ANALYSIS OF TRADE-OFFS AMONG CARBON SEQUESTRATION, TIMBER PRODUCTION AND WATER YIELD IN NORTH FLORIDA SLASH PINE FORESTS, USA

By

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A THESIS PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
OF THE REQUIREMENTS FOR THE DEGREE OF
MASTER OF SCIENCE

UNIVERSITY OF FLORIDA

2013

To my mother and my father

ACKNOWLEDGMENTS

First of all I would like to thank my family: my parents and my brothers, Wilson, Rodne, Widnord and Lever; my cousins, Loulou, Naomie and Jean-Yves; my uncle Gerard, for their love and encouragement from thousand miles away. Especially, I would like to thank Dr. Francisco Escobedo for all his advice and support during my master program. His patience, comprehension, motivation and above all friendship, this thesis work would have not been possible. I also would like to thank Dr. Amr Abd-Elrahman and Dr. Matthew Cohen for their valuable comments and contributions to help improve this thesis document, and also for their teaching that helped me develop my critical thinking. I want to thank Watershed Initiative for National Natural Resources and Environment (WINNER) project of the United States Agency for International development (USAID), The Florida Forest Service's Stewardship Ecosystem Service Survey study and the School of Natural Resources and Environment (SNRE) for financial support to realize this work and my master program, especially, Dr. Stephen Humphrey, Florence Sergile and Melissa Wokasch. This research would not have been done without support and contributions from Dr. Daniel McLaughlin, Dr. Carlos Gonzalez-Benecke and Dr. Nilesh Timilsina. I also would like to thank Alexandra Jean-Charles for supporting and encouraging me in difficult times. I also want to thank those special friends: Sonia Delphin, Olivia Jeanne, Yamil Rodriguez, Symithe Steeve Julien, Cassandra Coulanges, Peggy and Roger Sedlaceck for trusting me and give me all the motivation I need to finish this work. I want to thank my colleagues Arthur, Beneche, Dakson, Isnel, Lidwine, Pascale, Reginald and Dr. Lemane Delva, for being part of this process. I also want to thank my friends in Gainesville: Angelica, Martha, Oscar, Ivelisse, Silvia, Sofia, Kimmel, Marco, and Cristobal.

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May 2013

Chair: Francisco Escobedo
Major: Interdisciplinary Ecology

Managing forests solely for carbon sequestration and timber production objectives might have negative effects on the provision of water due to high losses from evapotranspiration. Therefore, information on the interactions among these three ecosystems services and how they are spatially bundled can provide useful insights for land management decision making. This study used a US Forest Services inventory dataset and computed leaf area index (LAI) to quantify levels of provision and service bundles of carbon sequestration, timber volume and water yield for slash pine sites in North Florida. Moreover, using a ranking classification approach, we determined spatially-explicit interactions among the services as well as the effect of drivers such as stand age, site productivity, silvicultural treatments, ownership, and disturbance regime on individual ecosystem services and on the interactions (i.e., trade-off/synergy). Results indicated that growth in biomass reduced water yield during the study period. Nevertheless, this trade-off varied across space, as revealed by the lack of correlation in the model of water yield as function of net carbon sequestration or timber volume. Specific areas of synergy among the 3 ecosystem services were found. Also, the

results indicated that, although the effect of some drivers was not statistically significant on individual services, all the drivers analyzed affected the interaction among the services, with stand age, silvicultural treatment and site quality, the most significant. Finally, the framework developed in this study can be used to assess and manage natural ecosystems for multiple and optimal provision of services to people.

CHAPTER 1
INTRODUCTION

Ecosystem Service Framework

The term ecosystem services has been increasingly used and cited in several scientific publications (Assessment, 2003; Costanza et al., 1998; Daily, 1997; Daily, 2000; Guo, Xiao, & Li, 2000). Nevertheless, this term is often used interchangeably with ecological processes, ecosystem functions or environmental services, ecosystem goods, etc. (Bennett, Peterson, & Gordon, 2009; Daily, 1997; de Groot, Alkemade, Braat, Hein, & Willemen, 2010). Understanding the differences between those terms is key for communication among users of the ecosystem service concept (de Groot et al., 2010). Ecosystem services are the benefits human receive as a result of the natural processes from the structures of an ecosystem (Daily, 2000; de Groot, Wilson, & Boumans, 2002; Egoh et al., 2007). However, because of the human controls on the biosphere, many ecosystem services are under threat (Heal, Daily, Ehrlich, & Salzman, 2001).

The Millennium Ecosystem Assessment (2003), has suggested four main categories for ecosystem services: provisioning services, including fresh water, timber production and all other direct goods or services provided to human being; regulating services, such as flood control and climate regulation; cultural services, related to recreation and aesthetic; and finally, supporting services, which are the natural processes upon which the other categories are dependent, such as nutrient cycling, soil formation, and primary production. Although there has been a growing interest in the scientific study of ecosystem services for conservation planning and management (de Groot et al., 2010) since the Millennium Ecosystem Assessment in 2005 (Carpenter, 2005), a lot of effort is still needed to fully understand the concept and to systematically characterize the concept in measurable biophysical terms (Daily, 2000).

Daily (1997), proposed four important elements to better comprehend the ecosystem service framework. First, it is important to identify the stocks of ecosystem structures that provide the ecosystem services by mapping their geographic location. Another element is the ecological characterization, which is a key element that relates to the ecosystem's functions and allows to exploration of the inherit interdependency between ecosystem services (e. g, carbon sequestration, timber and water) and the level at which the production of different services together can be sustainable. Finally, it is imperative to monitor changes in the provision of ecosystem services by using appropriate indicators (e. g., carbon sequestration, water quantity).

The quantification of ecosystem services is also an important step to the process of ecological compensation by which owners of biological resources (e.g., forest property owners) can be provided with financial benefits for conserving or managing their properties to provide services to others (Guo et al., 2000). For example, Guo et al. (200), used simulation models coupled with spatial analysis to quantitatively determine the capacity of forest ecosystem to regulate the water flow within a watershed and also included an economic valuation. Also Egoh, Reyers, Rouget, and Richardson (2011) quantitatively evaluated the bundle of ecosystem services supplied by grasslands in South Africa and suggested that when identifying the right target area for a given ecosystem service, conservation interventions tend to be less challenging and more cost-effective, especially when areas that provide that service are clustered.

Integrating ecosystem services into conservation planning can catalyze interest for developing sustainable approaches to assess the congruence and relevance of multiple ecosystem management goals (Balvanera et al., 2001; Egoh et al., 2007). The biodiversity approach, for example, considers species and habitats as providers of ecosystem goods and materials to society (Carpenter, 2005) and support for ecosystem functions that regulate the sustainability and viability of the services provided (Egoh et al., 2007).

Mapping of ecosystem services, which includes elements of biodiversity and ecological processes, can help identify providers (forest species) of ecosystem services (Kremen, 2005) by assessing and mapping their functional contribution (Balvanera et al., 2001) and can provide explicit arguments for inclusion in environmental assessments (Costanza et al., 1998; Kremen, 2005).

