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ALIMENTARIA Y DE BIOSISTEMAS

Land use changes and vegetation dynamics in a
silvopastoral system: Effect on their ecological
structure and carbon storage

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LOS VOCALES

EL SECRETARIO

To my Father

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ABSTRACT

The landscape in the old agricultural border in Nicaragua is a fragmented matrix in a livestock environment involving many different land uses with different tree cover and carbon storage. The municipality of Matiguás is a representative area of the livestock landscape in the sub-humid tropics. In order to assess land use and land cover (LULC) changes in Matiguás, the most representative traditional silvopastoral systems (TSPS) in the area were selected: shrubland, intervened secondary forest, pasture with high tree density, pasture with low tree density and degraded pasture. As an alternative to monitor and quantify these changes, remote sensing was used. In the last two decades, before Landsat images were free, developing countries could not afford monitoring through remote sensing because of the high cost of acquiring satellite imagery and commercial software. However, Landsat time series nowadays allows the characterization of changes in the vegetation across large areas over time. Nearly cloud free Landsat scenes: A Landsat 5 Thematic Mapper scene from 1986 and a Landsat 8 OLI scene from 2015 have been the datasets used in the study. A process chain following Jensen's (2009) four-step definition of the remote sensing process was conceptually developed and implemented, based on free open source software components and by applying the Random Forest algorithm. A conceptual LULC classification scheme representing TSPS was developed. Although the imagery shows a heterogeneous surface cover and mixed pixels, it was possible to achieve promising classification results with *out-of-the bag* errors below 13% for both images along with an overall accuracy level of 85.9% for the 2015 subset and 85.2% for the 1986 subset. The classification shows that from 1986 to 2015 (29 years) the intervened secondary forest increased 2.6 times while the degraded pasture decreased by 34.5%. The livestock landscape in Matiguás is in a state of constant transformation, but the main changes head toward the positive direction of tree cover recovery and an increase number of areas of natural regeneration.

Digging deeper into the analysis, each land use ecological structure was assessed in terms of tree richness and diversity index. Our study estimated the

above and belowground biomass carbon using allometric equations. Soil organic carbon was evaluated at four depths: 0-10 cm, 10-20, 20-40 and 40-100 cm. Of the land uses studied, shrubland had the highest diversity. The biomass carbon ranged from 1.9 to 13.2 t C ha⁻¹ for degraded pasture and intervened secondary forest, respectively. The highest soil organic carbon storage at 1 m depth was for intervened secondary forest (163 t C ha⁻¹), whereas degraded pastures had the lowest value (76 t C ha⁻¹), rather higher than biomass values. Since SOC is the largest pool of total carbon, its evaluation should be normalized down to a depth of at least one meter. Increasing tree coverage in degraded and low-tree density pastures can contribute not only to enhance carbon sequestration but also to restore degraded lands in livestock landscapes. Comparing the total biomass carbon stocks from 1986 and 2015 images, our results indicate an increasing trend in that period (24%).

The results indicated that Matiguás is characterized by a great dynamism in terms of land use changes with simultaneous processes of loss and gain of tree cover, with a net balance of gains. The transformation of the landscape does not mean a reduction of livestock areas, but rather, a change in the concept of livestock as we know. Natural regeneration is playing an important role, being a low cost solution to change land use, improving the tree cover in pasture and boost with the conservation and protection of aquatic resources, without a capital inversion.

Keywords: Silvopastoral systems, livestock landscape, Random Forest algorithm, free open source software, Pastures, Carbon sequestration, Land use change Sub humid tropical conditions Soil organic carbon, Nicaragua

RESUMEN

El paisaje en la antigua frontera agrícola de Nicaragua es una matriz fragmentada dentro de un entorno ganadero con usos de la tierra con distinta cubierta arbórea y almacenamiento de carbono. El municipio de Matiguás es una zona representativa del paisaje ganadero tropical sub-húmedo. Para evaluar los cambios en el uso de la tierra en el municipio de Matiguás, se seleccionaron los usos silvopastoriles tradicionales más representativos de la tierra en la zona: matorral, bosque secundario intervenido, pasto con alta densidad de árboles, pasto con árboles bajos densidad y pastos degradados. Como alternativa de monitorización y cuantificación de cambios, se utilizaron técnicas de teledetección. En las últimas dos décadas, antes de que las imágenes Landsat fueran de libre acceso, los países en desarrollo no podían permitirse el monitoreo a través de la teledetección debido al alto costo de adquisición de imágenes satelitales y software comercial. Sin embargo, la serie temporal de Landsat hoy en día permite la caracterización a través del tiempo de cambios de vegetación en zonas amplias. Para este estudio se han utilizado imágenes de Landsat casi sin nubes: una imagen Landsat 5 Thematic Mapper de 1986 y una imagen Landsat 8 OLI de 2015. Se desarrolló conceptualmente una cadena de procesos siguiendo la definición de cuatro pasos de Jensen (2009) del proceso de teledetección, y se implementó mediante componentes de software libre y la aplicación del algoritmo Random Forest. Se desarrolló un esquema de clasificación conceptual de usos de la tierra que representa a los sistemas silvopastoriles tradicionales. Aunque las imágenes muestran una cobertura de superficie heterogénea y píxeles mixtos, ha sido posible obtener resultados de clasificación prometedores con errores *out of the bag* (OOB) por debajo del 13% para ambas imágenes junto con un nivel de precisión global de 85.9 % para el subconjunto 2015 y 85.2% para el subconjunto 1986. La clasificación muestra que de 1986 a 2015 (29 años) el bosque secundario intervenido aumentó 2.6 veces mientras que el pasto degradado disminuyó en un 34.5%. El paisaje ganadero en Matiguás se encuentra en un estado de constante transformación, pero los principales cambios se dirigen hacia la

recuperación de la cubierta arbórea y al aumento en el número de áreas de regeneración natural.

Profundizando en el análisis, se evaluó la estructura ecológica de cada uso en términos de riqueza de árboles e índice de diversidad. En el estudio se estimó el carbono de la biomasa arbórea utilizando ecuaciones alométricas. El carbono orgánico del suelo se evaluó a cuatro profundidades: 0-10 cm, 10-20, 20-40 y 40-100 cm. De los usos de la tierra estudiados, el tacotal tenía la mayor diversidad. El carbono de la biomasa varió entre 1,9 y 13,2 t C ha⁻¹ en el pasto degradado y el bosque secundario intervenido respectivamente. El valor más elevado de almacenamiento de carbono orgánico del suelo a 1 m de profundidad se obtuvo en el bosque secundario intervenido (163 t C ha⁻¹), mientras que el pasto degradado tuvo el valor más bajo (76 t C ha⁻¹), en todos los casos muy superiores a los valores de carbono de la biomasa.

Dado que el carbono orgánico del suelo es el reservorio de C de mayor importancia, debería normalizarse su evaluación hasta una profundidad de al menos un metro. El aumento de árboles en pasturas degradadas y de baja densidad de árboles puede contribuir no solo a mejorar la captura de carbono sino también a restaurar las tierras degradadas en los paisajes ganaderos. Al comparar el carbono total de la biomasa arbórea en las imágenes de 1986 y 2015, se observa una tendencia creciente en ese período. Los resultados indicaron que Matiguás se caracteriza por un gran dinamismo en términos de cambios en el uso de la tierra con procesos simultáneos de pérdida y ganancia de cobertura arbórea, con un saldo neto de ganancias. La transformación del paisaje no significa una reducción de las áreas ganaderas, sino más bien un cambio en el concepto de ganadería. La regeneración natural está desempeñando un papel importante, ya que es una solución de bajo costo para cambiar el uso de la tierra, mejorar la cobertura arbórea en los pastizales e impulsar la conservación y protección de los recursos acuáticos, sin una inversión de capital.

Palabras clave: Sistemas silvopastoriles, paisaje ganadero, algoritmo Random Forest, Software libre, Secuestro de carbono, pasto, cambio de uso de la tierra, Condiciones sub húmedas tropicales, Carbono orgánico del suelo, Nicaragua

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Abbreviations

AGB: Above ground biomass

BA: Basal area

BGB: Belowground biomass

C: Cloud

CART: Classification and regression trees

CIB-10: Controlled Image Base 10 m

CO₂: Carbon dioxide

DBH: Diameter at breast height

DN'S: Digital numbers

DOS: Dark Object Subtraction

DP: Degraded pasture

EC: Error class

F: Forest

FOSS: Free open source software

GEF-WB: Global Environment Facility and World Bank

GIS: Geographic Information System

GR: Geometrical resolution

IDs: Images scene identifications

ISF: Intervened secondary forest

J-M: Jeffrey-Matusita

Landsat: Land satellite

LULC: Land use land cover

M: M-Statistic

MMU: Minimum mapping unit

MP: Map scale

OLI: Operational land imager

OOB: Out-of-the bag errors

PAST: PAleontological Statistics

PES: Paying for environmental services

PHD: High tree density pasture

PLD: Low tree density pasture

RF: Random forest

RIF: Rural infrastructure

RISEMP: Regional Integrated Silvopastoral Approaches to Ecosystem
Management Project

S: Shadow

SAR: Synthetic aperture radar

SCP: Semiautomatic classification plugin

SL: Shrubland

SOC: Soil organic carbon

SPOT: Satellite pour l'observation de la terre

SPS: Silvopastoral systems

TAO: Top of Atmosphere reflectance

TBC: Total biomass carbon

TM: thematic Mapper

TOF: Tree out forest

TSOC: Total soil organic carbon

TSPS: Traditional silvopastoral systems

USGS: United States Geological Survey

WB: Water body

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INTRODUCTION AND OBJECTIVES

1. Introduction

1.1. Fragmented livestock landscapes as a result of deforestation

Landscape is a natural and anthropogenic mosaic of landforms with complex biological interactions between physical, social and economic factors (Forman 1995). The relative function of such interactions depends upon the perspective and specific needs of each organism (Wu 2007). The patterns in this kind of landscape are determined by the structure and spatial distribution of the various elements, such as patches, corridors and matrices in a given area (Forman and Godron 1986). Patches are surface areas that differ in composition and structure from neighboring areas; corridors are basically narrow lines with a particular composition that differs from that of the adjacent area on either side, and matrices are commonly the widest part of the landscape and they are highly connected with other patches (Turner et al. 2001). Additionally, ecological objects such as animals, plants and soils are considered components of the landscape and are found distributed among the various landscape elements, while they vary in size, shape, number, type and configuration (Forman and Godron 1986).

A particular landscape can be characterized by four basic features: composition, structure, management, and regional context (Harvey et al. 2006; Scherr and McNeely 2008). Changes in both land use and land cover (LULC) transform not only the spatial distribution of the various landscape components but also alter the ecological processes and patterns of biodiversity (Turner 2001). The removal of native vegetation alters energy exchange processes, which are dependent on the amount of water and carbon that is exchanged between the soil surface and the atmosphere. This transformation reduces the strength of important elements of the natural resource base such as species biodiversity, soil

and water (Harvey et al. 2006).

Deforestation is one of the most important landscape transformations in the tropics, and has been taking place to fulfill human needs such as the establishment of farms, pastures, the extraction of timber for construction and fuel (Wright 2010; Wunder 2001). It is especially problematic in Central America because the region has one of the highest levels of biodiversity per square kilometer in the world due to its great diversity in climates, soils and altitudes (Reid and Miller 1989). The so called hamburger-connection in the 1960s and 1970s, with strong markets for meat and dairy products, stimulated livestock expansion in Central America with the consequent deforestation of thousands of forest areas. In the 70s the social tightness, provoked by a deep political crisis, increased the poverty gaps and the ecological pressure. In addition, the shift to socialist systems implemented cooperative management of natural resources and production. Notwithstanding, one decade later, the decline of export beef prices and the economic emergency in a war period, promoted continuous land abandonment and with this, the vegetation recovery in some areas started (Kaimowitz 1996). Many development projects were introduced focused on improving livestock production, thus causing pressure on the landscape. These development and research projects were directed to Matiguás municipality as a geographically strategic area, serving as a dry port between the Pacific and the Caribbean of the country, improving access roads and consolidating itself as the most important dairy area in the country (Polvorosa 2013).

The rate of deforestation decreased in the 1980s in Central America, but remained elevated, and in some cases increased, due to less violence, and better roads for transportation in forest regions. Already in the 90s, with peace restored, the cooperative system fell out and started a process of land reacquisition and with this, a degradation of natural resources, strongly accelerated by the pressure of displaced people, the structural adjustment policies (Baumeister and Fernández 2005) and globalization (Kull et al. 2007). From the second half of the 1990s, Central American rural landscape has been reconfigured, promoting and supporting the conservation and management of remaining forest patches (Hecht et al. 2014) by means the creation of co-

managed protected areas (Ravnborg 2008) and also the introduction of payments for environmental services (Pagiola et al. 2007) as a tool. Deforestation in Central America is still going forward, being singularly extreme in Honduras, Nicaragua, Guatemala and El Salvador (Saubes and Gálves 2015). Between 2000 and 2015 the forest cover fell throughout Central America, except Costa Rica (ERCA 2016). As a result, most tropical landscapes now consist of fragmented ecosystems with small patches of dense forest and logged areas surrounded by pastures (Alvarado et al. 2017, Tobar & Ibrahim 2010, Harvey et al. 2003).

With respect to Nicaragua, from the 13.04 million hectares of total surface, forest cover occupied 7 million hectares (54%) in the 50's, decreased to 4,5 million hectares (34,5%) in the 70's and lowered somewhat in the 90's to 4,1 million hectares (31,4%) (Kaimowitz 1996). In 2015 forest covered 25.9 % of the national territory (ERCAI 2016).

The consequences of such deforestation are devastating and represent an ongoing environmental degradation (McGroddy et al. 2015; Kaimowitz 1996). They include reducing the regional biodiversity with extinction of plant and animal species, decreasing evapotranspiration, increasing water runoff, contributing to water pollution and increasing the emission of greenhouse gases, such as methane (CH₄) and carbon dioxide (CO₂) to the atmosphere (Bennetzen et al. 2016) (Lindsey 2007; Swann et al. 2015). Tropical forest deforestation can lead to changes of temperature and rain patterns in other geographical regions (Lawrence and Vandecar 2015). Deforestation and environmental degradation entails not only the loss of soil and nutrients and desertification, but also puts human survival at global risk level (Zegeye 2017).

In this scenery, Nicaragua is the poorest country in Central America and the second poorest in Latin America and the Caribbean (World Bank 2014). It has 135,000 livestock farmers, 90 % of which are classified as smallholders. In 3.26 million hectares of land they raise approximately 5 million head of cattle, being the largest herd in Central America with a production of 768,000 metric tons of milk/year (Thecnoserve 2016, Lie and Rich 2016).

1.2. Land-use changes in livestock landscape

Commonly livestock landscapes are fragmented areas, composed mainly of pastures, with a few small patches of dense forests, connected by live fences. The amount of tree covers present in a particular pasture area is determined mainly by the farmer, who decides whether to plant, retain or remove trees in the landscape. Spatial distribution of these trees is determined by the degree of transformation undergone by a particular landscape (Forman 1995). According to Useche (2011), a typical livestock landscape in Central America is highly fragmented, being covered by forest 10% of the total area, only 2% of which is greater than 50 ha. Despite the degree of landscape fragmentation due to the expansion of livestock, traditional practices such as the implementation of live fences serve to create complex networks that connect more than 50% of forest patches acting as structural corridors reducing the degree of isolation between them (Lawrence 2004). Cattle raising needs to change the current trend of degradation of forested areas into one which generates livestock goods but also provides ecosystemic services. In order to successfully implement such a transformation, the tree cover density in pastureland must increase, soil degradation should be curbed and water sources should be protected, while efficient animal productivity rates can also be attained (Murgueitio et al. 2011).

The high rate of population growth increases the global demand for food, fuel and fiber. In countries with forests, the rapid rate of population growth increases the deforestation pressure for the conversion of natural areas to crop and cattle ranching land (Foley et al. 2011). Abandoned rural communities and the expansion of pasture and agricultural lands in tropical regions are both consequences of migration, in which the landscape is continuously modified by human activities, causing a loss of balance in the natural forests (Chazdon 2008; Lugo 2009; Hassler et al. 2011).

