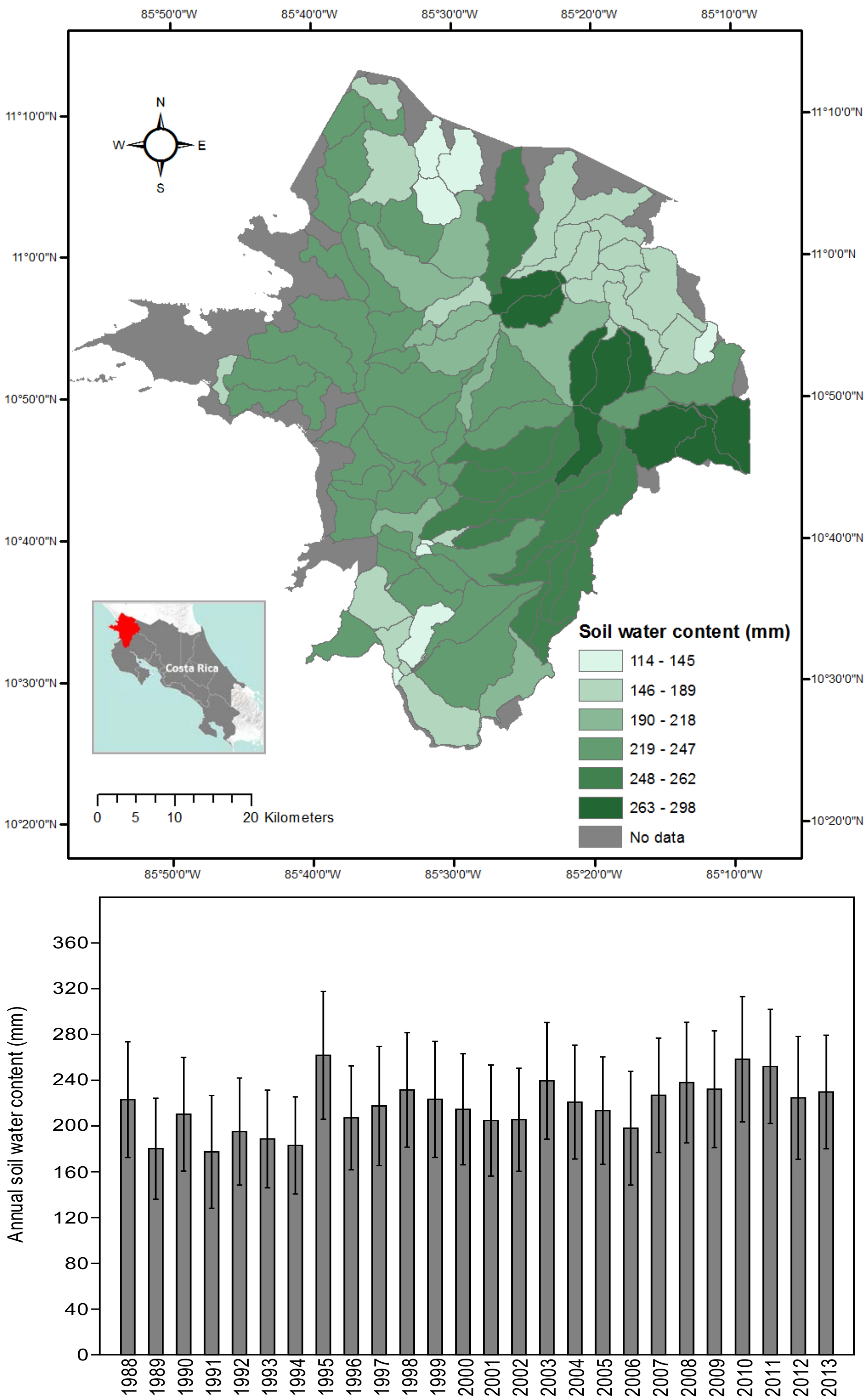


Figure 3.12 Estimates of mean annual soil water content in (mm) at sub-basin level simulated by SWAT and mean annual soil water content trends for total study area expressed in (mm) for the years 1988 to 2013.

