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Tropical Forests

The Challenges of Maintaining Ecosystem Services while Managing the Landscape

Edited by Juan A. Blanco,
Shih-Chieh Chang and Yueh-Hsin Lo

TROPICAL FORESTS - THE CHALLENGES OF MAINTAINING ECOSYSTEM SERVICES WHILE MANAGING THE LANDSCAPE

Edited by **Juan A. Blanco, Shih-Chieh Chang**
and **Yueh-Hsin Lo**

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Contents

Preface IX

Chapter 1 **Introductory Chapter: Land Use Change Ecosystem Services and Tropical Forests 1**
Shih-Chieh Chang, Juan A. Blanco and Yueh-Hsin Lo

Chapter 2 **Fauna Diversity in Tropical Rainforest: Threats from Land-Use Change 11**
Mohamed Zakaria, Muhammad Nawaz Rajpar, Ibrahim Ozdemir and Zamri Rosli

Chapter 3 **Detection of Amazon Forest Degradation Caused by Land Use Changes 51**
Paul Arellano, Kevin Tansey and Heiko Balzter

Chapter 4 **Analyzing the Contribution of Cameroon’s Council Forests to Climate Change Mitigation and Socioeconomic Development 79**
Dieudonne Alemagi, Lalisa Duguma, Peter Minang, Anderson Kehbila, Martin Yemefack and Zac Tchoundjeu

Chapter 5 **Analysis of Precipitation and Evapotranspiration in Atlantic Rainforest Remnants in Southeastern Brazil from Remote Sensing Data 93**
Gabriel de Oliveira, Elisabete C. Moraes, Nathaniel A. Brunsell, Yosio E. Shimabukuro, Luiz E.O.C. Aragão, Guilherme A.V. Mataveli and Thiago V. dos Santos

Chapter 6 **Ecological and Environmental Aspects of Nutrient Cycling in the Atlantic Forest, Brazil 113**
Márcio Viera, Marcos Vinicius Winckler Caldeira, Franciele Francisca Marmentini Rovani and Kallil Chaves Castro

Preface

Our planet is rapidly being modified at all its levels. Our use of fossil fuels is leading to changes in the composition to the atmosphere, which combined with our generation of residues is leading to changes in the composition of the oceans. We are also changing the ecological composition of many ecosystems by moving (by purpose or by accident) many plant, animal and microbe species around the globe. In addition, humans have changed whole landscape, substituting forests by grasslands, drying wetlands for croplands, abandoning farmlands to let forest regrow, flattening mountains for urban expansion, etc. This process is called land use change, an important part of the global change that has led to the beginning of a new geological era: the Anthropocene.

Tropical forests in particular are among the ecosystems most affect by land use change processes in the last decades. Clearing the forests for timber extraction, cropland establishment, cattle ranching, urban expansion, etc., are modifying important ecological processes and structures. Such changes are causing the lost or reduction of many ecosystem services provided by the tropical forests.

Tropical forests provide goods such as timber, fuel, food, medicinal plants, or water. They also regulate vital processes such as nutrient cycling, hydrological cycling, greenhouse gases, etc. In addition, tropical forests provide the framework for economic and cultural development of the communities living in them.

Keeping in mind the considerations described above, the aim of this book is to provide a set of examples that we hope can serve as an introduction for the readers to the work currently done in the tropics in the face of such important challenges. Although brief, we hope this book can be used as an introductory text on the importance and complexity of assessing ecosystem services in tropical forests as the planet enters in the new age of Anthropocene.

Biodiversity is therefore the pillar that sustains the ecosystem services provided by tropical forests. To understand the high biodiversity that tropical forests contain, the first chapter by Zakaria et al., provides detailed composition lists of flora and fauna that can be found in tropical forests. The authors also explore and comment how land use change can affect the faunal diversity in tropical rainforests.

To understand the current dimensions of land use change processes currently have in tropical forests, the chapter by Arellano et al. describes their work to detect land use change through satellite imagery. The authors used both field and satellite optical methods to measure chlorophyll content to detect the degradation caused by land use change. The authors also offer a non-destructive method to measure chlorophyll content and compare it with traditional methodology, commenting on the advantages and weaknesses of both techniques.

The chapter by Alemagi et al. describe the provision services of tropical forests in Cameroon. The authors discuss the contribution of council forests in Cameroon from the aspects of climate change mitigation and socio-economic development. They found these forests offered high values on both timber and non-timber benefits, and also serve as carbon stocks to help to mitigate the climate change. This chapter also provides an example of how tropical forests provide supporting services for cultural and economic development of the communities living on them.

Moving into an example of changes in regulatory services caused by land use change in tropical forests, the chapter by Oliveira et al. focuses on characterizing and analyzing the temporal dynamics of precipitation and evapotranspiration in the Atlantic rainforest of Brazil in the first decade of 21st century. By using the global remote sensing data and disaggregate into regional scale, they were able to analyze the changes in the hydrological variables in this region that can be linked to changes in land use change.

As an additional example of regulatory services provided by tropical forests, the book closes with a discussion on the role of land use change in the alteration of nutrient cycling by Viera et al. The authors explored nutrient cycling in the Atlantic forest of Brazil from both ecological and environmental aspects. In this chapter, the authors offer a general description about the biome and how different nutrients transferred in the ecosystem, and how land use change could affect them.

All things considered, these five chapters provide a first glimpse of the current research done on tropical forests and land use change processes. They are an introduction to the research being done around the globe in connection to this topic. We hope the readers from academia, management, conservation, and any other stakeholders will enjoy reading this book and regard it as an initial source of information and study cases on what is the role that biodiversity plays in ecosystems.

The Editors want to finish this preface acknowledging the collaboration and hard work of all the authors. We are also thankful to the Publishing Team of InTech for their continuous support and assistance during the creation of this book. Special thanks are due to Mrs. Andrea Koric for inviting us to lead this exciting project and for coordinating the different editorial tasks. Last but not least, we want to acknowledge InTech´s and the authors´ generosity and social commitment by making research from tropical and developing countries available for free.

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Chapter 1

Introductory Chapter: Land Use Change Ecosystem Services and Tropical Forests

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Additional information is available at the end of the chapter

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1. Introduction

Large regions of different ecosystems around the world (forests, grasslands, wetlands, farmlands, water bodies) are being managed for different uses, usually implicating the substitution of one ecosystem type for another. This process, known as land use change, is driven by the need to provide food, fiber, water, and shelter to more than seven billion people. Land use change has therefore moved from being a local environmental issue to becoming one of the most important causes of global change [1]. However, such changes in how humans use the land have caused global croplands, pastures, plantations, and urban areas to expand their surfaces in recent decades. In other words, humans are using an increasing share of the planet surface and its resources, accompanied by large increases in energy, water, and fertilizer consumption, along with considerable losses of biodiversity. As a consequence, ecosystems' structures and functions are being increasingly altered, potentially undermining the capacity of ecosystems to sustain food production, maintain freshwater, regulate climate and air quality, ameliorate infectious diseases, and provide a large list of ecosystem services, usually as ignored as important they are [1].

We therefore face the challenge on how to maintain ecosystem services provided by tropical forests, while at the same time tropical regions experience important land use changes. The challenge is made even more complex by the difficulty of providing rules of thumb that can be easily applied across many different types of tropical forests. Differences between regions in forestry and agricultural management, good consumption, trade, culture and of course in ecological structure and function make generalization almost impossible.

Globally, forest cover has been reduced by 7–11 million km^2 over the last 300 years, mainly to make room for agriculture and timber extraction [2, 3]. On the other hand, the increase in technification and market development has led to the expansion of intensively planted forests, first in North America and Europe, but increasingly in South America, Africa, and the Asia-Pacific region, covering now 1.9 million km^2 worldwide [4]. Although impressive, only the 3%

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of the world forest land is covered with productive forest plantations. However, this area expanded by 2 million ha annually in the 1990s and by 2.8 million ha in the 2000s [5].

All forest regions (tropical, subtropical, temperate, sub-boreal, and boreal) are being affected by land use change processes. In particular, tropical forests have suffered from the biggest changes (both positive and negative) of all the forest types although the loss rate is still 3.6 times bigger than the rate of surface gain [6]. These authors estimated that losses in tropical forests area accounted for 32% of total forest loss in the world, with half of those losses being concentrated in South American tropical forests. However, there are big differences among tropical countries in rates of loss and gain of forest area. For example, Brazil has recently shown a decline in annual forest area loss, moving from a high of over 40,000 km^2 $year^{-1}$ in 2004 to a low of under 20,000 km^2 $year^{-1}$ in 2011. On the other side, for the same period Indonesia has gone from losing 10,000 km^2 $year^{-1}$ in 2003 to over 20,000 km^2 $year^{-1}$ in 2012. In addition, subtropical forests are experiencing important land use change, with many planted forests being usually treated as crops, causing that old-growth natural forests to be relatively rare in these biomes [7]. As a result, although the absolute losses in surface are not as big as in the tropics, subtropical forests have experienced the largest relative changes in forest cover losses and the smallest relative gains [6].

Tropical forests have been extensively disturbed by human beings since long time, and the intensity and extent of disturbance will continue into the future [8]. Land use change in the tropics is caused mainly for agricultural use [9]. Land use change will affect ecosystem services, and climate change makes this a more complicated but emergent problem for human beings [10]. Many land use practices still widely extended in tropical forests (e.g., fuel-wood collection, forest grazing, and road expansion) can degrade forest ecosystem conditions—in terms of productivity, biomass, stand structure, and species composition—even without changing forest area. Changing the way the land is used also paves the way for the introduction of invasive species, including pests and pathogens that can degrade the original forests. Another major change is the alteration of fire regimes, by modifying fuel loads, removing coarse woody debris, increasing the number and frequency of ignition sources, and even modifying the local meteorological conditions [11]. On the other hand, human activity can also improve forest conditions, either by direct forest management or by unintended effects of other processes, such as increased nitrogen deposition, atmospheric concentrations of CO_2, and peatland drainage. Such processes have caused the increase in standing biomass of European forests by 40% between 1950 and 1990, while their area remained largely unchanged, accelerating forest growth in the twentieth century [12]. These forests have become a substantial sink of atmospheric carbon [13], although other ecosystem services including those provided by peatlands and biodiversity are likely diminished.

2. Land use change and biodiversity

All kinds of ecosystem services rely on the interplay of the organisms and the abiotic environmental factors of the ecosystems. Therefore, biodiversity of an ecosystem is the key property behind ecosystem services. Globally, the biodiversity is decreasing mainly due to the anthropogenic interferences [14]. Land use change has its first and direct impact on the land surface with the modification or removal of current organisms and thus will change the biodiversity to some extent. In the recent analysis of the intactness of biodiversity, as defined as the proportion of natural biodiversity remaining in local ecosystems, Newbold et al. [15] indicated that the 58% of the planet´s terrestrial ecological boundaries have been crossed. The main cause of this problem is the extensive land use changes that have disconnected natural ecosystems and rounded them up with human-made landscapes.

Land use change from forests worldwide has made ecosystem fragmentation a serious problem. Currently, 70% of the forest cover on Earth is within 1 km from the edge of the forests [16], indicating the loss of connectivity and the vulnerability to further disturbances. In a detailed modeling [17], the spatial patterns of fragmentation in Brazil were shown to have a strong effect on the final extent of influences on ecosystem services like biodiversity. For example, the farmland expansion on the forest edge would have much less impact on biodiversity and carbon storage compared to the farmland increase in the center of a forest. In the case of bird species richness, the fragmentation regime of forests plays a key role. Bregman et al. [18] analyzed the sensitivity to fragmentation of different bird species worldwide and found that the insectivores and large frugivorous are more negatively affected in larger forest fragmentations. This pattern is especially significant in the tropical area.

Barnes et al. [19] demonstrated a 45% reduction in soil invertebrate biodiversity after the conversion of tropical rainforests to oil palm plantations. They further calculated the change in ecosystem energy flux due to this land use change and found a surprisingly lower energy flux in oil palm plantations (51%) relative to what happens in the rainforest. Changes in biodiversity at the functional group level were also evident in a case study in Malaysian Borneo [20]. When comparing the community composition of dung beetles along a land use change gradient from primary forest to logged forest and oil palm plantation, the composition did change substantially. However, significant reduction in functional diversity only happened in the oil palm plantation.

Land use change modifies not just the biodiversity of higher plants and animals, but also that of microorganisms. Paula et al. [21] demonstrated that the change from Amazonian rainforests to pastures would decrease the microbial functional gene richness and diversity. The recovery from the disturbed lands to secondary forests may make the functional gene richness and diversity again similar to that in the primary forests, although not totally alike.

There are many different types of classifying ecosystem services, but a basic classification divides them into three main categories [22]. First, provisioning services are those related to goods generated by the forests that can be directly consumed: timber, food, water, fuel, medicinal plants, etc. Second, regulatory services are those that regulate the conditions in which humans inhabit the land and in which our economic activities take place: climate regulation, flood control, etc. Third, cultural services such as spiritual connection, recreation opportunities, cultural legacy, and sense of belonging are connected to ecosystems.

3. Provisioning services

Tropical forests maintain a high variety of plants, animals and microbes, and therefore many different species suitable for human consumption. In addition, to be a genetic reservoir for potential food sources [23], tropical forests can provide enough food to maintain the human population of traditional habitants [24], reaching values up to US $18.5 per hectare and year [25]. Fuelwood is also the main energy source for heating and cooking of millions of people in tropical countries. For example, in Mexico alone, 7 million of rural people depend on tropical forests [26]. Timber, usually of high quality and value, is among the most valued goods provided by tropical forests, sometimes being also the cause of the deforestation (often illegal) and land use change [27]. Similarly, traditional medicine from tropical communities is also providing new compounds for medicines, but at the same time can also cause local extinctions if their harvest is not controlled [25].

Among other goods, water is usually given from granted, but freshwater is a very valuable ecosystem service that comes mainly from higher elevation ecosystems. Ponette-González et al. [28] performed a meta-analysis of the effects of land use change on hydrological cycles of tropical high-elevation ecosystems. The types of land use change included the conversions from forest to grassland, agroforest to nonforest, nonforest to tree plantation, and recent glacier retreat. The deforestation did not lead to an expected substantial increase in downstream runoff in Latin America and the Caribbean and in Hawaii. On the other hand, Muñoz-Villers and McDonnell [29] compared the streamflow of three watersheds that have old-growth cloud forest, 20-year-old regenerated cloud forest, and heavily grazed pasture, respectively, in Mexico. The land use type of pasture produced 10% higher streamflow compared to the two forested catchments. Their results imply that a short period of 20 years of recovery from pasture to forest may be enough for the restoration of hydrological conditions.

4. Regulation services

Through plant-soil-atmosphere interactions, tropical forests have a major role in regulating atmospheric gases and therefore climate. Carbon emissions due to deforestation in the tropics were 810 Tg C year^{-1} between 2000 and 2005 [30], in which Brazil and Indonesia were the first two contributing countries with an emission rate of 340 and 105 Tg C year^{-1}, respectively. Soil carbon loss due to land use change in the tropical area was estimated to be 79 Pg CO_2 during the past 150 years (1860–2101, averaged from three different models) [31].

Peat swamp forests in Southeast Asia are an important carbon stock due to their predominant wet soil condition. However, the need for more farmland has largely changed the peatlands into different agricultural uses such as rice fields and oil palm plantations. Hergoualc'h and Verchot [32] demonstrated a very clear change in greenhouse gases (CO_2 + CH_4 + N_2O) budgets when original peatlands were converted to six different land use types including degraded forest, croplands and shrublands, rice fields, oil palm plantation, *Acacia crassicarpa* plantation, and Sago palm plantation. On average, the undisturbed peatlands are the strongest CH_4 source, which, however, could be offset by the CO_2 sink strength and thus remain the only net greenhouse gas sink of the magnitude of -1.3 ± 5.9 Mg CO_2-Eq ha^{-1} year^{-1}. The conversion of peatland into *Acacia crassicarpa* plantation turns the sink into the largest source of 72.0 ± 12.8 Mg CO_2-Eq ha^{-1} year^{-1}.

Coastal mangroves in many tropical countries have been destroyed and the land been used for aquafarming or other purposes like harbor construction. Kauffman et al. [33] showed an extremely high carbon emission accompanying the conversion of mangroves to shrimp ponds in the Dominican Republic. The carbon stocks of mangroves ranged from 706 to 1131 Mg C ha^{-1}, while that in the abandoned shrimp ponds were only 95 Mg C ha^{-1}. The estimated carbon emission of 2244–3799 Mg CO_2-Eq ha^{-1} was among the largest carbon emission due to land use change [33].

Land use change in tropical forests can also have indirect effects of the capacity of the ecosystems to regulate processes in water ecosystems. For example, land use change in a tropical watershed could change the decomposition rate of organic matter in tropical rivers [34].

Tropical forests also mitigate extreme weather. Structural complexity [35], together with other factors such as microtopography and soil features, modulates the impacts of extreme events [36]. In a model simulation of the precipitation regime under combined factors of land use change (transformation of rain forests to pasture) and different levels of soil water availability in the Amazonian rain forests, Bagley et al. [37] showed a clear reduction in precipitation and increase in drought degree under deforestation scenarios.

Tropical forests can also regulate air quality. Changes in air quality and atmospheric chemistry often arise when land use type has changed because the land-atmosphere fluxes of material and energy are to a certain extent vegetation-specific processes (e.g., see [38]). For example, isoprene is a biogenic volatile organic compound that emits naturally from forest vegetation. By deforestation, the emission of isoprene will decrease and the subsequent photochemical process of ozone formation will also decrease, leading to a decreased ozone deposition in the Amazonian rainforests [39]. On the other hand, the agricultural use of the deforested area has been shown to emit more NO_x to the atmosphere, mostly due to the higher N-fertilizer application.

In some tropical region, slash-and-burn is still a predominant method to create farmland [40]. The emissions from fires and smokes often cause regional problems of air quality. Marlier et al. [41] pointed out an important finding that ca. 80% of 2005–2009 fire emissions from Sumatra were related to degradation or land use maintenance. The fire emissions from land use conversion thus may have longer-term effect on the air quality.

5. Trade-off between different ecosystem services

Land use change may result in the increase in some ecosystem services but at the same time the reduction in other services. Such trade-offs always occur when management practices are oriented towards the production or use of a given ecosystem service, without taking into account the consequences for other services [23]. For example, the more forest that is transformed, services provided by plant-dominated ecosystems such as farmlands or pasture lands increase, with the production of agricultural and pastoral goods being increased, whereas the services provided by the tree-dominated forests decline. For example, Leh et al. [42] used InVEST model (Integrated Valuation of Ecosystem Services and Tradeoffs) to quantify the spatial pattern of ecosystem services including biodiversity, surface water yield, carbon storage, sediment retention, nitrogen retention, and phosphorous retention in the tropical African countries Ghana and Cote d'Ivoire. The land use scenarios from 2000 to 2005 and 2009 were used to analyze the change in those ecosystem services. By employing this tool, it is possible to quantitatively understand the change in ecosystem services at different spatial scales and thus makes the planning of land use strategy possible. The results of Leh et al.'s work emphasize the great challenges that we face to maintain ecosystem services provided by tropical forests, while land use change processes are becoming increasingly more important.

Another example of these complex trade-offs is the effect of land use change on freshwater availability when transforming tropical forests into other type of ecosystem. In theory, grasses and shrubs use less water than trees, having therefore lower evapotranspiration rates (Oliveira et al. this volume). This could lead to higher runoff and increased provision of water downstream [23]. However, clearing tropical forests also reduces infiltration rates, increasing erosion, soil evaporation, and runoff, which in turn can lead to reduction in water quality and decrease in water recharge rates (see above). The importance of trade-offs also appears when considering that ecosystem services also depend on the users: different stake holders value different services in different ways, and therefore, it is difficult to objectively determine whether a land use change is diminishing or increasing the provisioning of ecosystem services. It would depend on who is asked [23].

6. Final considerations

Tropical forests offer services of provision, regulation, and culture that are fundamental for the well-being of the societies that inhabit them, and for extension of all the Earth's inhabitants. The large extension and important biodiversity of these forests contribute to offer critical services for our society, which are being constantly modified by the management decisions that are part of the dynamics of human society. Food demand is one of the sectors that are related to flood control and climate regulation that tropical forests provide to a large section and the whole humanity, respectively. Management interventions such as forest restoration or payments for ecosystem services can help to recover or maintain ecosystem services that tropical forests offer.

Considering all the things, maintaining ecosystem services provided by tropical forests in the face of increasing land use change is a truly challenging task. Such task must start by understanding the components that make each tropical forest unique and how those components are linked and interact to create the ecological processes that maintain (and are maintained by) tropical forests. Then, understanding how human activities (economic, cultural, etc.) are dependent on such processes is the necessary step to analyze, and take decisions about, the consequences of land use change on the ecosystem services provided by tropical forests. It is time to address this challenge.

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Chapter 2

Fauna Diversity in Tropical Rainforest: Threats from Land-Use Change

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Additional information is available at the end of the chapter

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Abstract

Tropical rainforests are the cradle of life (perfect conditions for life) on Earth, i.e., rich in plant species composition (>250 plant species/hectare) and fauna diversity (>50% of animal species in the world). Rainforests occur near the Earth's equator and cover 6% of the Earth's surface across the tropical regions and are characterized by wet climate, i.e., heavy rainfall (125—660 cm), relative humidity (77—88%) and temperature (20—34°C). They are dominated by a wide range of broad-leaved trees that form dense canopy and the most complex ecosystem. Currently, the tropical rainforest ecosystem is changing faster than ever in human history due to anthropogenic activities, such as habitat loss and degradation due to deforestation for timber and conversion into agriculture fields (oil palm plantation), mining, fire, climate change, etc. The habitat loss and degradation had adversely influenced the distribution and richness of the fauna species. The current information on the fauna diversity of tropical rainforest is not sufficient and in the future, more research is required to document the various community parameters of the fauna species in order to conserve and protect them. For better future, conservation, and management, we must identify the major drivers of changes and how these factors alter the tropical rainforest.

Keywords: fauna, diversity, rainforest, landscape, vegetation

1. Introduction

Tropical rainforest usually occurs 10° north and south of the equator, where climate conditions are unique such as humid, warm, and wet. The monthly mean temperature is 18°C and the annual rainfall is not less than 168 cm. Tropical rainforest occurs in four main regions; Central and South America, Central and West Africa, Indo-Malaya and Australia [1]. They are storehouses of a range of food resources for a wide variety of fauna species as well as for human beings, raw material for buildings, and medicines [2, 3] and affect the climate [4, 5].

1.1. Ecological importance of tropical rainforest

Tropical rainforests are the most diverse in the vegetation structure and composition (**Figure 1**) that supported a diversity of fauna species such as birds, reptiles, mammals, amphibians, and invertebrates, which directly or indirectly depend on them for their survival and existence. They are rich in habitat diversity and provide a variety of resources for the avian species, such as food, habitat, and shelter [6]. Tropical rainforest is vital ecosystem, i.e., it provide crucial ecosystem services such as raw materials, reservoirs of biodiversity, soil protection, sources of timber, medicinal plants, carbon sequestration, and watershed protection [7–9].

1.2. Threats to tropical rainforest

Tropical rainforest covers less than 10% of the land area of the Earth, representing the largest biological diversity reservoir, i.e., >50% of known plant species grow in tropical rainforest. Despite being rich in fauna diversity, every year, huge areas of tropical rainforests are being lost and degraded due to human interference [10–15]. It has been stated that 25–50% of the world's tropical rainforest has been lost and degraded due to the land-use change such as

Figure 1. Aesthetic view of tropical rainforest.

deforestation for palm oil plantations, agriculture expansion, cattle ranches, mining, and development of housing societies [16–19], while the rest of the rainforest areas is under a major shift in the dynamic structure and productivity.

It has been reported that Southeast Asia had the highest rate of land-use change (such as deforestation of tropical rainforest for conversion into oil palm plantation, commercial logging for timber and development of human settlement) as compared to other regions [12, 20–22]. Deforestation and fragmentation due to agriculture expansion, human settlement, logging, and fire had altered the plant species composition, richness, and diversity [23–26]. Deforestation and fragmentation, over-exploitation, invasive species, and climate change are the major factors due to which the biodiversity of tropical forest had declined at an alarming rate. For example, some of the fauna species became extinct, while others became threatened and vulnerable due to habitat loss, fragmentation, and degradation.

Changes in the vegetation structure and composition due to deforestation and fragmentation may alter the habitat suitability and food productivity. Habitat suitability, i.e., vegetation structure, species composition, species richness, canopy layers, and food productivity are key drivers, which predominantly influence fauna community parameters such as species composition, relative abundance, species richness, species diversity, and the density of tropical rainforest. Furthermore, it has been stated that the deforestation in humid tropic may be in the range of 4.9–5.7 million ha/year. Likewise, each year, 2.3 million ha of humid forests had been degraded due to logging and fire activities. Similarly, around 2.2 million ha/year tropical moist deciduous and 0.7 million ha/year tropical dry forest has been deforested due to anthropogenic activities [27].

Deforestation may cause habitat loss and fragmentation that adversely affect the population and the community parameters such as species composition, relative abundance, species richness, species diversity, and density of different wildlife species [28–30]. However, the effect of habitat loss and fragmentation on the wildlife species may vary depending on remaining vegetation and the surrounded landscape [31, 32].

Land-use change such as deforestation, i.e., depletion of tree crown cover due to conversion of forested areas in agricultural fields, human settlements, excessive logging, and road constructions are major factors of habitat loss and degradation [10, 33, 34]. The habitat loss and degradation are responsible for biodiversity loss [35], low production of food, and habitat fragmentation [27, 36, 37] that ultimately affects different fauna species. Due to deforestation, large areas become isolated, i.e., temporal refuge, which serves as corridors for different wildlife species, especially bird species [38–40].

1.3. Floral composition

Tropical rainforests are the most rich tree species forest on the Earth and encompasses of broad-leaved trees with large buttress, and covered with climbers, epiphytes, and hemi-epiphytes. They have multi-layered canopy, i.e., upper, middle, and dense understory vegetation composition and are rich in diversity of flora and fauna, especially birds, mammals, reptiles, amphibians, and invertebrates [41–44]. Tropical rainforest is blessed with an enormous variety of flora species. The vegetation species composition of rainforests encompasses of four distinct layers of trees, namely; emergent, upper canopy, understory, and forest floor.

1.3.1. Emergent vegetation layer

Emergent or sunlit layer is dominated by broad-leaved, hardwood and evergreen. The trees may attain the height from 30.48 to 76.2 m and a trunk size up to 4.48 m around. The winds and sunlight are major environmental factors, which play a significant role (such as pollination and seed dispersal) in the tropical rainforest management ecosystem. The emergent layer is rich in the fauna species, such as birds (hummingbirds, macaw, harpy eagle, etc.), mammals (i.e., monkeys, bats, etc.), snakes, and insects such as butterflies, moths, etc. The birds and insects play a crucial role in the pollination of tropical rainforest plant species. The microclimate of this layer often fluctuates from time to time depending upon temperature and wind speed.

1.3.2. Canopy layer

The canopy is the main layer of tropical rainforest ecosystems, which is thick and dense like an umbrella. This layer is composed of a variety of vegetation structures and tree species composition such as philodendron, *strychnos toxifera*, rattan palms, etc. The trees may grow up to 18.29—27.42 m above the forest floor. Epiphytes such as orchids, mosses, ferns, and lichens are a common feature of this layer, which grow on tree trunks and branches. The canopy layer is rich in food diversity and an ideal habitat for a wide range of fauna species such as birds, mammals, reptiles, amphibians, and diversity of insect species. The members of fauna species are often observed flying, jumping, gliding, and hoping for canopy gaps.

1.3.3. Understory layer

The understory layer encompasses usually small trees, shrubs, ferns, and native bananas, which may attain 3.66 m height. Mosses, fungi, and algae often grow on the trees. This layer is rich in insects, such as bees, stick insects, ants, beetles, and butterflies, which serve as sources of food for a wide array of birds and reptiles. The fauna species encompass bats, monkeys, snakes, lizards, jaguars, frogs, and invertebrates.

