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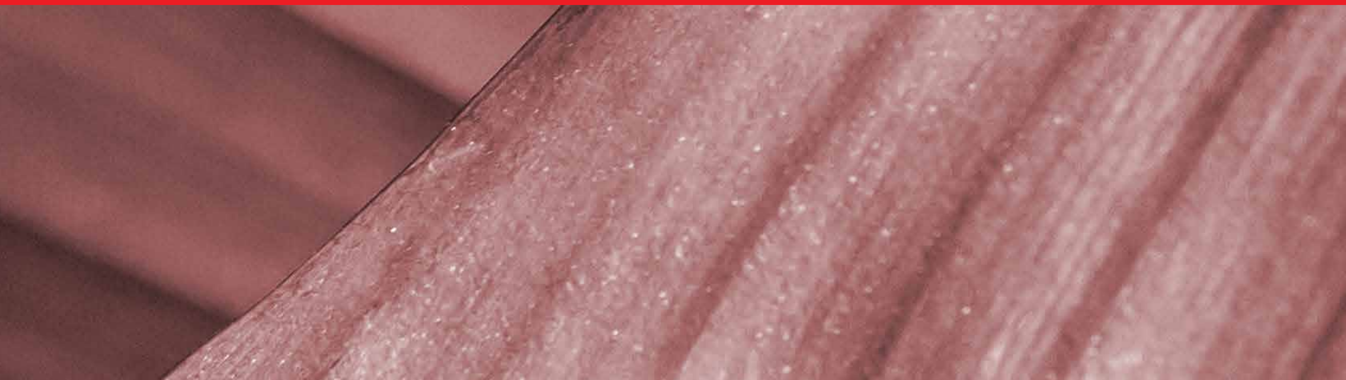
Forest Degradation Under Global Change

Edited by Pavel Samec



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IntechOpen Book Series

Sustainable Development

Volume 3

Aims and Scope of the Series

Transforming our World: the 2030 Agenda for Sustainable Development endorsed by United Nations and 193 Member States, came into effect on Jan 1, 2016, to guide decision making and actions to the year 2030 and beyond. Central to this Agenda are 17 Goals, 169 associated targets and over 230 indicators that are reviewed annually. The vision envisaged in the implementation of the SDGs is centered on the five Ps: People, Planet, Prosperity, Peace and Partnership. This call for renewed focused efforts ensure we have a safe and healthy planet for current and future generations.

This Series focuses on covering research and applied research involving the five Ps through the following topics:

1. Sustainable Economy and Fair Society that relates to SDG 1 on No Poverty, SDG 2 on Zero Hunger, SDG 8 on Decent Work and Economic Growth, SDG 10 on Reduced Inequalities, SDG 12 on Responsible Consumption and Production, and SDG 17 Partnership for the Goals

2. Health and Wellbeing focusing on SDG 3 on Good Health and Wellbeing and SDG 6 on Clean Water and Sanitation

3. Inclusivity and Social Equality involving SDG 4 on Quality Education, SDG 5 on Gender Equality, and SDG 16 on Peace, Justice and Strong Institutions

4. Climate Change and Environmental Sustainability comprising SDG 13 on Climate Action, SDG 14 on Life Below Water, and SDG 15 on Life on Land

5. Urban Planning and Environmental Management embracing SDG 7 on Affordable Clean Energy, SDG 9 on Industry, Innovation and Infrastructure, and SDG 11 on Sustainable Cities and Communities.

The series also seeks to support the use of cross cutting SDGs, as many of the goals listed above, targets and indicators are all interconnected to impact our lives and the decisions we make on a daily basis, making them impossible to tie to a single topic.

Meet the Series Editor

Usha Iyer-Raniga is a professor in the School of Property and Construction Management at RMIT University. Usha co-leads the One Planet Network's Sustainable Buildings and Construction Programme (SBC), a United Nations 10 Year Framework of Programmes on Sustainable Consumption and Production (UN 10FYP SCP) aligned with Sustainable Development Goal 12. The work also directly impacts SDG 11 on Sustainable Cities and Communities. She completed her undergraduate degree as an architect before obtaining her Masters degree from Canada and her Doctorate in Australia. Usha has been a keynote speaker as well as an invited speaker at national and international conferences, seminars and workshops. Her teaching experience includes teaching in Asian countries. She has advised Austrade, APEC, national, state and local governments. She serves as a reviewer and a member of the scientific committee for national and international refereed journals and refereed conferences. She is on the editorial board for refereed journals and has worked on Special Issues. Usha has served and continues to serve on the Boards of several not-for-profit organisations and she has also served as panel judge for a number of awards including the Premiers Sustainability Award in Victoria and the International Green Gown Awards. Usha has published over 100 publications, including research and consulting reports. Her publications cover a wide range of scientific and technical research publications that include edited books, book chapters, refereed journals, refereed conference papers and reports for local, state and federal government clients. She has also produced podcasts for various organisations and participated in media interviews. She has received state, national and international funding worth over USD $25 million. Usha has been awarded the Quarterly Franklin Membership by London Journals Press (UK). Her biography has been included in the Marquis Who's Who in the World® 2018, 2016 (33rd Edition), along with approximately 55,000 of the most accomplished men and women from around the world, including luminaries as U.N. Secretary-General Ban Ki-moon. In 2017, Usha was awarded the Marquis Who's Who Lifetime Achiever Award.

Meet the Volume Editor

Ing. Pavel Samec, Ph.D., graduated from Mendel University of Agriculture and Forestry, Czech Republic. The author has long been involved in landscape ecology. During his studies, he dealt with the theory of evolutionary ecology and past climate changes, as well as with the restoration of natural soil formation processes in polluted areas. From 2007 to 2011, he used his previous experience in the development of models of forest growth conditions and nutrient balances for the Ministry of Agriculture of the Czech Republic. Dr. Samec is the main author of twelve scientific books and the author or co-author of more than fifty scientific papers. He currently works for Mendel University and the Global Change Research Institute of the Czech Academy of Sciences.

Contents

Preface

Human activities cause the degradation of innumerable ecosystems. In particular, forest degradation leads to far-reaching nutrient cycle changes throughout all environments. Reversible and irreversible degradation processes cause ecosystem self-regulation to decline. While reversible processes tend to be a structural simplification and a decrease in native forest biodiversity under the preservation of the ecosystem's abiotic components, irreversible processes alter the character of the potential natural vegetation due to climate change, acidification, eutrophication, and soil erosion. Subsequent secondary ecosystems already affect global change processes in an uncertain way. The book discusses the irreversible degradation processes of forests worldwide.

This book is organized into three parts on general, climatic, and land-use degradative effects. Global environmental change involves a related sequence of the biophysical, ecosystem, and socioeconomic alterations that damage the life-sustaining abilities of the planet. Current global change is caused by the human transformation of the natural environment, but also by the reaction of human communities to the induced modifications. Human transformations of the natural environment are concentrated in a critical zone. The most vulnerable part of the terrestrial critical zone is composed of soil (pedosphere), which includes interfaces among the atmosphere, hydrosphere, lithosphere, and biosphere.

The most serious soil damage is due to land-use modifications after deforestation. While mere forest felling only gives rise to reversible changes in ecosystem functions, the combination of forest felling with soil erosion or conversion for the need of subsequent management utilization generates irreversible ecosystem degradation. Deforestation is instantly followed by declining evaporation and soil loss. The imminent consequences of deforestation are gradually leading to regional climate change, the loss of ecosystem recoverability, and uninhabitable landscapes. Ultimately, landscape uninhabitability is caused by uncontrollable soil erosion because of surface exposure to wind and landslides of weathered rocks impoverished from organic binders. Thus, this book discusses characteristics of human activities that disrupt the landscape and highlights the need to restore wildlife.

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Section 1

General Forest Degradation

Chapter 1

Perspective Chapter: Forest Degradation under Global Climate Change

Sandeep Sasidharan and Sankaran Kavileveettil

Abstract

Forests cover nearly one-third of the terrestrial surface and support life with energy, raw materials, and food and offer a range of services ranging from biodiversity conservation to climate regulation. The realization of this goods and services depends on the health of these pristine ecosystems. Forest degradation diminishes the utilitarian and ecosystem potentials of the forest and assessing this at local and global scales is draught with complexities and challenges. Recently, climate change has been identified as a major factor of forest degradation across the globe. Although native forests may be adapted to disturbances to a critical threshold level, the intensification of the stress will move the forests in a new trajectory. Evaluating the cause-effect relationship of forests and climate also play determinable roles in the forest-climate loop. Such analysis is critical in identifying the factors of degradation and would be crucial in developing strategies for restoring and conserving the forest ecosystems.

Keywords: forest degradation, climate tolerance, biophysical responses, plasticity responses, biological invasions

1. Introduction

Forests cover thirty-one percent of the total global land area and nearly half of these forests are relatively intact with more than one-third remaining as naturally regenerated forests of native species (primary forests), with no visible indications of disturbances. The total area under forests across the globe is estimated to be 4.06 billion hectares, and this ecosystem provides habitat for a vast variety of terrestrial flora and faunal species [1]. Besides biodiversity, forests provide a plethora of ecosystem services ranging from sociocultural benefits, to nature experiences and climate regulation. Unfortunately, these pristine natural resources are under tremendous threat both from natural and anthropogenic stresses.

The conventional forest management targeted only a minor subset of the accrued benefits from forests and specifically concentrated on harnessing their potential for timber production or recreation [2, 3]. The structure and composition of a large extend of the world's present forest systems are a direct or indirect result of such manipulations. With time, there was an increasing realization that forests provide services much beyond their traditional uses and have to be viewed from multiple dimensions to realize their full potential. One of the most important among these benefits is that forests are the most viable option for combating global climate change.

Forests exert regulations on the global climate in at least two ways: (i) forest ecosystems remove approximately 3 Pg C annually emitted to the atmosphere by anthropogenic activities and (ii) act as the major terrestrial sink holding more than two times the carbon in the atmosphere [4]. Though the climate mitigation role of forests is beyond doubt, a complex situation emerges on a reverse evaluation of the influence of climate on forests and their concomitant effects on the carbon cycle. Hence, evaluating the cause-effect relationship of forests and climate solely on carbon stocks and sequestration potential may not be sufficient. Forest structure, cover, compositional changes, and biophysical shifts (water and energy balances) play determinable roles in the forest-climate loop. Forest fires, pest outbreaks, invasive alien species, pollution, forest fragmentation, and soil erosion are degradative factors that hamper the functions and productivity of forests, thereby adding to atmospheric carbon. In the present chapter, we intend to analyze (i) biophysical responses of forests to climate change and (ii) the climate-change-induced impacts on forests.

2. Defining forest degradation

The concept of health applied to individual trees would appear quite simple and easily quantifiable by measures such as absence of disease or physiological stress. However, shifts in the monitoring scales from trees to forest as a system make the assessment very complex. Stand productivity is commonly treated as a measure of the health of natural and planted forests. Though this provides a good proximate estimation for utilitarian purposes, it neglects several other important features of forest ecosystems such as vegetation structure, species assemblage, carbon storage, nutrient cycling, and hydrological functions. This deficiency necessitates a more holistic definition of forest health with easily quantifiable attributes. Though giant leaps have been made in technologies and scientific techniques in forestry studies, researchers have struggled for decades with operational definitions of ecosystem degradation. Lanly and Jean-Paul [5] state that "The situation with respect to forest degradation is unsatisfactory particularly because of the imprecision and multiple, and very often subjective, interpretations of the term." Lund [6] could identify more than 50 definitions for forest degradation showcasing the inconsistencies and vagueness associated with the concept.

Ghazoul et al. [7] derived a concept of forest degradation by combining the theories and analogies of resilience and basins of attraction. The proposed basin of attraction represented the different ecosystem states, which continuously modify toward a single or multiple stable steady states. It should be noted that the term steady state does not mean a static state per se, but would be highly dynamic and with interactions between the abiotic and biotic elements that would continuously produce small changes in structure and composition at local levels, even without any disturbance. Even slight disturbances, however, would displace the ecosystems from their present stable steady state to an unstable state and will initiate changes that would enable the system to either return to their earlier state or achieve another stable state. The level of displacement of the system depends on the type, intensity, scale, and frequency of disturbance [8]. The system's ability to return to the earlier stable state (i.e., resilience) would depend on the intensity, frequency, and novelty of the disturbance causing degradation.

A robust approach to assess forest health has been put forth by FAO (The Food and Agriculture Organization of the United Nations), which combines the perspectives of "forest health and vitality" by considering biotic (e.g., weeds, insects and pathogens) and abiotic (e.g., pollution and drought) stresses and their effects on tree growth, survival, wood yield, wildlife habitat, non-timber forest products, cultural, esthetic, and recreation values. As such, a healthy forest system should be a balanced ecosystem that sustains complexity while providing ecosystem services. Such healthy systems would be recalcitrant to change and exhibit an excellent ability to recover from natural and anthropogenic stressors. Declining forest health or degradation is a global issue, but there exist problems in making realistic assessments because of difficulties in fixing appropriate timescales, reference states, thresholds, and ecosystem values.

The importance of having a proper definition for forest degradation would be crucial in properly defining the type of disturbance (natural or anthropogenic) that has led to a diminished status of the system. Forests are very complex and dynamic systems that constantly shift their structure and composition. Disturbance regimes and their subsequent interactions change forest structure and composition and are usually more expressed at the local to regional scales [9]. However, disturbances by climate change usually have wider ramifications that reverberate across the entire planet. Paleontological records of charcoal and pollen have attributed the rises in fire frequency and resultant degradation in boreal and temperate forest systems to changes in human management and climate [9].

3. Biophysical responses of forests to climate change

There has been an increasing concern on the potential impacts of climate change on forests [10–15]. Due to the comparatively larger size and life span of forests with respect to most agricultural systems, the former systems respond relatively slowly to changes compared with agroecosystems. In agriculture there remains an option to breed climate-resilient varieties or to shift entirely to a new landscape within a few decades. However, such quick fix solutions may not be available for most forested systems.

3.1 Climate tolerance by forest systems

Tree species in the forests have life spans of more than 200 years and, in most cases, would be able to tolerate a reasonable range of weather fluctuations [16]. However, extreme climatic events such as floods, drought, and wind throws can result in widespread tree mortality and species decline. Tree ring data have been used extensively to quantify the degree to which the growth was suppressed before the tree survival gets affected and to establish the relationship between tree growth decline and climate [17]. Loehle [16] reported that trees in Pacific Northwest are aged between 400 and 500 years, hence would have germinated alongside the Little Ice Age (approximately 550 years ago) and survived a complete cycle of cooling and warming. Studies by Stahle and Cleaveland [18] on Taxodium distichum (bald cypress press) and Scuderi [19] in Pinus balfouriana (foxtail pine) in North America also show that over the life span (several hundred years) of many trees, multiple cycles of weather extremes would have been a rule rather than an exception, and it can be safely expected that trees, to a large extend, can tolerate such fluctuations.

3.2 Plasticity response by trees to climate change

Trees adopt a conservative growth strategy by adjusting their growth rates with weather. Frequent adverse weather conditions projected in most climate models may not always lead to complete tree mortality, but may rather try to balance with the developing average climatic conditions over a period. This means that trees with long life spans would adopt conservative strategies to avoid rapid growth in short runs of good conditions and thereby death in adverse conditions [16]. Such strategies, however, have been less reported in short-lived and early successional species.

Physiological and morphological acclimation responses by plants also help them to adapt to changing climate. Herbaceous plants have been known to alter their shoot-root ratio to acclimate to growing conditions. However, there are serious gaps in research data with respect to such responses in larger trees. Drought and high intensity rains eroding the fertile topsoil are two prominent effects predicted in most climate change assessments. Trees have been reported to reduce shoot-root ratio and energy demands to decrease stresses due to water deficiency during drought [20, 21]. Such size and biomass adjustments are usually developed at young ages. On the other hand, mature trees usually get locked into the already established biomass partitioning patterns that they would have developed at young age. Such developed functionalities restrict their ability to alter resource demands, hence more likely to succumb to rapidly changing climatic conditions [22]. Under persisting severe climate conditions, trees that die are usually replaced by the same species (i.e., no change in forest composition), but with a more adaptive body organization to tolerate the changing climate situations. However, such adaptations would affect aboveground productivity and would have serious ramifications for the functioning of ecosystem. Species replacement in forest systems would occur only if better adapted species to new and changing climate conditions are present in the regeneration pool and if they are competent enough to outpace the inferior species [15].

3.3 Genetic responses by trees to climate change

Tree species have physiological ecotypes, each with very distinct limits of tolerance and adaptation to variation in climate. In other words, within each species there would be a broad range of climate optima facilitating wider adaptations to changing climate. Bonan and Sirois [23] observed that the eco-physiological responses did not vary much under a wide range of temperatures.

3.4 Forest soils under climate change

Soil provides the base matrix for ecosystems to grow and flourish providing all essential growth inputs. The vegetation that develops on a soil in turn supports the base matrix by way of nourishing, protecting, and cycling the resources. A disruption in this cycle would eventually add to the disruptive forces accentuating degradation. The degradation of surface soils of forests by natural or anthropogenic stressors can accelerate soil nutrient leaching especially in tropical humid regions [24, 25]. Thick root mats on sandy soils in the Amazon tropical rainforest have been found to retain up to 99% of the available nutrients, thereby preventing nutrient leaching losses [26, 27]. In forests affected by fire, concentrations of most primary (P, K) and secondary nutrients (Ca, Mg and S) increase rapidly immediately after burning, which promote root development, chlorophyll content, and reproduction in surviving plants [28]. However, most of these nutrients are lost within a short time through leaching and runoff, thereby furthering degradation and supporting ecosystems with lesser diversity and ecological potentials [29–31]. The degradative effects will be faster in coarse texture soils than in clayey soils due to the higher negative charges in clay that strongly bind and retain the nutrients from leaching losses [32] Topography also has a profound influence on soil nutrients with chances of nutrient losses higher on the upper slopes under high-intensity rains [30].

Trees exert direct influence on the availability and quality of water in forest ecosystems [33–35] and play an important role in regulating soil erosion and runoff rates, which gets offset by degradation. Accordingly, climate factors that affect degradation and vegetative cover modification will have substantial impacts on the water-related provisioning services. Interaction of forest systems with water and energy cycles provides the basic foundation for water resource distribution, carbon storage, and terrestrial temperature balances. Studies by Rodrigues et al. [35] showed that climate change could lead to a 24–46% rise in annual soil erosion in managed forest systems and that the losses were consistently higher in systems with lower site qualities. The projected increments in the mean annual temperatures and annual rainfalls under the different climate change scenarios are expected to aggravate erosion risks in forest systems. Apart from high-intensity rains, increased fire incidences by enhanced drought conditions would also open up canopy leading to soil erosion, thereby furthering forest degradation.

Besides a store house of nutrients, forest soil also contains a complex microbial community with diverse metabolic capabilities that regulate all biogeochemical transformations in these systems. These biogeochemical cycles control the nutrient transformation and maintain fertility of soil [36, 37]. Though degradation causes significant decline in beneficial microbial population, studies have also shown that the diversity of certain adaptable microbial groups may increase in some forest systems [38, 39]. Theoretical models predict that soil fungal communities may be more resistant to forest degradation than bacteria in tropical forests [40, 41].

3.5 Fire responses of forests under climate change

Forests are highly vulnerable to wildfires and such fire-induced changes are considered an important part of the landscape dynamics. Wildfires are responsible for degradation in about 4.8 million hectares of forest worldwide, and this accounts for nearly 23% of forest degradation by all factors [42]. Low, moderate, and high-intensity fires occur under a wide set of environmental and climatic variables such as very low humidity, high wind speeds, high temperature, and high dry biomass (fuel) load, all of which are accentuated under changing climate. Once initiated the propagation of forest fire usually follows three mechanisms, which degrade the existing forest systems:

i. crawling fire: the fire that spreads through low-level vegetation;

ii. crown fire: the fire that spreads through the top of the forest at an incredible speed. They can be extremely dangerous particularly on windy days;

iii. jumping or spotting fire: the burning fuel (branches and leaves) is carried by wind and likely to cause distant fires. Thus the fire can jump over a road, river, or even a firebreak.

Forest fires, besides effecting the vegetation dynamics, cause prominent disturbances in the system and act as an agent of environmental change with local to regional impacts on land use, productivity, carrying capacity, biodiversity, and regional to global impacts on hydrological, biogeochemical, and atmospheric processes. About 55% of the forest systems in India are affected annually by fires, which result in degradation and exacerbation of carbon dioxide levels in the atmosphere. Besides flora, forest fires affect the forest habitat and population and distribution of short range faunal species. Tropical wild fires produce high organic carbon emissions, trace gases, black carbon, and release almost 100 million tonnes of smoke aerosols into the atmosphere. These submicrometer smoke aerosols emitted in large quantities to the atmosphere play a major role on the radiation balance of the earth-atmospheric system. Further, fires also destroy the organic matter, which would be needed to maintain an optimum level of humus in forest soils. Frequent fires may also decrease the growth of grasses, herbs, and shrubs, which may result in increased soil erosion.

3.6 Biological invasions in forests under climate change

Invasive alien species (IAS) negatively impact forest ecosystems by extirpating native species, altering ecosystem functions, changing species composition, reducing food and cover for wildlife, and posing threats to biodiversity [43, 44]. Invasive alien plants can affect water availability to native species, increase forest fire hazards, and affect productivity of natural and planted forests. Although a comparison of the number of IAS and their extent of invasion in tropical and temperate forests is impeded due to data deficiency from the former, available information indicates that the number of IAS is increasing rapidly in temperate forests, and this may increase as high latitude areas get warmer with climate change [45].

Climate change (changes in global temperatures, precipitation, fire regimes, and occurrence of climate extremes) is reported to be a major driver of biological invasions by increasing susceptibility of all ecosystems to IAS [46, 47]. Climate change may also increase chances of introduction of new alien species, promote their spread by altering historical biogeographic limits, and enhancing probability of establishment and colonization [48]. If unsuitability of climate had previously prevented a species from establishing in a new biome, a change in climate may help its establishment, invasion, and spread [46, 49].

Climate change is thus a major driver of forest degradation since it promotes invasion by alien species in these ecosystems and causes a multitude of negative impacts. The damages due to climate-change-induced biological invasions can be more severe in forests, which are already degraded and fragmented due to human interventions or other causes. The interactions between climate change, land-use change, forest disturbance, and invasive alien plants may facilitate evolution of novel communities (assemblages of native and alien species) in natural forests [50]. Such interactions can also create new pathways of invasion and enhance vector efficacy [46]. On the other hand, deforestation, which leads to a local rise in atmospheric temperature, light availability, and increased supply of soil nutrients, can promote invasion by alien plants affecting natural regeneration of native tree species in forest ecosystems [51]. In short, the consequences of climate change on invasion by alien species and their impacts on forests are complex.

Intensity and frequency of fire in forests are facilitated by highly inflammable invasive alien plants especially where climate change causes extreme drought and rise in atmospheric temperature [52]. A shift in fire regimes may result from such fire events. And, forest fires may help quick regeneration of alien plants compared with native species and may alter carbon cycling in these ecosystems. Climate change is predicted to intensify the fire regimes in the future, which will help spread and establishment of invasive alien plants [53].

It is reported that warmer and extended autumns promote plant invasions in the understory of boreal forests and a fast-growing invasive herb interrupt regeneration of fir trees in forest gaps in balsam-fir-dominated boreal forests in Canada [54, 55]. Both are examples of climate-change-induced alien plant invasions in boreal forests. Climate warming is reported to promote the rapid spread of alien plants into higher altitude areas where they failed to colonize earlier due to unsuitable cold conditions [56]. This is an impending threat to mountain forests. Extreme climate events such as heavy winds, hurricanes, storms, and floods can help long-distance spread especially of propagules of alien plants, pests, and pathogens [57] causing forest health issues even in continents far-off from the source.

It is predicted that the current sources of IAS may change in the future due to shifts in geospatial matched climates worldwide. This will further impede management measures of IAS including preparedness and prevention. Also, there are indications that the distribution of invasive alien plants in terrestrial ecosystems may expand in the future under different climatic scenarios. The hotspots of these invasions are predicted to be located in South America, Europe, New Zealand, and northern and southern Africa [58]. Chances of invasion of woody and herbaceous alien plants in endangered ecoregions in these invasion hotspots in the changing climate scenarios are also projected.

It is well known that climate change may increase the susceptibility of forest flora and fauna to invasion by alien pests and pathogens by enhancing host susceptibility and range expansion [59]. All these factors will contribute to forest degradation significantly. With tree campaigns promoted across the world, there is a serious risk of fast-growing alien tree species, which are planted to sequester carbon in order to mitigate impacts of climate change, becoming invasive [60]. Adequate attention is yet to be paid on this emerging threat. In the near future, one of the major challenges for land managers and conservationists is that climate-change-induced impacts on forest and other ecosystems will make management actions against IAS a greater challenge [46].

4. Climate-degradation interactions in forest ecosystems

The effect of climate on forest systems can be both negative and positive. Climate change could force forests to periodic stresses such as droughts, intense rains, fires, and wind throws, which would adversely affect the tree resilience in these systems. Such stressors in an intense or frequent form can lead to large-scale mortality in susceptible forest systems and produce patches of dried forest. On the other hand, such climate-mediated stressors can facilitate a wide range of essential ecological process such as nutrient cycling, regeneration, and subsequently creation of new habitats at larger spatial scales. In short, climate stressors could modify the forest landscapes to encompass a diversity of mosaic of successional patches representing different stages of disturbance, recovery, biogeochemical cycles, stand structures, and new habitat niche for fauna [61–63].

Studies by Baccini et al. [64] showed that forest disturbance and degradation accounted for 46, 81, and 70%, of carbon losses from tropical forests in Asia, Africa, and America, respectively. The enhanced emissions from forest disturbance and degradation initiate a feeder breeder reaction wherein greenhouse gas content in the atmosphere gets accentuated, and this in turn would increase the extend and severity of degradation in terrestrial systems. On the other hand, increased CO_2 in the atmosphere is also increasingly viewed as a factor for improving net primary productivity of forests. Elevated atmospheric CO_2 content has been reported to increase photosynthesis, a process known as CO_2 fertilization [65]. The magnitude of the CO_2 fertilization effect would however depend on the carboxylation efficiency and leaf area index of plants to rising atmospheric CO_2 concentrations [66]. The forest responses to changing climate would also depend in part on whether the plant photosynthesis from elevated CO**2** content in the atmospheric compensate for enhanced physiological stresses arising from higher air temperatures. Sperry et al. [67] studied competing responses of optimization theory and mechanistic model of tree water transport and photosynthesis and observed that without acclimation, elevated CO_2 content could compromise the net primary productivity (NPP) of monoculture stands by way of increased temperature-driven vascular failures resulting in stress and mortality.

Under a changing climate scenario, the capacity of forests to sustain its productivity and ward of degradation would depend on whether the ratio of elevated CO_2 (ΔCa) to enhanced temperature (ΔT) exceeds the physiological thresholds. In a changing climate, plants would be forced to acclimatize over time, especially in plastic traits (e.g., leaf area), photosynthetic capacity (carboxylation capacity and electron transport capacity), and leaf responses to ΔCa and ΔT [68–70]. The tree density (basal area per ground area), forest leaf area per ground area, and rooting depth in combination would determine the competition for resources and ecosystems response to the ΔCa or ΔT. For forests that do not acclimate, the threshold ΔCa/ΔT must be >89 ppm°C − 1 and for systems that adapt to changing climate, a threshold ΔCa/ΔT of >67 ppm °C − 1 would be required to avoid chronic stress and subsequent degradation. A lower minimum of 55% of existing forest systems without acclimation and 71% of the forests with acclimation are expected to meet the physiological thresholds and negate degradation.

Extreme events associated with climate change can also put enormous pressures on forested systems, degrade them with respect to their ecological functions, and more importantly reduce their capacity to store carbon. Ciais et al., [71] estimated ca. 30% reduction in the gross primary productivity of forests and a strong anomalous net source of carbon dioxide amounting to 0.5 Pg C yr^{-1} in Europe following heat waves. The increase in future drought could turn forest systems into carbon sources accentuating carbon-climate feedbacks, particularly in the tropics and at higher latitudes [72, 73]. Several studies have reported forest degradation [74, 75] as a major factor responsible for the upward trend in carbon concentrations in the atmosphere. However, quantifying carbon losses due to forest degradation as opposed to estimating carbon stored in a stand replacing the lost forest is often challenging, hence ignored in most global estimation of carbon emissions.

In general, climate change enhances forest degradation by way of enhanced physiological stresses (e.g., under floods and droughts), incidence and spread of invasive alien species, insect pests and pathogens, increased fire incidences, and related degradation. Stressors for degradation can be placed under the broad categories of climate, biotic, and anthropogenic, which interact to produce combined effects on forest systems. For example, climate regulates the intensity and spread of forest fires even if they result from human activities. Similarly, plants stressed by extreme drought conditions would have lesser resources to resist disease or insect outbreak. Such instances would invariably decline the forest health, open up canopy, and give way for invasion by alien plants.

Projected climate changes in all likelihoods have been predicted to have profound influences on disturbance regimes and ensuing species demography [76]. The tolerance ability of species to variation in moisture and temperature regimes will be undoubtedly confounded by the alterations caused by disturbances [76]. For instance, climate-change-induced drought enhanced widespread insect pest outbreak leading to massive die-back of different tree species in the Colorado Plateau [77].

Besides the effect of climate on ecosystems, forests also regulate a large array of biophysical properties, which have a direct or indirect impact on the global climate change. Forests absorb a large proportion of sunlight incident on them, thereby having an effectively lower surface albedo than other land uses [78]. In regions with boreal forests, this could produce a cooling effect on deforestation, a reverse of that in the tropical regions. Forests also directly regulate climate by modifying the surface roughness and evapotranspiration. These forest-regulated biophysical factors in turn influence the exchanges of mass, energy, and water between the atmosphere and land surface [79]. These forest-determined surface changes can counteract perceived climate benefits from forest carbon sequestration and may have a negative pressure on the carbon stocks [80, 81].

5. Conclusions

The possibility of a future without healthy forests is a question that looms large and uncertain given the ambiguities in assessing and tracking the changes in forest systems. Deciphering the causes, predicted recovery trajectories and understanding the consequences of forest degradation would be crucial in restoring the lost services of forest systems. As the climate predictions indicate that most of the world forest systems will have to experience CO_2 and temperature levels outside their adaptable ranges, it has become very critical to define the tolerance thresholds and improve our efforts to monitor and restore forest systems. Many of the natural forests across the world have already survived a wide range of climatic conditions during their life span and have developed inherent ways to tide over the degradative forces of climate change. Human concerns of forest degradation reflect our dependence on the system for its products and services. Strategies to tide over the climate-induced changes are imperative for our survival on planet Earth.

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Conflict of interest

The authors declare no conflict of interest.

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Chapter 2

Soil Degradation Processes Linked to Long-Term Forest-Type Damage

Pavel Samec, Aleš Kučera and Gabriela Tomášová

Abstract

Forest degradation impairs ability of the whole landscape adaptation to environmental change. The impacts of forest degradation on landscape are caused by a self-organization decline. At the present time, the self-organization decline was largely due to nitrogen deposition and deforestation which exacerbated impacts of climate change. Nevertheless, forest degradation processes are either reversible or irreversible. Irreversible forest degradation begins with soil damage. In this paper, we present processes of forest soil degradation in relation to vulnerability of regulation adaptability on global environmental change. The regulatory forest capabilities were indicated through soil organic matter sequestration dynamics. We devided the degradation processes into quantitative and qualitative damages of physical or chemical soil properties. Quantitative soil degradation includes irreversible loss of an earth's body after claim, erosion or desertification, while qualitative degradation consists of predominantly reversible consequences after soil disintegration, leaching, acidification, salinization and intoxication. As a result of deforestation, the forest soil vulnerability is spreading through quantitative degradation replacing hitherto predominantly qualitative changes under continuous vegetation cover. Increasing needs to natural resources using and accompanying waste pollution destroy soil self-organization through biodiversity loss, simplification in functional links among living forms and substance losses from ecosystem. We concluded that subsequent irreversible changes in ecosystem self-organization cause a change of biome potential natural vegetation and the land usability decrease.

Keywords: global environmental change, pollution, nitrogen deposition, deforestation, soil self-organization

1. Introduction

Human activity induces degradation of many ecosystems, of which the forest degradation results in far-reaching alterations in nutrient cycles in other related types of the environment. The forest degradation, including impacts on ecosystem functions, is intensified by terrestrial environmental change. Forests are most affected by felling and critical loads of pollution. Both processes may be characterized by negative impacts on soil which subsequently causes a decline in the forest natural ability to regenerate [1]. The damage of forest soils signifies that the development of potential natural vegetation is endangered. Damaged forest soils do not allow restoration of original plant community due to disturbed mitigation of environmental fluctuations. Soil capability to mitigate environmental fluctuations resides in uninterrupted cycles of organic matter and continuous fertility. The disruption of forest soil nutrient cycles disadvantages management utilization including sustainable landscape management [2].

Global environmental change involves a related sequence of biophysical, ecosystem and socio-economic alterations that damage life-sustaining abilities of the planet [3]. The current global change is caused by human transformation of the natural environment, but also by the reaction of human communities to the induced modifications. Human transformations of natural environment are concentrated in a critical zone. The critical zone is range among sphere interfaces on the Earth's surface, where human changes in structure, chemical composition, radiation balance and biodiversity extend [4]. The most vulnerable part of the terrestrial critical zone is composed of soil (pedosphere), which includes interfaces among atmosphere, hydrosphere, lithosphere and biosphere. For these reasons, the soil damage has been altering character of the entire ecosystem over a long period [5].

The most serious soil damage is due to land use modifications after deforestation. While mere forest felling only gives rise to reversible changes in ecosystem functions, the combination of forest felling with soil erosion or conversion for the need of subsequent management utilization generates irreversible ecosystem degradation. Deforestation is instantly followed by declining evaporation and soil loss. The imminent consequences of deforestation are gradually leading to the regional climate change, the loss of ecosystem recoverability and uninhabitable landscape [6]. Regional climate change is mainly caused by reduction of water cycle between the lower-lying areas with higher evaporation and the higher-lying areas with higher atmospheric precipitation in the catchment. The evaporation reduction after deforestation is not sufficient to create cloudiness to make surfaces cooler. Subsequent decrease in precipitation over the higher-lying parts of the river basin deepens water shortage as well as further evaporation decrease in the lower-lying parts [7]. The forest ecosystem recoverability loss is caused mainly due to depletion of the organic matter from the exposed soils, which stimulates germination of tree seeds by means of hormonal effects and moisture retention [8]. Ultimate landscape uninhabitability is caused by uncontrollable soil erosion as a result of surface exposure to wind and landslides of weathered rocks impoverished of organic binders [9].