Land use changes in Latin America and the Caribbean, play an important role in the biodiversity conservation, carbon stock and preservation of protective areas (Aide et al. 2013). In Central America the landscape is very dynamic and is dominated by pasture mainly in state of degradation, without productivity, neither resistance or resilience to climate change (Sepúlveda and Ibrahim 2013). In previous decades, armed conflicts pushed Central Americans northward (Adams and Cuecuecha 2010). Many migrants did not have intention to return, but others went back to their native communities and brought their experiences about different ways of practicing agriculture back with them, together with savings that were able to invest in land (Davis and López-Carr 2014). Many studies have documented extensive deforestation as mentioned previously, but other studies report forest recovery. This contrasting trends can be due to demographic and socioeconomic changes. Migration can induce reforestation through natural regeneration (McDonald 2008) particularly in regions too steep or dry for modern agriculture (Grau and Aide 2008). On the other hand, social facts such as the growing global population, increasing per capita wealth, and the increasing global demand for agricultural products, can be important drivers of land change (Aide et al. 2013).

According to the Global Forest Resources Assessment 2000 (FAO 2001), Nicaragua had 6.2 million ha of forest in that year. Of those, 1.8 million ha were assigned to conservation and 4.4 million ha were productive forest. In the second half of the last century, the total amount of pastureland in Nicaragua grew from 0.8 million ha in 1950 to 4.2 million ha in 1997 (Szott et al. 2000), largely as a result of the clear-cutting of forest areas. The advancing agricultural frontier and cattle raising has consumed 1.8 million ha of forest and it constitutes a social conflict related to widespread poverty in the region. Indeed, more than a thin line separates the forested from the agricultural land. Till 2016, Nicaragua lost 1.11Mha of tree cover (GFW 2018). One of the main drivers of land use changes in Nicaragua the cattle raising activity which is concentrated in highly fragmented landscapes comprising small patches of forest within a matrix of pastures and other types of agricultural land use (Medina et al. 2007; Betancourt et al. 2003). Conversion of forests into other types of land use due to the current

economic pressure (Harvey et al. 2005a; Lamb et al. 2005) was high in most countries in Central America. Unlike other Central American countries Nicaragua did not experience a significant expansion in non-traditional intensive crop farming but livestock rearing was the main force behind land use change. Such changes in land use patterns impact not only the dynamics of the forest, but also have a global effect on the functionality of the ecosystems (Sánchez 2005) as well as on environmental services such as carbon sequestration and climate change mitigation.

The knowledge of the land use dynamics, particularly those affecting the TSPS, and their spatial distribution are important for implementing a monitoring and management system of natural resources (Mortelliti et al. 2010) as well as for evaluating the impact on provided the ecosystem services.

As mentioned before, a possible solution to this situation would be to increase areas of tree vegetation and to increase the connections between existing patches of forest (Kattan and Alvarez 1996; Murcia 1995 and Hobbs 1993). The natural regeneration has played an important role in the process of landscape restoration in Nicaragua, since it does not require any investment and improves vegetation diversity and density, which also serves as habitat for wildlife and as connectors between forest patches, carbon stock, water resources, and enhance the structural complexity within the agricultural landscape (Chazdon and Guariguata 2016; Manning et al. 2009; Harvey et al. 1998). Trees distributed in pasture belong to the category of "Trees outside Forest" TOF, which refers to trees on land not defined as forest and other wooded land. This may include agricultural land, meadows and pastures, or commercial and residential land. TOF can be productive such trees in agroforestry systems or be integrated into a livestock landscape as pastures with low and high tree density, shrubs, live fences, etc. where they perform a function (FAO 2001). There is an economic and ecological overlap between the functions of forests and TOF. Both of them make similar contributions, provide habitat for wildlife and help to preserve species of flora and fauna, protect water resources, serve as carbon storage etc. The difference lies mainly in the magnitude and specification of some functions provided by the forest (Kleinn 2000).

1.3. Silvopastoral systems

In the tropics, livestock production occurs in fragmented landscapes comprising a few, small fragments of dense forest surrounded by pastures. The sizes of forest fragments change within a wide range, but the tendency is to decrease their areas because of growing agricultural activities and human establishments (Chacón 2003). The Silvopastoral system (SPS) constitutes an alternative for cattle production where wooded perennial (trees and/or shrubs) interact with the traditional components (herbaceous covers and animals) under a system of integral handling (Ibrahim 1996). This is a system of sustainable production that, through transformations that improve the performance in production, generates environmental services while protecting and conserving the sources of water (Ibrahim et al. 2003). The silvopastoral system provides a more attractive landscape compared with the traditional pasture and they are less susceptible to create environmental problems related to the quality of water, diseases and by treatment to the animals (Agriculture Department USA 1997). Well designed and managed silvopastoral systems are win-win technologies and enhance productivity, provide environmental services, and may play an important role in retaining trees (Ibrahim et al. 1999; Gobbi 2001). The silvopastoral systems located between forest patches, like live fences, can serve as biological corridors between forest patches, thereby connecting populations, communities and ecological processes (Gascon et al. 2004; Laurence 2004; Schroth et al 2004); all these contribute to conserve biodiversity (Schroth et al. 2004). Silvopastoral systems are divided into two categories, improved silvopastoral systems and traditional silvopastoral systems, which are the approach of this research.

1.4. Traditional silvopastoral systems

Traditional silvopastoral systems (TSPS) are defined as the integration of tree remnants and shrubs in pastures with animals for economic, ecological and social sustainability (Montagnini et al. 2013). They constitute a new approach to

addressing agricultural landscapes based on an ecologically sustainable management within a landscape influenced by human activities. The aim is to create a resilient landscape where incentives and opportunity costs for the stakeholders are integrated, rural livelihoods, biodiversity, human environment interactions, and restoration ecology play an important role (Harvey et al. 2008, Chazdon et al. 2009).

Traditional silvopastoral systems usually are an alternative for the livestock farmers in Latin America. Well implemented, they improve the livestock production, through the animal welfare provided by the shadow in the paddocks at the grazing time, while protecting the soil from packing and erosion (Pagiola et al. 2004). At the same time TSPS enhance the reproductive parameters and the milk and meat production (Yamamoto et al. 2007; Betancourt et al. 2003). Traditional silvopastoral systems constitute a sustainable alternative for cattle production since they allow wooded perennial species (trees and/or shrubs) interact with the traditional components (herbaceous cover and cattle) under a system of integral handling (Ibrahim 1999). As mentioned above, this sustainable production system improves production and producer incomes, while it mitigates global warming (Ibrahim et al. 2003; Andrade and Ibrahim 2003) and conserves biodiversity (Jose 2009).

A pasture handled with a low load of cattle (0.5 and 0.6 animal units) retains more water than a forest ecosystem, due to the interception of water by grains (growth habit), allowing the infiltration of a greater amount of water in the soil (Ibrahim and Camargo 2001).

Land uses such as pasture with tree in high density have high values of biodiversity (Ibrahim et al. 2010) and contribute to the restoration of the structure and composition of the livestock landscapes supporting more diversity of species, and connecting protected areas. They are performing a major role in the survival of wildlife species (Pagiola 2004). One of the benefits of incorporating scattered trees in pasture is purely economic, and is reflected in the production of timber products as, for example, posts and firewood. This can serve to increase the amount of income generated by farms from 15 – 35 % (Holmann and Estrada

1997; Botero et al. 1999). Some stockbreeders use dispersed fodder trees to subsidize cattle diet during the dry season, when the quality and availability of grassland is low. Among fodder trees species commonly used are *Guazuma ulmifolia*, *Gliricidia sepium*, *Pithecellobium saman* and *Enterelobium cyclocarpum* (Zamora et al. 2001; Casasola et al. 2001).

In summary, TSPS are a better alternative to conventional pasture because they not only provide economic benefits to farmers, but also provide global environmental services (Pagiola et al. 2005) such as global warming mitigation (Noble et al. 2000; Andrade and Ibrahim 2003) due to their greater capacity for carbon sequestration (McGroddy et al. 2015) or biodiversity conservation (Jose 2009).

Traditional silvopastoral systems are an old practice and are very well spread in tropical countries as Argentina, Brazil, Chile New Zealand, United states of America, Paraguay, Uruguay, Colombia, and throughout the Central American isthmus (Cubbage et al. 2012). In Europe they happen in the Mediterranean region, but more as a conservation gadget. They are also used as grazing and browsing resources for shepherd and wild livestock (Ayanz 2005). But the largest distribution of TSPS still is in tropical areas. The broad spread out of the TSPS in the tropical livestock landscapes indicates a general trend to changing towards the growth of tree cover. In most cases they are inside agricultural landscapes (agroforestry systems) and have a strong impact on the climate change (Hecht 2014; Zomer et al. 2016).

Traditional silvopastoral systems may consist of trees dispersed in pastures with a variable density, or a scrubland. Tree density in a given pasture is dependent on a number of variables, including land use, the specific production system employed, resource demand and pasture management. Scrubland, also called, shrubland, heathland, or chaparral, is a diverse assortment of vegetation types sharing the common physical characteristic of dominance by shrubs (Britannica.com). Shrubs are commonly defined on the basis of stems characteristics as intermediate in the continuum from herbs to trees. There are many definitions of shrubs and it depends on the investigator and regional

context (Wilson 1995). In this case it is defined as native vegetation in natural succession with a height of less than 5 m. Shrubs, according to Murgueitio et al. (2011), will play an important role improving livestock landscape. Also the incorporation of shrubs in the landscape can increase the amount of carbon sequestered compared to a monoculture field of crop plants or pasture (Kirby and Potvin 2007).

1.5. Traditional silvopastoral systems and carbon storage

Terrestrial ecosystems play an important role in the amount of carbon that is fixed globally. The main carbon sinks in terrestrial ecosystems are found in tropical forests (62%), while approximately 54% of the global soil carbon stock is found in the forests of higher (northern) latitudes (Kanninen 2000). It has been observed that the amount of carbon stock varies from 60 to 230 t C ha⁻¹ in primary forests, and from 25 to 190 t C ha⁻¹ in secondary forests (Gardner et al. 1999). The transformation of natural ecosystems releases CO₂ stored in the vegetative biomass and soil into the atmosphere; the amount released is determined by the quantity of carbon stored in woody vegetation and soil organic matter (West et al. 2010). Given that tropical forests store greater amounts of carbon than other ecosystems (Chaturvedi et al. 2011), land use changes with the conversion of forest to agricultural systems have a significant impact on global greenhouse gas emissions (Solomon et al. 2007; Bernstein et al. 2008). Peters et al. (2012, cited by Harris 2012) emphasizes that, although the change of land uses is one of the anthropogenic activities that most affects the greenhouse emission, it is one of the components of the global carbon cycle that is more uncertain. In itself, the greenhouse gases emission in terms of carbon depends on many factors including among others, the type of soil cover, the area climatic conditions and the type of land use change to which they will be subjected. In Latin America, the greatest transformations of land uses are from forests to pastures. A total of 1,784 Mt CO₂ year⁻¹ of deforestation emissions took place during the period from

2000 to 2005 in Latin America, being Brazil the largest carbon emitter (Yepes et al. 2011).

The carbon stock in forests, agroforestry systems, and existing agricultural lands can be sustainably managed to enhance the amount of carbon fixed by the biomass and the soil. This might be a viable option to reduce the emission of C in an inexpensive, but effective way (Kanninen 2012). A management in this line is the increase of tree cover in the livestock landscape. This way, carbon storage in both biomass and soil is enhanced compared with pastures without trees (Zomer et al. 2016). Silvopastoral systems serve not only to trap greenhouse gases, but also to provide a local source of products (timber, firewood etc.), which reduces pressure on natural resources of forest and fossil fuels, making possible to retain the carbon that is already being stored in natural ecosystems. In the same way they enhance the water and biodiversity conservation and provide environmental services at regional and global scale (Ibrahim et al. 2003; Beer et al. 2003; Rojas et al. 2009).

Land use systems as SPS have been implemented for centuries in tropical and temperate areas. Some countries in central Europe like Italy (Sicily), Portugal (Montados), Greece (Crete), France, Denmark (Taatsrup) and Netherland, still maintain these systems in large areas of fruit trees plantations like almond, fig, olives, beech and oak woodland, which can be cut or grazed by the cattle and sheep. In East Germany, there are also remaining sites, which maintain the silvopastoral system, although they are declining (Eichhorn et al. 2006; Rigueiro-Rodríguez et al. 2008). On the other hand, China has a long history in SPS combining forestry with livestock and crops (Wu and Zu 1997). In North America, New Zealand and Australia there are SPS combining trees with forage or hay and livestock, and also improved silvopastoral systems as alley cropping (Nair and Nair 2003; Bandolin and Fisher 1991).

The above and belowground carbon contribution in SPS have been studied in different areas. In general, the average carbon storage by agroforestry practices (included the SPS) has been estimated from 9 to 63 Mg C ha⁻¹ for different climate zone (semiarid, sub humid, humid, and temperate regions (Montagnini and Nair 2004). In Florida a study compared the soil organic carbon (SOC) in SPS

and opened pasture (traditional livestock pasture systems). They found more carbon stored particularly at lower depths and in more stable fractions in SPS (Solomon et al. 2009). In the same way, in temperate areas of North America, Dixon (1994) found a carbon storage from 15 to 198 Mg ha⁻¹. Likewise, Howlett et al. (2011) reported values of SOC from 80.9 to 176.9 Mg ha⁻¹ for experimental silvopastoral plots in Galicia (Spain). In Latin America, the study of SPS has been intensified, attempting to determine the contribution made by each of their components to the global system, whether to carbon sequestration, water quality improvement or soil structure, and the economic contribution to the farmers, among other benefits. A SPS in southern Chilean Patagonia, stored between 148 and 252 Mg C ha⁻¹, distributed approximately 85% in soil, 12% in trees and 3% in pasture (Gordon et al. 2018).

In Central America silvopastoral systems have been promoted as a win win technology, which improve among others the productivity and provide environmental services (Gobbi 2001). In Costa Rica Andrade et al. (2008) reported values between 3.5 and 12.5 Mg ha⁻¹ for tree less and silvopastoral systems respectively. In the same way Hoosbek et al. (2018), found in southwestern Nicaragua, that the presence of tree in pasture enhances the topsoil C, N and P content and in subsoil the C content was also bigger, compared to tree less pasture. Furthermore, in Nicaragua, past measurements of carbon reservoirs in tropical sub humid regions showed that the total amount of carbon stored in a pasture with a high density of trees (132 t C ha⁻¹) was more than four times greater than that found at degraded pastures (29.5 t C ha⁻¹) (Ibrahim et al. 2007). The arboreal component of pastures in Matiguás municipality, on average, stored more carbon than degraded pasture due to the high tree density (Ruiz et al. 2004).

The amount of C fixed in SPS is particularly affected by tree species, density and their spatial distribution as well as the herbaceous species presents (Nyberg and Hogberg 1995; Jackson and Ash 1998). The total in carbon stored in SPS ranges from 68 to 204 t ha⁻¹, the majority of which is stored in the soil, while the annual rate of increase varies between 1.8 to 5.2 t ha⁻¹ (Pandey 2002).

A reliable method of gauging whether or not carbon has actually been

sequestered is critical to the development of a carbon trade mechanism. The field measurement of C stock change is capital intensive, using the technologies that are currently available (Bricklemeyer et al. 2007). Therefore, an estimation of C stock change that is based on agricultural soil practices and land management is currently preferred.

1.6. Land use/land cover mapping

Land use and land cover (LULC) change is a general term for the human modification of Earth's terrestrial surface of natural environment into semi-natural habitats such as arable fields, pastures, and managed woods (Ellis 2013). Spatial analysis of LULC is an essential method for the understanding and modeling of the earth as a system. The main data source for LULC analysis is remotely sensed imagery; done on orthophotos (georeferenced aerial imagery), on multispectral imagery derived from active satellite mounted sensors as well as from SAR (synthetic aperture radar) imagery. The most traditional kind of data source used is multispectral imagery, scanned by sensors as e.g. mounted on the Landsat, Sentinel-2 or Rapid Eye satellite platforms. In a traditional time consuming way, LULC mapping can be done manually, by digitizing on remotely sensed imagery realized in a GIS (Geographic Information System), carried out by an experienced analyst; this technique is time consuming. Therefore, several semi-automated classification techniques (e.g. supervised classification) have been developed (Keranen and Kolvoord 2008).

Classifying land cover types and analyzing land use changes using remote sensing offers a rapid and effective tool for mapping surface vegetation change in time and space through satellite imagery classification Landsat. It is also being used to furnish statistical and spatial data with regard to the state of deforestation for a given region. The use of operational remote sensing sensor systems provides an excellent basis for the analysis of a great variety of environmental issues, including the classification of land use/ land-cover vegetation on both a regional and global scale (Hansen 2000). Currently, the use of remote sensing is essential in the generation of detailed information regarding

natural resources, especially related to the monitoring of land use/land-cover and for the analysis of landscape change over time.