1.3.4. Forest floor

This is the bottom layer of tropical rainforest. This layer is dark due to dense ground vegetation and only 2% of sunlight reaches the floor. Due to less availability of sunlight, only few plant species can grow. This layer is rich in organic matter such as fallen leaves, seeds, fruits, and branches. Furthermore, this layer is rich in fungi and mosses. The fauna species of the forest floor include elephants, tigers, pumas, leopards, jaguars, ocelots, mongoose, tapirs, cassowaries, okapis, armadillos, pigs, and gorillas.

1.4. Environmental services provided by fauna in tropical rainforest

Faunas are the important component of the tropical rainforest ecosystem and provide a wide array of environmental services such as; they keep tropical rainforest systems in balance through pollinating a variety of plant species, dispersing seeds, controlling pest population and reducing the damage caused by different pest species, scavenging carcasses, and recycling nutrients back into the soil.

2. Fauna composition

Fauna species are not only confined to specific habitats but also utilize various habitats in search of food, shelter, and reproduction. Tropical rainforest is rich in fauna species such as birds, reptiles, mammals, amphibians, and invertebrates.

2.1. Bird species composition of tropical rainforest

Birds are highly motile animals, i.e., they may fly to different areas in search of food, shelter, and for breeding purposes. They are ecologically diverse and had occupied a wide array of habitats. Bird species depend on the vegetation structure and composition (such as trees, shrubs, and herbs) and food resources for their survival and reproduction [45, 46]. They are the functional group of tropical rainforest ecosystems as seed dispersers, pollinators, top predators, pest control, and scavengers [47–50].

Birds are conspicuous and an important component of tropical rainforest ecosystems, often exhibit distinction associated with vegetation structure and composition (**Figures 2–4**; **Table 1**). They are sensitive to habitat alteration and landscape modification [51–54]. This might be because the vegetation structure and composition may influence habitat selection and foraging efficiency of all birds. For example, large trees and ground dense herbaceous vegetation layers often harbor a higher avian abundance and diversity. This might be because old growth stands provide suitable nesting and breeding sites, plenty of food resources, and also provide

Figure 2. Asian paradise flycatcher—*Terpsiphone paradisi*.

Figure 3. Wallace's hawk eagle—*Nisaetus nanus*.

Figure 4. Oriental/Asian pied hornbill—*Anthracoceros albirostris*.

protection from predators and harsh weather [55, 56]. Likewise, ground vegetation also offers ideal habitat and safe breeding sites and shelter for different fauna species residing in dense ground cover vegetation, such as birds, mammals, reptiles, and amphibians. It has been illustrated that height and density of the tree [57], dense understory vegetation [58, 59], and logs and snags [60] are key elements, which affect avian distribution, richness, and diversity in tropical rainforest.

Habitat alteration due to land change use may alter the avian community parameters such as relative abundance, species richness, species diversity, and density [65]. This might be that bird community structure strongly associated with canopy openness and understory vegetation

Family	Scientific name	Common name	Habitat	Authors
Acanthizidae	*Gerygone chrysogaster*	Yellow-bellied Gerygone	Lowland tropical rainforest (Papua New Guinea)	[61]
	Gerygone magnirostris	Large-billed Gerygone	Tropical Rainforest (Australia)	[62]
	Gerygone sulphurea	Golden-bellied Gerygone	Tropical Rainforest (Malaysia)	[63]
Accipitridae	*Haliastur indus*	Brahminy Kite	Lowland tropical rainforest (Papua New Guinea)	[61]
	Harpyopsis novaeguineae	Papuan Harpy Eagle	Lowland tropical rainforest (Papua New Guinea)	[61]
	Henicopernis longicauda	Long-tailed Buzzard	Lowland tropical rainforest (Papua New Guinea)	[61]
	Accipiter virgatus	Besra	Tropical Rainforest (India)	[64]
	Spilornis cheela	Crested Serpent Eagle	Tropical Rainforest (India)	[64]
Aegithinidae	*Aegithina viridissima*	Green Iora	Hill dipterocarp tropical rainforest (Malaysia)	[6]
Alcedinidae	*Alcedo euryzona*	Blue-banded Kingfisher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Alcedo meninting	Blue-eared Kingfisher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Ceyx rufidorsa	Rufous-backed Kingfisher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Alcedo atthis	Common Kingfisher	Hill dipterocarp tropical rainforest (Malaysia), Lowland tropical rainforest (Papua New Guinea)	[6, 61]
	Alcedo azurea	Azure Kingfisher	Lowland tropical rainforest (Papua New Guinea)	[61]
	Alcedo pussio	Little Kingfisher	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ceyx lepidus	Variable Dwarf Kingfisher	Lowland tropical rainforest (Papua New Guinea)	[61]
	Melidora macrorrhina	Hook-billed Kingfisher	Lowland tropical rainforest (Papua New Guinea)	[61]
	Tanysiptera galatea	Common Paradise Kingfisher	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ceyx erithaca	Oriental Dwarf Kingfisher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
Apodidae	*Collocalia spodiopygius*	White-rumped Swiftlet	Tropical Rainforest (Australia)	[62]
Ardeidae	*Egretta garzetta*	Little Egret	Lowland tropical rainforest (Papua New Guinea)	[61]
	Gorsachius melanolophus	Malayan Night Heron	Tropical Rainforest (India)	[64]
Artamidae	*Cracticus cassicus*	Hooded Butcherbird	Lowland tropical rainforest (Papua New Guinea)	[61]
	Peltops blainvillii	Lowland Peltops	Lowland tropical rainforest (Papua New Guinea)	[61]

Family	Scientific name	Common name	Habitat	Authors
Bucerotidae	*Rhyticeros plicatus*	Papuan Hornbill	Lowland tropical rainforest (Papua New Guinea)	[61]
Bucerotidae	*Ocyceros griseus*	Malabar Grey Hornbill	Tropical Rainforest (India)	[64]
Cacatuidae	*Cacatua galerita*	Sulphur-crested Cockatoo	Lowland tropical rainforest (Papua New Guinea), Tropical Rainforest (Australia)	[61, 62]
Campephagidae	*Hemipus hirundinaceus*	Black-winged Flycatcher Shrike	Hill dipterocarp tropical rainforest (Malaysia)	[65]
	Boyer's cuckoo-shrike	Boyer's Cuckoo-shrike	Lowland tropical rainforest (Papua New Guinea)	[61]
	Campochaera sloetii	Golden Cuckoo-shrike	Lowland tropical rainforest (Papua New Guinea)	[61]
	Coracina melas	New Guinea Cuckoo-shrike	Lowland tropical rainforest (Papua New Guinea)	[61]
	Coracina novaehollandiae	Black-faced Cuckoo-shrike	Lowland tropical rainforest (Papua New Guinea)	[61]
	Coracina papuensis	White-bellied Cuckoo-Shrike	Lowland tropical rainforest (Papua New Guinea)	[61]
	Lalage leucomela	Varied Triller	Lowland tropical rainforest (Papua New Guinea), Tropical Rainforest (Australia)	[61, 62]
	Hemipus picatus	Bar-winged Flycatcher-shrike	Tropical Rainforest (India)	[64]
	Pericrocotus flammeus	Scarlet Minivet	Tropical Rainforest (India)	[64]
	Caprimulgus macrurus	Large-tailed Nightjar	Lowland tropical rainforest (Papua New Guinea)	[61]
Casuariidae	*Casuarius unappendiculatus*	Northern Cassowary	Lowland tropical rainforest (Papua New Guinea)	[61]
Chloropseidae	*Chloropsis cochinchinensis*	Blue-winged Leafbird	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Chloropsis cyanopogon	Lesser Green Leafbird	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Chloropsis aurifrons	Gold-fronted Leafbird	Tropical Rainforest (India)	[64]
Cisticolidae	*Prinia rufescens*	Rufescent Prinia	Hill dipterocarp tropical rainforest (Malaysia)	[65]
	Orthotomus atrogularis	Dark-necked Tailorbird	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Orthotomus sericeus	Rufous-tailed Tailorbird	Tropical Rainforest (Malaysia)	[63]
	Orthotomus sutorius	Common Tailorbird	Tropical Rainforest (Malaysia)	[63]
Colluricinclidae	*Colluricincla megarhyncha*	Little Shrike-thrush	Lowland tropical rainforest (Papua New Guinea)	[61]
	Colluricincla boweri	Bower's Shrike-thrush	Tropical Rainforest (Australia)	[62]

Family	Scientific name	Common name	Habitat	Authors
Columbidae	*Ducula pinon*	Pinon Imperial Pigeon	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ducula rufigaster	Purple Tailed Imperial Pigeon	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ducula zoeae	Zoe Imperial Pigeon	Lowland tropical rainforest (Papua New Guinea)	[61]
	Macropygia amboinensis	Brown Cuckoo-dove	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ptilinopus coronulatus	Coroneted Fruit Dove	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ptilinopus iozonus	Orange-bellied Fruit Dove	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ptilinopus perlatus	Pink-spotted Fruit Dove	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ptilinopus pulchellus	Beautiful Fruit Dove	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ptilinopus superbus	Superb Fruit-dove	Lowland tropical rainforest (Papua New Guinea)	[61]
	Reinwardtoena reinwardtii	Great Cuckoo-dove	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ptilinopus magnificus	Wompoo Fruit Dove	Tropical Rainforest (Australia), Lowland tropical rainforest (Papua New Guinea)	[61, 62]
	Chalcophaps indica	Emerald Dove	Tropical Rainforest (Australia), Isolated Tropical Rainforest (Malaysia)	[6, 62]
	Ducula badia	Mountain Imperial Pigeon	Tropical Rainforest (India)	[64]
	Geopelia striata	Zebra Dove	Tropical Rainforest (Malaysia)	[63]
	Streptopelia chinensis	Spotted Dove	Tropical Rainforest (Malaysia)	[63]
Coraciidae	*Eurystomus orientalis*	Common Dollarbird	Lowland tropical rainforest (Papua New Guinea)	[61]
Corvidae	*Platylophus galericulatus*	Crested Jay	Hill dipterocarp tropical rainforest (Malaysia)	[64]
	Corvus tristis	Bare-eyed Crow	Lowland tropical rainforest (Papua New Guinea)	[61]
	Dendrocitta leucogastra	White-bellied Treepie	Tropical Rainforest (India)	[64]
Cuculidae	*Cacomantis merulinus*	Plaintive Cuckoo	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Cacomantis variolosus	Brush Cuckoo	Lowland tropical rainforest (Papua New Guinea)	[61]
	Centropus bernsteini	Lesser Black Coucal	Lowland tropical rainforest (Papua New Guinea)	[61]
	Centropus menbeki	Greater Black Coucal	Lowland tropical rainforest (Papua New Guinea)	[61]

Family	Scientific name	Common name	Habitat	Authors
	Centropus phasianinus	Pheasant Coucal	Lowland tropical rainforest (Papua New Guinea)	[61]
	Chrysococcyx lucidus	Shining Bronze Cuckoo	Lowland tropical rainforest (Papua New Guinea)	[61]
	Eudynamys scolopaceus	Common Koel	Lowland tropical rainforest (Papua New Guinea)	[61]
	Probosciger aterrimus	Palm Cockatoo	Lowland tropical rainforest (Papua New Guinea)	[61]
	Scythrops novaehollandiae	Channel-billed Cuckoo	Lowland tropical rainforest (Papua New Guinea)	[61]
	Chrysococcyx minutillus	Little Bronze-Cuckoo	Tropical Rainforest (Australia)	[62]
	Cacomantis sonneratii	Banded Bay Cuckoo	Tropical Rainforest (Malaysia)	[63]
	Cuculus micropterus	Indian Cuckoo	Tropical Rainforest (Malaysia)	[63]
	Surniculus lugubris	Drongo Cuckoo	Tropical Rainforest (Malaysia)	[63]
Dendrocolaptidae	*Campylorhamphus pusillus*	Brown-billed Scythebill	Tropical forest (Costa Rica)	[65]
	Xiphorhynchus guttatus	Buff-throated Woodcreeper	Tropical forest (Costa Rica)	[65]
Dicaeidae	*Prionochilus percussus*	Crimson-breasted Flowerpecker	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Dicaeum trignostigma	Orange-bellied Flowerpecker	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Dicaeum geelvinkianum	Red-capped Flowerpecker	Lowland tropical rainforest (Papua New Guinea)	[61]
	Dicaeum hirundinaceum	Mistletoebird	Tropical Rainforest (Australia)	[62]
	Dicaeum concolor	Nilgiri Flowerpecker	Tropical Rainforest (India)	[64]
	Prionochilus maculatus	Yellow-breasted Flowerpecker	Tropical Rainforest (Malaysia)	[63]
Dicruridae	*Cheatorhynchus papuensis*	Pygmy Drongo	Lowland tropical rainforest (Papua New Guinea)	[61]
	Dicrurus bracteatus	Spangled Drongo	Lowland tropical rainforest (Papua New Guinea)	[61]
	Dicrurus aeneus	Bronzed Drongo	Tropical Rainforest (India), Isolated Tropical Rainforest (Malaysia)	[61, 64]
	Dicrurus paradiseus	Greater Racket-tailed Drongo	Tropical Rainforest (India), Isolated Tropical Rainforest (Malaysia)	[63, 64]
	Dicrurus annectans	Crow-billed Drongo	Tropical Rainforest (Malaysia)	[63]
	Dicrurus remifer	Lesser Racket-tailed Drongo	Tropical Rainforest (Malaysia)	[63]

Family	Scientific name	Common name	Habitat	Authors
Estrildidae	*Lonchura leucogastra*	White-bellied Munia	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Lonchura kelaarti	Black-throated Munia	Tropical Rainforest (India)	[64]
Eurylaimidae	*Calyptomena viridis*	Green Broadbill	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Cymbirhynchus macrorhynchos	Black-and-red Broadbill	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Eurylaimus javanicus	Banded Broadbill	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Corydon sumatranus	Dusky Broadbill	Tropical Rainforest (Malaysia)	[63]
Falconidae	*Microhierax fringillarius*	Black-thighed Falconet	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Herpetotheres cachinnans	Laughing Falcon	Tropical forest (Costa Rica)	[65]
Formicariidae	*Thamnophilus bridgesi*	Black-hooded Antshrike	Tropical forest (Costa Rica)	[65]
Furnariidae	*Automolus ochrolaemus*	Buff-throated Foliage-gleaner	Tropical forest (Costa Rica)	[65]
Halcyonidae	*Lacedo pulchella*	Banded Kingfisher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Todiramphus sanctus	Sacred Kingfisher	Lowland tropical rainforest (Papua New Guinea)	[61]
	Dacelo gaudichaud	Rufous-bellied Kookuburra	Lowland tropical rainforest (Papua New Guinea)	[61]
	Syma torotoro	Yellow-billed Kingfisher	Lowland tropical rainforest (Papua New Guinea)	[61]
Hirundinidae	*Hirundo tahitica*	Pacific Swallow	Lowland tropical rainforest (Papua New Guinea)	[61]
	Hirundo rustica	Barn Swallow	Hill dipterocarp tropical rainforest (Malaysia)	[6]
Icteridae	*Cacicus uropygialis*	Scarlet-rumped Cacique	Tropical forest (Costa Rica)	[65]
	Psarocolius wagleri	Chestnut-headed Oropendola	Tropical forest (Costa Rica)	[65]
Irenidae	*Irena puella*	Asian Fairy-bluebird	Tropical Rainforest (India), Hill dipterocarp tropical rainforest (Malaysia)	[6, 64]
Laniidae	*Lanius cristatus*	Brown Shrike	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Lanius tigrinus	Tiger Shrike	Hill dipterocarp tropical rainforest (Malaysia)	[6]
Leiotherichidae	*Garrulax jerdoni*	Kerala Laughingthrush	Tropical Rainforest (India)	[64]
	Garrulax delesserti	Wynaad Laughingthrush	Tropical Rainforest (India)	[64]
Maluridae	*Malurus amabilis*	Lovely Fairy-wren	Tropical Rainforest (Australia)	[62]

Family	Scientific name	Common name	Habitat	Authors
Megalaimidae	*Megalaima rubricapilla*	Crimson-fronted Barbet	Tropical Rainforest (India)	[64]
	Talegalla jobiensis	Brown-collared Brush Turkey	Lowland tropical rainforest (Papua New Guinea)	[61]
	Megapodius reinwardt	Orange-footed Scrubfowl	Tropical Rainforest (Australia)	[62]
Melanocharitidae	*Melanocharis nigra*	Black Berrypecker	Lowland tropical rainforest (Papua New Guinea)	[61]
	Oedistoma iliolophus	Plumed Longbill	Lowland tropical rainforest (Papua New Guinea)	[61]
	Toxorhamphus novaeguineae	Yellow-bellied Longbill	Lowland tropical rainforest (Papua New Guinea)	[61]
Meliphagidae	*Meliphaga analoga*	Mimic Honeyeater	Lowland tropical rainforest (Papua New Guinea)	[61]
	Meliphaga montana	White-marked Forest Honeyeater	Lowland tropical rainforest (Papua New Guinea)	[61]
	Philemon meyeri	Meyer's Friarbird	Lowland tropical rainforest (Papua New Guinea)	[61]
	Xanthotis flaviventer	Tawny-breasted Honeyeater	Lowland tropical rainforest (Papua New Guinea)	[61]
	Philemon buceroides	Helmeted Friarbird	Lowland tropical rainforest (Papua New Guinea), Tropical Rainforest (Australia)	[61, 62]
	Meliphaga notata	Yellow-spotted Honeyeater	Tropical Rainforest (Australia)	[62]
	Myzomela obscura	Dusky Honeyeater	Tropical Rainforest (Australia)	[62]
	Xanthotis macleayana	Macleay’s Honeyeater	Tropical Rainforest (Australia)	[62]
	Meliphaga gracilis	Graceful Honeyeater	Tropical Rainforest (Australia), Isolated Tropical Rainforest (Malaysia)	[62, 63]
Meropidae	*Merops viridis*	Blue-throated Bee-eater	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Merops ornatus	Rainbow Bee-eater	Tropical Rainforest (Australia), Lowland tropical rainforest (Papua New Guinea)	[61, 62]
Monarchidae	*Hypothymis azurea*	Black-naped Monarch	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia),	[6, 63]
	Terpsiphone paradisi	Asian Paradise Flycatcher	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 62]
	Arses telescopthalmus	Frilled Monarch	Lowland tropical rainforest (Papua New Guinea)	[61]
	Machaerirhynchus flaviventer	Yellow-breasted Boatbill	Lowland tropical rainforest (Papua New Guinea)	[61]
	Monarcha chrysomela	Golden Monarch	Lowland tropical rainforest (Papua New Guinea)	[61]

Family	Scientific name	Common name	Habitat	Authors
	Monarcha guttula	Spot-winged Monarch	Lowland tropical rainforest (Papua New Guinea)	[61]
	Monarcha manadensis	Hooded Monarch	Lowland tropical rainforest (Papua New Guinea)	[61]
	Monarcha rubiensis	Rufous Monarch	Lowland tropical rainforest (Papua New Guinea)	[61]
	Myiagra alecto	Shining Flycatcher	Lowland tropical rainforest (Papua New Guinea)	[61]
	Arses kaupi	Pied Monarch	Tropical Rainforest (Australia)	[62]
	Monarcha leucotis	White-eared Monarch	Tropical Rainforest (Australia)	[62]
	Monarcha trivirgatus	Spectacled Monarch	Tropical Rainforest (Australia)	[62]
Muscicapidae	*Ficedula nigrorufa*	Black-and-Orange Flycatcher	Tropical Rainforest (India)	[64]
	Muscicapa sibirica	Dark-sided Flycatcher	Tropical Rainforest (Malaysia)	[63]
	Philentoma pyrhopterum	Chestnut-winged Flycatcher	Tropical Rainforest (Malaysia)	[63]
	Enicurus ruficapillus	Chestnut-naped Forktail	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Eumyias thalassinus	Verditer Flycatcher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Ficedula mugimaki	Mugimaki Flycatcher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Ficedula zanthopygia	Yellow-rumped Flycatcher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Muscicapa dauurica	Asian Brown Flycatcher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Muscicapella hodgsoni	Pygmy Blue Flycatcher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Philentoma pyrhoptera	Rufous-winged Philentoma	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Rhinomyias umbratilis	Grey-chested Jungle Flycatcher	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Copsychus malabaricus	White-rumped Shama	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Microeca flavovirescens	Olive-yellow Flycatcher	Lowland tropical rainforest (Papua New Guinea)	[61]
	Brachypteryx major	White-bellied Robbin	Tropical Rainforest (India)	[64]
	Cyornis pallipes	White-bellied Blue Flycatcher	Tropical Rainforest (India)	[64]
	Eumyias albicaudatus	Nilgiri Flycatcher	Tropical Rainforest (India)	[64]
	Myophonus horsfieldii	Malabar Whistling Thrush	Tropical Rainforest (India)	[64]

Family	Scientific name	Common name	Habitat	Authors
	Culicicapa ceylonensis	Grey-headed Canary Flycatcher	Tropical Rainforest (India), Isolated Tropical Rainforest (Malaysia)	[63, 64]
	Ficedula elisae	Green-backed Flycatcher	Tropical Rainforest (Malaysia)	[63]
	Muscicapa dauurica	Asian Brown Flycatcher	Tropical Rainforest (Malaysia)	[63]
Nectarinidae	*Nectarinia minima*	Crimson-backed Sunbird	Tropical Rainforest (India)	[64]
	Arachnothera magna	Streaked Spiderhunter	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Arachnothera modesta	Grey-breasted Spiderhunter	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Hypogramma hypogrammicum	Purple-naped Sunbird	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Anthreptes simplex	Plain Sunbird	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Arachnothera flavigaster	Spectacled Spiderhunter	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Arachnothera longirostra	Little Spiderhunter	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Cinnyris jugularis	Olive-backed Sunbird	Lowland tropical rainforest (Papua New Guinea)	[61]
	Nectarinia jugularis	Yellow-bellied Sunbird	Tropical Rainforest (Australia)	[62]
	Nectarinia lotenia	Loten's Sunbird	Tropical Rainforest (India)	[64]
	Hypogramma hypogrammicum	Purple-naped Sunbird	Tropical Rainforest (Malaysia)	[63]
Oriolidae	*Oriolus xanthonotus*	Dark-throated Oriole	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Oriolus szalayi	Brown Oriole	Lowland tropical rainforest (Papua New Guinea)	[61]
	Pitohui kirhocephalus	Variable Pitohui	Lowland tropical rainforest (Papua New Guinea)	[61]
	Oriolus flavocinctus	Yellow Oriole	Tropical Rainforest (Australia)	[62]
	Oriolus chinensis	Black-naped Oriole	Tropical Rainforest (Malaysia)	[63]
Pachycephalidae	*Pachycephala hyperythra*	Rusty-breasted Whistler	Lowland tropical rainforest (Papua New Guinea)	[61]
	Pachycephala simplex	Grey Whistler	Lowland tropical rainforest (Papua New Guinea)	[61]
	Pitohui ferrugineus	Rusty Pitohui	Lowland tropical rainforest (Papua New Guinea)	[61]

Family	Scientific name	Common name	Habitat	Authors
Paradisaeidae	*Cicinnurus regius*	King Bird of Paradise	Lowland tropical rainforest (Papua New Guinea)	[61]
	Paradisaea minor	Lesser BOP	Lowland tropical rainforest (Papua New Guinea)	[61]
Pardalotidae	*Crateroscelis murina*	Rusty Mouse Warbler	Lowland tropical rainforest (Papua New Guinea)	[61]
Paridae	*Parus xanthogenys*	Himalayan Black-lored Tit	Tropical Rainforest (India)	[64]
Parulidae	*Dendroica virens*	Black-throated Green Warbler	Tropical forest (Costa Rica)	[65]
	Oporornis formosus	Kentucky Warbler	Tropical forest (Costa Rica)	[65]
	Setophaga ruticilla	American Redstart	Tropical forest (Costa Rica)	[65]
	Vermivora chrysoptera	Golden-winged Warbler	Tropical forest (Costa Rica)	[65]
	Wilsonia pusilla	Wilson's Warbler	Tropical forest (Costa Rica)	[65]
Pellorneidae	*Alcippe poioicephala*	Brown-cheeked Fulvetta	Tropical Rainforest (India)	[64]
	Alcippe brunnea	Brown Fulvetta	Tropical Rainforest (Malaysia)	[63]
Petroicidae	*Poecilodryas hypoleuca*	Black-sided Robin	Lowland tropical rainforest (Papua New Guinea)	[61]
	Microeca flavigaster	Lemon-bellied Flycatcher	Tropical Rainforest (Australia)	[62]
	Tregellasia capito	Pale-yellow Robin	Tropical Rainforest (Australia)	[62]
Phasianidae	*Galloperdix spadicea*	Red Spurfowl	Tropical Rainforest (India)	[64]
	Gallus sonneratii	Grey Junglefowl	Tropical Rainforest (India)	[64]
	Gallus gallus	Red Junglefowl	Tropical Rainforest (Malaysia)	[63]
Phylloscopidae	*Phylloscopus borealis*	Arctic Warbler	Tropical Rainforest (Malaysia)	[63]
Picidae	*Sasia abnormis*	Rufous Piculet	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Blythipicus rubiginosus	Maroon Woodpecker	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Meiglyptes tukki	Buff-necked Woodpecker	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Picus mentalis	Checker-throated Woodpecker	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Picus miniaceus	Banded Woodpecker		[6, 63]

Family	Scientific name	Common name	Habitat	Authors
			Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	
	Piculus rubiginosus	Golden-olive Woodpecker	Tropical forest (Costa Rica)	[65]
	Celeus brachyurus	Rufous Woodpecker	Tropical Rainforest (India)	[64]
	Dryocopus javensis	White-bellied Woodpecker	Tropical Rainforest (India)	[64]
	Picumnus innominatus	Speckled Piculet	Tropical Rainforest (India)	[64]
	Chrysocolaptes lucidus	Greater Flameback	Tropical Rainforest (India), Isolated Tropical Rainforest (Malaysia)	[63, 64]
	Dinopium javanense	Common Flameback	Tropical Rainforest (India), Isolated Tropical Rainforest (Malaysia)	[63, 64]
	Dinopium rafflesii	Olive-backed Woodpecker	Tropical Rainforest (Malaysia)	[63]
	Meiglyptes jugularis	Black and buff Woodpecker	Tropical Rainforest (Malaysia)	[63]
	Picus flavinucha	Greater Yellownape	Tropical Rainforest (Malaysia)	[63]
	Picus puniceus	Crimson-winged Woodpecker	Tropical Rainforest (Malaysia)	[63]
Pipridae	*Corapipo leucorrhoa*	White-bibbed Manakin	Tropical forest (Costa Rica)	[65]
	Manacus aurantiacus	Orange-collard Manakin	Tropical forest (Costa Rica)	[65]
Pittidae	*Pitta guajana*	Banded Pitta	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Pitta erythrogaster	Red-bellied Pitta	Lowland tropical rainforest (Papua New Guinea)	[61]
	Pitta sordida	Hooded Pitta	Tropical Rainforest (Malaysia)	[63]
Podargidae	*Batrachostomus stellatus*	Gould's Frogmouth	Hill dipterocarp tropical rainforest (Malaysia)	[6]
Psittaculidae	*Lorius lory*	Black Capped Lori	Lowland tropical rainforest (Papua New Guinea)	[61]
	Loriculus vernalis	Vernal Hanging Parrot	Tropical Rainforest (India)	[64]
	Eclectus roratus	Eclectus Parrot	Lowland tropical rainforest (Papua New Guinea)	[61]
	Geoffroyus geoffroyi	Red-cheeked Parrot	Lowland tropical rainforest (Papua New Guinea)	[61]
	Geoffroyus simplex	Blue-collared Parrot	Lowland tropical rainforest (Papua New Guinea)	[61]
	Micropsitta pusio	Buff-faced Pygmy Parrot	Lowland tropical rainforest (Papua New Guinea)	[61]