Carbon Sequestration, Timber Production and Water Yield from Forest Ecosystems

In the past century the global environment has been drastically impacted by human activity. For example, there has been an increase in the carbon dioxide concentrations in the atmosphere, inducing climate change (Friedlingstein, Dufresne, Cox, & Rayner, 2003) primarily due to the burning of fossil fuels, inappropriate forest management and other anthropogenic activities. Climate change, together with the severe reduction of biodiversity, has fueled the interest to manage natural ecosystems, while accounting for their multifunctionality (Schwenk, Donovan, Keeton, & Nunery, 2012). Along with wetland and grassland habitats, terrestrial forest ecosystems represent a bigger reservoir of carbon than the atmospheric stocks (Lal, 2004). Given the potential of forest ecosystem to sequester and store carbon dioxide in their living biomass, afforestation and reforestation (hereon jointly referred to as forestation) programs are one of the strategies used by instruments such as the Kyoto Protocol to mitigate global climate change (Jackson et al., 2005; Jindal, Swallow, & Kerr, 2008).

Land use patterns involving reforestation or afforestation have potential impact on other ecosystem services and functions such as stream flow, food security, and loss of biodiversity (Canadell & Raupach, 2008). In addition, carbon sequestration and storage in terrestrial forest biomass can be widely recognized when the social value is accounted for (Tallis et al., 2011). This importance and value of forest ecosystems in climate change mitigation and their corresponding social value due to avoided disturbance from non-emission(Nordhaus, 2007; Tol, 2009), make these ecosystem service, and therefore the carbon cycle, the focus of many scientific publications (Amthor, 1995; Anderson et al., 2010; Canadell & Raupach, 2008; Dixon et al., 1994; Falkowski et al., 2000; Friedlingstein et al., 2003).

The carbon stored in terrestrial forest ecosystems has different fates: it can be released into the atmosphere, sequestered or conserved in the soil (Brown et al., 1996). The amounts of carbon biomass present in woody plants are products of important ecosystems processes. Indeed, the carbon is produced by photosynthesis and consumed through ecosystem respiration dead material harvesting, etc (Brown, Schroeder and Kern 1999). One of the challenges for forest managers is to provide incentive to landowners for improvement of their forest land to retain more carbon (Subak, 2002). The biomass accumulation in aboveground parts of live trees and the biomass present in the underground coarse roots represents carbon offsets (Dwivedi, Alavalapati, Susaeta, & Stainback, 2009). Although its impact on climate change is positive as an offset strategy to reduce carbon dioxide in the atmosphere, forest management with carbon storage purposes is still under threat since the carbon stored in forest biomass may be released back to the atmosphere should a disturbance occur (Canadell and Raupach 2008).

Timber Production in the US Southeast Region

Biomass in the form of timber production is also an important ecosystem service because it can generate profit to landowners and communities that properly manage forest plantations (Tallis et al., 2011). The rate at which timber is harvested influences the sustainability of this service as well other services related to the functioning of forest ecosystems. Forest management is important both economically and ecologically to the US Southeast region, especially in North Florida (Hendry & Gholz, 1986). This activity contributes greatly to the supply of timber product (Dwivedi et al, 2009). Also, many studies recognized the role of pine plantations in the Southeast of United States for their contribution to carbon sequestration (Shan, Morris, & Hendrick, 2001). The forest industry contributes to Florida’s economy by providing more than 64 thousand jobs, which represents 13.0 billion dollars in the state’s gross domestic product (GDP) (Brown et al., 2012). In 2010, approximately 50% (nearly 17.4 million acres) of Florida was covered by forests. Trends in ownership and land use changes affect how forest lands are managed in Florida(Brown et al., 2012). The Nonindustrial Private Forests (NIPF) account for 65% (around 10.3 million acres) of timberlands (i.e., lands available for timber production) in Florida. However, public ownership is on the rise, from 28% in 2007 to 30% (4,751,700 acres) in 2010 (Brown et al., 2012).

Forest Ecosystem Structures and Water Yield

Although carbon and timber are receiving a lot of attention for their driving effects on climate change and the energy budget -and their market and profit opportunities-, respectively, water remains one of the most valuable resources for people on Earth. Via its cycle, water drives most of the biological interactions between all ecosystems on Earth (Chapin Iii & Matson, 2011). Because of this control on living biomass, the hydrologic cycle can also influence the carbon cycle and thus the energy budget (Chahine, 1992). Therefore, a strong interaction exists between the carbon and the hydrologic cycles that warrant a useful approach by studying them together. Hydrologic services provided by terrestrial ecosystems can be defined as the benefits people receive from the impact of these ecosystems on fresh water (Bennett et al., 2009; Brauman, Daily, Duarte, & Mooney, 2007).

Vegetation plays a role in both the carbon (GrÜNzweig, Lin, Rotenberg, Schwartz, & Yakir, 2003; Kirschbaum, 2003) and hydrological cycles (Farley, Jobbágy, & Jackson, 2005; Scheffer, Holmgren, Brovkin, & Claussen, 2005). Through photosynthesis, plants fix the chemical (Churkina, Running, Schloss, & Intercomparison, 1999) energy required by their tissues to sequester carbon in the terrestrial ecosystems (Chapin Iii & Matson, 2011). Just as plant distribution and composition influence the hydrologic cycle (Dunn & Mackay, 1995; Gerten, Schaphoff, Haberlandt, Lucht, & Sitch, 2004), , water controls the productivity and distribution (Stephenson, 1990) of plant communities in the terrestrial ecosystems. Within a given watershed, an increase in forest cover will reduce the surface water flow as a consequence of an increase in evapotranspiration, whereas a deforested catchment will increase runoff (Bosch & Hewlett, 1982; Brown, Podger, Davidson, Dowling, & Zhang, 2007; Brown, Zhang, McMahon, Western, & Vertessy, 2005). These mechanisms are driven mainly by the composition and pattern of leaf area (Neilson, 1995), through stomatal conductance and transpiration rates (Skiles & Hanson, 1994), and rooting behavior (Milly, 1997).

Evapotranspiration is one of the key ecosystem processes that relate carbon sequestration to water yield and subsequent water supply and climate regulation services. For example, the latent heat flux that drives the transfer of water to the atmosphere and measured in energy units is identical to evapotranspiration, measured in units of water (Chapin Iii & Matson, 2011). Some authors have attempted to model evapotranspiration (ET) using a dimension analysis method (describing a process using dimensionless group of variables) to estimate ET from forest ecosystems at multiple temporal scales. For example Running et al. (1989) mapped evapotranspiration in western Montana (USA), predominated by coniferous forests and a comprehensive assessment was made by Zhang et al. (2001), who estimated the mean evapotranspiration response following the conversion of grasslands to a forested cover.