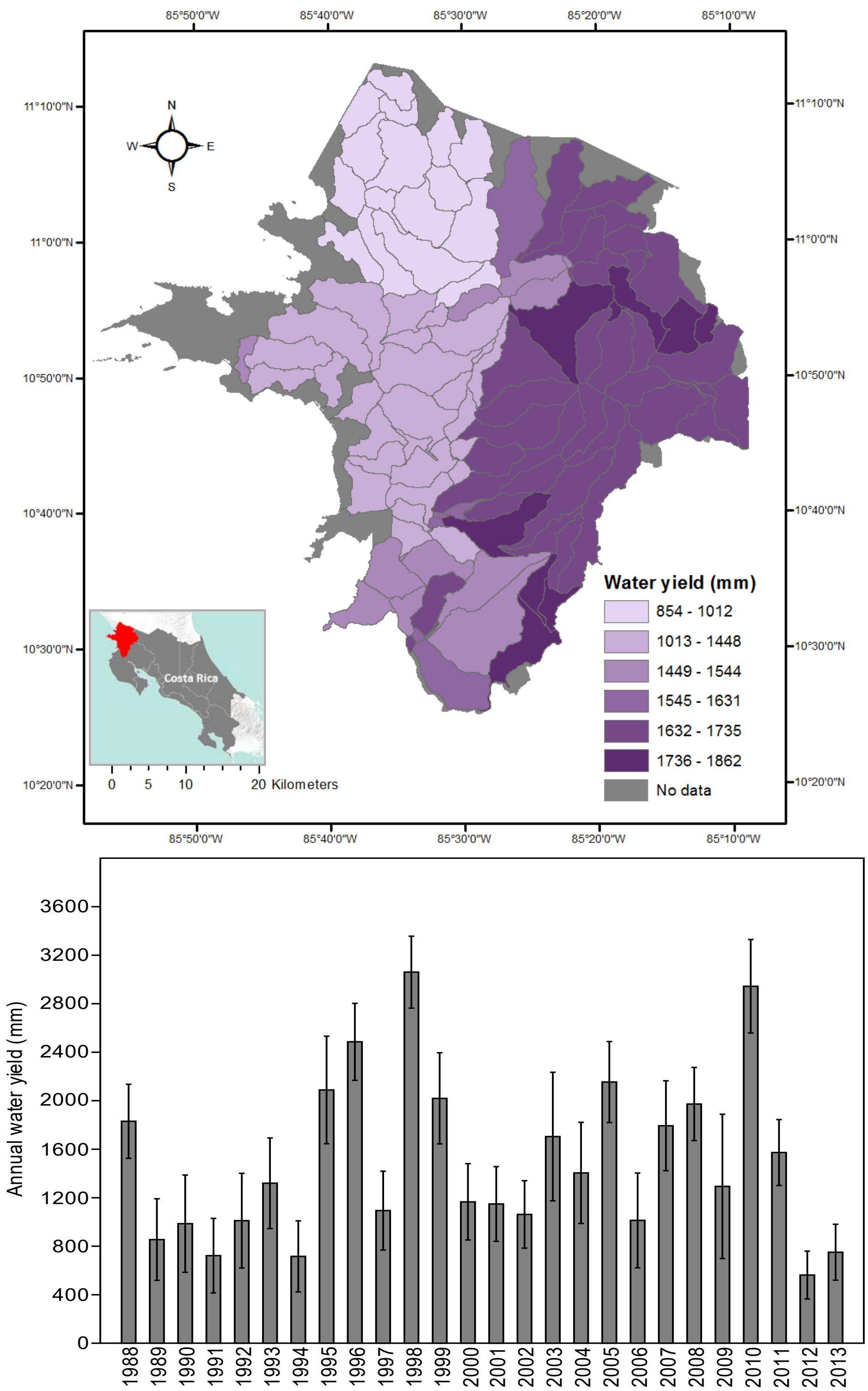


Figure 3.13 Estimates of mean annual water yield in (mm) at sub-basin level simulated by SWAT and mean annual water yield trends for total study area expressed in (mm) for the years 1988 to 2013.

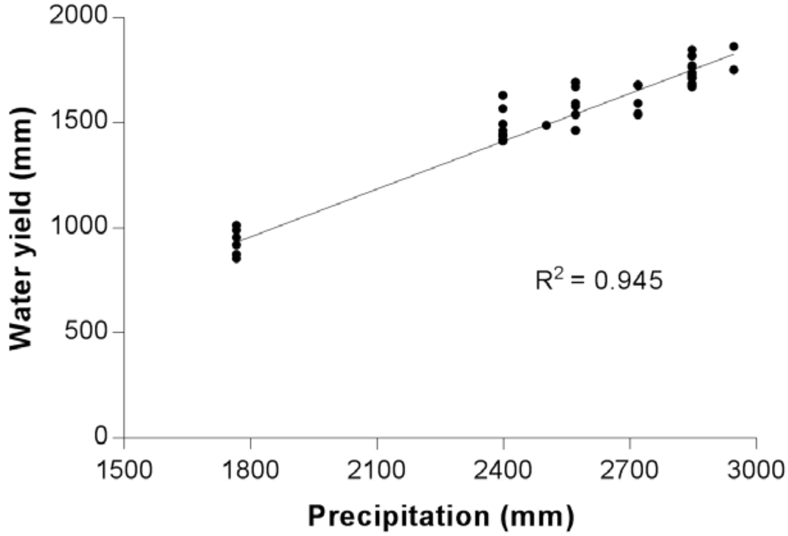


Figure 3.14 Mean annual water yield (mm) vs. mean annual precipitation (mm) in the study area. The ratio also represents the proportion of precipitation that remains as water yield after losses for the provision of water ES.

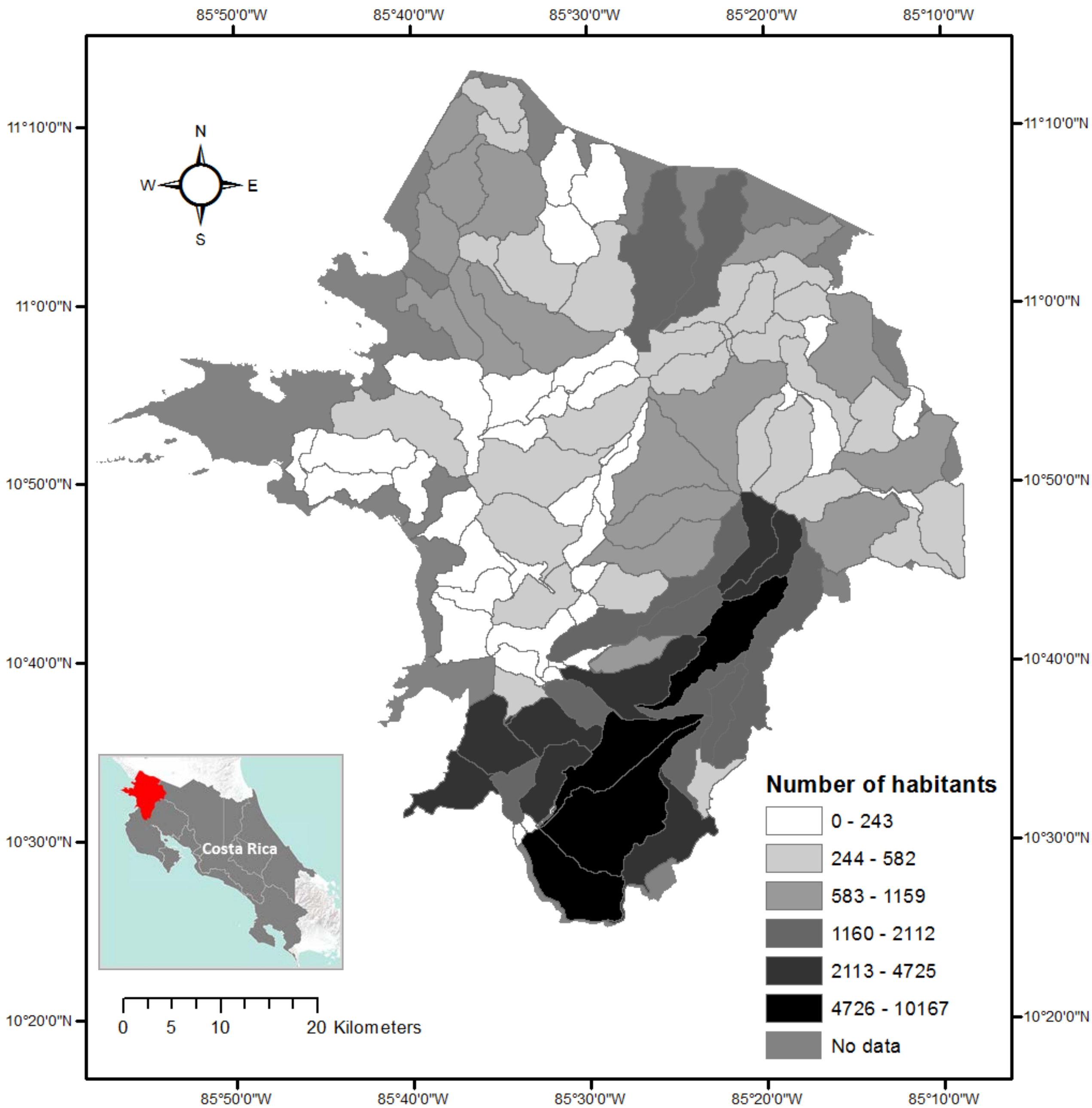


Figure 3.15 Spatial allocation of the population at each sub-basin unit in the study area expressed in number of habitants per sub-basin. Darker colors represent higher population densities.