Family	Scientific name	Common name	Habitat	Authors
	Pseudeos fuscata	Dusky Lory	Lowland tropical rainforest (Papua New Guinea)	[61]
	Psittaculirostris edwardsii	Edward's Fig Parrot	Lowland tropical rainforest (Papua New Guinea)	[61]
	Cyclopsitta diophthalma	Double-eyed Fig-parrot	Tropical Rainforest (Australia)	[62]
	Trichoglossus haematodus	Rainbow Lorikeet	Tropical Rainforest (Australia), Lowland tropical rainforest (Papua New Guinea)	[61, 62]
	Psittacula columboides	Blue-winged Parakeet	Tropical Rainforest (India)	[64]
	Psittacula cyanocephala	Plum-headed Parakeet	Tropical Rainforest (India)	[64]
Psophodidae	*Psophodes olivaceus*	Eastern Whipbird	Tropical Rainforest (Australia)	[62]
Ptilonorhynchidae	*Ailuroedus buccoides*	White-eared Catbird	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ailuroedus melanotis	Spotted Catbird	Tropical Rainforest (Australia)	[62]
Pycnonotidae	*Alophoixus ochraceus*	Ochraceous Bulbul	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Alophoixus phaeocephalus	Yellow-bellied Bulbul	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Ixos malaccensis	Streaked Bulbul	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Pycnonotus cyaniventris	Grey-bellied Bulbul	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Pycnonotus finalysoni	Stripe-throated Bulbul	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Pycnonotus melanoleucos	Black-and-white Bulbul	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Iole olivacea	Buff-vented Bulbul	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia),	[6, 63]
	Pycnonotus simplex	Cream-vented Bulbul	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Pycnonotus bruuneus	Red-eyed Bulbul	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Pycnonotus erythropthalmos	Spectacled Bulbul	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Pycnonotus eutilotus	Puff-backed Bulbul	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Pycnonotus plumosus	Olive-winged Bulbul		[6, 63]

Family	Scientific name	Common name	Habitat	Authors
			Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	
	Tricholestes criniger	Hairy-backed Bulbul	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Hypsipetes leucocephalus	Black Bulbul	Tropical Rainforest (India)	[64]
	Iole indica	Yellow-browed Bulbul	Tropical Rainforest (India)	[64]
	Pycnonotus jocosus	Red-whiskered Bulbul	Tropical Rainforest (India)	[64]
	Pycnonotus melanicterus	Black-capped Bulbul	Tropical Rainforest (India)	[64]
	Pycnonotus priocephalus	Grey-headed Bulbul	Tropical Rainforest (India)	[64]
	Alophoixus finschii	Finsch’s Bulbul	Tropical Rainforest (Malaysia)	[63]
	Pycnonotus atriceps	Black-headed Bulbul	Tropical Rainforest (Malaysia)	[63]
	Pycnonotus goiavier	Yellow-vented Bulbul	Tropical Rainforest (Malaysia)	[63]
Ramphastidae	*Calormphus fuliginosus*	Brown Barbet	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Pteroglossus frantzii	Fiery-billed Aracari	Tropical forest (Costa Rica)	[65]
Rhipiduridae	*Rhipidura rufiventris*	Northern Fantail	Lowland tropical rainforest (Papua New Guinea)	[61]
	Rhipidura threnothorax	Sooty Thicket Fantail	Lowland tropical rainforest (Papua New Guinea)	[61]
	Rhipidura fuliginosa	Grey Fantail	Tropical Rainforest (Australia)	[62]
	Rhipidura rufifrons	Rufous Fantail	Tropical Rainforest (Australia)	[62]
	Rhipidura javanica	Pied Fantail	Tropical Rainforest (Malaysia)	[63]
	Rhipidura perlata	Spotted Fantail	Hill dipterocarp tropical rainforest (Malaysia)	[65]
	Rhipidura leucothorax	White-bellied Thicket-Fantail	Lowland tropical rainforest (Papua New Guinea)	[61]
Sittidae	*Sitta frontalis*	Velvet-fronted Nuthatch	Tropical Rainforest (India)	[64]
Strigidae	*Otus rufescens*	Reddish Scope Owl	Hill dipterocarp tropical rainforest (Malaysia)	[6]
Sturnidae	*Aplonis cantoroides*	Singing Starling	Lowland tropical rainforest (Papua New Guinea)	[61]
	Mino dumontii	Yellow Faced Myna	Lowland tropical rainforest (Papua New Guinea)	[61]

Family	Scientific name	Common name	Habitat	Authors
	Aplonis metallica	Metallic Starling	Tropical Rainforest (Australia), Lowland tropical rainforest (Papua New Guinea)	[61, 62]
	Gracula religiosa	Hill Myna	Tropical Rainforest (India)	[64]
Tephrodornithidae	*Tephrodornis gularis*	Large Woodshrike	Tropical Rainforest (India)	[64]
Thamnophilidae	*Cercomacra tyrannina*	Dusky Antbird	Tropical forest, Costa Rica	[65]
Thraupidae	*Piranga flava*	Hepatic Tanager	Tropical forest (Costa Rica)	[65]
	Tangara guttata	Speckled Tanager	Tropical forest (Costa Rica)	[65]
	Tangara gyrola	Bay-headed Tanager	Tropical forest (Costa Rica)	[65]
Timaliidae	*Malacocincla sepiaria*	Horsfield's Babbler	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Malacopteron affine	Sooty-capped Babbler	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Pellorneum capistratum	Black-capped Babbler	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Stachyris erythroptera	Chestnut-winged Babbler	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Stachyris maculata	Chestnut-rumped Babbler	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Stachyris poliocephala	Grey-headed Babbler	Hill dipterocarp tropical rainforest (Malaysia)	[65]
	Trichastoma bicolor	Ferruginous Babbler	Hill dipterocarp tropical rainforest (Malaysia)	[65]
	Trichastoma rostratum	White-chested Babbler	Hill dipterocarp tropical rainforest (Malaysia)	[65]
	Macronous gularis	Striped Tit-babbler	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Macronus ptilosus	Fluffy-backed Tit-Babbler	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Malacocincla malaccensis	Short-tailed Babbler	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Malacopteron cinereum	Scaly-crowned Babbler	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Malacopteron magnirostre	Moustached Babbler	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Malacopteron magnum	Rufous-crowned Babbler		[6, 63]

Family	Scientific name	Common name	Habitat	Authors
			Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	
	Stachyris nigricollis	Black-throated Babbler	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Garritornis isidorei	Isidore's Rufous Babbler	Lowland tropical rainforest (Papua New Guinea)	[61]
	Ptilorrhoa caerulescens	Blue Jewel-babbler	Lowland tropical rainforest (Papua New Guinea)	[61]
	Megalaima viridis	White-cheeked Barbet	Tropical Rainforest (India)	[64]
	Pellorneum ruficeps	Puff-throated Babbler	Tropical Rainforest (India)	[64]
	Pomatorhinus horsfieldii	Indian Scimitar Babbler	Tropical Rainforest (India)	[64]
	Rhopocichla atriceps	Dark-fronted Babbler	Tropical Rainforest (India)	[64]
	Turdoides subrufus	Rufous Babbler	Tropical Rainforest (India)	[64]
	Pellorneum capistratum	Black-caped Babbler	Tropical Rainforest (Malaysia)	[63]
	Napothera epilepidota	Eyebrowed Wren-babbler	Tropical Rainforest (Malaysia)	[63]
	Stachyris erythroptera	Chestnut-winged Babbler	Tropical Rainforest (Malaysia)	[63]
	Yuhina zantholeuca	White-bellied Yuhina	Tropical Rainforest (Malaysia)	[63]
Tityridae	*Pachyramphus aglaiae*	Rose-throated Becard	Tropical forest (Costa Rica)	[65]
Trochilidae	*Campylopterus hemileucurus*	Violet Sabrewing	Tropical forest (Costa Rica)	[65]
	Phaethornis guy	Green Hermit	Tropical forest (Costa Rica)	[65]
	Phaethornis longuemareus	Little Hermit	Tropical forest (Costa Rica)	[65]
Troglodytidae	*Thryothorus rufalbus*	Rufous-and-white Wren	Tropical forest (Costa Rica)	[65]
Trogonidae	*Harpactes diardii*	Diard’s Trogon	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Harpactes orrhophaeus	Cinnamon-rumped Trogon	Hill dipterocarp tropical rainforest (Malaysia)	[6]
	Trogon bairdii	Baird's Trogon	Tropical forest (Costa Rica)	[65]
	Harpactes fasciatus	Malabar Trogon	Tropical Rainforest (India)	[64]
Turdidae	*Copsychus saularis*	Oriental Magpie Robin		[6, 63]

Family	Scientific name	Common name	Habitat	Authors
			Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	
	Luscinia cyane	Siberian Blue Robin	Hill dipterocarp tropical rainforest (Malaysia), Isolated Tropical Rainforest (Malaysia)	[6, 63]
	Turdus merula	Common Blackbird	Tropical Rainforest (India)	[64]
	Zoothera citrina	Orange-headed Thrush	Tropical Rainforest (India)	[64]
Vireonidae	*Hylophilus decurtatus*	Lesser Greenlet	Tropical forest (Costa Rica)	[65]
	Vireo flavifrons	Yellow-throated Vireo	Tropical forest (Costa Rica)	[65]
	Vireolanius pulchellus	Green Shrike-vireo	Tropical forest (Costa Rica)	[65]
Zosteropidae	*Zosterops lateralis*	Silvereye	Tropical Rainforest (Australia)	[62]
	Zosterops palpebrosus	Oriental White-eye	Tropical Rainforest (India), Isolated Tropical Rainforest (Malaysia)	[63, 64]

Table 1. List of bird species that occur in tropical rainforest.

cover. Forest logging [66–68], habitat degradation and fragmentation [69], slash-and-burn agriculture [61], and fires are major factors, which had adversely affected the population of the avian species in different forest ecosystems [51]. These factors altered the vegetation structure and composition, which affects the avian richness and diversity by affecting the food resources, increased nest predation and brood parasitism. The diversity and richness of food resources are closely associated with the vegetation structure and composition, such as foliage, flowers, fruits, and barks. Furthermore, large-scale logging for valuable timber harvesting,

Figure 5. Bornean pygmy elephant—*Elephas maximus borneensis*.

Figure 6. Sambar deer—*Rusa unicolor*.

Family	Scientific name	Common name	Habitat	Authors
Emballonuridae	*Saccopteryx bilineata*	Greater Sac-winged Bat	Tropical Rainforest (Mexico)	[73]
	Diclidurus virgo	White Bat	Tropical Rainforest (Mexico)	[73]
Mormoopidae	*Pteronotus davyi*	Davy's Naked-backed Bat	Tropical Rainforest (Mexico)	[73]
	Mormoops megalophylla	Ghost-faced Bat	Tropical Rainforest (Mexico)	[73]
	Pteronotus parnellii	Parnell's Mustached Bat	Tropical Rainforest (Mexico)	[73]
Muridae	*Rattus annandalei*	Annandale's Rat	Primary Rainforest (Malaysia)	[74]
	Niviventer fulvescens	Chestnut White-bellied Rat	Primary Rainforest (Malaysia)	[74]
	Niviventer cremoriventer	Dark-tailed Tree Rat	Primary Rainforest (Malaysia)	[74]
	Leopoldamys edwardsi	Edwards's Long-tailed Giant Rat	Tropical Rainforest (Indonesia)	[75]
	Lenothrix canus	Gray Tree Rat	Primary Rainforest (Malaysia)	[74]
	Leopoldamys sabanus	Long-tailed Giant Rat	Primary Rainforest (Malaysia), Tropical Rainforest (Indonesia)	[74, 75]
	Niviventer rapit	Long-tailed Mountain Rat	Tropical Rainforest (Indonesia)	[75]
	Rattus tiomanicus	Malayan Field Rat	Primary Rainforest (Malaysia)	[74]
	Sundamys muelleri	Muller's Giant Sunda Rat	Primary Rainforest (Malaysia), Tropical Rainforest (Indonesia)	[74, 75]
	Maxomys rajah	Rajah Spiny Rat	Primary Rainforest (Malaysia), Tropical Rainforest (Indonesia)	[74, 75]
	Maxomys surifer	Red Spiny Rat	Primary Rainforest (Malaysia)	[74]
	Maxomys whiteheadi	Whitehead's Spiny Rat	Primary Rainforest (Malaysia), Tropical Rainforest (Indonesia)	[74, 75]
Natalidae	*Natalus stramineus*	Mexican Funnel-eared Bat	Tropical Rainforest (Mexico)	[73]
Phyllostomidae	*Desmodus rotundus*	Common Vampire Bat	Tropical Rainforest (Mexico)	[73]
	Choeroniscus godmani	Godman's Long-tailed Bat	Tropical Rainforest (Mexico)	[73]
	Mimon bennettii	Golden Bat	Tropical Rainforest (Mexico)	[73]

Family	Scientific name	Common name	Habitat	Authors
	Artibeus lituratus	Great Fruit-eating Bats	Tropical Rainforest (Mexico)	[73]
	Chiroderma villosum	Hairy Big-eyed Bat	Tropical Rainforest (Mexico)	[73]
	Platyrrhinus helleri	Heller's Broad-nosed Bat	Tropical Rainforest (Mexico)	[73]
	Sturnira ludovici	Highland Yellow-shouldered Bat	Tropical Rainforest (Mexico)	[73]
	Artibeus jamaicensis	Jamaican Fruit-eating Bat	Tropical Rainforest (Mexico)	[73]
	Vampyressa pusilla	Little Yellow-eared Bat	Tropical Rainforest (Mexico)	[73]
	Leptonycteris sanborni	Long-nosed Bat	Tropical Rainforest (Mexico)	[73]
	Lampronycteris brachyotis	Orange-throated Big-eared Bat	Tropical Rainforest (Mexico)	[73]
	Phyllostomus discolor	Pale Spear-nosed Bat	Tropical Rainforest (Mexico)	[73]
	Glossophaga soricina	Pallas's Long-tongued Bat	Tropical Rainforest (Mexico)	[73]
	Artibeus phaeotis	Pygmy Fruit-eating Bat	Tropical Rainforest (Mexico)	[73]
	Carollia brevicauda	Silky Short-tailed Bat	Tropical Rainforest (Mexico)	[73]
	Vampyrodes major	Stripe-faced Bat	Tropical Rainforest (Mexico)	[73]
	Uroderma bilobatum	Tent-making Bat	Tropical Rainforest (Mexico)	[73]
	Artibeus watsoni	Thomas's Fruit-eating Bat	Tropical Rainforest (Mexico)	[73]
	Artibeus toltecus	Toltec Fruit-eating Bat	Tropical Rainforest (Mexico)	[73]
	Hylonycteris underwoodi	Underwood's Long-tongued Bat	Tropical Rainforest (Mexico)	[73]
	Enchisthenes hartii	Velvety Fruit-eating Bat	Tropical Rainforest (Mexico)	[73]
	Centurio senex	Wrinkle-faced Bat	Tropical Rainforest (Mexico)	[73]
	Sturnira lilium	Yellow-shouldered Bat	Tropical Rainforest (Mexico)	[73]
Sciuridae	*Sundasciurus lowii*	Low's Squirrel	Primary Rainforest (Malaysia)	[74]
	Callosciurus notatus	Plantain Squirrel	Primary Rainforest (Malaysia)	[74]
	Callosciurus prevostii	Prevost's Squirrel	Primary Rainforest (Malaysia)	[74]

Family	Scientific name	Common name	Habitat	Authors
	Sundasciurus tenuis	Slender Squirrel	Tropical Rainforest (Indonesia)	[75]
	Lariscus insignis	Three-striped Ground Squirrel	Tropical Rainforest (Indonesia)	[75]
Thyropteridae	*Thyroptera tricolor*	Spix's Disk-winged Bat	Tropical Rainforest (Mexico)	[73]
Tupaiidae	*Tupaia glis*	Common Treeshrew	Primary Rainforest (Malaysia), Tropical Rainforest (Indonesia)	[74, 75]
	Tupaia tana	Large Treeshrew	Tropical Rainforest (Indonesia)	[75]
Vespertilionidae	*Antrozous sp.*	Pallid Bat	Tropical Rainforest (Mexico)	[73]

Table 2. List of mammal species that occur in tropical rainforest.

Family	Scientific name	Common name	Habitat	Authors
Agamidae	*Gonyocephalus semperi*	White-Spotted Angle head	Tropical Rainforest (Philippine)	[94]
Colubridae	*Boiga dendrophilia*	Mangrove Blunt-Headed Snake	Tropical Rainforest (Philippine)	[94]
	Psammodynastes pulverulentus	Dark- Spotted Mock Viper	Tropical Rainforest (Philippine)	[94]
	Oxyrhabdium leporinum	Banded Philippine Burrowing Snake	Tropical Rainforest (Philippine)	[94]
	Oligodon maculatus	Barred ShortHeaded Snake	Tropical Rainforest (Philippine)	[94]
	Calamaria gervaisii	Gervais' Worm Snake	Tropical Rainforest (Philippine)	[94]
	Lycodon dumerili	Dumeril's Wolf Snake	Tropical Rainforest (Philippine)	[94]
	Tropidolaemus sp.	Wagler's Pit Viper	Tropical Rainforest (Philippine)	[94]
	Phyton reticulatus	Reticulated Phyton	Tropical Rainforest (Philippine)	[94]
Gekkonidae	*Gekko mindorensis*	Mindoro Narrow-Disked Gecko	Tropical Rainforest (Philippine)	[94]
Scincidae	*Sphenomorphus variegatus*	Black-Spotted Sphenomorphus	Tropical Rainforest (Philippine)	[94]
	Sphenomorphus beyeri	Beyer's Sphenomorphus	Tropical Rainforest (Philippine)	[94]
	Lipinia pulchella	Yellow-Striped Slender Tree Skink	Tropical Rainforest (Philippine)	[94]
	Eutropis multicarinata borealis	Northern Two-Striped Mabuya	Tropical Rainforest (Philippine)	[94]
	Eutropis englei	Six-Striped Mabouya	Tropical Rainforest (Philippine)	[94]

Table 3. List of reptile species that occur in tropical rainforest.

damage to forest, and replacement of native vegetation by exotic species [70] are the main problems, which affect the fauna species. Loss of forested areas is responsible for the loss of biodiversity.

2.2. Mammal species composition of tropical rainforest

Tropical rainforest had harbored rich mammal diversity and density due to richness of plant communities and higher productivity (**Figures 5** and **6**; **Table 2**). Mammals are a versatile group of animals and a major component of the tropical rainforest ecosystem, i.e., they serve a wide range of ecosystem functions; such as pollination, seed dispersal, pest control, herb control, food source for other animals and nutrient cycling. In addition to ecological functions, the mammals also provide a wide array of benefits to human beings, such as food, recreation, and source of income, i.e., various byproducts such as bush meat, skin, oil, musk, fur, etc. [71, 72].

Unfortunately, these rich mammal communities are facing severe threats from human activities such as over exploitation (intensive hunting), land-use change (habitat loss and degradation), and climate change [76–78]. These populations of different mammal species had declined abruptly due to change in land use, i.e., habitat fragmentation and degradation due to logging, and deforestation and habitat loss due to agriculture expansion and excessive hunting [79–83]. It has been reported that around one-fifth of mammal species in the wild are at risk of extinction due to human activities such as deforestation for agriculture expansion, logging for timber, and excessive hunting [84]. It has been stated that changes in vegetation cover may affect the richness of food resources and habitat preferences of the mammalian species [85, 86]. This could be due to fact that home range preferences of the mammal species and their population are strongly associated with the vegetation structure and composition [87].

The primates residing in a rainforest are habitat specific, some occupy large continuous forested areas such as Diademed Sifakas—*Propithecus diadema*—while others prefer fragmented forested areas such as Black Howler Monkey—*Alouatta pigra*—for their survival and reproduction [25, 88–90]. Monkeys are diet specific, they consume a variety of food resources such as fruits, seeds, flowers, leaves, arthropods, etc. [91, 92], and their diet is strongly influenced by the plant species composition and richness of the particular dwelling habitat [93].

2.3. Reptile species composition of tropical rainforest

Reptiles are carnivorous in nature and play a significant role in controlling various pests present in the forest, such as beetles, arthropods, caterpillars, termites, bugs, rats, mice, etc.,

Figure 7. Red-eye tree frog—*Agalychnis callidryas* (Male).

Family	**Scientific Name**	**Common Name**	**Habitat**	**Authors**
Bufonidae	*Ingerophrynus divergens*	Malayan Dwarf Toad	Lowland Tropical Rainforest (Malaysia)	[106]
	Ansonia muelleri	Mueller's Toad	Tropical Rainforest (Philippine)	[94]
Dicroglossidae	*Limnonectes finchi*	Finch's Wart Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Limnonectes ingeri	Inger's Wart frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Limnonectes leporinus	Giant River Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Limnonectes malesianus	Malaysian Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Occidozyga baluensis	Balu Oriental Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Occidozyga laevis	Puddle Frog	Lowland Tropical Rainforest (Malaysia)	[106]
Microhylidae	*Chaperina fusca*	Brown Thorny Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Kalophrynus pleurostigma	Black-spotted Sticky Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Kaloula baleata	Smooth-fingered Narrow-mouthed Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Metaphrynella sundana	Borneo Tree-hole Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Microhyla borneensis	Matang Narrow-mouthed Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Kalophrynus pleurostigma	Black-spotted Narrow-mouthed Frog	Tropical Rainforest (Philippine)	[94]
Ranidae	*Hylarana erythraea*	Common Green Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Hylarana glandulosa	Rough-sided Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Hylarana nicobariensis	Cricket Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Hylarana raniceps	White-lipped Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Staurois natator	Rock Frog	Tropical Rainforest (Philippine)	[94]
	Rana grandocula	Big-eyed Frog	Tropical Rainforest (Philippine)	[94]
	Limnonectes magnus	Mindanao Fanged Frog	Tropical Rainforest (Philippine)	[94]
	Platymantis corrugata	Rough-backed Forest Frog	Tropical Rainforest (Philippine)	[94]

Family	**Scientific Name**	**Common Name**	**Habitat**	**Authors**
	Megophrys stejnegeri	Mindanao Horned Frog	Tropical Rainforest (Philippine)	[94]
Rhacophoridae	*Nyctixalus pictus*	Cinnamon Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Polypedates colletti	Black-spotted Tree Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Polypedates leucomystax	Common Tree Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Polypedates macrotis	Dark-eared Tree Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Polypedates otilophus	Borneo Eared Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Rhacophorus appendiculatus	Frilled Tree Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Rhacophorus dulitensis	Jade Tree Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Rhacophorus harrissoni	Brown Tree Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Rhacophorus pardalis	Harlequin Tree Frog	Lowland Tropical Rainforest (Malaysia)	[106]
	Polypedates leucomystax	Four-lined Tree Frog	Tropical Rainforest (Philippine)	[94]
	Philautus acutirostris	Pointed-Snouted Tree Frog	Tropical Rainforest (Philippine)	[94]

Table 4. List of amphibian species that occur in tropical rainforest.

which may cause severe loss, such as defoliation, seed, and wood damage. Even though they are crucially important for tropical forest ecosystems (**Table 3**), they are facing critical threats from human induced factors such as land-use change (i.e., deforestation, fragmentation and degradation) that have altered the natural habitat of the reptile species that directly or indirectly depend upon tropical rainforests for their survival and reproduction.

Reptiles are highly sensitive species compared to other fauna species, i.e., they become more vulnerable due to land use change, i.e., habitat alteration [95]. This might be because they have a small home range, which is adversely affected by habitat loss and degradation [96, 97]. For example, deforestation may cause severe habitat loss, fragmentation, and degradation which adversely affect the population, community parameters of reptiles inhabiting in a tropical rainforest ecosystem [98–100].

Anthropogenic activities had altered the reptilian intact habitat through land-use change, their habitats becomes degraded and lost thus ultimately becoming unsuitable for them. This is because forested reptile prefers dense and moist habitat, which provides them shelter and rich food resources for their survival, protection, and reproduction. Deforestation may disturb their breeding sites, reduce home range, and increase visibility for predators. Likewise, fragmentation reduced their home range, while degradation reduced their food resources and breeding behavior. Furthermore, land-use changes such as deforestation, fragmentation, and degradation may alter microclimatic conditions of particular dwelling habitats such as temperature, relative humidity, rainfall, and sunlight that ultimately modify the vegetation structure and composition.

2.4. Amphibian species composition of tropical rainforest

Amphibians are cold-blooded vertebrates and are carnivorous in nature. They play a key role to control the various pests, thus balancing the nature [101]. Amphibians are a significant component of the tropical rainforest ecosystem (**Figure 7**; **Table 4**) and play an important role in pest control. Habitat loss due highest deforestation is responsible for one-third population decline of the amphibians [10, 102–104]. One-fifth amphibians of Southeast Asia are reported as threatened species [105]. This is because they have small home ranges, i.e., specific aquatic habitat, higher vulnerability to habitat change, and visibility to predators. In addition, over-harvesting from natural habitat for food supply (human consumption), medicine (traditional use), and pet trade also had exerted great pressure on the population of amphibians [104].

2.5. Invertebrate species composition of tropical rainforest

In tropical rainforests, logging creates gaps and alters the habitat structure and microclimatic conditions, e.g., temperature, relative humidity, and light [107, 108], which influence on the invertebrate diversity and distribution. After logging, new habitat with a different microclimate may develop which tend to be unsuitable for a wide array of invertebrates [109, 110]. This indicates that land-use changes influence invertebrate diversity, richness, and distribution. It has been stated that disturbing the habitat affects invertebrate colonization and distribution [111, 112]. Basset [113] reported that the canopy of tropical rain forest is rich in Coleoptera, Hymenoptera, Lepidoptera, and Araneae taxa. However, their home range and foraging habitats may vary from species to species depending upon the types of vegetation, forest types, and bio-geographical regions.

3. Conclusion and future perspective

Even though, faunas are a crucial component of tropical rainforest ecosystems, detailed information on different aspects of fauna community parameters such as species composition, distribution, diversity, richness and population trend, impact of anthropogenic activities, associated with microclimate and habitat variables is still lacking. The current review highlighted that tropical rainforest is an ideal productive habitat for a wide array of fauna species, i.e., birds, mammals, reptiles, amphibians, and invertebrates. These fauna are a major component of the food web of the rainforest ecosystem and functions. Furthermore, it was revealed that the diversity of rainforest fauna is facing many threats that directly or indirectly affected the population; community parameters of various fauna species inhabited the tropical rainforest. There is an urgent need to study various fauna species of tropical rainforest in order to reduce the impact of human activities and for future conservation and management. We hope that the findings of this chapter will provide the ways and means to conserve the fauna in and around the tropical rainforest.

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Chapter 3

Detection of Amazon Forest Degradation Caused by Land Use Changes

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Additional information is available at the end of the chapter

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Abstract

Field and satellite optical methods for estimation of chlorophyll content were applied in three study sites of the Ecuadorian Amazon rainforest. Those sites represent a wide range of land use disturbance in secondary and pristine lowland rainforest. The first field method is based on transmittance from the SPAD-502 chlorophyll meter index, the second field method is based on reflectance measurements collected by a spectroradiometer, and the third method estimates chlorophyll content from the PROSPECT radiative transfer model. For the first method, seven models that account for a wide range of vegetation species showed similar average leaf chlorophyll contents until 80 units of SPAD-502. An average of the results of these models was computed and used as ground truth from where a generalized second-order polynomial model was created. For the second method, five chlorophyll indices based on reflectance measurements provided similar chlorophyll content estimations for all SPAD range (15–95 units). The third method estimates chlorophyll content based on the inversion process of the PROSPECT model. The satellite methods estimate vegetation indices sensitive to chlorophyll content from space. All methods have shown to be an alternative approach to detect forest degradation at local and regional levels caused by forest disturbances and land use changes.