Forest ecosystem restoration is made impossible within the recent global change, except for soil erosion by pollution. Nitrogen pollution from industry and agriculture has become a major environmental driver of the forest growth [10]. However, atmospheric pollution with the available nitrogen forms is manifested contradictorily within different soil types in forests. The forests situated on optimally fertile soils were generally favorably affected by nitrogen pollution while the predominant forests located on poor soils were damaged. On the one hand, adequate nitrogen intake supports plant growth and, on the other hand, it increases demands on other mineral resources which are declining as a result of human changes in the environment [11]. The unnaturally increasing disparity between plant demands and dwindling nutrient resources causes growth decrease and gradual ecosystem degradation even in hitherto unspoilt areas [12]. Even though the largest nitrogen deposition occurs in the vicinity of pollution sources with lower precipitation, higher concentrations of available nitrogen in wet deposition acidify ecosystems significantly. Approximately 70–80% of nitrogen released from industrial products falls back to the Earth's surface [13]. Of the nitrogen inputs, 5% penetrates the groundwater, 12%

is released into the atmosphere, 30% is immobilized in soil organic matter and 53% is removed with the crop. The utilization of nitrogen by the plant production is still declining, whereas the rate of nitrogen losses by leaching and gasification as well as immobilization in the soil increases in proportion to the amount of fertilizers [14]. Nitrogen supplied to the soil by means of fertilizers results in faster depletion of available bases, making the soil more susceptible to acidification [15].

The environmental nitrogen load is becoming an increasingly important driver of the global ecosystem change as it has exceeded the critical level in large areas of most continents [16]. Exceeding critical nitrogen loads extended plant susceptibility to drought [2, 17, 18]. The widespread plant susceptibility is compound of growing sensitivity of terrestrial ecosystems to climate change. Subsequently, the processes of the climate change and alterations in complex growth conditions for plant communities lead to a deviation in development of prospective natural vegetation or to biome alteration [19]. Therefore, the soil protection is becoming a tool to mitigate the effects of the global terrestrial change maintaining ecosystem link among forests, water cycle and human civilization [5].

2. Forest soil degradation processes

2.1 Impacts on self-organization

Soil degradation may be ranked to one of the most dangerous human activities on the Earth's surface because soil is not instantly renewable. Degradation commences with vegetation coverage damage as a result of which evaporation decreases. The evaporation diminution foreshadows regional warming which contradictorily results in an intensification of water cycle into short, intense rainfall more frequently accompanied by soil drift or even flash floods [20]. Soil degradation affects ecosystem self-organization. The disruption of soil self-organization is initiated with the decrease in the diversity of functional connections within microbial communities. Disintegration of soil functional interconnectedness involves biodiversity loss and substitution of symbioses for decomposers (saprophytes) that do not exchange available nutrients among organisms, but cause leakage of substances from the ecosystem [21]. Forest soil degradation destroys irreplaceable natural values that improve adaptability of cultural landscape to climate change [22]. The disruption of soil self-organization damages both the continuity of crop production and success of ecosystem restoration.

Soil degradation is divided into quantitative or qualitative one (**Figure 1**). Quantitative soil degradation represents the physical loss of a soil body. Qualitative soil degradation involves unfavorable alterations in soil physical or chemical properties that limit ecosystem functions [23]. Soil losses occur through claim, erosion or desertification:

- Claim is usually accompanied by soil sealing, where the land is either merely covered or removed and replaced with building materials. The claim completely destroys soil infiltration capacity. As a result of clearing, the radiation balance and heat capacity alter locally and surface runoff increases sharply [24].

- Soil erosion is surface soil drift by gravitational shifts, water or wind. The human activity has intensified soil erosion after vegetation removal, excessive grazing and inappropriate tillage, developments affecting landscape and pollution

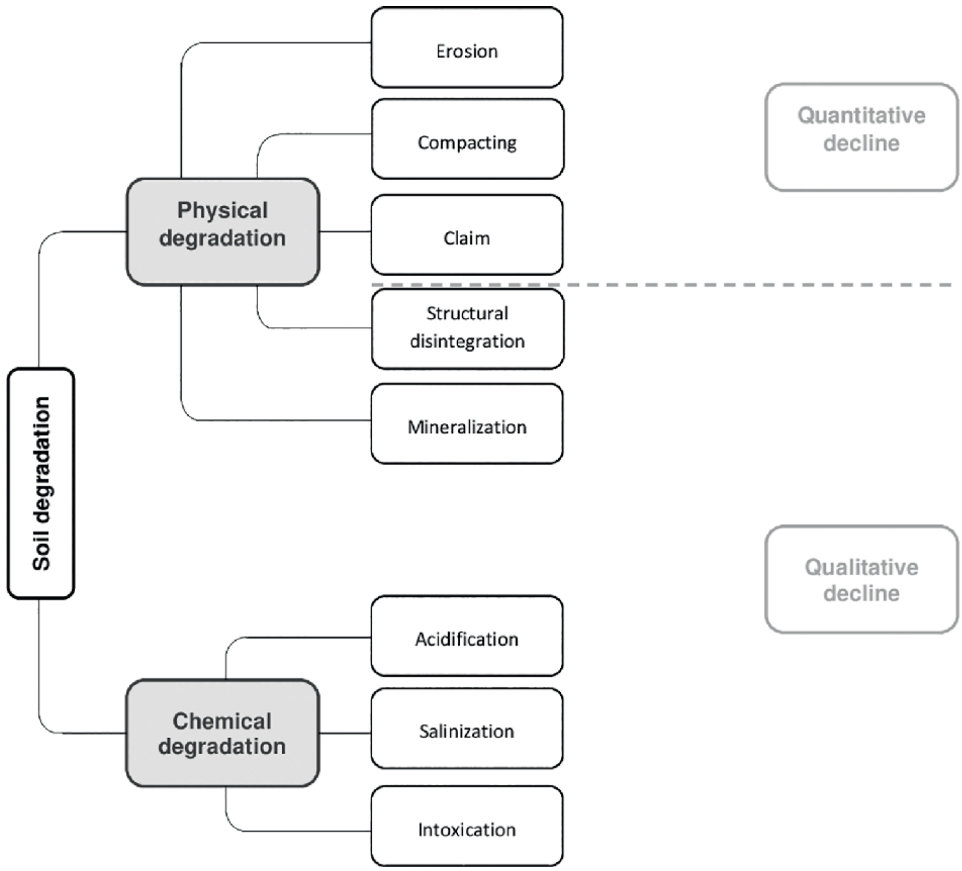

Figure 1.
Division of soil degradation processes along quantitative and qualitative impacts on physical or chemical properties.

which stopped formation of organo-mineral particles aggregating the soil into more cohesive peds (**Figure 2**). The vulnerability through erosion (erodibility) depends on weatherability of soil-forming substrate, soil cohesion, climate and land use (**Table 1**).

- Desertification is unnatural spread of wastelands after permanent vegetation removal. Causes of unnatural desertification are mainly disproportionate grazing, fires and erosion followed by loss of soil water retention capacity. Deserts spread the fastest in areas naturally adapted to seasonal drought [25]. Approximately 10–20% of the world's semi-deserts and steppes are threatened by desertification. The accompanying phenomena of desertification are decrease in groundwater levels or salinization which make it impossible to restore vegetation and lead to wasteland homeostasis [9].

Qualitative soil degradation is produced by excessive losses of the organic matter, the reduction of the biological activity, acidification, contamination, technological compaction (pedocompaction), technical or wind salinization and

Figure 2.
Coupled occurences of soil erosion and spreading of desert in arid environments seriously threat forest restoration due to water availability decrease.

Erodibility	Parent rock	Soil units
Very easy	Eolian deposits	Leptosols, Retisols, Anthroposols
Easy	Clayey shales, basaltic tuffs	Leptosols, Luvisols, Cambisols
Medium	Carbonate deposits, sandstones	Calcaric Leptosol, Chromic Cambisol, Ferralic Podzol
Medium hard	Breccias, graywackes, phyllites, andesites	Stagnic Podzol, Stagnosols, Stagnic Cambisol
Hard	Metamorphites, basalts, diorites	Gleysols, Stagnic Luvisol, Haplic Cambisol
Very hard	Igneous rocks, shales	Epi-humic Cambisol, Vertic Chernozem, Histosols

Table 1.
The vulnerability of forest soils by erosion (erodibility) along various parent rocks and developed soil units.

technical modifications of soil properties. Although qualitative soil degradation potentially occurs in much smaller areas than quantitative one and its effects are usually reversible, they have a similar overlap on landscape functions in the form of reduced water retention capacity and biodiversity, increased runoff and substance imbalances.

Degradation of forest soils is distinctive mainly to the qualitative damage. Qualitative degradation in forests prevails due to the long-term growth of continuous tree species communities. Tree species instantly impede quantitative damage to soils; on the other hand, periodic windthrows during storms mingle the mass among soil horizons, as well as move the soil down the slope. These post-disturbance movements

Figure 3.
A series of differently degraded soil bodies in mountain conditions: introskeletal erosion in Dystric Hyperskeletic Leptosol (A); surface scarification of Entic Podzol (B) and accumulated Spolic Garbic Urbic Technosol (C).

of earth bodies divide microrelief and homogenize soil properties, but at the same time their arrangement is concentrated along the effects of individual global climate changes [26]. However, predominant qualitative degradation of forest soils is manifested by deterioration of physical or chemical properties of solid soil bodies due to external human activities (**Figure 3**) [27].

Deterioration of forest renewal as a consequence of qualitative soil degradation commences with fertility change. Forest felling is accompanied by accelerated leaching of nitrogen substances which can only be ceased by sufficient available calcium [28]. Leaching is preceded by increase in C/N which indicates decrease in ability of soil organic matter to bind mineral nutrients. Soils damaged by compaction of the profile middle part, texturally significantly differentiated or hydromorphic, are exposed to a slow-motion water flow, which expels air for the root growth and similarly increases C/N [29]. The root systems grow merely shallowly with water stagnation and the nutrient loss, making forest stands more susceptible to soil moisture fluctuations [30]. Thus, felling of forests threatened by qualitative erosion impairs the ecosystem ability to restore as a result of exposure to episodic drought [2].

2.2 Physical degradation

Degradation of soil physical properties includes structural damage, pore loss and compaction. The processes of physical soil damage result in both loss of water retention capacity and humus loss by introskeletal erosion. The decline in soil water retention capacity is usually caused by repeated heavy machinery moving. Heavy machinery moving worsens soil aeration and water permeability. Reduced aeration is reflected in the decrease of blank spaces for plant roots and the consequent reduction in the biological activity.

Forest soils are less endangered by physical degradation than agricultural ones owing to dampening effect of surface humus. Nevertheless, topsoil compaction can initiate introskeletal erosion. The mechanical degradation threat is descending from Histosols and gleyed soils to granularly light drying soil bodies [31]. The risks by mechanical damage to the forest soils vary along the grain-size composition, relief exposure and groundwater level (**Table 2**) [32].

Risk	Soil series	Inprint depth	Consistency	Critical pressure (kPa)	
				Dry conditions	Wet conditions
Very high	Histosols+Gleysols	≥35	Cohesionless	30–50	5–12
High	Stagnosols+stagnic soil groups	25–35	Viscous	50–140	12–22
Medium	Stagnic Cambisols-Stagnic Luvisols-Fluvisols	15–25	Crombly	140–300	18–50
Moderate	unhydromorphic Cambisols+Luvisol s+Chernozems+Re gosols	7–15	Cohesive	300–600	50–80
Insignificant	Leptosols+Podzols	<7	Skeletic	600	80–120

Table 2.
Characteristics of forest soil compaction risk after logging machinery movement.

- A very high risk results from high groundwater level and incoherent soil-forming substrate. Forests endangered by a very high risk of soil erosion are mostly found on Histosols or Gleysols, but also on water-affected Luvisols, Arenosols or Podzols.

- A high risk emerges from extremely developed hydromorphic features related to heavy skeletality of soil bodies. Forests exposed to high erosion risk are covering Planosols, Stagnosols and Gleysols, including gleyed subtypes on clayey shales or claystones the most.

- A medium risk is associated with the medium level of forest site gleyfication. Forests at the medium soil erosion risk are located on Stagnic Cambisols, Stagnic Luvisols or Stagnic Fluvisols.

- A moderate risk is related to site desiccation. Forests moderately endangered by erosion means may be found on Cambisols, Luvisols, Chernosols or on Fluvisols developed from sandy substrates.

- An insignificant risk is conditioned by soil cohesion, medium skeletability and merely slightly by sloping relief. Forests insignificantly exposed to soil erosion occur at unexposed sites constituted by Leptosols, Cambisols, Podzols or by Chernosols.

Introskeletal erosion represents a predominantly vertical subsidence of fine-grained soil particles through blank spaces among skeleton to the base of rock mantle. The introskeletal erosion risk resides in unstable occurrence of surface humus. Introskeletal erosion is triggered after removal of the vegetation cover in the exposed sites. Its result is the loss of whole fine-grained matter, followed by impossibility of restoring plant community and permanent exposure of relief [33]. The threat to the site by introskeletal erosion is distributed along exposure of relief and soil skeletability [34]:

- An extreme risk accompanies periglacial brash in arcto-alpine conditions. Extremely endangered sites are only merely sparsely populated by forests. Emerging plant communities are very sensitive to any changes in growth conditions, so they require consistent protection.

- A very strong risk accompanies shallow soils. Forests endangered by very strong risk of introskeletal erosion are most frequently found along upper tree vegetation limit that is sensitive to global warming [35].

- A strong risk is accompanied with brash or stone fields (**Figure 4**). Forests exposed to the strong risk of introskeletal erosion are mostly concentrated on long rocky slopes below the upper limit of tree species vegetation.

- A medium risk is characteristic of islet occurrences of brash on rocky slopes. Forests exposed to the medium risk of introskeletal erosion typically occur in the middle parts of mountain ranges.

- A low risk is specific for sparse outcrops of subsoil decay on medium rocky slopes. Forests threatened by erosion on a small scale occur on gentle slopes with deeply developed soils

2.3 Chemical degradation

2.3.1 Acidification

Degradation of soil chemical properties is the intensification of naturally processed weathering and substance leaching. Chemical soil degradation includes acidification, salinization and intoxication. Acidification is the most extensive process of forest soil degradation causing decline in site fertility [36]. Soil acidification is gradual decreased in neutralizing capacity. In nature, acidification is elicited mainly by water autoprotolysis, naturally acid atmospheric precipitation, organic acids activities, but also by formation of strong acids after reactions of water with atmospheric gases (CO_2, SO_2) or with some rock-forming minerals (chlorides, sulphates or carbonates). The resulting acids (formal HCl, H_2CO_3 and H_2SO_3) can cause very intensive decomposition of original minerals into salts [37].

The intensification of soil acidification was caused by fertilization, crop cultivation and industrial pollution. Industrially emitted CO_2, SO_2 and NO_x create formal acids and soil bases are excessively depleted to neutralize them. The base loss slows down humus formation; on the other hand, raw humus is a significant source of organic acids. The slow-motion formation of humus is reflected in decrease of organo-mineral colloid genesis as a result of which the number of binding sites for exchange cations on the active surfaces of soil particles decreases. The decomposition of variable organo-mineral colloids limits base cations exchanges to stable mineral colloids. Nonetheless, mineral colloids can capture only 0.2–25% of exchangeable cations, unlike organic particles [38].

The soil resists acidification impacts by exchange reactions between inputs of acid-forming H_3O^+ and available sources of releasable cations. Soil cation sources are

Figure 4.
Introskeletal erosion leaves rocky flows without surface humus instead soil, where plants can to root hardly, thus forest site gets features of disperse platforms with dwarf vegetation.

active depending on pH (**Figure 5**). Intensified acidification of forest soils is naturally slowed down either by deciduous tree species or by weathering of the soil-forming substrates. The influence of tree species predominates in surface soil horizons while the influences of soil-forming substrate predominate in the subsurface horizons. Significant acidification in surface horizons of forest soils most often affects the transitional ecosystem types [39]. Even though the mitigating effect of tree species does not overcome impacts of weathering, the optimal tree species composition actively

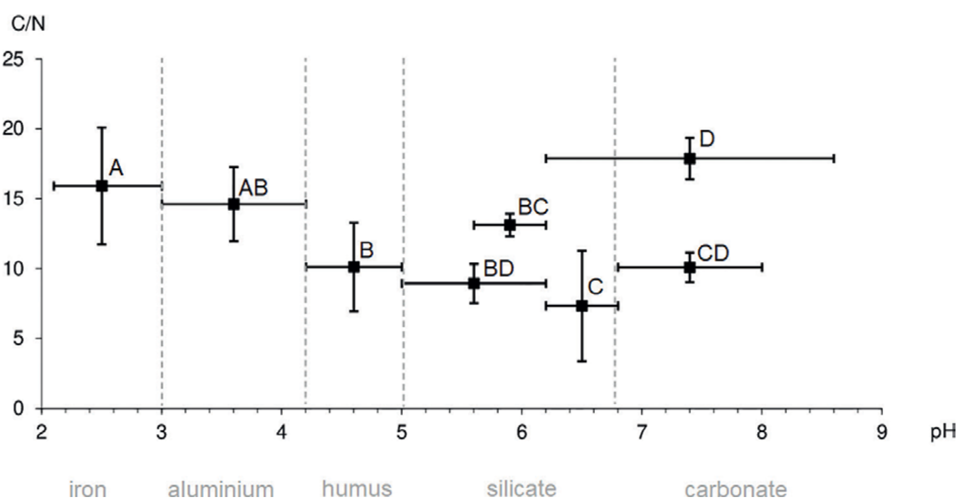

Figure 5.
Intervals of soil acidity (pH) and organic matter C/N ratio divide trophic (A – Oligotrophic; AB – Oligomesotrophic; B – Mesotrophic; BC – Mesotrophically nitrophilous; C – Nitrophilous; CD – Nitrophilous-base; BD – Mesotrophically base; D – Base) series among zones buffering acidification through specific neutralization. Data according to [39].

reducing C/N prolongs weathering effects. On the other hand, soil cation release by weathering maintains intensified acidification as a reversible process. Weathering counteracts acidification by means of electrochemically controlled soil-forming substrate decomposition [12, 40, 41]:

1. **The carbonate zone** (pH > 6.8) is employed by dissolving the CO_3^{2-} compounds, whereby the incoming H^+ is neutralized into soluble salts. The consequence of these acid–base reactions is a gradual loss of carbonates by dissolution and leaching, which can only be forestalled in soils formed directly from carbonate substrates and rocks.

2. **The silicate zone** (pH 5.0–6.8) occurs either within soils from which carbonates have already been leached or where silicates predominate. Acids cause decomposition of silicates from which base cations are released and deuterogenous clay minerals are formed.

3. **The exchange zone** (pH 4.2–5.0) may be found in those soils where there is a disproportion between base cations released during weathering of silicates and H^+ inputs. The excess of entering H^+ is trapped on surface of organic colloids to release bases.

4. **The aluminum zone** (pH 3.0–4.2) subdues effects of acidic inputs by releasing Al^{3+} in the presence of sesquioxides with simultaneous formation of organic complexes. Soil fertility decreases, nutrients are further leached and the biological activity decreases.

5. **The iron zone** (pH < 3.0) occurs in those soils where acidic inputs are subdued by the dissolution of iron oxides, Fe^{3+} migration and destruction of clayey minerals. Nutrients are excessively leached out from these soil bodies, the concentration of toxic substances in soil profile increases and the biological activity is usually concentrated only into raw surface humus.

The unnatural decomposition of soil minerals triggers irreversible acidification. Acidification may be mitigated merely after the removal of acidifying substances sources. The acidification of forest soils affected exchange zone the most, switching to active aluminum zone [42]. The damage to the forest ecosystem by release of active Al^{3+} followed due to occupation of exchange sites on soil particle surfaces instead of bases, the lack of which limited root growth. The roots were concentrated shallowly below the surface so that new focal points of biological activity and humus ceased to form deeper in the soil [43]. Introducing the other side of the fact, the marginally widespread transition from the aluminum zone to iron one was ensued by loss at the ability of forest ecosystems to restore from the damage (**Figure 6**).

Air pollution has significantly accelerated soil acidification, especially in the areas of forests transformed into homogeneous stands of coniferous tree species. While cultivation of homogeneous coniferous forests homogenized formation of acidic humus causing micropodzolization and increased base cation leaching, the pollution after acid deposition reduced not only the forest increment but also decomposition of organic matter [44]. Forest increment was reduced by direct damage to the assimilation apparatus, by stimulating sensitivity to seasonal drought or frost and by reduction in soil symbioses mediating nutrient deficiencies. The decline of mycorrhizal

Figure 6.
Irreversible damaged forests are characteristic by predominantingly dead tree storey and by absent young woods due to lost soil organic matter irreplaceably stabilizing moisture during seed germination.

fungi was followed by increase in frequency of saprophytic to saproparasitic fungi, which diverted organic matter decomposition to complete leaching from the ecosystem [45]. The susceptibility of mycorrhizal symbioses to pollution resulted in limited accessibility to phosphorus necessary for nucleic acid synthesis [46]. The disturbed phosphorus cycle triggered decrease in increment as well as seed germination leading to forest self-organization loss [47].

2.3.2 Salinization

Soil salinization is the process of accumulating surpluses of mineral salts. Salinization of forest soils is a rare phenomenon, but it threatens 23% of agricultural land, mainly in arid areas [48]. Forest soils are salinized in areal or linear extent. Areal salinization is caused by high groundwater mineral levels, the use of saline water for irrigation, waste materials for fertilization or deposition of solids. Linear salinization occurs alongside roadsides maintained by chemical salting during winter or along river banks. The recent climate change is expanding areas of salinized soils with rising sea level along the coast or estuaries. On the other hand, the natural risk of soil acidification subdues consequences of salinization [49].

The impacts of salinization in forests are associated with extreme soil chemical properties. Salinization highlights malfunctions of water and nutrient uptake by plants. Above all, the disproportionate sodium input (sodification) disrupts ration among exchangeable bases in the soil environment, thereby disrupting effects of alkalization on soil structure. Significant Na^+ inputs displace other cations from soil sorption complex and disperse soil particles. Sodium displacement of cations results in deficient nutrition, but at the same time crushes soil structure, thus water availability fluctuates. Sodium surplus in plant tissues reduces osmotic pressure, whereby cells lose ability to absorb other substances from soil solution [27]. While conifers are susceptible to soil salinization, deciduous tree species are tolerant to it. The younger plants are more susceptible than the older ones.

Areal forest salinization is most at risk in floodplains due to variability of water flow. The regulation of water flow caused groundwater level fall in some river basins while it resulted to water level permanent increase in some other ones. The groundwater level decline was typically ensued by ecosystem desiccation due to the fact that riparian forests are mostly located in submontane locations with insufficient precipitation [50]. By contrast, rising groundwater levels after water regulation meant change in availability of mineral ions, with impacts on soil microbial activity and ability of the ecosystem to sequester carbon. The increase in level of saline water inflicts decrease in soil microbial activity and consequently decrease in vegetation growth [51]. On the one hand, decreases in growth processes are caused by loss of oxygen in soil environment, on the other hand, by increasing concentrations of Na^+, Cl^-, SO_4^{2-} ions, including Fe and Mn compounds. In particular, SO_4^{2-} in the soil solution is converted to toxic sulphide when there is a lack of oxygen. Although Fe and Mn are biogenic elements that catalyze soil organic matter decomposition, bound in sulphides block microbial metabolism [52]. The main forest salinization danger with groundwater is inability to adapt on climate change due to spread of microaerobic conditions [53, 54].

2.3.3 Intoxication

Intoxication of soils with heavy metals, radioactive or petroleum substances is a rare but very hazardous process of cumulative pollution. Especially, heavy metals merely slowly participate in biogeochemical cycles and accumulate in the ecosystem because they are either microbiogenic (Cu and Zn), or xenobiotic (e.g. Cd, Co, Pb, Hg, Ni).

Soil intoxication occurs by deposition means. Sources of cumulative pollution are point or dispersed ones. The point sources of heavy metals are smelters, thermal power stations or municipalities by watercourses. The dispersed sources are represented by polluted water, inappropriate distribution of industrial or sewage sludge or operation of internal combustion engines. Fluvisols, which are among the most intoxicated soils, are usually located between watercourses and agricultural soils [48].

Xenobiotics are toxic to most organisms. They mainly affect energy balance of living cells and their division. Heavy metals mainly bring about halting respiration as a consequence of interactions with SH- groups at intracellular enzymes and their complexes, they disrupt semipermeability of cell membranes and their proton gradient. Soil environment pollution with heavy metals significantly reduces density of microbial occurrences and directly damages plants. The rate of soil contamination damages microbial activity significantly more than the differences in heavy metal contents among different sites [55]. However, the decline of susceptible species is being replaced by expansion of resistant species populations, including pests abundantly infesting damaged plants [56].

Resistant phytophagous arthropods adapt to the environment contaminated with heavy metals by searching for less contaminated nourishment, sufficient release of metals in excrements, by sloughing or by means of other tissues (in the adipose body, epithelium of the digestive tract) [57]. The importance of phytophagous insects for the movement of heavy metals in ecosystem lies in the fact that these invertebrates are an important link in food web that receive toxic substances directly from plants, especially Cd and Zn [58, 59], but no Ni and Fe [60].

The risk of heavy metal accumulation with toxic manifestations threatens not only tree species, but also predators. Numerous ground beetles (Carabidae) and ants (Formicidae) are mainly food-bound to phytophagous insects, earthworms, countless larvae and springtails [61]. On the other hand, the immobilization of heavy metals takes place in cells of specialized metallogeneic microorganisms by binding to metallothionein-based amino acids. Immobilization of heavy metals in amino acids changes course of humus formation. The rate of biosorption on the soil microbial active surfaces typically decreases in order Zn > Cd > Pb > Cu > Cr [62]. Alterations in the forest ecosystems as a result of heavy metal pollution include (**Figure 7**):

- the reduction of phospholipid fatty acids content in the bacterial cells and raw humus. Even though the total content of phospholipid acids is directly proportional to ATP synthesis, it decreases under the toxic load. The consequence of these processes is alterations in overall functional diversity of microbial community and inability to decompose deposition by aromatic hydrocarbons [63].

- promoting release of mobile humus substances and decrease of insoluble substances. On the other hand, the great ability of Pb to form complexes with insoluble humus substances maintains its immobilization while Cd and Zn tend to form soluble complexes. Although more heavy metals are bound to soluble humus substances than to insoluble organic compounds, the greater release of soluble compounds also contributes to pollutant migration in the soil and to their penetration into groundwater [64].

Petroleum products belong to secondary persistent organic compounds, similar to benzo(a)pyrene, polychlorinated dibenzo-p-dioxins or dibenzofurans, which are removed from the soil for more than 2 years [65]. Like benzo(a)pyrene, they consist of polycyclic aromatic hydrocarbons that directly harm the health of organisms. Oil pollution is significantly more caused by human activity than by natural (geogenic) sources. It begins with mining, combustion of petroleum products (fossil fuels), accidental or operational spills and corrosion of industrial materials. Oil products in the forest ecosystem load surface humus the most, which at the same time prevents their

Figure 7.
Forests dying at regions loaded by acid deposition were transformed to substitute stands of resistant introduced tree species which have provided cover for regeneration of indigenous forest communities after pollution decrease.

penetration into deeper occurring soil. The load of surface humus decreases activity of soil microorganisms. The decrease of soil biological activity is mostly caused by aromatic nuclei imitating lignin, which either block formation of amino acids or replace carbon compounds in fungi [66]. Subsequently, the humus decomposition is disrupted, foreshadowing disruption of processes to get available nutrients from the soil. Nevertheless, the load of petroleum substances is irregularly concentrated in surroundings of industrial areas and vertically along different intensities of wood logging in floodplains, hillycountries, highlands and high-mountain forests [67].

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Section 2

Impacts of Climate Change

Chapter 3

Impact of Climate Variability on Forest Vegetation Zones in Malawi

Edward Missanjo and Maggie Munthali

Abstract

A study was conducted to evaluate the influence of climate variability on forest type and forest living biomass. Three scenarios were used in the assessment. Namely: Near century (2011–2040), mid-century (2041–2070), and end-century (2071–2100). Holdridge Life Zone model and GAP Formind modified were used for the assessment. The results show that three forest vegetation zones will be observed from near century to end century. Namely: dry forest, very dry forest and thorn woodland forest. Under near century climate conditions, there are two forest vegetation zones occurring: dry forest and very dry forest. Under mid-century climate conditions, thorn woodland forest will emerge, and dry forest will disappear in the end-century. There will be a significant decrease in forest living biomass (1000 kg ha^{-1} yr^{-1}) from near century to end-century. The study has demonstrated that future climate change will be conducive to growth and expansion of very dry forest vegetation zone, which causes positive effects on reforestation planning and adaptive strategies. Therefore, the study suggests the following as some possible strategies to adapt climate change: promotion of natural regeneration of tree species, promotion of tree site matching, production and promotion of new tree seed varieties; and seed banking for drought resistant tree species.

Keywords: adaptation, climate change, vegetation zone, forest biomass, climate condition

1. Introduction

Forests contribute directly to Malawi's GDP through domestic and export product sales, employment and tourism. Forest supply more than 96% of the country's energy need. Apart from energy, trees provide timber and non-timber forest products [1, 2]. Malawi's forest cover 23,677 km^2. This represents 25% of the total land area. The forest is mainly categorized into two: Miombo woodlands and plantation forests. Miombo woodlands covers 96% (22,857 km^2) of the forest, while the plantation forests cover the remaining 4% (820 km^2) of the forest. Brachystegia is the most predominant tree genus in miombo woodlands while pine and eucalyptus are the most mutual trees in the plantations [2].

Despite the important role forests play in Malawi, the forests are being cut and degraded much faster than they are regenerating. It is estimated that the rate of deforestation is 2.8% per annum [3–5]. Furthermore, recent study has shown that between 2019 and 2025, Malawi's demand for wood fuel will exceed sustainable supply [1]. In addition, the adverse impact of climate change on the environment has been widely reported [6–9]. Malawi has not been spared from the adverse impacts of climate variability and change as evidenced by recent floods and drought [10].

Therefore, it is important for Malawi to identify measures and strategies for adapting the climate variability and change. The purpose of this study was: (1) to determine how many forest vegetation zones exist in Malawi, (2) to determine how the structure of forest vegetation zones respond to future climate change in the aspects of boundaries, areas, and forest living biomass. This study is of importance for improving our understanding of the effect of climate change on vegetation zones and for planning the adaptation strategies of future ecological restoration programmes in Malawi and the surrounding region.

2. Material and methods

2.1 Study area

Malawi is placed in Southeast Africa. It is bordered by Zambia on the west and northwest, by Tanzania on the north and northeast, and by Mozambique on the east and southwest (**Figure 1**). Malawi's sub-tropical climate is characterized into three seasons: Cool-dry, warm-wet, and hot-dry seasons. Cool-dry season is evident from May to August with a mean temperature variety of 4–10°C. In cool-dry season, specifically in June and July frost may fall in some areas. Warm-wet season stretches from November to April with an annual mean rainfall series of 725 mm to 2500 mm.

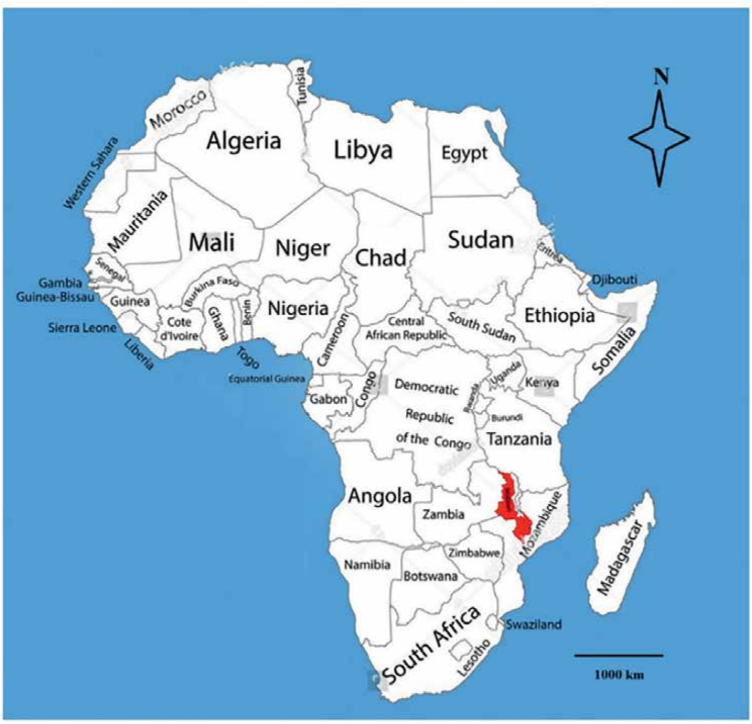

Figure 1.
Location of Malawi in Africa.

This is the season through which 95% of the annual rainfall falls. Hot-dry season lasts from September to October with a mean temperature range of 25–37°C [11].