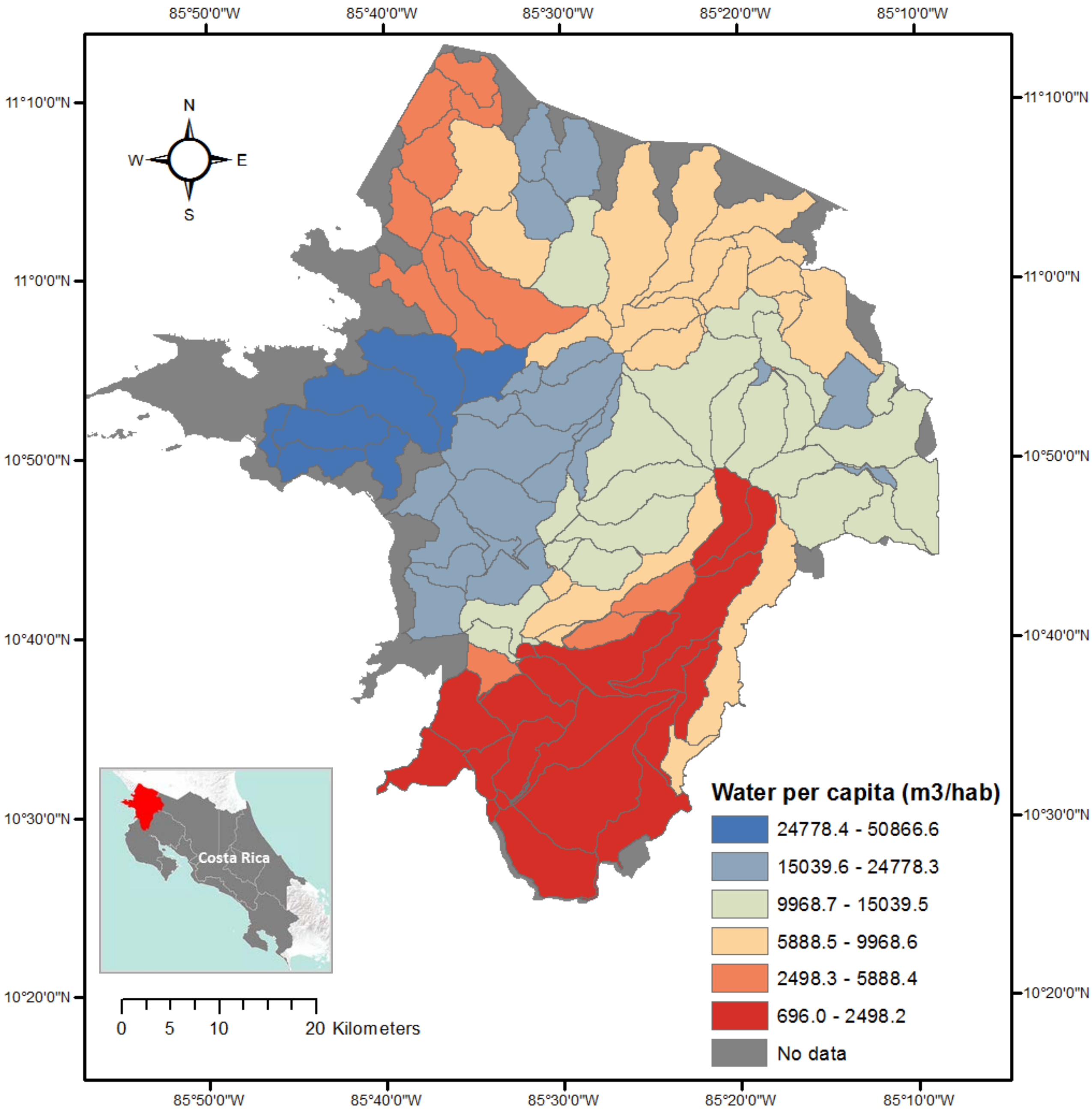


Figure 3.16 Resulting estimates of mean annual water per capita expressed in (m^3 hab^{-1}) at each sub-basin unit. Darker blue colors represent higher amounts of water per capita whereas red colors represent lower quantities of water per capita.