Keywords: tropical forest, chlorophyll content, remote sensing, land use, forest degradation, photosynthesis

1. Introduction

The Amazon rainforest holds half of the tropical forested area of the world [1] and accounts for 30% of global biomass productivity [2] and 25% of global biodiversity [3]. Evaporation and

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condensation in tropical forests play a pivotal role in the regional and global atmospheric circulation [4], and the rivers' system produces about 20% of the world's fresh water discharge [5]. Photosynthesis and respiration process are more than twice the carbon of the annual rate of anthropogenic fossil fuel emissions [6]. Tropical forests store large amounts of carbon in high diversity ecosystems and play an important role in the global carbon cycle due to its net primary productivity (NPP). According to the estimates of Ref. [7], Amazon forests contain 70–80 billion metric tons (Pg or 10^{15} g) of carbon in plant biomass and assimilate 4–6 Pg of carbon each year in NPP. Despite its importance, a better understanding is needed of the interactions between the tropical forest and the global processes, such as climate change. During the last decades, the Amazon forest has been threatened by deforestation, selective logging, hunting, fire, and global and regional climate changes [4, 5].

Tropical forest deforestation and degradation have raised international concerns since they contribute approximately 20% to the global greenhouse gases (GHGs) emissions [8]. Reducing emissions from deforestation and forest degradation (REDD) is a United Nations Framework Convention on Climate Change (UNFCCC) initiative that developed a financial framework and mechanisms to reduce forest losses and the associated GHGs emissions aiming to prevent further deforestation and consequently mitigate climate change.

Deforestation is defined as the "permanent" conversion of a forest type to another land cover. "Forest degradation" is a reduction in biomass density within a forest cover. The relative contribution of deforestation and degradation to the net emissions of carbon is not readily distinguished [9]. Research has aimed to quantify global deforestation from satellite and census data, but there is an ongoing debate on the uncertainties of the estimates [10]. On the other hand, forest degradation has been more difficult to measure with remote sensing and there are no estimates for the entire tropics [9]. Therefore, accurate estimations of photosynthetic activity of forested areas are needed to quantify forest degradation and evaluate environmental services provided by flora in the tropical forest.

Photosynthesis is probably the most important biochemical process on earth. It allows plants to absorb certain wavelengths of the incoming radiation from the sun and transform its energy into organic compounds. Photosynthetically active radiation (PAR) is the amount of sunlight in the 400–700 nm wavelength range that is available for photosynthesis. Its agents are the photosynthetic pigments in the chloroplasts of which chlorophyll is the most important.

The leaf chlorophyll content is closely related to the plant's health and physiology. This characteristic has been considered to assess vegetation stress in agricultural areas and forest plantations [11–14], but studies of chlorophyll content in tropical rainforest environments, and specifically in the Amazon rainforest, are rare [15, 16]. A better knowledge of leaf chlorophyll content in the tropical forest is required to contribute to detecting and modeling vegetation stress during drought or pollution events by using satellite data and in this way better understand the potential of photosynthetic capacity and its implications in regional and global carbon cycle and climate models.

Traditional methods for estimating pigment content in vegetation need to be performed in a well-equipped laboratory. They require the extraction of plant pigments from the leaves by applying organic solvents such as dimethyl sulfuoxide (DSMO), methanol, ethanol, acetone, or ether. Depending on the solvent being used, the position of the maximum absorption of plant pigments varies due to the differences in polarity and the loss of pigment-protein interaction [17]. The extracted foliar solution is analyzed by a spectrophotometer in specific absorption wavelength ranges. Finally, absorbance is converted to chlorophyll concentration by applying equations described in the literature [18–21].

Alternative, nondestructive methods for chlorophyll estimation are available from spectral methods for plant pigment estimation. These methods are based on measuring light reflectance and transmittance properties of the vegetation using field spectroradiometers that can be carried in a rucksack, or from spectroradiometers on board of drones, planes, and satellites. They provide indirect estimations of relative pigment content expressed as an index, which needs to be converted to foliar pigment content through often a linear, a polynomial, or an exponential model. During the years, various vegetation indices (VIs) have been developed and applied to remotely sensed satellite images to quantitatively characterize the physiological status of vegetation. VIs are dimensionless measures that indicate relative abundance and activity of green vegetation, including leaf-area-index (LAI), percentage green cover, chlorophyll content, green biomass, and absorbed photosynthetically active radiation (APAR) [22]. VIs are obtained by adding, multiplying, or taking ratios of reflectance in two or more spectral bands of a pixel. These indices are classified into red/NIR ratios, green, red edge, and derivative indices. A useful description of chlorophyll indices can be found in [12, 17] and carotenoid indices [23–25].

This chapter focuses on the analysis of several optical approaches to estimate chlorophyll content in the tropical forest. The study sites were carefully selected across of a forest gradient degradation caused by land uses changes during the last decades. The optical approaches considered are transmittance, reflectance, and radiative transfer models at leave levels; and satellite-derived vegetation indexes at regional level. The objective of this study was to identify suitable methods to detect forest degradation caused by land use changes, deforestation, forest degradation, and pollution in the Amazon rainforest.

2. Alternative methods to measure chlorophyll content

2.1. Chlorophyll meter SPAD-502

Chlorophyll meters based on transmittance have been produced and are available commercially. They offer an inexpensive, easy, rapid, and portable approach for an indirect estimation of chlorophyll content. One of these is the SPAD-502 chlorophyll meter (SPAD-502, Konica-Minolta, Osaka, Japan) which bases its measurements on the light that is transmitted by the leaf in two wavelength regions: the first is located in the red region at 650 nm, which corresponds to the chlorophyll absorption peak unaffected by carotene, and the second is located in the infrared region at 940 nm where chlorophyll absorption is extremely low. The light emitted by the instrument and transmitted by the leaf is measured by the receptor and converted into electrical signals. Finally, a chlorophyll index is calculated by using the ratio of the intensity of the transmitted light [26]. Chlorophyll meters have been used extensively in agriculture to estimate chlorophyll and nitrogen in different species [27–31] and also in forest studies [15, 32–36]. Furthermore, chlorophyll meters have been used in the indirect assessment of foliar nitrogen [29, 30, 37], and carotenoid content [29, 38].

Chlorophyll content estimates in the tropical rainforest are rare. A published generalized homographic model for trees of the Amazon region [15] has been used as standard model to estimate chlorophyll content for more than 700 Amazonian tree species. A comparison of chlorophyll estimation between the homographic model and the second-order polynomial model proposed in this study illustrates good agreement for a wide range of SPAD-502 reading (15–95 units).

The accuracy of the SPAD-502 decreases at high chlorophyll index readings. When applying the proposed second-order polynomial model, caution should be taken for readings higher than 80 where estimation increases markedly compared to other optical methods (reflectance indices and PROSPECT) assessed in this study. Moreover, SPAD index has shown to be a valuable indicator to detect main impacts of land use changes in the tropical forest.

2.2. Reflectance indices

Another spectral method for chlorophyll content estimation is based on reflectance measurements to create pigment indices. Such indices take into account between two and four spectral bands and have shown high accuracy. Despite the literature offers several pigment indices, the majority of them have been tested in just specific plant species or vegetation type. As a result, they have become plant or vegetation specific. Estimations of chlorophyll content based on reflectance indices have been widely used [23–25, 33, 39–42].

Chlorophyll indices are increasingly being used in crops and forest assessments but also in ecology and Earth science. Several calibration models have been described in the literature, most of which, however, have been calibrated and validated in few or closely related plant species with a limited number of samples. Under these conditions, most of the models can only be applied to specific species and environmental conditions [23, 32, 43]. There is no scientific consensus as to whether a universal model can be found that can be applied for species-rich forest stands in different latitudes, phenological stages, and leaf structures [17]. Feret et al. [25] noted this limitation of the spectral indices and proposed new indices for chlorophyll and carotenoid estimation. They were based on a vegetation dataset collected in various ecosystems around the world including a wide variety of plant physiology and leaf structure.

2.3. Radiative transfer models: PROSPECT model

Based on the relationship between reflectance and the biochemical and biophysical properties of the leaves and canopies, models have been created in order to simulate the interaction of the light with the plant leaves through the radiative transfer theory. The leaf optical properties spectra (PROSPECT) model describes radiative transfer within a broadleaf with a plate model [44]. Plate models treat internal leaf structure as sheets or plates and calculate multiple reflections of diffuse radiation between these interfaces [13]. PROSPECT is based on the representation of the leaf as one or several absorbing plates with a rough surface giving rise to isotropic scattering. The model estimates the directional-hemispherical reflectance and transmittance of leaves across the solar spectrum from 400 to 2500 nm [45].

A leaf structure parameter of the model is represented by N, which is the number of compact layers specifying the average number of air/cell wall interfaces within the mesophyll. The leaf biophysical parameters of the model are represented by chlorophyll $a + b$ content (C_{ab}) and equivalent water thickness (C_w). The latest versions of the model include the parameters dry matter content (C_m) and brown pigments content (C_{bp}). Inversion of PROSPECT revealed good agreement between measured and predicted leaf chlorophyll concentrations [13, 45].

2.4. MTCI satellite vegetation index

The medium resolution imaging spectrometer (MERIS) terrestrial chlorophyll index (MTCI) is a standard product derived from MERIS satellite from the European Space Agency (ESA), which provides estimations of chlorophyll content of vegetation (amount of chlorophyll per unit area of ground) at global level. MTCI index is simple to calculate, sensitive to high values of chlorophyll content [46, 47] and estimations are independent to soil and atmospheric conditions, spatial resolution, and illumination and observation geometry [48]. Validation of MTCI index and ground chlorophyll content across a range of crop types and environmental conditions resulted in a strong relationship of $R^2 = 0.8$ and root mean square error (RMSE) = 192 g per MERIS pixel [49]. Moreover, the strong relationship of MTCI and canopy chlorophyll content has been used to estimate gross primary production (GPP) across a range of ecosystems. Boyd et al. [50] applied MTCI index, together with radiation information (photosynthetically active radiation—PAR and fraction of photosynthetically active radiation—fPAR), into models which extended the accuracy of GPP estimated.

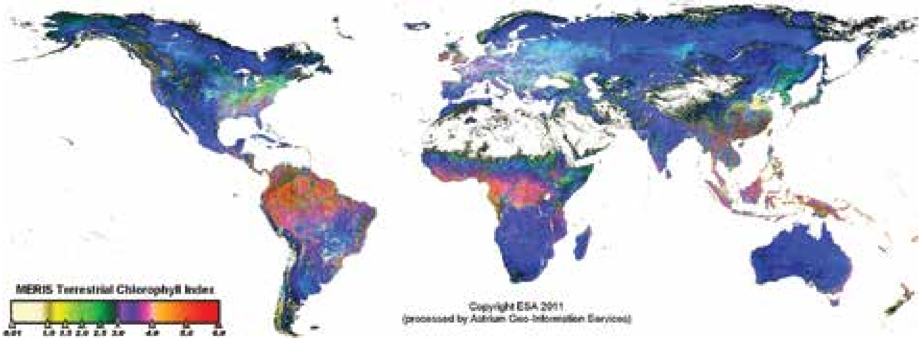

Figure 1. Global coverage of MERIS Terrestrial Chlorophyll Index at 31 May 2011. Processed by Astrium Geo-Information Services. Copyright ESA-2011.

MTCI is computed by the ratio of the difference in reflectance between band 10 and band 9 and the difference in reflectance between band 9 and band 8 of the MERIS standard band setting:

$$MTCI = \frac{R_{\text{Band10}} - R_{\text{Band9}}}{R_{\text{Band9}} - R_{\text{Band8}}} = \frac{R_{753.75} - R_{708.75}}{R_{708.75} - R_{681.25}} \quad (1)$$

where $R_{753.75}$, $R_{708.75}$, and $R_{681.25}$ are the MERIS reflectance at wavelength 753.75, 708.75, and 681.25 nm, respectively.

Figure 1 illustrates the global (Level 3) MERIS terrestrial chlorophyll index (MTCI) estimated at 31 May 2011. Highest MTCI values are located in the tropical forest biomes around the world.

2.5. The red-edge position (REP)

The red-edge position is a unique feature of green plants related to leaf chlorophyll content and to LAI. REP is defined as the inflection point (or sharp change) of the low red reflectance caused by chlorophyll absorption near 680 nm and high infrared reflectance governed by the internal structure of leaves near 750 nm [51]. REP has been used as an indicator of chlorophyll content in vegetation, as increasing chlorophyll content implies an enlargement of the chlorophyll absorption peak: this moves the red-edge to longer wavelengths while a decrease in chlorophyll shifts the red-edge toward shorter wavelengths [12]. However, the REP has been reported not to be an accurate indicator of chlorophyll content in vegetated areas showing high chlorophyll content values because of the asymptotic relationship between REP and chlorophyll content [52, 53].

Several methods have been proposed to estimate REP from spectral data coming from field and satellite sensors. Dawson and Curran [54] developed a three-point Lagrangian interpolation technique, but this method has shown some problems when the reflectance spectrum exhibits more than one maximum in its first derivative [51]. Another method was developed by Guyot and Baret [55], which applies a linear model to the red-NIR slope. This method has been reported to be robust when it was applied to various datasets [11]. A third method identifies the red-edge inflection point as the maximum of a curve fitted to the first derivative of the reflectance spectrum. This method has been closely related to chlorophyll content per unit area at leaf and canopy level [56] and has shown sensitivity to detect vegetation stress by quantifying changes in chlorophyll content [57].

3. Materials and methods

Fieldwork was undertaken from April to Jul 2012 at three sites in the Amazon tropical rainforest of Ecuador (**Figure 1**). The first and second study sites are located in a lowland evergreen secondary forest in Sucumbios province, Tarapoa region (0°11′ S, 76°20′ W). Site 1 has a history of petroleum pollution during the last decades. Mean annual rainfall is 3800 mm and the average annual temperature is 23°C with relative humidity close to 90% [58]. The area is located at 232–238 m above mean sea level. The third study site is a highly diverse lowland evergreen primary forest located in the Orellana province, in the northern section of Yasuní National Park (0°41′ S, 76°24′ W). The area lies 216–248 m above mean sea level and receives an annual average of 3081 mm rainfall with peaks in October and November. Mean monthly temperatures vary from 22°C to 34°C [59]. In this site, the Pontifical Catholic University of Ecuador established and manages permanent forest dynamics plots of 50 hectares where over 150,000 mapped trees ≥1 cm in diameter at breast height (dbh) from over 1100 species have been identified [60].

On one hand, the three study sites are located in the lowland Amazon forest sharing very similar ecological and environmental conditions. On the other hand, the forest in the three sites is substantially different due to the land use changes occurred during the last decades. Site 1 and Site 2 are disturbed forest that was exposed to selective logging, agricultural activities, petroleum industry impacts, and secondary forest regrowth over the last 20 years following diminishing human influence. Site 3 is a pristine primary tropical rainforest with legal protection status where a research project on plant and animal species diversity is currently conducted. Studies consider that the plant species richness in this area is among the highest in the world [61].

3.1. Sampling process

Well-developed branches were carefully selected and collected by using a telescopic pruner, tree-climbing techniques, and canopy towers at different levels of the vertical profile of the forest (**Figure 2**). The collected branches were sealed in large polyethylene bags to maintain their moisture content and stored in ice coolers. The foliar material was transported to a local site, and fully expanded mature leaves with no damage by herbivores or pathogens were selected for analysis. A total of 1134 samples were collected in the three fieldwork sites. The sampling process accounted for three levels of the vertical profile and included a wide range

Figure 2. Map of the study area—north-east Amazon region of Ecuador. Site 1 and Site 2 are located in Sucumbios province and Site 3 is located in Orellana province. Background is a Landsat image. Source of zoom-in map: Color shaded relief image, WorldSat International, Inc.

of vegetation heterogeneity related to species distribution, phenological stage, and leaf structure. (Detailed information about the sampled process can be found in Ref. [62].)

3.2. Chlorophyll meter readings

Depending on the size and shape of the leaf, different cork borers of variable size between 2.5 and 8.5 cm diameter were used to clip a leaf disk from the central and widest portion of the leaf blade, avoiding the major veins (**Figure 3**).

Figure 3. Photographs of leaf sampling process. (a) Collecting leaves using the telescopic pruner (b) climbing trees (c) telescopic pruner 9 m long (d) climbing trees techniques and (e) canopy towers in the study area.

All leaf disks were clipped from the midpoint of the leaves since it has been documented that it is the best position from which to take chlorophyll readings [37]. Three readings were taken from each disk using a portable SPAD-502 chlorophyll meter at different positions of each leaf disk, and a mean index value was used in further analysis.

3.3. Spectroradiometer measurements

Reflectance and "trans-flectance" (a term used in this study to describe the measurement of "double" transmittance) were measured for each leaf disk using an ASD FieldSpec Hand-Held-2 spectroradiometer (Analytical Spectral Devices Inc., Boulder, CO, USA). This instrument provides a wavelength range of 325–1075 nm with a resolution of full width at half maximum of 3.5 nm and sampling interval of 1 nm. The spectrometer is attached with a plant probe to an internal 4.05 W halogen light source and a leaf clip that includes rotating head with both white and black reference panels (**Figure 4**).

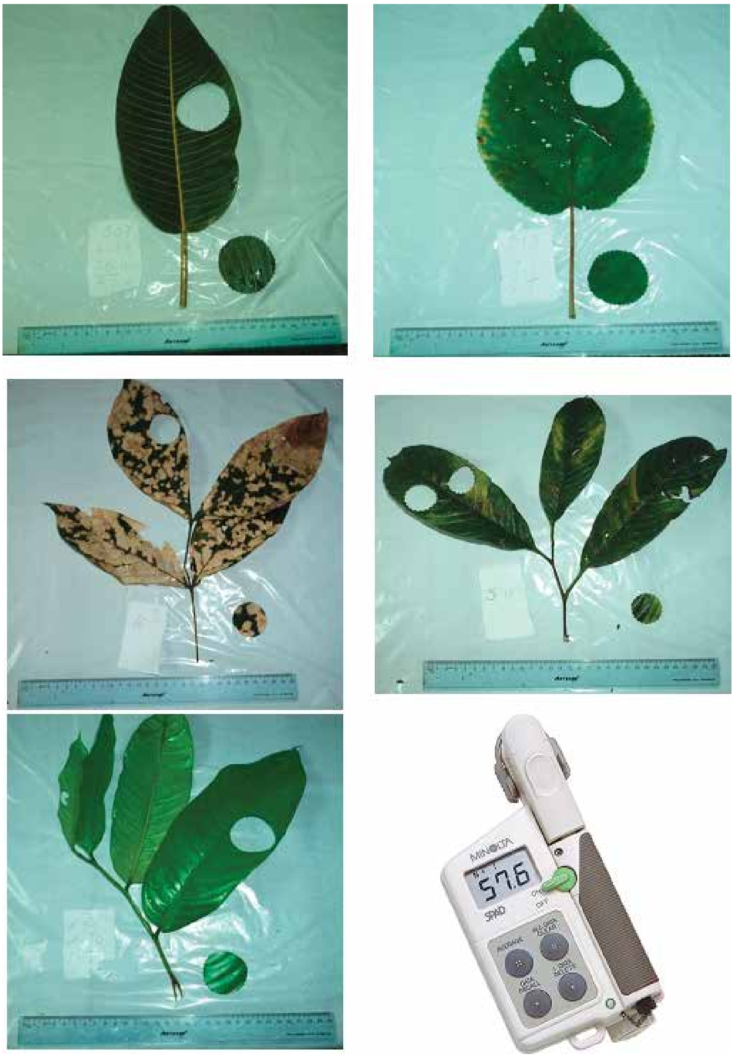

Figure 4. Leaf samples and clipped disks from plants of different species and health status. The last photo shows the SPAD-502 meter.

3.4. Chlorophyll indices based on SPAD-502 readings (transmittance)

Several published calibration models based on SPAD-502 readings were applied in this study. **Table 1** describes seven published polynomial, exponential, or homographic calibration models for chlorophyll content estimation from SPAD-502 chlorophyll meter readings. Selected calibration models cover a heterogeneous range of plants species, plant physiology, phenology, and growing conditions, which is a characteristic of the vegetation in tropical forests. All selected models have shown good agreement with traditional methods applied in a laboratory.

ID	Model	Units	Tested in	Number of samples	SPAD-502 range	Chl range (μm cm^{-2})	R^2
1	$\text{Chl} = 62.05e^{(X*0.0408)}$	mg cm^{-2}	6 Amazonian trees species	30–50 leaves per specie	3–80	~0–100	0.79
2	$\text{Chl} = (117.1*X)/(148.84–X)$	μg cm^{-2}	13 Amazonian trees species	391	0–80	0–150	0.89
3	$\text{Chl} = 2E\text{-}05X^2 + 1\text{E-}04\text{X} + 0.0038$	mg cm^{-2}	Lindera melissifolia	145	3.8–47.3	4–50	0.90
4	$\text{Chl} = 5.52E\text{-}04 + 4.04E\text{-}04X + 1.25E\text{-}05X^2$	mg cm^{-2}	Paper birch	100	~0–45	0.4–45.5	0.96
5	$\text{Chl} = 10.6 + 7.39X + 0.114X^2$	μmol m^{-2}	Soybean and maize	na.	0–70	~0–90	0.96
6	$\text{Chl} = 10(X^{0.265})$	μmol m^{-2}	Soybean and maize	na.	na.	~0–90	0.94
7	$\text{Chl} = 10(X^{0.264})$	μmol m^{-2}	Maize	na.	na.	na.	0.79

na. = Not available.
Source: (1) Ref. [35], (2) Ref. [15], (3) Ref. [31], (4) Ref. [33], (5) Ref. [28], (6) Ref. [28], and (7) Ref. [28].

Table 1. Indices of chlorophyll content estimation (μm cm^{-2}) based on SPAD-502 chlorophyll meter models applied in this study.

3.5. Chlorophyll indices based on reflectance indices

Five reflectance indices for chlorophyll content estimation found in the literature are described in **Table 2**. They considered the visible, red edge, and near infrared ranges. Chlorophyll content was estimated by applying linear or polynomial models for specific plant species when deriving these models. Selection criteria for reflectance indices were based on their ability to estimate chlorophyll content in a wide range of plant species, plant physiology, phenology, and growing conditions, which is a characteristic of the vegetation in tropical forests.

3.6. PROSPECT radiative transfer model

The inversion of the PROSPECT model using leaf reflectance and transmittance was applied in this chapter in order to estimate chlorophyll concentration. Foliar chlorophyll content (C_{ab}) was computed by the inversion process of PROSPECT 5 for the range of 400–1075 nm using reflectance and transmittance in the sampling interval of 1 nm for the 1134 leaf samples. Brown pigments (C_{bp}) and water content (C_w) were neutralized since foliar samples are green vegetation and the spectra does not show water absorption features.

ID	**Index**	**Model**	**Units**	**Tested in**	**Samples**	**Chl rang*e* (μm cm^{-2})**	**RMS*E***	**R^2**
8	$[1/(R_{680\text{-}730})]-[1/(R_{780\text{-}800})]*R_{755\text{-}780}$	Chl = $3.96*X^2 + 23.86*X - 3.31$	μg cm^{-2}	Temperate and tropical tree species and crops	1417	0.3–106.7	6.53	na.
9	R_{708}/R_{775}	Chl = $96.8*X^2 - 209.76*X + 115.08$	μg cm^{-2}	Temperate and tropical tree species and crops	1417	0.3–106.7	6.6	na.
10	$(R_{780} - R_{712})/(R_{780} + R_{712})$	Chl = $40.65*X^2 + 121.88*X - 0.77$	μg cm^{-2}	Temperate and tropical tree species and crops	1417	0.3–106.7	6.25	na.
11	$(R_{750\text{-}800})/(R_{710\text{-}730})$ -1	Chl = $716.32 * X$	mg m^{-2}	Maize and soybean	82	~0–100	6.07	0.95
12	$(R_{770\text{-}800})/(R_{720\text{-}730})$ -1	Chl=$37.904 + 1353.7X$	mg m^{-2}	Maize	2300	1–80.5	3.8	0.94

na. = Not available.
Source: (8) Ref. [25], (9) Ref. [25], (10) Ref. [25], (11) Ref. [40], and (12) Ref. [42].

Table 2. Chlorophyll content indices based on reflectance derived from spectroradiometer data.

3.7. MTCI index

In this study, MTCI was applied to foliar reflectance data collected at leaf level by the following equation:

$$MTCI_{\text{foliar reflectance data}} = \frac{R_{754} - R_{709}}{R_{709} - R_{681}} \tag{2}$$

where R_{754}, $R_{709.75}$, and R_{681} are the foliar reflectance at wavelength 754, 709, and 681 nm, respectively.

3.8. REP: first derivative method

The red-edge inflection point was estimated by the first derivative method:

$$D_{\lambda(i)} = \frac{R_{\lambda(i)} - R_{\lambda(i-1)}}{\Delta\lambda} \tag{3}$$

where $R_{\lambda(i)}$ and $R_{\lambda(i\text{-}1)}$ are reflectance at wavelength i and ($i - 1$), respectively.

3.9. Vegetation indices from satellite images: MTCI index

USGS EO-1 Hyperion image was that acquired on 15 February 2005. Hyperion data have a spatial resolution of 30 m^2 with each pixel covering the spectral range, 400–2500 nm. A single image is 7.65 km wide (cross-track) by 185 km long (along-track) covering the study sites 1 and 2 (secondary disturbed sites). After atmospheric and radiometric corrections, see more details in [62], MTCI index was derived to assess from space the main impacts of land use changes on chlorophyll content in the tropical forest.

4. Results

4.1. Chlorophyll content based on SPAD indices

Models 1– 7 shown in **Table 3** were applied to the SPAD-502 chlorophyll meter readings from the tropical forest study sites and the descriptive statistics of the estimates are shown in **Table 3**.

Model	Reference	Max.	Min.	Mean	SD
1	[35]	292.83	11.62	67.02	39.70
2	[15]	203.45	13.48	72.16	28.30
3	[31]	191.73	10.06	72.82	30.21
4	[33]	150.27	9.71	62.43	23.21
5	[28]	154.04	13.48	69.46	23.10
6	[28]	194.22	10.31	72.78	29.87
7	[28]	187.56	10.18	70.80	28.82

Table 3. Descriptive statistics of leaf chlorophyll content (μg cm^{-2}) based on seven published SPAD-502 chlorophyll meter models.

Figure 5 illustrates the chlorophyll content estimations for each model, its average values across models, and the confidence interval of 95% for the binned SPAD-502 readings. Estimations for the first six bins (range 15–80 SPAD-502 index) reported similar values. Average values at the higher SPAD index bin (80–95) show increase differences between models.

Figure 5. (a) ASD Hand Held 2 spectrophotometer (b) plant probe + leaf clip.

4.2. Chlorophyll content based on reflectance indices

Reflectance indices and their respective models were applied to the reflectance spectra to the samples collected for this study. The resulting descriptive statistics are shown in **Table 4**. Most of the mean chlorophyll estimations are lower than their counterpart based on SPAD-502 index.

Model	Reference	Max.	Min.	Mean	SD
8	[25]	126.82	1.06	57.92	17.48
9	[25]	78.63	5.05	53.17	11.26
10	[25]	85.80	6.22	54.37	12.29
11	[40]	101.66	5.86	50.05	14.59
12	[42]	136.69	12.51	65.77	18.91

Table 4. Descriptive statistics of chlorophyll concentration (μg cm^{-2}) from the reflectance models based on the spectroradiometer data.

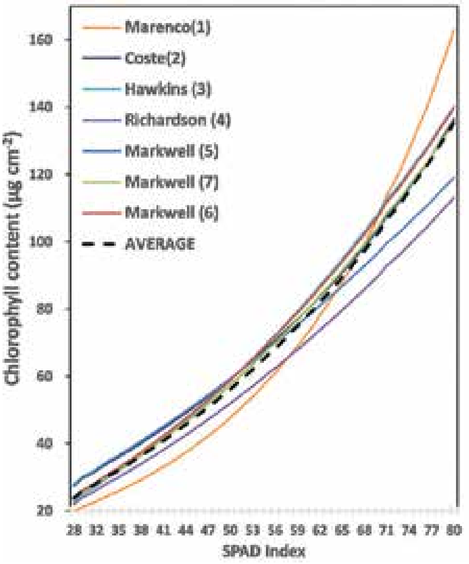

Figure 6. Estimated chlorophyll content for each SPAD-502 calibration model applied to the total samples of our dataset. The black line represents the average value across models and its confidential interval of 95% for the binned SPAD-502 readings.