2.2 Simulation process

2.2.1 Climate layers and future scenarios

The assessment was conducted based on five climatological zones of Malawi (**Figure 2**). Seasonal climatic characteristics and other information about the five climatological zones in Malawi are presented in **Table 1**. The assessment used three scenarios: Near-century; mid-century; and end-century with the following time frames: 2011–2040; 2041–2070; and 2071–2100, respectively. The projected precipitations and temperatures for the three scenarios were obtained from the Ministry of Forestry and Natural Resources in the Department of Climatic Change and Meteorological Services

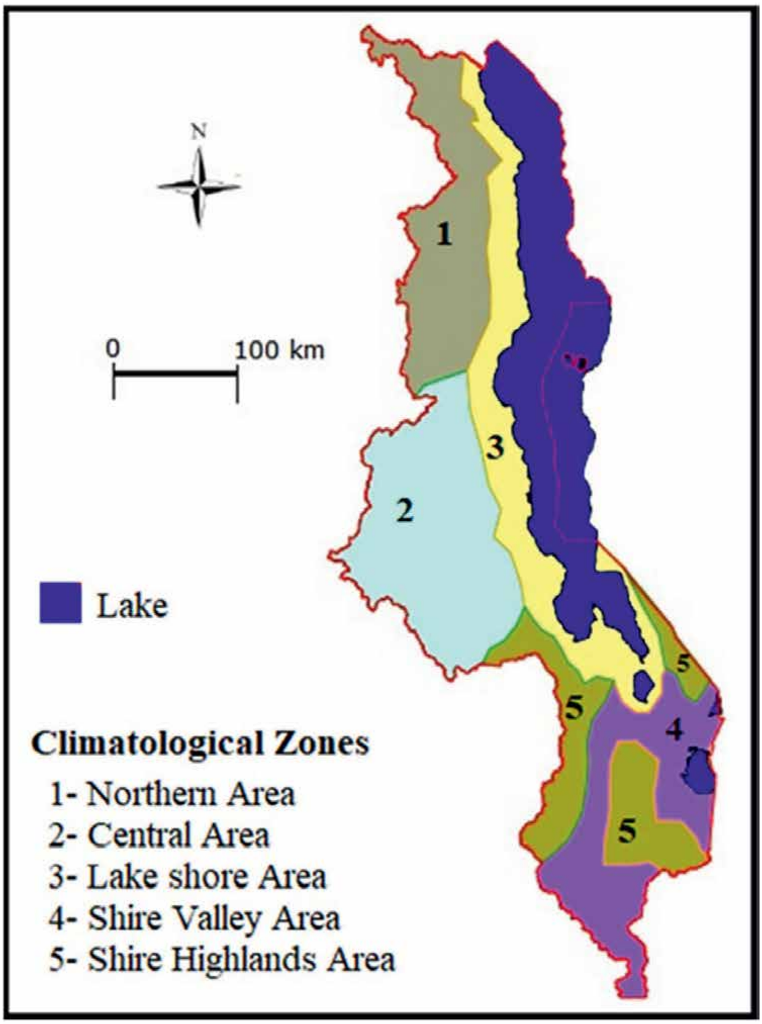

Figure 2.
Malawi's five climatological zones.

Climatological zone	Climate area (km^2)	Seasonal climatic characteristics
Shire valley area	18,305.01	These are Malawi's lowest areas with altitudes below 50 metres above the sea level. Covers areas along the shire river, Lake Chirwa and Lake Chiuta Valleys. The area has a semi-arid climate prevailing. The wet season is oppressive and mostly cloudy, the dry season is clear, and it is hot year-round. Over the course of the year, the temperature typically varies from 19°C to 36°C and is rarely below 17°C or above 40°C. The rain period of the year lasts for 7 months from October to May with an average rainfall per month of 12 mm. The month with the most rains is January with an average rainfall of 200 mm.
Shire highlands area	15,268.33	These are the plateau areas in the southern part of Malawi and Kirk Range up to Dedza. The plateau varies in elevation from 600 to 1100 mm above the sea level. The highest peak is Sapitwa in Mulanje Massif at 3002 m followed by Zomba Mountain at 2087 m. The area experiences extreme seasonal variation. The wet season is muggy and mostly cloudy, the dry season is clear, and it is warm year-round. Over the course of the year, the temperature typically varies from 13°C to 28°C and is rarely below 12°C or above 32°C. The rain period of the year lasts for 7 months from October to May and the month with the most rains is January with an average rainfall of 260 mm.
Lake shore area	20,074.26	These are areas along Lake Malawi. The wet season is hot, oppressive, and overcast and the dry season is warm, windy, and mostly clear. Over the course of the year, the temperature typically varies from 16°C to 32°C and is rarely below 13°C or above 35°C. The rain period of the year lasts for 6 months from October to April and the month with the most rains is January with an average rainfall of 250 mm.
Central area	25,916.06	It covers the following areas Lilongwe, Mchinji, Dowa, Ntchisi, Kasungu, and part of Mzimba district. The climate is warm and temperate. The summers are much rainier than the winters. The average annual temperature is 20.4°C. In a year, the rainfall is 739 mm. The driest month is June, with 1 mm of rain. The greatest amount of precipitation occurs in January, with an average of 211 mm.
Northern area	18,143.93	It covers all areas north of Mzimba district expect the Lake shore area. The climate is classified as warm and temperate. When compared with winter, the summers have much more rainfall. The average annual temperature is 19.2°C. Precipitation is about 1487 mm per year. The least amount of rainfall occurs in September. The average in this month is 15 mm. In January, the precipitation reaches its peak, with an average of 303 mm. The temperatures are highest on average in November, at around 21.9°C. At 15.3°C on average, July is the coldest month of the year

Table 1.
The area and seasonal climatic characteristics of climatological zones in Malawi.

(DCCMS), Blantyre, Malawi. Briefly, projections for future precipitation and temperature were developed using the 20 global scale general circulation models (GCMs). These are downscaled outputs used in the Intergovernmental Panel on Climate Change Fifth Assessment report. The GCMs were used in concurrence with two representative concentration pathways (RCPs; RCP4.5 and RCP8.5) [12]. Observed

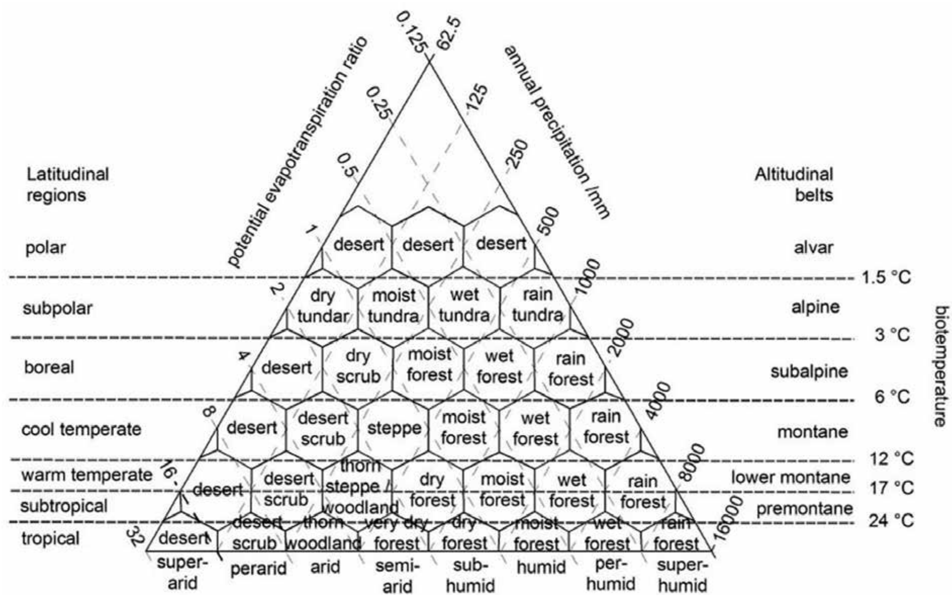

Figure 3.
The Holdridge Life Zone model concept framework that divides the world territorial ecosystems into 39 vegetation zones.

data temperatures and precipitation data used was for the period 1961–2010 while daily temperatures and precipitation data used was for 1971–2000.

2.2.2 Holdridge life zone model

Holdridge life zone (HLZ) model was used to assess climate change impact on forest type while QGIS3.2 was used to produce the forest type maps. HLZ model is well explained by Li et al. [6]. Briefly, The HLZ model is a classic climate- vegetation model designed by Holdridge [13]. It divides world territorial ecosystems into 39 vegetation zones (**Figure 3**). The 39 vegetation zones are mapped in a triangular coordinate system with three key climatic variables [6, 13]. The three key climatic variables are: annual bio temperature (ABT), annual precipitation (AP), and potential evapotranspiration ratio (PER) [6]. In this study ABT, AP and PER were estimated using the following equations [6]:

$$ABT = \frac{1}{12}\sum_{i=1}^{12} T_i \quad (1)$$

$$AP = \sum_{i=1}^{12} P_i \quad (2)$$

$$PER = 59.93\,\text{x}\,\frac{ABT}{AP} \quad (3)$$

Where T_i is monthly mean temperature and P_i is monthly precipitation. In addition, QGIS 3.2 was also used to analyze the change area of the vegetation zone.

2.2.3 GAP formind modified model and biomass estimation

The impact of climate variability on forest living biomass was assessed using the GAP-Formind modified model, while the above and below ground biomass were estimated using the following site-specific models [14].

$$AGB = 0.21691 \text{ x } DBH^{2.318391} \tag{4}$$

$$BGB = 0.284615 \text{ x } DBH^{1.992658} \tag{5}$$

Where AGB and BGB are above and below ground biomass (kg dry matter per tree), respectively and DBH is the diameter at breast height (cm). The total living biomass per tree was estimated by adding up AGB to BGB. In Addition, the following National Forest Inventory (NFI) data for different climatological zones were used for the estimation of biomass:

- Liwonde National Park, Lengwe National Park and Mwabvi Wildlife Reserve (Shire valley area)

- Chongoni and Dzonzi-Mvai Forest Reserves (Shire highlands area)

- Dzalanyama and Ntchisi Forest Reserves (Central area)

- Chinyakula Village Forest Area in Nkhatabay (Lake shore area)

- Misuku and Perekezi Forest Reserves, Chileta and Chowe Village Forest Areas in Rumphi and Chitipa, respectively (Northern area)

3. Results

3.1 Impact of climate change on forest type

Climate change projections indicate that some forests would significantly change while others would not change (**Figure 4**). For instance, the central area and northern area forests would be converted from dry forest in near century to very dry forest in mid-century. The central area forests would further be converted from very dry forest in mid-century to thorn woodland forest in end-century. The lake shore area forests would be converted from very dry forest in near century to thorn woodland forest in mid-century. On the other hand, the shire highlands forest would be converted from dry forest in mid-century to very dry forest in end-century. Interestingly, the shire valley forests would not be affected by climate change.

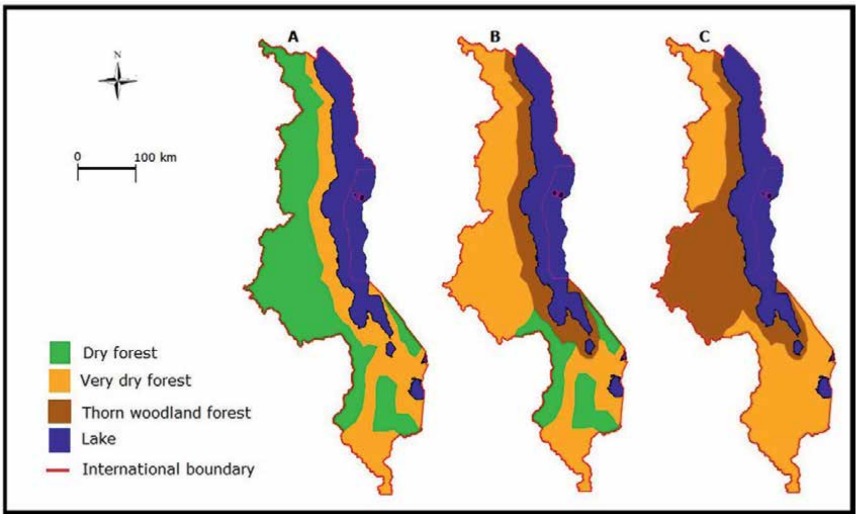

Figure 4.
Impact of climate change on forest type using Holdridge Life Zone (HLZ) model under three scenarios, A: Near century (2011–2040); B: Mid-century (2041–2070); and C: End-century (2071–2100).

3.2 Spatial distribution patterns of vegetations zones

The results on the spatial distribution pattern of forest zones in Malawi are presented in **Figure 5**. Three forest vegetation zones will be observed in the near century, mid-century and end-century climate change scenarios from the concept framework of Holdridge Life Zone model system. They are dry forest, very dry forest and thorn woodland forest. Under near century (2011–2040) climate conditions, there are two forest vegetation zones occurring in Malawi: dry forest and very dry forest. Under mid-century (2041–2070) climate conditions, one new forest vegetation zone will emerge in Malawi (thorn woodland forest), and dry forest will disappear in under end-century (2071–2100) climate conditions.

3.3 Area distribution change of forest vegetations zones

Summary of the results on the area distribution change of forest vegetation zones in Malawi are presented in **Table 2**. The results show that dry forest vegetation zone shall decrease its area proportion from 60.7% to 15.6% from near century to mid-century. On the other hand, very dry forest vegetation will expand its area proportion from 39.3% to 63.8% under the same climate condition scenario. Most notably is that, thorn woodland forest would shall emerge under the same climate condition scenario. Furthermore, the results indicate that thorn woodland forest would increase its area proportion from 20.5% to 47.1% from the mid-century to the end-century. Consequently, very dry forest would decrease its area proportion from 63.8% to 52.9%, while dry forest will disappear.

3.4 Effect of climate variability on forest living biomass

Summary of the results on the effect of climate variability on forest living biomass, are presented in **Figure 6**. The results show a significant decrease in forest

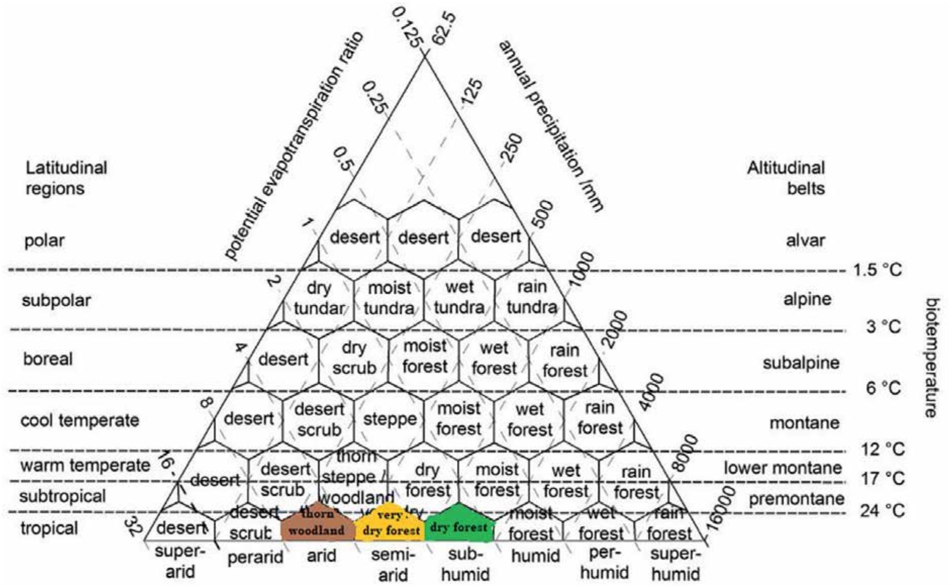

Figure 5.
Three forest vegetation zones that occurs and that will be observed in Malawi under Holdridge Life Zone model system.

Vegetation zone	Climate area (km^2)		
	Near-century (2011–2041)	Mid-century (2041–2070)	End-century (2071–2100)
Dry forest	59,328.32	15,268.33	—
Very dry forest	38,379.27	62,365.00	51,717.27
Thorn woodland forest	—	20,074.26	45,990.32

Table 2.
The area of forest vegetation zones under near-century, mid-century and end-century climate change scenario.

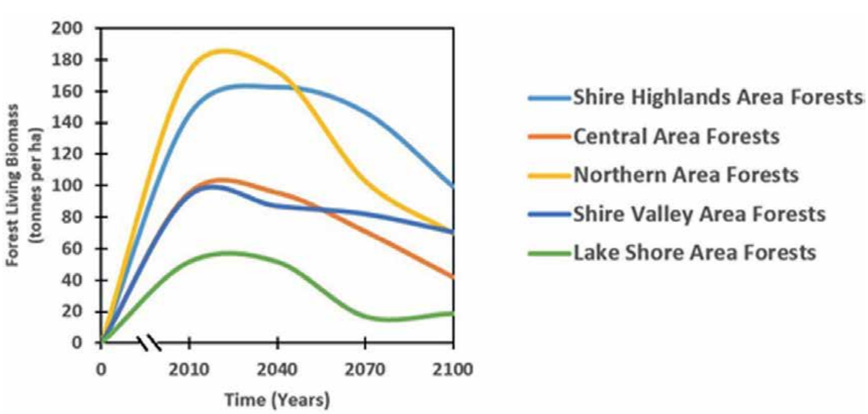

Figure 6.
Prediction on the impact of climatic change on forest living biomass for different climatological zone forests in Malawi.

living biomass from near-century to mid-century for both lake shore area forests (1200 kg ha^{-1} yr^{-1}) and northern area forests (2300 kg ha^{-1} yr^{-1}). Likewise, projections show a significant decrease in forest living biomass from mid-century to end-century for both shire highlands forests (1600 kg ha^{-1} yr^{-1}) and central area forests (1000 kg ha^{-1} yr^{-1}). Conversely, the projections show that forest living biomass for shire valley area forests would not be highly affected by climate variability.

4. Discussion

Numerous research studies have been conducted to examine the potential effects of climate change on the distribution of terrestrial vegetation at regional, national and district scales [6, 15–17]. The present study shows that forest vegetation zone responses to climate change are significantly different under the three climate conditions scenarios. It has been demonstrated that dry forest vegetation zone will likely disappear in the end-century. In addition, one new vegetation zones (thorn woodland forest) will appear in the mid-century scenario.

In mid-century and end-century more than 50% of the forest vegetation would be very dry forest. This indicates that mid-century and end-century climate changes will be beneficial for the growth and expansion of the very dry forest. The results reported by Li et al. [6] also support the present findings.

Various climate-vegetation models are used for determining the impact of climate change on forest vegetation type [16]. The present study used HLZ model because of its simplicity, it only needs three parameters to be used, hence more advantageous than other models [6, 18, 19]. Even though the parameters used in HLZ model may sufficiently simulate vegetation patterns, the actual patterns can be described by a function of additional factors that are not clearly considered in the model, which are caused by human [6]. Li et al. [6] and Josa et al. [20] argued that human activities may change the response of vegetation to climate change through transformation of land use types. Therefore, it is said that HLZ model simulates ecosystem potential functions rather than the actual ecosystem structure [6, 20]. Equally, the present study is in agreement to that argument.

The forest vegetation zones simulated under the three climate conditions scenarios in the present study can provide important reference information for policy makers in planning regional vegetation restoration. Some of the strategies that can be used to adapt to the climate change would be promotion of natural regeneration of tree species, promotion of tree site matching, production and promotion of new tree seed varieties; and seed banking for drought resistant tree species.

5. Conclusion

The present study assessed the impact of climate change on forest type, forest living biomass, basal area, and number of stems. HLZ model and GAP Formind modified were used under three climate condition scenarios; near century (2011–2041); mid-century (2041–2070); and end-century (2071–2100). The results show that two forest vegetation occurs in Malawi (dry forest and very dry forest) in near century. Thorn woodland forest will emerge in the mid- century, while dry forest will disappear in the end-century. Furthermore, the results indicate an overall significant decrease in forest living biomass due to climate change in the end-century. Therefore, future climate change will be conducive to growth and expansion of very dry forest vegetation zone, which causes positive effects on reforestation projects in the region.

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Conflict of interest

The authors declare no conflict of interest.

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Chapter 4

Forest Degradation in Tanzania: A Systematic Literature Review

Emmanuel F. Nzunda and Amri S. Yusuph

Abstract

Forest degradation is a process in which the biological diversity of a forest area is permanently reduced due to one or more factors. Forest degradation continues at an alarming rate, contributing significantly to the loss of biodiversity around the world. This chapter presents the findings of a systematic literature review of forest degradation in Tanzania. The PRISMA method was employed in the study's search, document selection, and data analysis. There were more studies more recently due to the increasing interest in forest degradation as an important aspect of forest management. Most terms are mentioned less frequently in the document title than in the document as a whole, indicating research gaps for the research topics represented by the research terms. Some terms are covered less than expected, given their significance in forest degradation. The estimated annual volume removals exceed the estimated mean annual increment, indicating forest management in Tanzania is not sustainable. The most mentioned region was Dar es Salaam, while the list mentioned was Rukwa. It is expected that forest stakeholders will find the analysis presented in this study useful. Furthermore, the stakeholders will find interest in addressing temporal, spatial, and thematic research gaps highlighted in this chapter.

Keywords: forest degradation research, temporal and thematic gaps, factors and drivers, monitoring and assessment, forest management institutions, spatial distribution

1. Introduction

Tanzania's forests cover approximately 45.7 million ha, or about 55 % of the country's total land area, as of 2022 [1]. The forests provide essential goods and services [2]. In addition to providing economic benefits, Tanzanian forests and other woodlands serve as important habitats for a variety of animals and plant species. Over 10,000 plant species have been identified in these forests, with 305 being classified as threatened by the IUCN Red List and 276 being classified as endangered [3]. Despite the valuable goods and services that forests provide, high rates of deforestation and forest degradation persist [4, 5]. Forest degradation is a subset of the larger issue of land degradation. Deforestation and forest degradation continue at alarming rates, contributing significantly to the ongoing loss of biodiversity around the world. According to the last Global Forest Resources Assessment report (2020), deforestation has occurred at a rate of 469,000 ha per year. Forest degradation is defined as changes within the forest that have a negative impact on the structure or function of the stand and/or site, reducing the capacity to supply products and/or services [6]. Forest degradation alone contributed 25% of total emissions from deforestation and forest degradation even exceeding emissions from deforestation in some countries [7].

The major causes of forest degradation in the unreserved forest include heavy pressure from agricultural expansion, livestock grazing, development of human settlements, overgrazing, firewood and charcoal production, uncontrolled fires, timber extraction, development of infrastructure/industry, refugees, and most recently, the introduction of large-scale agriculture for bio-fuel production [5]. Several studies in Tanzania have investigated the various drivers, trends, and methods for assessing deforestation and forest degradation [4, 8, 9].

To sustainably provide goods and services from the forests, effective forest management initiatives are required. If these initiatives are implemented, Tanzania may ultimately be able to reduce emissions caused by deforestation and forest degradation. This would help it meet its emission reduction targets. This chapter discusses the findings of a systematic literature review of forest degradation in Tanzania, in terms of (1) trends in research on forest degradation, (2) research topics covered and gaps in research, (3) extent of forest degradation, (4) drivers of forest degradation, (5) methods of monitoring and assessment of forest degradation, (6) institutions involved in efforts to control forest degradation, and (7) spatial distribution of studies on forest degradation.

2. Methodology

2.1 Data sources

The research method used in this study was a systematic literature review. The PRISMA (Preferred reporting items for systematic reviews and meta-analyses) method was used to select the articles [10]. Scopus and Google Scholar were used to search for articles. **Table 1** shows the keywords used to search and retrieve published articles in the Scopus and Google Scholar databases.

A total of 437 records were retrieved after searching Scopus and Google Scholar for articles. To avoid bias in the search, we conducted hand searches by following prominent scholars in the field, resulting in the collection of 33 papers. The collected documents were loaded into Zotero (reference manager software). During the screening process, duplicate records were removed. Rayyan was then used to screen the article titles and abstracts (a web-based program). There were 334 papers screened for titles and abstracts, as well as records that did not include any of the search criteria in their title, abstract, or keywords, as well as research conducted outside of Tanzania and those deemed irreverent. As a result, 162 records were chosen for the subsequent stages. The eligibility phase entailed assessing a range of article attributes in order to choose the most relevant articles for further research. Records that were not concerning forest degradation did not have a PDF, or depended primarily on secondary data were excluded. Finally, an in-depth review of 71 papers was carried out. **Table 2** outlines the inclusion criteria that were considered.

Keywords	Area restriction
Forest OR "Forest degradation" OR methods OR "forest quality" OR "qualitative OR quantity" OR quantitative OR "forest health" OR "land degradation" OR species OR tree species; animal species OR "forest species" OR "plant species" OR diversity OR "forest diversity" OR "species diversity" OR "plant diversity" OR "animal diversity" OR " tree diversity OR forest harvesting OR "forest inventory" OR "remote sensing" OR "forest biomass" OR "tree biomass" OR "biomass assessment" OR extent OR factors OR Drivers OR "socioeconomic drivers" OR "financial resources" OR " human resources" OR "physical resources" OR transportation OR infrastructure OR biophysical drivers OR "carbon stock" OR REDD+ OR "carbon sequestration " OR "firewood extraction" OR "mining in forests" OR "forest grazing " OR livestock OR "loss of biodiversity" OR "forest biodiversity" OR "species biodiversity" OR "plant biodiversity" OR "animal biodiversity"	AND (Tanzania)

Table 1.
Keywords used in the Scopus and Google scholar database to search published articles.

Inclusion criteria	Exclusion criteria
Text in English	Text in other languages other than English
Research articles, or book chapter	Publication type is other than article, review, or book chapter
Addressing forest degradation and deforestation	Not addressing forest degradation and deforestation
Study done in Tanzania	Study done outside Tanzania
Time frame: 1985 to 2022	Before 1985
Study includes primary data	Study includes secondary data

Table 2.
Inclusion and exclusion criteria.

2.2 Data analysis

Data analysis was performed by examining the frequency of mention of a research term either as keywords in titles, abstracts, or in the whole document. The analysis was done in VOSviewer software supplemented by analysis using Zotero and MS Excel. Results were summarized as a VOSviewer item map, tables, and graphs showing the frequency of mention of a research term or administrative region of Tanzania (**Figure 1**).

3. Results and discussion

3.1 Trend in research on forest degradation in Tanzania

There were more studies more recently, especially increasing sharply after the year 2000 (**Figure 2**). However, there is also an obvious pattern of decline in the number of studies after peaking in 2016 and 2017. The increasing interest in forest degradation may be related to more recognition of forest degradation as an important aspect of forest management, especially in comparison to deforestation, which

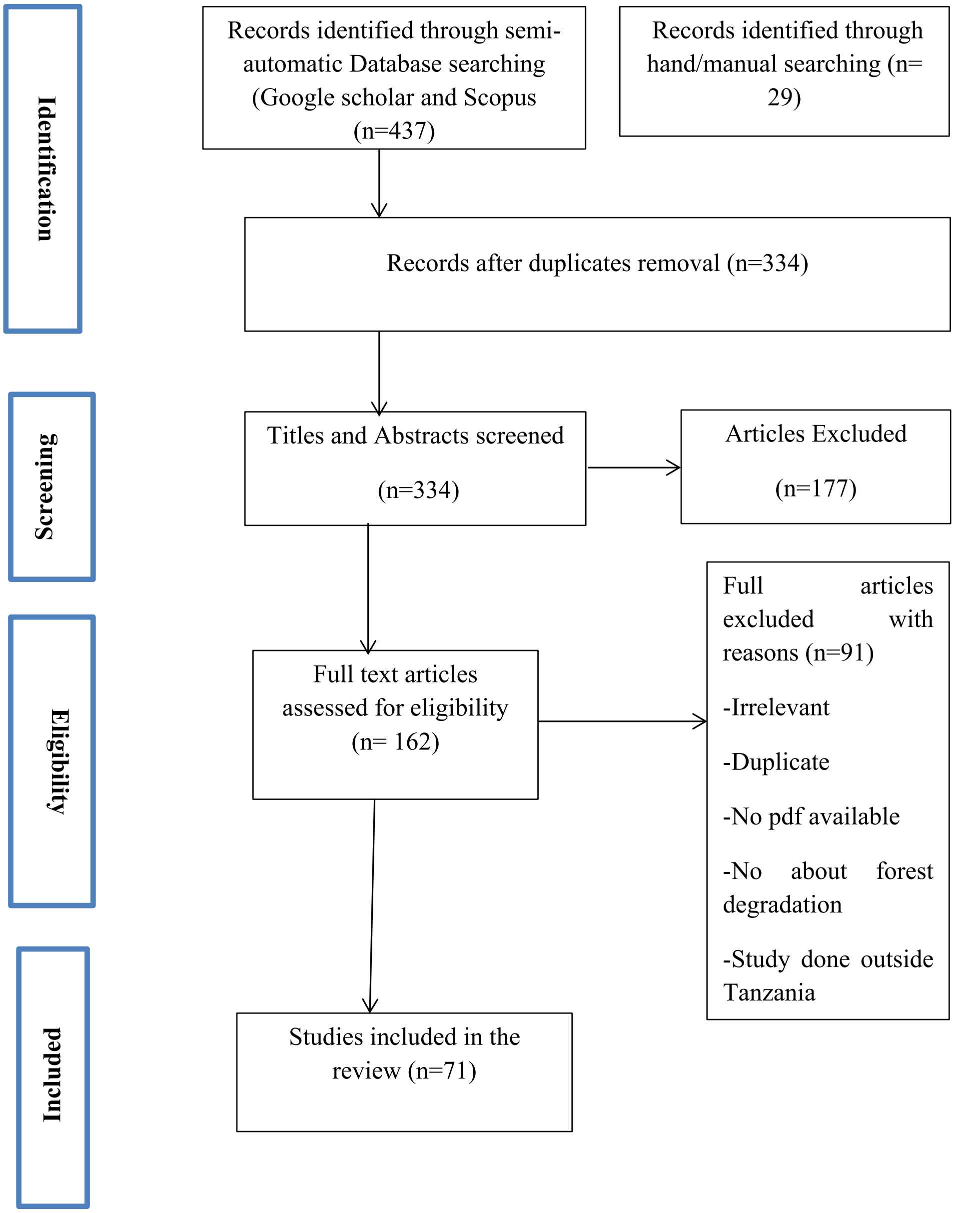

Figure 1.
The PRISMA framework used in article screening.

received and continues to receive more attention than forest degradation. Even REDD+ began initially as RED and then became REDD and finally REDD+ as forest degradation represented by the second D and its correlates represented by the + in REDD+ became more appreciated [11]. The increase in a number of publications toward 2016 and the decline after 2017 may also be related to a big research program that funded many projects related to forest management in general and forest degradation in one way or another. This was the Climate Change Impacts, Adaptation and Mitigation (CCIAM) program between 2009 and 2014. Before and after this program

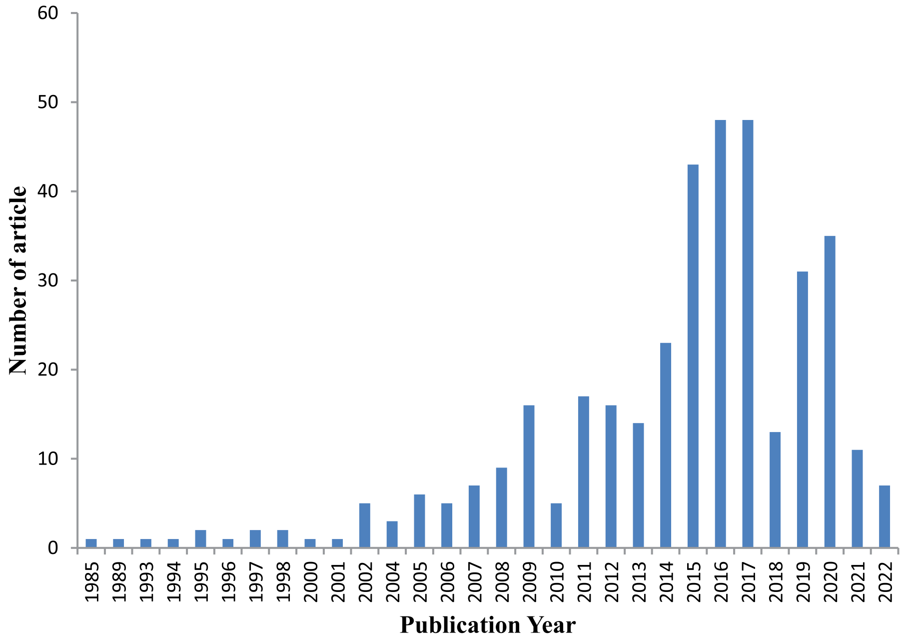

Figure 2.
Trend in research in forest degradation in Tanzania indicated by a number of publications per year.

the fewer number of publications may indicate the significance of a focused research program that enhances the number of country-specific projects that get funded. This is unlike the case where researchers have to compete for funds from international research funding baskets.

3.2 Research topics covered and gaps in research in forest degradation in Tanzania

The most frequent keywords taken from titles and abstracts were Tanzania, trees, and REDD+ (**Figure 3**). Forest transition model, drivers, and monitoring were among the least frequent keywords (**Figure 3**). Eight clusters were formed by the mapping of keyword occurrences (**Figure 3**). These keyword clusters appear to be based on numerical correlations without a meaningful interpretation of research themes. Such research themes could have been, for instance, research study area, method of data collection, and method of data analysis. However, from the item map terms related to methods, such as monitoring, spatial analysis, and forest transition model, are in three different clusters (**Figure 3**).

Most terms are mentioned less frequently in the document title than in the document as a whole (**Table 3**). This indicates that more studies are general on forest degradation than specific on the research terms. This represents research gaps for the research topics represented by the research terms. The most covered research term is human, while the least covered is biophysical (**Table 3**). Some terms are covered less than expected, given their significance in forest degradation. This includes the term mining. The number of studies one may find on a research topic may depend on the term used to search the studies. For example, agriculture returns more research documents than cultivation. In some cases, this is affected by small differences in the

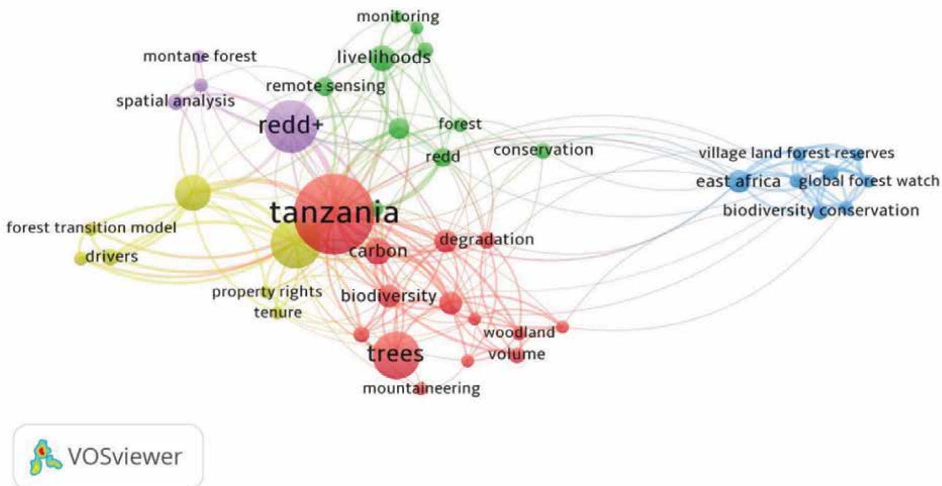

Figure 3.
VOSviewer item map showing keywords that were frequently used to describe clusters in research studies on forest degradation in Tanzania. Larger circles represent a higher frequency of occurrence. Circles in the same color are in the same cluster and have a higher statistical similarity than others. No meaningful research themes could be assigned to the clusters.

spelling out of the research term. For example, spelling socioeconomic returns only 13 documents, whereas spelling socioeconomic returns 43 documents, a difference of 42% of the total number of documents (**Table 3**).