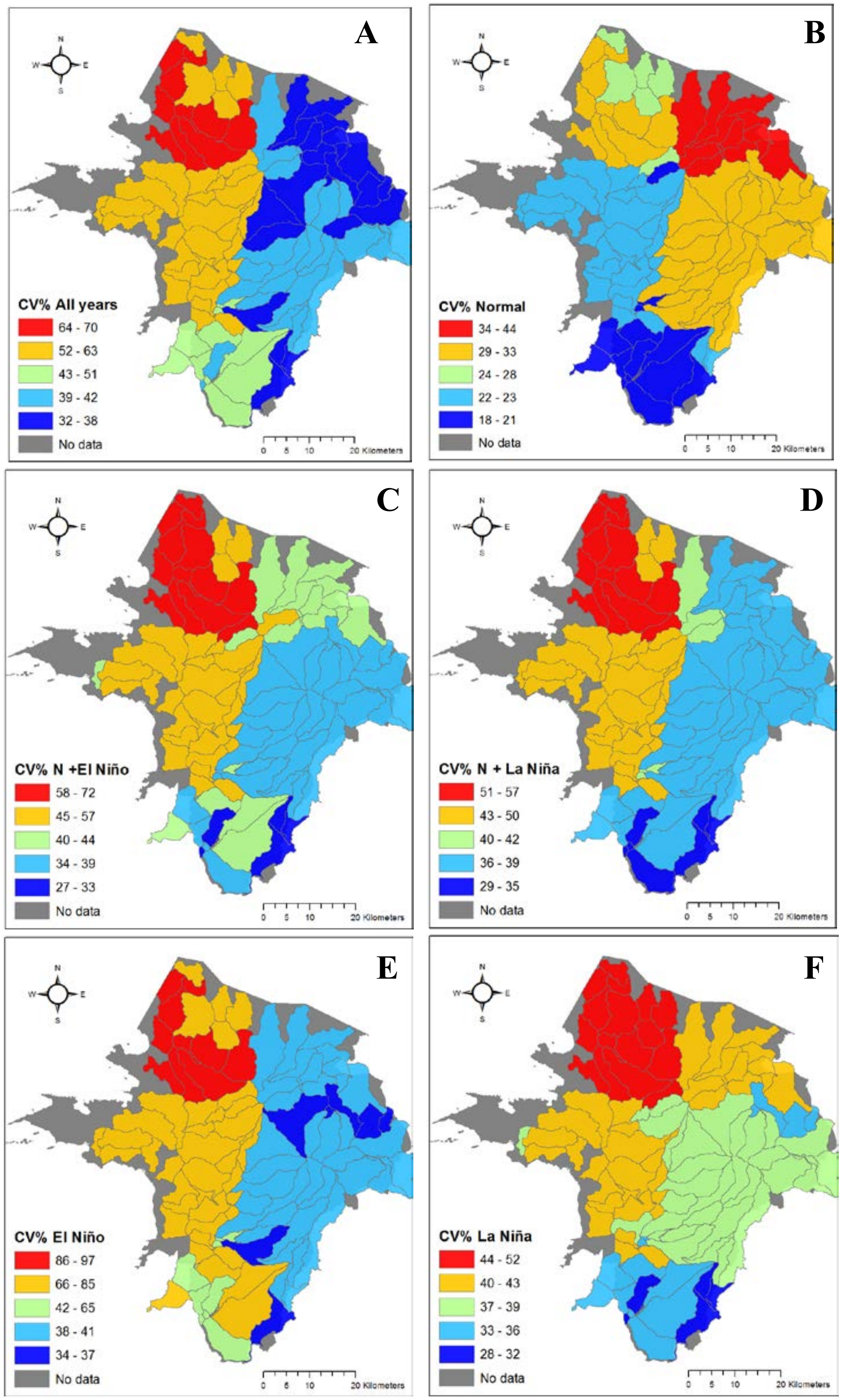


Figure 3.17 Coefficient of Variation for different water provisioning scenarios. A) For all years in the trend; B) For all normal years; C) For all normal years including El Niño; D) For all normal years including La Niña; E) Just for El Niño years and F) Just for La Niña years.

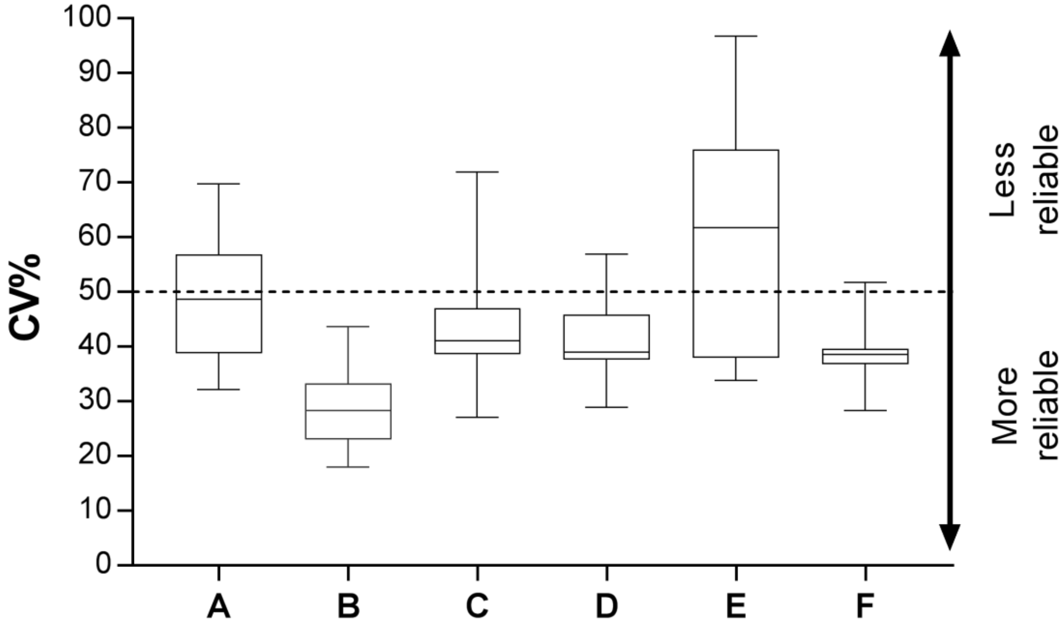


Figure 3.18 Distribution of the Coefficient of Variation for assessing the reliability of the annual provision of water for all six scenarios.

Where:

A. First scenario including all years in the trend

B. Second scenario selecting only normal years in the trend

C. Third scenario for all normal years including El Niño years

D. Fourth scenario for all normal years including La Niña years

E. Fifth scenario selecting only El Niño years

F. Sixth scenario selecting only La Niña years

3.8 Tables

Table 3.1 Land use categories used to simulate water yield in SWAT model.

No.	Land use category	Area (km2)
1	Forest	2303.7
2	Pasture	679.5
3	Agriculture	423.8
4	Wetland	2.3
5	Urban	29.1
6	Other	5.8

Table 3.2 Soil classes used to simulate water yield in SWAT model according to USDA Soil Taxonomy Classification.

No.	Soil class	Area (km2)
1	A-ENTS	9.2
2	H-ULTS	486.6
3	O-ENTS	1684.6
4	ENTS-UEPTS	10.4
5	UANDS	400.9
6	UDEPTS	69.1
7	U-ANDS	393.1
8	U-EPTS	287.4
9	U-ERTS	48.0
10	U-OLLS	5.0
11	OLLS-UEPTS	50.9

Table 3.3 List of meteorological stations used in the simulation from the year 1985 to 2013.

No.	Name	Latitude	Longitude	Elevation	Description
1	105-853	10,46	-85,313	135	Daily records of precipitation, minimum and maximum temperatures, solar radiation, wind speed and relative humidity from years 1985 to 2013.
2	105-856	10,46	-85,625	102	
3	108-853	10,772	-85,313	894	
4	108-856	10,772	-85,625	192	
5	111-853	11,084	-85,313	139	
6	111-856	11,084	-85,625	199	

Table 3.4 List of local meteorological stations used for validation for different time periods.