1.7. Justification of research

The livestock activity in Nicaragua is a very important economic sector, representing approximately 7% of the national gross domestic product, with 5 million head of cattle (Thecnoserve 2016, Lie and Rich 2016). Unfortunately, the development of farming activity in Nicaragua generates harmful environmental effects such as pasture degradation, soil erosion, water contamination (through the use of fertilizers and pesticides), landscape fragmentation, loss of connectivity between forest patches and deforestation, inter alia (Moss 2008).

For these reasons, the Global Environment Facility and World Bank (GEF- WB) launched the Silvopastoral Project, which identified the destruction of forests as an issue of major importance for the country, giving top-priority to the improvement of natural resource management. The project was designed to assist three countries (Nicaragua, Costa Rica and Colombia) to develop technological choices in controlling livestock induced deforestation, to address socio economic issues linked to livestock grazing and to identify means to overcome barriers (e g: financial, knowledge, policy) to the adoption of silvopastoral systems. In many ways, the Silvopastoral Project could be considered as a research project which provided incentives to farmers to adopt silvopastoral practices that generate environmental services. The project was also designed to assess whether or not such rewards for environmental services could be effective in altering local behavior patterns. Under the Silvopastoral Project, the Regional Integrated Silvopastoral Approaches to Ecosystem Management Project (RISEMP) was a research project carried out in the study area for a period of 6 years-period (2002 – 2008), which generated field information about C stocks in a variety of silvopastoral land-uses.

Despite the great information generated by this project, it is still needed to investigate the transformations occurred in the livestock landscape in Nicaragua

via LULC changes affecting TSPS and the impact of these changes on the total amount of C stored. This way we can determine if total C is being released to the atmosphere and therefore, contributing to greenhouse gas emissions, or instead, carbon is being stored, contributing to mitigate global warming.

For that purpose, the present study aims to demonstrate the feasibility of detecting LULC changes in highly fragmented livestock landscapes, with a special focus on TSPS via remote sensing techniques by using free and open source software (FOSS) tools. The implementation of a FOSS based image processing chain for the analysis of Land use changes affecting TSPS enhances the reproducibility of the study and can serve as a conceptual model for researchers and other target groups to realize further research.

The target group for the LULC analysis of TSPS mainly comprises the authorities and researchers in developing countries in the tropical region where funding is often low. Moreover, the acquisition costs for highly specialized image processing software and high spatial resolution imagery can only be raised within limited project timeframes. This frequently hampers the continuity of the research once the project is finished.

Once identified the LULC affecting TSPS in a long enough time period, the link with the C storage of the main TSPS allows an estimation of changes in the total C storage that have taken place in the territory. This research, therefore, contributes to the information required for decision-making regarding the most adequate land use change strategies for improving climate change mitigation and the recovery of degraded lands. For this study we rely on previous results obtained in the project RISEM (Regional Integrated Silvopastoral Approaches to Ecosystem Management (<http://bit.ly/2AL6c6h>)).

1.8. Objectives

The general objective of this study was to analyze the land use changes occurred within the livestock landscape of Nicaragua in the past, assessing the impact on the total carbon storage.

To accomplish this goal, specific objectives were proposed:

- I. To determine the land-use changes associated to TSPS occurred in the livestock landscape of a representative area in Nicaragua for a period of 29 years.
- II. To determine the ecological structure as well as the total carbon stored in the main TSPS land-uses present in the study area, assessing the contribution of biomass and soil to the total carbon storage.
- III. To assess the effect of the TSPS land-use changes previously identified on the total amount of C stored in the territory.

This research, therefore, contributes to the information required for decision-making regarding the most adequate land use change strategies for improving climate change mitigation and the recovery of degraded lands

MATERIALS AND METHODS

2. Materials and methods

The research work is divided into three parts. The first part is a retrospective land use change analysis of the last 29 years (1986-2015) in the study area. This analysis is based on the use of free open source software and data as QGis, R and satellite images Landsat TM 5. The second part is made up of the estimation of above and belowground carbon and the ecological structure of the TSPS land uses studied at the preceding study. For this analysis, data previously acquired in the RISEM project, in which the doctoral student was a scientific researcher, was used. The third and final part merge the two previous analyzes in such a way that it illustrates the multi-temporal carbon content at the landscape level, showing the effect on total carbon resulting from the change in land uses.

2.1. Study area

The research took place in Nicaragua, in the municipality of Matiguás (12°50'N / 85°27'O), longitude East and altitude between 200 and 500 meter above sea level (Fig. 1). Matiguás municipality covers 1710 km² and is located in the department of Matagalpa in North Central Nicaragua, one of the largest cattle raising regions of the country. The prevailing climate is sub-humid tropical with mean annual temperatures between 28°C and 32°C and an annual rainfall from 1200 to 2000 mm (Levard et al. 2001; Yamamoto et al. 2007). According the Köppen-Geiger climate classification (Kottek et al. 2006), this climate is classified as "tropical wet and dry or savannah climate" (Aw). The annual rainy season in Matiguás lasts from May until December, with a dry period from January until April (Fig. 2). Matiguás is located in a transition area between the dry and the moist tropical forest (Harvey et al. 2003, Meyrat 2000) and belongs to the Holdridge's Tropical Moist Forest life zone (Holdridge 2000). Vertisols, which are

soils rich in expansive clays, are the predominant soils in the area.

Figure 1. (a) Location of the study area Matiguás within Central America (base map data source: Natural Earth, 2018); (b) Topographical map of Matiguás (digital elevation model source: Aster Global Digital Elevation Map from the USGS EarthExplorer (2018)).

The soil use is characterized by the predominance of natural and improved pastures, with trees. There are also patches of primary, secondary and riparian forests (Ruiz 2004). The landscape is highly fragmented (Sánchez et al. 2005). The main reason for choosing the study site is that Matiguás is a typical example of a landscape that lies within Nicaragua's old agricultural frontier (Maldidier and Marchetti 1996) with cattle ranching (milk and meat production) as the main economic activity (Medina et al. 2007). Its represent the main source of employment and resources for its householders.

Figure 2. Monthly precipitation(mm) and temperature (° C and °F) graphs of Matiguás, Nicaragua. Source: climate-data.org.

They also develop agricultural activities, such as the production of corn, beans, and sweet potatoes in small scale, that is intended for own consumption (Gobbi 2001). Another land use is occupied by two natural reserves, with the Sierra Quirragua Natural Reserve being the most important of them with an area of more than 10,000 ha.

Matiguás has been one of the intervention areas of the previously mentioned GEF-Silvopastoral Project. Under this project and during the first years of this century, the establishment of SPS was promoted through a payment system for environmental services. With this, it was intended to generate global environmental services and improve the socioeconomic conditions of farmers in the area.

2.2. Land use / land cover mapping

The focus of the present study is the change of land uses caused by livestock activity and TSPS promotion. For that, five TSPS classes were selected, taking their significance in land degradation trends and their representativeness in the area (Pagiola 2007) into account. For the class selection the previous work realized within the framework of the RISEM project was considered. Supplementary data supported the class selection, such as geographical data of the research area (e.g. orthophotos, satellite imagery, soil and vegetation maps). For the selection of suitable geographical data, the recommendations of Andrade and Ibrahim (2003) were followed. The five TSPS land uses selected are: Shrubland (SL), Intervened secondary forest (ISF), High tree density pasture (PHD), Low tree density pasture (PLD) and Degraded pasture (DP). To this five land uses classes native forest was incorporated (F), which doesn't have a livestock purposes, with the aim of complementing the analysis of land use changes in the study area and assess the progress of the deforestation (Table 1 and Fig. 3).

Table 1. TSPS LULC classification scheme

LULC class	Symbol	Description
Forest	F	Native Forest without intervention in the last 30 years, more than 80% of tree cover.
Shrubland	SL	Native vegetation in natural succession with less than 5 m height.
Intervened secondary forest	ISF	Native disturbed forest (partial logging or emergent tree removal, non-timber forest, harvesting and/or hunting).
High tree density pasture	PHD	Pasture dominated by introduced or improved grass species with high vigor and productivity, the existing trees are mature, with a tree density of more than 30 trees per hectare.
Low tree density pasture	PLD	Area dominated by natural or naturalized grass species, trees with more than 5 cm of DBH and 2 m height.
Degraded pasture	DP	Pasture with less than 50% pasture and/or forage coverage, minimum presence of trees and shrubs; with obvious signs of erosion.
Water body	WB	River, lake, water reservoirs.
Rural infrastructure	RI	Rural roads, rural housing, farmyards.
Error class	EC	
Shadow	S	
Cloud	C	

DBH: diameter at breast height.

Figure 3. Photos of the different land use/land cover classes studied.

2.2.1. Satellite data acquisition

The study follows the approach of using open access data and multi-temporal multispectral sensor data from the Landsat program (Anjos 2016). Landsat imagery constitutes one of most important data source for remote sensing due to its free availability. Landsat has a long tradition, being the first operational passive optical earth observing satellite system (Fig. 4). Since the 1970's Landsat imagery has been used for projects of various disciplines: Agriculture, Forestry, Geology Geography and Cartography.

Figure 4. Graphical illustration of Landsat program from 1972, present and future launch.

Source: <https://landsat.usgs.gov/landsat-missions-timelinelandsat.usgs.gov>

Land cover monitoring through image analysis must consider the variations in vegetation reflectance due to phenologic changes between dry and rainy seasons. Hence, finding images around the same date of a year is a crucial precondition of remote sensing analysis. For the present study, an in depth search for cloud free Landsat imagery in the archives of the United States Geological Survey (USGS) Earth explorer (USGS 2018) from 1972 to 2015 resulted in two almost cloud cover free images (less than 5%) from the dry season (from December to March). The Landsat scenes found are a Landsat 5 Thematic Mapper (TM) scene from 28 January 1986 and a Landsat 8 Operational land imager (OLI) scene from 28 January 2015 (scene name p016r051_5dx19860128 and LC80160512015028LGN00 respectively; both with the same row and path number: WRS_PATH = 016, WRS_ROW = 051).

2.2.2. Proposal for a FOSS process chain for TSPS LULC analysis

The present study develops the FOSS process scheme stated in Fig. 5, whose denomination adduces to a licensing practice for software, in this case QGIS, as an alternative to ERDAS, ENVI, ARGIS, IMAGINE, among others. This proposed classification scheme is based on the four steps developed by Jensen (2009): (i)

the first trace is the remote sensing data requirements, such as, *in situ* measurement, ancillary data, collect information; (ii) remote sensing data acquisition which is overriding the spatial, spectral, temporal, radiometric and thematic resolution; (iii) data analysis or data to information conversion, attending as the analog and digital image processing and , finally, (iv) the information presentation, hither the information is summarized, in form of an improve image, map, spatial data base, statistics or graphs. This method was applied in this study only with Landsat images, but can be used as a conceptual basis for the inclusion of other types of cloud free imagery, if available (e.g. Sentinel-2). The process scheme uses the semiautomatic classification plugin of QGIS, which has the goal to provide a FOSS alternative for land cover monitoring using remote sensing image analysis (Congedo 2014). At the same time, we have relied on the use of Gnu-R which is a free software mass-collaboration project, applied by R, a programming language and free software, in which several R packages have been used (e.g. Raster, Random Forest, RStoolbox, sp, asbio). This sequence of process components is proposed due to the high grade of interoperability between the software components (Fig. 5).

A substantial part of the research was the conceptual development of a reusable TSPS LULC classification scheme prior to any supervised image processing. The classes selected and the knowledge gained in the RISEM project was included in the development. To get an overview over the spectral characteristics of the images an unsupervised classification (K-Means clustering) was performed by using five iterations with 100 statistically built classes. The class number was lowered by 10 in each iteration, permitting to alter the total number of clusters by splitting, merging, or deleting clusters. As previously indicated in Table 1, the final TSPS LULC classification scheme contains five TSPS relevant land use classes plus native forest, and additionally, waterbodies (WB), rural infrastructure (RI), shadow and cloud error classes were included.

Figure 5. Methodological flowchart of the FOSS based data to information conversion of Jensen's (2009) representation of the TSPS-LULC analysis process.

2.2.3. Image pre-processing

The image preprocessing consists of several parts, the radiometric, geometric correction and the atmospheric correction of the satellite images. As radiometric correction, a histogram stretch was applied in QGIS, with the goal to enhance the visualization of the images. As geometric correction the pixel of the Landsat 5 TM image from 1986 have been co-registered to the Landsat 8 OLI image. The QGIS semiautomatic classification plugin (SCP) (Congedo 2016) plugin was used to perform an atmospheric correction by applying the method of Dark Object Subtraction 1 (DOS 1) developed by Chavez (1996). DOS 1 searches the darkest pixel value in each spectral band, then the scattering is removed by subtracting this value from all other pixel values (Chavez 1996). Further, the semiautomatic classification plugin was applied to convert the reflectance values from relative values (DN's, digital numbers) to absolute Top of Atmosphere reflectance values (TAO).

Several methods have been used to get to a final decision about the thematic classes. Separability was tested with the SCP, the QGIS plugin ROI-explorer (Bullock and Holden 2015), a plugin that allows to visualize the standard deviations and means of spectral signatures within the training polygons and mainly by using different Gnu-R libraries. Subsequently the realized preprocessing steps and tools to check the spectral separability will be explained: First an unsupervised classification was performed, applying k-means clustering. K-means was applied by using the unsupervised classification function of the R library RSToolbox (Leutner et al. 2018). The algorithm statistically locates the cluster centers in the multidimensional (multispectral) feature space of the input data; depending on the distance vector of each pixel to the class centers, each pixel in the imagery receives an assignation to a class. The assignation is repeated in an iterative way, till no more changes in the location of the class mean vectors appears (Lillesand et al. 2014). The K-Means cluster was performed five times with an initial class number of 100 classes, which was lowered by 10 in each run, permitting to alter the total number of clusters by splitting, merging or deleting clusters.

After the K-means clustering 10 possible thematic classes have been identified. For those classes 500 training areas have been digitized and used for further tests of the spectral separability. Spectral scatter plots of the pixels within the train areas have been scatter plotted by using the R library ggplot (Wickham et al. 2016) and the Jeffries-Matusita and M-statistics spectral separability metric has been calculated to test the spectral separability. The Jeffrey-Matusita (J-M) distance pairwise calculates the spectral distance between the thematic classes, perfectly separable classes have a distance value of 2, identical classes of 0 (Matusita 1966). M-Statistic (M) measures peaks in the distribution of spectral values, large values mean a good separation between spectral classes (Kaufman and Remer 1994). The spectral metrics were calculated by applying the separability functions in R's spatialEco package (Evans 2018).

2.2.4. Image processing

For many years, image processing and analysis techniques have been developed to improve the interpretation and to obtain as much information as possible from the images. The target of each research will determine the algorithms to apply (<https://crisp.nus.edu.sg>). For this study a Random Forest (RF) algorithm was chosen to classify the LULC features of the study area. Random Forest is a supervised tree based ensemble learning method that comprises several non-influencing classification and regression trees (CART); it uses a random set of features for node splitting (Breimann 2001) (Fig. 5). The RF algorithm combines different basic classifiers to generate divergent possibilities of data classifications (Rodriguez-Galiano et al. 2012). Random vectors are created, each of them grown as trees, based on a randomly selected subset of the data (in our case, the spectral information) (Lawrence et al. 2006). This increases the diversity of the trees as each tree grows from a different randomly selected training dataset (Guo et al. 2011), which reduces the generalization error (Rodriguez-Galiano et al. 2012). For the final classification result, each of the trees gives one single voice on the possible class and thus the results of the

classifications are collected, with the final class being derived in a rule based manner (Pal 2005; Rodriguez–Galiano et al. 2012). One of the biggest advantages of RF is the low number of parameters that is needed to get a good performance (Feng et al. 2015). Compared with neural networks or support vector machines, RF is easier to implement but still very effective and comprehensive as all trees can be revised, while the internal decisions of a neural network are often not comprehensible (Gieslason et al. 2006; Feng et al. 2015) (Fig 6). The Random Forest classification was carried out by using R' Random Forest library by Liaw and Wiener (2002)

Figure 6. Random forest decision trees.