Figure 6 illustrates the estimations of chlorophyll content for each reflectance model. It includes the average values across models and the 95% confidence interval for the binned SPAD-502 readings. It is interesting to observe that chlorophyll estimations become insensitive for SPAD reading greater than 80.

4.3. Comparison between the three methods for chlorophyll estimation

Figure 7 shows the comparison between average chlorophyll estimations from the three methods used in this study. Estimations until bin 50–60 are relatively similar. Estimation from SPAD then increased exponentially while estimations from reflectance and PROSPECT model are close to each other until bin 70–80, differences then increased since the asymptotic behavior of reflectance models estimations.

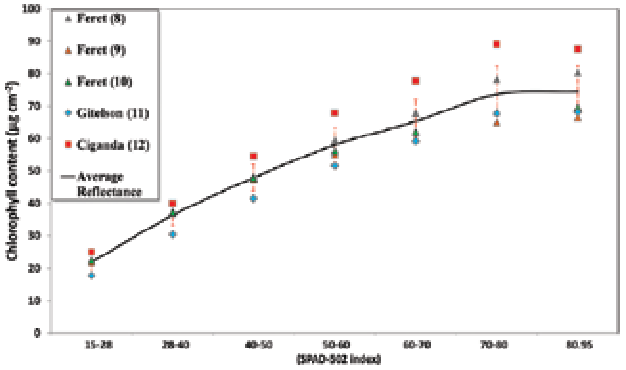

Figure 7. Average chlorophyll content estimates from five reflectance models (errors bars at 1.96 standard deviations) compared to estimated ground truth chlorophyll content based on SPAD-502 chlorophyll meter readings (error bars at 1.96 standard deviations).

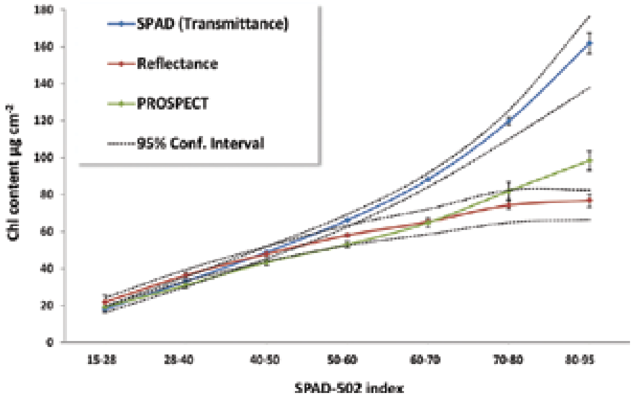

Figure 8. Comparison of average chlorophyll content estimates from the SPAD-502 chlorophyll meter index and the averages of all spectroradiometer-based chlorophyll estimates (error bars at 1.96 standard deviations).

Figure 8 illustrates the comparison of average chlorophyll content estimates from the SPAD-502 chlorophyll meter index and the averages of all spectroradiometer-based chlorophyll estimates. **Figure 9** presents the correspondent boxplots for the three approaches used in this study.

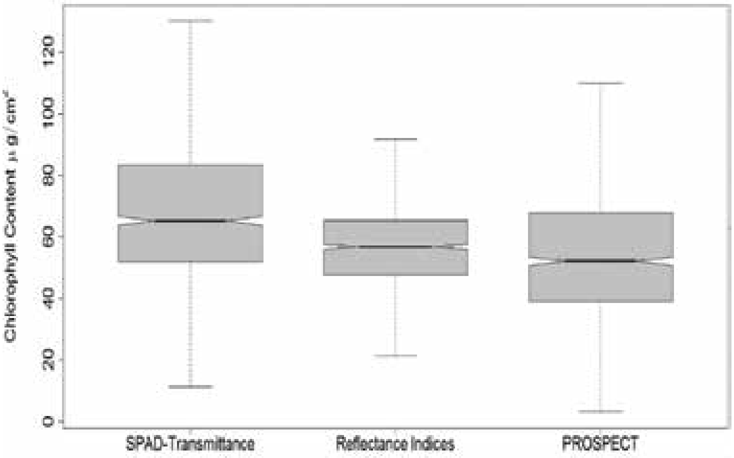

Figure 9. Boxplots of the three estimation of chlorophyll content (outliers not included).

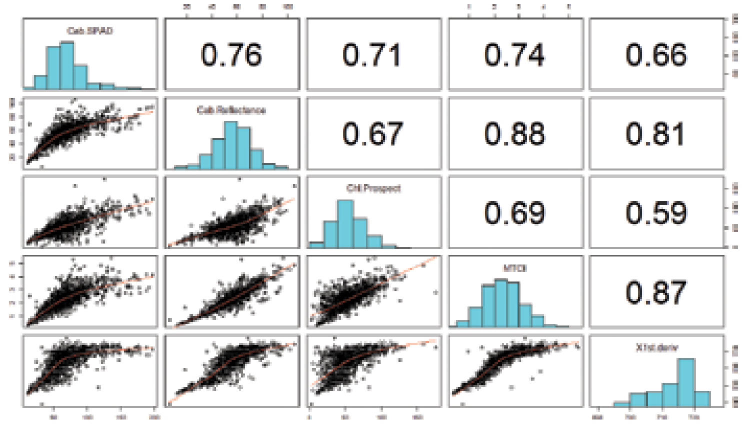

Figure 10. Scatter plots, histograms, and Pearson correlation between three chlorophyll estimations (SPAD, reflectance, and PROSPECT) and MTCI index and REP.

Figure 10 shows the correlations between the three chlorophyll estimations (SPAD-502, reflectance, and PROSPECT) applied in this study. Additionally, correlations with MTCI and REP are presented. Pearson correlation demonstrates a strong correspondence between the three methods calculated at leave level (SPAD-502, reflectance indices, and PROSPECT). Chlorophyll content estimates by the second-order polynomial based on SPAD-502 models and reflectance models agree in 0.76 while SPAD-502 models and PROSPECT agreed in 0.71. The lowest correlation (r = 0.67) is presented by estimations from reflectance models and PROSPECT model despite the fact that both methods are estimated from reflectance measurements. A strong correlation between them was found. MTCI and SPAD-502 correlate in 0.74, MTCI and reflectance models correlate in 0.88, and MTCI and PROSPECT correlate in

0.69. Correlation coefficients between REP and SPAD-502 model, reflectance models, PROSPECT, and MTCI are 0.66, 0.81, 0.59, and 0.87, respectively.

Figure 11 shows the estimations of leaf chlorophyll content based on SPAD index, MTCI and Ratio of derivatives. For the first two methods, chlorophyll content in the oil spill is significantly lower compared to the non-polluted sites.

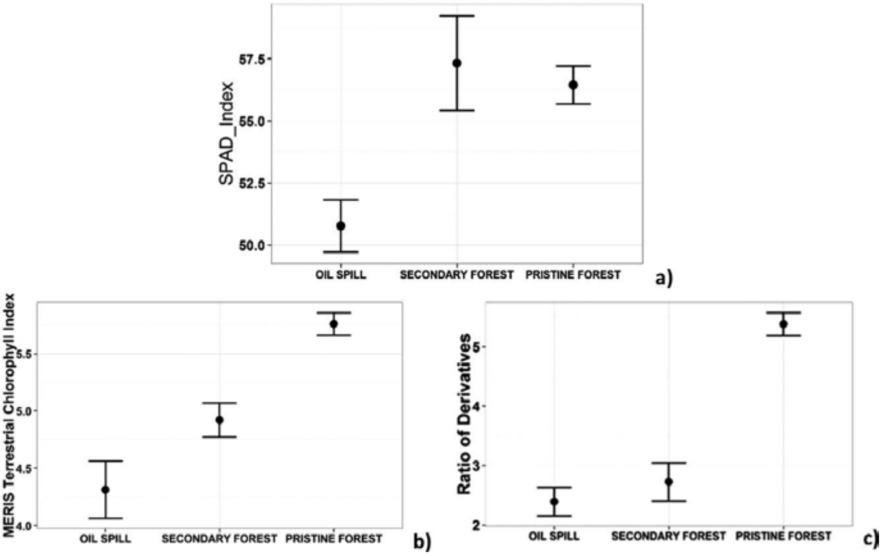

Figure 11. (a) SPAD chlorophyll index for the three study sites; (b) MERIS terrestrial chlorophyll index and (c) REP red-edge position-first derivatives for the three study sites.

4.4. Chlorophyll content evaluation

SPAD 502 chlorophyll content index and REP index were estimated for the three study sites. The results from **Figure 10(a)** and **(c)** shows that chlorophyll content was significantly lower (99.9%) at the secondary forest affected by pollution (Site 1) which allow us to conclude that forest degradation at local level can be detected using a portable chlorophyll content instrument. On the other hand, MTCI index derived from the satellite image also shows significantly lower values in the Site 1 **(Figure 10b)**, which confirm that chlorophyll content is a suitable indicator of land uses changes, and it can be applied at regional level to detect forest degradation caused by land use changes in the tropical forest.

MTCI index at regional level was computed using the Hyperion satellite images of the area corresponding to Site 1 and Site 2. **Figure 12** illustrates the results. Lower levels of chlorophyll (less than four) are found around the petroleum facilities and routes. On the other hand, higher levels of chlorophyll content (more than four) were found in areas still covered by the secondary forest.

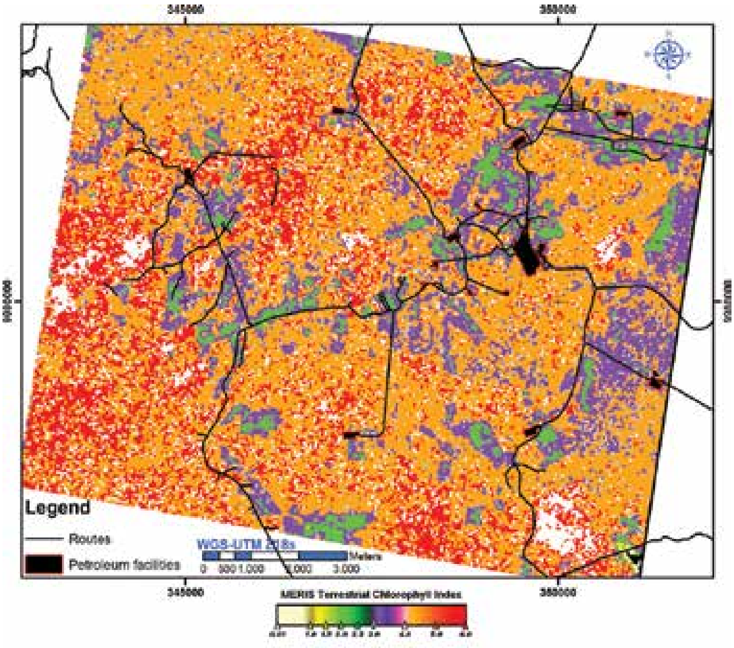

Figure 12. MTCI index computed from the Hyperion Satellite images of the study area of Site 1 and Site 2.

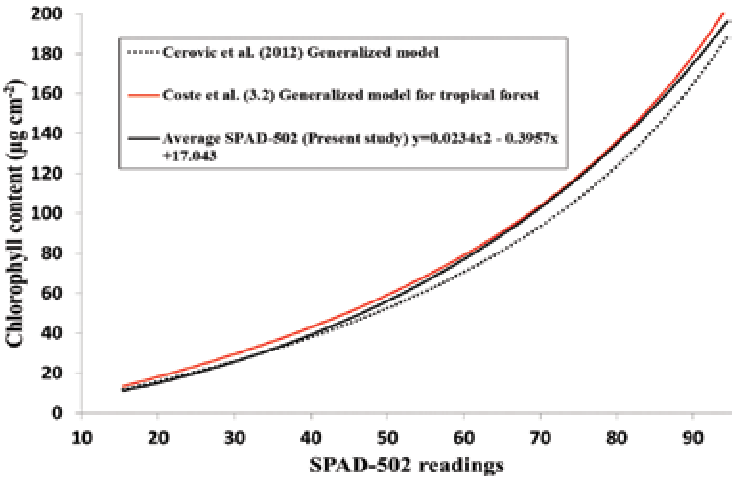

Figure 13. Comparison of three generalized models derived from SPAD-502 readings. The second-order polynomial model proposed in this study (black line), the homographic model proposed by Cerovic et al. [36] (dotted line), and the homographic model proposed by Coste et al. (2012) for trees from the Amazon forest.

Based on the results of the seven SPAD-502 published calibration models we compute their average in order to obtain a general model for chlorophyll content estimation which accomplish for a wide range of vegetation species and physiological stage. The resulting general model is a second order polynomial in a range of 15 to 95 SPAD index readings. This general model is proposed as ground truth chlorophyll which is assessed by comparing it to a reference published generalized model based on SPAD-502 readings and traditional methods in a laboratory. The first reference model is a homographic model proposed by Cerovic et al. (2012) and computed from seven (polynomial, exponential and homographic) models applied to a variety of plant species. The second model is the generalised homographic model for tropical trees proposed by Coste et al. (2010) which was discussed before as Model 2 in **Table 1**. **Figure 13** illustrates the comparison of the three models.

5. Discussion

Five methods for the estimation of chlorophyll content were applied to the collection of over 1100 leaf samples from the Ecuadorian Amazon rainforest, which represents a wide range of vegetation species growing in a disturbed and a pristine lowland rainforest. The first method is an optical method based on transmittance from the SPAD-502 chlorophyll meter index, the second method, also optical, is based on reflectance measurements collected by a spectroradiometer, and the third method is based on radiative transfer approach using the inversion process of the PROSPECT model. The other two methods are based on vegetation indices derived from satellite images.

For the first method, seven models that account for a wide range of vegetation species, phenological stage, and leaf structure showed close estimations between them until 80 SPAD-502 index (**Table 3** and **Figure 5**). At higher indices the differences increase. This can be explained by the fact that the calibration models considered a maximum SPAD-502 range of 80 units, meanwhile our database register readings beyond this range until 95 units. The best accuracy claimed by the instrument reaches its maxima until 50 units; therefore, higher values may be less accurate.

Based on the results of the seven SPAD-502 published calibration models, we compute their average in order to obtain a general model for chlorophyll content estimation which accomplish for a wide range of vegetation species and physiological stage. The resulting general model is a second-order polynomial in a range of 15–95 SPAD index readings. This general model is proposed as ground truth chlorophyll which is assessed by comparing it to a reference published generalized model based on SPAD-502 readings and traditional methods in a laboratory. The first reference model is a homographic model proposed by Cerovic et al. [36] and computed from seven (polynomial, exponential, and homographic) models applied to a variety of plant species. The second model is the generalized homographic model for tropical trees proposed by Coste et al. [15], which was discussed before as Model 2. **Figure 11** illustrates the comparison of the three models.

The proposed second-order polynomial model has the same concave shape and very close chlorophyll estimations along the range 15-95 SPAD-502 readings than the two homographic models. Homographic models have the generalized equation proposed by Cerovic et al. [36] and claims to be probably more accurate and certainly more rapid and portable than wet methods when used in crop plants. The model proposed by Coste et al. [15] was developed for the tropical forest from the Amazon region and has been a reference model for estimating chlorophyll content based on SPAD-502 readings.

Indeed, published SPAD-502 models applied to tropical rainforest vegetation are rare. A literature search by the authors only found two models (Model 1 and Model 2) developed for several species of the Amazon forest. Both experiments with tropical trees of the Amazon exhibited higher SPAD-502 readings which are comparable with our dataset. Those models account for a wide range of species, leaf structure, and phenology, and claim good accuracy for chlorophyll content estimation in multispecies forest stands. The homographic model proposed by Coste et al. [15] (Model 2) has been used to estimate chlorophyll content in a study that considered 1084 trees from 758 species across a broad environment gradient of 13 sites (seasonal flooded, clay terra firma, and white-sand forest) at opposite ends of Amazonia in Guiana and Peru [63]. The study relies on chlorophyll estimations based on the SPAD-502 model without considering traditional methods in a laboratory which prove the ability of a rapid and portable method of chlorophyll content in remote areas where analysis in a laboratory is not available.

Based on the comparison to published homographic models for multispecies, it is derived that the second-order polynomial calibration model offers a good approximation of chlorophyll content in tropical forest species. This is because of its close performance compared to the models proposed by Cerovic et al. [36] and Coste et al. [15] (**Figure 10**), and its homographic nature takes into consideration the reduced performance of chlorophyll meters at high chlorophyll contents. Indeed a homographic nature of SPAD-502 model has been applied to a wide range of tropical species from the Amazonia [63].

Estimations from the second method based on five reflectance models illustrate good agreements along all range of SPAD-bins (15–95 units). **Table 4** and **Figure 6** illustrate the results of these methods showing a saturation curve at the higher SPAD bind (80–95).

The observed maximum values of chlorophyll estimation from SPAD-502 (**Table 3**) are considerably higher than maximum values from reflectance indices (**Table 4**), which reflect the exponential increase of SPAD-502 models after 80 SPAD-502 units and the asymptotic nature of reflectance indices after this range. Differences between average estimations are less distinctive.

The first two methods are compared with the third method which is based on the inversion process of the PROSPECT model. **Figure 7** illustrates that the mean values are close to each other until 50–60, and after that the estimations based on SPAD-502 models increase faster than the other two methods. The method based on reflectance models and the PROSPECT model show close mean values until bin 70–80. Analysis of variance (ANOVA) and pairwise comparison between the three methods shown in **Table 5** indicate significant difference between the methods. Results from the lower SPAD-502 bin reported no differences between the methods.

	ANOVA	Pairwise comparisons between chlorophyll estimation methods (Holm adjustment method)		
	p-value	SPAD vs. reflectance	SPAD vs. PROSPECT	Reflectance vs. PROSPECT
All dataset	***	***	***	*
<28	ns	ns	ns	ns
28–40	***	**	.	***
40–50	***	ns	***	***
50–60	***	***	***	***
60–70	***	***	***	ns
70–80	***	***	***	**
80>	***	***	***	***

ns, nonsignificant.
***Strongly significant (0.1%) or lowest significant (10%).
**Highly significant (1%).
*Significant (5%).

Table 5. ANOVA and pairwise comparison between the three chlorophyll methods for chlorophyll estimation based on the binned SPAD-502 index.

Table 5 shows ANOVA and pairwise comparison between the three chlorophyll methods for chlorophyll estimation based on the binned SPAD-502 index.

Figure 10 presented that the chlorophyll estimations at leave level (SPAD-502, reflectance indices, and PROSPECT model) and estimations at regional level (satellite images) applied in this study show strong correlations between them. This finding demonstrates that a combination of field-based methods at leaf level with remote sensing methods at regional level may provide a good opportunity to evaluate forest health caused by land use changes. As it was stated in the introduction, forest degradation and its related changes in ecosystem services have not been fully assessed using remote sensing techniques, especially in high diverse tropical forest. The estimations of MTCI index in Site 1 and Site 2 shown in **Figure 12** have demonstrated lower levels of chlorophyll content caused by land use changes, specifically due the influence of petroleum facilities cause forest degradation. Therefore, in those areas accurate estimations of photosynthetic activity of forested areas are needed to quantify forest degradation and evaluate environmental services provided by flora in the tropical forest.

6. Conclusion

Three optical methods for estimation of chlorophyll content at leaf level were applied to the collection of over 1100 leaf samples collected in the Ecuadorian Amazon rainforest, which represents a wide range of vegetation species growing in a disturbed and a pristine lowland rainforest. The first method is based on transmittance from the SPAD-502 chlorophyll meter index, the second method is based on reflectance measurements collected by a spectroradiometer, and the third method estimates chlorophyll content from the radiative transfer PROSPECT model. For the first method, seven models that account for a wide range of vegetation species showed similar average leaf chlorophyll contents until 80 units of SPAD-502. An average of the results of these models was computed and used as ground truth from where a generalized second-order polynomial model was created. For the second method, five chlorophyll indices based on reflectance measurements provided similar chlorophyll content estimations for all SPAD range (15–95 units). The third method estimates chlorophyll content based on the inversion process of the PROSPECT model.

Comparison between the three methods shows that estimations until bin 50–60 are relatively similar, and estimations from SPAD increased exponentially. Estimations from reflectance and the PROSPECT model are close to each other until bin 70–80, after that differences increased since the asymptotic behavior of reflectance models estimations. A strong coefficient of correlations between the proposed generalized model and reflectance and PROSPECT approaches result in 0.76 and 0.71, respectively. Comparisons with MTCI and REP indicate correlations of 0.74 and 0.66, respectively.

The results of this study show that the relatively lightweight handheld field spectroradiometer can be used at field level to estimate leaf chlorophyll content in remote tropical rainforest ecosystems that are difficult to access. They provide a rapid and portable method for such remote areas where traditional chemical extraction methods for chlorophyll estimation are not viable. A general second-order polynomial calibration model for chlorophyll content estimation which accounts for a wide range of plant species, phenological stage, and leaf structure based on spectral measures offers an alternative approach for chlorophyll estimation. At a regional level, vegetation indices derived from satellite images are an efficient approach to detect chlorophyll content differences in vegetation exposed to main impacts of land use changes in the Amazon forest. These methods can be applied to regional scale to monitor the effects environmental services provided by the tropical forest and to detect forest degradation caused by land use changes.

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Chapter 4

Analyzing the Contribution of Cameroon's Council Forests to Climate Change Mitigation and Socioeconomic Development

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Additional information is available at the end of the chapter

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Abstract

Council forests were officially enacted in Cameroon in 1994 as part of the forestry law reform. The law provided rural councils with the legal right to create their own forests estate within the Permanent Forest Estate (PFE) of the State, following the preparation of a management plan approved by the forest administration. In this chapter, we analyze the socioeconomic and climate change mitigation potentials of these forests and propose possible options for improving their socioeconomic importance as well as their ability to mitigate climate change. Results indicate that Cameroon's council forests provide socioeconomic opportunities to communities in which they are located including employment and revenue from the sale of timber and nontimber forest products emanating from these forests. Additionally, given their diversity in terms of the various forest types (e.g., humid dense evergreen forests, humid dense semideciduous forests, and gallery forests), these forests have enormous carbon stocks which can provide huge opportunities for international climate initiatives such as the REDD+ mechanism to be initiated within them as a potential for mitigating global climate change. The chapter identifies and discusses possible options for improving the socioeconomic and climate change mitigation potential of these forests. Progress on the options the chapter opines, will help in improving the contributions of these forests to socioeconomic development and climate change mitigation.

Keywords: council forests, forestry law reform, socio-economic importance, climate change mitigation potentials, options and scenarios

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1. Introduction

Cameroon's Technical Centre for Communal Forestry (CTFC, 2010) defines a council forest as a forest that constitutes part of Cameroon's permanent forest, which is governed by an agreement between the municipality and the Ministry of Forests and Fauna (MINFOF). Cheteu [1] notes that Cameroon's council forests are usually endowed with a management plan executed by the council or municipality under the supervision of MINFOF.

The rate of deforestation in Cameroon forests including council forest remains one of the highest in the Congo Basin. According to the Food and Agricultural Organization (FAO), between 2000 and 2010, the annual rate of deforestation of Cameroon's tropical rainforest of the Congo Basin was approximately 1.04%. Furthermore, it is reported that about 75% of the forest in Cameroon (including council forest) has been degraded as a result of forest exploitation. As a result, development and conservation experts as well as indigenous rights advocates have embraced the management of council forest to address deforestation and forest degradation as well as improve the livelihoods of communities that depend on these forests. Management of these forests allows local populations to benefit from forests and its resources, as opposed to outside entrepreneurs or economic and political elites. By acquiring rights over natural resources, and related increase in organizational strength, the residents of the municipality as well as the local population can also improve participation in democratic processes. Communal control over these forests therefore decreases the opportunities of nonlocals to engage in destructive forest use, resulting in a positive conservation impact [2]. For instance, within the Guinean forest block, Liberia's council forests are significant for their rich biodiversity as they contain approximately 225 timber species, 2900 flowering plant species, 140 mammal species, 600 bird species, 75 reptile and amphibian species, and over 1000 species of insect [3]. While in Guatemala, the highland council forest landscapes are strategically important because of their location in the higher watershed areas that contribute to the maintenance of water sources; an aspect that is acquiring greater interest and relevance in light of declining water supplies in the area [4].

A majority of studies carried out to date on council forests in Cameroon have sought to describe the state of these forests and the participation of forest-dependent communities in their management [5–9]. At the same time, however, fundamental questions, such as: what are the opportunities associated with council forests in Cameroon especially in terms of their potentials for climate change mitigation and socioeconomic development of rural communities; what threats to these forests faced, and what are the possible options or scenarios that could be used to promote their sustainability, have been seriously overlooked. The purpose of this chapter, therefore, is to examine this much-neglected area of the debate using findings obtained from field work in Cameroon and a comprehensive review of relevant literature. Hopefully, policy makers at the national and subnational level in Cameroon will be able to incorporate the findings of this investigation into their strategic plans designed to advance sustainable management of these forests.

2. Historical evolution of council forests in Cameroon

Following independence in 1960 from France and 1961 from Britain, in order to protect and manage Cameroon's natural resources, a series of laws and decrees were enacted. Prominent among them were the 11th July 1968 Law which brought the notion of forest conservation and its resources, Order No. 73118 of 22nd May 1973 creating protected areas and recognizing the use rights of the local population, and Law No. 83/13 of 27th November 1983 fixing the forest, fauna, and fishery system. These regulations instituted a policy which insisted on the necessity of using forest resources in a rational manner, with the aim of maximizing productivity so as to offer the necessary revenue needed to perpetuate the wellbeing of the local population [10].

While all these regulations brought hope to nationals, they were not sufficient to guarantee the populations' participation in the management of forest resources; especially the conservation of the existing forests. The wind of sustainable development dictated by the international community around the 1990s doubled the determination of the Cameroonian government to effectively fight against poverty while maintaining its forest resources, led to the creation of the Ministry of Environment and Forestry (MINEF) in 1992.

In 1994, a new forestry law was voted by the national assembly promulgated by the President of the Republic of Cameroon. Central to this law, was the sustainable management of forest resources with a strong implication of the local population. The latter, which constituted one of the most important innovations of this law, was marked by the creation and regulation of decentralized forest management models including *inter alia* council forests [5].

With the adoption of the decentralization laws of 2004 and the ongoing process to strengthen the role of the council in the development of their area and the management of natural resources, the option of council forest continuously evolved while attracting more attention from the councils [8]. Since 2004, council forest landscapes in Cameroon have increased rapidly from 13 council forests in 2004 (collectively covering 325,500 ha); 18 council forests in 2006 (collectively covering 413,622 ha); 31 council forests in 2009 (collectively covering 734,751 ha); to a total of 34 areas designated as council forests (collectively covering 827,285 ha) as of June 2011.

Cameroon's council forests provide environmental services such as raw materials (mostly timber) that provide significant revenue for economic development in Cameroon. Additionally, this forest is rich in nontimber forest products (NTFPs) that are harvested by forest-dependent communities for commercialization and for subsistence purposes. It is also important to note that Cameroon's council forest also contains an enormous amount of carbon. The ecosystem of these forests can also provide other fundamental environmental services including watershed management, soil quality improvement, biofuels from forest residues, and biodiversity.

The establishment of council forest has also altered the land use changes in the regions where they have been set up as many municipalities have contracted their forest to large-scale logging companies for forest exploitations. These companies are engaged in large scale and intensive logging operations and can devastate a council forest in a little time interval.

3. Methods

3.1. Carbon stocks assessment

We conducted field work to estimate carbon stocks within 10 council forests in Cameroon. The decision to choose these case study council forests was borne out of the consideration that they were the only council forests that were operational in the country and/or had a management plan. Above ground carbon in these forests was estimated as follows.

3.1.1. Sampling

In each forest, a rectangular sample plot was selected randomly in the forest zone where there was no human activity (area of the forest were there was no degradation). The area of the main rectangular plot was $20 \times 100 = 2000\ \text{m}^2$. Within this main rectangular plot, the diameter of trees (DBH) greater than 30 cm were measured and recorded. Additionally, within this main plot, trees with DBH between 5 and 30 cm were also measured and recorded. Understory (like epiphytes, ferns, and herbs) and litter with a DBH of less than 5 cm were also harvested within the main plot and were weight to determine their fresh weight. Next, they were dried at 65°C in order to determine their dry weight.