Leaf are index (LAI) is defined as the ratio of leaf area to ground cover for broadleaf plant canopies or projected needle forests (Iiames, Congalton, Pilant, & Lewis, 2008; Myneni, Ramakrishna, Nemani, & Running, 1997; Running & Coughlan, 1988) and is one of the most widely used forest attributes for parameterization of forest ecosystem function models (Fassnacht, Gower, MacKenzie, Nordheim, & Lillesand, 1997; Grier & Running, 1977; Jensen & Binford, 2004; Myneni et al., 1997). In fact, it's the main forest structure characteristic that allows the exchange of CO_2 and water vapor, between the atmosphere and forest canopies (Fassnacht et al., 1997; Iiames et al., 2008). In addition, the relationship between leaf area index (LAI) and water availability has been widely investigated (Vose et al., 1994).

For example, a study conducted in Australia (1999), (Watson, Vertessy, & Grayson, 1999) found that: i) LAI is positively correlated with evapotranspiration, which in turn negatively affects water yield; ii) In a region where soil and water are not limited, the relationship between evapotranspiration and LAI is linear. Also, for most ecosystems it is widely recognized that biomass production is related to the photosynthetic surface area (Gholz, 1982). Generally an increase in net primary production (NPP) is associated with increase in LAI, which induces a higher water loss from interception and transpiration (Gerten et al., 2004). Additionally, LAI can be spatially analyzed, thus facilitating the identification of areas of interest to better study the interaction between carbon storage and water yield. Some studies have used LAI to map photosynthetic vegetation structures (Kucharik et al., 2000; Running et al., 1989). Therefore, LAI can be considered the common driver between water yield and the aboveground carbon stored in forest ecosystems and used to identify potential areas where trade-offs might be occurring.

The assessment of the patterns of LAI development at the stand level is often realized using a chronosequence approach, which is the estimation of LAI from stands, classified according to different age classes (Clark, Gholz, & Castro, 2004; Gholz & Clark, 2002; Vose et al., 1994). Also, some studies use models derived from field data from chronosequence approaches or from repeated observations on the same stand (McLaughlin, Kaplan, & Cohen, 2012). Models developed by Gholz and Clark (2002), supported by McLaughlin et al. (2012), from studies on slash pine, showed that LAI is highly correlated with stand age. Also, Gholz (1982), Grier and Running (1977), graphed LAI versus water balance, LAI versus biomass production, and NPP (Net Primary Production) versus water balance. Their studies found that LAI and biomass are positively correlated, LAI and water balance, as well as NPP and water balance, negatively correlated.

Spatial Analyses and Trade-offs

Additionally, understanding the interactions among bundles of ecosystem services represents a challenge in land management decision making (Bennett et al., 2009; Rodríguez, 2006). Bennett et al. (2009) provides three reasons that support the need to better understand these interactions among ecosystem services: 1) Focusing management of natural systems to produce one ecosystem service can substantially alter the provision of other important services (Bennett et al., 2009), Millennium Ecosystem Assessment 2005 and Diaz and Rosenberg 2008); 2) Management practices can reduce trade-offs between sets of ecosystem services by focusing on the natural processes that generate the services (Pretty et al., 2006); and 3) Lack of information of the processes behind the interactions between ecosystem services can lead to unexpected changes (Gordon, Peterson, & Bennett, 2008).

Ecosystem services are in interaction (Figure 1-1) when the level of provision of one service directly affects the level of another service (Bennett et al. 2009). The interaction can be unidirectional in the case where the supply of one service alters the provision of the other but there is no reciprocity. Interactions among services can also be bidirectional when the impact is reciprocal, i.e., the provision of service affects the provision of the other and vice-versa. Furthermore, interaction can be negative or positive, where positive interactions happen when the occurrence of one service increases the provision of the other Interactions and negative when the level of provision of one service decreases the level of supply of the others. Thus these “Trade-offs” occur when two ecosystem services, which interact among each other, the production of one compromises the provision of the other. An opposite response to the same driver can also create trade-off between two ecosystem services. Meanwhile, a “synergy” occurs when the production of the 2 ecosystem services simultaneously increase due to interaction between them or because how they respond to a particular driver (Bennett et al. 2009, Schwenk et al. 2012, Nelson et al. 2009).

An analysis of the trade-offs among key ecosystem services can provide useful information on how one service can affect the provision others (Brauman et al., 2007). Usually, spatial information at the landscape scale only focuses on the land use land cover (LULC) change, neglecting the biophysical function behind those issues (de Groot et al., 2010). To better understand the trade-offs and synergies among a set of ecosystem services, it is imperative to analyze the landscape drivers behind their relationship (Bennett et al., 2009). The study of relationship between ecosystem services and the values that they generate is important to address decision making that can help manage trade-offs in land use change (de Groot et al., 2010). To date, local and regional studies that account for these relationships are still limited (de Groot et al., 2010). Therefore, studies on the assessment of a land use/cover’s capacity to provide a bundle of ecosystem services and the different drivers that influence trade-off can provide useful information to ecosystem managers.

Furthermore, decision making can be more efficient if, besides relying on location of features and functions, it can also be based on the quantification of the bundle of provision of services (Troy & Wilson, 2006). Mapping ecosystem services can require analysis of extensive spatial data when focused on a regional scale, which may lead to the use of available models from literature that developed underlying ecosystem functions (de Groot et al., 2010).

Mapping of ecosystem service approach is used by many other authors in other parts of the world. (Timilsina et al., 2012) introduced a spatial framework to identify priority areas for carbon storage in the state of Florida. Specifically, his study consisted of mapping hotspots of carbon storage from a range of forest types and management schemes. Gimona and van der Horst (2007) investigated multifunctionality hotspots for three bundled ecosystem services in Scotland by using a weighting scheme approach to create benefit maps for the three services. Other approaches may use the mapping process as a step to identify important areas for biodiversity conservation, areas that provide important ecosystem services and therefore assess conflict and spatial congruence (Egoh et al., 2007; Polasky et al., 2008). Thus, spatially explicit analyses are crucial to understand the biophysical processes behind the production of ecosystem services ((Brown & Schroeder, 1999).

Understanding the multifunctionality of landscape systems can help identify areas of trade-offs and synergies, and therefore better manage conflicts (Chan, Shaw, Cameron, Underwood, & Daily, 2006; Egoh et al., 2008; Egoh, Reyers, Rouget, & Richardson, 2011; Groot et al., 2007). Also, the approach of mapping ecosystem functions and the relevance of this exercise remain a challenge (Willemen et al. 2008, de Groot et al. 2010). Therefore use of Geographic Information System (GIS) (Egoh, Reyers, Rouget, Bode, & Richardson, 2009; Gimona & van der Horst, 2007) coupled with available observational data (e. g. Forest Inventory data) can help develop models that identify local patterns or bundles of geographic phenomena and to better quantify ecosystem services for resources allocation purposes (Troy & Wilson, 2006).