3.3 Extent of forest degradation in Tanzania

One study estimated the extent of forest degradation in miombo woodlands for the whole country [12]. Miombo woodlands represent more than 90% of forest cover in Tanzania, and hence, may give a country-wide picture of the extent of forest degradation. On the basis of that study, annual volumes, aboveground biomass removed, and belowground biomass removed were 1.71 ± 0.54$m^3 ha^{-1}$ $year^{-1}$, 1.23 ± 0.37 t ha^{-1} $year^{-1}$, and 0.43 ± 0.12 t ha^{-1} $year^{-1}$, respectively. The corresponding aboveground and belowground carbon removed were found to be 0.6 ± 0.18 tC ha^{-1} $year^{-1}$ and 0.21 ± 0.05 tC ha^{-1} $year^{-1}$, respectively. The estimated annual volume removals exceed the estimated mean annual increment of 1.6 ± 0.2 m^3 ha^{-1} $year^{-1}$ in miombo woodlands. This indicates forest management in Tanzania is not sustainable.

3.4 Drivers of forest degradation in Tanzania

Various studies have found a variety of causes of forest degradation, including deforestation and forest degradation, agricultural expansion, wood extraction and settlement area agriculture, fuel wood production, unsustainable timber extraction, and pasture expansion [4, 13, 14]. Demand for land and forest resources, as well as the combination of social, political, cultural, and technological variables,

Research term	Number of mentions in title	Number of mentions in all document	Percent of documents mentioning the research term	Research term	Number of mentions in title	Number of mentions in all document	Percent of documents mentioning the research term
Biophysical	0	8	11	biomass	2	30	42
Quantity	0	10	14	carbon	0	31	44
Qualitative	0	11	15	REDD+	7	31	44
Sequestration	0	12	17	REDD	8	31	44
Fuelwood	1	12	17	health	0	32	45
Mining	0	12	17	harvesting	0	32	45
Socioeconomic	1	13	18	forest diversity	0	33	46
Infrastructure	0	13	18	species diversity	0	33	46
Forest inventory	0	14	20	tree diversity	0	33	46
Inventory	0	15	21	fire	1	33	46
Quantitative	0	18	25	extent	1	35	49
Remote sensing	1	18	25	social	0	35	49
Physical	0	20	28	land degradation	0	39	55
Firewood	0	22	31	plant	2	39	55
Cultivation	0	24	34	animal	3	40	56
Financial	0	24	34	diversity	0	40	56
Grazing	0	24	34	Degradation	7	41	58
livestock	0	24	34	economic	2	42	59
Transport	0	25	35	socioeconomic	1	43	61
Quality	0	26	37	Agriculture	0	45	63
Fuel	2	27	38	factors	2	45	63
Plant diversity	0	28	39	Tree	4	49	69

Research term	Number of mentions in title	Number of mentions in all document	Percent of documents mentioning the research term	Research term	Number of mentions in title	Number of mentions in all document	Percent of documents mentioning the research term
Tree biomass	1	28	39	Forest	9	51	72
Drivers	4	28	39	Biodiversity	3	51	72
animal diversity	0	29	41	human	3	52	73
Forest biomass	0	30	42				

Table 3.
Research topics covered and gaps in research on forest degradation are indicated by number and percent of documents mentioning a research term.

can all contribute to deforestation and forest degradation [4]. Specifically on forest degradation due to stem removals, eleven drivers were identified, namely, forest fires, firewood collection, grazing by wildlife, domesticated animals, carving, poles, shifting cultivation, timber, and mining activities [15]. A higher number of stems/ha/year were removed by shifting cultivation, followed by charcoal, natural death, firewood collection, and poles. In terms of carbon, however, higher above-ground carbon removals were caused by timber followed by fire, shifting cultivation, charcoal, and natural death.

When agriculture leads to the permanent conversion of forest into agricultural land use that is deforestation. However, when there is shifting cultivation within a landscape that is defined primarily as a forest, that is, forest degradation. Forest degradation is also defined when there is agriculture under the forest canopy. This is practiced for some crops that are shade-tolerant or even shade-demanding, such as some types of spice crops. Agriculture is the primary cause of deforestation and forest degradation worldwide, accounting for over 80% of total deforestation in developing nations [2]. Agricultural expansion in form of shifting cultivation and encroachment, which is characterized by a low level of technology and o low agricultural yield [16]. A variety of factors, including cultivation techniques, soil fertility, market potential, population expansion, and perhaps pertinent policies, will influence the length of the fallow and cultivation periods [17]. When the fallow period is shortened for shifting cultivation, carbon stocks are degraded. Agricultural expansion rather than intensification was due to inadequate farming technology, low productivity that resulted from it, and low returns on inputs [18].

Fuelwood and charcoal production are major contributors to forest degradation in Tanzania. This is owing to the fact that the primary source of energy is in the majority of Tanzanian households. Wood biomass is collected for use as fuel or charcoal for both home and commercial purposes. The extraction of adequate fuelwood stocks to damage the forest is a significant aspect. Bailis et al. [19] created a geographical supply and demand technique to quantify greenhouse gas emissions associated with wood harvesting for fuel consumption. Woodfuel Integrated Supply/Demand Overview Mapping (WISDOM). Their findings revealed that the distance to the road had a substantial impact on the occurrence of stumps cut for charcoal. According to field observations and interviews, the geology of the area (slope, soil, presence of stones, and availability of drinking water) and preferred species are important variables in determining the location of charcoal production.

Selective timber harvesting can degrade forest carbon stocks, especially in humid tropical forests [7]. Biomass is lost during harvesting operations due to a number of factors, including wood tree felling and damage to trees surrounding the felled trees. There is both illegal and legitimate timber harvesting. Biomass loss has the potential to deplete humid tropical forest carbon stocks [17].

The influence of fire on forests is multifaceted; fires might be ground fires that burn smaller trees and understory or major stand-replacing fires. Fires reduce the rate of forest regrowth and succession from grassland to forest, resulting in continuous grassland and bushland [17].

When grazing animals are allowed on forest land, they both browse and trample young and regenerating trees, killing or damaging them. This causes forest degradation because young seedlings do not survive, and tree girdling causes the eventual death of larger trees (**Table 4**).

Region	Forest degradation driver
Iringa, Morogoro, Tabora	Agriculture (Crop and livestock production)
Shinyanga, Singida	Farming (food crops and cash crops), firewood
Lindi, Mtwara, Pwani	Logging, charcoal
Iringa, Morogoro, Tanga	Illegal logging, fire
Kagera, Mwanza	Farming (food crops and cash crops) subsistence (food crops production), charcoal
Manyara, Morogoro, Tabora	Charcoal

Adapted from [20].

Table 4.
Main regions for forest degradation in Tanzania.

3.5 Methods of monitoring and assessment of forest degradation

The effects of forest degradation are expected to vary depending on location, forest type, and degree of degradation, making it challenging to detect with medium resolution remote sensing, such as Landsat. There are two types of accounting: land-based and activity-based. Activity-based accounting analyzes emissions independently for each activity and evaluates numerous human activities that cause forest degradation. Regardless of the activities that occur, land-based accounting determines the change in carbon stocks in a specific area of land [17].

Estimates of subsistence wood extraction may be obtained using indirect remote sensing methods. Biomass sampling yields zero-inflated continuous data that challenges conventional statistical approaches. To predict biomass loss as a function of distance to the nearest settlement, Dons et al. [21] employed Tweedie Compound Poisson distributions from the exponential dispersion family in conjunction with GLM. Estimating removals is commonly done by measuring the diameter at breast height (D). The calculated D is then used to calculate biomass and volume using allometric formulae used SD to generate equations for calculating volume, aboveground biomass, and belowground biomass in Tanzanian miombo woods. They analyzed land use and cover maps over a 15-year period by dividing the estimated individual tree volume by the estimated age of the stump.

Distance to populations, urban centers, and roadways are examples of proxy variables that can be used to measure forest degradation [13]. Spatial models that imply a relationship between forest degradation and distance from populated areas, highways, and the forest boundary can be a valuable tool for determining the degree of forest deterioration. In this case, spatial models that imply a relationship between forest degradation and distance from populated areas, highways, and the forest boundary can be a useful tool for estimating the degree of forest deterioration [13]. Focusing on remotely sensed deforestation might miss significant declines in forest quality. Ahrends et al. [8] recommend the use of fast field assessments in conjunction with remote sensing to provide early warning and to allow for prompt and adequately focused conservation and policy responses.

3.6 Institutions involved in efforts to control forest degradation

To properly manage forest resources, it is not sufficient to comprehend different tree species, their regeneration strategies and extraction techniques and rates, for instance. Institutions must be involved in determining who has access to environmental resources, how much can be extracted, when, and how. Tanzania has made major advancements in the management of its forest resources since the early 1990s. Community-based forest management (CBFM) and joint forest management have been implemented as part of the phases (JFM). The two strategies are collectively known as participatory forest management (PFM) [22].

Community-based forest management (CBFM) in Tanzania has contributed to improving forest conditions and in some cases improving livelihoods. PFM has been demonstrated to reinforce forest bureaucrats' and other specialists' dominance at the expense of local autonomy and decision-making, which was both a normative objective in and of itself and a major premise underlying its promises of enhancing local livelihoods and forest conservation [23]. Tanzania's Forest Act of 2002 transferred forest resource ownership and management responsibilities to local communities. The objectives of community-based forest management (CBFM) include enhancing rural livelihoods, protecting and regenerating forest resources, and fostering good governance. State-owned forests on the reserved property are managed jointly by community organizations that rely on the forests for their livelihood [23].

3.7 Spatial distribution of studies on forest degradation in Tanzania

The most mentioned region was Dar es Salaam, while the list mentioned was Rukwa when the names of the regions were searched from the whole document (**Figure 4**). However, when the names of the regions were searched only from the titles of the documents, only Kigoma had two mentions. The other mentioned regions were only mentioned once, while the rest had no mention in the title. The spatial distribution of studies on forest degradation may be influenced by a number of factors pertaining to a region, including existing forests and their characteristics, drivers of forest degradation, and proximity and accessibility of the region for research.

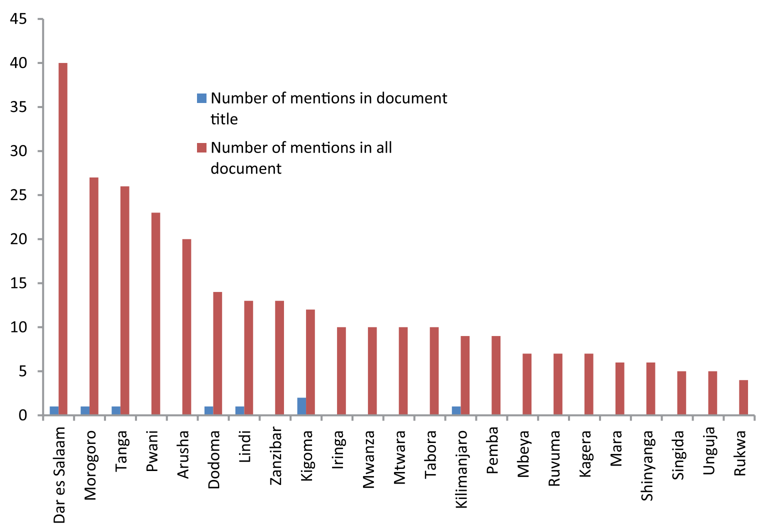

Figure 4.
The spatial distribution of studies on forest degradation in Tanzania is indicated by number of times a region is mentioned in the title or the number of documents mentioning the region.

4. Conclusions and recommendations

There were more studies more recently, especially increasing sharply after the year 2000, peaking in 2016 and 2017, and then declining. The increasing interest in forest degradation may be related to more recognition of forest degradation as an important aspect of forest management, especially in comparison to deforestation, which received and continues to receive more attention than forest degradation. The increase in the number of publications toward 2016 and the decline after 2017 may also be related to a big research program that funded many projects related to forest management in general and forest degradation in one way or another. Before and after this program the fewer number of publications may indicate the significance of a focused research program that enhances the number of country-specific projects that get funded. The most frequent keywords taken from titles and abstracts were Tanzania, trees, and REDD+. Forest transition model, drivers, and monitoring were among the least frequent keywords. Keyword clusters formed by VOSviewer appear to be based on numerical correlations without meaningful interpretation of research themes.

Most terms are mentioned less frequently in the document title than in the document as a whole. This indicates that more studies are general on forest degradation than specific on the research terms. This represents research gaps for the research topics represented by the research terms. Some terms are covered less than expected, given their significance in forest degradation. The number of studies one may find on a research topic may depend on the term used to search the studies. For example, agriculture returns more research documents than cultivation. In some cases, this is affected by small differences in the spelling out of the research term. The estimated annual volume removals exceed the estimated mean annual increment indicating forest management in Tanzania is not sustainable. Various studies have found a variety of causes of forest degradation, including deforestation and forest degradation, agricultural expansion, wood extraction and settlement area agriculture, fuel wood production, unsustainable timber extraction, and pasture expansion. The effects of forest degradation are expected to vary depending on location, forest type, and degree of degradation, making it challenging to detect with medium resolution remote sensing, such as Landsat. There are two types of accounting: land-based and activity-based. Activity-based accounting analyses emissions independently for each activity and evaluates numerous human activities that cause forest degradation. Regardless of the activities that occur, land-based accounting determines the change in carbon stocks in a specific area of land. Tanzania has made major advancements in the management of its forest resources since the early 1990s. Community-based forest management (CBFM) and joint forest management (JFM) have been implemented. The most mentioned region was Dar es Salaam, while the list mentioned was Rukwa when the names of the regions were searched from the whole document. However, when the names of the regions were searched only from the titles of the documents, only Kigoma had two mentions. The other mentioned regions were only mentioned once, while the rest had no mention in the title. It is expected that forest stakeholders will find the analysis presented in this study useful. Furthermore, the stakeholders will find interest in addressing temporal, spatial, and thematic research gaps highlighted by this chapter. Temporally, the declining number of publications reported needs to be addressed. Spatially, some administrative regions are underrepresented in the literature. Thematically, more specific research on topics related to forest degradation needs to be carried out.

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Section 3

Impacts of Human Landuse

Chapter 5

Degradation of Forest Reserves in Asunafo Forest District, Ghana

Kenneth Peprah, Raymond Aabeyir and Gervase Kuuwaabong

Abstract

Forest reserve degradation is a global concern because it is a storage facility of global biodiversity. In addition, forest reserves contain the wealth of several poor countries, particularly in Africa. Such is the situation in Ghana, which possesses portions of the tropical African rainforest. The timber species thereof has been harvested to create wealth since the 1800s. The wealth of the soils for cocoa production was realised in the first decade of the nineteenth century in Asunafo. Hence, the desire to reserve portions of the forest as protected areas began in 1910. Therefore, the aim of this study is an investigation of the degradation of the forest reserves of Asunafo. The methods of the study include a survey of farmers, key informant interviews, community meetings, and transect drives. The results reveal a progressive increase in the human population, expansion of settlements, and a drastic reduction in the forest reserves by –24.59%. The timber industry, cocoa farming, and population increase have caused forest reserve degradation, a loss of wildlife habitats, an out-migration of elephants, buffalos, and chimpanzees, a loss of plant biodiversity, and an invasion by weeds. The Government of Ghana should increase efforts to halt forest reserve degradation.

Keywords: degradation, forest reserve, close forest, Asunafo, Ghana

1. Introduction

The theme of this chapter is to provide a warning signal that forest reserves have also come under attack by human-induced changes resulting in forest degradation. This situation is threatening because the forest reserves in question form part of Africa's tropical rainforest. Hence, the issues of global change in the twenty-first century equally apply to forest reserve degradation. Factors responsible for these changes are global warming and the associated abnormal water shortages [drought and dry spells], land use changes, biological invasion, loss of biodiversity, changes in gas emissions, and ozone layer changes [1, 2]. These changes are experienced globally or happen in isolated locations but are ubiquitous in spread to constitute a global change [1]. This chapter places in context, the general factors driving global change in forest degradation. These include increases in population, nation states' strife to achieve economic development, and the use of resources amid technology application [3]. The problem of forest reserves, particularly, the ones located in the African tropical rainforest, is the rapidity of the global change. Steffen et al. [4] put global change into an environmental context in which the Earth's biosphere is lively but is rapidly affected by the consequences of human activities, which threaten the very sustainability of the biosphere on which the survival of humans depends. Generally, global change and global environmental change are often used interchangeably beginning with Earth systems or planetary-scale changes [5]. Pyhälä et al. [5] argue that global change research often uses a top-down approach (global model), but the bottom-up approach (local to global) is more desirable because of the use of primary data and remedies that occur at all levels.

Within the biosphere, the basic unit of analysis is an ecosystem [6]. Besides the constant flux in the ecosystem, and temporary or progressive change, the impact of global change on the ecosystem is undeniable [7, 8]. The consequential impacts on forest ecosystems are the widespread forest degradation in discrete locations constituting global-scale forest ecosystem adverse change [1, 7]. Wildfires, diseases, invasive species, insect pests, and extreme climatic events are degrading global forests [9]. These degradation factors have impeded efforts at enhancing biodiversity conservation and food and fiber security [10]. It must be added that portions of global-scale degraded forests have gone beyond recovery levels (forest loss) [11].

In terms of forest reserve degradation, some gaps still exist in the literature. However, the literature indicates a common methodology of forest reserve degradation research as the use of remote sensing and geographic information systems. Even though the methods of research are similar different results often come out revealing striking additions to knowledge. Akingbogun et al. [12] used remote sensing techniques and geographic information systems analysis to reveal that forest reserves found closer to major African cities suffer from pressures of population increase and human abuse. These factors are converting large portions of the reserves into bare ground. Using similar methods of research, Mutesi et al. [13] showed human factors such as population increase, poverty, and harvesting of forest reserve resources in rural areas have degraded forest reserves. The reserve has substantially changed to shrub land. Pacheco-Angulo et al. [14] indicated that deforestation and forest degradation are abusive to forest reserves and that the former releases a lot more carbon dioxide emissions than the latter. Deforestation and forest degradation are primarily driven by charcoal production, firewood collection, and logging. The rest of the driving forces are infrastructure development and crop farming. These factors exacerbate climate change impacts, land degradation, and biodiversity loss [15]. Even though Orimoogunje [16] also revealed deforestation and forest degradation, the study found forest regeneration propelled by local community control and enforcement of rules. Again, the degradation was one in which original species of the forest were being replaced by species tolerant of the factors of global change. Maghah and Fokeng [17] added to the degradation factors eucalyptus colonization and livestock grazing in the forest reserve leading to forest loss. The migration of large herds of cattle to settle in areas near forest reserves is an increasing phenomenon in West Africa [17]. Sinasson et al. [18] reveal the harm of forest degradation on non-timber forest resources; that, their regeneration is seriously decreasing. It is apparent from the above that grass invasion, loss of prominent plant species, and out-migration of some keystone animal species, as aspects of forest reserve degradation constitute gaps in the literature that warrant further research; and this study reports in this chapter on catering of the gaps.

2. Methodology

2.1 Theoretical underpinnings

The Malthusian theory argues that "the power of population is indefinitely greater than the power in the earth to produce subsistence for man" ([19], p. 13). The disequilibrium between population growth and food supply would lead to unsustainable exploitation of land resources and eventually cause forest degradation. This would inevitably affect the political economy, family life, prosperity, good citizenry, and marriage life. Subsequently, living standards, disease, famine, infant mortality, mob action, and political dictatorship would increase resulting in misery, vice, and moral restraint ([20], pp. 5–19). Could population growth be an aggravating factor in the process of forest degradation at Asunafo? The Malthusian theory becomes useful in addressing this question. On the contrary, Boserup [21] posits that instead of environmental degradation, the population increase would bring about sustainable environmental management through agricultural intensification and a rise in cropping frequency. This would be occasioned by increased technology, which would help to expand productivity to cater for the needs of the increased population. Consequently, productivity would increase. As this happens, market prices, substitutes, inventions, and government policies would control environmental degradation. However, the failure of the market economy and policies to efficiently manage, develop, and allocate resources would exacerbate environmental degradation.

2.2 Data sourcing and analysis

Data sourcing began with desk study through literature review, computer generation of study sites, and secondary data gathering. Specific secondary data include the following: historical records (1700–2000) on Ahafo District; rainfall, humidity, and temperature for Goaso synoptic station, and satellite images of Asunafo for 1986, 2003, and 2020 [22–24]. The desk research was followed by a 4-day drive across the districts. The drives included trips from:

- Goaso through Ayomso, Ayum, and Bonkoni Forest Reserves to Dominase.
- Goaso – Dantano – Akrodie – Ayum, and Subim Forest Reserves – Asummura.
- Goaso through Mim, Bediako, and Dominase, and ended at Kasapin.
- Goaso through Kukuom, Nobeko, Kwapong, Sankore, and Camp No. 1 to Abuom.

The next step was the verification of the selected 21 case study communities. The selection process was informed by Ghana Statistical Service's choice of the 20 largest communities in each district in the country ([25], p. 54). However, Goaso, Kukuom, Mim, Kwapong, and Sankore were replaced by nearby communities because of lack of logistics to cater for such large towns.

Subsequently, 21 community meetings were organized early in the morning for farmers in each community. Attendance of the meetings was recorded in a field notebook, and the proceedings were recorded on audio cassettes. About eight (8) farmers attended the smallest meeting, while 81 farmers attended the largest meeting. The total number of farmers who attended the 21 meetings was 774, and they formed the population frame for the study. A sample size of 264 farmers was selected based on a formula: n = N/1 + N(e)2. About 15 research assistants were drawn from third and fourth-year undergraduate students to assist in the data gathering through the administration of questionnaires, key informant interviews, and personal observation. The rest were transect drives, community meetings, field visits, and photography. Archival data on Ahafo District were provided by the Regional Public Records and Archives Administration (PRAAD) of the Brong Ahafo Region. The study categorizes the period for a considerable outcome into three, namely: short-term, medium-term, and long-term. The short-term deals with a period of fewer than 2 years, the medium-term ranges between 2 and 6 years, and the long-term considers more than 6 years. Data analysis involved the use of ENVI 4.3, ArcGIS 9.3, SPSS, and Microsoft Excel.

3. Results and discussion

3.1 Sociodemographics of the Asunafo area

According to the 1960 population census of Ghana, about 48,043 people occupied Asunafo. The 1970 population was 82,275, that is, about a 71% increase between 1960 and 1970. There was a 49% increase in population between 1970 and 1984 to 122,585. The population again increased by 43% between 1984 and 2000 to 174,026 ([26], p. xx; [27], p. 8; [28], p. 29). Currently, the population of Asunafo South District is 91,693 and that of Asunafo North Municipal District is 150,198. The total of Asunafo is 241,891 from the 2021 population and housing census (http://asunafonorth.ghanadistricts.gov.gh/).

In terms of human occupancy of the Asunafo area, the indigenes were Ahafo. The Ahafo and other migrated Akan people constitute about 68.8% of the population. The rest of the people are from the following ethnic groups: Mole-Dagbani (13.8%), Ewe (5%), Ga-Dangme (3.3), Gurma (3.1%), Guan (1.8%), Grusi (1.6%), Mende (1.5%), and foreigners (1.1%). The largest percentage of the people (86%) depend on fuel wood for cooking, while 9.3% use charcoal. Apart from these fuel sources, kerosene (1.1%), gas (0.6%), electricity (0.5%), coconut husk/maize stock (0.1%), others (0.1%), and none/no cooking (2.2%) have also been recorded ([29], p. 30, 55).

The two dominant economic activities at Asunafo are farming and logging. These activities have long histories. Farming history began in 1900 and timber activities in 1947 ([30]: 167). Individual smallholder farmers normally crop an average of 1.75 acres for food and 5 acres for cocoa ([31], p. 15). Many farmers grow cocoa while a few add oil palm and citrus. Plantain (*Musa ABB*) is the most important subsistence food crop. Large timber firms with huge export capacities have also carried out timber extraction. Notable ones include Mim Timber Company and Gliksten West Africa Limited. Competition between large and small timber merchants has been riffed in the area.

There are three relevant state institutions about forest reserves in Asunafo. The institutions are the Ministry of Food and Agriculture, the Ghana Cocoa Board (COCOBOD), and the Forestry Commission. The mission of the Ministry of Food and Agriculture is to offer quality agricultural services, sustainably enhance the growth and development of the sector, ensure farmer collaboration with the private sector, improve food security, and ensure the conservation of natural resources. The Ministry of Food and Agriculture, in the Brong Ahafo Region, has observed that soil fertility replenishment through natural fallow is inadequately poor in the region. This reason underlies the Ministry of Food and Agriculture's training of 51 farmers in the region on sedentary farming systems, land and water management as well as tree cover depletion minimization [32]. The Ministry of Food and Agriculture maintains a District Agricultural Office, which takes care of agricultural activities in the two Asunafo Districts. About the COCOBOD, three offices operate separately at Asunafo: Quality Control Company (QCC), Cocoa Swollen Shoot Virus Disease Control Unit (CSSVDCU), and Seed Production Unit (SPU). They are located in Goaso. The Quality Control Company supervises the work of cocoa buying companies. "The mission of the QCC is to develop and provide systematic strategies that will ensure the supply of best grade cocoa and other produce both on the local and international markets" [33]. The CSSVDCU and SPU are, however, indirectly related to land degradation control. The CSSVDCU works directly with cocoa farmers by spraying farms and providing appropriate cocoa agrochemicals. It is solely responsible for the government's free and mass cocoa spraying. The SPU develops cocoa seeds and seedlings for planting. The Forestry Commission maintains two offices at Asunafo – Wildlife Division and Forestry. Both offices are located at Goaso. Its mission is "to sustainably develop and manage Ghana's forestry and wildlife" ([34]: 1). Forestry Commission is specifically concerned with the protection and management of "permanent forest estates and protected areas in the various ecological zones of the country to conserve Ghana's biophysical heritage" ([34]: 1). It also has the responsibility of "advising and providing technical services for forest plantation for restoration of degraded forest and environmental conservation in general" ([34]: 3). The mandate of the Forestry Commission is confined to the protected areas.

3.2 Land use and land cover changes between 1986 and 2020

In this section, the focus is on the close forest (which constitutes the forest reserve). There are six (6) forest reserves, namely Abonyere, Ayum, Bia-Tano, Bonkoni, Bonsam Bepo, and Subim. They contain several economic trees. For instance, Aboniyere contains 37 timber species, Ayum 40, Bia-Tano 56, Bonkoni 43, Bosam Bepo 36, and Subim 40. **Figure 1** shows a portion of the Ayum Forest Reserve.

The information generated through community meetings and key informant interviews suggests that Asunafo used to be a dense forest. This belief is supported by historical records, which indicate that the general land cover type of Asunafo was a thick forest dotted with human settlements [35, 36], although the intensity and extent of forest cover have decreased with time. For instance, a vegetation map of Ghana produced in 1921 shows a forest land cover divided into evergreen and deciduous forest cover types. **Figures 2** and **3** show a portion of the floor of the Bonkoni Forest Reserve indicating lianes and climbers.

Figure 4 pre-dates the creation of forest reserves in 1939. The red box depicts the land cover types in Asunafo during the 1920s. The western half of the Asunafo area shows evergreen forest while the eastern side portrays deciduous forest. Detailed characteristics of the forest indicate that:

The evergreen forest in general consists of trees forming a closed canopy from 20 to 150 feet or more in height, and interlaced by innumerable wood lianes. Below, where the light is sufficient to permit it, is a mass of shrubs from a few inches to several feet

Figure 1.
A section of the Ayum Forest reserve.

Figure 2.
A portion of the floor of Bonkoni Forest reserve with lianes and climbers shown behind the researchers with dim.

Figure 3.
A portion of the floor of the Bonkoni Forest reserve with sunshine.

> *high, bound together by lesser wood lianes and herbaceous climbers and interspersed with tall herbs. As a rule, the smaller herbaceous flora is scanty but varies considerably according to the amount of light that is permitted through the canopy ([37]: 14).*

However, the deciduous forest in its original state has an upper layer of dominant trees that show a single crown. Underneath, the crowns of the middle layer trees form a closed canopy. The forest floor is made of shrubs, lianes, climbers, and herbaceous cover depending on the amount of sunshine received. Trees, especially, the dominant ones shed their leaves during the Harmattan season ([37]: 17). The community meetings revealed that tree cover has drastically reduced due to logging. As a result, grasses and *Euphorbia heterophylla* have invaded portions of the forest reserve.

In **Figure 5**, it is obvious that the close forest has been declining in size from 56.3% in 1986 to 54% in the year 2003 to 51.8% in the year 2020.

The farmers of the area have observed a decline in the close forest (forest reserves). In the community meetings, farmers described the close forest decline as reduced tree cover and replacement of the dense forest with grass and shrubs. The study has verified the claim by farmers using satellite images of different years. **Figures 6**–**8** respectively indicate land use and land cover types for 1986, 2003, and 2020 satellite images. These images were downloaded from the path/row 195/55. In the analysis, the land use/cover classification includes four classes. The land use/cover classes include: close forest (very dense canopy formed by tree crowns mainly in the forest reserves), open forest (varied from a dense bush fallow through cocoa farms with isolated tree crowns to mixed food crop farms), shrubland/grasses (made up of a variety of shrubs and grasses as well as bare ground), and bare/settlements (were human habitations either dispersed or nucleated built up surfaces).

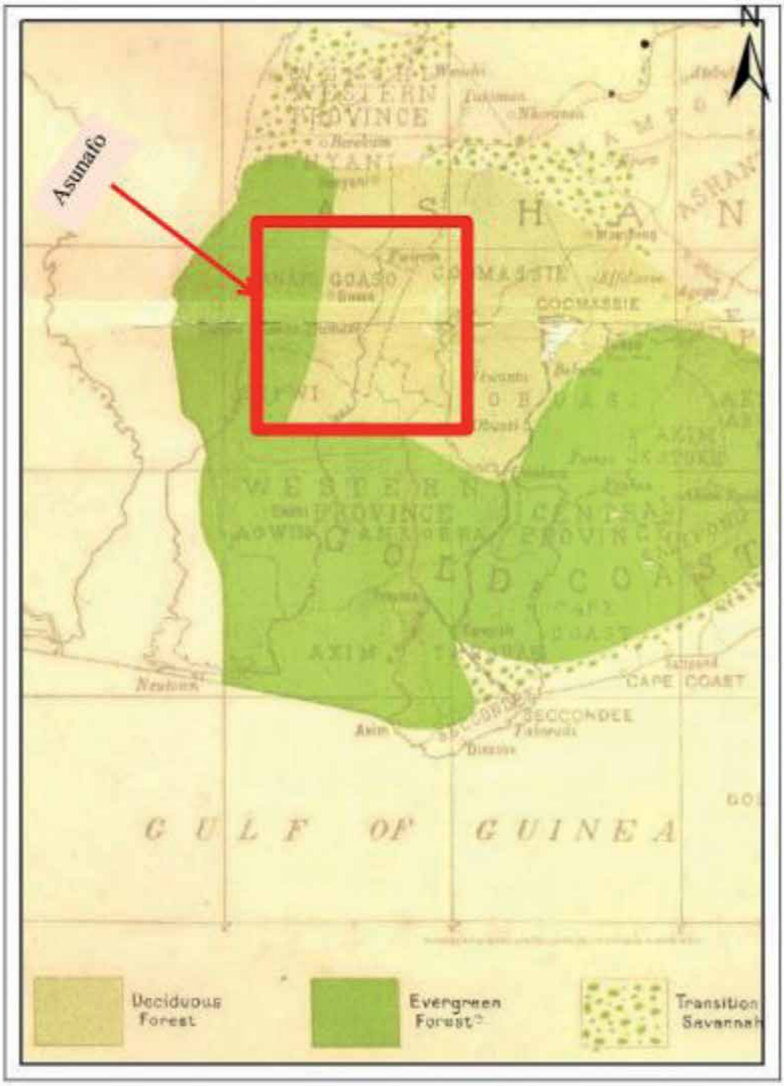

Figure 4.
Vegetation map of Ghana in 1921, Evergreen and deciduous Forest of Asunafo in the red box The citation "Forestry Commission, 2008; Potapov et el., 2017" has been changed to match the date in the reference list. Please check here and in subsequent occurrences, and correct if necessary.source: [37]: 13.

The Landsat TM 1986 shows that:

- the close forest was the largest land cover type at Asunafo in 1986, which occupied about 56.3% of the area under study.

The Landsat ETM+ 2003 shows that:

- between 1986 and 2003, the close forest remained the largest land cover type with 54% space of the Asunafo forest.

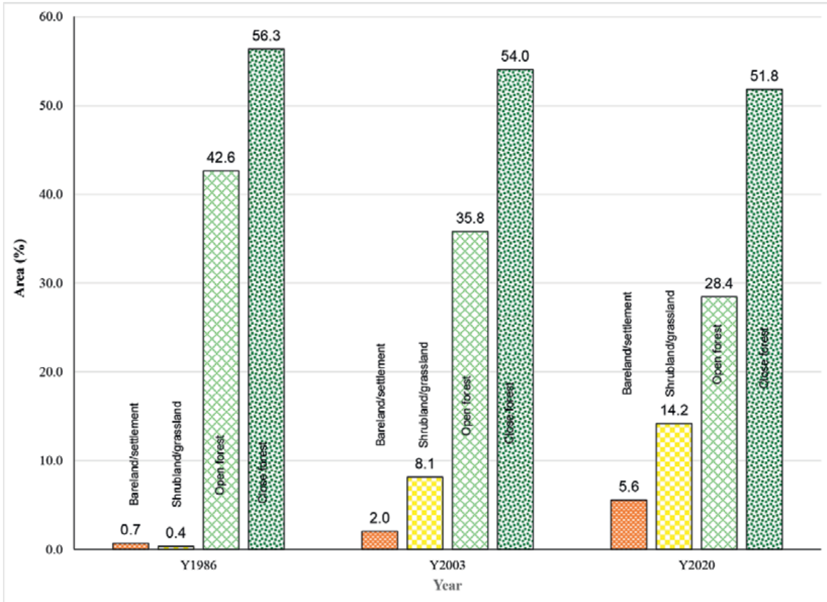

Figure 5.
Land use and land cover change between the years 1986, 2003, and 2020.