2.2.5. Supervised classification

A set of 500 homogeneously distributed training areas was digitized in each LULC class and used to train the model. An R script was written that made use of the R Random Forest package based on Breiman's and Cutler's original FORTRAN

script (Liaw and Wiener 2001). We have published our script on Github (Cárdenas and Schernthanner 2017). We followed Breiman's (2001) recommendations and put aside data to calculate the out-of-the bag errors (OOB) error. The OOB error is a performance measure for assessing the accuracy of RF models. By fine-tuning the model, the OOB model error could be lowered below 13%; which refers to the search of the best-fitting model parameters by applying a stepwise increase in the number of RF trees and the number of variables splitting at each tree node.

To determine the number of trees, the mean error for 1,500 model runs, with an increasing number of trees, was performed for each LULC class of the two image datasets. Above 500 trees and three variables at each tree node, there was no more decreases in the mean error of the RF models was observed and the classification error tended remain constant. For this reason, 500 RF trees and three variables at each tree node were chosen as optimal parameters for the operational stage of the classification. Twenty-five ha were selected as the minimum mapping unit (MMU) based on the MMU used in the Corine Land Cover Program (Copernicus Programme 2016). These units, which are training polygons varying in size and form, have been digitized according to underlying homogenous areas, representing the different land use classes. For digitizing the training areas expert knowledge of image interpretation and extensive knowledge of the research area is indispensable. This expert knowledge was acquired in several field studies (weekly field visits in a period of 5 years from 2002 - 2007).

The relationship between the image scale and the resulting map scale was determined by applying the simple Equation (1) (Tobler 1987):

$$MP = \frac{1}{2(GR)1000} \quad \text{Equation (1)}$$

Where MP is the map scale and GR is the geometrical resolution measured in m.

According to Tobler's formula, the thematic classification results of the study allow a medium map scale of 1: 60,000.

Following the classification, a change detection was performed on the thematic classification results by calculating the pixel by pixel difference between the resultant thematic layers. Shadow and cloud pixels were removed from the calculation of the area by being recoded to pixel with no data values. For this, the raster map algebra in QGIS was applied. The state of LULC transition was determined pixel by pixel. For each pixel, the change from the 1986 LULC class to the 2015 LULC class was calculated.

2.2.6. Post-classification and accuracy assessment

Prior to the change detection analysis, an accuracy assessment of the classified thematic layers was performed. To minimize the salt and pepper effects of a pixel based classification and produce more homogeneous land cover areas, a 3 x 3-pixel window majority filter was applied. The model validation and the assessment of classification accuracy consisted of two components: the RF model's internal error measurements and an accuracy assessment by randomly distributed reference data. As the measurements of the RF model's internal error, OOB, and the decreasing RF mean error rate were calculated. A pragmatic validation strategy based on the principles of expert knowledge of image interpretation had to be applied due to the absence of reference data. For the interpretation of the 2015 satellite imagery, satellite base maps from Bing maps (Bing map aerial) and the panchromatic band 8 of the Landsat 8 OLI image was used. For the interpretation of the 1986 satellite imagery, two ancillary panchromatic Satellite Pour l'Observation de la Terre (SPOT) Controlled Image Base 10 m, (CIB-10) images scene identifications (IDs): SPOTCIBN13W086, SPOTCIBN14W086) were used.

The validation of the classification results was based on the comparison with an interpretation of LULC classes at the stratified randomly distributed reference

points on the original Landsat images. According to the number of sampling points, Congalton and Green (2008) have stated the requirement for a minimum of 50 sample points per ground cover class. López et al. (2006) have stated a requirement for 100 sample points for 95% confidence level. Conforming López's and Congalton and Green's recommendations, we used 50 to 100 randomly distributed reference points per stratum for the TSPS LULC classes that are relevant for the analysis. In the case of error classes and non TSPS LULC classes 30 to 50 stratified reference points were used, depending on the presence in the scene. In case of overlapping with the training samples, the reference samples have been excluded from the validation.

For each class the user and producer accuracy was calculated; moreover, the Cohen's kappa coefficient (κ) of the overall classifications accuracy was calculated (Thomlinson et al. 1999; Zharikov et al. 2005). The kappa coefficient is a statistic which measures the agreement between two individuals for qualitative items. It is generally thought to be a more robust measure than only percentage because it takes into account that agreement may be due to chance. Misclassifications between the assigned class and the real class (ground truth) always occur because of the continuity of vegetation, the spectral overlap of different types of coverage, and the particular resolution chosen for the map (Steele et al. 1998).

2.3. Carbon storage analysis

2.3.1. Experimental design for carbon and ecological analysis

A forest inventory was conducted within the RISEM project in each TSPS land use in order to estimate the carbon storage associated with the above and belowground biomass as well as with soil in the livestock landscape. The doctoral

student was in charge of data for Nicaragua. Ecological characterization was also carried out looking for relationships between diversity and C storage. Considering that native forest is not directly related with the livestock activity and that it is concentrated in a natural reserve, without presence on the open landscape, it was not included on the analysis.

Four replicates of each one of the five TSPS land uses were randomly selected comprising twenty sampling areas, which were located in small to large size livestock farms from Matiguás municipality. In each of these areas, three sampling points, separated 50 m from each other along the slope were selected. Sampling strategies for biomass or soil carbon were centered around these sampling points (Fig. 7).

Plots for estimation of biomass and ecological structure in pastureland were selected following a methodology suggested by MacDicken (1997). Circular plots of 1000 m² centered in the previous selected points were established, resulting in a total of 3000 m² sampled in each replicate. With respect to plots for ISF and SL, an adaptation of the methodology proposed by Segura and Kanninen (2002) was selected. These plots consisted of rectangular areas of 25 m x 10 m centered on the previous chosen points, with the longest length along the slope (Fig. 7). In these plots, the species of every tree with a DBH (diameter at breast height) greater than 5 cm, saplings (<1.5 m high and DBH < 4.9 cm) and seedlings (0.3 to 1.5 m high) were identified for the forest inventory.

With respect to sampling for soil organic carbon analyses, there was one principal trial pit (1 m x 1 m x 1 m) located in each point and two small trial pits (0.4 m x 0.4 m x 0.4 m) on either side of the principal pit. Each principal pit was spaced 50 m from its nearest neighbor, while the small pits were spaced 15 m from the corresponding principal pit (Fig. 7).

Figure 7. Experimental design in each one of the 20 sampling areas. In the center, three sampling pits, one in each of the selected points following the slope, with two small pits on each side for soil organic carbon sampling. On the right, circular plots in each pastureland replicate, centered on each main pit for ecological characterization of pastureland. On the left rectangular plots in each shrub or forest replicates for ecological characterization.

2.3.2. Tree biomass carbon

Based in the inventory previously carried out, the aboveground tree biomass of each land use was estimated in the present study using the allometric models described in Table 2. With respect to ISF, a model developed by Brown (1997) for tropical moist regions with trees having a DBH between 5 and 148 cm was applied. For pasture with trees (DP, PLD, PHD) and SL, a model developed in Matiguás (Ruiz, 2004) was selected. Due to the fact that the analysis of tree roots, which constitute the belowground biomass (BGB), is a destructive and extremely expensive method, a model proposed by the IPCC was applied to estimate fine roots (Buendia et al. 2006) and a value of 30% of the aboveground biomass (AGB) was used for coarse roots (Brown 2002). To calculate the carbon, AGB and BGB were multiplied by 0.42, following Amézquita (2004).

Table 2 Allometric equations used to estimate the tree aboveground and fine roots biomass on TSPS

Land uses	Allometric Equation	R ²
ISF	$B = \exp(-2.134 + 2.53 \ln(DBH)^2)$	0.97
DP, PLD, PHD, SL	$\log B = -2.18 + 0.08 \cdot DBH - 0.0006 \cdot (DBH)^2$	0.94
Fine Roots	$Y = \exp[-1.0587 + 0.8836 \ln(AGB)]$	0.84

B= Biomass; ISF= Intervened secondary forests; DP= Degraded pasture; PLD= Pasture with low tree density; PHD= Pasture with high tree density; SL=Shrubland; AGB=Aboveground biomass; Y= Fine roots biomass.

2.3.3. Soil organic carbon

For the estimation of SOC, samples were taken from each of the three principal trial pits at four different depths: 0-10 cm, 10-20 cm, 20-40 cm and 40-100 cm, giving 12 samples for each sampling plot. Subsequently the corresponding samples from the same depth of the three pits of in each plot were combined and homogenized to give a composite sample. That gave a total of 4 samples

per plot; 16 samples per land use and 80 samples altogether. SOC concentration was determined in the laboratory by the Walkley and Black Method. For each layer, soil bulk density was measured from three undisturbed soil core samples, using steel rings of 5 cm diameter by 5 cm height. The total soil organic carbon (TSOC) stored in a depth of 1 m was calculated as the sum of SOC in each layer weighted by thickness.

The carbon contents of each component (above and belowground biomass and TSOC) were summed to give the total carbon storage for the entire system.

2.3.4. Ecological structure analysis

To estimate the vegetation and composition dynamic, the basal area (BA) was estimated using the following equation:

$$BA = 0,00007854 \times DBH$$

Derived from the following equation proposed by Amateis and Burkhardt (1987):

$$BA = \pi (DBH/2)^2 = 0.00007854 \times (DBH)^2$$

where:

BA= Basal area (m²)

π = 3.1416

DBH= diameter at breast height

Additionally, for each tree the height was estimated using an altimeter.

The tree density was calculated as the number of individuals per ha. Rarefaction curves have been implemented to estimate the richness distribution for the land uses studied (Krebs 1989). Furthermore, the Margalef (1958) index was calculated to assess the diversity of species for the different land uses:

$$R = (S - 1) / \ln N$$

where S is the number of taxa, and N is the number of individuals in the sample. The Margalef index refers to values below 2.0 as low-diversity areas and values above 5.0 as indicative of high biodiversity (Margalef 1958).

In addition, Fisher's alpha diversity index was chosen to estimate diversity because of its ability to discriminate information, its relative sample size sensitivity, and robustness (Magurran 1981). Fisher's alpha diversity index (Fisher et al. 1943) was calculated as:

$$S = \alpha \ln \left(1 + \frac{N}{\alpha} \right)$$

where S is the number of species, N is the number of individuals and α is Fisher's alpha diversity index.

2.3.5. Statistical analysis

For the statistical analysis of the ecological structure, aboveground and belowground biomass and soil carbon storage on the five TSPS land uses, nonparametric statistics (Kruskal Wallis test) were carried out to test whether the observed differences were significant ($P < 0.05$). All analyses were performed with the statistical software package InfoStat. For the estimation of richness and the diversity indices (Margalef and Fisher alpha), the software package PAleontological STatistics (PAST) was used. The calculations were carried out using the R Package Vegan (Oksanen et al. 2014).

2.4. TSPS carbon maps from LULC maps

For the carbon maps elaboration, the carbon value formerly estimated for each land use, was extrapolated to the land use map previously generated, getting then the carbon distribution map at landscape level for 1986 and 2015. To obtain the carbon stock lost or accumulated in the period of almost 30 years elapsed from 1986 to 2015, the two carbon content maps (1986 and 2015) were compared in a carbon matrix table. This way, we can assess the effect of the

TSPS land-use changes occurred in the period 1986-2015 on the amount of C stored in the territory.

RESULTS

3. Results

3.1. Land use change analysis

First question arising from this study is if the methodology developed to detect LULC changes in this highly fragmented landscape via remote sensing techniques by using FOSS tools is adequate. To do that, general accuracy and Cohen's Kappa coefficient, which tell us what percentage of the image in analysis has been mapped correctly, were run on the classification of remotely sensed imagery. For 2015 image classification, an overall accuracy of 85.9% and a Cohen's Kappa coefficient of 84.1 were obtained. For the image of 1986, an overall accuracy of 85.2% and a Kappa value of 83.4 were obtained. The result was close to what was recommended in the literature (Congalton and Green, 2008). Therefore, the derived distributions of land cover classes were acceptable, since thematic maps derived from remote sensing are never completely accurate (Shao and Wu 2008). The classifications showed a good OOB error (below 13%) for both scenes. The thematic datasets derived from the satellite images had an overall accuracy within an acceptable error range. Producer's and user's accuracy in the error matrix, exposed in the Table 3 for 2015 and 4 for 1986, show the ability of random forest to classify land uses in a livestock landscape matrix, being F and SL the land uses classes that performed better, followed by PHD

Table 3. Confusion Matrix and Statistics Landsat 2015.

LULC CLASS	F	SL	ISF	PHD	PLD	DP	WB	RI	C	S	TOTAL	USERS ACCURACY ↓ (%)
F	106	4	3	0	2	0	0	0	0	0	115	92.2
SL	0	93	2	0	2	0	0	0	0	0	97	93.0
ISF	0	9	78	7	2	4	0	0	0	0	100	78.0
PHD	0	2	5	91	0	2	0	0	0	0	100	91.0
PLD	0	2	1	1	83	13	0	0	0	0	100	83.0
DP	0	0	1	1	14		0	1	0	0	99	82.8
WB	0	1	0	2	1	0	29	1	0	4	38	76.3
RI	0	4	0	0	5	4	0	33	0	0	46	71.7
C	0	1	2	0	0	0	0	0	68	2	73	93.2
S	4	0	1	0	0	0	3	0	0	28	36	77.8
TOTAL	110	116	93	102	109	105	32	35	68	34	804	
PRODUCER S ACCURAC Y (%)	→ 96.4	80.2	83.9	89.2	76.1	78.1	90.6	94.3	100	82.4		

F = Native forest; SL = Shrubland; ISF = Intervened secondary forests; PHD = Pasture with high tree density; PLD = Pasture with low tree density; DP = Degraded pastures; WB = Water body; RI = Rural infrastructure; S = Shadow; C = Cloud.

Table 4. Confusion Matrix and Statistics Landsat 1986.

LULC CLASS	F	SL	ISF	PHD	PLD	DP	WB	RI	C	S	TOTAL	USERS ACCURACY (%) ↓
F	82	4	13	0	0	1	0	0	0	0	100	82.0
SL	0	72	18	5	3	2	0	0	0	0	100	72.0
ISF	0	0	96	3	0	0	0	1	0	0	100	96.0
PHD	0	0	4	84	12	0	0	0	0	0	100	84.0
PLD	0	0	0	0	83	5	0	0	1	0	89	93.3
DP	0	0	0	0	5	87	0	8	0	0	100	87.0
WB	1	0	1	0	0	0	35	0	0	3	40	87.5
RI	0	2	0	0	0	5	0	48	2	0	57	84.2
C	0	0	0	0	0	0	0	0	31	0	31	100
S	8	0	1	0	0	0	6	0	0	40	55	72.7
TOTAL	91	78	133	92	103	100	41	57	34	43	772	
PRODUCERS ACCURACY (%) →	90.1	92.3	72.2	91.3	80.6	87.0	85.4	84.2	91.2	93.0		

F = Native forest; SL = Shrubland; ISF = Intervened secondary forests; PHD = Pasture with high tree density; PLD = Pasture with low tree density; DP = Degraded pastures; WB = Water body; RI = Rural infrastructure; S = Shadow; C = Cloud

According to the Landsat image analysis, the study area is a matrix of land uses, typically disturbed by livestock and agricultural activities. The landscape is dominated mostly by paddocks, forest patches, trees outside the forest, and degraded areas. The landscape in the study area was very dynamic during the 1986–2015 period (Table 5). Over the 29-year period studied, the occurrence of F, SL, PHD, and DP decreased while that of ISF and PLD increased. In absolute terms, the most notable change was for ISF, which went from occupying 16.4% of the territory in 1986 to occupying 43.1% in 2015, with an increase of almost 46,000 ha. In contrast, PHD lost 15,000 ha and pasture almost 13,000 ha in the same period. Percentagewise, the largest increase was experienced by PLD, which went from occupying 2.4% of the territory to occupy 7.5%, as long as the largest percentage reduction occurred in SL which went from occupying 19.8% of the territory to 6.8% during the same period. While in 1986 the land use was relatively balanced with a value around 20% (except for F and PLD that were less), in 2015, ISF occupied more than 40% of the territory of Matiguás, figure that reaches almost 50% when considering the area occupied by the land uses under study (152,765 ha).

Table 5. Land use area distribution (ha) and occupancy (%) in 1986 and 2015, and changes between 1986 and 2015.