Study Objective

Ecosystem services are the final, measurable outputs from ecosystem functions -in both managed and natural forests- that directly benefit humans. So, understanding the relationship between different sets of ecosystem services (Kareiva, Heather, H., C., & Stephen, 2011; Kremen, 2005) through spatial analysis is possible by combining available field data with models in a geographic information platform (de Groot et al., 2010; Guo et al., 2000). Therefore, the purpose of this study is to examine, spatially and statistically, carbon sequestration, timber production and water yield interactions in slash pine ecosystems in North Florida.

More specifically, since forest plantations use water to sequester carbon and store it in their biomass, we proposed to assess the trade-offs between carbon sequestration, timber production and water yield. In order to accomplish this we first used inventory data from the FIA, available functional models, and the literature for slash pine in Florida to quantify a bundle of ecosystem services: carbon sequestration, timber production, and water yield. In this study we used two models developed by McLaughlin et al. (2012), where: the first one predicts leaf area index (LAI) from stand basal area and the second one, stand level water use as a function of LAI.

With the ecosystem services quantified at the plot-level, a GIS-based classification algorithm was used to assess the interaction among the level of provision of 3 key ecosystem services during a defined study period (2002-2010): water yield, carbon sequestration, and timber volume growth. In this classification the ecosystem services were ranked in 3 classes, high medium and low, then an equation was used to estimate a three-digit code, which identifies where there were trade-offs or synergies among the three services. Furthermore, we determined the effect of biophysical drivers, such as stand age, and others such as silvicultural treatments and disturbance regime, on these ecosystem services and their interactions. Therefore, we propose the following questions and hypotheses:

- Question 1: Are specific areas of slash pine areas in north Florida reducing water yield?
- Hypothesis 1: Higher reduction in water yield can be identified by FIA plots with higher carbon sequestration rates and timber volumes.
- Question 2: How do forest structure (e. g., basal area, and stand age), management (e. g., site quality, silvicultural treatments) and other human and ecological factors (e.g., ownership and disturbance regimes) drive carbon sequestration, timber production and water yield interactions?
- Hypothesis 2: Drivers that positively affect carbon sequestration rates and/or volume of timber, will negatively affect water yield (Bennett et al., 2009; McLaughlin et al., 2012).

The results of this framework can be useful because they can help identify areas that maximize the provision of these three ecosystem services. Furthermore, the relevance of this framework is the application of its results in the development of management alternatives that maintain consistent levels of ecosystem service provision while not compromising certain management or conservation goals.

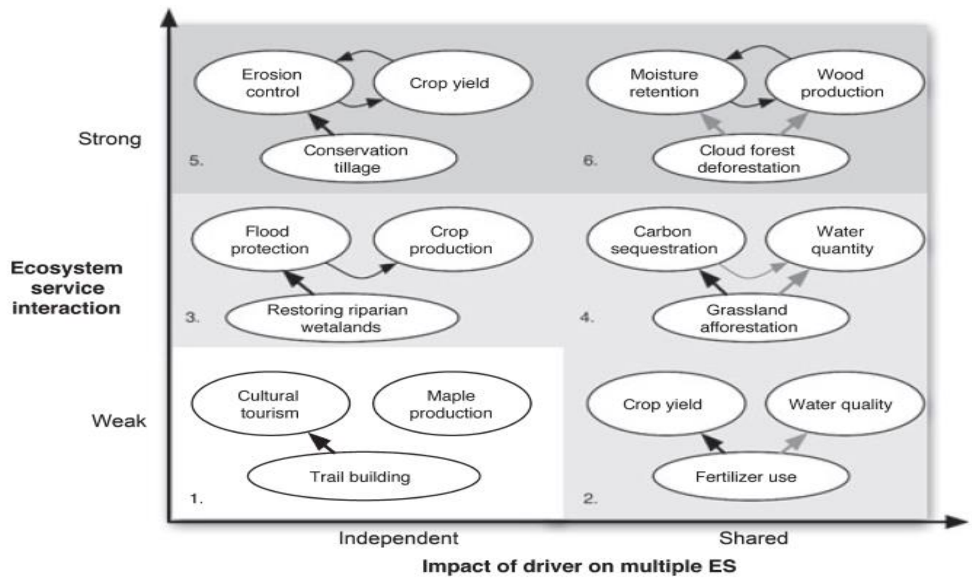

Figure 1-1. Examples of relationship between sets of ecosystem services, due to interactions between them or due to response to the same driver. The black arrows show positive effects and the grey negative effects [Adapted from Bennett, E. M., G. D. Peterson and L. J. Gordon (2009) Understanding relationships among multiple ecosystem services (Page 1397, Figure 2). Ecology Letters, 12, 1394-1404

CHAPTER 2
METHODOLOGY

Study Areas

The study area was the northern region of Florida because it is the most forested part of the state (Carter & Jokela, 2003). The study area encompasses two of the four Florida Forest Inventory and Analysis (FIA) survey units: the Northeast and Northwest. In 2007, these regions covered 6.6 and 5.5 million acres, respectively, which represent 76% of the total of timberland surveyed in Florida in the cycle 8 period (i.e., inventory carried out over the period 2002 to 2007). Northern Florida is dominated by pine flatwoods ecosystems (Timilsina et al., 2012) but other potential natural vegetation occurrences are: sand pine (Pinus clausa), in the deep-sand and xeric zones; bald cypress (Taxodium distichum L.), water tupelo (Nyssa aquatica L), and other hardwoods species, including laurel oak (Quercus laurifolia), live oak (Quercus geminata), sweetbay (Magnolia virginiana), sweetgum (Liquidambar styraciflua), spruce pine (Pinus glabra), red maple (Acer rubrum L.), etc., in the flood plains. Due to human impact over the past century, the dominant longleaf pine (Pinus palustris) ecosystems have been replaced by slash pine (Pinus elliottii), which represent around 69% of conifers in Florida.

Because of its capacity to grow fast and its adaptation to low fertility soils of US Coastal plain (Gholz et al., 1991), Slash pine (Pinus elliottii) is one of the dominant planted species in that region (Shan et al., 2001). One of the advantages of Slash pine plantations is that management and subsequent ecosystem models are easier to be developed due to the homogenous structure with relatively simple understory species composition (Cropper Jr & Gholz, 1993). However, leaf area index (LAI), which regulates water availability, varies greatly across slash pine plantation in North Florida (Gholz & Clark, 2002).

Slash pine forests are found mostly on lower elevations, where the natural habitats are formed over Quartzipsamments, nutrient-poor and mostly poorly drained soils, which contain deep loamy or clayey particles at the subsurface (Hendry & Gholz, 1986). The climate is humid subtropical with a mean annual temperature of 19 °C, and the average precipitation varies from 1000 to 1500 mm annually (McNab et al., 2005). The surface water system is characterized by fresh water springs and natural lakes which are found on limestone rocks formations, and the presence of some major rivers (McNab et al., 2005). A common disturbance regime is fire that occurs in moderate or low intensity in sand pine and longleaf pine areas (McNab et al., 2005).