The Landsat ETM+ 2020 clearly shows that 17 years on (that is, 2003–2020):

- the close forest maintained the largest portion of the Asunafo forest with 51.8%.

Figure 9 shows a summary of the win-loss of forest space from the years 1986–2003, 2003–2020, and 1986–2020. The three scenarios indicate that bare land and settlement gain spaces from the other land use/cover in all three situations. The same applies to shrub land and grassland. However, there is a mixture of wins and losses for the open forest category. The close forest displayed a decline (degradation).

Table 1 shows the exchange of land use/cover types in the three different periods of 17 years breaks.

Between 1986 and 2003, the close forest gained a space of 14.21% but lost to the other land uses/covers a space of 36.13%. There was a difference of −21.29% of losses for the close forest. Between 2003 and 2020, the gains in forest space for the close forest were about 18.17%. Once again, there were losses of space to the other land uses/cover of about 20.84%. The difference is a negative value of about −2.67%. In the third scenario, between 1986 and 2020, the gains in forest spaces were about 10.80% and the losses were 35.39% with a negative difference of about −24.59%.

3.3 Driving forces and pressures responsible for the forest reserve degradation

From the discussion on the sociodemographics, the increases in the human population have a resultant effect on the spaces required for settlements. Hence, settlement creation is a major factor in the degradation of forest reserves. In the specific case of Asunafo, archival records indicate that the area was a virgin or dense forest inhabited by several wildlife particularly elephants, chimpanzees, and buffaloes [35, 38, 39].

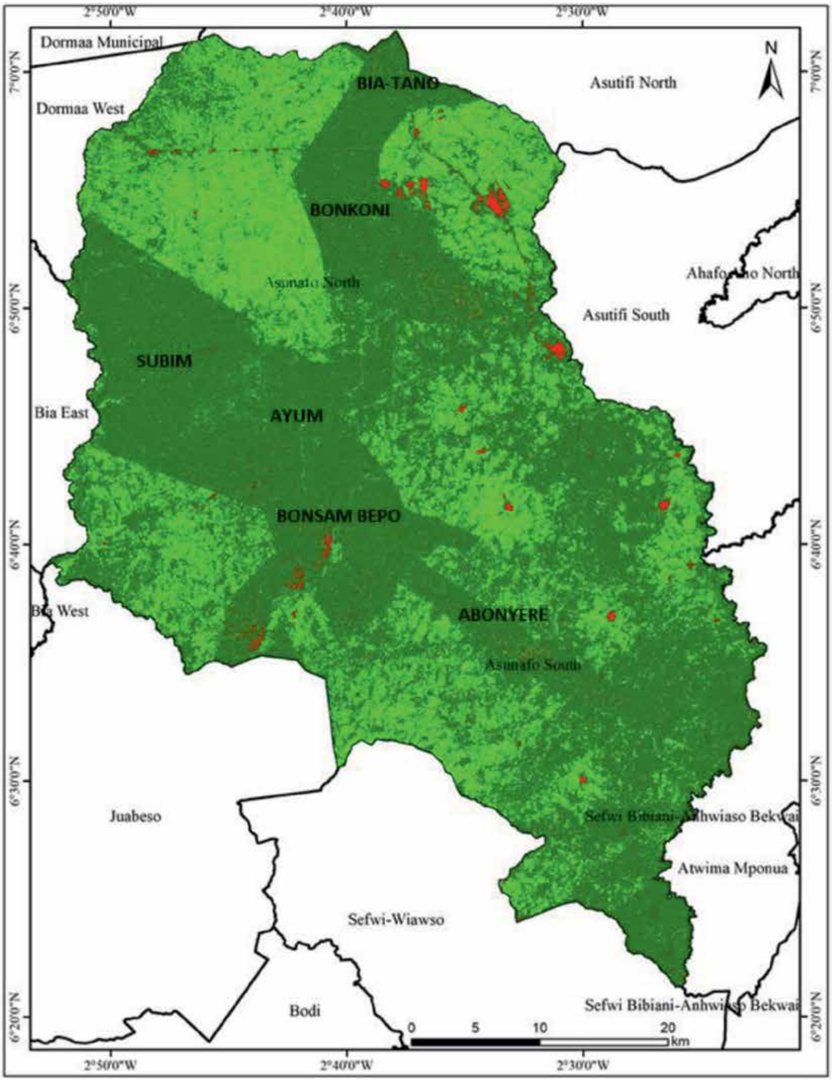

Figure 6.
Land use/cover map of Asunafo north and south for 1986.

Human impact on the dense forest was limited and restricted to settlement creation, food gathering, and hunting. However, the introduction of cocoa farming in the 1902 and export of timber industries in the 1940s have caused drastic deforestation ([30]: 167; [40]: 26). In Ghana, areas that have suffered deforestation are reportedly undergoing land degradation ([41]: 1).

Another cause of deforestation of the forest reserves in Asunafo is timber extraction. It was the timber contractors that opened the closed forest by building roads. These roads were made for use by tractors and caterpillars, and some of the roads were motorable by timber-carrying vehicles [42]. **Figure 10** is an example of a road through the Bonkoni Forest Reserve taken during the field visits.

The timber companies also cleared large portions of the forest reserves in some central places where the timber logs were assembled and later loaded on the timber-carrying trucks to be transported to the sawmills. According to Logan [43],

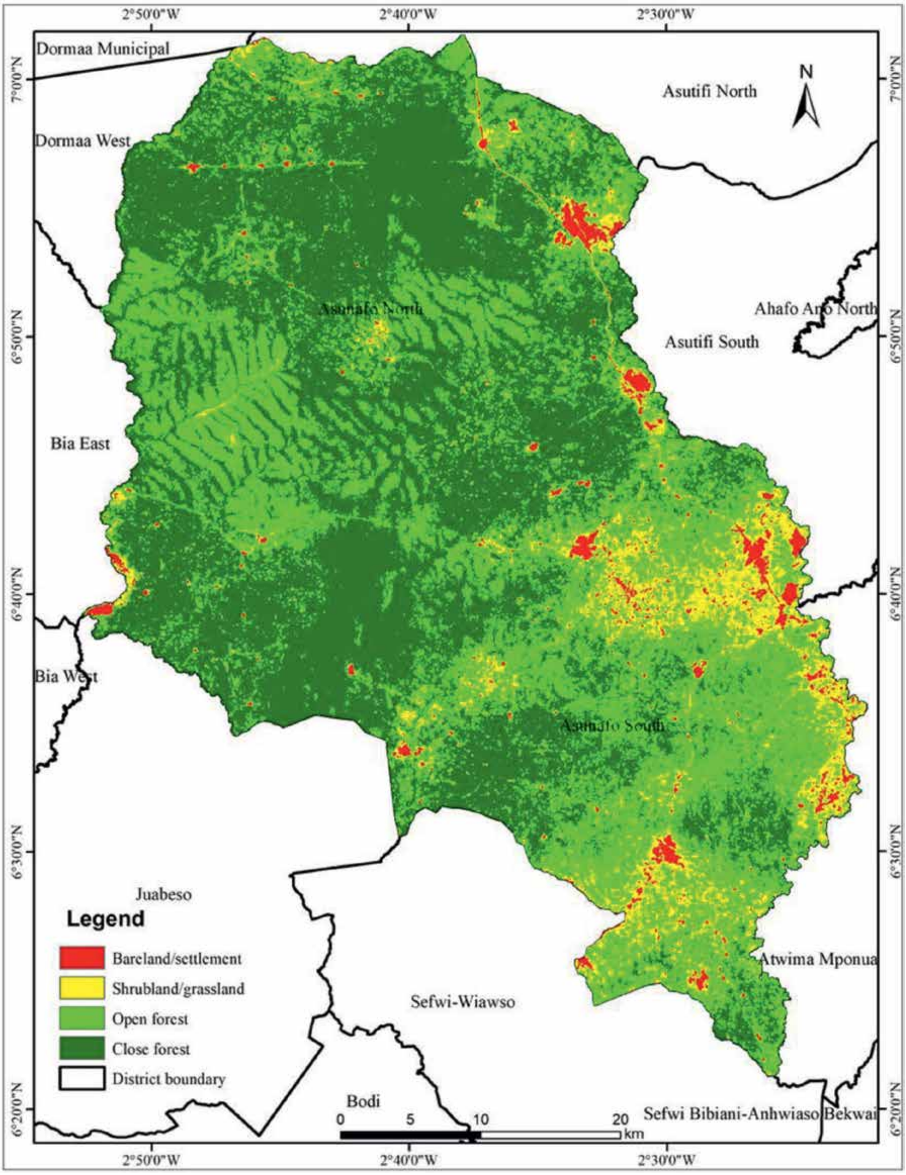

Figure 7.
Land use/cover map of Asunafo north and south for 2003.

the exports of timber logs from the Gold Coast began in the year 1888 with Mahogany. The Forestry Department of the Gold Coast (now Ghana) started in 1909. In 1911, the Forest Bill was passed by the legislative council, and in 1927, the Forest Ordinance was made law. These dates are provided to show the early attempts at conserving the close forest. However, timber merchants continued with timber extraction. By 1927, several millions of Mahogany were exported from the country and a shortage in the forest began to be felt by the Forestry Department ([43], pp. 52). After independence in the year 1957, the timber industry continued to operate within the close forest with little or insignificant difference. Two timber extraction companies that caused massive deforestation in the Asunafo area are W. A. Gliksten Limited and Mim Timber Company.

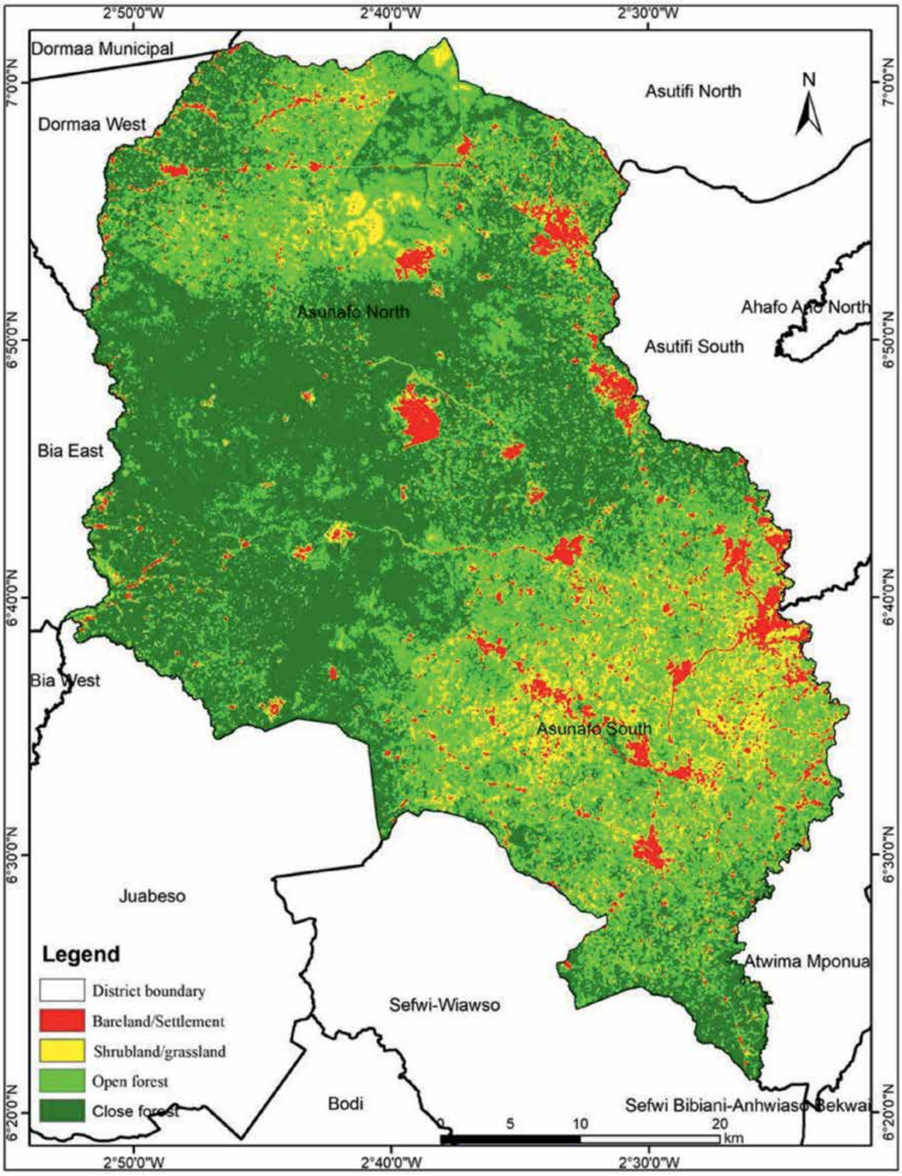

Figure 8.
Land use/cover map of Asunafo north and south for 2020.

Together, the two foreign-owned companies exported about 80% of their timber output [40, 44]. In addition to small local timber merchants, there were medium to large-scale Ghanaian-owned timber firms in Ashanti and Brong Ahafo Regions that extracted timber from Asunafo. No wonder, Asunafo Forest suffered from timber shortages as early as 1965 [45]. **Figure 11** shows a truck with timber logs from the Asunafo forest area. Previously, a truck of this nature could carry one log or a maximum of two logs due to the sizes of the logs. Presently, five or more logs are carried by the same truck size.

The human population follows several livelihood options. The main livelihood of the people of Asunafo has been cocoa farming. Clearing of the closed forest for cocoa

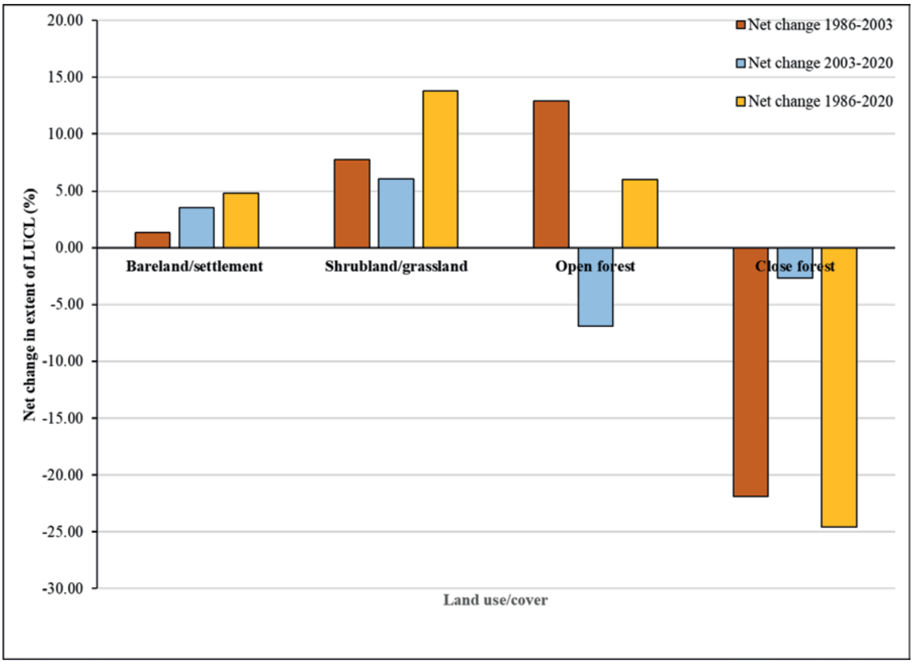

Figure 9.
Land use/cover net change (%) from 1986 to 2003, from 2003 to 2020, and from 1986 to 2020 for Asunafo north and south.

A: From the year 1986 to 2003							
Year				**2003**		**Total_1986**	**Loss_1986**
	Land use/ Land Cover	**Builtup**	**Shrub/ grassland**	**Open forest**	**Close forest**		
1986	Bareland/ settlement	0.31	0.07	0.14	0.15	0.68	0.36
	Shrubland/ grassland	0.03	0.03	0.12	0.12	0.30	0.27
	Open forest	0.48	2.08	6.38	13.94	22.88	16.50
	Close forest	1.16	5.84	29.13	40.01	76.14	36.13
	Total_2003	1.98	8.02	35.77	54.22	100.00	53.27
Gain_2003		1.67	7.99	29.39	14.21	53.27	
B: From the year 2003 to 2020							
Year				**2020**		**Total_2003**	**Loss_2003**
2003	Bareland/ settlement	1.22	0.36	0.24	0.16	1.98	0.76
	Shrubland/ grassland	1.30	2.24	2.73	1.74	8.02	5.78
	Open forest	1.65	6.22	11.64	16.27	35.77	24.13
	Close forest	1.32	5.26	14.26	33.38	54.22	20.84
	Total_2020	5.49	14.08	28.87	51.56	100.00	51.51
Gain_2020		4.27	11.84	17.23	18.17	51.51	

C: From the year 1986 to 2020							
Year		**2020**				**Total_1986**	**Loss_1986**
1986	Bareland/ settlement	0.33	0.08	0.12	0.14	0.67	0.34
	Shrubland/ grassland	0.05	0.04	0.07	0.14	0.30	0.26
	Open forest	1.27	3.44	7.64	10.52	22.88	15.24
	Close forest	3.84	10.52	21.03	40.75	76.14	35.39
Total_2020		5.49	14.08	28.87	51.56	100.00	51.22
Gain_2020		5.16	14.03	21.23	10.80	51.22	

Table 1.
Land use/cover change matrix (%) for Asunafo north and south.

Figure 10.
A new timber road through the Bonkoni Forest reserve, showing the destruction of vegetation and rich topsoil.

farming was another aggravating factor in the close forest degradation. According to Logan [43], the native Ghanaian communal land holding turned to private holdings due to the spread of cocoa farms. Before 1923, the Government had realized the threats of cocoa farming to the closed forest. The preparation of land for cocoa meant the felling of large trees to allow the sunlight to reach the cocoa plants, and it started as early as the first decade of the twentieth century [42]. The rich farmers manage relatively larger cocoa farms. The poor often maintain smaller cocoa farms and rely

Figure 11.
A truck with timber logs from the Asunafo forest area.

Figure 12.
Cocoa agroforestry at Asunafo Forest area.

solely on income from cocoa and the sale of labor on daily basis as "by-day laborers". The forest vegetation, climate, and associated edaphic features were supportive of cocoa farming. After the opening of the dense forest with accessible roads by Mim

Timber Company and W. A. Gliksten Limited in the 1940s, the transportation of dry cocoa beans and carting of other farm produce became possible. Another reason was the high-income returns generated from cocoa farming. Again, the land was owned by families; hence, industrious members of the family took to cocoa farming due to easy entry requirements such as the acquisition of simple farm implements and family labor. The farmers experienced low crop yields because of the old cocoa trees. Some of the cocoa trees from 1921 are still available and are called “Tetteh Quarshie.” **Figure 12** shows cocoa agroforestry, and **Figure 13** displays cocoa trees alone.

In the case of the Asunafo forest area, mining of gold and other minerals is still not a focus of forest degradation. Since the year 1983 wildfire, which destroyed the forest reserves, the Forestry Commission has been able to keep the forest reserve from fire outbreaks.

3.4 The consequences of the forest reserve degradation

3.4.1 Loss of wildlife habitat

The farmers of the area have observed a drastic reduction and loss of wildlife in the Asunafo Forest area. An indicator of forest reserve degradation is the absence of traces of some wildlife notably elephants and chimpanzees as well as a drastic reduction in the tortoise and snails population. In addition, forest guards have not encountered buffalos for a long time. There is a drastic reduction in the population of deer, waterbucks, antelopes, and other wildlife species. There was a deliberate attempt to remove elephants from the Asunafo forest to allow cocoa farms to flourish. By so doing, forest guards were invited from Damongo Game Reserve to kill the elephants at Asunafo. **Figure 14** displays the destruction of the elephant population for 39 years

Figure 13.
A section of a cocoa farm at Asunafo with no tree cover.

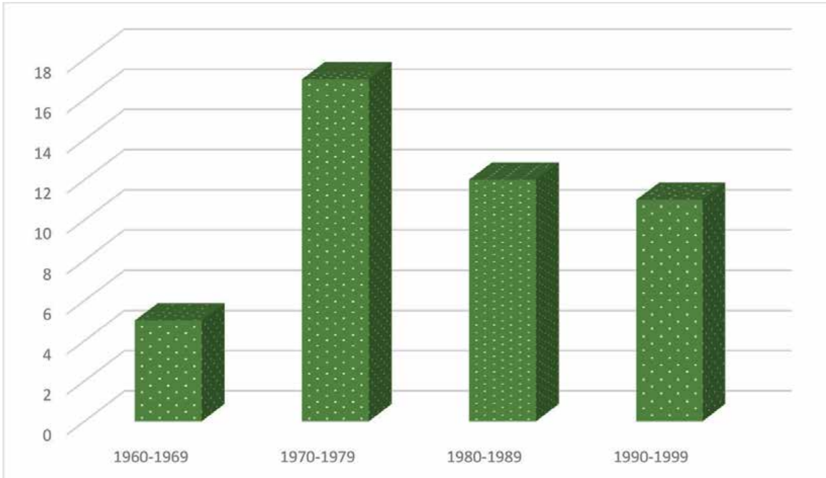

Figure 14.
Deliberate killing of elephants.

(1960–1999). Elephants and chimpanzees have migrated from the Asunafo forest due to the destruction of their habitat. Habitat destruction is an indication of forest reserve degradation. Presently, elephants and chimpanzees are no longer encountered directly or their traces are not found in the forest [34].

3.4.2 Loss of plant species

The harvesting of non-timber forest products has resulted in the dwindling of some plant species in the forest reserves such as "Anwonomo/Aworomo" (*Aumatococcus daniefii*), canes, and raphia palm. In the case of Garcinia spp., traditional chewing sticks for the cleaning of teeth, as a substitute to paste and toothbrush have gone extinct. The situation of the timber species in the forest reserves is crucial for forest degradation. It was very difficult to locate economic trees measuring 130 cm dbh (diameter at breast height) in the year 2001 in any of the forest reserves at Asunafo [34]. The forest ecosystem of Asunafo faces timber provision problems due to unsustainable timber activities.

3.4.3 Invasive species

The vehicular movement in the forest is believed to have introduced invasive species, particularly grass. There are progressive decreases in native species because of the increases in invasive species. The presence of *Panicum maximum* or Guinea grass, *Pennisetum purpureum* or elephant grass, and *Rottboellia cochinchinensis* or itchgrass is considered invasive because they are savanna species. Also, the key informant interviews and the farm visits reveal three other invasive weeds. The presence of any one of them indicates forest degradation. In addition to wildfire outbreaks, the traditional farming practices of slash and burn have supported the continuous migration and stay of such weeds. The weeds identified by farmers are *Pupalia lappacea*, *Ageratum conyzoides* (*Asteraceae*, Billy goat weed), and *Spigelia loganiaceae*. These weeds grow very fast, spread quickly, out-compete crops, and are successful invaders on native habitats [46].

4. Conclusion

The sociodemographic figures indicate a progressive population increase in the Asunafo forest area resulting from a natural increase in birth rate and overrun death rates due to increased and improvements in health care. A contributing factor is a migration to the forest area for greener pastures in the timber industry, cocoa farming, and hunting. It can be concluded from the analysis of land use/cover of the Asunafo forest area between 1986 and 2020 that the closed forest, which includes the forest reserves, has declined drastically by about −24.59%. The major causes of the depletion of the forest reserves are progressive population increases, activities of the timber industry, and cocoa farming by Ghanaians since the year 1921. The spread of wildfire during the 1982–1983 drought was a one-time contributing factor. However, mining activities for gold and other minerals play no part in the forest reserve degradation for now. The consequential timber and cocoa farming industries have negative repercussions on the forest reserves resulting in degradation. Their effects have been aggravated by the expansion and spread of human settlement around the forest reserves. For instance, wildlife habitats are lost leading to the out-migration of elephants, buffalos, and chimpanzees. In addition, several plant species are either extinct or suffered a serious reduction in productivity. A deliberate government response is required to restore the Asunafo Forest Reserves to a desirable state.

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Chapter 6

Assessment of Diversity, Growth Characteristics and Aboveground Biomass of Tree Species in Selected Urban Green Areas of Osogbo, Osun State

Omolara Aremu, Olusola O. Adetoro and Olusegun Awotoye

Abstract

This study assessed the abundance and diversity of trees, estimated the growth characteristics and determined the aboveground biomass of the trees within three selected green areas, namely Riparian Corridor was abbreviated as Riparian corridor (RC), Industrial sites (IS), and Residential sites (RS) in Osogbo, Southwestern Nigeria. Species Diversity Index, Relative Dominance, and Importance Value Index of trees were also estimated. Trees' diversity and ranking were determined using the R statistical package. A total number of 124 tree stems were enumerated and (RC), (IS), and (RS) had 49, 38, and 37 tree stems belonging to 27, 18 and 20 species respectively. *Albizia zygia* (Mimosaceae) was the most abundant species in both RC and IS, while *Milicia excelsa* (Moraceae) was the most abundant in the RS. Growth variables were recorded as 1.18 m^2, 5.01 m^2, and 11.06 m^2 (basal area), and 13.49 m^3, 64.03 m^3 and 122.39 m^3 (volume) for RC, IS, and RS, respectively. The highest mean aboveground biomass was recorded in the RS (28325.20±7639.57 Kg C ha^{-1}). There was no significant difference ($P \geq 0.01$) between the aboveground biomass of RC and IS but a significant difference ($P \geq 0.01$) existed between the aboveground biomass of RC and RS. There is a continuous transition of the urban forest.

Keywords: urban green areas, tree diversity, growth characteristics, aboveground biomass

1. Introduction

Urban green areas (green spaces) encompass all vegetation found in the urban environment, including blue spaces such as lakes or rivers and their adjacent greens and economic values [1]. The continuous changes in the natural ecosystem from disturbances, especially human activities, have been a global concern for decades. This has not only resulted in continuous reductions in its volume but has also led to a reduction in the services provided, such as water protection, flood control, air filtration and carbon sequestration [2, 3]. Also, this has positioned urban green spaces as a better option for tree species conservation, since increasing prosperity and residential density require more green infrastructure (green areas) and trees to serve amenity functions [4]. In addition to the various critical functions that urban vegetation support in the ecosystem [5–7], the lives of urban residents depend on the availability and lushness of green areas as well as the abundance and diversity of urban trees [8]. Trees are a valuable asset to the urban community [9] which occur in many different sets of species, genera, orders and families with a variety of growth forms, shapes, vegetative and reproductive characteristics, leading to their great range of diversity [10]. Despite several documentations on the great potential for preserving high tree species diversity, several cities are still experiencing loss of green areas and trees diversity, especially in developing countries as a result of rapid economic and urban growth rate [11, 12]. This results in land cover conversion into other land-use forms which affects urban forest cover as well as the functions and services they provided. [13] noted that urban areas are responsible for more than 70% of the anthropogenic release of carbon dioxide and 76% of wood used for industrial purposes. Increasing urban green biomass contributes to an increase in atmospheric carbon sequestration in the urban's terrestrial biosphere. [14] ascertained that the largest single source of CO_2 emissions comes from fossil fuel combustion, followed by industry, residential and commercial activities. Therefore, the huge release of emissions requires urgent mitigation measures.

However, the long-term ecosystem service provision by urban trees is dependent on diversity. Urban centers can also support the great number of trees diversity [15]. The connectivity of the urban forest patches in the urban landscape may be critical for maintaining the species population and diversity [16]. Trees' population and diversity can greatly be increased through human direct or indirect activities in the urban environment. These aids increase in connectivity and tree populations by planting more than one type of species in streets, parking lots, gardens and other areas within the urban centers. Species diversity is necessary for an adaptable urban ecosystem [17] which promotes resistance and resilience to disturbance, such as pests, fire outbreaks and destructions from various human activities. Therefore, the focus should be on increasing diversity to ensure ecosystem stability with more emphasis on the conservation of native species. The introduction of exotic species in urban areas can spread to neighboring rural areas as invasive plants, compete with and threaten the existence of native species and affect the ecosystem negatively by altering its processes [18, 19]. It has been predicted that current species could be at risk of extinction in the nearest future [20]. Therefore, the first step towards ensuring sustainable management of urban biodiversity is by quantifying the green infrastructure. This is done by conducting a tree inventory which may serve as a tool for the establishment of a baseline for setting management objectives by determining the resources present and where they are [21]. However, insufficient information about the status of green infrastructure in most urban cities in Nigeria remains a serious research gap, therefore this study was designed to assess the abundance and diversity of these trees within the three urban green areas of Osogbo.

2. Materials and method

2.1 Study area

The study was conducted in Osogbo, the capital city of Osun state due to its commercial importance. Osogbo covers a total area of 144 km^2 and is located within latitudes $7^0 43'$ N to $7^0 56'$ N and longitudes $4^0 33'$ E to $4^0 35'$ E, with an elevation beyond 502 m above sea level [22]. The population size of Osogbo for 1991, 2006 and projected population for 2016 were 187,219, 288,455 and 395,500 respectively [23]. The climate of the study area is a tropical hinterland type with a mean annual rainfall of 1200 mm to 1400 mm and a mean annual temperature of about 27°C [24]. Osogbo falls within the lowland tropical rainforest vegetation characterized by multiple canopies and lianas, most of which had since given way to secondary forest and derived savannah [25] due to intensive cultivation and bush burning for several years. Tender forest trees become replaced with fire-tolerant species and vegetation changes in features within short distances.

2.2 Woody trees sampling

Purposive selection of the study sites was done based on the urban forest cover and study objectives where three green spaces were identified: Riparian vegetation (site A), Industrial site (site B) and Residential site (site C) after the reconnaissance survey inside the urban center. Ten sample plots of 25 m x 25 m each were selected at random at the three sites. Tree species (≥ DBH 10 cm, h ≥ 1.3 m) within the sampled plots were identified. The Diameter at Breast Height and height (≥ DBH 10 cm, h ≥ 1.3 m) of each tree were also measured. Saplings of woody tree species (≥1 cm < 10 cm) were also identified and recorded.

2.3 Data analysis

The tree data collected were analyzed to determine the following parameters:

2.3.1 Biodiversity assessment

Shannon-Weiner Diversity Index, Species evenness and Rényi diversity ordering technique were estimated using the R-statistical package. The Shannon -Weiner Index is well recognized for measuring wood species abundance and richness [26, 27]. It measures the average probability of where a specie will belong when randomly predicted individually [28] and this is considered in the study to assess specie abundance and richness. This was incorporated in the Rényi Diversity profile. Rényi Diversity Profile which is used to order the diversity of tree species within the different physiognomy [29, 30] was used in this study. Studies have shown that only one diversity index may not be sufficient enough to provide information on ordering sites from high to low. Rényi profile incorporated species richness, evenness, Shannon's index, Simpson's index and Berger-Parker's index in a simple index. R statistical package was employed for accurate and effective computation [31]. Species Relative Density (RD) was determined to assess species relative distributions across the different habitats.

Species relative density (RD) for each tree species was determined by:

$$RD\ (\%) = (N/ni)\ \text{x}\ 100 \tag{1}$$

ni = number of individual species; N = Total number of species in the entire community.

2.3.2 Growth characteristics and aboveground biomass of sampled urban trees

Growth characteristics are important in the determination of the health status and biomass of trees and were measured using growth parameters such as girth size, basal area and volume. Species Relative Dominance (RD_o %) was used to assess the relative space occupancy and management practice of forested lands.

- Species Relative Dominance (RD_o %) [10]

$$RD_o = \text{Total basal cover of individual species} + \text{Total basal cover of all species} \times 100 \quad (2)$$

Where Ba = ^{2} /4

Importance Value Index relates how dominant a species is or the share of each species in a tree community.

- Importance Value Index (IVI) [31, 32]

$$IVI = RD + RF + RD_o \quad (3)$$

Where RF = number of chances of a species occurrence / total number of plots x 100

Where, ni = number of individuals of species i, N = total number of all individual trees of all species in the entire community, Bai = basal area of all trees belonging to a particular species i, Ban = basal area of all trees in a habitat, RD = Relative Density, RDo = Relative Dominance, RF = Relative Frequency.

The improved pantropical biomass Equation for tropical rainforest developed by [33] was employed for the estimation of the above-ground biomass content of tree species.

$$AGB_{est}\ (Kg) = 0.0673\ x\ \left(\rho D^2 H\right)^{0.976} \quad (4)$$

AGB is aboveground biomass, ρ = wood density, D = Diameter at breast height, H = Height.

Aboveground carbon storage of tree species [7, 34].

$$AGB_{carbon} = AGB_{est}\ X\ 0.5 \quad (5)$$

The mean aboveground biomass of the three study sites was subjected to One-Way Analysis of Variance (ANOVA) to test for their significance difference and Least Significance Difference (LSD) when ANOVA was significant. Descriptive statistics was employed for the presentation of the results.

3. Results and discussion

3.1 Tree species abundance and distribution

A total number of 124 individual trees consisting of 52 species, 43 genera and 22 families were identified in the three sites. In Site A, a total of 49 tree stems of 27 species were enumerated. Site B had 38 trees consisting of 18 species and site C had 37 trees stem of 20 species. *Albizia zygia* of Mimosaceae family was the most frequently encountered species in sites A and B, while *Milicia excelsa* of Moraceae family was most abundant in site C. The results revealed the extent of urban trees management in the selected green areas. A total of 170 tree saplings belonging to 15 families and 28 species were also encountered in the sampling sites. The higher proportion of saplings is attributed to the regeneration process occurring in the different sites which are sufficient key to maintaining forest continuity and urban greening.