No.	Name	Latitude	Longitude	Elevation	Description
1	La Cruz Centro (LCC)	11,083	-85,633	150	Daily records of precipitation and minimum and maximum from years 1990 to 2008.
2	Santa Rosa (SR)	10,840	-85,618	303	Daily records of precipitation and minimum and maximum from years 1994 to 2006.
3	Finca Brasil del Oro (FBDO)	10,983	-85,347	380	Daily records of precipitation and minimum and maximum from years 2003 to 2014.

Table 3.5 Land use distribution for the major watersheds at the ACG.

	Major watersheds	Total Area (km^2)	Forest (km^2)	Pasture (km^2)	Agriculture (km^2)	Wetland (km^2)	Urban (km^2)	Other (km^2)
1	Santa Elena Bay	228.0	166.3	55.0	5.5	0.06	0.1	0.07
2	Papagayo Gulf	165.3	123.7	39.4	1.7	0.3	0.08	0.0
3	Nicaragua Lake	1361.6	1007.4	202.8	148.6	0.0	0.2	2.6
4	Tempisque River	1469.0	821.7	366.1	253.7	0.1	27.2	0.1
	Total	3223.9	2119.2	663.4	409.6	0.5	28.5	2.7

Table 3.6 Total water balance simulated for the Guanacaste Conservation Area (ACG).

GUANACASTE CONSERVATION AREA	
Mean annual precipitation (mm)	2501.97
Mean Annual evapotranspiration (mm)	795.49
Mean annual water soil content (mm)	218.58
Mean annual water yield (mm)	1487.89
Catchment Area (km^2)	3090
Total Population (Hab)	98509
Mean annual water per capita m^3/Hab	11357.58

Table 3.7 Total water balance simulated for the major watersheds at the Guanacaste Conservation Area (ACG).

	ACG MAJOR WATERSHEDS			
	Santa Elena Bay	**Papagayo Gulf**	**Nicaragua Lake**	**Tempisque River**
Mean annual precipitation (mm)	2465.1	2502.1	2451.9	2540.7
Mean Annual evapotranspiration (mm)	801.3	785.8	795.3	795.9
Mean annual water soil content (mm)	258.6	238.8	220.9	210.1
Mean annual water yield (mm)	1405.2	1477.5	1435.7	1534. 6
CV %	30.7	24.2	31.1	26.6
Total Population (Hab.)	1293	8636	40160	48599
Area (km2)	108.4	144.9	1425.9	1411.3
Mean annual water per capita m^3/Hab.	12744.1	10871.8	10909.8	11456.1

Chapter 4: Conclusions and Future Work

The main objective of this thesis was to determinate the potential of Tropical Dry Forests (TDFs) for the supply of two key ecosystem services (ES): carbon sequestration and water provision. In the precedent chapters I have presented the biophysical quantities and economic values of the carbon sequestration service in TDFs of Mexico, Costa Rica, Brazil and Bolivia, this by using a subset of spatial-based tools. I also provide an ES modeling approach for the biophysical quantification and reliability assessment of water provisioning services at a regional scale in a TDF of Costa Rica.

4.1 Synthesis of significant contributions

The results of Chapter 2 "**Assessing terrestrial carbon sequestration in the tropical dry forests of America**" have shown the relevance of TDFs for the supply of carbon sequestration services and for sustaining human well-being through the regulation of climate. Main biophysical trends in TDF sites showed higher and consistent annual sequestration rates at northern sites and lower but increasing trends at southern latitudes. A major finding above all studied TDFs was the optimum efficiency for the transfer of CO_2 as terrestrial biomass (66% of GPP is stored as terrestrial NPP in TDFs). Moreover, variations in sequestration rates can be explained by differences in forest structural properties (e.g., age, growth stages, species composition) (Mora et al., 2017), and the difficulty of considering human disturbances associated to TDFs given the strong influence of anthropogenic activities (Sanchez-Azofeifa et al., 2005). Furthermore, the different spectrums of land management associated to each TDF site (e.g., higher deforestation rates and extensive clearing in countries like Bolivia and Brazil contrast with the emphasis on conservation and forest restoration in countries like Costa Rica).

I also demonstrated fundamental contributions of the carbon sequestration service for national economic accounts, by determining the monetary value in USD of a hectare of TDFs in different countries of America considering a range of Social Costs of Carbon (SCC). The results showed that the economic values vary according to the biophysical sequestration rates of each TDF site. Estimated values can also be extrapolated to calculate the total revenue of a landowner's property, a protected area, or a large forest fragment as was showed in this study. The lack of local carbon markets in the majority of the studied TDFs was addressed by selecting three SCC global reference values being the FUND model the lowest value (Tol, 2009), DICE model for the medium value (Nordhaus, 2017) and the PAGE model for the highest value (Hope, 2011). Although the use of global reference values can serve as a foundation to determine the general contributions of TDFs to national accounts, there is a clear need for the implementation of local carbon markets and more appealing offset programs to increase social achievements for local communities, landowners and other stakeholders (Hall, 2012). A successful example of the implementation of incentives for the conservancy of forest areas, environmental policies, and stakeholder involvement was seen in Costa Rica with the restoration of the TDF (Janzen and Hallwachs, 2016).

In addition, given the growing interest of decision makers for faster and more straightforward ways to understand how much carbon is stored in landscapes for carbon markets and incentive programs (Gibbs et al., 2007). I provided a systematic methodology for a complete assessment of the CO_2 sequestration service through the integration of a remote sensing based approach (for the biophysical component) and a benefit transfer method (for the economic component). Although methods that rely on direct measurements and forest inventories (Brown, 1997) remain the best tool to estimate biophysical carbon stocks, I demonstrated that a remote sensing based approach is a good and consistent proxy for an initial quantification of the carbon sequestration service at regional scales and over several periods of time. Moreover, scientific challenges nowadays are focusing on the different integrations of remote sensing methods with e-flux site based measurements (Cui et al., 2016) and ecosystem modeling (Nightingale et al., 2012) to have more efficient estimates. Furthermore collecting an appropriate and stratified site sample data is a critical step for improving sequestration estimates across all methods (Gibbs et al., 2007).

Although the quantification of CO_2 sequestration in tropical forests is a challenge, Gibbs et al. (2007) state that one of the predominant challenges to be integrated into the determination of carbon budgets is the consideration of carbon emissions from deforestation and degradation. A task that often relies on the use of land use maps for identifying cleared areas. However, discrepancies and uncertainties regarding the land cover methods and datasets should be considered given the occurrence of issues regarding the estimation of areas, comparative analysis and semantics (Kalacska et al., 2008).