<i>LULC</i>	<i>1986</i>		<i>2015</i>		<i>Change (1986-2015)</i>	
	(ha)	(%)	(ha)	(%)	(ha)	(%)
<i>F</i>	13,248	7.7 (8.7)*	8,514	5.0 (5.6)	-4,734	-35.7
<i>SL</i>	33,792	19.8 (22.1)	11,592	6.8 (7.6)	-2,200	-65.7
<i>ISF</i>	28,046	16.4 (18.4)	73,781	43.1 (48.3)	+45,736	163.1
<i>PHD</i>	37,163	21.7 (24.3)	22,112	12.9 (14.5)	-15,051	-40.5
<i>PLD</i>	4,087	2.4 (2.7)	12,893	7.5 (8.4)	+8,806	215.5
<i>DP</i>	36,427	21.3 (23.8)	23,871	14.0 (15.6)	-12,556	-34.5
<i>Other</i>	18,235		18,235			
<i>Total</i>	171,000		171,000			

F = Native forest; SL = Shrubland; ISF = Intervened secondary forests; PHD = Pasture with high tree density; PLD = Pasture with low tree density; DP = Degraded pastures; * = Occupancy over the area covered by the classes analyzed: 152,765 ha.

The land cover changes from one class to another, as shown in Table 6, represent the gains and losses. It should be noted that bidirectional changes occur frequently in this area with a highly complex pattern of changes. In the case of non-intervened forest (F), among the 13,248 ha in 1986, 3,833 ha remained intact in 2015, while the rest have been transformed into other classes, mainly ISF (7,365 ha), and into other classes to a much lesser extent. Non-intervened forest also experienced gains. For example, 1,094 ha that were SL and 2,417 ha that were ISF in 1986 changed to F by 2015. It is very important to notice that the most unexpected change in land use was an area of 0.17% that belonged to DP and was transformed into forest without intervention. This phenomenon could be explained in the redefinition of the protected area "Serrania Quirragua," which was declared in November 1991. This entails a new distribution of land use, which means the liberation, restoration, and protection of areas pastured in the state of degradation that are currently located in the forest category. However, as mentioned above, there is still a net loss of F.

With regard to ISF, 16,427 of the 28,046 ha in 1986 were intact in 2015, while the remaining 11,619 ha was distributed evenly among other classes. As for gains in ISF, these have occurred mainly at the expense of SL (22,449 ha) and PHD (14,656 ha), as well as other classes, resulting in a net gain of 45,736 ha. The positive outcome that almost 12,000 hectares of DP in 1986 have become ISF needs to be highlighted.

Table 6. Land use/cover change matrix, 1986-2015 Matiguás, Nicaragua (in hectares).

<i>LULC</i>	<i>1986</i>		<i>2015</i>					
	Total	(%)	F	SL	ISF	PHD	PLD	DP
<i>F</i>	13,249	8.7	3,833*	554	7,365	799	278	422
<i>SL</i>	33,793	22.1	1,094	2,902*	22,449	3,232	1,551	2,563
<i>ISF</i>	28,046	18.4	2,417	1,785	16,427*	2,829	1,797	2,788
<i>PHD</i>	37,163	24.3	856	3,914	14,656	9,820*	3,495	4,420
<i>PLD</i>	4,088	2.7	45	345	968	1,442	574*	711
<i>DP</i>	36,428	23.8	270	2,090	11,914	3,988	5,197	12,966*
<i>Total</i>	152,766		8,514	11,592	73,781	22,112	12,893	23,871
<i>(%)</i>			5.6	7.6	48.3	14.5	8.4	15.6

F = Native forest; SL = Shrubland; ISF = Intervened secondary forests; PHD = Pasture with high tree density; PLD = Pasture with low tree density; DP = Degraded pastures; * = correspond to ha kept intact.

With regard to the spatial distribution of land use change (Fig. 9, 10, 11, 13), there is no clear pattern in the case of SL, ISF, PHD, and DP. In the case of F, however, there was an increase around the protected area of a natural reserve, while the losses were concentrated in fragmented areas with forest patches in 1986, mainly located in the southern half of the district (Fig 8)

Figure 8. Native forest cover maps generated for 1986 and 2015.

Figure 9. Shrubland cover maps generated for 1986 and 2015.

Figure 10. Intervened secondary forest cover maps generated for 1986 and 2015.

Figure 11. High tree density pasture cover maps generated for 1986 and 2015.

Figure 12. Low tree density pasture cover maps generated for 1986 and 2015.

Figure 13. Degraded pastures cover maps generated for 1986 and 2015.

3.2. Ecological structure of the five silvopastoral land uses

Of the last six land uses studied, a forest inventory was not available for the native forest, which is located in a close natural reserve since the beginning of the 90s, and has no livestock orientation. Among the five TSPS land uses evaluated, SL had the highest number of species (33). Significant differences for tree density and basal area between DP and three land uses (SL, ISF, and PHD) were found. PHD had the highest basal area followed by ISF and SL (Table 7), whereas DP had the lowest. To estimate species richness and diversity the Margalef index was used, with SL having significantly higher values than the other uses. Similarly, the values of Fisher's alpha index show that there is a significantly different diversity of species for SL compared with the other land uses (Table 7). Individual rarefaction curves were calculated to compare the species richness and support the result of the richness and diversity indices. Unlike the Margalef and Fisher indices, the rarefaction curves suggest that not only SL but also ISF differ from the other land uses in terms of richness (Fig. 14).

Table 7. Ecological structure of the five TSPS land uses land cover studied

LULC	# Sp	Tree density (n ha ⁻¹)	Basal Area (m ² ha ⁻¹)	Richness and diversity	
				Margalef	Fisher's alpha
SL	33	75.0 (51.77) b	2.05 (0.66) b	3,1 (0.12) b	5.0 (0.29) b
ISF	26	82.7 (30.45) b	2.20 (0.65) b	2.3 (0.24) a	3.0 (0.43) a
PHD	19	105.9 (53.22) b	2.35 (1.88) b	2.2 (0.13) a	3.3 (0.28) a
PLD	19	24.7 (26.81) ab	1.25 (0.34) ab	2.2 (0.09) a	3.3 (0.32) a
DP	20	24.3 (6.58) a	0.55 (0.16) a	2.1 (0.13) a	3.1 (0.22) a

Sp= species; SL=Shrubland; ISF= Intervened secondary forests; PHD= Pasture with high tree density; PLD= Pasture with low tree density; DP= Degraded pasture; Medians (standard error) within a column with different letters are significantly different (P<0.05).

Figure 14. Individual rarefaction curves of the five TSPS land uses studied: SL=Shrubland; ISF= Intervened secondary forests; PHD= Pasture with high tree density; PLD= Pasture with low tree density; DP= Degraded pasture

3.3. Total carbon storage in the silvopastoral land uses

Estimates of carbon storage in the aboveground tree biomass (Fig. 15) show that ISF (8.4 t C ha^{-1}) and PHD (6.3 t C ha^{-1}) accumulate the most C. On the other hand, DP (1.2 t C ha^{-1}) had the lowest storage of C. The land uses SL and PLD had values between these two extremes and there were no significant differences between them. The carbon content in fine and coarse roots followed the same pattern. For both above and belowground biomass carbon, significant differences were found between DP and ISF, whilst the other land uses had no statistical differences among them ($P > 0.05$). With regards to the total carbon stored in the biomass, DP had the lowest amount of total carbon (1.9 t C ha^{-1}), while ISF had the highest value (13.2 t C ha^{-1}), followed by PHD (9.8 t C ha^{-1}) (Fig. 15).

Soil organic carbon content per layer is shown in Fig. 15. In the 0-10 cm layer, SOC ranged from 23.2 to 35.2 t C ha^{-1} but no significant differences were found among

land uses. The C content varied significantly in the 10-20 cm layer, ranging from 12.6 to 29.1 t C ha⁻¹ for DP and SL, respectively. In the deeper soil layers (20-40 and 40-100 cm) ISF had the highest values (38.7 and 65.4 t C ha⁻¹, respectively). Degraded pasture had the lowest values in all four soil layers, but was only significantly different from ISF in the 10-20, 20-40 and 40-100 cm layers. The other land uses had no significant differences between them (Fig. 16).

Figure 15. Median carbon contents of the aboveground biomass (AGB) and belowground biomass (BGB) (coarse and fine roots) for five different land use classes: SL=Shrubland; ISF= Intervened secondary forests; PHD= Pasture with high tree density; PLD= Pasture with low tree density; DP= Degraded pasture. Error bars correspond to standard errors of the medians; different letters indicate statistical differences among LULC ($P < 0.05$).

With regards to the TSOC, significant differences were found among land uses. Intervened secondary forests had the largest value (163.6 t C ha⁻¹), followed by SL (122.2 t C ha⁻¹), both of which were significantly larger than DP (76.9 t C ha⁻¹), which had the lowest values. PLD (109.5 t C ha⁻¹) and PHD (108.1 t C ha⁻¹) had similar carbon contents and were not significantly different from the other land

uses.

The estimation of total carbon, ranged from 79.4 to 171 t C ha⁻¹ for DP and ISF, respectively. The SOC was the largest contributor to the carbon stock in all land uses under study ranging from 91 to 98% of the total C (Fig. 17).

Figure 16. Median soil organic carbon content for the five different land use classes: SL=Shrubland; ISF= Intervened secondary forests; PHD= Pasture with high tree density; PLD= Pasture with low tree density; DP= Degraded pasture, at four depths 0-10, 10-20, 20-40, 40-100 cm. Error bars show the standard errors; different letters indicate statistical differences among LULC ($P < 0.05$).

Figure 17. Total carbon storage in the biomass (TBC) and in the soil down to a depth of 100 cm (TSOC) for the different land uses. SL=Shrubland; ISF= Intervened secondary forests; PHD= pasture with high tree density; PLD= pasture with low tree density; DP= Degraded pasture. Different letters indicate significant differences ($P < 0.05$); error bars show the standard errors of the total carbon.

3.4. Effect of LULC changes between 1986 and 2015 on C storage

The study area has demonstrated to be a constantly changing area as seen from 1986-2015 LULC comparison (Table 6). Due to the different C amount stored by each TSPS use, the TSPS LULC changes occurred in Matiguás in this period affect importantly the C distribution in the livestock landscape as well as the net total C storage.

If we compare arboreal TBC between 1986 and 2015 (Table 8), the storage for the TSPS in 1986 was 1,016,156 t of carbon while for the year 2015 was estimated at 1,259,087 t C, resulting in an increase of 24%, or 242,930 t C in three decades at landscape level. This carbon includes the aboveground biomass carbon and roots carbon (coarse and fine roots carbon). Only 29 % (358,969 t) of the total C stored in 2015 comes from areas that did not experience change in use (diagonal figures). The zones of greater stability are normally under SF, ISF, (213,880 t C) and PHD (96,236 t C), while only a small part of the C retained in LDP in 2015 (2,732 t C) corresponds to C that was contributed by this same use in 1986 (Table 8).

Table 8. Total Biomass Carbon (TBC) flux in five TSPS land uses between 1986 and 2015 in a livestock landscape in Matiguás, Nicaragua.

LULC	1986	2015				
	Total	SL	ISF	PHD	PLD	DP
SL	237,707	21,098	292,286	31,674	7,383	4,947
ISF	333,651	12,977	213,880	27,724	8,554	5,381
PHD	355,789	28,455	190,821	96,236	16,636	8,531
PLD	19,230	2,508	12,603	14,132	2,732	1,372
DP	69,779	15,194	155,120	39,082	24,738	25,024
Total	1,016,156	80,232	864,710	208,848	60,043	45,254

SL=Shrubland; ISF= Intervened secondary forests; PHD= Pasture with high tree density; PLD= Pasture with low tree density; DP= Degraded pasture.

Regarding the contribution of each land use to the total aboveground biomass carbon stored in TSPS land uses, almost all uses except PLD had a relatively similar contribution in 1986 (Fig. 18). On the other hand, in 2015, the use with the highest contribution became the ISF that contributed 69% of the total aboveground biomass carbon, while in 1986 it only accounted for 32%. For its part, SL went from contributing with 23% in 1986 to be the smallest contributor in 2015 assuming only 6 % of the total.

Figure 18. Total Biomass carbon stored on each TSPS land use studied in the two study period.

The LULC changes also impacted the spatial distribution of the total biomass carbon as shown by the corresponding maps generated for 1986 and 2015 (Fig. 19). The distribution of C does not follow a clear pattern and responds to the migration of land uses experienced in that period.

Figure 19. Map of distribution of tree total biomass carbon content at landscape level for 1986 and 2015.

DISCUSSION

4. Discussion

4.1. Land use changes through remote sensing analysis

Remote sensing imagery offers the possibility to analyze land cover change over large areas without the need of any extensive costly fieldwork. Despite a high number of mixed pixels, the highly fragmented land use and land cover, and the medium geometric resolution of the imagery used in this study, random forest resulted in very promising classification accuracies. Therefore, the described methodology serves as a valuable basis for future LULC analysis for challenging datasets. The use of satellite imagery with higher geometric resolution might be desired if land cover change is to be evaluated in greater detail and/or for classifications with more LULC types. Moreover, cloud free imagery would help to refine and improve the conclusions regarding the LULC changes and, therefore, efforts should be made to acquire better imagery. The technical implementation of the methodology described in this study was strictly based on the use of FOSS software components and open data; therefore, its application to studies of other research groups is conceivable.

With regards to the differences in LULC between 1986 and 2015, the large extent of changes between the studied land use is very high, motivated mainly by anthropogenic drivers, in many cases in opposing directions such as the abandonment of some areas and the intensification of others. Contrary to

expectations, the results show a positive evolution in Matiguás over the past 29 years during which the overall forest cover has increased and DP have declined. Since our study covers 29 years without evaluating the intermediate years, we may not have detected rapid deforestation processes such as those found between 1983 and 1992 by Pujol et al. (2017) in southeastern Nicaragua. Without excluding the possibility that this deforestation also occurred in our study area during that period, the fact is that in 2015 there is an appreciable recovery of forest cover with respect to 1986.

However, the net loss of forest between 1986 and 2015 (4,734 ha) is a negative process that has affected 2.8% of the study area. This loss is relatively low because Matiguás belongs to the region of the old agricultural frontier, which was practically deforested in the '40s, '50s and '60s. The scant forest that remains in the study area belongs to the "Sierra Quirragua" nature reserve, declared as a protected natural area in 1991. For this reason, the percentage of dense forest reduction found in this study (34.5%) is not as high as in other areas of the new agricultural frontier; in fact, it is closer to the value found by Chávez (2012) between 1980 and 2010 in another municipality of Matagalpa (31% reduction). Primary forest remnants have always been subject to extraction for domestic consumption purposes such as firewood and fence posts (Agrawal et al. 2013; López et al. 2013). This clearly explains the conversion of 7,365 ha into intervened secondary forests during the 29-year period. By contrast, the decline of logging in certain areas favored the recovery of 3,511 ha of F that were previously SL and ISF in 1986.

The increase of ISF in the study area resulted mainly from the shift from SL, PHD, and to a lesser extent, DP. This transformation of the landscape suggests that the lack or reduction of anthropogenic activities has accelerated the natural regeneration in SL and stimulated the tree cover in silvopastoral land use such as PHD. In the case of DP, the low productivity of these degraded soils favored abandonment and subsequent regeneration. Ruiz et al. (2013) have also highlighted the importance of abandonment in the land use change in the North Central Region of Nicaragua between 1993 and 2011 that resulted in a substantial increase of SL. These findings agree with Gillespie et al. (2000), who have argued that most of the existing forests in Central America were generated by the abandonment of agricultural activity. On the other hand, Useche et al. (2011) have suggested that the motivation behind tree protection was the conservation of water resources as lack of water affected livestock production.

In contrast to SL, which has mainly evolved into forests, a considerable portion of PHD has deteriorated and lost forest mass. It should be mentioned that PHD is a type of land use that can be exploited for wood of high commercial value, fence posts, and firewood. This fact seems to explain the logging that has occurred in the areas previously occupied by PHD, resulting in 8,000 ha (4.6%) that were classed as PLD and DP in 2015. The opposite trend occurred in DP, since one third of the areas occupied by this class in 1986 had been transformed into land use with higher tree coverage and ecological sustainability, such as PLD, PHD, SL, and ISF, by 2015. This change may have been stimulated by incentives such as paying for environmental services (PES) (Casasola et al. 2013) or by the concerns of livestock producers about water resources (Useche et al. 2011).