3.1.2. Biomass and carbon stock calculations

Aboveground biomass (AGB) was estimated for each council forest using the allometric equation developed by Chave et al. [11] for moist tropical forest:

$$AGB = \rho \times \exp[-1.499 + 2.148 \ln 2.148 \ln(\text{DBH}) + 0.207 \ln \ln(\text{DBH})^2 - 0.0281 \ln \ln(\text{DBH})^3],$$

where ρ is wood specific density, DBH (cm) is the diameter at breast height, and AGB (kg) is dry mass. The estimation of the aboveground C stock was based on the assumption that all the AGB biomass pools contain 47.5% of C [12, 13]. The total C stored in each council forest was calculated using the C stock per unit area and the total area of the council forest.

3.2. Structured interviews

Structured interviews were held with officials of the ten council forest to determine (i) the socioeconomic potentials of these forests particularly in terms of their contribution to employment at the local level and income generated from these forests through the sale of timber and nontimber forest products (NTFPs), and (ii) the threats, difficulties or constraints that these forests are currently facing particularly in terms of bush fire, illegal logging, illegal hunting, management conflicts, nonrespect of contracts by loggers, high costs of exploitation, abusive exploitation of nontimber forest products, and absence of public participation in forest management decision-making process.

4. Results

For illustrative purposes, results of this study are presented in five sections including: (1) provision of income to the municipality and local communities; (2) provision of employment opportunities; (3) valorization of forest resources; (4) provision of environmental services; and (5) threats faced by council forests

4.1. Provision of income to the municipality and local communities

Like other forests domain in Cameroon, council forests provide raw materials (mostly timber) that generate substantial income for economic development of the municipality in which they are located (**Figure 1**). Additionally, they are rich in nontimber forest products (NTFPs) which could be harvested by the municipality and other local communities for commercialization and subsistence purposes. For example, in a personal communication with an official of the Dimako council forest, it was reported that in June 2012, 2,484,000 FCFA[1] was obtained from the sale of Djansang (*Ricinodendron heudelotti*) emanating from this forest.

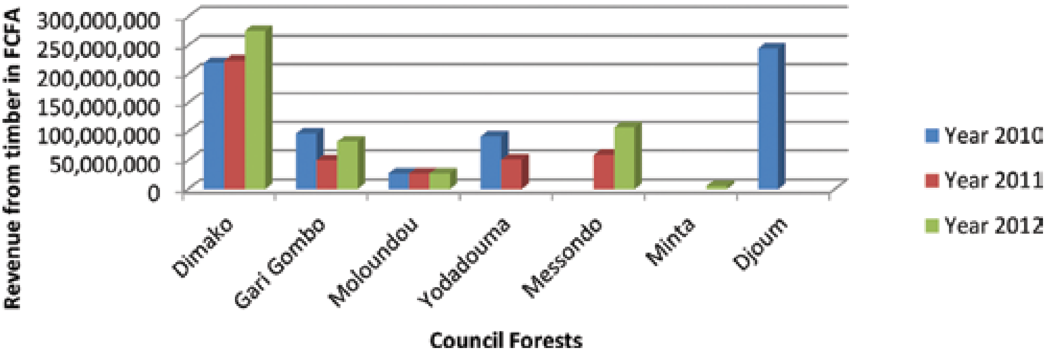

Figure 1. Revenue (in FCFA) generated from timber emanating from the case study council forests.

4.2. Provision of employment opportunities

Council forests provide local employment opportunities. As Cuny [9] notes, about 85% of council forests employees are local indigenes. The contribution of the case study council forests to local employment and in terms of average monthly income is depicted in **Figures 2** and **3**, respectively. Apart from the Dimako council forest where almost all the logging operations are carried out by people directly employed by the council, most of the other councils have contracted their forest to large-scale logging companies for the purpose of timber extraction.

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[1] 2,484,000 FCFA is almost equal to 5000 USD.

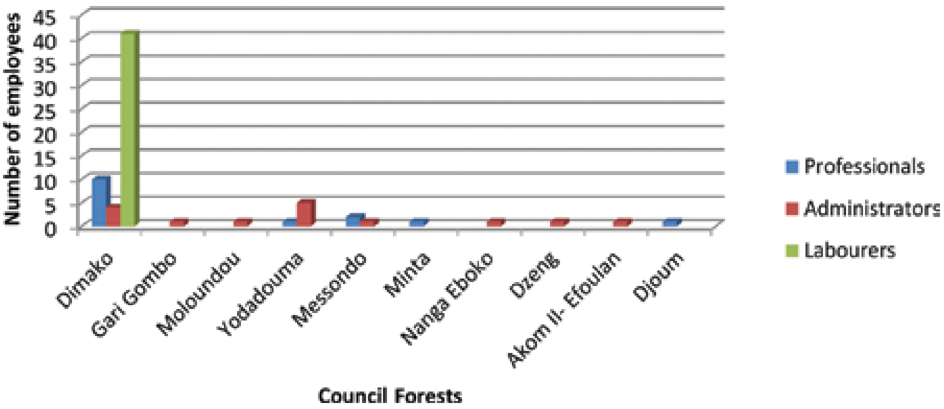

Figure 2. Number of people working with the case study council forests.

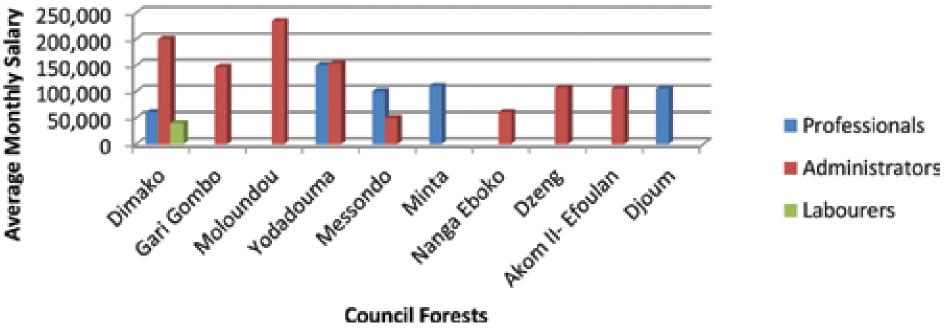

Figure 3. Average monthly wages for people working with the case study council forests.

4.3. Valorization of forest resources

According to the forestry and decentralization laws of Cameroon, council forestry is an opportunity for the valorization of forest resources emanating from council forests. The sustainability of valued-added wood products has been well documented. For instance, as Kozak [14] puts it:

"Catalyzing the value-added wood products sector has been embraced by most stake-holders – government, industry, organized labor, communities, Aboriginal peoples, environmental groups – as a sensible and rational vehicle to transform the forest sector…. Deriving more value and creating more jobs per volume of wood cut is seen as a conservation-based strategy for attaining the tenuous balance between economic well-being, environmental sustainability, and community health and vitality."

Market opportunities for value-added wood products currently exist both within Cameroon and all over the world [15]. Indeed, in the United States alone, higher value wood products represent a US$200 billion market [14]. Apart from value-added wood products, valorization of NTFPs from council forests in Cameroon can provide a great opportunity for improving the livelihoods and income of forest dependent communities as well as municipalities were the forest is located. As Tieguhong et al. [16] explain, valorization of nontimber forest products (NTFPs) in African communities can increase the revenue of these communities, thus contributing to poverty alleviation.

4.4. Provision of environmental services

Given the diversity of Cameroon's forest in terms of the various forest types (e.g., humid dense evergreen forests, humid dense semideciduous forests, and gallery forests), these landscapes have enormous carbon stocks. This can provide huge opportunities for international climate initiatives such as the REDD+ mechanism to be initiated in these forests as a potential for mitigating global climate change. In this study, carbon stocks within the case study council forests were evaluated (see **Figure 4**). **Figure 4** shows the carbon contained in the biomass within each forest. Most of the carbon is concentrated in the tree biomass, followed by dead trees. The carbon content of other features (litter, understorey, and palm trees) was very negligible and could not be seen in **Figure 4**.

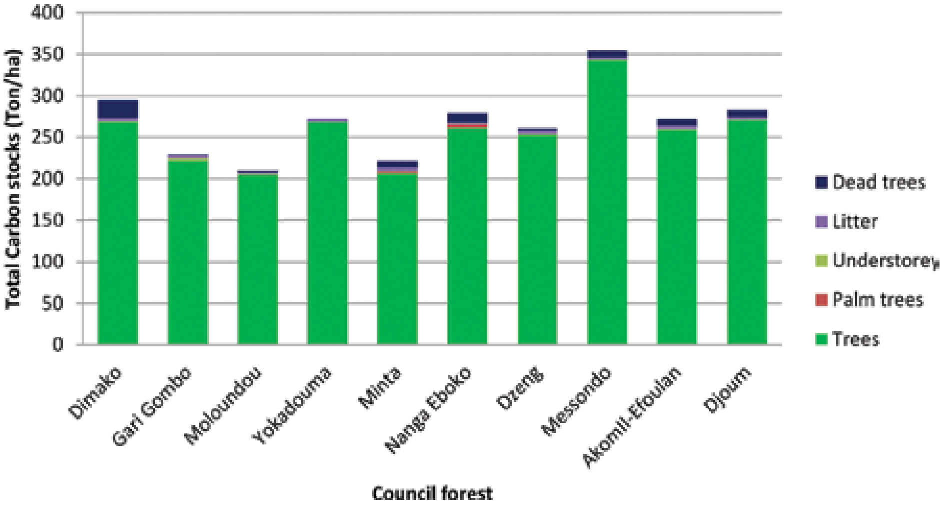

Figure 4. Total carbon stocks within the case study council forests.

4.5. Threats to Cameroon's council forests

4.5.1. Illegal logging

According to Cuny [9], illegal logging is a common practice within some council forests in Cameroon. In a visit to one of the council forest in Cameroon, Om Bilong et al. [17] noted a prominent case of illegal logging practices. In a series of personal interviews with some council officials, it was revealed that actors involved in this illicit practice include "unidentified persons coming from other communities with local residents as accomplice and as a result of poor forest monitoring from forest guards." The issue of poor forest monitoring as one of main factors unpinning illegal logging in Cameroon is supported in the prevailing literature [15, 18, 19]. Others have identified poor forest governance from the relevant ministries as the root cause for this illegality [15, 20, 21]. Indeed, as Cerutti et al. [21] report, each year, Cameroon's State officials may be collecting an estimated sum of 6 million Euros in the form of informal payments or bribes from illegal chainsaw loggers operating in the country.

4.5.2. Potential source of conflict

The council forest of Efoulan for example is linked to that of AKOM II; this can be a potential source of conflict between the two municipalities in future if practices put in place to promote the sustainable management of these forests are not clearly established in terms of objectives and targets.

4.5.3. High cost involved

The financial expenditures involved in establishing a council forest is usually high; about 50 million FCFC without including the fees for gazettement, exploitation, monitoring, and revision of the management plan [9]. These difficult and almost unrealistic financial requirements involved in the process of setting up a council forest have left local authorities at the mercy of private donors thereby relinquishing their autonomy in decision making (every donor has its own requirements which sometime run contrary to the objectives of the local council).

4.5.4. Illegal hunting

One major threat to council forest in Cameroon is illegal hunting. According to Lindsey et al. [22], illegal hunting is the hunting of protected species, without licenses/permits, in areas where it is prohibited, or using prohibited methods. During personal interviews with council forest officials, it was reported that actors involved in this practice include local indigenes and individuals from Central Africa Republic. They noted that they indulge in this illegal exercise for commercialization and subsistence purposes. Some major drivers of illegal hunting include increase demand of bush meat in both rural and urban areas, absence of other alternative livelihoods, and inadequate enforcement of regulations [22].

5. Discussion

The previous section has provided the socioeconomic opportunities offered by council forests in Cameroon, their climate change mitigation potentials, as well as salient threats faced by these forests. Possible options for addressing these threats and improving their climate change mitigation potentials and socioeconomic importance are highlighted and discussed in the information that follows.

5.1. Reduced impact logging

Reduced impact logging is one option that could be used to improve carbon stocks in the case study council forests. The term Reduced impact logging and its acronym RIL were first used in 1993. Traditionally, it often describes a set of forest management techniques that reduce logging impacts and improve productivity. Putz et al. [23] define it as "intensive planned and carefully controlled timber harvesting conducted by trained workers in ways that minimize the deleterious impacts of logging." Over the past two decades, sets of timber harvesting guidelines designed to mitigate the deleterious environmental impacts of tree feeling, yarding, and hauling have been known as RIL techniques. Although RIL techniques have been described as covering a variety of practices with no standard definition, De Blas and Manuel [24] define RIL techniques as: the delimitation of protected forest areas within concessions; the determination and use of minimum tree diameter at breast height; the development of a management plan and a logging inventory; minimizing the width and density of logging road networks; planning of logging roads; setting a maximum ceiling on the number of trees felled by hectare; use of directional felling; optimizing timber transport road networks; and planning of timber yards.

Despite its variability in countries, most RIL guidelines are also components of most forest management plans, often starting with recommendations related to designation of forest management units and progresses rapidly through issues related to assignment of annual coupes (i.e., cutting areas), before considering in more detail issues related to road and log landing planning, layout, and construction [23]. In Cameroon like the entire Congo basin area, RIL techniques are included into forestry laws especially those associated with mandatory management plans; i.e., preharvest planning of logging roads, determining diameter at breast height, or timber yards planning [24]. Although improved forest management (RIL inclusive) was not included in the Kyoto Protocol as an option for carbon sequestration, ample evidence is already available that selective logging using RIL techniques increases forest retention of carbon relative to conventionally logging [23]. **Figure 5** shows the total amount of carbon in the case study council forest as well as the amount of carbon obtained from conventional logging compared with RIL. It is glaring that relative to conventional logging, carbon stock in these forests increases when RIL is practiced. The calculations were done following the work done by Durrieu de Madron et al. [25] on the estimation of the impact of various type of forest exploitation on C stock in Central Africa. According this work, the extraction one cubic meter of timber per hectare would lead to the loss of 0.73 t of carbon. In conventional logging operations, if 20 m^3 of timber is exploited per hectare, a total carbon stock of 20 × 0.73 t of carbon are loss per logged hectare, plus C loss due to logging skid tracks (7% of the productive area × 0.00195 kg C/m^2) and roads (1% of the productive area × 0.028 kg C/m^2). Thus, for instance, an exploitation of 180,000 ha of forest under conventional logging would then lead to the loss of 20 × 0.73 × 180,000 t C + 245,800 t C (from skid tracks) + 504,000 t C (from roads). That makes a total of about 3,378,000 t C. The application of RIL would preserve about 517,700 t C from this loss. These figures were therefore used to estimate the impact of RIL and conventional logging on carbon stocks in each council forest.

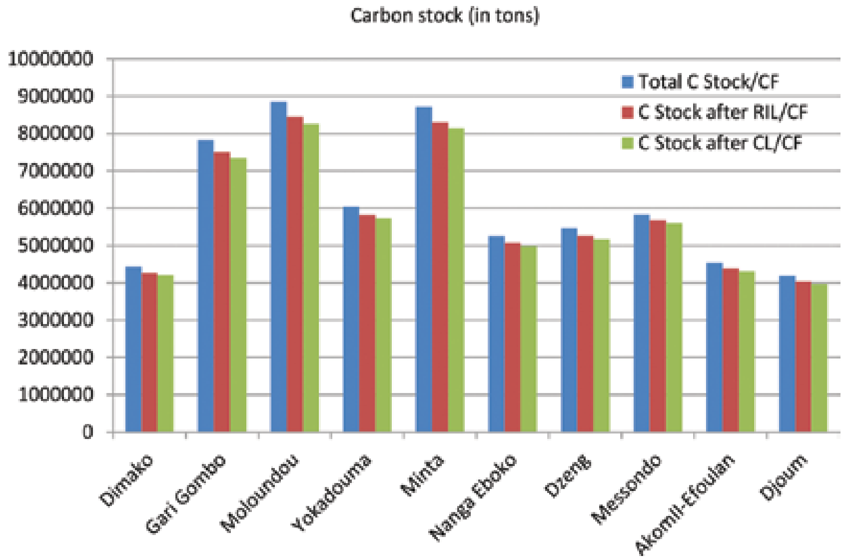

Figure 5. Total carbon stock and carbon stock in the case study council forests after RIL and conventional logging.

5.2. Reforestation

Reforestation constitutes another possible approach that could be employed to improve carbon stocks within Cameroon's council forests. Under the definitions of the Marrakesh Accords, reforestation refers to the direct human-induced conversion of nonforested land to forested land through planting, seeding, and/or the human-induced promotion of natural seed sources, on land that was forested but that has been converted to nonforested land [26]. Simply put, reforestation is planting trees or other activities geared towards the expansion of forest cover in general, though with particular reference to natural forest succession [27], or areas cleared of forests through timber harvesting and/or natural disaster.

Climate change mitigation through carbon sequestration is usually the primary benefit of reforestation as efforts to increase terrestrial carbon sequestration are based on the premise that reforestation adds to the planet's net carbon storage and helps moderate global warming by slowing the growth of carbon emissions in the atmosphere. However, there are many other ecological benefits of reforestation outside of carbon sequestration. Reforestation of degraded lands provides restoration of forest ecosystem goods and services (especially forest-based carbon), biodiversity conservation, improved air and water quality as well as improved soil fertility, structure and sustainability [27], and habitats for wildlife.

5.3. Promotion of good governance in order to combat illegal logging

Accountability, transparency, and jail terms for defaulters should be more aggressively promoted and applied around managing forest resources and ensuring that the proceeds derived from these economic activities are used to enhance the overall objectives of both the council forest and surrounding communities. The jailing of the former major of the council of Yokadouma for embezzlement of proceeds from the Yokadouma council forest provides a step in the right direction.

5.4. Improve capacity and data

In order to promote effective and efficient monitoring, there is a need to improve capacity and data. One immediate option of doing this is to create strategic partnerships with international organizations like the Food and Agricultural Organization (FAO) that is currently implementing a regional project on REDD+ MRV in the country. The project envisages training with inputs from the Brazilian Institute for Space Science (INPE) and also negotiates remote sensing data from them that might be helpful. However, moving toward community managed MRV approaches may be more useful as this has been demonstrated to be potentially more effective and efficient elsewhere in Asia [28].

5.5. Promote easy access to credit schemes

The inability of council forest operators to cope with the exorbitant cost associated with their establishment and management results partly from their inability to secure financial support and credit schemes from most financial institutions. The government has done minimal efforts to lobby on their behalf, which is fundamental to improving their ability to procure this support. Thus, it is recommended here that cash flow to councils be improved with an important starting point being a fundamental change in the lending schemes of most financial institutions especially those operating in the country.

5.6. Develop alternative livelihood projects and enforcement of wildlife laws.

This chapter presses the need for alternative livelihood projects like bee keeping and the cultivation of snails to be promoted in forest-dependent communities that live around council forest landscapes in Cameroon as a strategy of reducing illegal hunting. As Lindsey et al. [22] explain, for illicit hunter to be curbed, livelihood interventions that generate alternative protein supply must be combined with proper enforcement of the relevant legal regulatory framework that prevails in that jurisdiction.

6. Conclusions

This chapter attempts to analyze council forest landscapes in Cameroon in terms of their socioeconomic and climate change mitigation potentials. The chapter posits that while there are some strengths and opportunities associated with these landscapes, they are fraught with glaring weaknesses as well as threats which could undermine importance services that they provide. Thus, it is argued that promoting good governance in order to combat illegal logging, improving capacity and data, facilitating easy access to credit schemes, developing alternative livelihood projects, RIL, reforestation, and proper enforcement of wildlife laws are keys to advancing the sustainable management of these landscapes. Advances on these main recommendations will help in improving the contributions of council forest landscapes to desired sustainable development pathways. Finally, lessons learned from this study could be replicated to other countries in West Africa (like Liberia) where the Guinean forest block are significant for their rich biodiversity.

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Chapter 5

Analysis of Precipitation and Evapotranspiration in Atlantic Rainforest Remnants in Southeastern Brazil from Remote Sensing Data

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Additional information is available at the end of the chapter

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Abstract

The Atlantic Rainforest has been intensely devastated since the beginning of the colonization of Brazil, mainly due to wood extraction and urban and rural settlement. Although the Atlantic Rainforest has been reduced and fragmented, its remnants are important sources of heat and water vapor to the atmosphere. The present study aimed to characterize and to analyze the temporal dynamics of precipitation and evapotranspiration in the Atlantic Rainforest remnants in São Paulo state, southeastern Brazil, for the period from January 2000 to December 2010. To achieve this, global precipitation and evapotranspiration data from TRMM satellite and MOD16 algorithm as well as forest remnant maps produced by SOS Mata Atlântica Foundation and Brazilian National Institute for Space Research (INPE) were used. Results found in this study demonstrated that the use of remote sensing was an important tool for analyzing hydrological variables in Atlantic Rainforest remnants, which can contribute to better understand the interaction between tropical forests and the atmosphere, and for generating input data necessary for surface models coupled to atmospheric general circulation models.

Keywords: hydrological variables, Atlantic Rainforest, South America, TRMM, MOD16, remote sensing

1. Introduction

The Atlantic Rainforest stretches across Brazil, Argentina, and Paraguay, and is considered the second largest tropical forest in the American continent [1]. In Brazil, the Atlantic Rainforest covers 17 states, from Ceará to Rio Grande do Sul, and is located along coastal and inland regions, over mountains and plateaus [2]. The great longitudinal and latitudinal extension and, consequently, the wide variability in rainfall and temperature provide, combined with altitude gradient and ancient vegetation formations, a high degree of biodiversity and endemism [3].

The Atlantic Rainforest was heavily devastated since the beginning of Brazilian colonization, mainly due to wood extraction and rural and urban settlement. Thus, from the original forest cover, only isolated remnants with variable sizes in different successional stages were left [4]. Recent studies show that only 8.5% of the original Atlantic Rainforest, estimated in 1.3 million km^2, still exists [5]. The state of São Paulo (henceforth referred to as SP), despite its high levels of agricultural and urban development, presents the largest remnants of Atlantic Rainforest in the country. Estimates show that 13.9% of the original Atlantic Rainforest still exists in SP [6].

Although Atlantic Rainforest has been reduced and fragmented, its remnants are an important source of heat and water vapor to the atmosphere. This is because latent heat released as evapotranspiration influences the atmospheric circulation in the tropics and the water vapor contributes to the regional precipitation regime [7]. In this context, knowing the annual and interannual variability of precipitation and evaporative processes in tropical biomes is necessary for a better understanding of the energy and water partitioning between surface and atmosphere, which allows for a better parameterization of the boundary layer processes used in climate and weather forecasting models [8, 9].

Usually, precipitation and evapotranspiration are measured by instruments equipped in conventional meteorological stations; however, these measurements are expensive and do not represent well the spatial variability of these processes [10]. Hence, the use of remote sensing techniques becomes a methodological alternative since it enables to obtain different biophysical parameters at the Earth's surface with high temporal and spatial coverage. The Tropical Rainfall Measuring Mission (TRMM) [11] and the MOD16 algorithm [12, 13], developed, respectively, to estimate global surface precipitation and evapotranspiration, have been widely used by the scientific community in large-scale hydrological studies [14].

The purpose of this study was to characterize and analyze, based on both TRMM and MOD16 imagery, the temporal dynamics of precipitation and evapotranspiration in the Atlantic Rainforest remnants of SP, southeastern Brazil, during a 10-year period (January 2000 to December 2010). We have implemented a wavelet transform to evaluate the temporal variability of these parameters. Wavelet analysis is becoming a common tool for researches involving remote sensing and land-atmosphere interactions. It provides an efficient method for extracting relevant information from large datasets and has been applied to a wide range of variables and different types of ecosystems [15].

2. Materials and methods

2.1. Study area

The study area is located in SP, southeastern region of Brazil (**Figure 1**). The Atlantic Rainforest remnants are mainly located in the slopes of Serra do Mar (1), Bocaina (2), and Mantiqueira (3) mountains and Ribeira (4) and Paraíba (5) valleys, where the natural vegetation cover was less affected due to the difficulty of agricultural mechanization [16]. The main formations of Atlantic Rainforest observed in SP are dense ombrophilous Forest, mixed ombrophilous forest and seasonal semideciduous forest [6].

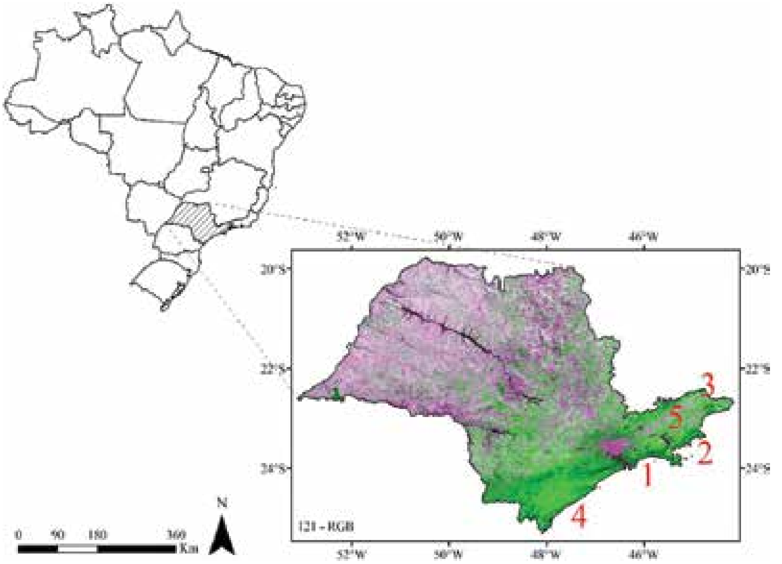

Figure 1. Map showing the location of São Paulo State, Brazil. The color composite was obtained from MODIS/Aqua images of June 23, 2006. The green areas represent dense vegetation, while the beige, magenta, and black areas represent, respectively, agriculture (mostly pasture and sugarcane plantations), bare soil (or urban areas), and water bodies. The numbers 1–5 show, respectively, the location of the slopes of Serra do Mar, Bocaina and Mantiqueira mountains and Ribeira and Paraíba valleys.

2.2. TRMM data

The TRMM satellite was designed from a cooperative program between National Aeronautics and Space Administration (NASA) and Japan Aerospace Exploration Agency (JAXA). Its main goal is to monitor the distribution of precipitation in tropical and subtropical regions [11]. The satellite was launched in 1997 and has three main sensors onboard for studying precipitation: (i) precipitation radar (PR), (ii) microwave imager (TMI), and (iii) Visible and Infrared Scanner (VIRS). PR is an active sensor, the first of its kind in orbit, presenting as the most important characteristic for studying precipitation providing a three-dimensional view of the structure of precipitation [17]. TMI is a passive microwave radiometer operating in five frequencies that provide information about the integrated content of the precipitation column, intensity and type of precipitation. The VIRS sensor, derived from the AVHRR/NOAA sensor, has five spectral bands in visible and infrared regions performing observations of clouds, such as cover, type, and top temperature [18].

For this study, monthly precipitation data derived from TRMM (3B43 product) version 7 (v7) were used, covering the period from January 2000 to December 2010. The 3B43 product is calculated using data from multiple satellites, in addition to TRMM, as well as meteorological stations data from the Global Precipitation Climatological Center (GPCC) and the Climate Assessment and Monitoring System (CAMS) [19]. 3B43 imagery were acquired from the electronic address (http://gdata1.sci.gsfc.nasa.gov/daac-bin/G3/gui.cgi?instance_id=TRMM_Monthly), presenting spatial resolution of ~30 km in mm month^{-1}. Images were processed in ENVI version 4.5, where the steps of reprojecting, resampling of pixels to 1 km (same spatial resolution of MOD16 data) using nearest neighbor method, and clipping to Atlantic Rainforest remnants in SP were performed.