Timberland class represents 92% of the total forest cover in Florida, in 2010. The yellow pine forest group (i.e., slash pine, loblolly pine, longleaf pine); which occupies 46% of timberland in Florida, is the dominant forest type group. Slash pine is the most dominant yellow pine type group and also the dominant forest type, when considered individually. It represents 30% of the total timberland in the state (4.8 million acres). In Florida, The Northeastern and Northwestern FIA survey units contain 76% of all merchantable volume (14.7 billion cubic feet). Although yellow pine plantation in the Northwest unit has doubled from 1995 to 2007, the Northeast still accommodates the majority of planted yellow pine (Brown et al., 2012).

Estimating Ecosystem Services

Forest Inventory and Analysis (FIA) Program

The Forest Service Inventory and Analysis (FIA) of the U.S. Department of Agriculture (USDA) is a program dictated by the McSweeney-McNary Forest Research

Act 1928 and the 1974 Planning Act Forest and Rangeland Renewable Resources, whose mission is to monitor the conditions of forestlands nationwide (Jenkins, Chojnacky, Heath, & Birdsey, 2003; Woudenberg et al., 2010). Since 1998, the periodic inventory data has changed to an annual inventory.

The FIA dataset provides plot-level carbon data for all forest ecosystems and timber volume production for timberlands, and is collected periodically statewide (Woudenberg et al., 2010). These data are largely used in ecosystem risk and management assessments, such as forest health and timber production inventories for economic purposes (Coulston, Reams, McRoberts, & Smith, 2006). The FIA uses a sampling method divided in 3 phases. Phase 1 consists of a stratification of the land at the state level using remote sensing techniques (Woudenberg et al., 2010). In phase 2, data are sampled on new or previously measured plots of 1 acre. More specifically, the actual plot consists of 4 macroplots of 1/4 acre. Each macroplot contains a subplot of 24 feet radius where trees of 5 inches diameter or more are sampled. Each subplot contains a microplot, on which smaller trees (diameter less than 5 inches) are measured (Figure 2-1).The FIA inventories follow the national standard fixed-radius sampling procedure (Woudenberg et al., 2010). In addition to the publicly available inventory database, in May 2002 the FIA program created the Spatial Data Services group (FIA SDS) to provide data for spatial analysis, given the privacy policy enacted by the Congress (Timilsina et al., 2012; Woudenberg et al., 2010).

The FIA data was used in this study to estimate the provision of the ecosystem services during the period 2002-2011, (i.e., carbon sequestration and timber volume). Also, although the water yield estimated in this study is based on models from the literature reviewed; some parameters were from the FIA database (e. g., basal area, stand age). This dataset was used mainly because of its public availability. Moreover, since the focus of this research is on net biomass change over time (i.e., net carbon sequestration), the fact that FIA remeasurements were matched at the tree level (Woudenberg et al., 2010), is a useful contribution of this study and for future research.

Carbon Sequestration Estimation

The carbon estimates in this analysis are based on data from the FIA inventories carried out during cycle 8 and 9. The cycle 8 inventory includes data from inventory 2002 to 2007, except 2005, while the cycle 9 contains inventories from inventory year 2009 to 2011. Often, the market for carbon credits focuses on changes in carbon storage (Hoover, Birdsey, Heath, & Stout, 2000), therefore, our objective was net carbon sequestration over a time period. We used slash pine stands in the Northeastern and Northwestern survey units and plot identification number to match plots sampled in cycle 8 to the ones remeasured in cycle 9. The result was 377 study plots, which are well distributed over the entire study area (Figure 2-2).

We downloaded the FIA plot-level dataset, available in shape file format, from the Florida Geographic Data Library (http://www.fgdl.org). Additionally, since the plot dataset does not include carbon estimates at the tree-level, we obtained tree-level data from the FIA websites: http://www.fia.fs.fed.us/tools-data/. The plot-level shapefiles provide data on the condition of the plot and information such as ownership class, stand age, site productivity class, disturbance, silvicultural treatment regimes, measured basal area, and understory and aboveground carbon. The tree-level dataset provides information on each individual tree sampled and includes the status condition (whether the tree is live, felled, or dead), estimated carbon values measured from the dry biomass portions of the tree (stump, bole, sapling, roots, etc.) for both aboveground and underground components, and estimated timber volumes according to the market standard (Woudenberg et al., 2010).

Furthermore, with the tree-level dataset, we use Microsoft Excel's filter tool to select plot-level standing live and dead trees based on if the status condition class (STATUSCD) was equal to 1 (i.e., live trees) or 2 (standing dead). Moreover, using the FIA column name (CN), which is a unique number to identify a sampled tree, we matched the 2002-2007 and the 2009-2011 data.

Since the carbon values provided by the FIA database at the tree-level are in pounds of carbon per tree, these per-tree values were converted to megagrams or tons of carbon per hectare (Mg C/ha) using the conversion factor TPA_UNADJ (i.e., trees per acre unadjusted), which represent a theoretical number of a specific tree in 1 acre (0.404686 ha) given the plot size (Table 2-1). These values were: i) 0.999 for trees in macroplots, ii) 6.01 for trees in subplots and iii) 74.96 for trees in microplots (Woudenberg et al., 2010). Using the pivot table tool in Microsoft Excel, we summed the values of all trees per plot. In addition, this tree table was matched to the plot table using the plot column number (PLT_CN), a unique number that link a tree record to the plot record. The carbon storage value for each plot is the sum of the aboveground carbon in trees and was derived from the following equation (1):

$$y = {}^{1}/_{2}\,(x_1 + x_2 + x_3 + x_4 + x_5) \tag{2-1}$$

where x_1, is the dry biomass in the merchantable bole; x_2, the dry biomass in the tree stump; x_3, the dry biomass in the top of the tree; x_4, dry biomass of saplings and x_5, the Dry biomass of woodland tree species .

To determine the net annual carbon sequestration between measurement cycles, we used the following equation (2):

$$CSQTNET = (CSTG1 - CSTG2)/REMPER \tag{2-2}$$

where, CSQTNET was the net annual Carbon sequestered during the time period, in megagrams of carbon per hectare per year (MgC/ha/yr). This value can be a negative number, since this a net change in carbon storage. Both carbon storage (CSTG1) in the first cycle and carbon storage (CSTG2) in the second cycle were in megagram of carbon per hectare. Finally, REMPER was the remeasurement period, the number of years between measurements for remeasured plots.

Timber Volume Estimation

We used the same tree-level procedure as for carbon sequestration estimation to calculate the timber volume during the period of study. Since generally commercial timber production is a valuable commodity (Tallis et al., 2011), we used timber volume estimates in cycle 9 as the ecosystem service of interest. The attribute from the FIA database considered was the VOLCFSND, which is defined as sound cubic-foot volume in the merchantable stem of trees. This measurement was recorded only for tree diameters at Breast Height (i.e.DBH; 1.4m) greater than 5 inches (Woudenberg et al., 2010). We converted tree-level values (cubic feet per tree) into cubic meter per hectare, using the conversion factor TPA_UNADJ. Using the pivot table tool in Microsoft Excel, we summed the values of all trees per plot. This tree table was matched to the plot table using the PLT_CN, as we did previously for carbon sequestration estimates.