The study area, therefore, is characterized by a great dynamism in terms of land use changes, which include simultaneous processes of abandonment and subsequent natural regeneration, increased tree cover, and intensification of livestock activities, probably motivated by social, ecological, and economic factors. Hence, there is no a clear sequence of degradation. Areas of forest regeneration coexist with areas where loss of tree cover has been predominant due to the intensification or expansion of livestock activity, demographic pressure, or natural disasters (INAFOR 2009). When considering both processes at the same time, the net change in Matiguás from 1986 to 2015 has favored the expansion of tree cover and the reduction of degraded areas with resultant environmental benefits.

4.2. Diversity and total carbon storage in the silvopastoral land uses

The findings of this study show that, although PLD, DP and PHD have continuous grazing activity, they retain a considerable number of trees, particularly in the case of PHD. These trees are remnants of forest or the product of tree selection by farmers, driven by different interests (Harvey et al. 2011). In spite of this, these land uses have a considerable degree of richness, contributing to preserve tree diversity in agricultural landscapes (Gordon et al. 2004). It should be noted that the higher density of trees in PHD does not translate into greater diversity than in PLD and DP, suggesting that many of the trees present in PHD belong to the same tree species. This abundance of trees in PHD benefits animal welfare because of the shade provided by tree cover (Betancourt et al. 2003).

Of the three land uses with highest BA and tree density (SL, ISF and PHD), SL has a substantially higher diversity value, if we consider the Margalef and Fisher's alpha values, suggesting that in the process of natural succession there is a recovery of diversity. It should be mentioned that in livestock landscape areas, SL has a relatively high tree diversity which could be related to the favorable climate conditions and non-intervention during the rainy season, allowing the growth of seedlings, fodder tree, leguminous and non-leguminous species, inter alia (Shelton 2000). However, dynamics change in the dry season, when forage availability is reduced and woody vegetation becomes an important source for cattle feeding (Dagang and Nair 2003), which represents an economical value for SL, but also a threat for its diversity. Contrary to expectations, the Margalef and Fisher's alpha indices barely differentiate ISF from other land uses that should, in principle, be less diverse (PLD and DP). However, the rarefaction curves do separate this land use from the others, showing it to be more similar to SL. In the case of ISF, not only browsing pressure during the dry season, but also logging contribute to the loss of diversity.

Of the five land uses studied, ISF and PHD had the largest carbon storage in the above and belowground biomass and DP had the smallest. Our range of values for aboveground carbon are consistent with data reported for pastures with trees in the area: values between 8.2 and 17.5 t C ha⁻¹ (Ruiz 2004), or, between 1.2 and 19.1 t C ha⁻¹ reported by Chacón-León and Harvey (2013); values AGB multiplied by 0.42. For ISF, Ibrahim et al. (2007) found values of 23 t C ha⁻¹ in Nicaragua (humid tropical forest), much lower than those found for the same land use in Costa Rica (90.8 t C ha⁻¹, annual T= 27.2°C; annual P= 2043 mm; sub-humid

tropical forest). Kirby and Potvin (2007) reported values of 75 t C ha^{-1} (AGB) in agroforest uses in Panamá (annual $T = 26^{\circ}\text{C}$; annual $P = 2500 \text{ mm}$; tropical moist forest) similar to those found in Costa Rica. Many factors as climate variations, soil type, or age, may contribute to explain this variability, but if part of the difference is due to management, then, ISF shows a remarkable C sequestration potential with the appropriate management.

The values of TSOC in our study ($76 - 163.6 \text{ t C ha}^{-1}$) at 1 m depth agree with various studies. For example, Amézquita et al. (2010) found values between 95 and 181 t C ha^{-1} in Pocora, a sub-humid zone of Costa Rica. In Venezuela, Delaney et al. (1997) found a wide range of values ($125\text{-}257 \text{ t C ha}^{-1}$) for forests, with 125 t C ha^{-1} corresponding to dry forests in conditions similar to those of this study. Our results also agree with those of Andrade et al. (2008) who found an average value of 110 t C ha^{-1} (down to 0.6 m) in Guanacaste (Costa Rica). However, other studies report higher values if we only take into account what is stored in the upper layers (Ruiz et al. 2004; McGroddy et al. 2015).

With respect to the vertical distribution of SOC, more than 50% is accumulated in the top 20 cm of the soil in all land uses except ISF, which has less than 40% of the total in this layer. Andrade et al. (2008) also found that more than half of the SOC accumulates in the top 20 cm in silvopastoral land uses, although their study only went down to a depth of 60 cm. In the top 20 cm there were relatively few differences between the five land uses. However, below 20 cm, ISF starts to accumulate more SOC than the other land uses, resulting in a higher total value at 1 m. Thus, TSOC in ISF is double that of DP down to 1 m. In agreement with

Takimoto et al. (2008), it is advisable to carry out SOC studies to a standard depth of 1 m, since smaller depths can give erroneous results.

The largest storage of TSOC found in ISF ($163.6 \text{ t C ha}^{-1}$) agrees with some studies but not with others. The comparison is difficult because land uses do not always coincide. For instance, Amézquita et al. (2004) found that a secondary forest in Colombia stored more TSOC (152 t C ha^{-1}) than a DP (97 t C ha^{-1}) and a mixed forage bank (86 t C ha^{-1}). However, the same authors found no differences between land uses for TSOC in Esparza (Costa Rica) and furthermore, the values tended to be lower in a secondary forest (116 t C ha^{-1}) than in a silvopastoral system (130 t C ha^{-1}). Kirby and Potvin (2007) found no differences between forest, agroforestry and pasture but their study only went down to a depth of 40 cm. Ruiz et al. (2004) found no differences between shrubland and native pasture or improved pasture with trees (SOC to 60 cm depth). In these previous studies, the quantification of SOC to a shallower depth may explain the lack of differences. On the other hand, the relatively similar values of TSOC found for the three types of silvopastoral systems (PHD, PLD and DP) is surprising since one would expect an increase in SOC with increasing tree density and basal area. McGroddy et al. (2015) also found no relationship between SOC content and tree density or biomass. The lack of significant differences between DP, PLD and PHD, and the fact that the values are relatively high, suggests that logging may have occurred recently and that the present SOC content still reflects previous land use (Mosquera et al. 2012). This may have been favored by the protection that the high clay content of these soils provides to SOC (Takimoto et al. 2008).

Soil organic carbon was the largest pool of total carbon for all the land uses studies, which agrees with other studies (Ruiz et al. 2004; McGroddy et al. 2015; Takimoto et al. 2008). While in our study the contribution by TSOOC does not fall below 90%, other studies have found differences depending on land use. Amézquita et al. (2010) report values of 62% in native forests, 90% in silvopastoral systems and 95-98% in pastures. However, Kirby and Potvin (2007) found values of 90% in pastures but only 13% in forests and 31% in agroforestry systems. The fact that the latter study only went down to 40 cm could mean that they underestimated the C of the soils in the forest and agroforestry land uses and, therefore, underestimated its contribution to the total. Again, this emphasizes the importance of normalizing the study depth for SOC.

In this study, we explored the links between biodiversity and C storage but we did not find any evidence of relationships between tree diversity and biomass C storage or SOC. The higher tree diversity in SL did not translate into a higher C storage, despite the fact that some studies highlight the benefits of complementing species (Lasky et al. 2014; Saha et al. 2009). Our result is in agreement with Kirby and Potvin (2007) who attributed the lack of relationship between diversity and SOC to the high soil variability. Since SL is a land use in transition, in our case it is also unlikely that diversity is related to SOC due to the fact that changes in soil C occur slowly, particularly in clayey soils and SOC, therefore, still reflects previous land uses. Since the connection between biodiversity and C storage still seems unclear, new studies would be necessary to confirm or reject this relationship.

The higher storage of total C in ISF, SL and PHD compared with DP indicates that deforestation, even when leaving a certain number of trees per ha (like in DP), results in a considerable loss of C and, therefore, greenhouse gas emissions. Although highly variable, the results suggest that increasing tree cover in degraded pastures would tend to increase carbon storage by 40%, in addition to other benefits. Thus, through appropriate management, degraded pastures have a high potential for carbon sequestration, even more if we consider that this degraded land use occupies a significant part of the region (Szott 2000).

The results of this study clearly show a positive balance in the period 1986-2015 with respect to total tree biomass carbon in the area of Matiguás. However, it is hard to distinguish between natural expansion or human induced afforestation. As previously mentioned, the land use that experienced greater expansion is ISF (increase of 163%) growing at the expense of SL, HDP and DP. The rearrangement of land uses implies a number of changes, that range from variations on the structure of the landscape to changes the net carbon stock. According to Chazdon et al. (2016), tropical secondary forests regeneration, subsequent to the removal of cover vegetation can actively store carbon in aboveground biomass, partially counteracting carbon emissions from deforestation. Overall, there was a net increase in global forest area, resulting in a net relatively positive carbon flux in the whole landscape. The strong increase in total C experienced in Matiguás in almost 30 years is due to that the secondary forest intervened has been the use that has increased the most while being the use with the highest content in C. Tropical forest have the potential to actively store biomass carbon,

comparable to old-growth, being not the same for soil carbon (Arneth et al. 2017; Poorter et al. 2016).

Regarding the effect of LULC changes in SOC, the response of SOC is slower than that of biomass carbon. In Matiguás, the high content of expansive clay in soils may temper even more SOC changes. This fact together with the high LULC dynamism and bidirectional changes in the area, makes not very rigorous to assign the reference SOC value previously obtained, to the areas showing in 2015 new silvopastoral uses, especially in the cases with higher tree-density, i.e. ISF, SL or PHD. But as a hypothetical exercise, in case of assuming that 2015 LULC store their corresponding biomass carbon as well as their SOC values, then, Matiguás landscape would have stored 4 Mt C more in almost three decades. Therefore, in this period, carbon sequestration in Matiguás would have ranged between 0.24 and 4 Mt C.

In order to explain this net carbon increase, it should be mentioned that from the 40's to the 70's the area had been practically deforested. During the 1980s, the area was a war zone, what provided a way to stop deforestation. The landscape started to change and improve the forest cover. After the war, a global study of socioeconomic needs, made evident the need of offering assistance to producers. Studies demonstrating the relationship between productivity and tree cover, convinced land holders of the need of having trees on their land. The area had also experienced in *payments for environmental services* through the Regional Integrated Silvopastoral Ecosystems Management project (RISEMP)

In summary, under the climatic conditions of Matiguás, the change of land use from ISF to pastures, especially those with low tree densities, means a loss of C storage, whereas forest recovery with changes from other TSPS uses to ISF mean a gain of C. Because of the importance that livestock activities have in most tropical areas, traditional silvopastoral systems are a preferable alternative to extensive cattle ranching from an economic and environmental point of view. The recovery of deforested areas can be boosted through the implementation of production systems that are applied and executed by the stakeholders.

The enhancement of the landscape matrix in livestock areas through TSPS has the potential to provide many benefits. Not only direct financial benefits for the farmer, but also global benefits as a result of improvements to soil and water quality, carbon storage in soil and aboveground biomass, linkage of isolated forest patches, the provision of habitats for wildlife and increases in biodiversity among other things. Since the systems are so complex, it is necessary to do more targeted research to better understand each of the interactions, contributions and linkages between the component.

CONCLUSIONS

5. Conclusions

The analysis of the last 30 years shows that Matiguás landscape is characterized by a great dynamism in terms of land-use changes with simultaneous processes of gains and losses of tree cover. Bidirectional movements always aimed to the livestock activity given the dominance of the traditional silvopastoral systems in the landscape.

From 1986 to 2015, the silvopastoral expansion together with the abandonment and subsequent natural regeneration has contributed to improve tree cover in the livestock landscape of Matiguás. Intervened secondary forest was the TSPS that increase most resulting in the main land use of the municipality and covering 43% of total area. Intervened secondary forest grew mainly at the expense of SL and PHD. In this process of tree recovery, degraded pasture reduced by 35%.

The dynamism in the pattern of changes in land use implies a permanent change in the structure of the landscape, the biodiversity and the carbon storage. This study showed that tree diversity is not linked to a higher tree density: SL, with fewer trees, showed higher diversity than ISF; and PHD showed similar diversity that PLD and DP. This fact may be attributed to management reasons.

Diversity and carbon storage were neither related in this study, which is explained by their different response time. Given that changes in soil C may occur slowly in the clayed soils of the area, and the fact that SOC constitutes the largest pool of total carbon for all the TSPS studied, then, connection between diversity and total carbon storage may be obscured.

Intervened secondary forest makes one of the highest contributions to the biomass carbon, the soil organic carbon and the total carbon storage, followed by SL and PHD. Degraded pasture, with the lowest total carbon, shows a high potential for carbon sequestration.

Due to the importance of carbon monitoring in LULC changes, it is of special relevance normalizing the study of C storage in the soil. It is, therefore, important that SOC is quantified to a sufficient depth to show the differences between land uses, being suggested a depth of 1 m.

Given the strong increment shown by ISF and other TSPS with high tree-density in the period 1986-2015 and their greatest C storage capacity, the C stored in the tree biomass increased by 24% at landscape level. In addition, assuming that areas with the new TSPS land uses have reached their reference SOC values (soil organic pools are stable and at equilibrium with vegetation), then Matiguás municipality could be considered an important area from the point of view of carbon sequestration.

This achievement has been done with the transformation of the peasant visions, which was possible as a result of decades of investment projects, bank loans, payments for environmental services, and permanent technical assistance packages. All together helped the peasant to make better decisions when looking for a long term welfare and thinking about the ecological implications particularly related with water resources.

However, bidirectional changes make the study area highly unstable. Although the results indicate that there is a recovery of forest cover and a decline in the degraded pastures, the high dynamism of the area could lead to undesirable land use changes. Therefore, efforts in the line of the previously mentioned, and especially those aimed to TSPS promotion, should be maintained in time to avoid a new cycle of degraded pastures expansion and the corresponding carbon stock loss.

Given that TSPS are very dynamic land uses and they play a positive role in the carbon stored in the landscape as well as provide other important ecological benefits, there is a need to integrate TSPS in the forest policies. Thus, Nicaragua needs a policy to promote forestry as an attractive activity for small to medium livestock farmers as well as to stimulate the use of TSPS to replace extensive

livestock systems in order to obtain environmental and economic benefits at the same time.

From a methodological point of view, the methodology here developed based in remote sensing techniques by using free data and open source software tools has proved to be valid in detecting LULC changes in highly fragmented landscapes, despite low geometrical resolution of the LANDSAT images. The use of free images and the application of free open source software is of special interest for monitoring purposes in developing countries, where this type of fragmented livestock landscapes are common. But, for the classification images is important to provide full information of the study area as well as trainings areas, etc. Therefore, for further study, the use of FOSS data is recommended for general and specific analysis if accompanied by field observations.

REFERENCES

6. References

- Amateis, R. L., and H. E. Burkhart. 1987. Cubic-foot volume equations for loblolly trees in cutover, site-prepared plantations. *South. J. Appl. For.* 11:190-192.
- Aide, T. M., Clark, M. L., Grau, H. R., López-Carr, D., Levy, M. A., Redo, D., ... and Muñiz, M. (2013). Deforestation and reforestation of Latin America and the Caribbean (2001–2010). *Biotropica*, 45(2), 262-271.
- Adams Jr, R. H., and Cuecuecha, A. (2010). Remittances, household expenditure and investment in Guatemala. *World Development*, 38(11), 1626-1641.
- Agrawal, A., Cashore, B., Hardin, R., Shepherd, G., Benson, C., and Miller, D. (2013). Economic contributions of forests. Background Paper 1, United Nations Forest on Forests (UNFF). 10th Session, Istanbul, Turkey. http://www.un.org/esa/forests/pdf/session_documents/unff10/EcoContrForests.pdf.
- Arnell, A., Sitch, S., Pongratz, J., Stocker, B. D., Ciais, P., Poulter, B., and Gasser, T. (2017). Historical carbon dioxide emissions caused by land-use changes are possibly larger than assumed. *Nature Geoscience*, 10(2), 79.
- Amézquita, M. C., Ibrahim, M., Llanderal, T., Buurman, P., and Amézquita, E. (2004). Carbon sequestration in pastures, silvo-pastoral systems and forests in four regions of the Latin American tropics. *Journal of Sustainable Forestry*, 21(1), 31-49.
- Amézquita, M. C., Murgueitio, E., Ibrahim, M., and Ramírez, B. (2010). Carbon sequestration in pasture and silvopastoral systems compared with native forests in ecosystems of tropical America. In *Grassland carbon sequestration: Management, policy and economics. Proceedings of the workshop on the role of grassland carbon sequestration in the mitigation of climate change: Integrated Crop Management Vol. 11*, pp. 153-161).