The results from Chapter 3 "**Quantifying water provision services in a tropical dry forest: the case of Guanacaste Conservation Area**" established a first baseline for the biophysical assessment of the supply of water provisioning services at the Guanacaste

Conservation Area (ACG) in Costa Rica. Results suggest that the ACG has an appropriate provision of water overtime. Nevertheless, geographical singularities were relevant to determine differences in the provision of water and its relation with human settlements. For instance, the less populated region in the west presents lower annual water yield rates in contrast to the higher water yield of the eastern and southeastern parts which possess a higher population density. By mapping significant areas for the provisioning of water as well as regions with restricted availability I showed that the water provision service is spatially explicit (Fisher et al., 2008). The advantages of using mapping tools for quantifying ecosystem services and for supporting stakeholder communication and decision making processes were also addressed by Burkhard et al. (2013).

Also, since the scarcity of water is a major constraint in TDFs (Sánchez-Azofeifa and Portillo-Quintero, 2011), I estimated the potential effect of extreme climatic events, such as El Niño and La Niña, on the historical reliability of the provision of water services. Results suggest that El Niño has a high impact on the reliability of the provision of water services (Maass et al., 2017). This is a critical element to be considered in the future since the provision of water is not only important for human consumption but also for the supply of multiple ecosystem services such as productivity, soil cycling, and others.

I also demonstrated the importance of using a hydrological process-based model (SWAT) to provide better estimates of the hydrological variables needed for modeling the provision of water services. Vigerstol and Eukema (2011) established that although traditional hydrological balances can provide a good general picture of the provision of water, there is a lot to learn about the connections of land use, soil, and vegetation to water modeling. In this sense, I demonstrated the importance of considering the soil characteristics and land use properties of the landscape for improving hydrological modeling. This is a crucial element to consider in TDF areas, given the major influences of anthropogenic activities (e.g., agriculture and cattle ranching) that rely on the consumption of water. In the ACG, this matrix of different land uses and forested areas has originated what Cao and Sanchez-Azofeifa (2017) referred as the "agro-landscape," a region where the ideas of sustainability and human consumption try to sustain in equilibrium, and that depends mainly on the regular provision of water services.

Furthermore, although a lot remains to explore about the use of SWAT in an ecosystem service context (Francesconi et al., 2016), the approach presented in this thesis has proven to be a useful tool for the spatial and temporal assessment of water provisioning services. All the elements included in this approach can be easily extrapolated to a national or regional scale. However, incorporating a correct collection of input data sets and local data for calibration and validation will undoubtedly improve the hydrological modeling process. All these elements will consequentially support the next stages of ecosystem services assessment, which are the economic valuation and decision making.

4.2 Future work and challenges

In spite of the substantial ecological research conducted in TDFs (Sanchez-Azofeifa et al., 2005; Balvanera, 2012), further understanding of the linkages between ecological processes and human well-being in an ecosystem services context is in its early stages (Maass et al., 2005). In this context, one of the predominant challenges is the development of consistent and robust ways of measuring, mapping, modeling and valuing multiple ES in TDFs. For Calvo-Rodriguez et al. (2016) there is a fundamental need to create standard frameworks to reach such understanding. Long-term and large-scale standardized studies are therefore critical for supporting future comparisons in space and time. Also, given the critical impact of climate change on the state of ecosystem services, analyzing the future effects that these changes can have on ES supply constitutes an important challenge. Runting et al. (2017) stated that despite increasing literature on climate change impacts on ecosystem services, no quantitative syntheses or scenario planning studies exists.

Another consideration for future ecosystem services research in TDF is the fact that ecosystem processes and services are closely related, thus, dealing with synergies and trade-offs is unavoidable. According to Costanza et al. (2017) these occur mainly between provisioning and regulating or supporting services, and in the majority of the time arise from the management choices made by humans. Locatelli et al. (2015) identified three ways of approaching such study: across space, across time, and according to their reversibility.

While the biophysical dimension of ES studies in TDFs has been increasingly studied in Latin America over the last years (Calvo-Rodriguez et al., 2016) there is still an emergent need to integrate the economic dimension in further research. A key challenge in any ecosystem accounting and valuation process is imperfect information (Costanza et al., 2017), as outlined in chapter two of this thesis, most of the avoided damages are implicit for society and hard to quantify. Also, the value of an ecosystem service highly depends on measurements and treatment of biophysical information which also involves uncertainties in the estimation. However, even providing a first “ES unit value” is fundamental for recognizing the contributions of ES to national accounts. These challenges are currently trying to be addressed by the development of standard accounting frameworks such as the United Nations System of Environmental Accounting (SEEA, 2012) and the World Bank-led global partnership WAVES (for Wealth Accounting and the Valuation of Ecosystem Services).

Moreover, ES valuation processes often involve the participation of many stakeholders in TDFs, and their perception of value and well-being might not be the same. For example, the perceived value of a forest area might be different for a farmer, an indigenous community or a decision maker. In this context, major challenges rely on the development of integrated social–ecological research with a continuous dialog among stakeholders in collaboration with national and local decision makers. This will encourage the construction of a common vision for regional sustainable development and to conterest the deterioration of regulating, supporting, and provisioning services (Hall, 2012).

Although we may never have an exact estimate of the quantity or value of ecosystem services in TDFs, the results and methods provided in this thesis are an initial baseline for future research and assessments. Finally, this thesis is abundantly clear about the essential contributions of provisioning and regulating services of TDFs for human well-being and livelihoods.

4.3 References

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CHAPTER 3: QUANTIFYING WATER PROVISION SERVICES IN A TROPICAL DRY FOREST: THE CASE OF THE GUANACASTE CONSERVATION AREA

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CHAPTER 4: CONCLUSIONS AND FUTURE WORK

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APPENDICES

Appendix 1. Database of all monthly GPP values expressed in tC ha^{-1}, processed from MODIS17A from the year 2001 to 2015 for all TDFs sites.