Generally, the percentage of native species was 67% while 31% was exotic. Site A had a higher percentage (81%) of native species than Sites B (72%) and C (40%). while Site C had the highest percentage of exotic species (60%). Studies on urban green areas and forestry have reported variations in the proportion of trees origin where residential areas had more exotic tree species with higher percentages of native trees in cities [6, 19]. The higher percentage of native species in this study could be attributed to cultural and traditional beliefs as well as indigenous knowledge of dwellers on the usefulness of the trees and the incentives that benefited from the native tree species [35]. They also act as mitigation for poverty. This could also be true of urban areas because not all urban dwellers are financially buoyant and satisfied as life in the urban can be more expensive. [36] also signaled this in a field survey within Ibadan metropolis, Oyo State, Southwestern Nigeria. The higher percentage of the trees serve medicinal value (29%), followed by miscellaneous values (23%) such as ornament, clothing, dye-making, paper, tobacco, cosmetics and fencing. Trees used primarily as fuel and charcoal are 9% of the tree species population. Tree species used as a food source (15%) are trees that supply fruits, seeds and condiments, or spices. Their products may also be sold during fruit seasons (**Figure 1**). Trees for shade and support (10%) are used on farmlands for the protection of crops against harsh environmental conditions and as support for stem tubers. Trees used for construction and furniture trees constitute 99% of the total population. Also, primarily 5% of the tree species enumerated help in soil nutrient enhancement and protection.

Diversity indices are a more compact method of comparing the diversity (variety) of species. Shannon-Wiener diversity estimation for the three sites has values within the expected range of 1.5 - 3.5 [10, 28, 37]. This shows that the sites are rich and diverse in tree species. Site A had the highest diversity index value of 3.15, while Site B had the lowest diversity index value of 2.55. Describing species diversity as a single

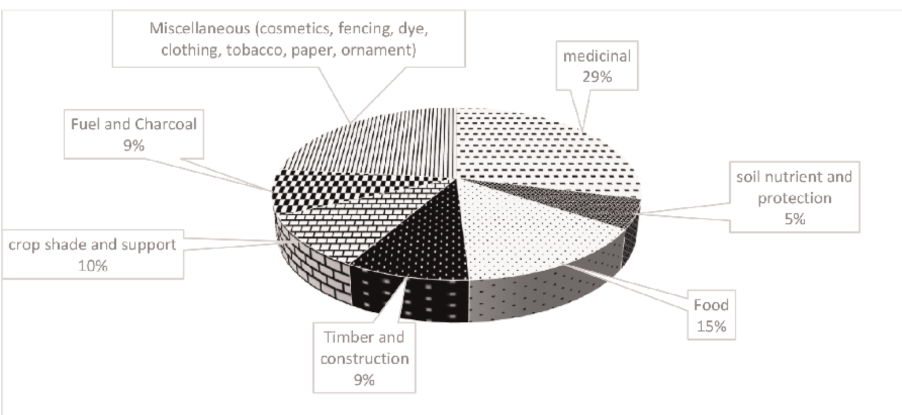

Figure 1.
Ecosystem Services of Tree Species within the study sites.

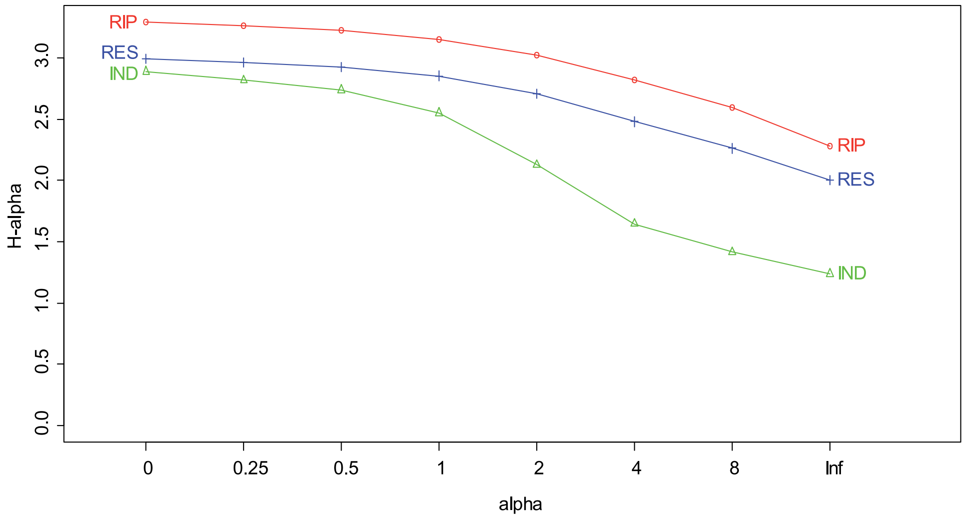

Figure 2.
Rényi diversity profile. H = Rényi diversity profile, alpha = diversity parameter, RIP = riparian vegetation, RES = residential site, IND = industrial site.

value has been reported to compromise much of the detailed structure of a community and that different measures may lead to different rankings among communities [38, 39]. Therefore, a diversity profile that portrays the simultaneous values of a large collection of diversity indices in a single diversity spectrum has been recommended. Rényi diversity profile revealed that the riparian site was the most diverse. Ordering of the sites by Rényi Profile diversity followed; Riparian > Residential > Industrial site (**Figure 2**). Comparing the slope of the three-diversity profile, it is revealed that riparian sites and residential sites have similar and higher species evenness than the industrial site. The more horizontal the shape of the side profile, the higher the species evenness [30, 31]. However, the Riparian and Industrial sites have more connectivity (**Figure 3**) to the number of tree species common to both of them, The nine [9] tree species common to both sites are *Albizia adianthifolia, A. zygia, Brachystegia eurycoma, Ceiba pentandra, Ficus exasperata, Gliricidia sepium, Holarrhena floribunda, Lecaniodiscus cupanioides* and *Margaritaria discoidea*. The agricultural practice was also common within the two sites (Riparian and Industrial sites) in the past. This could be affirmed by the presence of tree species which are usually found within regrowth vegetation previously used for agricultural practice. Species evenness, as a basic component of diversity that measures the equitability of species spread [40], was observed to be highest in site A (0.9681), followed by site C (0.9529) and site B (0.8826). More evenly distribution of trees within the riparian vegetation could be attributed to less competition for space among the tree species and high competition among the tree species within the industrial site as a result of stem proximity which could have also led to competition.

The growth characteristics of the trees in each site varied with growth parameters. Growth variables were estimated as 1.18 m^2, 5.01 m^2 and 11.06 m^2 (basal area), and 13.49 m^3, 64.03 m^3 and 122.39 m^3 (volume) for sites A, B and C respectively. Site A had the highest percentage of small-sized trees (65.31%) while large and largest-sized trees were absent. Industrial sites had 42.11% of the trees within the small-sized range and only 2.63% within the largest-sized range, while site C had 43.24% and 10.81% within the medium and largest-sized ranges respectively (**Figure 4a**). Generally, a

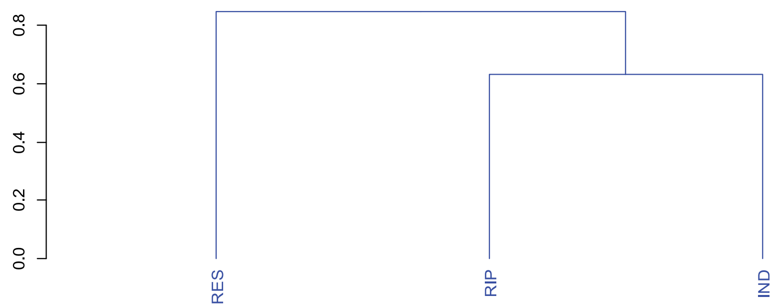

Figure 3.
Clustering analysis of trees diversity within the three sites.

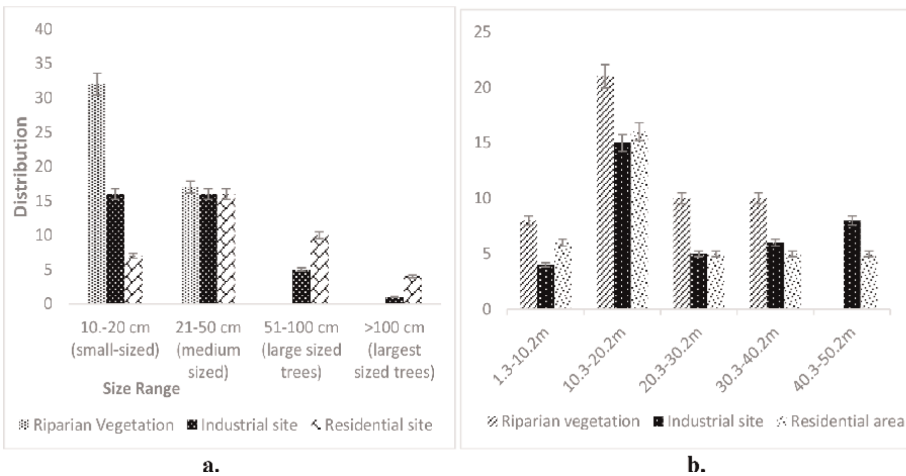

Figure 4.
(a) Size-class distribution (mean ± sd) and (b) height-class distribution (mean ± sd) of trees enumerated.

higher percentage of the trees (41%) were within the class of 10.3-20.2 m. Site A had no record for trees heights within class 40.3-50.2 m (**Figure 4b**). Site B had only 10.5% of trees within class 1.3-10.2 m. The variation in the DBH and heights contributed to the differences in estimated AGB (**Table 1**). The first four species that contributed to the highest growth characteristics in the three sites are presented in **Table 2**. *C. pentandra* in site A; *Chrysophyllum albidum* in site B and *Brachystegia eurycoma* in site C contributed to the highest basal area, volume and RDo. However, the highest RDe, RF and IVI were contributed to by *A. zygia* in sites A and B, and *M. excelsa* in sites C. The variations explain the management practices for trees preservation and exploitation in the different sites. From the field survey, it was observed that the riparian vegetation is an open area for public use (agriculture, logging and settlement). The residential site is a restricted area to loggers and forest exploiters. The industrial site is also restricted to exploiters but faced with the challenge of encroachment by secret exploiters. Forest resources exploitation for services such as medicine, food and cosmetics may have contributed to reduced trees growth. The depletion of these economically important species populations as habitat degradation and over-exploitation are the two main causative agents [41, 42]. The multipurpose utility of species could also indicate high pressure on them. Moreover, native species with the least abundance could be considered vulnerable in their different habitats. This agrees with a study carried out by [10]

	Species	AGB (Kg C ha^{-1})	Mean DBH (cm)	Height Class (m)
Top ten species				
	Chrysophyllum albidum	190929.20	131.0	47.0
	Milicia excelsa	94237.25	88.6	28.2
	Peltophorum pterocarpum	64652.00	89.0	32.6
	Delonix regia	60275.43	99.2	28.2
	Brachystegia eurycoma	41711.32	26.8	31.2
	Holarrhena floribunda	19068.44	44.9	27.0
	Sterculia tragacantha	17681.00	43.7	24.7
	Terminalia catappa	11606.86	27.0	7.4
	Albizia zygia	11421.60	34.1	29.4
	Persea americana	10981.86	57.8	14.9
Least ten species				
	Spondias mombin	158.92	10.0	8.2
	Croton zambesicus	237.63	10.5	9.3
	Anacardium occidentale	421.03	17.6	6.9
	Baphia nitida	482.94	16.4	14.8
	Ficus sp	573.16	15.0	14.4
	Dictyandra arborescens	592.82	17.0	7.3
	Voacanga africana	607.84	13.1	17.0
	Blighia sapida	639.35	10.7	16.2
	Albizia ferruginea	673.24	12.6	19.0
	Citrus sineensis	779.18	19.6	7.6

Table 1.
Aboveground biomass, size and height of top ten and least ten tree species in the study location.

that species with low relative density and relative dominance are at the top list of vulnerable species under threat of extinction.

The highest aboveground biomass was recorded in site C, while site A had the least value (**Table 1**). The size class 21-50 cm contributed most to the tree aboveground biomass in site C with a mean value of 5062.29 ± 730.90 Kg C ha^{-1} (5.06 ± 0.73 t C ha^{-1}), while site C recorded the highest aboveground value for >100 cm with a mean value 134531.84 ± 29018.85 Kg C ha^{-1} 134.53 ± 29.02 t C ha^{-1}). Generally, the size class 10-20 cm contributed the least to the aboveground biomass in all the land use (**Table 3**). The contributions of ten species with the highest aboveground biomass were recorded in **Table 1** where *C. albidum* (site C) had the highest AGB. This is attributed to its large size and height. (131 cm and 47 m respectively). *M. excelsa* also had a mean DBH of 88.6 cm and a height of 28.2 m while its carbon content was estimated as 94,237.25 Kg C ha $^{-1}$ (94.24 t C ha $^{-1}$). These are examples of native species with high carbon content. Protective and maintenance measures are necessary for the conservation of native species that could promote the uptake of high carbon in the atmosphere. Moreover, the highest AGB values for site C were contributed to by the highest girth sizes of the trees which also relates to the management practice of tree preservation within the site. Size,

	Species	BA	VOL	RDo	RDe	RF	IVI
SITE A	*Albizia zygia*	0.03	0.27	3.23	10.20	50	63.43
	Anthonotha macrophylla	0.07	1.17	7.53	2.04	25	34.57
	Ceiba pentandra	0.16	2.65	17.20	2.04	25	44.25
	Cola acuminata	0.08	0.77	8.60	2.04	25	35.64
SITE B	*Albizia adianthifolia*	0.10	0.42	4.00	5.26	100	109.26
	A. zygia	0.17	1.97	6.84	28.95	100	135.79
	C. pentandra	0.30	2.48	12.21	5.26	50	67.47
	Chrysophyllum albidum	1.35	21.12	54.69	2.63	50	107.32
SITE C	*Brachystegia eurycoma*	1.13	16.25	21.46	2.70	25	49.16
	Delonix regia	0.77	7.28	14.69	5.41	25	45.09
	Holarrhena floribunda	0.41	4.38	7.75	8.11	75	90.86
	Milicia excelsa	0.75	9.51	14.31	13.51	100	127.82

Table 2.
The first four species with highest growth characteristics in the different sites.

Size class (cm)	Riparian vegetation (Kg C ha^{-1})	Industrial site (Kg C ha^{-1})	Residential site (Kg C ha^{-1})
10-20	1062.77 ± 134.83	885.27 ± 156.48	620.88 ± 118.69
21-50	5062.29 ± 730.90	6855.49 ± 1323.02	4272.29 ± 1073.59
51-100	0	36742.80 ± 6005.92	43720.21 ± 7629.14
>100	0	0	134531.84 ± 29018.85

Table 3.
Size-class distribution of tree aboveground biomass (kg C ha^{-1}) recorded across the study sites.

age and species are major factors that influence the amount of carbon that trees can absorb. [43] reported that matured trees can absorb up to 48 lbs. of CO_2 per year. [44] reported that at maturity, trees can store approximately 1000 times more than saplings. The result of the Analysis of Variance (ANOVA) performed on the aboveground biomass among the three study sites revealed that there was no significant difference between sites A and B. There was also no significant difference between sites B and C. However, a significant difference occurred between sites A and C at probability level 0.01 (99% confidence interval).

4. Conclusion

Protection of urban tropical forest trees, as noticed within the residential and industrial sites, contributes effectively to tropical urban vegetation, urban diversity and ecosystem benefits. Therefore, proper urban planning, preservation and sustainable utility of biodiversity within the urban areas is also a way to decelerate the rapid rate of biodiversity loss that results from population explosion, urban expansion and pressure on the ecosystem.

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Chapter 7

Physical Quality of Soils in a Toposequence of a Forest Fragment under Livestock Activity in a Watershed in South Brazil

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Abstract

The conservation of native forests is fundamental to the preservation of hydric resources in the landscape. The use of animals in forest fragments has resulted in degradations in the soil, resulting in the grating of these. Thus, soil classes were studied and physical parameters of forest soils were evaluated in areas without and with cattle grazing in the "Arroio Pelotas" watershed, Pelotas, Rio Grande do Sul, extreme south of Brazil. The results were submitted to statistical analysis with the Kruskal–Wallis nonparametric test with a significance level of 5%. The means of the physical parameters of soil in the same toposequence and layers with and without the presence of livestock were compared. By analyzing soil physical attributes (density, macroporosity, and microporosity) it can be seen that the structural quality of the soil is affected by the access of animals inside the forest fragments, especially in the upper layer of the soil (0–5 cm deep). In forest fragments without access to animals, the physical structure of the soil presented the best conditions of macroporosity and, consequently, greater protection of nutrients, microorganisms, and water resources. Therefore, it is concluded that conservation by the isolation of protective forests in rural property planning benefits the quality of forest soils.

Keywords: physical attributes of the soil, cattle grazing, forest soils, environmental management, soil degradation

1. Introduction

Forests are important for the conservation of water resources at the landscape level. Thus, the need for conservation and recovery of forests in watersheds is fundamental, according to the new Brazilian forest code [1]. Currently, forests are negatively impacted in different ways. Livestock activity carried out without planning in rural properties can determine negative environmental impacts on the environment and production. The influences of livestock in forest fragment part of the interference in vegetation until the degradation of forest soils, where the animals travel in search of fodder and end up exploring part of the fragments [2]. Soil degradation in livestock areas ends up resulting in low-quality pastures by overgrazing, gullies, sanding, pasture compactions and in the path of animals, floods, siltations, and loss of ravines on the banks of watercourses, among others.

The access of animals in forest areas, as well as their exploitation, promotes a grating and fragmentation of forest areas [2], with this there is a greater luminosity input inside that stimulates the growth of grasses in these areas. These conditions intensify the trampling of the animal and, consequently, soil compaction, due to the demand for fodder in forest areas. Compaction is influenced by texture, aggregation, soil moisture at the time of management practices, traffic and frequency of vehicle trafficking, intentional soil manipulations, and the loss of organic matter [3–5].

Compaction by animal trampled results in changes in soil physical properties, with a rearrangement of mineral and organic solid particles due to the mechanical force applied to the surface and transmitted mainly by soil solids [6]. Compaction is defined as the process of soil porosity decrease, especially the macroporosity occupied by air, increasing the density and resistance to root penetration in the soils. It also decreases water infiltration along the soil profile, causing surface erosion and removals of high soil volumes [7]. Compaction promotes the reduction of the pores of the inter and intra-aggregates, resulting in a dense and massive porous system, which may lead to more pronounced horizontal water flow, and consequently, greater soil erosion [8, 9].

Forest growth is affected by compaction and, consequently, their productivity is reduced, as well as impairs the protection and conservation of soils and water. Increased compaction negatively affects seed germination of native species, root growth of plants, and productivity of mature commercial plantations [5, 10]. This is due to the restriction of root development, through low water infiltration and redistribution, limitation of adsorption and/or absorption of nutrients in the soil, and the precarious aeration of the soil [5, 11].

The main cause of compaction in forest soils is the trafficking of machinery for the management and harvesting of forests, as well as the trafficking of people and animals in the area [12]. Studies also show that anthropic soil compaction occurs with different management practices according to each activity. For example, by the displacement of agricultural machinery in the forest, grain and fruit harvest [9, 13, 14], in fallow areas, no-tillage and no-tillage system of grains [15, 16], and trampled by large domestic animals [17, 18].

The results of soil quality compaction are the breakdown and reduction of macroporosity. In study on orange orchard, Lima et al. [19] found superficial compaction by the farm and by the traffic of machines, affecting the shape and distribution of the poorly space, and the most affected were the biopores.

According to Han et al. [20], pores derived from biological activity (or biopores) formed by the mineralization of the root system and organisms of the soil macrofauna. It is important to understand that vegetables and animals produce biopores and other various ecosystem services to the rural property ecosystem, as shown in **Table 1**. In addition to the roots, soil fauna produces biopores. This soil fauna includes microscopic organisms, such as nematodes, mites, and colêmbolos, to easily visible organisms, such as earthworms, spiders, ants, termites, and beetles [21].

With compaction, there are changes in the distribution of soil aggregate size, changes in macroporosity and microporosity, increased mechanical impediment, decreased pore continuity, loss of water retention and infiltration capacity, weak internal drainage, and increased heat conduction. All these environmental impacts promoted by compaction affect soil quality, whether an agricultural soil of cultivation or pasture, as in the forest soils of this study.

Service category	Service	Ecosystem process	Contribution of fauna
Provision	Available water	Infiltration and storage of water in the soil	Bioturbation, increased infiltration, and water retention in the soil.
	Food	Biomass production	Food for humans and animals.
Support*	Nutrient cycling	Decomposition and humification	Fragmentation, ingestion, stimulation of the de-compositorcing microbial community.
		Regulation of nutrient losses	Mineralization prevents leaching and continuous circulation of nutrients.
	Primary productivity	Stimulation of symbiont activity and growth-promoting microorganisms	Stimulation of symbiotes in the rhizosphere, intestines and coprolites, and change in the activity of microorganisms promoting plant growth.
		Protection against pests and diseases	Production of repellent substances. Pest control by increasing biodiversity and population balance.
Regulation*	Flood control	Drainage, infiltration, and storage of water in the soil	Change in roughness and soil structure, increasing porosity and surface biopores that increase retention and infiltration.
	Climate regulation	Carbon sequestration	Formation of stable aggregates rich in organic matter in the form of substances.
	Pollination	Pollination	Insects with the phase of life in the soil contribute to pollination.
Cultural	Recreation	Social and natural interaction	Food for some organisms, creation or collections as a hobby, theme for exhibitions, art, literature.
	Education	It only cares who knows the soil	Instrument for environmental education and information in rural extension.

**Main services of soil animals (indirect).*
Adapted from Parron et al. [21] according to Millennium Ecosystem Assessment (2005).

Table 1.
Ecosystem services of soil fauna.

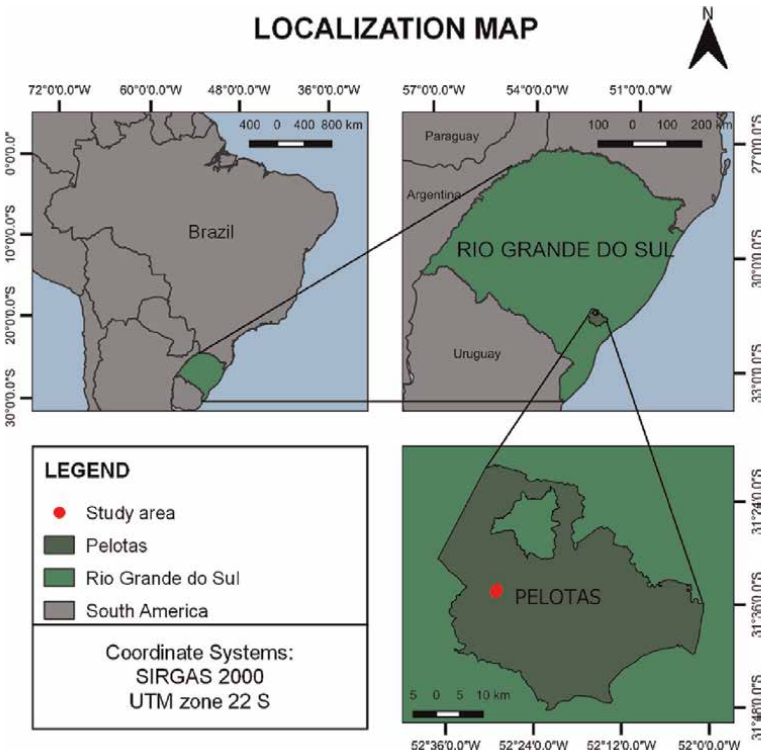

Figure 1.
Location map of the study area.

There are few studies on soil quality within native forests in Brazil. Thus, there is a need to evaluate the environmental impacts of livestock on forest soils, in a context of producing with good environmental practices and with social, economic, and environmental sustainability. For this, the objective was to evaluate soil physics parameters in soil toposequence shards in fragments of native forests with and without livestock, on the banks of the watercourse toward the top of the hill in a part of the Arroio Pelotas watershed, in the state of Rio Grande do Sul, and in the extreme south of Brazil (**Figure 1**).

2. Material and methods

2.1 Study area

2.1.1 The watershed of the Arroio Pelotas

The study area in the municipality of Pelotas is inserted in the Arroio Pelotas watershed (**Figure 2**). The hydrographic basin of this watercourse is located under two phytogeographic regions: pioneer formation areas and the semidecidual seasonal forest, according to the classification of the Brazilian Institute of Geography and Statistics [22]. The two phytogeographic regions are distinct and conditioned by the geomorphological characteristics of the relief. The areas of pioneer formations, as [23]

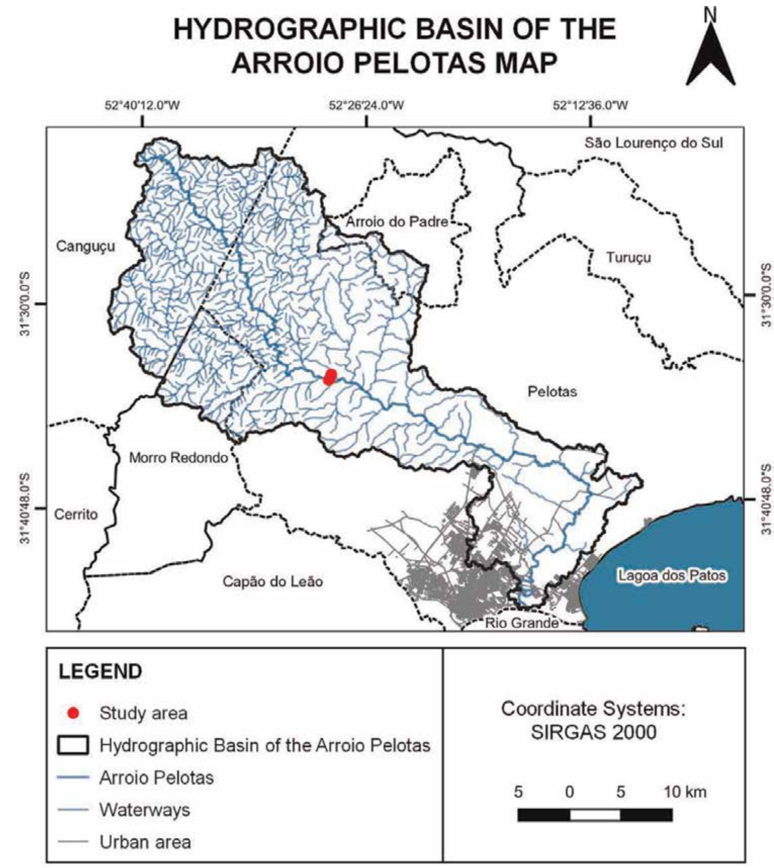

Figure 2.
Hydrographic basin of the Arroio Pelotas map, Pelotas, RS, Brazil.

called restinga are located predominantly in the internal and external coastal plain with vegetation of dry and humid fields, bathed, riparian forests, and capons of restinga forests. The other region, called semidecidual seasonal forest, was the site of this study and forest vegetation is typical (**Figure 3**) [22].

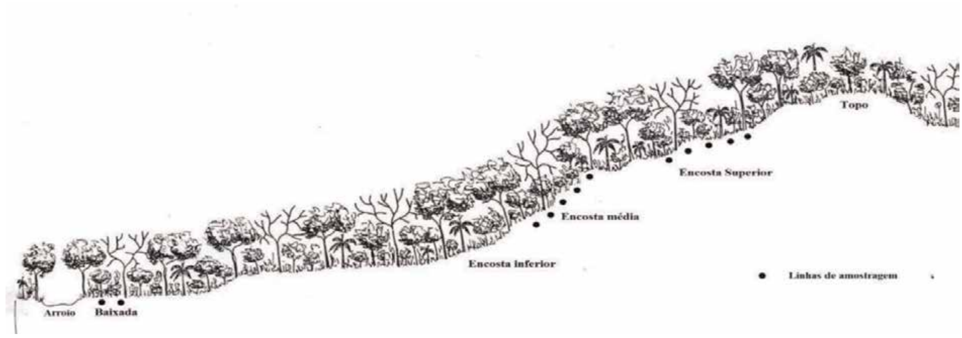

Figure 3.
Illustrative scheme of the topographic profile of the forest areas sampled along the relief in Pelotas, RS, Brazil. •= quematic distribution of soil sample collections. Source: Adapted Image of Phytogeographic Manual of Brazilian Vegetation IBGE [22].

2.1.2 Soils in the Arroio Pelotas river basin

The area is inserted in the geomorphological province of "escudo Sul-Rio-Grandense" besides being ancient, is geologically very complex, because it comprises several plutonic igneous rocks, mainly of granite composition associated with belts of metamorphic rocks, which were covered by sequences of sedimentary rocks and volcanic rocks [24]. On the edges of this heterogeneous geological province, is inserted approximately half of the area of the municipality of Pelotas. In this region, three main soil units are found: association LUVISSOLO HÁPLICO Órtico with NEOSSOLO REGOLÍTICO Distro-úmbrico, ARGISSOLO VERMELHO-AMARELO Distrófico and PLANOSSOLO HÁPLICO Eutrófico associado with GLEISSOLO MELÂNICO Tb Eutrófico, covering respectively the high, middle and low relief of the slope of the Escudo Sul-Rio-Grandense [24].

2.2 Soil sampling

2.2.1 Description and collection of soil profiles

For soil classification, the environmental characteristics of the site (external characteristics, such as relief, drainage, erosion, vegetation, and source material) were described, which constitutes the general description, and the morphological characteristics of the soil, such as thickness, arrangement, transition and characteristics between horizons and horizon characteristics (color, texture, structure, and consistency, etc.) as Santos et al. [25]. Soil profiles were described and collected at the top, on the upper slope, on the middle slope, and on the lowered slope. Soil horizon samples were physically and chemically characterized in the laboratories of the Soil Department of FAEM/UFPel[1] following the methodology set out in Teixeira et al. [26], and were subsequently classified according to the Sistema Brasileiro de Classificação de Solos, developed by the National Soil Research Center [27], and world reference base for soil resources [28].

2.2.2 Survey of soil physical attributes along the toposequence

Deformed samples (unpreserved structure) and undisturbed samples (preserved structure) of soil were collected in six different treatments distributed along two environmental gradients of the toposequence in the Pelotas Stream. For each gradient with three sample blocks distributed on the upper slope, on the middle slope, and on the lowered slope on the margin of the watercourse (**Figure 3**). The low slope and top relief sections were not sampled by the absence of adequate forest fragments in the study region. The gradients were divided into an area with access to cattle grazing and areas without grazing, witness for comparison. The control of areas of isolated forest fragments without access to animals and in conditions of relief similar to treatments with livestock access. The maximum distance between the areas is approximately 1400 m. The altitude of the plots ranged from 39 to 116 meters at sea level. Forest fragments were distributed in three small rural properties.

The undisturbed samples (preserved soil structure) were collected using the volumetric ring method [26]. The rings were 3 cm tall and 4.8 cm in diameter, with an

[1] FAEM: Faculty of Agronomy Eliseu Maciel of Federal University of Pelotas (UFPel).

average volume of 54.9 cm^3 per sample. For the collection in the field, the insertion site of the ring was cleaned in the soil, removing the litter until the soil was fully discovered for the insertion of the rings. After the rings were inserted, using shovels, the ring was carefully removed from the ground, so as not to disaggregate the sample. Subsequently, each ring was wrapped in aluminum foil, sealed and labeled. The undeformed samples on the foil were carefully transported to avoid damage to the soil structure contained inside the ring.

Fourteen samples were collected per sample block, totaling 42 total samples per gradient, seven rings in the 0–5 cm layer and seven other rings in the 5–10 cm layer. The sampling points were approximately 10 meters from each other, inside the sample block of the collection of native forest vegetation (**Figure 3**). With the excess of soil in the layers of 0–5 and 5–10 cm, the deformed samples were collected, where they were placed in plastic bags, sealed, and labeled.

The deformed and undeformed samples were taken to the pedology laboratory, sample preparation room, and to the soil chemistry and physics laboratories of Faculty of Agronomy Eliseu Maciel of Federal University of Pelotas. The deformed samples were placed to dry at room temperature in the sample preparation room. After 5–7 days they were destroyed with a wooden roll and passed in 2 mm sieves, and then placed in plastic bags labeled and stored in the Pedology Laboratory/UFPel, constituting the air-dried soil samples. The undeformed samples were stored in the pedology laboratory and taken to the soil physics laboratory FAEM/UFPel, for the preparation and evaluation of attributes of soil density, microporosity (pores smaller than 0.05 mm), macroporosity (pores greater than 0.05 mm), and total porosity.

The determinations of the chemical attributes of the soil were made according to Tedesco et al. [29] and Teixeira et al. [26]. The determinations of pH (H_2O), pH (KCl), Ca, Mg, K, Na, and Al e P available were carried out in accordance with Tedesco et al. [29]. Organic carbon and H + Al were used in the study by Teixeira et al. [26]. The texture of the soil profile layers was analyzed by the granulometry method following the methodology proposed by Teixeira et al. [26].