- Anjos, D., D. Lu., Dutra, L., and S. Sant'Anna. (2016). "Change detection techniques using multisensor data." In *Remote Sensing Handbook—Data, Characterization, Classification and Accuracies*, edited by PS Thenkabail, 377–397. CRC Pres.
- Andrade, H. J., Brook, R., and Ibrahim, M. (2008). Growth, production and carbon sequestration of silvopastoral systems with native timber species in the dry lowlands of Costa Rica. *Plant and Soil*, 308(1-2), 11-22.
- Andrade, H., and Ibrahim, M. (2003). ¿Cómo monitorear el secuestro de carbono en los sistemas silvopastoriles? *Agroforestería en las Américas*, 10(39-40), 109-116.
- Baumeister, E., and Fernández, E. (2005). Análisis de la tenencia de la tierra en Nicaragua a partir del censo agropecuario 2001. Managua: Ministerio de Agricultura Ganadería y Forestal (MAGFOR), Instituto Nicaragüense de Estadísticas y Censos (INEC), and the Food and Agriculture Organization of the United Nations (FAO).
- Beer, J., Harvey, C. A., Ibrahim, M., Harmand, J. M., Somarriba Chávez, E., and Jiménez Otárola, F. (2003). Servicios ambientales de los sistemas agroforestales.
- Bernstein, L., Bosch, P., Canziani, O., Chen, Z., Christ, R., and Riahi, K. (2008). IPCC, 2007: climate change 2007: synthesis report.
- Botero, J., Ibrahim, M., Bouman, B., Andrade, H., and Camargo, J. C. (1999). Modelaje de opciones silvopastoriles sostenibles para el sistema ganadero de doble propósito en el trópico húmedo. *Agroforestería en las Américas*, 6(23), 60.
- Betancourt, K., Ibrahim, M., Harvey, C., and Vargas, B. (2003). Efecto de la cobertura arbórea sobre el comportamiento animal en fincas ganaderas de doble propósito en Matiguás, Matagalpa, Nicaragua. *Agroforestería en las Américas*, 10(39-40), 47-51.

- Bennetzen, E. H., Smith, P., and Porter, J. R. (2016). Decoupling of greenhouse gas emissions from global agricultural production: 1970–2050. *Global change biology*, 22(2), 763-781.
- Breiman, L. (2001). Random forests. *Machine learning*, 45(1), 5-32.
- Bricklemeyer, R. S., Lawrence, R. L., Miller, P. R., and Battogtokh, N. (2007). Monitoring and verifying agricultural practices related to soil carbon sequestration with satellite imagery. *Agriculture, ecosystems and environment*, 118(1-4), 201-210.
- Brown, S. (1997). Estimating biomass and biomass change of tropical forests: a primer. Food and Agriculture Organization (Forestry Paper 134).
- Brown, S. (2002). Measuring, monitoring, and verification of carbon benefits for forest-based projects. *Philosophical Transactions of the Royal Society of London A: Mathematical, Physical and Engineering Sciences*, 360(1797), 1669-1683.
- Bullock, E., and Holden, C. (2015). ROIExplorer. QGIS python plugin. Online: https://github.com/beeoda/roi_plugin
- Cárdenas, A., and Schernthanner, H. (2017). Random_Forest_Landsat, GitHub repository, Available from: https://github.com/hatschito/Random_Forest_Landsat/
- Casasola, F., Ibrahim, M., Harvey, C., and Kleinn, C. (2001). Caracterización y productividad de sistemas silvopastoriles tradicionales en Moropotente, Estelí, Nicaragua. *Agroforestería en las Américas*, 8(30), 17-20.
- Casasola Coto, F., Ibrahim, M., Ramírez, E., Villanueva Najarro, C., Sepúlveda López, C. J., and Araya, J. L. (2013). Pago por servicios ambientales y cambios en los usos de la tierra en paisajes dominados por la ganadería en el trópico subhúmedo de Nicaragua y Costa Rica.

- Congalton, R. G., and Green, K. (2008). Assessing the accuracy of remotely sensed data: principles and practices. CRC press.
- Congedo, L. (2016). Semi-automatic classification plugin documentation. Release, 4(0.1), 29.
- Congedo, L. (2014). Land Cover Classification of Cropland: a Tutorial Using the Semi-Automatic Classification Plugin for QGIS.
- Chávez, E. (2012). Análisis hidrológico ante impactos del cambio climático y cambios de uso del suelo en la cuenca del río Compasagua, Nicaragua.
- Chavez, P. S. (1996). Image-based atmospheric corrections-revisited and improved. Photogrammetric engineering and remote sensing, 62(9), 1025-1035.
- Chacón León, M. (2003). Cobertura arbórea y cercas vivas en un paisaje fragmentado Río Frío, Costa Rica Tree cover and livefences in a fragmented landscape in Rio Frio, Costa Rica (No. Thesis C431cob). CATIE, Turrialba (Costa Rica).
- Chacón-León, M., and Harvey, C. A. (2013). Reservas de biomasa de árboles dispersos en potreros y mitigación al cambio climático. Agronomía Mesoamericana, 24(1).
- Chazdon, R. L. (2008). Beyond deforestation: restoring forests and ecosystem services on degraded lands. science, 320(5882), 1458-1460.
- Chazdon, R. L., and Guariguata, M. R. (2016). Natural regeneration as a tool for large-scale forest restoration in the tropics: prospects and challenges. Biotropica, 48(6), 716-730.
- Chazdon, R. L., Harvey, C. A., Komar, O., Griffith, D. M., Ferguson, B. G., Martínez-Ramos, M., and Philpott, S. M. (2009). Beyond reserves: A research agenda for conserving biodiversity in human-modified tropical landscapes. Biotropica, 41(2), 142-153.

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- Cubbage, F., Balmelli, G., Bussoni, A., Noellemeyer, E., Pachas, A. N., Fassola, H., ... and de Silva, M. L. (2012). Comparing silvopastoral systems and prospects in eight regions of the world. *Agroforestry Systems*, 86(3), 303-314.
- Dagang, A. B., and Nair, P. K. R. (2003). Silvopastoral research and adoption in Central America: recent findings and recommendations for future directions. *Agroforestry systems*, 59(2), 149-155.
- Delaney, M., Brown, S., Lugo, A. E., Torres-Lezama, A., and Quintero, N. B. (1997). The distribution of organic carbon in major components of forests located in five life zones of Venezuela. *Journal of tropical ecology*, 13(5), 697-708.
- ERCA: Estado de la region Centro Americana. (2016). Quinto Informe Estado de la Región en Desarrollo Humano Sostenible Programa Estado de la Nación – Región. Apdo. 1174 - 1200, Pavas - Costa Rica.
- Eichhorn, M. P., Paris, P., Herzog, F., Incoll, L. D., Liagre, F., Mantzanas, K., and Pisanelli, A. (2006). Silvoarable systems in Europe—past, present and future prospects. *Agroforestry systems*, 67(1), 29-50.
- Evans, S. J. (2018). spatialEco. package version 1.1-0. Online: <https://cran.r-project.org/web/packages/spatialEco/index.html>
- FAO. (2001). Food and Agriculture Organization. Global Forest Resources Assessment 2000 Main Report. FAO Forestry paper 140. FAO, Rome, Italy. Available at <http://www.fao.org/docrep/004/Y1997E/Y1997E00.HTM>
- Feng, Q., Liu, J., and Gong, J. (2015). UAV remote sensing for urban vegetation mapping using random forest and texture analysis. *Remote Sensing*, 7(1), 1074-1094.
- Fisher, R. A., Corbet, A. S., and Williams, C. B. (1943). The relation between the number of species and the number of individuals in a random sample of an animal population. *The Journal of Animal Ecology*, 42-58.

- Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., ... and Balzer, C. (2011). Solutions for a cultivated planet. *Nature*, 478(7369), 337.
- Forman R. T. T. (1995). *Land Mosaics- the ecology of landscape and regions*. Cambridge, UK. 632
- Forman, R. T. T. and Godron M. (1986). *Landscape ecology*. John Wiley, New York.
- Gardner, C. M., Laryea, K. B., and Unger, P. W. (1999). *Soil physical constraints to plant growth and crop production*. Land and Water Development Division, Food and Agriculture Organization.
- Gillespie, T. W., Grijalva, A., and Farris, C. N. (2000). Diversity, composition, and structure of tropical dry forests in Central America. *Plant ecology*, 147(1), 37-47.
- Gordon, A. M., Newman, S. M., and Coleman, B. (Eds.). (2018). *Temperate agroforestry systems*. CABI.
- Gobbi, J. A. (2001). *Evaluación socioeconómica. Proyecto Regional Integrated Silvopastoral Approaches to Ecosystems Management*. Turrialba, CR: GEF-CATIE, 200.
- Gordon, J. E., Hawthorne, W.D., Reyes-Garcia, A., Sandoval, G., and Barrance, A.J. (2004). Assessing landscapes: a case study of tree and shrub diversity in the seasonally dry tropical forests of Oaxaca, Mexico and southern Honduras. *Biological conservation*, 117(4), 429-442.
- Guo, L., Chehata, N., Mallet, C., and Boukir, S. (2011). Relevance of airborne lidar and multispectral image data for urban scene classification using Random Forests. *ISPRS Journal of Photogrammetry and Remote Sensing*, 66(1), 56-66.
- GFW: Global forest Watch. (2018). <https://www.globalforestwatch.org/country/NIC>

-
- GLCF: Global land cover facility. MODIS Land Cover - Product MCD12Q1. (2017). <http://glcf.umd.edu/> Accessed: 2017-05- 20.
- Grau, H. R., and Aide, M. (2008). Globalization and land-use transitions in Latin America. *Ecology and society*, 13(2).
- Harvey, C., Alpízar, F., Chacón, M., and Madrigal, R. (2005a). Assessing linkages between agriculture and biodiversity in Central America: Historical overview and future perspectives. The Nature Conservancy.
- Harvey, C. A., Haber, W. A., Mejias, F., and Solano, R. (1998). Remnant trees in Costa Rican pastures: tools for conservation? *Agroforestry Today*, 10(3), 7-9.
- Harvey, C. A., Komar, O., Chazdon, R., Ferguson, B. G., Finegan, B., Griffith, D. M., ... and Van Breugel, M. (2008). Integrating agricultural landscapes with biodiversity conservation in the Mesoamerican hotspot. *Conservation biology*, 22(1), 8-15.
- Harvey, C. A., Medina, A., Sánchez, D. M., Vílchez, S., Hernández, B., Saenz, J. C., ... and Sinclair, F. L. (2006). Patterns of animal diversity in different forms of tree cover in agricultural landscapes. *Ecological applications*, 16(5), 1986-1999.
- Harvey C. A., Villanueva C., Esquivel H., Gómez R., Ibrahim M., López M., Martinez J., Muñoz D., Restrepo C., Saénz J. C., Villacís J., and Sinclair F. L. (2011). Conservation value of dispersed tree cover threatened by pasture management. *Forest Ecology and Management*, 261(10), 1664-1674.
- Harvey, C. A., Villanueva, C., Villacís, J., Chacón, M., Muñoz, D., López, M., ... and Navas, A. (2003). Contribución de las cercas vivas a la productividad e integridad ecológica de los paisajes agrícolas en América Central. *Agroforestería en las Américas*, 10(39-40), 30-39.
- Hansen, M. C., DeFries, R. S., Townshend, J. R., and Sohlberg, R. (2000). Global land cover classification at 1 km spatial resolution using a classification tree approach. *International journal of remote sensing*, 21(6-7), 1331-1364.

- Hassler, S. K., Zimmermann, B., van Breugel, M., Hall, J. S., and Elsenbeer, H. (2011). Recovery of saturated hydraulic conductivity under secondary succession on former pasture in the humid tropics. *Forest Ecology and Management*, 261(10), 1634-1642.
- Hecht, S. B. (2014). Forests lost and found in tropical Latin America: the woodland 'green revolution'. *Journal of Peasant Studies*, 41(5), 877-909.
- Holmann, F. J., and Estrada, R. D. (1997). Alternativas agropecuarias en la Región Pacífico Central de Costa Rica: Un modelo de simulación aplicable a sistemas de doble propósito. Publicación CIAT (Colombia).
- Hoosbeek, M. R., Remme, R. P., & Rusch, G. M. (2018). Trees enhance soil carbon sequestration and nutrient cycling in a silvopastoral system in south-western Nicaragua. *Agroforestry Systems*, 92, 263-273. <https://doi.org/10.1007/s10457-016-0049-2>.
- Howlett, D. S., Mosquera-Losada, M. R., Nair, P. K., Nair, V. D., and Rigueiro-Rodríguez, A. (2011). Soil carbon storage in silvopastoral systems and a treeless pasture in northwestern Spain. *Journal of environmental quality*, 40(3), 825-832.
- Ibrahim, M., Beer, J., Ruiz, A., Locatelli, B., and Andrade Castañeda, H. J. (2004). Fijación y almacenamiento de carbono en sistemas silvopastoriles y competitividad económica de fincas ganaderas en Matiguás, Nicaragua.
- Ibrahim, M., Camargo, J. C., and Carvalho, M. M. (2001). Produtividade e serviços ambientais de sistemas silvipastoris: experiências do Catie. *SISTEMAS AGROFLORESTAIS PECUÁRIOS: Opções de sustentabilidade para áreas tropicais e subtropicais*. Juiz de Fora: Embrapa Gado de Leite, 331-347.
- Ibrahim, M., Camero, A., Camargo, J. C., and Andrade, H. J. (1999). sistemas silvopastoriles en América Central: experiencias de CATIE. In *Congreso Latinoamericano sobre Sistemas Agroforestales para la Producción Agrícola*. Fundación CIPAV: Cali.

- Ibrahim, M., Chacón, M., Cuartas, C., Naranjo, J., Ponce, G., Vega, P., ... and Rojas, J. (2013). Almacenamiento de carbono en el suelo y la biomasa arbórea en sistemas de usos de la tierra en paisajes ganaderos de Colombia, Costa Rica y Nicaragua.
- Ibrahim, M., Guerra, L., Casasola, F., and Neely, C. (2010). Importance of silvopastoral systems for mitigation of climate change and harnessing of environmental benefits. *Grassland carbon sequestration: management, policy and economics*, 11, 189.
- Ibrahim, M., and Harvey, C. A. (2003). Diseño y manejo de la cobertura arbórea en fincas ganaderas.
- Ibrahim, M., Villanueva, C., Casasola, F., and Rojas, J. (2006). Sistemas silvopastoriles como una herramienta para el mejoramiento de la productividad y restauración de la integridad ecológica de paisajes ganaderos. *Pastos y Forrajes*, 29(4), 383-419.
- Ibrahim, M., Villanueva, C., and Mora, J. (2005). Traditional and improved silvopastoral systems and their importance in sustainability of livestock farms. *Silvopastoralism and sustainable land management*. (MR Mosquera, A. Riguerio and J. McAdam, Eds.). CAB. Wallingford, UK, 13.
- Jackson, J., and Ash, A. J. (1998). Tree-grass relationships in open eucalypt woodlands of northeastern Australia: influence of trees on pasture productivity, forage quality and species distribution. *Agroforestry systems*, 40(2), 159-176.
- Jensen, J. R. (2009). *Remote sensing of the environment: An earth resource perspective* 2/e. Pearson Education India.
- Jose, S. (2009). Agroforestry for ecosystem services and environmental benefits: an overview. *Agroforestry systems*, 76(1), 1-10.

- Kanninen, M. (2000). Secuestro de carbono en los bosques: El papel de los bosques en el ciclo global de carbono. *Intensificación de la Ganadería en Centroamérica: Beneficios Económicos y Ambientales*. Eds. C Pomareda, H Steinfeld, 137-149.
- Kanninen, M., and Mery, G. (2012). Carbon sinks in different forest ecosystems in Latin America. *World Forests, Markets and Policies*, 3, 68.
- Kattan, G. H., and Alvarez-López, H. (1996). Preservation and management of biodiversity in fragmented landscapes in the Colombian Andes. *Forest patches in tropical landscapes*, 3-18.
- Kaimowitz D. (1996). Livestock and deforestation in Central America in the 1980s and 1990s: a policy perspective (No. 9). Cifor.
- Kaufman Y., and L. Remer (1994) Detection of forests using mid-IR reflectance: An application for aerosol studies. *IEEE T. Geosci.Remote*. 32(3):672-683.
- Keranen, K., and Kolvoord, R. (2008). *Making spatial decisions using GIS* (Vol. 4). ESRI, Inc.
- Kirby, K.R., and Potvin, C. (2007) Variation in carbon storage among tree species: implications for the management of a small-scale carbon sink project. *Forest Ecology and Management*, 246(2), 208-221.
- Kleinn, C. (2000). On large-area inventory and assessment of trees outside forests. *Unasylva* (English ed.), 51(200), 3-10.
- Kottek, M., Grieser, J., Beck, C., Rudolf, B., and Rubel F. (2006). World map of the Köppen-Geiger climate classification updated. *Meteorologische Zeitschrift*, 15(3), 259-263.
- Krebs, C.J. (1989). *Ecological methodology* (No. QH541. 15. S72. K74 1999.). New York: Harper and Row.