2.3. MOD16 data

The MOD16 algorithm [12, 13] was developed in the context of the Earth Observing System/NASA (EOS/NASA) program, aiming to estimate global evapotranspiration using data from Moderate Resolution Imaging Spectroradiometer (MODIS) sensor (Terra and Aqua) and meteorological data from Global Modeling and Assimilation Office (GMAO). In general terms, MOD16 is a revision of the algorithm proposed by [20], who adapted the Penman-Monteith equation (Eq. (1)) to be used with remote sensing data:

$$\lambda E = \frac{sA + \rho C_p \left(e_{\text{sat}} - e\right) / r_a}{s + \gamma\left(1 + r_s / r_a\right)} \tag{1}$$

where λE is the latent heat flux (W m^{-2}) and λ represents the latent heat of evaporation (J kg^{-1}), $s = d(e_{\text{sat}})/dT$ is the slope of the curve which relates saturated pressure of water (e_{sat}) and temperature (Pa K^{-1}), A is the energy available at surface (W m^{-2}), ρ represents air density (kg m $^{-3}$), C_p is the specific heat of air (J kg^{-1} K^{-1}), e is the real pressure of water vapor (Pa), r_s is the surface resistance, r_a is the aerodynamic resistance (s m^{-1}), and γ represents the psychrometric constant (66 Pa K^{-1}).

MODIS input data required for MOD16 algorithm have spatial resolution between 500 m and 1 km, and include global products of land use and land cover (MOD12Q1), leaf area index (LAI) and photosynthetically active radiation (PAR-MOD15A2), and albedo (MCD43B2). Regarding the meteorological parameters required for the algorithm, daily reanalysis data of GMAO referring to incident solar radiation, air temperature, and water vapor pressure, with spatial resolution of 1.00° × 1.25°, are used [13, 21]. In summary, MOD16 data have a spatial resolution of 1 km and covers ~109 million km^2 of vegetated global areas. Among the products generated, we highlight the potential and actual evapotranspiration and potential and actual latent heat flux products, in intervals of eight (MOD16A2) and 30 days (MOD16A3) [22].

MOD16 data were acquired from the Numerical Terradynamic Simulation Group/The University of Montana repository (http://www.ntsg.umt.edu/project/mod16). Tiles H13V10 and H13V11, corresponding to the monthly real evapotranspiration product, in mm month^{-1}, were selected for the period between January 2000 and December 2010. As MOD16 data are available in sinusoidal projection, images were initially reprojected to geographic coordinates with datum WGS 84 and converted to GeoTIFF format using the MODIS Reprojection Tool (MRT). Then, a number of steps were undertaken using ENVI 4.5. These steps included clipping of the study area, multiplication by scale factors, and application of the land-water and urban areas mask over the datasets.

2.4. Atlantic rainforest remnants

Since the 1980s, the SOS Mata Atlântica Foundation, jointly with the National Institute for Space Research (INPE), is regularly mapping forest cover in the Atlantic Rainforest biome. These institutions use remote sensing imagery to produce the "Atlas of Forest Remnants of Atlantic Rainforest." Resulting maps enable us to determine the spatial distribution of forest remnants and ecosystems associated to Atlantic Rainforest, keep track of changes in vegetation cover, and generate permanently improved and updated information of this biome [23, 24].

The spatial distribution of the forest remnants was obtained from the database provided by [5]. This database was used to update the period corresponding to 2011–2012, that is, to update changes occurred in the polygons previously classified as forest fragments (forest remnants, mangrove, or restinga) in previous versions of the "Atlas." To this end, images of the LISS III/RESOURCESAT-1 orbital sensor corresponding to the second semester of 2012 were used. Vector files of the forest fragments polygons were acquired from the electronic address http://mapas.sosma.org.br/ and edited in ARCGIS version 9.3. The editing process consisted in selecting polygons of forest remnants with area equal or greater than 100 ha. Selecting only ≥100 ha polygons intended to create a spatial homogeneity of the analyzed areas. It should be noted that the study comprehends the period between 2000 and 2010 and it is understood that the forest remnants mapped by [5], referring to the update of 2011–2012, are representative of the period analyzed.

2.5. Wavelet analysis

Wavelet analysis has become a widely used method to study variations of energy in environmental time series [25, 26]. The decomposition of a time series in the time-frequency space allows the determination of dominant modes of variability and its variation modes in time [27]. Time series for TRMM and MOD16 were analyzed with continuous wavelet transform using the algorithm developed by [28]. Generally, continuous wavelet transform is used to visualize, in a three-dimensional diagram, the relationship between components of different frequencies according to the time scale of the series studied [29]. Several functions are used to generate wavelets; in this study, the Morlet complex function was used, which is composed of a plane wave modulated by a Gaussian envelope, as shown in Eq. (2):

$$\Psi(\eta) = \pi^{-1/4} e^{i\omega_0\eta} e^{-\eta^2/2} \tag{2}$$

where η is the dimensionless time parameter, and ω_0 represents the dimensionless frequency. Here, it is important to point out that Torrence and Compo algorithm was compiled in MATLAB version 7.9.0 and that the analysis was performed exclusively for forest remnants of Atlantic Rainforest. Therefore, values used to generate wavelets referred to the monthly average precipitation (TRMM) and evapotranspiration (MOD16) in the analyzed forest remnants.

3. Results and discussion

3.1. Precipitation and evapotranspiration in São Paulo State between 2000 and 2010

Figure 2 shows the spatial distribution of monthly average precipitation (January to December) in SP between 2000 and 2010 obtained from TRMM satellite data.

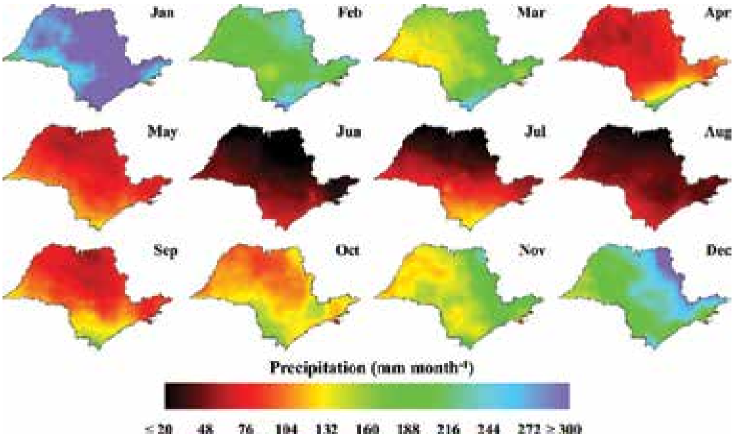

Figure 2. Spatial distribution of monthly average precipitation (mm month^{-1}) in SP for the period between 2000 and 2010.

Generally, it is noted that images from January to March and from October to December show higher precipitation as compared to April to September. This reflects the well-defined rainfall regime in SP: the rainy season (October to March) and the dry season (April to September) [30]. It is possible to note that in most of the year highest values of precipitation are located in the Coastal Plain and Serra do Mar regions. This dynamic is associated to frontal systems (cold fronts) and the South Atlantic Convergence Zone (SACZ), which occur during the year in SP and act mainly in the areas near the coast, as well as the fact that the Serra do Mar conditions the formation of orographic rainfall through the condensation of humid winds from the ocean [31]. In contrast, lower values of precipitation are observed over the year in the Western Plateau region, where organized local convection is the main source of rainfall [32].

Figure 3 shows the monthly precipitation in SP between 2000 and 2010. Monthly precipitation ranged between 4.3 (August 2004) and 386.9 mm month^{-1} (January 2003), which indicates an absolute variation of 382.6 mm month^{-1}. On average, monthly precipitation between 2000 and 2010 was 128.9 mm month^{-1}.

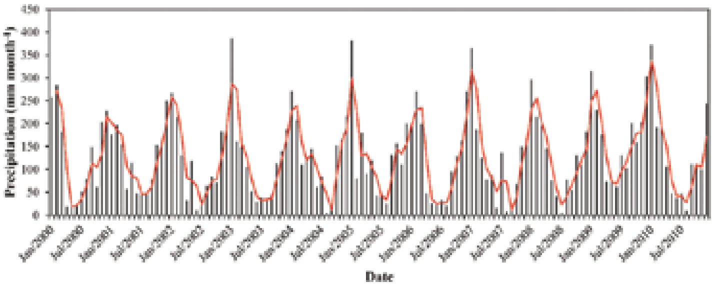

Figure 3. Monthly precipitation (mm month^{-1}) in SP between January of 2000 and December 2010. The red line represents the moving average of the time series (period = 2).

Average monthly precipitation ranged from 35.8 to 298.5 mm month^{-1}, where June is the driest month and January is the wettest. This result is observed in the January and June images shown in **Figure 2**. These two images differ significantly when compared to the other images, especially the image of January, since the image of June has some resemblance to the image of August. In June, it is possible to note that most of precipitation is lower than 48 mm month^{-1}, except in the Southern region, where values close to 76 mm month^{-1} were found. Regarding to the image of January, most of precipitation is higher than 300 mm month^{-1}, except in the western edge of the state, where values of ~216 mm month^{-1} were found.

Analysis of the dry season (April to September) and the rainy season (October to March) has revealed that the average monthly precipitation was, respectively, 64.5 and 193.2 mm month^{-1}. Therefore, average month precipitation in the rainy season was ~200% higher than the observed average in the dry season. Annual precipitation in São Paulo State ranged between 1403.5 and 2029.5 mm year^{-1}. In this sense, 2002 was the least rainy year, while 2009 was the most rainy year. Average annual precipitation was 1546.5 mm year^{-1}, with ~25% of that occurring in the months corresponding to the dry season and ~75% of the average annual precipitation in the months corresponding to the rainy season. Monthly precipitation in 2002 ranged between 10.4 (June) and 267.8 mm month^{-1} (January), while in 2009 monthly precipitation ranged from 62.4 (June) to 314.7 mm month^{-1} (January).

It is important to note that TRMM satellite estimates were not validated in this study. In this context, researches present in literature suggest relative errors ranging from ~5 [33] to ~25% [34]. Still, it is noted that the results regarding the precipitation regime in SP are consistent with several observation meteorological studies conducted in the state, such as [35, 36].

Figure 4 shows the spatial distribution of monthly average actual evapotranspiration (January to December) in São Paulo State between 2000 and 2010, derived from MOD16 algorithm.

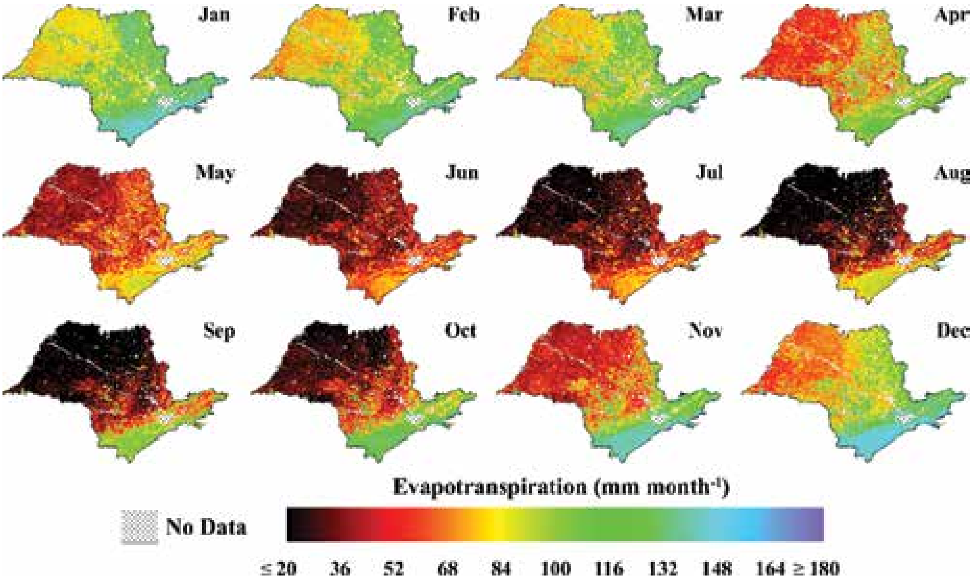

Figure 4. Spatial distribution of monthly average actual evapotranspiration (mm month^{-1}) in SP corresponding to the period between 2000 and 2010.

Visual inspection of **Figure 4** reveals a spatial and temporal pattern for evapotranspiration similar to the one found in precipitation (**Figure 2**). However, evapotranspiration images provide a better perception of subtle changes along the state. Generally, images corresponding to the rainy season have higher values for evapotranspiration when compared to images of the dry season. Evaporation depends on variation in solar radiation, local atmospheric circulation process, which regulates the precipitation system and air and soil moisture conditions, and vegetation conditions, which show considerable changes following the rainy or dry season [37]. Among these conditions, solar radiation stands out, whose incident amount depends, among other factors, on the season [38]. Therefore, this pattern is expected because highest incidence of solar radiation occurs during the rainy season [39]. It is also worth mentioning that throughout the year highest values of evapotranspiration are located in the southern and eastern SP, while lowest values are situated in the northern and western regions of the state.

Figure 5 shows monthly average actual evapotranspiration in SP between 2000 and 2010. Monthly evapotranspiration varied between 26.1 and 116.8 mm month^{-1}, representing an absolute variation of 90.7 mm month^{-1}. Accordingly, lowest monthly value was found in July 2000 and the highest in January 2003. Considering the period between 2000 and 2010, monthly evapotranspiration corresponded, on average, to 68.2 mm month^{-1}.

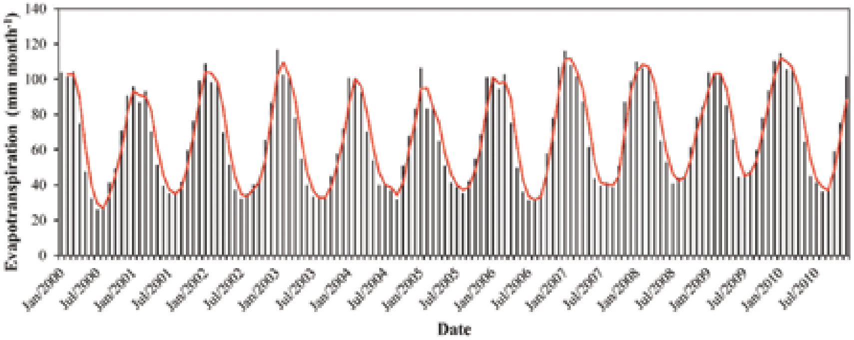

Figure 5. Monthly actual evapotranspiration (mm month^{-1}) in SP between January of 2000 and December 2010. The red line represents the moving average of the time series (period = 2).

The months of August and January presented, respectively, lowest and highest monthly average evapotranspiration (values of 36.6 and 107.1 mm month^{-1}). Relating precipitation and evapotranspiration, it denotes that August had the second lowest monthly average precipitation, while January had the highest monthly average precipitation.

Monthly average evapotranspiration in the dry season was 48.0 mm month^{-1}, while in the rainy season it corresponded to 88.3 mm month^{-1}, which shows an increase of ~84% in evapotranspiration during the wetter period of the year in São Paulo State. Annual evapotraspiration values ranged between 765.7 and 942.0 mm year^{-1}, with 2003 and 2009 presenting, respectively, lowest and highest estimates. In 2003, monthly evapotranspiration ranged from 32.7 (August) to 116.8 mm month^{-1} (January), while in 2009 monthly evapotranspiration ranged between 44.6 (June) and 110.4 mm month^{-1} (December). Regarding yearly average evapotranspiration, the estimate found for the period between 2000 and 2010 was 817.9 mm year^{-1}. On average, for the period between 2000 and 2010, evapotranspiration accounted for ~53% of precipitation in São Paulo State.

It should be noted that MOD16 algorithm estimates were not validated for this study. Ideally, validation process should be performed using surface measurements throughout SP in order to identify biases in the estimates found according to the conditions studied. However, there is a lack of such information for the study area, both the spatial and temporal perspective, which prevents this type of analysis. For comparison, [22], in a validation study for the MOD16 algorithm, found relative errors of 18–22% in tropical forest areas, 20% in seasonal flooding areas and 33% in agricultural areas. Finally, it should be noted that results found about the evapotranspiration regime in SP agree with the results from a modeling study using the Simple Biosphere Model (SiB2) performed by [40].

3.2. Precipitation and evapotranspiration in Atlantic Rainforest remnants between 2000 and 2010

Figure 6 shows the remnants of the Atlantic Rainforest in São Paulo State with area ≥ 100 ha, and the overlapping of the remnants mapped in São José do Rio Preto region (northwestern

SP) with a MODIS/Terra sensor image from June 27, 2010. It is possible to note that most of the remnants are located in South and East portions of the State, in contrast to the northern and western regions [16]. Yet, as observed in the highlighted image, polygons mapped by [5] are properly adjusted to MODIS images, which are the basis of the MOD16 algorithm used in this study.

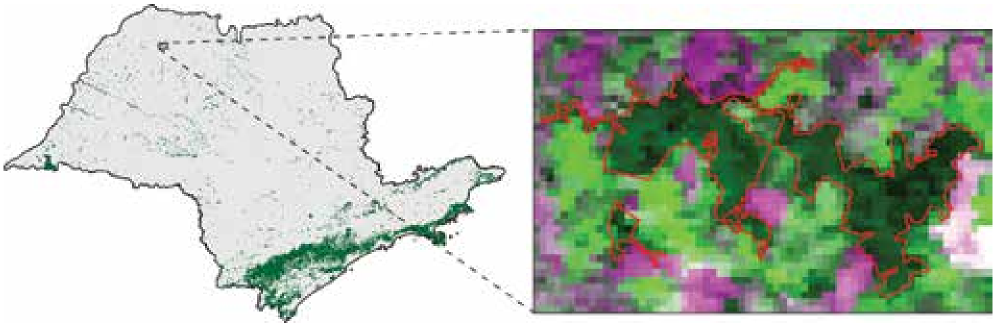

Figure 6. Atlantic Rainforest remnants in São Paulo State with an area equal or greater than 100 ha. Highlighted image shows the overlapping between remnants polygons mapped in São José do Rio Preto region and a MODIS/Terra sensor image (R1G2B1) from June 27, 2010.

For the period of 2011–2012, 25,554 polygons were mapped in São Paulo State, totalizing an area of ~2,421,538 ha. After the selection of the polygons with area ≥ 100 ha, 2054 were found, representing an area of ~1,914,331 ha. In here, it is important to mention that analysis related to precipitation and evapotranspiration were realized only for Atlantic Rainforest remnants with area equal to or greater than 100 ha.

Figure 7 shows precipitation (monthly and monthly average) in Atlantic Rainforest remnants between 2000 and 2010. It is possible to note a strong seasonality in Atlantic Rainforest remnants precipitation, similar behavior found in previous analysis for São Paulo State (Section 3.1). Monthly precipitation ranged between 11.7 and 460.1 mm month^{-1}, values found, respectively, in July 2008 and January 2010. Considering the entire period (2000–2010) monthly average precipitation was 114.7 mm month^{-1}. [41] observed, in Atlantic Rainforest areas in São Paulo State, monthly precipitation ranging between 1.5 and 347.3 mm month^{-1}. Moreover, during the period analyzed, monthly precipitation in Atlantic Rainforest remnants was, on average, ~12% higher than that estimated for SP.

Monthly average precipitation ranged from 49.6 (June) to 309.5 mm month^{-1} (January). In this sense, Donato et al. [41] estimated, for Atlantic Rainforest areas in São Paulo State, monthly average precipitation between 33.8 (August) and 272.0 mm month^{-1} (January), similar to those obtained in this study.

During the dry season, monthly average precipitation in remnants was 85.7 mm month^{-1}, while in the rainy season was 203.8 mm month^{-1}. Therefore, monthly average precipitation in the Atlantic Rainforest remnants was ~138% higher in the rainy season. Annual precipitation ranged from 1426.6 (2007) to 2185.4 mm year^{-1} (2009). Thus, annual precipitation showed an absolute and relative variation of 758.8 mm year^{-1} and ~53%, respectively. In 2007, monthly precipitation fluctuated between 18.7 (June) and 268.2 mm month^{-1} (January), while in 2009 it ranged from 63.3 (June) to 298.6 mm month^{-1} (January). Annual average precipitation in Atlantic Rainforest remnants was 1737.0 mm year^{-1}. In addition, ~30% of the annual average precipitation occurred during the dry season, and ~70% was concentrated in the rainy season. Similar studies by [41, 42] found, for Atlantic Rainforest remnants areas in SP, annual average precipitation of 1784.0 and 1974.1 mm year^{-1}, respectively.

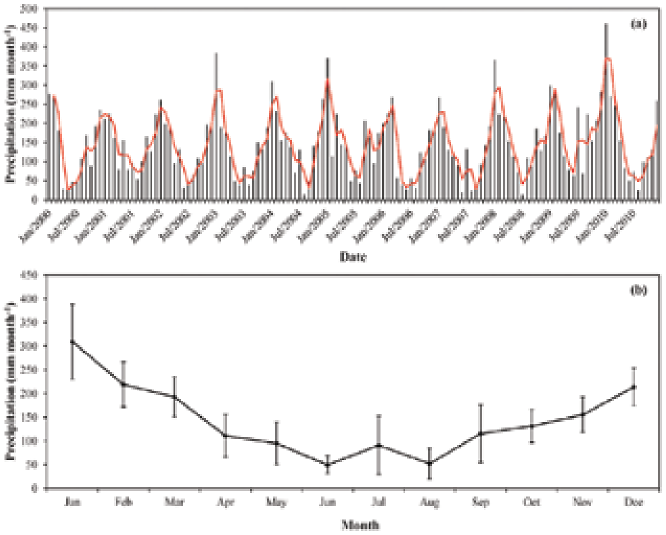

Figure 7. Monthly (mm month^{-1}) (a) and monthly average (mm month^{-1}) (b) precipitation in Atlantic Rainforest remnants of São Paulo State between January 2000 and December 2010. In (a), the red line represents the moving average of the time series (period = 2), and in (b), vertical bars represent the standard deviation.

Figure 8 shows actual evapotranspiration (monthly and monthly average) in Atlantic Rainforest remnants between 2000 and 2010. It is possible to note the temporal variability of the values found, characterizing the seasonality of this parameter and presenting well-defined dry and rainy seasons, as mentioned in Section 3.1. Considering the period studied, monthly evapotranspiration oscillated between 55.3 and 144.3 mm month^{-1}. Accordingly, lowest value was found in July 2000, while the highest in December 2002. On average, considering the period between 2000 and 2010, monthly evapotranspiration was 104.03 mm month^{-1}. [43], considering an experimental microbasin located in an Atlantic Rainforest area in the municipality of Cunha, obtained monthly evapotranspiration values oscillating between 26.5 and 142.3 mm month^{-1}, similar to those obtained in the present study. It is worth mentioning that, considering the period analyzed, monthly evapotranspiration in Atlantic Rainforest remnants was, on average, ~52% higher than monthly evapotranspiration in SP.

Monthly average evapotranspiration ranged from 63.2 (June) to 139.3 mm month^{-1} (December). Comparing these results with monthly average precipitation, June was the month with lowest precipitation, while December was the third wettest month. In this context, [44], in a study conducted at the Serra do Mar State Park, found monthly average evapotranspiration between 35.8 (July) and 95.0 mm month^{-1} (January).

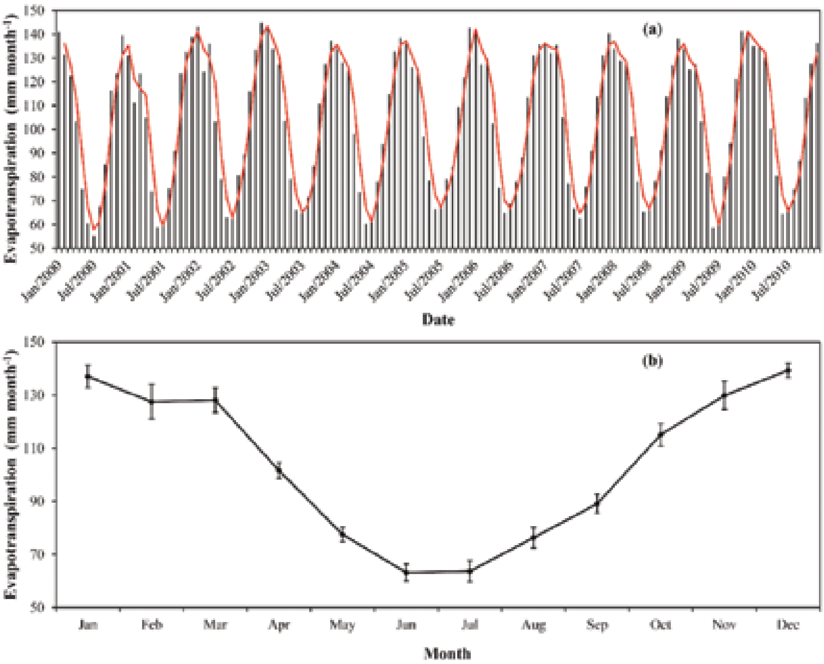

Figure 8. Monthly (mm month^{-1}) (a) and monthly average (mm month^{-1}) (b) actual evapotranspiration in Atlantic Rainforest remnants of São Paulo State between January 2000 and December 2010. In (a), the red line represents the moving average of the time series (period = 2), and in (b), vertical bars represent the standard deviation.

Monthly average evapotranspiration for the dry season was 78.6 mm month^{-1}, while during the rainy season was 129.5 mm month^{-1}. Considering these results, monthly average evapotranspiration in the Atlantic Rainforest remnants was ~65% higher in the rainy season when compared to the dry season. Annual evapotranspiration ranged from 1220.4 (2000) to 1275.2 (2002) mm year^{-1}, an absolute variation of 55 mm year^{-1} and relative variation of ~5%. Monthly evapotranspiration for 2000 and 2002 ranged, respectively, from 55.3 (July) to 140.7 mm month $^{-1}$ (January), and from 62.5 (July) to 144.6 mm month^{-1} (December). Annual average evapotranspiration was 1248.3 mm year^{-1}, with dry and rainy season month representing, respectively, ~38 and ~62% of the total. Considering annual average, evapotranspiration represented ~72% of the precipitation in Atlantic Rainforest remnants, suggesting a low hydric production (~28%). Usually, evapotranspiration studies in tropical forests show values ranging, on average, from 1000 to 1400 mm year^{-1} [45]. Regarding Atlantic Rainforest in São Paulo State, [41] found annual average of 697.5 mm year^{-1} for evapotranspiration, ~44% lower than the result found in this study.

Figure 9 shows the continuous wavelet transform power spectrum for normalized time series of precipitation and evapotranspiration. In general, it is possible to observe that the main oscillation mode in precipitation and evapotranspiration of Atlantic Rainforest remnants time series is concentrated between 8 and 16 months, showing, as previously mentioned, a strong seasonal or intraannual behavior.

For precipitation, maximum energy peak was observed between 10 and 14 months (seasonal mark), occurring between March 2001 and November 2009. In addition, less intense peaks of energy are highlighted for a period of 1.5 month (January 2003 and February 2003), 1–3 months (January 2005 to April 2005 and June 2009 to September 2009), and 5–7 months (January 2009 to December 2009), being the last period not statistically significant considering a 95% confidence interval. These less intense peaks of energy of 1.5 month, 1–3 months, and 5–7 months are related to high precipitation episodes, mainly in January 2003 (383.2 mm month^{-1}), in January 2005 (370.2 mm month^{-1}), and February, July and September 2009 (281.4, 242.5, and 223.3 mm month^{-1}, respectively). Yet, as previously mentioned, 2009 presented the highest values for annual precipitation. Therefore, these high values of precipitation could be related to the occurrence of frontal systems (cold fronts), the SACZ, and South American Low Level Jet (SALLJ). Cold fronts are very common in São Paulo State and cause intense and isolated rainfall in different regions of the state [32]. SACZ and SALLJ exert an important control in the frequency of extreme precipitation events in Southeastern Brazil, acting in intraseasonal and interannual scales [35]. It is important mentioning that the El Niño event contributes to the action of SACZ in São Paulo State, increasing the probability of intense rainfall in the state during the years that the phenomenon occurs [46].