CHAMELA												
YEAR	**JAN**	**FEB**	**MAR**	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	**DEC**
2001	1.21	0.84	1.15	1.00	0.84	0.32	0.56	0.47	0.43	1.04	1.51	0.70
2002	0.90	0.79	1.07	0.87	0.91	1.12	0.92	0.67	0.41	0.72	0.69	0.56
2003	1.16	1.09	0.79	0.83	0.90	0.45	0.45	0.32	0.42	0.41	0.61	1.12
2004	1.03	1.29	0.94	1.03	0.72	0.61	0.44	0.47	0.35	0.83	1.38	1.05
2005	0.84	0.77	1.03	0.74	0.86	0.93	0.33	0.82	0.36	0.61	1.16	1.12
2006	0.90	0.80	0.86	0.92	0.89	0.87	0.42	0.49	0.44	0.52	0.88	1.32
2007	1.10	1.03	0.72	0.79	0.85	0.79	0.67	0.43	0.35	0.51	0.87	1.35
2008	1.11	1.00	0.87	0.95	0.86	0.78	0.51	0.47	0.28	0.50	0.73	1.19
2009	1.20	1.00	1.06	0.99	0.87	0.73	0.25	0.44	0.41	0.67	0.58	1.35
2010	1.03	1.08	1.39	1.55	1.03	1.09	0.47	0.33	0.43	0.79	1.07	1.57
2011	1.14	0.94	0.97	0.99	0.94	0.64	0.52	0.54	0.43	0.91	0.77	0.79
2012	1.21	1.05	1.24	1.14	1.04	1.09	0.54	0.51	0.46	0.81	0.66	1.08
2013	1.13	1.34	0.89	0.97	0.93	0.68	0.33	0.59	0.35	0.64	1.13	0.56
2014	0.71	1.31	1.46	0.96	0.87	0.64	0.69	0.46	0.43	0.74	0.63	0.68
2015	0.99	1.15	1.40	1.46	1.39	0.93	0.75	0.46	0.66	0.28	0.89	1.57
YUCATAN												
YEAR	**JAN**	**FEB**	**MAR**	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	**DEC**
2001	1.13	0.89	0.72	0.80	0.48	0.80	0.64	0.67	0.72	0.71	0.77	0.72
2002	1.25	1.00	0.88	0.63	0.47	0.35	0.68	0.90	0.42	0.88	1.32	0.70
2003	1.24	0.70	0.73	0.83	0.58	0.62	0.66	0.71	0.82	0.77	0.80	0.88
2004	1.32	0.81	1.18	0.77	0.68	0.65	0.77	0.73	0.47	1.09	0.81	0.71
2005	1.02	1.10	1.11	0.57	0.58	0.47	0.79	0.68	0.67	0.73	0.84	1.08
2006	0.99	0.85	1.09	0.95	0.63	0.34	0.57	0.79	0.66	0.86	0.73	0.66
2007	0.93	0.77	0.77	0.88	0.50	0.80	0.71	0.62	0.55	0.83	0.72	0.84
2008	0.92	0.75	1.12	1.00	0.82	0.28	0.86	1.07	0.68	0.55	0.90	0.75
2009	0.95	0.85	1.19	0.80	0.56	0.42	0.70	0.46	0.48	0.52	0.76	0.64
2010	0.66	0.68	1.35	0.64	0.68	0.41	0.52	0.69	0.56	0.82	0.90	0.69
2011	0.94	0.78	0.60	0.68	0.65	0.40	0.93	0.74	0.51	0.64	0.89	0.62
2012	0.62	0.60	0.94	0.75	0.46	0.51	0.92	0.72	0.43	0.66	0.95	0.73
2013	0.95	0.98	1.08	0.83	0.60	0.51	0.88	0.65	0.51	0.85	0.70	0.71
2014	0.92	1.02	0.45	0.91	0.59	0.71	0.46	0.36	0.57	0.64	1.17	0.59
2015	0.64	0.93	0.82	0.86	0.71	0.35	0.37	0.35	0.35	0.57	0.79	2.33

SANTA ROSA												
YEAR	**JAN**	**FEB**	**MAR**	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	**DEC**
2001	1.66	1.44	1.37	1.05	0.47	0.42	0.93	0.61	0.57	1.00	1.48	1.56
2002	1.98	1.60	1.34	1.08	0.55	0.58	0.75	0.66	0.78	0.83	1.34	1.90
2003	2.04	1.37	1.09	0.90	0.57	0.52	0.48	0.67	0.89	0.57	0.66	1.37
2004	1.96	1.72	1.55	0.94	0.62	0.55	0.69	0.68	0.95	0.86	1.47	1.69
2005	1.94	1.56	1.15	0.87	0.67	0.34	0.87	0.58	0.46	0.51	1.00	1.59
2006	1.80	1.69	1.71	0.95	0.83	0.60	0.79	0.72	0.73	0.94	1.09	1.54
2007	2.06	1.56	1.28	0.77	0.40	0.59	0.67	0.65	0.65	0.54	1.10	1.63
2008	1.96	1.49	1.38	1.09	0.55	0.63	0.56	0.76	0.54	0.53	1.33	1.41
2009	1.94	1.53	1.67	1.26	0.69	0.48	0.73	0.74	0.83	0.98	1.42	1.90
2010	1.86	1.14	1.23	0.68	1.17	0.47	0.37	0.58	0.49	1.05	1.23	1.89
2011	1.84	1.57	1.49	1.06	0.52	0.45	0.47	0.75	0.66	0.44	1.50	1.47
2012	2.16	1.61	1.64	1.06	0.38	0.73	0.76	0.77	0.83	0.92	1.56	1.85
2013	1.91	1.45	1.28	1.02	0.66	0.53	0.65	0.64	0.33	0.77	1.37	1.66
2014	2.22	1.73	1.38	0.82	0.92	0.73	0.91	0.65	0.63	0.66	1.37	1.68
2015	1.98	1.57	1.35	0.94	0.77	0.78	1.00	0.80	0.66	0.78	1.12	2.62
MATA SECA												
YEAR	**JAN**	**FEB**	**MAR**	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	**DEC**
2001	1.61	1.17	1.88	1.19	0.69	0.65	0.59	0.61	0.41	0.36	0.53	1.06
2002	1.04	0.99	1.18	1.22	0.64	0.64	0.58	0.60	0.57	0.47	1.03	0.56
2003	1.09	1.31	1.45	1.63	1.04	0.70	0.67	0.55	0.39	0.40	0.61	0.92
2004	0.47	0.41	1.30	1.69	1.39	0.91	0.76	0.73	0.46	0.39	0.68	0.83
2005	0.64	1.33	1.44	1.62	1.11	0.62	0.65	0.60	0.28	0.34	0.82	0.73
2006	1.69	1.39	0.91	1.78	1.44	0.86	0.55	0.56	0.49	0.56	1.00	1.26
2007	1.47	1.27	1.78	1.25	0.83	0.72	0.68	0.73	0.52	0.37	0.21	1.39
2008	1.13	1.11	0.85	1.38	1.12	0.86	0.80	0.65	0.35	0.29	0.17	0.74
2009	1.31	1.52	1.50	1.55	1.38	0.76	0.61	0.57	0.39	0.38	1.50	1.30
2010	1.52	1.09	1.30	1.46	1.45	1.01	0.81	0.74	0.50	0.39	1.05	0.60
2011	1.45	1.36	1.06	1.82	1.44	0.89	0.71	0.59	0.43	0.61	0.69	0.94
2012	1.51	1.32	1.61	1.06	0.64	0.55	0.59	0.68	0.37	0.41	0.61	1.65
2013	0.87	1.25	1.14	1.47	0.82	0.68	0.58	0.59	0.41	0.69	0.96	0.55
2014	1.88	1.75	1.44	1.40	0.97	0.63	0.61	0.56	0.37	0.45	1.19	1.06
2015	0.82	1.22	1.35	0.80	1.10	0.59	0.52	0.44	0.16	0.20	0.44	1.37