The pH in water (active acidity) was determined at the ratio of 1:1 (soil:water) by immersion of a glass electrode connected to a pot. The pH in KCl (potential acidity – H + Al) was extracted with calcium acetate and determined by titration with NaOH. Calcium extraction (Ca^{+2}), magnesium (Mg^{+2}) e aluminum (Al^{+3}) exchangeable were made with KCl 1 mol L^{-1} and determined in the atomic absorption spectrophotometer (Ca^{+2} e Mg^{+2}) and by titration with NaOH (Al^{+3}). Potassium content (K^+) available and sodium (Na^+) were estimated by the double extractor method of $Mehlich^{-1}$ and analyzed by flame photometry. Based on the results of the analyses, the base saturation was calculated (BS%), the cation exchange capacity (CEC), and aluminum saturation (m%). Base saturation is calculated by the sum of the bases Ca^{+2}, Mg^{+2}, and K^+ e Na^+. A CTC = (Sum of bases + H^+ + Al^{+3}). Aluminum saturation (m%) is calculated by Eq. (1) below:

$$\mathrm{m}\ (\%) = \left[\mathrm{Al}^{+3}/\left(\mathrm{SB} + \mathrm{Al}^{+3}\right)\right] \times 100 \tag{1}$$

SB = sum of bases

Soil physical attributes were determined by the volumetric ring method, using the dry soil mass in a greenhouse to 105°C for 24 hours in the volumetric ring of known volume [26]. To calculate soil density by mass, the equation was used (2):

$$\mathrm{Bd} = \mathrm{Dsm}/\mathrm{Va} \tag{2}$$

Bd = soil bulk density ($g.cm^{-3}$)
Dsm = dry soil mass contained inside the volumetric ring (g)
Va = volumetric ring volume (cm^3)
The determination of the porosity of the total soil was performed by the tension table method, using volumetric rings [26]. Through total porosity, the microporosity and macroporosity of the soil were determined, using the height of the suction water column of 60 cm to obtain a 0.006 MPa. The total porosity was calculated by Eq. (3) shown below:

$$Pt = [(P1 - P3) - P4/Va] \times 100 \tag{3}$$

Pt = total porosity (%)
P1 = saturated sample weight with fabric and rubber alloy (g)
P3 = weight of rubber alloy and saturated fabric (g)
P4 = weight of the kiln-dried sample (g)
Va = volumetric ring volume (cm^3)
Microporosity (mp) was calculated from Eq. (4) below:

$$mp = [(P2 - P4)/Va] \times 100 \tag{4}$$

mp = micropores (%)
P2 = balance sample weight (g)
P4 = weight of the kiln-dried sample (g)
Va = volumetric ring volume (cm^3)
Macroporosity (Mp) was calculated by the difference of total porosity by microporosity (mp), through the Eq. (5):

$$Mp = Pt\text{–}mp \tag{5}$$

Mp = macroporosity (%)
Pt = total porosity (%)
In the soil physics laboratory, for each volumetric ring, the aluminum foil packaging was first removed. After removal of the package, the sample toilet was performed with the use of a stiletto, scissors, knife, and water spray. The toilet consisted of the process of removing excess soil at the lower and upper ends of the ring to equalize the ring volume with the volume of the soil sample. Following the toilet procedure, at the bottom of the ring was placed a piece of appropriate fabric fastened with rubber alloy [26]. The fabric has the function of handling possible sections of soil that detach from the ring. In the sequence, each ring was numbered and placed for saturation in trays by capillary rise, with water placed up to ¾ from the height of the ring, outdoors in the laboratory environment for at least 48 hours [26]. Subsequently, the rings were removed from the water and weighed on a precision scale, and then the saturated rings were placed for drainage on a tension table and a tension corresponding to 60 cm of water column height corresponding to macropore drainage was applied.

The samples of the volumetric rings were placed in Richard's pressure chamber and remained until the drainage of the excess moisture for each voltage ceased. After the volumetric rings came out pressure chamber, these were weighed to obtain the weight of the ring in balance and discounted the weights of the fabric and rubber alloy saturated. Subsequently, the rings in equilibrium were placed to dry in a greenhouse for 48 hours at a temperature of 105°C [26]. The rings were removed from the greenhouse and weighed on a precision scale to obtain the weight of the dry soil.

After obtaining the weight of the saturated soil, the balance soil weight and the dry soil weight of each volumetric ring, the soil of the ring was removed, which was washed, dried, and weighed to obtain the weight of the ring without soil to use in the subtraction in the weight of the samples. For each volumetric ring, it was measured, the diameter and height for calculating the volumetric ring volume, using the Eq. (6) below:

$$Va = д \times r2 \times h \tag{6}$$

Va = volumetric ring volume (m^3)
д = pi = 3.1428
r = geometric radius of the ring (m)
h = height (m)

The results were submitted to statistical analysis, which was performed by the PAST software with the Kruskal–Wallis nonparametric test with a significance level of 5%. The means of the physical parameters of soil in the same toposequence and layers with and without the presence of livestock were compared.

3. Results and discussion

The pedological survey was carried out with a methodology that aims to identify, characterize, and classify soil profiles [25]. Soil classification resulted in three soil classes at the order level: Leptosol, regosol, and acrisol, according to the classification proposed by IUSS working group WRB [28], and in two soil classes at the order level: Neossolo and Argissolo, according to the classification proposed by Santos et al. [27] (**Table 2**; **Figure 4**). It has been classified as Argissolo in the downloaded and Neossolos on the middle slope, top slope, and top (**Figure 5**). Soil classification was performed by means of analytical data interpretation (**Tables 3** and **4**), physic and chemical attributes, and morphological description of profiles.

The class of Argissolos in general, are soils of varying depth, from strong to imperfectly drained, with reddish or yellowish colors and rarely brunadas or grayish. It has a sandy texture to clayey on horizon A and medium to very clayey on horizon B, always having a textural gradient of clay from horizon A to B, which characterizes the textural horizon B (Bt). The transition usually between horizons A and Bt is clear, abrupt, or gradual. They are strong to moderately acidic soils, predominantly kaolinitic with high or low base saturation [27].

Soil profile	SiBCS	WRB
P1	NEOSSOLO LITÓLICO Distrófico fragmentário (RLd)	Dystric Leptosol (LPdy)
P2	NEOSSOLO REGOLÍTICO Distrófico leptofragmentário (RRd)	Leptic Regosol (Rgle)
P3	NEOSSOLO REGOLÍTICO Eutrófico típico (RRe)	Eutric Regosol (RGeu)
P4	ARGISSOLO AMARELO Distrófico típico (PAd)	Haplic Acrisol (ACha)

SiBCS = Sistema Brasileiro de Classificação de solos [27]; WRB = World reference base for soil resources [28].

Table 2.
Equivalence of classification systems.

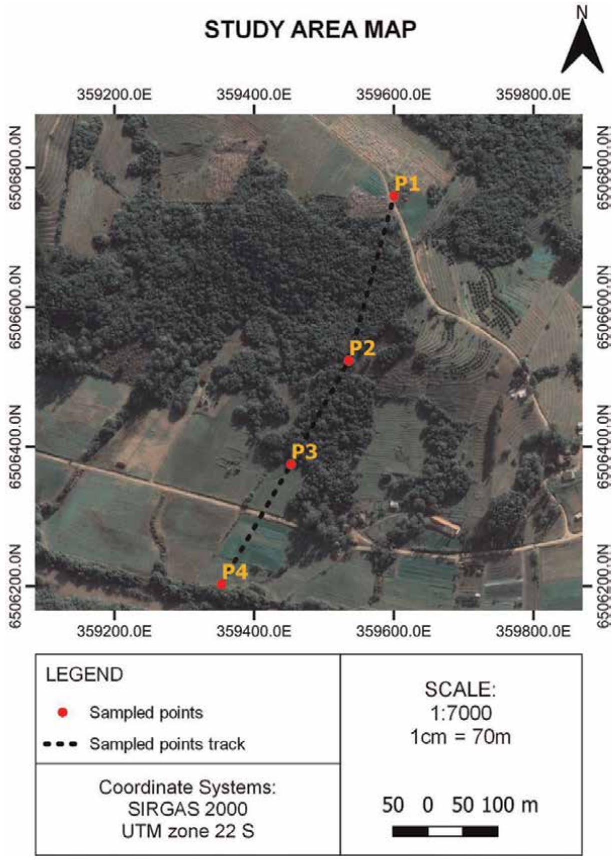

Figure 4.
Study area map. P1 – NEOSSOLO LITÓLICO Distrófico fragmentário (RLd) /Dystric Leptosol (LPdv), P2 – NEOSSOLO REGOLÍTICO Distrófico leptofragmentário (RRd)/Leptic Regosol (RGle), P3 – NEOSSOLO REGOLÍTICO Eutrófico típico (RRe)/Eutric Regosol (RGeu) and P4 – ARGISSOLO AMARELO Distrófico típico (PAd)/Haplic Acrisol (ACha).

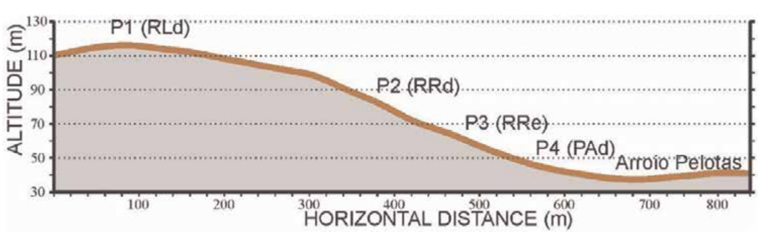

Figure 5.
Toposequence of the study area with the respective topographic profiles in the landscape and the respective soils. P1 – NEOSSOLO LITÓLICO Distrófico fragmentário (RLd)/Dystric Leptosol (LPdv), P2 – NEOSSOLO REGOLÍTICO Distrófico leptofragmentário (RRd)/Leptic Regosol (RGle), P3 – NEOSSOLO REGOLÍTICO Eutrófico típico (RRe)/Eutric Regosol (RGeu) and P4 – ARGISSOLO AMARELO Distrófico típico (PAd)/Haplic Acrisol (ACha).

In downloaded, profile P4 (**Figure 4**), the soil was classified as ARGISSOLO AMARELO Distrófico típico, presenting sequence of horizons A-AB1-AB2-Bt, deep soil (>100 cm), with yellowish color, indicating a sign of good drainage and low base

Soil profile	Hz	pH H_2O	pH HCl	Ca	Mg	K	Na	Al	H	CEC	BS	m	OC	P
				$cmol_c.kg^{-1}$							%		$g.kg^{-1}$	$mg.kg^{-1}$
	NEOSSOLO LITÓLICO Distrófico fragmentário													
P1	A	7.3	5.4	3.5	1.4	0.2	0.0	0.1	2.9	8.2	62	2.8	11.8	9.9
	A/CR/R	6.4	4.4	2.9	1.4	0.2	0.0	0.7	5.1	10.3	44	13.2	10.1	3.5
	CR/R	5.3	3.9	2.6	1.4	0.2	0.0	1.9	4.6	10.7	39	31.3	6.0	2.0
	NEOSSOLO REGOLÍTICO Distrófico leptofragmentário													
P2	A	4.9	4.1	3.8	0.9	0.2	0.0	1.1	7.1	13.0	38	17.8	25.7	2.8
	AC	4.9	3.8	1.8	0.4	0.1	0.0	1.9	5.5	9.7	23	46.3	10.1	1.0
	C	4.9	3.9	1	0.2	0	0.0	1.2	2.9	5.4	24	48.3	5.4	0.8
	C/CR	5	3.8	1.2	0.3	0	0.0	1.4	2.6	5.6	29	46.6	4.9	1.1
	NEOSSOLO REGOLÍTICO Eutrófico típico													
P3	A	5.2	4.2	5.9	1	0.1	0.1	0.8	6.1	14.0	51	9.8	16.3	2.0
	AC	5.6	4.2	3.8	0.4	0.0	0.1	0.5	3.1	8.0	54	11.0	6.4	1.1
	C1	5.6	4.1	3.1	0.7	0.0	0.1	0.5	1.5	5.9	67	11.0	2.7	1.7
	C2	5.6	4.0	3.2	0.8	0.0	0.1	0.6	0.9	5.7	73	13.1	2.0	1.4
	ARGISSOLO AMARELO Distrófico típico													
P4	A	5.5	4.7	2.2	0.6	0.1	0	0.6	1.4	4.9	59	16.6	7.0	24.5
	AB1	5.3	4.1	1.4	0.7	0.1	0	0.7	2.3	5.1	42	23.9	5.8	21.9
	AB2	5.2	4.0	1.8	0.7	0.0	0	0.4	2.9	6.0	44	14.1	5.6	7.0
	Bt	4.9	3.8	2.9	0.9	0.1	0	1.2	3.0	8.0	48	23.2	5.2	4.3

P1 – NEOSSOLO LITÓLICO Distrófico fragmentário (RLd) /Dystric Leptosol (LPdv), P2 – NEOSSOLO REGOLÍTICO Distrófico leptofragmentário (RRd) /Leptic Regosol (RGle), P3 – NEOSSOLO REGOLÍTICO Eutrófico típico (RRe) /Eutric Regosol (RGeu) and P4 – ARGISSOLO AMARELO Distrófico típico (PAd)/Haplic Acrisol (ACha). Hz= Soil profile horizon; CEC= cation exchange capability; BS= base saturation; m= aluminum saturation; OC= organic carbon.

Table 3.
Chemical characteristics of the toposequence profiles in the Arroio Pelotas, RS, Brasil.

saturation (<50%). These characteristics, together with the chemical attributes described in **Table 2**, corroborate with Streck et al. [24] which gives these soils parameters of low natural fertility, high acidity, and low cation exchange capacity (CEC).

The second soil class identified was that of the Neossolos, present in the sampling points P1, P2 e P3 (**Figure 4**). The Neossolos are characterized by being soils of recent formation, consisting of mineral or organic material that does not present express changes in relation to the source material, due to the low intensity of action of the pedogenetic processes, and that does not present horizon B diagnosis [27]. Despite having good natural fertility, this class has limitations in the production of annual crops, since it has a strong restriction on the entry of mechanization due to stoniness and rockiness, which impairs root development and water storage by low effective profundiade [24].

Soil profile	Horizon	Depth	Thick sand	Fine sand	Silt	Clay
		cm		$(g.kg^{-1})$		
		NEOSSOLO LITÓLICO Distrófico fragmentário				
P1	A	0–10/15	207	374	297	123
	A/CR/R	10/15–16/25	182	339	328	151
	CR/R	16/25–40+	116	345	355	184
		NEOSSOLO REGOLÍTICO Distrófico leptofragmentário				
P2	A	0–20	298	260	276	167
	AC	20–38	191	331	305	174
	C	38–60/70	228	370	276	126
	C/CR	60/70–85+	343	245	268	144
		NEOSSOLO REGOLÍTICO Eutrófico típico				
P3	A	0–22	205	226	371	198
	AC	22–42	284	228	364	124
	C1	42–65/70	312	245	339	105
	C2	65/70–100+	290	262	344	104
		ARGISSOLO AMARELO Distrófico típico				
P4	A	0–25	296	554	113	37
	AB1	25–50	204	530	201	65
	AB2	50–80	130	553	234	83
	Bt	80–100+	23	464	392	121

P1 – NEOSSOLO LITÓLICO Distrófico fragmentário (RLd) /Dystric Leptosol (LPdv), P2 – NEOSSOLO REGOLÍTICO Distrófico leptofragmentário (RRd) /Leptic Regosol (RGle), P3 – NEOSSOLO REGOLÍTICO Eutrófico típico (RRe) /Eutric Regosol (RGeu) and P4 – ARGISSOLO AMARELO Distrófico típico (PAd) /Haplic Acrisol (ACha).

Table 4.
Granulometric characteristics of the toposequence profiles in the Arroio Pelotas, RS, Brasil.

At the top, profile P1 (**Figure 4**), the soil was classified as NEOSSOLO LITÓLICO Distrófico fragmentário, presenting a sequence of horizons A-A/CR/R-CR/R, an effective profunity less than 50 cm that configures with shallow soil, high base saturation on horizon A (>50%) and lithic contact between 50 and 100 cm from the surface of the soil.

On the top slope, profile P2 (**Figure 4**). The was classified as NEOSSOLO REGOLÍTICO Distrófico leptofragmentário, presenting a sequence of horizons A-AC-C-C/CR, an effective depth between 50 and 100 cm configuring as shallow soil, low base saturation along the profile (<50%) and fragmentary lytic contact at a depth greater than 50 cm and less than or equal to 100 cm from the soil surface.

On the middle slope the soil was classified, the P3 (**Figure 4**), in NEOSSOLO REGOLÍTICO Eutrófico típico presenting a sequence of horizons A-AC-C1–C2, an effective depth greater than 100 cm configuring as deep soil, high base saturation (>50%) and lithic contact greater than 50 cm from the soil surface.

The average results for soil density determinations, total porosity, macroporosity, and microporosity are presented in **Table 5**. Forest soil density varied between 1.00

Position on relief	Macroporosity		Microporosity		Total porosity		Density	
	0–5 cm	5–10 cm	0–5 cm	5–10 cm	0–5 cm	5–10 cm	0–5 cm	5–10 cm
	%						Mg m^{-3}	
Upper slope SG	31.4a	25.5a	27.8a	24.5a	59.2	50	1.00a	1.24a
Upper slope CG	19.6b	21.5a	37.9b	29.3a	57.5	50.8	1.09a	1.27a
Middle slope SG	28.2a	21.0a	21.1a	23.2a	49.3	45.1	1.06a	1.26a
Middle slope CG	18.9b	21.4a	31.2b	22a	51.7	50	1.21a	1.45b
Lowland (ciliar) SG	19.8a	18.8a	32.5a	31.6a	52.3	50.3	1.06a	1.20a
Lowland (ciliar) CG	12.0b	21.4a	41.4a	30.8a	52.1	52.9	1.15a	1.23a

SG: no access of cattle in the forest and CG: with access of cattle to the interior of the forest. Means followed by the same letter do not differ statistically from each other by the Kruskal–Wallis nonparametric test ($p < 0.05$).

Table 5.
Physical parameters of the forest soils in the 0–5 cm and 5–10 cm layers in toposequence in the Arroio Pelotas Hydrographic Basin, Rio Grande do Sul, Brasil.

Mg m^3 and 1.45 Mg m^3. Regardless of the position in the relief or access of the animals inside the fragments, the density was lower in the upper layer (0–5 cm) ranging between 1.00 Mg m^3 and 1.21 Mg m^3. While in the lower layer (5–10 cm), the density varied between 1.20 Mg m^3 and 1.45 Mg m^3. Although forest areas with animal access showed a soil density generally higher than the fragments without access, statistical difference was detected only in the lower layer (5–10 cm) on the middle slope. The critical density values found for sandy soils under agricultural use are cited and range from 1.6 to 1.8 [19], well above that found in this study.

This lower soil density in the upper layer (0–5 cm) was found in studies in natural forests [19, 30, 31]. This general pattern of natural or anthropic soils is caused by the greater intake of organic matter in the most superficial layer of the forest soil. The organic matter from the vegetation has a specific mass of lower density and greater aggregation power among the mineral particles of the soil, resulting in the better structural quality of the soils and, consequently, in the greater amount of pore space and space for root growth.

The lowest soil density in areas occupied by native forests has already been found in different Brazilian Biomes [15, 17, 19, 30–37]. In the same watershed, Flores et al. [32] found in Argissolo Vermelho, that the density of native forest and native field soil was lower than in the cultivation areas, with greater differences in the layer of higher organic matter intake (0 a 0.05 m). In this layer, the native forest had the lowest density (1.06 Mg m^{-3}) while no-till was the largest (1.45 Mg m^{-3}), and after five years, no-tillage had half the organic matter of the surface layer of the native forest [32].

In the studies of different Brazilian biomes that used native forests as control of edafica quality in agricultural properties, the density was always lower in native forests compared to more intense land uses, such as annual crops of conventional management, and no-tillage and pastures. It is important to highlight that the density values of the forest soil studied, in all glebes, they had density below the appropriate critical density for plant growth. However, densities close to or above the critical boundary should probably occur on the animal tracks within forest fragments.

The pattern of the lower density of forest soils compared to other forms of landscape use, such as crops, orchards, and pastures, is a consequence of the lower or no soil revolving in these forest environments. Thus, the decompaction of forest soil is largely influenced by the intense and constant activity of edafica fauna and woody plants with a large volume of the root system. Edaphic organisms, mainly microorganisms, have an important role to decompile biomass produced in the different strata of the forest and deposited in the litter, which is incorporated into soil organic matter and promotes better conditions in physical attributes, such as aggregation, porosity, and rate of water infiltration, energy flow, as well as chemical attributes such as nutrient cycling and increased cation exchange capacity [38] in forest soils.

Since then, soil density has been an indirect environmental indicator of the degree of particulate ness. For soils of similar classes, the higher the soil density, the higher its compaction, the lower its total porosity and, consequently, the greater the restrictions on the development of the root system of young plants, water infiltration, and soil aeration [5, 17].

The total porosity of forest soils ranged from 45% to 59%. However, macroporosity and microporosity had different distributions between the relief segments and between the soil layers. The areas without grazing presented nominally higher values in macroporosity and with the statistical difference in the three parts of the relief in the surface layer of the soil (0–5 cm). The forests with cattle access presented macropores in a smaller proportion, ranging from 12–20% in the upper soil layer (0–5 cm) and close values, in the three areas of the relief, to 21.5% in the lower layer (5–10 cm). In the protected areas of grazing, the proportion of macropores was between 19–31% in the 0–5 cm layer and between 18–25% in the 5–10 cm layer. On the other hand, the micropores were more abundant in forest areas with the entry of animals, between 31 and 41% of the porosity of the samples of the 0–5 cm layer and between 22–31% for the 5–10 cm layer. In the areas without the presence of the animals, the microporosity values were lower, between 21 and 32% in the upper layer and 23–32% in the samples of the lower layer. The microporosity was different between the areas with and without the cattle, however, even with higher microporosity values, in the lowered, there was a statistical difference between the middle and upper slope in the surface layer (0–5 cm). There was no statistical difference in the lower layer (5–10 cm) in all parts of the relief. The total porosity of forest soils ranged from 45 to 59% in the upper and lower layer.

Soil porosity is the intense result of biological activity and the diameter distribution of these pores reflects the condition to determine the physical-water behavior in soils [39]. In the study of the toposequence in Pelotas, the size of the pores indicates alteration due to the presence of animals inside the fragments of protective forests. The macroporosity of the soil surface layer was the parameter that was impaired by the compaction of cattle. The trampling of the animals is indicating a reduction in soil quality, being favorable for greater compaction of the surface layer, due to the increase in soil density and the reduction of macroporosity [40].

In relation to forest species, soil esthesity porosity is fundamental for seed germination and plant growth. A smaller volume of aerating space in the soil restricts the full development of the root system. In this sense, data on critical values for plant growth in different studies showed a minimum limiting value of soil aeration amount (macroposority) of at least 10% [8]. The toposequence soils in Pelotas, despite the loss of aeration space and water infiltration in the unfenced areas, showed macroporosity values above minimum environmental quality values. Even the forest fragments with the presence of the animals maintained adequate soil aeration conditions for plant growth with adaptations to grazing and animal trampling, such as carne-de-vaca (*Styrax leprosus*), embira (*Daphnopsis racemosa*), pitanga (*Eugenia uniflora*), *Cestrum*

(Cestrum parquii) e mamica-de-cadela (*Zanthoxylom fagara*), spiny vines, among other dominant.

Other land uses show less adequate conditions of aerated ground space. In different management systems of agricultural property in Pelotas, CRUZ et al. [41] found in conventional systems, no-tillage and native livestock field, lower aeration values, between 8 and 13% macroporosity for Argissolo Vermelho. Already evaluating soil quality in cultivated areas a few kilometers from the study, FLORES et al. [32] found greater aerate structure in the native forest and in the no-tillage system, 19 and 14%, respectively, already in the conventional planting system and in the native field with high livestock grazing, 8% macroporosity occurred. In Argissolo Vermelho distrófico in the same geological unit of this study, SUZUKI et al. [37] evaluated pasture, production forest, and forest of anthropized native protection in relation to compaction intensity. The authors found in the pasture with more than five years the lowest macroporosity (3.6%), while in the planting of *Eucalyptus saligna* with 20 years, the best macroporosity (19.9%). An intermediate macroporosity and practically below the minimum required plants occurred in the eucalyptus saligna forest in recent formation (four years) and in the anthropized native forest and used as shelter by cattle for five years [37].

4. Conclusions

The higher porosity of esthetic and water infiltration in forest soils, compared to other uses of the landscape, such as crops, orchards, and pastures, is the result of less soil revolving and its permanent vegetation cover. Forest vegetation constantly supplies organic matter to the soil through the litter intake of the treetops, the rhizosphere relationships in the root system, and its mycorrhizae relationships. The long time without soil revolving in production forests or protective forests occurs the action of organisms, plants, and animals in the incorporation of organic matter and the structuring of soil aggregates promoting improvements in forest soil quality by the stability of the environment.

The classification of the soils presented different classes in the toposequence, with Neossolos at the top and on the slopes and Argissolos in the lowlands of the relief, in the riparian forest. The presence of cattle inside the forest fragments, regardless of the position in the relief, indicates a tendency to promote negative effects and the loss of macroporosity in the surface layer of the soil. The soils of the forest fragments with the presence of the animals presented lower significant values of macroporosity in the surface layer, contributing to a certain compaction, compromising the distribution of porosity, and generating negative effects on soil physical quality.

This study evaluated the effects of cattle grazing on the physical quality of forest soils in the extreme south of the Atlantic forest, resulting in technical-scientific subsidies for the implementation of forest fragment isolation in landscapes planned to improve soil quality in the protected forests of watersheds. Thus, conservation by the isolation of protective forests in the planning of rural property benefits the quality of forest soils. Further studies are needed in the Arroio Pelotas watershed to characterize soil physics patterns according to the type of land use, to mitigate environmental impacts of livestock, and encourage a more sustainable activity.

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Conflict of interest

The authors declare no conflict of interest.

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Chapter 8

Drugs and Biodiversity Loss: Narcotraffic-Linked Landscape Change in Guatemala

Steven N. Winter, Gillian Eastwood and Manuel A. Barrios-Izás

Abstract

Characteristic of the Anthropocene, human impacts have resulted in worldwide losses in forested land cover, which can directly and indirectly drive biodiversity loss. The global illicit drug trade is one source of deforestation directly implicated with habitat loss in Central America, typically for drug trafficking and livestock production for money laundering. Given reports of deforestation in Central America linked to narcotraffic, we explored vegetation changes within Guatemala's highly biodiverse Maya Biosphere Reserve by examining trends suggestive of deforestation in a protected area. As such, we collected satellite-derived data in the form of enhanced vegetation index (EVI), as well as history of burned areas, published human-"footprint" data, official population density, and artificial light activity in Laguna del Tigre National Park from 2002 to 2020 for descriptive analysis. We found consistent reductions in EVI and trends of anomalous losses of vegetation despite a baseline accounting for variation within the park. Analyses revealed weak correlations ($R^2 \leq 0.26$) between EVI losses and official sources of anthropogenic data, which may be attributable to the data's limited spatial and temporal resolution. Alarmingly, simple analyses identified vegetation losses within a protected area, thus emphasizing the need for additional monitoring and science-based, but interdisciplinary policies to protect this biodiversity hotspot.

Keywords: deforestation, narcotrafficking, biodiversity loss, money laundering, habitat, enhanced vegetation index

1. Introduction

Drug trafficking and money laundering via livestock production contribute to deforestation directly implicated with habitat loss in Central America [1, 2]. In light of reports of deforestation in Central America being connected to narcotrafficking, we examined trends suggestive of deforestation within Laguna del Tigre National Park in Guatemala's highly biodiverse Maya Biosphere Reserve. Our explorations are generalizable and use openly accessible data sources, encouraging transference of our descriptive analyses to other study areas with limited resource availability or safety concerns precluding abilities to collect data *in situ*.

Global biodiversity is being lost at an unprecedented rate creating a critical environmental crisis [3]. The world's biodiversity is concentrated in hotspots–regions with spectacularly high levels of species richness and endemism. Nearly half the world's vascular plant species and one-third of terrestrial vertebrates are endemic to 25 such hotspots [4, 5]. Historically, biodiversity hotspots covered 12% of the land's surface, but even by 2003, their intact habitat covered only 1.4% of the land [6]. None of these hotspots have more than one-third of their original pristine habitat remaining [5].

Tropical forests account for the majority of all biodiversity hotspots [4]. Yet as well as the highest species richness, in tropical forests, we also find some of the most acute human pressures on natural environments [7]; the conservation of tropical forests being the most pressing biodiversity issue today. Tropical evergreen and deciduous forests spanned ~17 million km^2 globally but have now declined to ~11 million km^2 [8]. Such forests are expected to continue to shrink further this century, with 11–36% of forests existing in 2000 projected to disappear by 2050 [8, 9]. Main drivers of biodiversity loss in tropical ecosystems are related with the conversion of tropical forest to intensive agriculture or ranch cattle fields, forest fires, mining, logging, hunting, and illegal trade [10–14]. Over half of tropical or subtropical forests still in existence today have been substantially altered. Indeed, a quarter of the remaining tropical rainforest has been fragmented, with one-fifth of these forests selectively logged at some level from 2000 to 2005 [8].

Mesoamerica is one of three global tropical forest biodiversity hotspot regions that have lost much of their forested area over the last 30 years (others being Sundaland—all of Indonesia and Indo-Burma) [15]. Central America—part of the Mesoamerica region—covers only 2% of the world's territory but is home to 12% of the planet's biological diversity [16]. For example, the Central American country of Guatemala has 2779 species identified by the International Union for the Conservation of Nature (IUCN), of which 189 species are listed as threatened [17]. Unfortunately, research effort in this megadiverse country is poor, especially for hyper-diverse taxa, which has higher number of species and endemism in the tropics [4, 18], and biodiversity research is disproportionately focused on temperate regions [17].

Through means of drug-crop cultivation or trafficking, the global illicit drug trade is a major driver of tropical forest loss, termed "narco-deforestation" [2]. Drug-crop cultivation has played a disproportionately large role in deforesting and degrading some of the world's most biodiverse ecosystems, including those in national parks and indigenous territories [19, 20]. Drug trafficking is also associated with deforestation and habitat degradation in Central and South America [2, 21, 22]. However, due to their remoteness, large tracts of contiguous forest can equally be beneficial for concealing illicit activity, drawing narcotrafficking operations to biodiversity hotspots [23]. In general, land conversion to directly support trafficking efforts is a relatively small portion of the impact of the drug trade on biodiversity; however, as an order to launder profits, traffickers convert vast tracts of forest landscapes to agricultural enterprises, such as cattle ranches and oil palm plantations, as a modus operandi to conceal financial transactions [20, 24].

Cocaine trafficking in Honduras, Guatemala, and Nicaragua is estimated to account for between 15% and 30% of annual forest loss in these three countries over the past decade, and 30–60% of loss occurred within nationally and internationally designated protected areas [24]. Associated deforestation could be driven directly as a product of drug-movement (e.g., creation of illegal aircraft landing strips and pathways out) or indirectly via "drug-ranches"—a front of ranching and concealed money laundering as mentioned above.

Among Central America's protected areas, narco-deforestation has impacted Guatemala's Maya Biosphere Reserve and La Mosquitia, at the border between Honduras and Nicaragua; and to a lesser extent, the Jiquilisco region in El Salvador, the Osa peninsula in Costa Rica, and Darién National Park in Panama [16]. Cattle ranch laundering might be facilitated in northern Guatemala due to the low presence of law enforcement agencies for border security [25], increase of illegal economies and clandestine routes through the border [26], low control of cattle monetary transactions [1], the increase of narco-airstrips since mid-2000s [27], and corruption [28].

Here, we explore forest loss trends in Guatemala's Laguna del Tigre National Park—an area with a history of increasing narcotrafficking and the core zone of the Maya Biosphere Reserve, Guatemala's largest protected area complex. Recent large-scale drug trafficking interdiction, such as the US "War on drugs," has shifted international supply lines from previous shipping channels through the Caribbean or Pacific direct to Mexico, to now passage overland through Central America [29]. We identify spatial aggregations of deforestation, as well as their magnitude and extent using vegetation data derived from satellites. Our results demonstrate a fine-scale baseline of narco-deforestation in Guatemala's largest and most pristine protected area.

2. Methods

2.1 Study area

The Laguna del Tigre National Park (LTNP) is part of a conglomerate of protected areas together known as Maya Biosphere Reserve (MBR). This tropical humid forest extends through the Yucatan Peninsula in northern Guatemala, southeast Mexico, and northern Belize (**Figure 1**). LTNP occupies 3379 km^2 composed mainly of tropical rainforest, pastures, and wetlands [30], being one of the largest inland RAMSAR sites in Mesoamerica [31]. The altitude of the LTNP ranges from 60 to 182 m, the weather is hot (25–35°C) and humid (85%), and annual precipitation reaches 1629 mm. Within the National Park boundaries live already regionally or globally endangered vertebrate species such as jaguar (*Panthera onca*, (Linnaeus 1758)), cougar (*Puma concolor*, (Linnaeus 1771)), Geoffroy's spider monkey (*Ateles geoffroyi*, (Kuhl 1820)), Harpy eagle (*Harpia harpyja*, (Linnaeus 1718)), Mealy parrot (*Amazona guatemalae*, (Sclater 1860)), Morelet's crocodile (*Crocodylus moreletii*, Duméril & Bibron 1851), tapir (*Tapirus bairdii*, Gill 1865), scarlet macaw (*Ara macao*, (Linnaeus 1758)), Spectral bat (*Vampyrum spectrum*, (Linnaeus 1758)), and Yucatán black howler monkey (*Alouatta pigra*, (Lawrence 1933)) [32–34]. LTPN also hosts endangered tree species such as *Dalbergia stevensonii* (Standl 1927; IUCN: Critically endangered) and *Swietenia macrophylla* (King 1886; IUCN: Vulnerable) [35, 36]. In addition to wildlife conservation, Maya Biosphere Reserve is part of the Yucatan basin—the development center of Mayan civilization (approx. 800 BC–900 AD), holding several archeological sites, some of them recognized within the World Heritage Convention.

Massive immigration and extensive deforestation occurred in the Petén basin in the last three decades of the twentieth century [37]. Immigration to northwestern Petén was pushed during the 1980s by poverty and civil war [38], further, during the mid-1980s and 1990s, forests were cleared for establishing subsistence farming, roads, and maize [39]. Later in the early 2000s, forest clearance increased even more in northwest Petén, especially in LTNP, for the establishment of ranching fields

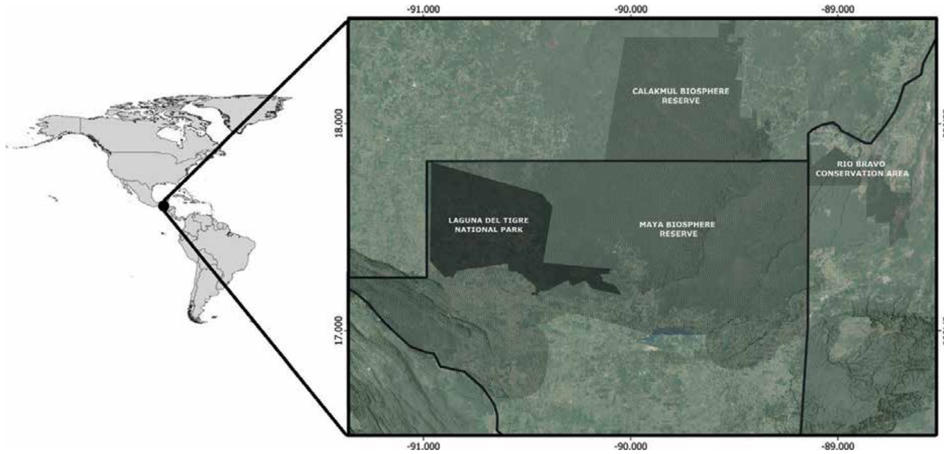

Figure 1.
Map of contiguous protected areas from Guatemala, Mexico, and Belize. Laguna del Tigre National Park (dark shaded) is part of a complex of protected areas immersed within the Maya biosphere reserve (2,090,667.00 ha) at western border. LTNP became part of the RAMSAR convention sites in 1990 for promoting the conservation of 335,060 ha. Illegal activities during the last three decades have been shifting natural ecosystems into ranchery fields.

related to illegal activities that overpass the capacities of governmental institutions in charge of the protection of the environment or cultural heritage (for example, Ministry of Environment, National Council of Protected Areas, or the Institute of Anthropology and History) [29].