Water Yield Estimation

Since (LAI) is identified as key driver for the ecosystem services being studied as explained above, and because LAI measurements are not reported in the FIA database, our estimations of water yield were based on modeled relationships between (LAI) and specific stand structure attributes. Several forest structures influence the amount of leaves available that directly affect forest productivity and water yield. Gonzalez-Benecke et al. (2012) developed an integrated model where LAI is predicted from stand index, tree density and basal area. This model captures many factors that could influence LAI at the stand level and also uses data from 15 year- time period and was measured frequently (Gonzalez-Benecke, Jokela, & Martin, 2012). However, McLaughlin et al. (2012) developed a non-linear model for slash pine stands in Florida, where mean annual leaf area index is predicted from stand basal area (Figure 2-3); thus, we estimated LAI, using McLaughlin et al.'s (2012) model.

Moreover, actual evapotranspiration was estimated using forest structure data for Slash pine ecosystems in Florida based on McLaughlin et al. (2012), who developed models that relate ecosystem water use and forest stand structure. Their results revealed that LAI is the best predictor of water use, which therefore can be used to predict water yield. In the studies reviewed by McLaughlin et al. (2012), different methods were used to measure evapotranspiration (ET) (Table 1), as ET was estimated using either Eddy Covariance (EC) measurements (Bracho et al., 2008; Gholz & Clark, 2002; Knowles, 1996), model simulation (Ewel & Gholz, 1991) or Eddy Covariance (EC) measurements combined with model simulation (Liu, Riekerk, & Gholz, 1998).

We used the relationship between leaf are index (LAI) and the ratio of evapotranspiration to precipitation (ET/PPT), developed in the same study by McLaughlin et al. (2012) to calculate the annual water yielded by the FIA plots in cycle 9

(2009-2011). First of all, evapotranspiration-precipitation (ET-PPT) ratio was obtained from the linear regression model (Figure 2-4), using equation 2-3:

$$ET/PPT = 0.06 * LAI + 0.54 \tag{2-3}$$

where ET/PPT is the ratio of evapotranspiration to precipitation and LAI, leaf area index in m^2m^{-2}.

Water yield for each plot was computed using the following equation:

$$WY = (1 - ET/PPT) * MAP * 10000 \tag{2-4}$$

where WY is water yield in cubic meter per hectare, on a year basis (m3/ha/year) per year; ET/PPT is the ratio of evapotranspiration to precipitation, which was previously predicted from LAI (dimensionless); MAP is mean annual precipitation in meter (m), from a 10-year period (2001-2011), which includes our study time period (2002-2011). These precipitation data were obtained from a grid downloaded from the PRISM Climate Group at Oregon State University website and are presented in Table 2-1 (http://www.prism.oregonstate.edu/products/matrix.phtml). The precipitation values were assigned to each plot using ArcGIS 10’s “Raster to Point” tool.

The estimates in this study, however do not consider the resulting water yield that could be observed at the ground level or surface water runoff (McLaughlin et al., 2012). Also the water yield estimates do not account for any changes (e.g., seasonal climate fluctuations) that could have occurred during the study period considered.

Trade-off Analysis

As hypothesized, trade-offs are expected between growth in biomass (i.e., carbon sequestration and timber volume growth) and water yield (Farley et al., 2005; Jackson et al., 2005), therefore we used our estimated ecosystem service values to develop two simplified regression models with Excel to explore relationships between water yield and net carbon sequestration and/or timber volume. As displayed in figures 2-5 and 2-6, the graphs are showing a regression line that could be used to define a production frontier for assessing trade-offs between net carbon sequestration and water yield or between timber volume and water yield. However, as revealed by small adjusted R^2 (0.14 and 0.29), there is a weak relationship between dependent and independent variables, suggesting there is a variability across sites. Also, there is no indication of the relative three-way interaction among the level of provision within the ecosystem service bundle. Therefore another method was used to analyze the interactions between the ecosystem services.

The ranking method is a common method used to rank the level of production of several ecosystem service goals (timber, carbon or water) and to prioritize areas where a particular goal is being achieved at the highest level when compared to others (Carr & Zwick, 2007). Thus, this framework essentially can be used to classify the level of provision of a finite set of ecosystem services. Therefore, we used this ranking method as the conceptual basis for a spatial classification framework for ecosystem service tradeoffs. The framework was derived based on the Land-Use Conflict Identification Strategy (LUCIS) model developed by Carr and Zwick (2007). Their objective was to classify lands based on suitability analysis in order to determine preferences or appropriateness for agriculture, conservation or urban uses. In contrast to the LUCIS model which used raster analysis; our approach was based on vector (point feature) analysis. The reason was because our variables represent discreet phenomena and therefore cannot be interpolated in space (Mitchell, 2009).

Since, the data for the 3 ecosystem services present different ranges in values (Table 2-2) we normalized the values using a scale from 0 to 1 (Figure 2-7) by dividing all the values by the maximum values for each ecosystem service. This same approach was used by Carr and Zwick (2007) to determine land use preferences and conflicts between agriculture, conservation and urban areas. In ArcGIS Version 10, we classified the normalized values using the Natural Break (Jenks) classification method, as opposed to Manual, Equal Interval and Standard Deviation classification methods, because the data values of the three services are not normally distributed, as suggested by Carr and Zwick (2007). Specifically, the natural breaks or Jenks-natural-breaks is a data classification method based on Jenks optimization procedure (Mennis & Liu, 2005). The algorithm used by this method arranges the values into different classes, relying on an iterative process where different breaks in the dataset are used to minimize the variance within classes and maximize the variance between classes, as much as possible (Brewer & Pickle, 2003).

This method produced 3 classes for each service, which were coded respectively 1, 2 and 3. The codes 3, 2 and 1 represent high, medium and low, respectively. Since we were working with net estimates of the variables, the negative values were classified as 1 (low). Moreover, we determined interaction (i.e., synergy of tradeoff) codes by combining the level of the individual ecosystem service in an “ecosystem service bundle” using the following formula:

$$IC = CSL * 100 + TVL * 10 + WYL \tag{2-5}$$

where IC is the three digit code defining the type of interaction (a priori defined); CSL, the carbon sequestration level; TVL, the timber volume level and WYL, the water yield level.

The output codes for the interactions were a series of numbers between 111 and 333 (Figure 2-8), which were further classified into plots where one of the ecosystem service is dominant over the others (i.e., trade-off) and plots where at least two of the ecosystem services are dominant (i.e., synergy) (Figure 3-4). Furthermore, in ArcGIS, we created maps which display the spatial distribution of the FIA plots where trade-offs or synergies, as classified using our interaction classification framework, occur between the 3 ecosystem services (i.e. bundle) and areas where water yield is dominant compared to areas where carbon or timber is dominant. Table 3-2 presents the calculated values of the three services used in the classification framework.