SAN MATIAS												
YEAR	**JAN**	**FEB**	**MAR**	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	**DEC**
2001	0.15	0.09	0.59	0.28	0.36	0.49	0.30	0.07	0.06	0.08	0.23	0.39
2002	0.36	0.17	0.35	0.28	0.28	0.37	0.29	0.11	0.12	0.05	0.18	0.18
2003	0.24	0.29	0.26	0.57	0.49	0.52	0.41	0.42	0.25	0.19	0.21	0.08
2004	0.22	0.23	0.21	0.32	0.61	0.53	0.62	0.25	0.08	0.24	0.35	0.21
2005	0.34	0.28	0.29	0.14	0.45	0.28	0.55	0.29	0.28	0.19	0.14	0.24
2006	0.43	0.27	0.25	0.30	0.68	0.55	0.27	0.28	0.33	0.23	0.21	0.52
2007	0.42	0.22	0.48	0.34	0.50	0.39	0.48	0.39	0.08	0.16	0.35	0.34
2008	0.28	0.45	0.40	0.91	0.70	0.85	0.32	0.25	0.24	0.20	0.25	0.21
2009	0.41	0.31	0.50	0.67	0.71	0.91	0.81	0.75	0.38	0.36	0.24	0.34
2010	0.52	0.66	0.91	0.83	0.80	0.56	0.46	0.29	0.15	0.38	0.74	0.62
2011	0.56	0.28	0.57	1.46	1.37	0.95	0.81	0.39	0.28	0.20	0.25	0.26
2012	0.48	0.78	0.75	0.64	0.91	1.08	0.99	0.30	0.21	0.27	0.26	0.35
2013	0.34	0.40	0.53	0.77	0.82	0.77	0.81	0.44	0.25	0.48	0.49	0.33
2014	0.60	0.30	0.86	1.45	1.08	1.39	1.22	0.67	0.14	0.04	0.34	0.46
2015	0.56	0.37	0.43	0.66	0.73	0.85	1.01	0.16	0.20	0.30	0.30	0.76
TUCAVACA												
YEAR	**JAN**	**FEB**	**MAR**	**APR**	**MAY**	**JUN**	**JUL**	**AUG**	**SEP**	**OCT**	**NOV**	**DEC**
2001	0.13	0.17	0.60	0.21	0.32	0.55	0.31	0.07	0.08	0.04	0.15	0.34
2002	0.21	0.21	0.20	0.21	0.25	0.47	0.35	0.12	0.14	0.05	0.12	0.19
2003	0.15	0.20	0.21	0.60	0.56	0.32	0.44	0.38	0.26	0.18	0.22	0.14
2004	0.16	0.23	0.12	0.30	0.72	0.58	0.66	0.34	0.12	0.21	0.30	0.19
2005	0.30	0.16	0.20	0.23	0.52	0.38	0.59	0.27	0.41	0.25	0.19	0.20
2006	0.36	0.11	0.15	0.26	0.57	0.77	0.38	0.31	0.35	0.24	0.17	0.40
2007	0.33	0.18	0.30	0.24	0.62	0.50	0.57	0.48	0.08	0.20	0.55	0.32
2008	0.28	0.28	0.39	0.72	0.71	0.87	0.27	0.28	0.29	0.16	0.50	0.21
2009	0.50	0.29	0.52	0.70	0.59	0.93	0.85	0.89	0.56	0.47	0.15	0.35
2010	0.58	0.39	0.65	0.46	0.77	0.61	0.35	0.35	0.25	0.40	0.49	0.26
2011	0.58	0.14	0.84	1.22	1.09	0.69	0.73	0.42	0.36	0.16	0.13	0.21
2012	0.35	0.86	0.55	0.48	1.00	0.94	0.87	0.33	0.23	0.33	0.15	0.21
2013	0.30	0.32	0.33	0.55	0.70	0.82	0.88	0.59	0.29	0.39	0.42	0.21
2014	0.47	0.26	0.69	1.11	0.76	1.20	1.23	0.77	0.15	0.06	0.28	0.27
2015	0.27	0.27	0.41	0.74	0.64	0.83	1.19	0.26	0.23	0.36	0.29	0.65

Appendix 2. Comparison of different estimates of CO_2 sequestration rates from previous studies for different types of forests. The type of indicator selected and method of quantification are also presented. For TDFs stages of growth such as early (E), intermediate (I) and late (L) are also identified.

Forest Type	Sequestration rates	Indicator	Method	Author
Tropical semi-deciduous forest	78 t CO_2 ha^{-1} yr^{-1}	Above ground biomass	Forest inventory data and allometric equations	Brown (1997)
Tropical moist forest	230 CO_2 ha^{-1} yr^{-1}	Above ground biomass	Forest inventory data and allometric equations	Brown (1997)
Tropical wet forest	194 - 214 CO_2 ha^{-1} yr^{-1}	Above ground biomass	Forest inventory data and allometric equations	Brown (1997)
Tropical Pan American forest	129 CO_2 ha^{-1} yr^{-1}	Biomass	Forest inventory data	Achard (2004)
Tropical Amazonia	186 CO_2 ha^{-1} yr^{-1}	Biomass	Forest inventory data	Achard (2004)
TDF	(E) 31,8 Mg CO_2 ha^{-1} yr^{-1} (I) 60,9 Mg CO_2 ha^{-1} yr^{-1} (L) 88,9 Mg CO_2 ha^{-1} yr^{-1}	Above ground biomass	Allometric equations	Kalacska et al. (2008)
TDF	(E) 11.8 Mg CO_2 ha^{-1} yr^{-1} (I) 32.6 Mg CO_2 ha^{-1} yr^{-1} (L) 27.8 Mg CO_2 ha^{-1} yr^{-1}	Annual NPP	CASA Model	Cao et al. (2016)