2.2 Landscape data collection

Remotely sensed vegetation indices quantify spectral signatures (i.e., variations in wavelength reflectance) found in photosynthetically active radiation, and correlate with coarse scale landscape conditions that vary with soil type, precipitation, elevation, and temperature gradients [40]. Vegetation indices also correlate with vegetative primary productivity, which lend to their application in ecological research [41], and demonstrate the impact from land cover change on biodiversity [42]. The enhanced vegetation index (EVI) is a widely used measure of vegetation phenology that is sensitive to increased biomass and variation in canopy cover [43, 44], making it ideal in assessing land cover change in tropical landscapes [45].

The Moderate Resolution Imaging Spectroradiometer (MODIS) sensor onboard NASA's Terra satellite collects multispectral data worldwide, including data on vegetation phenology in the form of vegetation indices [46]. The MODIS sensor collects EVI data at 250-meter spatial resolution and 16-day temporal resolution. Values for EVI span from −1, representing no vegetation, to 1, corresponding to high vegetative biomass [44].

Fire regimes are natural in many terrestrial ecosystems [47]. Fires can fluctuate in tropical Central America because of climatic phenomena (i.e., El Ninõ-Southern Oscillation) inducing drought with varying magnitudes [48], and between ecoregions, with fires typically less common in tropical humid forests (i.e., the ecoregion of northern Guatemala [49]). Nevertheless, smaller fires more commonly associated with human activities have amassed to significant burned areas in Central America [48], including northern Guatemala [49].

The MODIS sensor can also quantify whether burn-sensitive vegetation has undergone uncharacteristically permanent changes, consistent with recent history of burning, and statistical algorithms have been developed to binarily categorize these areas as either "unburned" or "burned," with values of 0 or 1, respectively [50]. Data are collected on burned areas at 500-meter spatial resolution and averaged monthly [50]. We collected MODIS data at the finest available resolution (i.e., 250-meter resolution EVI and 500-meter resolution burned area) for Guatemala spanning years 2002–2020 collected seasonally using the *MODIStsp* R package [51] from April 1 until April 30, which correspond to the Guatemala's summer season, when clear skies result in low image obstruction due to clouds, rain, and oversaturation of soils reflecting artifactual water bodies.

Radiance from artificial, nighttime lights has been associated with human settlements [52]. In addition to being used to monitor expansions in urban development [53, 54], artificial light has been shown to identify local-level development in rural areas [55]. Artificial light data can provide information on human populations where official reporting may not be conducted or unreliable [55]. The Visible Infrared Imaging Radiometer Suite (VIIRS) sensor aboard NASA's Suomi NPP satellite records radiance observed from nighttime lights [52, 56]. Radiance is recorded as a continuous measurement of nanowatts per steradian per square meter (nW cm^{-2} sr^{-1}), where values range from 0 (no radiance detected) to 255 (highest radiance). We collected 2012 nighttime light data at 30-meter spatial resolution with the use of imagery from the NASA Worldview application [57].

Human influences in natural areas have been shown to impact biodiversity [58], enhance extinction risk [59], and promote colonization of invasive species [60]. The human footprint is a cumulative and multivariable metric used to quantify humans' indirect and direct effects on wilderness areas [61]. High human pressures have been quantified in areas with higher biodiversity [62], and a recent assessment identified that the majority of Guatemala was highly human-modified [63], emphasizing scarcity of wilderness areas in the country.

A standardized, quantitative analysis combined human-related pressures (e.g., major road and waterways, crop and pasture lands, built areas, and human population density, among other variables) to create a global map of the human footprint, which was validated via satellite imagery [62]. We utilized this published 1-kilometer spatial resolution rasters of the 2009 assessment of the global human footprint [62]. Human footprint is measured on a scale from 0 (no human influence) to 100 (highest degrees of human influence).

2.3 Data analysis

We prepared EVI raster data for descriptive analyses by cropping and masking data from MODIS tiles to the extent of LTNP. We generated 19 annual EVI rasters by averaging the two rasters collected from April 1 to April 30 from each year, then using these averages to represent annual EVI values spanning the 19-year study period (i.e., 2002–2020). All raster calculations were performed in R [64], using functionality in the *raster* package [65]. Visualizations were performed using the *raster*, *rasterVis*, and *plotfunctions*, and *rgdal* R packages [65–67].

We examined changes in annual EVI values over time by calculating the annual difference in annual EVI values from 2002 (i.e., the initial collection year), as well as in annual timesteps (e.g., 2008 EVI values subtracted from those in 2007). Next, we evaluated the magnitude of annual EVI change from 2002 by converting the differences in annual EVI values from 2002 into percentages of the initial values. That is, pixel-level percentages were calculated by subtracting each year's EVI raster values from 2002's raster values, then dividing their difference by the absolute value of the 2002 raster, and multiplying decimals by 100. We examined the distribution of areas with loss and gain in EVI by isolating net positive and net negative percent changes and averaging raster values.

Next, we examined each year's mean EVI values for anomalies (i.e., values deemed uncharacteristically high or low relative to variation during a defined baseline period). Specifically, we adapted a formula for calculating anomalies in climatic phenomena [49, 66, 68], which essentially calculates a z-score transformation common in statistics for understanding the measure of standard deviations that a sample is away from its mean [69]. Using the first 5 years (i.e., 2002–2006) as the baseline, annual anomalies (A_i) were calculated from subtracting each of the 14 annual mean EVI rasters during 2007–2020 ($\bar{X}_i$) from the baseline years' mean raster ($\bar{X}_b$) and dividing the difference from the raster characterizing the standard deviation in EVI during baseline years (σ_b) (Eq. (1)). We calculated a cumulative anomaly raster to understand the geographic distribution of the overall magnitude of anomalies and where uncharacteristic values have been found.

$$A_i = \frac{\bar{X}_i - \bar{X}_b}{\sigma_b} \tag{1}$$

Finally, we assessed fire and published human data for co-occurrence and relationships with changes in EVI. To understand fire data, we collected monthly MODIS and cropped images to the study area extent and summarized the magnitude of burned areas from 2002 to 2020 by calculating the sum of all binary (i.e., burned or unburned) rasters collected. Finally, we transformed continuous rasters into four categories of magnitude of burned areas (i.e., none, low, moderate, and high) using natural breaks (Jenks). We did not alter other published data sources. Percent EVI change rasters were examined for simple correlations between burned areas, 2009 human footprint data, and 2012 artificial light raster data within Laguna del Tigre National Park.

3. Results

Our results indicate there was considerable variation in the percent change of EVI in LTNP over the study period of 2002–2020. We found percent loss of EVI over broader geographic areas than percent gained, pixels of which were aggregated in the southern extent of the national park (**Figure 2A**). Similarly, for EVI percentages relative to 2002, the majority of values were negative for most (63%) years indicating negative changes (**Figure 2B**). When comparing differences in raw EVI values at annual timesteps, we noted considerable variation between years (**Appendix Figure A1**); however, the culmination of these differences manifested as signals of EVI loss consistently throughout the national park (**Figure 3**). We noted the geographic distribution of higher annual percent gain in reference to 2002 occurred along the southern extent of the park; however, high

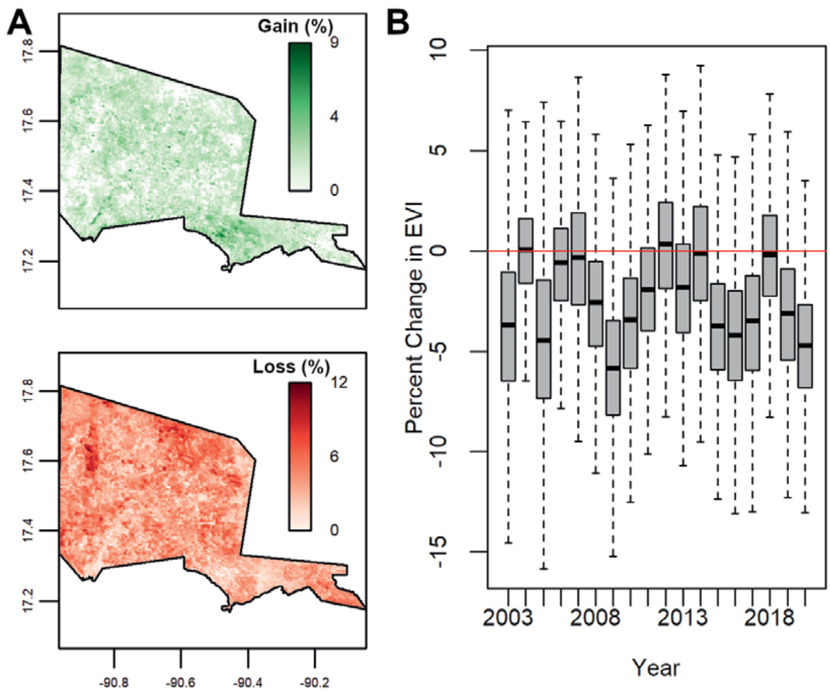

Figure 2.
Percent change in EVI from 2002 within Laguna del Tigre National Park. Annual EVI values from 2003 to 2020 were converted to percentages relative to 2002 and averaged into maps. A) Maps were isolated to increases (top) and decreases (bottom) in EVI percentage. Larger, widespread areas experienced more EVI percent loss (i.e., darker red colors in bottom panel) relative to clustered areas with strong increases in EVI (i.e., smaller dark green areas in top panel). B) Boxplots represent percent change in EVI values relative to 2002 (y-axis) grouped by year (x-axis). Red horizontal line represents no change (i.e., values consistent with 2002). Note the majority of values as denoted by boxplots are below the red line for nearly all years indicating negative changes in EVI percentages relative to 2002.

percent loss in reference to 2002 occurred throughout the national park, with more pronounced in areas adjacent to the Guatemala-Mexico border (**Appendix Figure A2**).

Trends were noted of negative anomaly values consistent with uncharacteristic EVI loss in most years. Some geographic similarities in anomalies were found between years (i.e., negative anomaly values were observed along the western boundaries of the park and Mexico border; **Figure 4**). Also, larger negative anomaly values were more geographically widespread than positive anomalies, which generally occurred in clusters. This negative trend has occurred in the majority of anomaly values in more recent years (**Appendix Figure A3**), consistent with negative percent change in EVI values previously noted (**Figure 2B**). When mapped, we identified negative anomaly values in the southwestern portions of the park with relatively smaller areas experiencing positive EVI anomalies (**Figure 5**).

Increased metrics of human activity or influence (i.e., artificial light and human footprint) were found in a location consistent with a local oil refinery and in areas that are uncharacterized for human development. Analysis of burned areas revealed that fires have been present throughout the national park during the study period with notable, consistent burning observed in the northeastern portions of the park (**Figure 6**). Nevertheless, we found weak relationships between both fire and

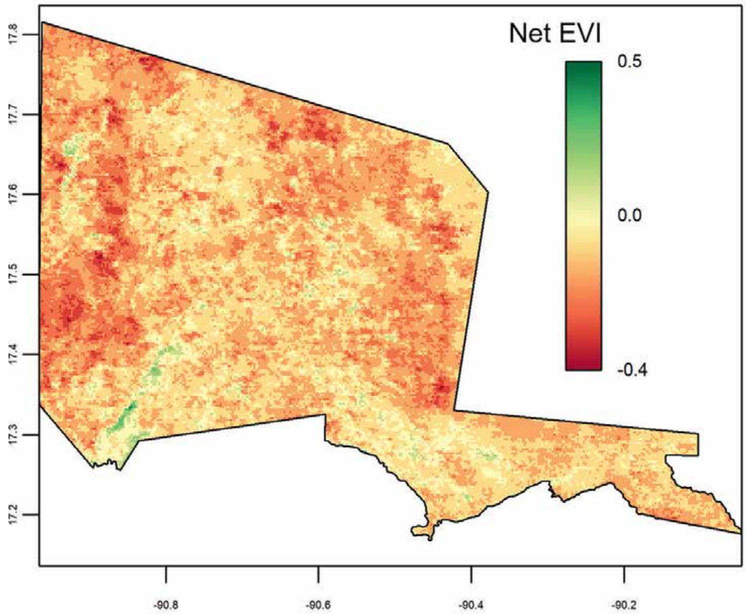

Figure 3.
Cumulative differences in raw EVI values from annual timesteps spanning 2002–2020. Annual differences (e.g., EVI values from 2004 subtracted from 2003) of raw EVI values were accumulated and summed to obtain net differences seen each year across the 18-year span. Considerable and consistent EVI loss was observed along the western border of Laguna del Tigre National Park.

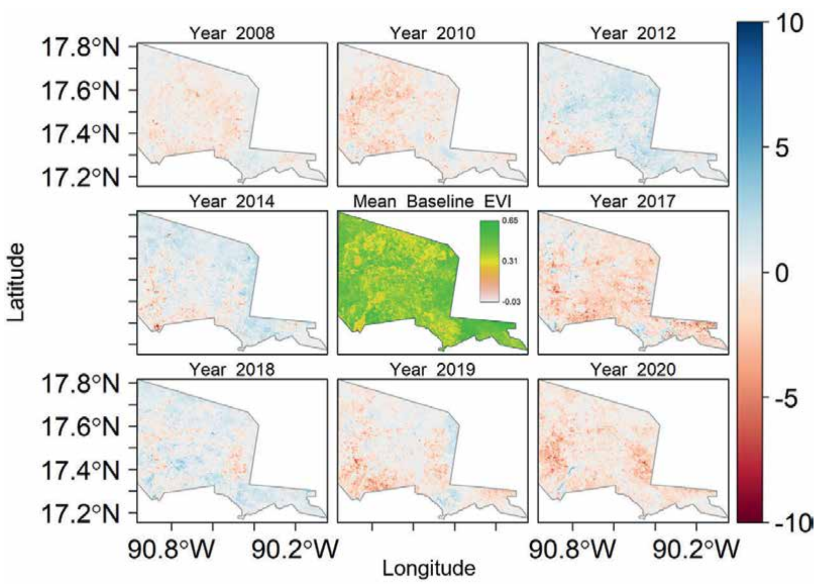

Figure 4.
Annual anomalies in EVI in Laguna del Tigre National Park. Small multiple level plots along the perimeter of plots show annual anomalies for even-numbered years. Middle plot shows the mean EVI values during the baseline (2002–2006). Anomaly values identify the number of standard deviations away from baseline variation (i.e., z-score) using years 2002–2006 as the baseline. We found negative annual EVI anomaly values (dark-red-colored pixels) were more abundant than positive anomaly values (dark-blue-colored pixels).

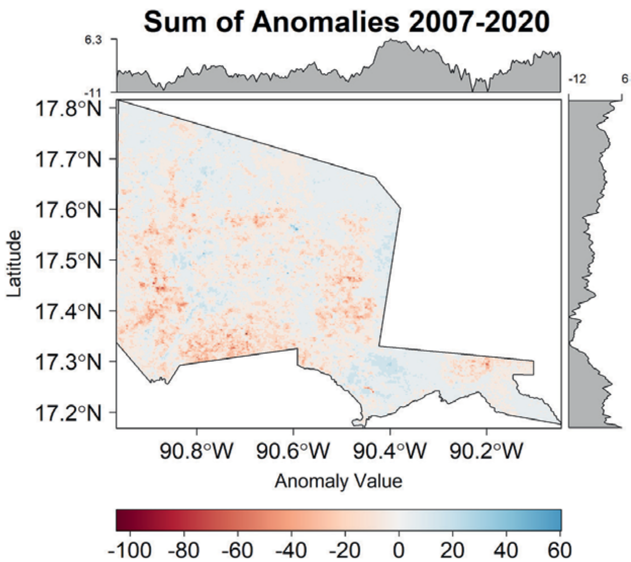

Figure 5.
Cumulative anomalies within Laguna del Tigre National Park. Map extent and black line show boundary of Laguna del Tigre National Park. Pixel colors show the magnitude of anomaly values seen from 2007 to 2020, where red colors represent areas with negative anomalies (i.e., areas experiencing negative EVI relative to baseline of 2002–2006), while blue-colored pixels identify areas with positive anomalies. We note strong signals of consistent negative anomalies in the southwestern portions of the park with relatively smaller areas experiencing positive anomalies consistent with higher EVI relative to the baseline.

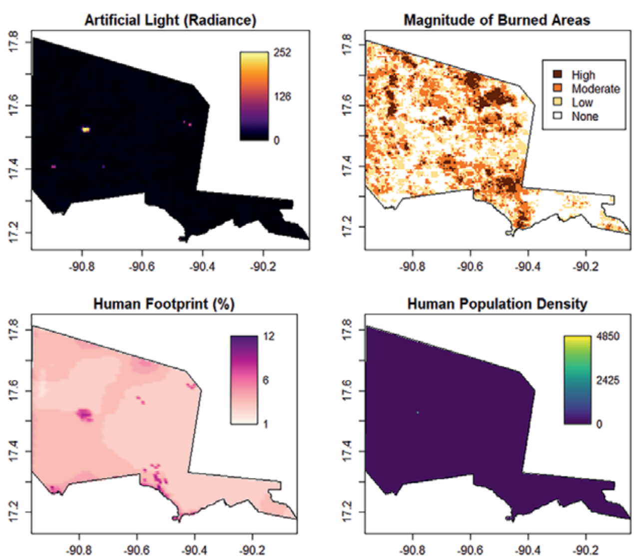

Figure 6.
Geographic distribution of variables associated with disturbance in Laguna del Tigre National Park, Guatemala. Four panels show signs of artificial light observed from nighttime light radiance (top left), magnitude of burned areas represented in categories (Jenks) (top right), percent of human footprint using methods from Venter et al. (2018) (bottom left), and human population density (bottom right). Note higher values near the center of the national park occurred at the site of a local oil refinery, while other areas of night light and burned areas were found in areas uncharacterized for human development.

published human data sources and percent change in 2020 EVI values from 2002 (R^2 values ≤ |0.26|) (**Appendix Figure A4**).

4. Discussion

Our analyses identified anomalous losses in forest cover within the national park. This finding demonstrates that biodiversity hotspot areas are experiencing vegetative losses consistent with deforestation even inside national parks. Previous reports noted forests are cut for clandestine roads and landing strips, and drug trafficking intensifies preexisting pressures on forests by infusing already weakly governed frontiers with unprecedented amounts of cash and weapons [2]. In Guatemala's Maya Biosphere Reserve, drug traffickers deforest the protected area to illegally ranch cattle, which serves as a mechanism of money laundering, drug smuggling, and territory control [1]. From a social-justice perspective, deforestation from drug paddocks is linked to flood disasters, without forests acting as a natural "shield" against extreme weather events. Thus, indigenous communities in Guatemala are becoming more vulnerable to natural disasters. Additionally, land of indigenous communities is taken by drug traffickers [16].

We identified consistent losses in EVI within Laguna del Tigre National Park since the beginning of the twenty-first century; some losses of EVI occurred in regions where MODIS sensors revealed a history of frequent burning, and anomaly calculations have identified that many changes within recent years are uncharacteristic for the region. From 2002, the initial year of data collection and earliest available time period to the turn of the millennium when drug trafficking intensified [70], we noted that losses in EVI were becoming increasingly common throughout much of the national park. We have shown this by examining both the changes in raw vegetation index values from 2002 and by accounting for the relative weight of these changes as percentages of their initial values. Further, regions where we observed dramatic EVI losses occurred along the periphery of the national park along the Mexico-Guatemala border. Our findings are consistent with previous studies confirming forest loss within the Maya Biosphere Preserve and Guatemala in general [24, 71]. Criminal activity in the reserve in Northern Guatemala intensified at the start of the 2000s, accelerating the destruction of the western half of the reserve. An important factor is that LTNP is ideally situated to refuel drug aircrafts flying from South America and transfer narcotics to trucks for the short drive to Mexico. Ranchers have built dozens of airstrips, including one dubbed the "international airport," which had three runways, and more than a dozen abandoned aircraft, which generated a loss of 40,000 hectares of forest [72]. Drug trafficking activities in northern Central America have been facilitated by corruption in the highest authorities of local and national governments; just in the past 5 years, former presidents, vice presidents, and Security Ministers have been prosecuted or are required for extradition by the US justice system [73–75].

In general, fires in Central America have been documented for their significant variability [49]. Our analysis revealed that many areas within the national park have experienced frequent fires over the nearly two decade-long study period, consistent with past studies [71]. Identifying direct causes of fires was outside of the scope of our analyses and thus remain unknown; however, fires within this ecoregion have been recently attributed to farming practices [49] and are mostly absent from areas where official sources report human populations. Indeed, the strongest source of artificial light, reported human populations, and highest human footprint corresponds to the local oil refinery located within the park. Yet, some burned areas and strong negative anomalies in EVI loss were observed in areas where mismatches exist between artificial light being detected and a lack of official reports of human populations (i.e., areas in the southwest and northeast corners in the park), plausibly suggesting illegal settlements and slash-and-burn activities [55, 71]. In Colombia, for example, approximately more than 1 million hectares of forest have been destroyed in the production of unlawful crops. It is estimated that for every hectare of coca, four hectares of forest are damaged, most often through a crude slash-and-burn farming technique. This deforestation causes soil erosion, among other environmental problems (Foundations Recovery Network). Such mismatches observed herein geographically and correlation-wise could emphasize the need for better data on monitoring human activities in remote regions.

Some variation in forest cover within dynamic and natural landscapes can be expected, which could be identified using EVI given the variable's close relationship with climatic conditions that occur in cyclic patterns in this region (i.e., El Nino Southern Oscillation; [40]). Still, our anomaly calculations identified that recent trends in EVI values extending beyond the assumed baseline years were uncharacteristically negative, suggesting losses of vegetation. Negative anomalies were most notable along the western extent of the national park, along the Mexico border. These findings are broadly consistent with a recent spatiotemporal assessment that identified significant anomalous forest loss in Guatemala [24], but our results differ in the areas in which anomalous losses were detected, likely resulting from differences in study areas. That is, Guatemala's highest anomalous forest loss was reported in central regions of the Department of Petén [24]—a region outside of our study area; we present anomalous changes explicit to LTNP.

Our analyses are strictly descriptive to quantify dynamics in EVI that have been observed over a nearly two-decade study period. We recognize that the analyses performed provide poor capabilities on inferring causation relating narcotraffic to observed land cover change; instead, linkages between the two are supported with previous studies [2, 24] and the well-known history of criminal activities within the park [70]. Generalizability and ease of accessing data sources encourage transference of our analyses to other study areas with limited abilities to collect data *in-situ* that may be constrained by resource availability in developing countries or logistics relating to safety concerns, such as LTNP. Instead of MODIS satellite data, LANDSAT imagery could be an interesting alternative to define changes with finer scale resolution and collect local human footprint metrics in lieu of coarse scale human data that did not yield clear relationships. Our analyses serve as a local assessment of land-use change from which additional modeling activities and investigations (e.g., integrating EVI in prediction models, such as [76]) are both warranted and critically needed.

Worryingly, indigenous communities in biodiversity hotspots are being displaced from their territories due to deforestation [77]. Since 2000, deforestation rates in Honduras, Guatemala, and Nicaragua have been among the highest in Latin America and the world; after 2005, the rates increased [2]. Further, assuming conservatively that tropical primary forests support two-thirds of the wildlife species within each group, tropical forest loss/degradation will result in global richness declines of 43.8% in ants, 29.9% in dung beetles, and 19.9% in trees [78]. Examining 10 different species groups, disturbed habitats were shown to include 41% fewer species than the undisturbed forests [79]. When biodiversity does not decrease by as much as expected, a closer look can show that disturbed ecosystems become dominated by widely dispersed, highly abundant, and often invasive species such as the pig (*Sus scrofa*), black rat (*Rattus rattus*), cane toad (*Rhinella marina*), i.e., substituting endemic species for damaging ones [79].

Loss of forest coverage risks modification of the micro- and macroclimates, affecting local and broader temperature regimes, humidity, precipitation, as well creating suitable ecological niches for propagation of disease vectors and the risk of emerging infectious disease [80]. Primary forests are irreplaceable for sustaining tropical biodiversity [7]; for example, even when forest consumption halts, the habitat loss inhibits recovery of plant diversity as forests regrow [81]. For both birds and mammals, the proportion of deforestation in both insular Southeast Asia [82] and Brazil's Atlantic Forest [83, 84] consistently predicts the numbers of threatened species [5]. Even minimal deforestation has had severe consequences for vertebrate biodiversity [3]. Unless new large-scale conservation efforts are put in place to protect intact forests, deforestation rates will not slow to avert a new wave of global extinctions [3]. Current scenarios for future forest loss predict bleak futures for critically endangered species; for example, 70% loss of Sumatran orangutans by 2030 [85], as well as driving a biodiversity crisis with abrupt declines in species richness if species loss reaches a threshold from forest loss [86].

Finally, aside from cocaine transit, it is unknown how these tropical forests are impacted by other drug commodities. For example, Latin America (most notably Mexico and Colombia, and to a far lesser extent, Guatemala) now accounts for most of the heroin supply to North America (most notably the United States) [87], while also supplying smaller heroin markets of South America. Guatemala, along with Costa Rica, also has sizable cannabis cultivation [88]. Solutions need to be found to tackle challenging issues concerning drug trafficking, such as governance corruption or reducing market drivers for the demand of both cattle (imported "narco-beef") and of cocaine.

5. Conclusion

Tropical forest coverage plays a vital role in climate regulation, biodiversity, and disease prevention. Following the analyses presented here, there are numerous avenues still to investigate and questions that need clarifying. For example, can such findings lead to more effective protection of park habitat given corruption in the current political climate? Will results encourage new funding opportunities to assess whether cocaine has entered protected park borders, or to study the extent to which drug crop eradication drives deforestation by progressively displacing drug farmers into new, more remote environments? [19, 20, 77]. Further steps should address how changes in forest loss associated with drugs can motivate policy for both conservation and enforcement, how governance corruption can be curbed to limit ties of criminal organizations with elements of police, army, government, and courts, and how landscape changes to primary forest can be reduced. Clearly, interdisciplinary approaches will be needed to understand and resolve conservation problems imposed by narcotraffic within Laguna del Tigre National Park; our analyses serve as an important step in characterizing ecological changes in this biodiversity hotspot.

Acknowledgements

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A. Appendix

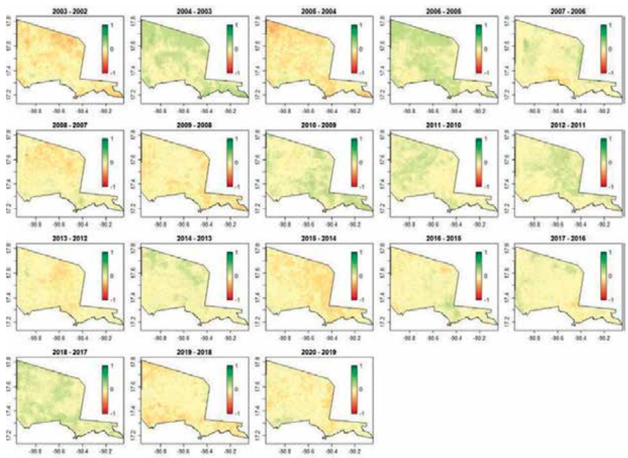

Figure A1.
Differences in EVI at annual timesteps from 2002 to 2020. Maps of Laguna del Tigre show differences in EVI in sequential annual timesteps. Red colors represent negative changes in EVI (losses), and green colors show positive gains in EVI from 1 year to the next. Each timestep displays considerable annual variation in EVI.

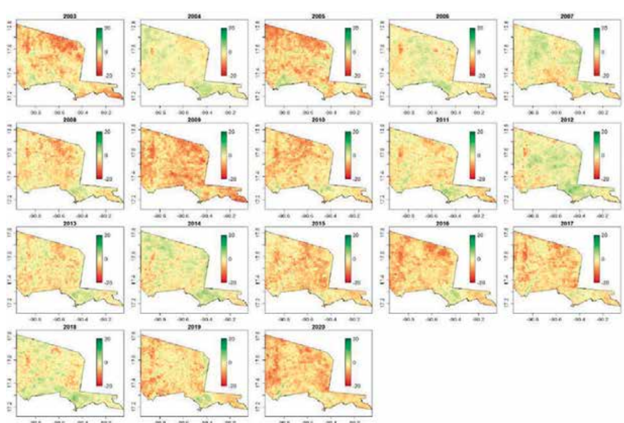

Figure A2.
Percent change in annual EVI from 2002. Changes in EVI were converted to percentages to show magnitude of change in corresponding years to EVI values observed in 2002. Colors indicate percentage increases (green) and decreases (red) in EVI relative to 2002. Note consistent and considerable decreases in EVI in the western portions of the national park.

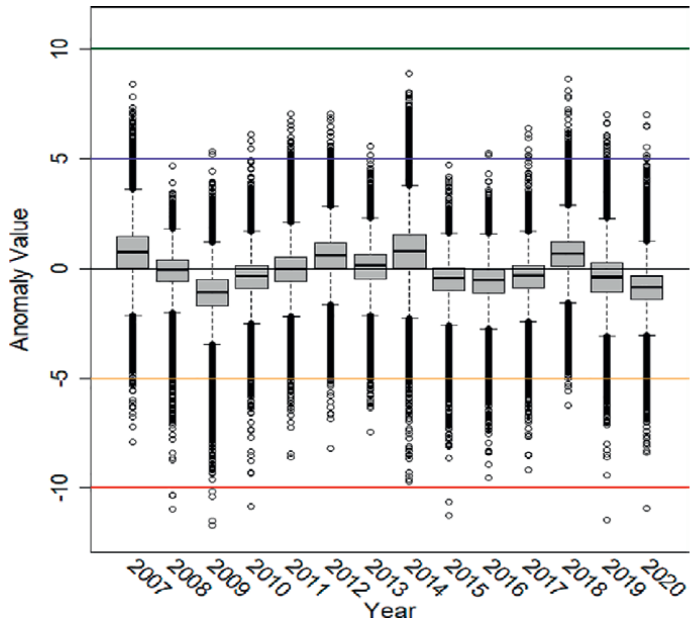

Figure A3.
Time series of anomaly values in EVI from Laguna del Tigre. Boxplots represent grouped anomaly values from yearly EVI rasters in Laguna del Tigre National Park. Years (x-axis) span from 2007 to 2020 extending beyond the baseline used for anomaly calculations (i.e., 2002–2006). Observed EVI anomalies were variable over time, though we note the high frequency of negative mean anomalies between 2015 and 2020 (83.3%).

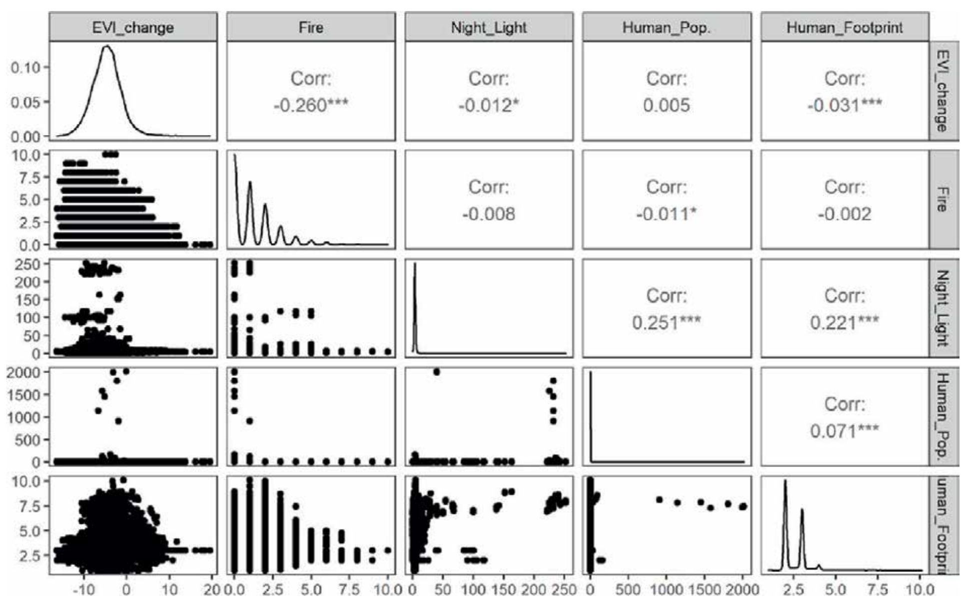

Figure A4.
Correlations between EVI change and human disturbance metrics. We found weak correlations ($R^2 \leq |0.26|$) between the percent change in EVI in 2020 from 2002, fire data, and published human data (e.g., artificial night light, human population density, and human footprint). Each subplot within pairs plot shows raw data (bottom diagonal), histogram density curves (diagonal), or pairwise Pearson's correlation coefficients (top diagonal) between variables indicated on top and far right panels.

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Human activities cause the degradation of innumerable ecosystems, including forests. The degradation of forests leads to far-reaching nutrient cycle changes throughout all environments. Reversible or irreversible degradation processes cause declines in an ecosystem's ability to self-regulate. Reversible processes decrease native forest biodiversity while preserving the ecosystem's abiotic components, whereas irreversible processes alter the character of the potential natural vegetation. This book discusses forest degradation in three sections on general, climatic, and land-use degradative effects.

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