For the purpose of this framework, we defined a synergy as the situation when two of the ecosystem services being analyzed in a bundle, are produced at the same or higher (high-high, medium-medium) level than the third one (Bennett et al., 2009; Raudsepp-Hearne, Peterson, & Bennett, 2010). More explicitly, when two of the services are generated at: 1) the same level 3 (high) while the other is at the level 2 (medium) or 2) when two of the services are generated at the same level 2 (medium) while the other at the level 1 (low), or 3) when the three services are yielded at the same level, either 1 (low), 2 (medium) or 3 (high). On the other hand, we defined a trade-off as the situation when one of the ecosystem services is dominant compared to the other one (Bennett et al., 2009; Raudsepp-Hearne et al., 2010). In this case a trade-off would be when one of the services is generated at the level 3 and the other two at level 1 or 2, or when one of services is produced at the level 2 and the other ones at the level 1.

Statistical Analysis of Drivers

We also tested the effect of different drivers on the provision of each individual ecosystem service as well as the effect of the same drivers on the resulting interactions in the ecosystem service bundle, based on the codes generated from the classification framework. Among the drivers tested, was stand age (McLaughlin et al., 2012; Timilsina et al., 2012). This variable is to be interpreted carefully because of inconsistencies in the method used for its estimation by the FIA. Although the time period considered in this study is relatively short (7 years), considering the ecosystem services being analyzed, the stand age used from the second inventory period (2009-2011) ranged from 2 to 139 years with 75% being lower than 49 years, which provides a good range to test temporal variation of the ecosystem processes.

Other variables used were ownership, site productivity class, disturbance regime and silvicultural treatments (Woudenberg et al., 2010). Ownership (land tenure) was reported as public (e.g. state, federal, national park service, national forest system, etc..), and private but were reclassified into 2 dummy variables, 0 for public ownership and 1 for private ownership. Variations among the public land tenure classes (e.g., Corporate and non-governmental conservation organizations) were not accounted for. In addition, we assumed that the plots remain under the same land tenure status over the analysis period. The data analyzed consisted of approximately 72% of the plots managed under private ownership and 28, under public land.

For the site productivity class, the slash pine forest stands studied were growing on sites quality classified as 1, 2, 3, 4, 5 and 6, with productivity ranging >15.8, 15.7 and

11.6, 8.4 and 11.5, 6.0 and 8.3, 3.5 and 5.9, and 1.4 and 3.4 cubic meter per hectare per year (m3/ha/year), respectively (Woudenberg et al., 2010). As far as disturbance regime, we considered disturbances reported from the second year (2009-2010), that occurred from 2003 to 2011. The dataset reported disturbance regimes that have been caused by disease on sampling or seedling trees, fire that is either prescribed or natural at the crown or ground level, and livestock or animal grazing or any anthropogenic damage. In the statistical analysis performed, we used two classes: 0 (undisturbed) when there is no presence of disturbance and 1 (disturbed) for plots damaged by any of the reported factors. Approximately 89% of the plots being analyzed showed no presence of disturbance.

Finally, we tested the influence of silvicultural treatments on the provision of the three ecosystem services (Jerome K, 2009; Shan et al., 2001). Plots were treated using several stand improvement activities such as cutting, the use of fertilizers, herbicides or other activities with the objective of enhancing the commercial value of a stand (Woudenberg et al., 2010). About 78% of the plots were not treated, with only 22% receiving silvicultural treatments at least once during the time period between the inventory cycles.

All the statistical analyses were described in this study were analyzed in the Statistical Applications Software's JMP Pro 10 package. We used two different types of statistical tests to analyze the effect of the drivers on the interaction among net carbon sequestration, timber volume and water yield. First, we used a multiple regression analysis, with a backward selection expressed by the following equation:

$$y = (a + b_1x_1 + b_2x_2 + b_3x_3 + bx_4 + b_5x_5 + b_6x_6 \qquad (2\text{-}6)$$

where y represents the value of the dependent variable that is being explained, in this case net carbon sequestration, timber volume or water yield. The constant “a” is intercept of the graph described by Equation 4. And x1, x2, x3, x4, x5 and x6, are the independent variables, represent by stand age, silvicultural treatment, ownership, site productivity and disturbance regime, respectively. The coefficients b1, b2, b3, b4, b5 and b6, represent the change in Y that correspond to a unit change in stand age, basal area, silvicultural treatment, ownership, site productivity and disturbance regime, respectively.

Homoscedasticity of the residuals was examined using scatter plots and the visual relationship between residual-predicted values. Normality was examined with the normal quantile plot and the goodness of fit using the Shapiro-Wilks test. When assumptions were not met, data were log transformed for the three variables. A P value of <0.05 and parameter estimates were used to interpret the statistical significance of a particular driver for net carbon sequestration, net volume growth and water yield.

Second, we used a multiple logistic regression to test the effect of these same drivers (e.g. stand age, basal area, silvicultural treatments, ownership, site quality and disturbance) on the interactions among net carbon sequestration and water yield as well as timber volume and water yield. Two dummy variables (i.e., 0 and 1) were analyzed using a similar approach as a study of forest carbon hotspots in Florida, by Timilsina et al. (2012). Specifically, we analyzed the interaction codes as 2 dummy variables; trade-off plots were “0” and the synergistic plots were “1”. The results of the multiple logistic regression were interpreted using chi square of the Likelihood Ratio tests, parameter estimates and/or odds ratios, and all were tested for significance using $\alpha = 0.05$.

Table 2-1. Description of the forest inventory and analysis (FIA) attributes and other input data used in the estimation of the ecosystem services

Input data	Description	Range of values	Units	Source
Leaf Are Index (LAI)	Ratio of leaf area to ground cover for broadleaf plant canopies or projected for needle forests	0.16 – 7.12	m2/m2	(McLaughlin et al., 2012)
Evapotranspiration-precipitation ratio (ET/PPT)	Ratio of evapotranspiration and precipitation (PPT), which indicates the portion of precipitation that has been evaporated and/or lost through transpiration.	0.549 – 0.967	-	(McLaughlin et al., 2012)
Mean annual precipitation (MAP)	Average annual precipitation for the period 2001-2011	1.105 – 1.638	m	http://www.prism.oregonstate.edu/
Carbon Tree aboveground	Carbon value in the aboveground portion of the tree, except foliage, recorded for live trees with a diameter greater than 1 inch.	Period1: 0.00 – 122.700 Period2: 0.00 – 99.183	Mg/ha	FIA Manual
Inventory year	Year in which the data were recorded.	Period1: 2002- 2007 Period2: 2009- 2011	Year	FIA Manual
Owner class code	A code indicating the status ownership of the plot during the inventory.	11 National Forest System 12 National Grassland 13 Other Forest Service 21 National Park Service 22 Bureau of Land Management 23 Fish and Wildlife Service 24 Department of Defense/Energy 25 Other federal 31 State 32 Local (County, Municipal, etc.) 33 Other non-federal public 46 Private	-	FIA Manual