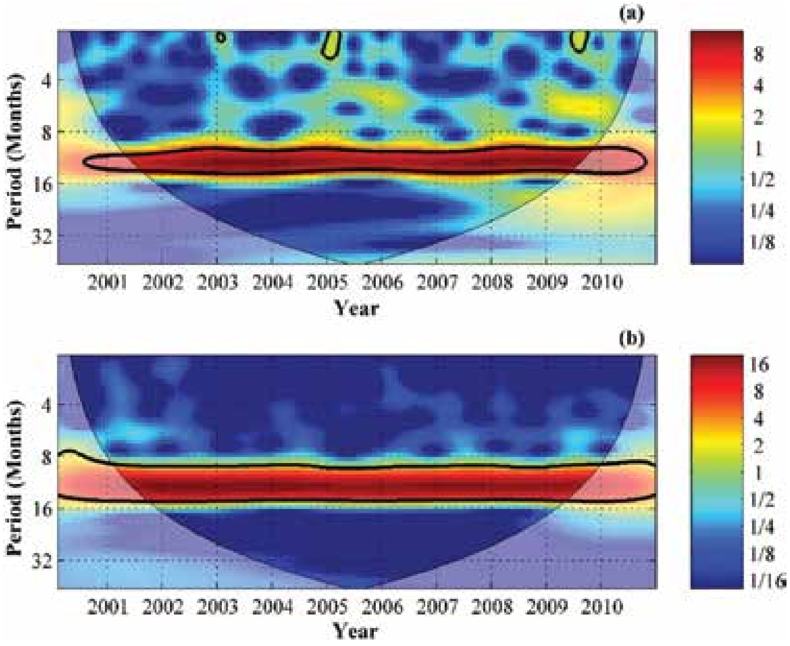

Figure 9. Continuous wavelet transform power spectrums for normalized time series of precipitation (a) and evapotranspiration (b). U-shaped curve represents the cone of influence, below which edge effects are important.

Regarding evapotranspiration, the maximum peak of energy is identified in the 9–15-month period (seasonal mark), occurring between January 2001 and December 2009. It is possible to observe less intense peaks of energy in the period of 5–7 months (January 2001 to March 2002, and December 2009 to March 2010); however, they are not statistically significant. Note that 2002 and 2009 showed the highest values of annual evapotranspiration. Therefore, the action of atmospheric systems, and their influence over meteorological variables (e.g., air temperature, wind speed, and air and soil moisture) [47], could have provided conditions that favored the increase of evapotranspirative processes in Atlantic Rainforest remnants considering the intraseasonal scale between 2001/2002 and 2009/2010 periods.

4. Conclusions

Combining TRMM satellite data and MOD16 algorithm enabled mapping the spatial distribution and evaluating precipitation and evapotranspiration in São Paulo State, as well as analyzing the temporal dynamics of these variables in Atlantic Rainforest remnants for the period between 2000 and 2010. Generally, the precipitation and evapotranspiration trends (considering both São Paulo State and forest remnants) revealed a strong seasonal pattern, with highest values concentrated in the rainy season (October to March) and lowest values in the dry season (April to September).

Regarding to São Paulo State, highest values of precipitation and evapotranspiration were found in southern and eastern regions, while lowest values were located in the northern and western portions of the state. The time series analysis showed that monthly averages for precipitation and evapotranspiration were, respectively, ~200 and ~84% higher during the rainy season when compared to the dry season. Considering annual averages, evapotranspiration corresponded to ~53% of precipitation in São Paulo State.

In regard to Atlantic Rainforest remnants, time series analysis showed that during the rainy season precipitation and evapotranspiration were, respectively, ~138 and ~65% higher than those observed during the dry season. In terms of annual averages, evapotranspiration accounted for ~72% of precipitation, indicating a low hydric production (~28%). Considering the entire period, monthly averages of precipitation and evapotranspiration were, respectively, ~12 and ~52% higher than the monthly averages for São Paulo State, which demonstrates the contribution of these remnants to the regional hydrologic regime. The higher amounts of precipitation are observed in the coastal region where most of the remnants are located and maybe there is an artifact, however if we analyze the evapotranspiration maps it is possible to note that the evapotranspiration is really low in the western part of Sao Paulo state due the presence of large areas of agriculture and pasture and a reduced number of forest remnants. Analysis of wavelet transform for precipitation and evapotranspiration time series in Atlantic Forest remnants showed that the main oscillation mode is concentrated between 8 and 16 months, revealing a seasonal or intra-annual behavior. It is important to note that the wavelets analysis allowed to conduct a more comprehensive evaluation of the behavior of precipitation and evapotranspiration through

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time. It can be a useful tool to verify trends of temporal shifts in environmental parameters [48–53], which in its turn can affect the ecosystem services delivered by tropical forest remnants. In this sense, just to point out, the trends of temporal shifts in precipitation and evapotranspiration observed in our study were related, in general, to the SACZ, SALLJ, and El Niño.

Results found in this study demonstrated that the use of remote sensing was an important tool for analyzing hydrological variables in Atlantic Rainforest remnants, which can contribute to better understanding the interaction between tropical forests and the atmosphere, and for generating input data necessary for surface models coupled to atmospheric general circulation models. Accordingly, future studies should be performed to (i) validate MOD16 algorithm for Atlantic Rainforest conditions, (ii) analyze potential artifacts related to the spatial distribution of the land cover and environmental parameters, (iii) identify other phenomena that could be related to intraseasonal and interannual variations in precipitation and evapotranspiration occurred in Atlantic Rainforest remnants, (iv) analyze precipitation and evapotranspiration in specific forest formations of Atlantic Rainforest (e.g., dense ombrophylous forest, mixed ombrophylous forest, and seasonal semideciduous forest), (v) evaluate the differences of precipitation and evapotranspiration between forest remnants and different land use types (e.g., pasture, agriculture, urban areas, etc.), which can play an important role to understand more specifically what is the impact of land use changes in ecosystem services in tropical regions, and (vi) analyze the relationship between precipitation and evapotranspiration in Atlantic Rainforest remnants using other biophysical variables, such as surface albedo and vegetation indices.

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Chapter 6

Ecological and Environmental Aspects of Nutrient Cycling in the Atlantic Forest, Brazil

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Additional information is available at the end of the chapter

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Abstract

In developing countries, where population growth is on the rise, intense anthropogenic actions in natural forests are observed usually in the form of burnings and shallow cuts. With aiming to deploy crops or even promote irrational exploitation of forest products. In this context, preservation of natural forests (tropical and subtropical forests) depends on the knowledge of their dynamics. This information is important to allow exploration of natural forests sustainably or to subsidize conservation actions. In planted and native forests, the biogeochemical cycling of nutrients predominantly occurs through production and decomposition of litterfall. The information provided in this chapter, in particular with regard to nutrient cycling, is an important basis for understanding the structure and dynamics of nutrients in the ecosystem. We characterized nutrient stocks and elucidate some aspects of forest growth and productivity. This information is important to enhance biodiversity conservation and generate ecosystem goods and services in the Atlantic Forest Biome. Even with the intense change of land use (from forest to agricultural, pasture and urbanization), the region has high diversity of endemic species, and is considered a priority area for biodiversity conservation.

Keywords: nutrient transfer, forest soils, biogeochemical cycling, tropical forests

1. Introduction

The Atlantic Forest biome is currently at an advanced change process from its original and primitive form, due to intensive occupation and exploitation over the past five centuries. The devastation of the Atlantic Forest, at large, has been attributed to intensive use of timber species of interest (mainly *Caesalpinia echinata*, popularly known as Brazilwood), and the establishment of areas for agriculture, pasture and urbanization. The advancement and establishment of agricultural areas and, consequently, fallen forests have reduced native forest massifs to fragmented forests, which has greatly compromised biological diversity and conservation of these forest ecotypes [1]. Even with the intense land-use change, with only 12.5% of the original cover remaining (only fragments larger than 3.0 ha), the Atlantic Forest currently shows more than 15,000 plant species and more than 2000 species of vertebrate animals [2]. The biome has high diversity of endemic species, and is considered a priority area for conservation (hotspots). In it, 383 species of animals threatened with extinction are found [2].

Studies on native forests are of vital importance for a better understanding of the behavior of intrinsic characteristics to the ecosystem and must be performed before these ecosystems have all their original area changed by men [3]. The understanding of intrinsic characteristics aids to adopt proper programs for the recovery of degraded ecosystems. Therefore, a significant part of the areas that were changed due to changes in land use can be recovered. They can present again the ecological interactions necessary to ensure the biodiversity of fauna and flora. The recovery of ecosystems as a strategy to reverse the degradation process and enhance biodiversity conservation and provide ecosystem goods and services is already being implemented [4].

Mainly in tropical and subtropical regions, it is of utmost importance to have further information concerning the dynamics of nutrients in different compartments of a forest ecosystem. It is important in order to employ silvicultural practices to effectively ensure sustainable long-term management of altered ecosystem by land-use change. Nutrient cycling occurs naturally, in part, by the throughfall of tree canopies and trunks by rainfall and through the deposition of senescent tissues (litter) and after their decomposition [5]. This process, nutrient cycling (plant-soil-plant), enables the development of forests in soils with low nutritional levels [6]. The organic material that accumulates under the forest works as a big sponge able to retain water, reduce evaporation and sudden variations of soil temperature, thus preventing erosion, improving soil structure and promoting the cycling of nutrients [7].

In addition to these benefits, the understanding of nutrient cycling through litterfall in forests is one of the key aspects to be studied for planning the use of tree species to recover degraded areas or for timber production [7]. The content of nutrients supplied to the forest soil can influence production capacity as well as the potential of environmental recovery, because the nutrients resulting from organic material cause changes to the chemical and physical characteristics of the soil [3].

In this chapter, we will present some information about the nutrients cycling in the Atlantic Forest biome, the most important biome in socio-economic terms of Brazil. We will show the current status and characterization of existing forest types in the biome, description of nutrient cycles and factors affecting cycling in forests and indication and analysis of results of studies carried out throughout the biome and the potential of practical use of the data in areas with land-use change.

2. Atlantic Forest biome

The Atlantic Forest biome consists of forest formations [Dense Ombrophilous Forest, Mixed Ombrophilous Forest (also known as Araucaria Forest), Open Ombrophilous Forest, Semideciduous Seasonal Forest, Deciduous Seasonal Forest and Evergreen Seasonal Forest] and pioneer formations, such as Sandbanks, Mangroves and Grassland [8]. The biome represents 13.04% of the Brazilian territory of which only 22% are in native vegetation at different regeneration stages [9].

The significant biodiversity of the Atlantic Forest biome is related to geographical variations in this region. Longitude, latitude and altitude affect the climatic variables, forming regions with distinct characteristics, increasing species diversity. The area of the Brazilian Atlantic Forest covers a large latitudinal extent (from 3°S to 30°S) and longitudinal (approximately 17°) and significant altitudinal variations (from sea level to altitudes above 2700 m in the Mantiqueira Hills) [10, 11] (**Figure 1**).

Figure 1. Distribution of Atlantic Forest Biome in Brazil. Adapted from Ref. [9].

The main forest types found in the Atlantic Forest biome are classified according to the floristic composition and environmental variables, such as precipitation and temperature. In the following section, we show some features of the main forest formation in the Atlantic Forest according to Veloso [12] and the Brazilian Institute of Geography and Statistics [13, 14].

The Ombrophilous Forest is classified as Dense, Open and Mixed formation. Dense Ombrophilous forest is characterized by the presence of medium and large trees, in addition to lianas and epiphytes in abundance, due to the constant moisture from the ocean. The coastline extends from the Northeast to the extreme South of Brazil. Its occurrence is connected to hot and humid tropical climate without dry season, with rainfall well distributed throughout the year (eventually there may occur in some regions dry periods until 60 days) and average temperature is 25°C. In Open Ombrophilous Forest, we find arboreal vegetation more sparse and with lower shrubby density. It occupies areas with climatic gradients ranging between two and four dry months. Average temperatures range between 24°C and 25°C. Finally, Mixed Ombrophilous Forest is strongly characterized by the predominance in the upper stratum of *Araucaria angustifolia* and genera of the family Lauraceae (e.g., *Ocotea* and *Nectandra*). It consists of 2776 forest species, and 946 are endemic [10]. The physiognomy occurs in areas of wet climate and without water deficit. The average annual temperature is around 18°C. The Dense and Open Ombrophilous Forests had most forest species (9661) as well as most endemic species (5164) [10].

Seasonal Forest is classified as Deciduous, Semideciduous and Evergreen. For the first, Deciduous Seasonal Forest, it is characterized by a large number of deciduous trees, accounting for more than 50% of individuals of the forest component. It consists of 165 endemic forest species of the total of 1113 found in the forest typology [10]. In the tropical region, its occurrence is conditioned to a long dry period (more than seven months). In the subtropical region, however, this forest formation occurs in areas with long cold periods, for more than five months with average temperatures below 15°C. On the other hand, Semideciduous Seasonal Forest is composed of deciduous trees, which represent 20–50% of individuals of the forest component. It has the second largest number of forest species (3841) of the Atlantic Forest of which 1081 are endemic [10]. Their occurrence in the tropical region is defined by two well-defined pluviometric periods, one dry and one rainy with average annual temperature around 21°C. However, in the subtropical region, this formation occurs in a short dry period followed by a sharp drop in temperature, with averages below 15°C in the cold period. The last type is the Evergreen Seasonal Forest, which is composed of deciduous trees, which account for less than 20% of individuals of the forest component. This forest occurs under tropical climate with a rainy and dry season, with about four to six months of dry weather. Still, the arboreal component does not seem to undergo water stress, which causes low leaf shedding.

Currently, approximately 7% of the biome natural areas are well preserved in fragments larger than 100 ha [15]. The biome consists of about 20,000 plant species of which 8000 (i.e., 40%) are endemic [16]. The analysis of species distribution in the different forest formations [10] showed that more than half of the wealth (60%) and most endemics (80%) are found in the Atlantic Forest. Due to their high levels of richness and endemism, the Atlantic Forest is among the top five hotspots in the world [16].

This region is of great importance for Brazil, because more than half of the national population is spread across the Atlantic Forest biome and this region accounts for much of the economic activity in the country. In addition, water resources that serve about 70% of the Brazilian population are located in this biome [17]. However, with the intense land-use change and the consequent fragmentation of this biome, biodiversity loss is noticeable and there is an eminent need for conservation. Due to the importance of this vegetation component, law n. 11,428 was enacted in 2006 [8] to regulate the use of native plants in the Atlantic Forest biome.

3. Nutrient cycling in forests

Biomass production in a forest ecosystem is conditioned to several factors, namely light, water, CO_2 concentration, chlorophyll content, temperature, nutrients, genetic adaptation and competition, among others [18, 19]. Among these factors, nutrients stand out as an essential element for the primary productivity of the forest ecosystem [20]. Nutrient cycling in forests is defined as the transfer of elements between the different components of the ecosystem. This transfer is controlled by climate, site, abiotic factors (topography, source material) and biotic agents [21]. Therefore, nutrient cycling in tropical forests is distinct from that in temperate zones. For example, the amount of nutrients on the forest floor and the length of deposition are shorter in tropical forests than in boreal forests, due to slow decomposition in regions of cold climate and high altitudes [21].

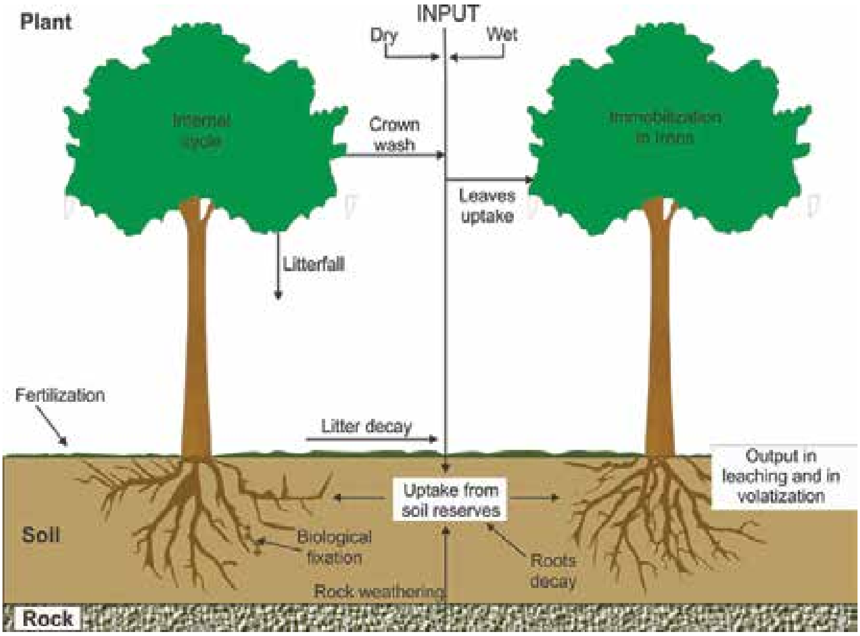

Figure 2. Scheme of nutrient cycling dynamics in a forest. Adapted from Refs. [24, 25].

Nutrient cycling in forests can be generalized into three models: geochemical, biogeochemical and biochemical cycling [22]. Geochemical cycling is characterized by the input and output of nutrients in the ecosystem. Atmospheric deposition (wet and dry), fertilization, biological fixation and rocks weathering are responsible for most nutrients input [23]. While, leaching, volatilization and harvest biomass are responsible for most nutrients output [24]. The biogeochemical cycle is characterized by the transfer of nutrients between the plant and the soil. In this cycle, plants absorb nutrients form soil reserves and then return them to the soil via litterfall (litter decay), roots decay or plant death [24]. Biochemical cycling is the translocation of nutrients inside the plant (internal cycle). Once soil nutrients are absorbed, some of these elements are in constant mobilization within the plant, mostly from older to younger tissues.

The dynamic process of nutrient cycling in native or exotic forest ecosystems is shown in **Figure 2**.

4. Nutrient cycling in the Atlantic Forest

The biogeochemical cycling is one of the most studied nutrient cycles in the Atlantic Forest, mainly in terms of deposition, accumulation and decomposition of litterfall. This litter is composed predominantly of leaves, branches, bark, trunks of fallen trees, flowers, fruit, dead animals, etc. In general, the percentage of leaves in relation to the other litter components ranges from 60% to 80% of the total material. The biomass of senescent leaves that fall onto the forest floor represents part of net primary production (NPP) of vegetation [26, 27].

Most nutrients uptaken by the trees return to the soil through senescence of their organic components. The intensity of nutrient cycling depends mainly of the deposition of organic material. It is considered the most important form of nutrient transfer from the plant to the forest soil in the ecosystem [28]. According to Viera and Schumacher [28], there is variation between species regarding the amount of nutrients retained and returned. For them, there are species that retain most nutrients absorbed, while others return most nutrients absorbed, and there are also those in which retention is equal to return. This retention and return ratio is linked to different translocation rates of species [29], age, soil and climate conditions [3], as well as environmental aspects, varying from species to species [5].

The continuous supply of litterfall enables storage of soil organic carbon (SOC) and nutrient availability. These nutrients, after litter decomposition, help to keep soil fertility in native forests [30, 31]. Litter provides nutrients, energy and matter to microorganisms in the soil and roots, which is important in tropical forests where litterfall is intense and decomposition is faster [30, 32] than in temperate forests. Litterfall is responsible for important environmental services. It helps intercept rainfall and its storage in the soil increases infiltration rate and surface flow conditioning of water and soil [33], thus avoiding the beginning of erosion processes.

In the Atlantic Forest, due to the different types of forest formations, we can observe a diversity of environments, where each one offers a distinct pattern of litter deposition and accumulation

(**Table 1**). For example, seasonal forests have a seasonal deposition pattern due to a period of lower precipitation and low temperatures, triggering leaf abscission. The amount of litter is also influenced by the replacement of mature, older and less efficient foliar tissue by new leaves [27, 34, 35].

Forest type	Succession	Deposition	Accumulation	Reference
		(Mg ha^{-1})		
Dense Ombrophilous	Primary	7.4	7.3	[36]
Dense Ombrophilous	Secondary	5.6	–	[37]
Dense Ombrophilous	Secondary	–	8.6	[38]
Dense Ombrophilous	Early[1]	–	4.5	[39]
	Intemediate[2]	–	5.0	
	Advanced[3]	–	5.2	
Dense Ombrophilous	Early[1]	5.2	–	[40]
	Intermediate[2]	5.4	–	
	Advanced[3]	5.3	–	
Dense Ombrophilous	Secondary	9.8	–	[41]
Dense Ombrophilous	Secondary	10.0	–	[42]
Dense Ombrophilous	Secondary	4.7	–	[35]
Mixed Ombrophilous	Primary	6.0	–	[43]
Mixed Ombrophilous	Secondary	6.3	–	[44]
Mixed Ombrophilous	Primary	10.3	14.3	[34]
Mixed Ombrophilous	Secondary	–	8.0	[45]
Semideciduous Seasonal	Secondary	–	5.5	[46]
Semideciduous Seasonal	Secondary	9.3	–	[47]
Semideciduous Seasonal	Primary	8.2	–	[48]
Semideciduous Seasonal	Secondary	11.7	–	[49]
Deciduous Seasonal	Secondary	5.9	–	[50]
Deciduous Seasonal	Secondary	–	8.0	[51]

Note: Secondary forest in early (1), intermediate (2) and advanced (3) stages of succession.

Table 1. Annual deposition and accumulation of litterfall in the soil in different forest types in the Brazilian Atlantic Forest.

In tropical forests, such as the Atlantic Forest, litterfall deposition is influenced by latitude and altitude. According to Alves et al. [52], the vegetation structure can vary greatly according to the altitude, since lower altitudinal gradients can present significant changes in edaphic conditions, due to topographic and climate variations. Thus, species that grow in environments with adequate light, water and nutrient availability have high productivity compared to those that develop in environments with low availability of these resources. For example, Montane Forests are less productive than Lowland Forests, since temperature reduction, increased cloudiness, lower reserves of nutrients in the soil and water saturation of the soil are factors that limit the NPP in Montane Forests [26, 53]. In addition, the Atlantic Forest located at higher altitudes is more susceptible to the action of winds, more intense thermal inversions and greater terrain slope. All these aspects, along with its solar orientation, can increase or reduce incident radiation that will affect the phytosociological structure and composition of the forest.

The different types of the Atlantic Forest biome feature a distinct nutrient transfer via litter deposition. This may be linked to the different developmental stages of the forest. In each stage, the vegetation displays distinct control forms of nutrient demands through storage and redistribution in biomass [54] (**Table 2**).

Forest type	**Succession**	**N**	**P**	**K**	**Ca**	**Mg**	**S**	**Reference**
				kg ha^{-1} year^{-1}				
Semideciduous Seasonal	Secondary	150.3	7.3	45.2	291.5	30.5	10.7	[55]
Semideciduous Seasonal	Secondary	172.2	8.9	67.7	216.9	27.3	13.6	[47]
Semideciduous Seasonal	Primary	294.2	3.2	108.3	462.2	33.9	–	[48]
Semideciduous Seasonal	–	217.8	11.6	52.8	199.8	38.7	–	[56]
Deciduous Seasonal	Secondary	123.2	5.1	26.4	131.6	15.6	7.1	[50]
Dense Ombrophilous	Secondary	–	5.0	49.7	170.7	26.4	–	[42]
Dense Ombrophilous	Secondary	123.7	14.4	4.9	–	–	–	[57]

Table 2. Nutrients transferred to the soil annually via litter deposition in different forest types in the Brazilian Atlantic Forest.

Under similar climate and soil conditions, variation in litter accumulation occurs by both the amount and the composition (contents of lignin, polyphenols and nutrients) of the material deposited, influencing decomposition speed and nutrient release [58]. In general, N and Ca are the nutrients that are most accumulated on the soil in the Atlantic Forest (**Table 3**). In forests established in weathered soils, accumulated litterfall ensures nutrient cycling. This litter, along with the soil, regulates many fundamental processes in the dynamics of ecosystems, such as primary production and nutrient release [59].

The amount of nutrients in litter deposed or accumulated varies according to the forest type and edafoclimatic conditions. Abiotic and biotic factors affect litter production, namely the vegetation type, altitude, latitude, rainfall, temperature, light incidence, relief, water availability and soil characteristics [60]. Likewise, nutrient concentration and content in this litter vary according to the soil type, vegetation, population density, the ability of species to absorb, use and translocate nutrients before leaf senescence, as well as the percentage of leaves in relation to other components of the natural habitat (soil and climate conditions) and the tree age [29, 61].

Forest type	**Succession**	**N**	**P**	**K**	**Ca**	**Mg**	**S**	**Reference**
		kg ha^{-1} year^{-1}						
Semideciduous Seasonal	Secondary	105.9	4.4	12.9	249.1	16.5	7.1	[55]
Semideciduous Seasonal	Secondary	94.9	4.1	14.0	161.0	12.1	7.4	[46]
Dense Ombrophilous	Secondary	218.0	3.4	8.5	61.0	14.9	–	[38]
Dense Ombrophilous	Early^1	67.5	2.6	11.8	40.2	12.9	7.1	[39]
	Intermediate^2	73.1	2.8	11.7	60.9	13.1	7.3	
	Advanced^3	88.8	2.8	9.0	41.2	13.9	9.4	
Mixed Ombrophilous	Secondary	95.7	5.4	45.3	36.8	7.6	14.8	[45]

Note: Secondary forest in early (1), intermediate (2) and advanced (3) stages of succession.

Table 3. Nutrients stored in accumulated litter on the soil in different forest types in the Brazilian Atlantic Forest.

The availability of nutrients in the accumulated litterfall occurs during decomposition. Decomposition is controlled by the nature of the scavenging community (animals and microorganisms), by the organic matter characteristics, which determines its degradability (quality) and by the physical-chemical aspects of the environment, which operates in the edaphic or microscale conditions [62].

Similar to litter decomposition, the rate at which nutrients are released depends on the chemical composition of the litter, the structural nature of the nutrient in the litter and the availability of external nutrient sources [63]. The release of nutrients in the litter depends on its quality, on macro- and micro-climatic variables and on biotic activities. The climate factors that influence litter decomposition the most are temperature and soil moisture [63]. According to the authors, another primordial factor responsible for higher or lower decomposition rate is the structural composition of tissues because tissues that contain higher contents of cellulose, hemicellulose and lignin are more resistant to decomposition than tissues with lower contents of these compounds.

5. Final remarks

The lessons learned with landscape change in the Atlantic Forest, especially during the last few decades, indicate the need to develop programs of environmental conservation and restoration. Environmental education and scientific research are also important to allow a sustainable management of world forests. Therefore, knowing the different factors that influence the development and maintenance of a natural forest ecosystem is necessary to prevent fragmentation of new forest areas.

Nutrient cycling is one of the fundamental processes in the functioning of forests. It helps to understand the great complexity of relationships and flows between different compartments of nutrients and carbon to manage forest ecosystems sustainably. This means that mechanisms in this ecosystem have not been thoroughly understood, hindering the proper management of this resource. Therefore, there is the need to understand the nutrient cyclic processes in different forest ecosystems, as identified for the Atlantic Forest, where the amount of nutrients in litter deposed or accumulated varies according to the forest type and edafoclimatic conditions. Understanding these characteristics aids to adopt programs for the recovery of fragmented and degraded ecosystems specific for each forest type.

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Large regions of the planet have been transformed from their natural composition into different human-made landscapes (farmlands, forest plantations, pastures, etc.). Such process, called land use change, is one of the major components of the current global change, which has brought the planet into a new geological era: the Anthropocene. Land use change is particularly important in tropical forests, as this ecosystem type is still heavily affected by deforestation for timber extraction, agricultural land creation of urban expansion. Changing land use has important implications for the services that tropical forests provide: production of goods such as timber, food or water; regulation of process such as nutrient cycling, carbon sequestration, local weather or climate extremes; generating the framework for economic and cultural activity, etc. Therefore, keeping ecosystem services when changing the use of the tropical lands is a major challenge in tropical regions. This brief book, by showcasing different research work done in tropical countries, provides a first introduction on this topic, discussing issues such as biodiversity loss, changes in local weather or nutrient cycling patterns, and economic activities around tropical forests, and tools to detect and quantify the importance of land use change.

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