-
- Lamb, D., Erskine, P. D., and Parrotta, J. A. (2005). Restoration of degraded tropical forest landscapes. *Science*, 310(5754), 1628-1632.
- Lasky, J. R., Uriarte, M., Boukili, V. K., Erickson, D. L., John Kress, W., and Chazdon RL. (2014). The relationship between tree biodiversity and biomass dynamics changes with tropical forest succession. *Ecology letters*, 17(9), 1158-1167.
- Lawrence D (2004) Erosion of tree diversity during 200 years of shifting cultivation in Bornean rain forest. *Ecological Applications*, 14(6), 1855-1869.
- Lawrence, D., and Vandecar, K. (2015). Effects of tropical deforestation on climate and agriculture. *Nature Climate Change* 5.1 (2015): 27.
- Lawrence, R. L., Wood, S. D., and Sheley, R. L. (2006). Mapping invasive plants using hyperspectral imagery and Breiman Cutler classifications (RandomForest). *Remote Sensing of Environment*, 100(3), 356-362.
- León Lau Williams, W. (2009). Resultados del inventario nacional forestal: Nicaragua, 2007-2008. Ministerio Agropecuario Forestal, Managua (Nicaragua). Instituto Nacional Forestal, Managua (Nicaragua).
- Leutner, B., Horning, N., Schwalb-Willmann J., and Hijmans R-J. (2018). Rstoolbox. R package version 0.2.1. Online: <https://github.com/bleutner/RStoolbox>
- Levard, L., Marín López, Y., and Navarro I. (2001). Municipio de Matiguás: Potencialidades y limitantes del desarrollo agropecuario.
- Lillesand, T., Kiefer, R. W., and Chipman, J. (2014). Remote sensing and image interpretation. John Wiley & Sons.
- Lie, H., and Rich, K. M. (2016). Modeling Dynamic Processes in Smallholder Dairy Value Chains in Nicaragua: A System Dynamics Approach. *Int. J. Food System Dynamics* 7 (4), 328-340

- Lindsey, R. (2007). Tropical deforestation. NASA's Earth Observatory, 508.
- López, F., Gómez, R., López, M., Harvey, C. A., and Sinclair, F. L. (2013). Toma de decisiones de productores ganaderos sobre el manejo de los árboles en potreros en Matiguás. Nicaragua.
- López, A., and Gordo, A. (2006). Sample Size and Confidence when Applying the NSSDA. Grupo de Investigación en Ingeniería Cartográfica, Universidad de Jaén, Jaén, España, < <http://www.cartesia.org/geodoc/icc2005/pdf/poster/TEMA7/Francisco>, 20.
- López-Carr, D., Pricope, N. G., Aukema, J. E., Jankowska, M. M., Funk, C., Husak, G., & Michaelsen, J. (2014). A spatial analysis of population dynamics and climate change in Africa: potential vulnerability hot spots emerge where precipitation declines and demographic pressures coincide. *Population and Environment*, 35(3), 323-339.
- Lugo, A. E. (2009). The emerging era of novel tropical forests. *Biotropica*, 41(5), 589-591.
- McDonald, R. I., Motzkin, G., & Foster, D. R. (2008). The effect of logging on vegetation composition in Western Massachusetts. *Forest ecology and management*, 255(12), 4021-4031.
- MacDicken, K. G. (1997). A guide to monitoring carbon storage in forestry and agroforestry projects.
- Magurran, A. E. (1981). Biological diversity and woodland management: an investigation with special reference to Banagher, Co. Derry, N. Ireland (Doctoral dissertation, New University of Ulster).
- Maldidier, C., and Marchetti, P. (1996). El Campesino-Finquero y el potencial económico del campesinado nicaragüense. UCA.
- Margalef, R. (1958). Information theory in ecology. *Gen. Systems* 3, 36-71.

-
- McGroddy, M. E., Lerner, A. M., Burbano, D. V., Schneider, L. C., and Rudel, T. K. (2015). Carbon stocks in silvopastoral systems: A study from four communities in southeastern Ecuador. *Biotropica*, 47(4), 407-415.
- Meyrat, A. (2000). Los ecosistemas y formaciones vegetales de Nicaragua. Protierra/MARENA/CBA. Managua, Nicaragua.
- Medina, A., Harvey, C. A., Merlo, D. S., Vilchez, S., and Hernández, B. (2007). Bat diversity and movement in an agricultural landscape in Matiguás, Nicaragua. *Biotropica*, 39(1), 120-128.
- Montagnini, F., Ibrahim, I., and Murgueitio, E. (2013). Silvopastoral systems and climate change mitigation in Latin America. *Bois et Forêts des Tropiques*, 316(2), 3-16.
- Montagnini, F., and Nair, P. K. R. (2004). Carbon sequestration: an underexploited environmental benefit of agroforestry systems. In *New vistas in agroforestry* (pp. 281-295). Springer, Dordrecht.
- Mortelliti, A., Amori, G., and Boitani, L. (2010). The role of habitat quality in fragmented landscapes: a conceptual overview and prospectus for future research. *Oecologia*, 163(2), 535-547.
- Mosquera, O., Burman, P., Ramirez, B. L., and Amezquita, M. C. (2012). Carbon stocks and dynamics under improved tropical pasture and silvopastoral systems in Colombian Amazonia. *Geoderma*, 189, 81-86.
- Moss, B. (2008). Water pollution by agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363(1491), 659-666.
- Murcia, C. (1995). Edge effects in fragmented forests: implications for conservation. *Trends in ecology and evolution*, 10(2), 58-62.

- Murgueitio, E., Calle, Z., Uribe, F., Calle, A., and Solorio, B. (2011). Native trees and shrubs for the productive rehabilitation of tropical cattle ranching lands. *Forest Ecology and Management*, 261(10), 1654-1663.
- Nair, P. R., and Nair, V. D. (2003). Carbon storage in North American agroforestry systems. The potential of US forest soils to sequester carbon and mitigate the greenhouse effect, 333-346.
- Noble, I., Bolin, B., Ravindranath, N. H., Verardo, D. J., and Dokken, D. J. (2000). *Land use, land use change, and forestry*. Cambridge University Press.
- Nyberg, G., and Högberg, P. (1995). Effects of young agroforestry trees on soils in on-farm situations in western Kenya. *Agroforestry systems*, 32(1), 45-52.
- Oksanen, J., Blanchet, F., Kindt, R., Legendre, P., Minchin, P. R., O'Hara, R. B., ... and Wagner, H. (2014). *vegan: Community Ecology Package*. R package version 2.1-41/r2867.
- Pagiola, S., Arcenas, A., and Platais, G. (2005). Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. *World development*, 33(2), 237-253.
- Pagiola, S., Ramírez, E., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., and Ruíz, J. P. (2007). Paying for the environmental services of silvopastoral practices in Nicaragua. *Ecological Economics*, 64(2), 374-385.
- Pal, M. (2005). Random forest classifier for remote sensing classification. *International Journal of Remote Sensing*, 26(1), 217-222.
- Pandey, D. N. (2002). Carbon sequestration in agroforestry systems. *Climate policy*, 2(4), 367-377.

- Polvorosa, J. C. (2013). Opportunities and constraints for small and medium-sized farmers in the context of the booming dairy value chains in Nicaragua: case study of Matiguás (Doctoral dissertation, University of Antwerp).
- Pujol, P., Sauri, D., Martí, C., and Pujadas, M. (2017). Cambios en el uso del suelo en el sudeste de Nicaragua, 1983-1992. *Encuentro*, (51), 93-103.
- Ravnborg, H. M. (2008). Organising to protect: Protecting landscapes and livelihoods in the Nicaraguan hillsides. *Conservation and Society*, 6(4), 283.
- Reid, W. V. 8c MILLER, KR 1989. Keeping Options Alive: the Scientific Basis for Conserving Biodiversity. World Resources Institute, Washington, DC.
- Rigueiro-Rodríguez, A., McAdam, J., and Mosquera-Losada, M. R. (2008). Agroforestry in Europe: current status and future prospects (Vol. 6). Springer Science and Business Media.
- Rodriguez-Galiano, V. F., Ghimire, B., Rogan, J., Chica-Olmo, M., and Rigol-Sanchez, J. P. (2012). An assessment of the effectiveness of a random forest classifier for land-cover classification. *ISPRS Journal of Photogrammetry and Remote Sensing*, 67, 93-104.
- Ruiz, V., Savé, R., and Herrera, A. (2013). Análisis multitemporal del cambio de uso del suelo, en el Paisaje Terrestre Protegido Mirafior Moropotente Nicaragua, 1993–2011. *Ecosistemas*, 22(3).
- Ruiz, A., Ibrahim, M., Beer, J., Locatelli, B., and Andrade Castañeda, H. J. (2004) Fijación y almacenamiento de carbono en sistemas silvopastoriles y competitividad económica de fincas ganaderas en Matiguás, Nicaragua.
- Saha, S. K., Nair, P. R., Nair, V. D., and Kumar, B. M. (2009). Soil carbon stock in relation to plant diversity of homegardens in Kerala, India. *Agroforestry systems*, 76(1), 53-65.

- Sánchez Merlos, D., Harvey, C. A., Grijalva, A., Medina, A., Vílchez, S., and Hernández, B. (2005). Diversidad, composición y estructura de la vegetación en un agropaisaje ganadero en Matiguás, Nicaragua. *Revista de biología tropical*, 53(3-4), 387-414.
- San Miguel Ayanz, A. (2005). Mediterranean European silvopastoral systems. *Silvopastoralism and sustainable land management*. CABI Publishing, Wallingford, UK, 36-40.
- Saubes, N., y Gálvez, J. (2015). Estado de la conservación y uso de los recursos naturales en Centroamérica. Quinto informe estado de la región. Programa Estado de la Nación – Región. Apdo. 1174 - 1200, Pavas - Costa Rica
- Segura, M., and Kanninen, M. (2002). Inventario para estimar carbono en ecosistemas forestales tropicales. Orozco, L. y Brumer, C. *Inventarios forestales para bosques latifoliados en América Central*. Turrialba: CATIE, 202-222.
- Sepúlveda López, C. J., and Ibrahim, M. (2013). Políticas y sistemas de incentivos para el fomento y adopción de buenas prácticas agrícolas: como una medida de adaptación al cambio climático en América Central.
- Scherr, S. J., and McNeely, J. A. (2008). Biodiversity conservation and agricultural sustainability: towards a new paradigm of 'ecoagriculture' landscapes. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363(1491), 477-494.
- Schwartz, N. B., Uriarte, M., DeFries, R., Gutierrez-Velez, V. H., and Pinedo-Vasquez, M. A. (2017). Land-use dynamics influence estimates of carbon sequestration potential in tropical second-growth forest. *Environmental Research Letters*, 12(7), 074023.
- Shelton, M. (2000). Tropical forage tree legumes: Key development issues.

-
- Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B. and Miller, H. L. (2007). Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change.
- Solomon, S., Plattner, G. K., Knutti, R., & Friedlingstein, P. (2009). Irreversible climate change due to carbon dioxide emissions. *Proceedings of the national academy of sciences*, 106(6), 1704-1709.
- Steele, B. M., Winne, J. C., and Redmond, R. L. (1998). Estimation and mapping of misclassification probabilities for thematic land cover maps. *Remote Sensing of Environment*, 66(2), 192-202.
- Swann, A. L., Longo, M., Knox, R. G., Lee, E., and Moorcroft, P. R. (2015). Future deforestation in the Amazon and consequences for South American climate. *Agricultural and Forest Meteorology*, 214, 12-24.
- Shao, G., and Wu, J. (2008). On the accuracy of landscape pattern analysis using remote sensing data. *Landscape Ecology*, 23(5), 505-511.
- Takimoto, A., Nair, P. R., and Nair, V. D. (2008). Carbon stock and sequestration potential of traditional and improved agroforestry systems in the West African Sahel. *Agriculture, Ecosystems and Environment*, 125(1-4), 159-166.
- Technoserve. (2016). Building a Competitive and Inclusive Livestock Sector in Nicaragua. A Case Study of the Ganadería Empresarial Project (2012-2016)
- Thomlinson, J. R., Bolstad, P. V., and Cohen, W. B. (1999). Coordinating methodologies for scaling landcover classifications from site-specific to global: Steps toward validating global map products. *Remote Sensing of Environment*, 70(1), 16-28.
- Tobar, D. E., and Ibrahim, M. (2010). ¿ Las cercas vivas ayudan a la conservación de la diversidad de mariposas en paisajes agropecuarios?. *Revista de Biología Tropical*, 58(1), 447-463.

- Tobler, W. (1987). Measuring spatial resolution. In *Proceedings of the International Workshop on Geographic Information Systems* (Vol. 48).
- Turner, M. G., Gardner, R. H., and O'Neill, R. V. (2001). *Landscape ecology in theory and practice* (Vol. 401). New York: Springer.
- Useche, D., Harvey, C. A., and DeClerck, F. A. (2011). *Conservación de bosques tropicales en fincas ganaderas privadas de Centroamérica. Estudio de caso: Matiguás, Nicaragua*.
- USGS. (2018). EarthExplorer. Available from: <https://earthexplorer.usgs.gov/>
- West, P. C., Gibbs, H. K., Monfreda, C., Wagner, J., Barford, C. C., Carpenter, S. R., and Foley, J. A. (2010). Trading carbon for food: Global comparison of carbon stocks vs. crop yields on agricultural land. *Proceedings of the National Academy of Sciences*, 107(46), 19645-19648.
- Wickham, H., Chan, W., Rstudio (2016). Ggplot. R package version 2.21. Online: <https://github.com/tidyverse/ggplot2>
- Wilson, B. F. (1995). Shrub stems: form and function. In *Plant Stems* (pp. 91-102).
- Wright, S. J. (2010). The future of tropical forests. *Annals of the New York Academy of Sciences*, 1195(1), 1-27.
- Wu, J. (2007). Scale and scaling: a cross-disciplinary perspective. In *Key topics in landscape ecology*. Cambridge University Press.
- Wu, Y. and Z. Zhu. (1997). Chapter 5. Temperate Agroforestry in China. In *Temperate Agroforestry Systems*. A.M. Gordon and S.M. Newman (eds). CAB International, Wallingford, UK.
- Wunder, S. (2001). Poverty alleviation and tropical forests—what scope for synergies? *World development*, 29(11), 1817-1833.

- Yamamoto, W., Ap Dewi, I., and Ibrahim, M. (2007). Effects of silvopastoral areas on milk production at dual-purpose cattle farms at the semi-humid old agricultural frontier in central Nicaragua. *Agricultural Systems*, 94(2), 368-375.
- Zamora, S., García, J., Bonilla, G., Aguilar, H., Harvey, C. A., and Ibrahim, M. (2001). ¿Cómo utilizar los frutos de guanacaste (*Enterolobium cyclocarpum*), guácimo (*Guazuma ulmifolia*), genízaro (*Pithecellobium saman*) y jícara (*Crescentia alata*) en alimentación animal?.
- Zegeye, H. (2017). Major drivers and consequences of deforestation in Ethiopia: Implications for forest conservation. *Asian Journal of Science and Technology*, 08 (08), 5166-5175.
- Zharikov, Y., Skilleter, G. A., Loneragan, N. R., Taranto, T., and Cameron, B. E. (2005). Mapping and characterising subtropical estuarine landscapes using aerial photography and GIS for potential application in wildlife conservation and management. *Biological Conservation*, 125(1), 87-100.
- Zomer, R. J., Neufeldt, H., Xu, J., Ahrends, A., Bossio, D., Trabucco, A., ... and Wang, M. (2016). Global Tree Cover and Biomass Carbon on Agricultural Land: The contribution of agroforestry to global and national carbon budgets. *Scientific reports*, 6, 29987.