Table 2-1 Continued

Input data	Description	Range of values	Units	Source
Stand age	The estimated age of the plot using field records and or/ local procedures.	Period2: 2 to 139 years	Year	FIA Manual
Site productivity class code	A code indicating the potential growth in cubic meter per hectare per year, which is related to the capacity of a forest land to grow biomass.	(1) 15.8+; (2) 11.6 -15.7; (3) 8.4 -11.5; (4) 6.0 -8.3; (5) 3.5 -5.9; (6) 1.4 -3.4; (7) 0.0 -1.3	m3/ha/ year	FIA Manual
Disturbance code	A code indicating the type of disturbance that was happened since the last inventory of a plot or within the last 5 years, in case of a new plot. To be recorded, the disturbance must have affected at least 1 acre and have caused damages to 25% of the trees in the plot.	0 No visible disturbance 10 Insect Damage 20 Disease Damage 30 Fire damage (from crown and ground fire, either prescribed or natural) 40 Animal Damage 41 Beaver 42 Porcupine 43 Deer/ungulate 44 Bear 45 Rabbit 46 Domestic animal/livestock 50 Weather Damage 51 Ice 52 Wind (includes hurricane, tornado) 53 Flooding (weather induced) 54 Drought 60 Vegetation (suppression, competition, vines) 70 Unknown / not sure / other 80 Human-caused damage 90 Geologic disturbances	-	FIA Manual

Table 2-1 Continued

Input data	Description	Range of values	Units	Source
Disturbance year	The estimated year in which a disturbance type have happened.	From 2003 to 2010	-	FIA Manual
Stand treatment code	A code indicating the type of stand treatment that was applied since the last inventory of a plot or within the last 5 years, in case of a new plot. To be recorded, the treatment type must have affected at least 1 acre.	00 No observable treatment. 10 Cutting 20 Site preparation – Any practices realized with the intention of preparing a site natural or artificial regeneration 30 Artificial regeneration – Following a disturbance or treatment (usually cutting), a new stand where at least 50% of the live trees present resulted from planting or direct seeding. 40 Natural regeneration – Following a disturbance or treatment (usually cutting), a new stand where at least 50% of the live trees present (of any size) were established through the growth of existing trees and/or natural seeding or sprouting. 50 Other silvicultural treatment – The use of fertilizers, herbicides, girdling, pruning, or other activities (not covered by codes 10-40) designed to improve the commercial value of the residual stand.	-	FIA Manual
Basal area of live trees	Stand basal area of live trees which DBH or DRC is greater than 1 inch.	Basal area	m2/ha	FIA Manual
Remeasurement period	Number of years between inventories for remeasured plots	1.9-8.9	year	FIA Manual

Table 2-2. Descriptive statistics of the quantitative variables and drivers used in the analysis

Variable	Minimum	Maximum	Mean	Std. Deviation
Net Carbon Sequestration (Mg/ha/yr)	-11.716	9.155	0.573	2.653
Timber Volume (m3/yr)	0.000	329.08	55.575	59.571
Water Yield (m^3/ha/yr)	461.113	6298.73	2223.479	1230.033
Stand age (yeas)	2	138	33.873	24.044

N=377 for all variables

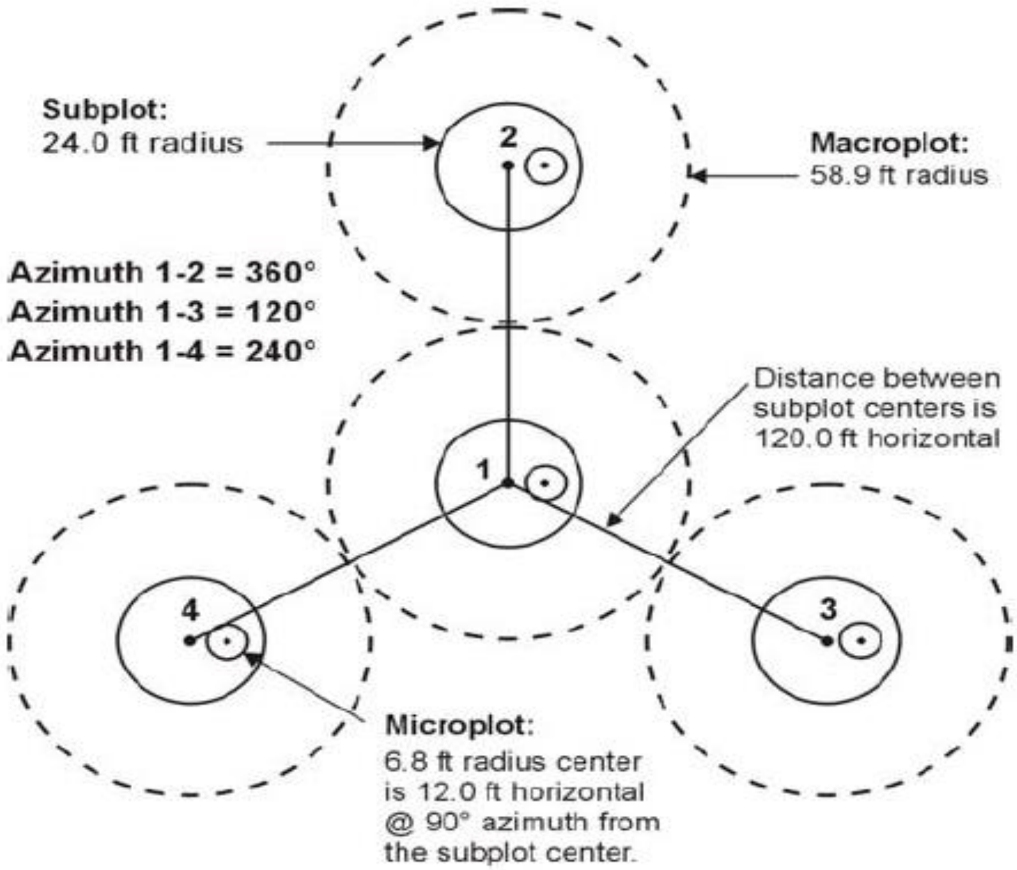

Figure 2-1. Forest Inventory Analysis (FIA) plot design [Source:Woudenberg, S. W., B. L. Conkling, B. M. O'Connell, E. B. LaPoint, J. A. Turner and K. L. Waddel. 2010. The Forest Inventory and Analysis Database: Database Description and User's Manual Version 4.0 for Phase 2 (Page 8, Figure2). ed. F. S. U.S. Department of Agriculture, Rocky Mountain Research Station. Fort Collins, CO.

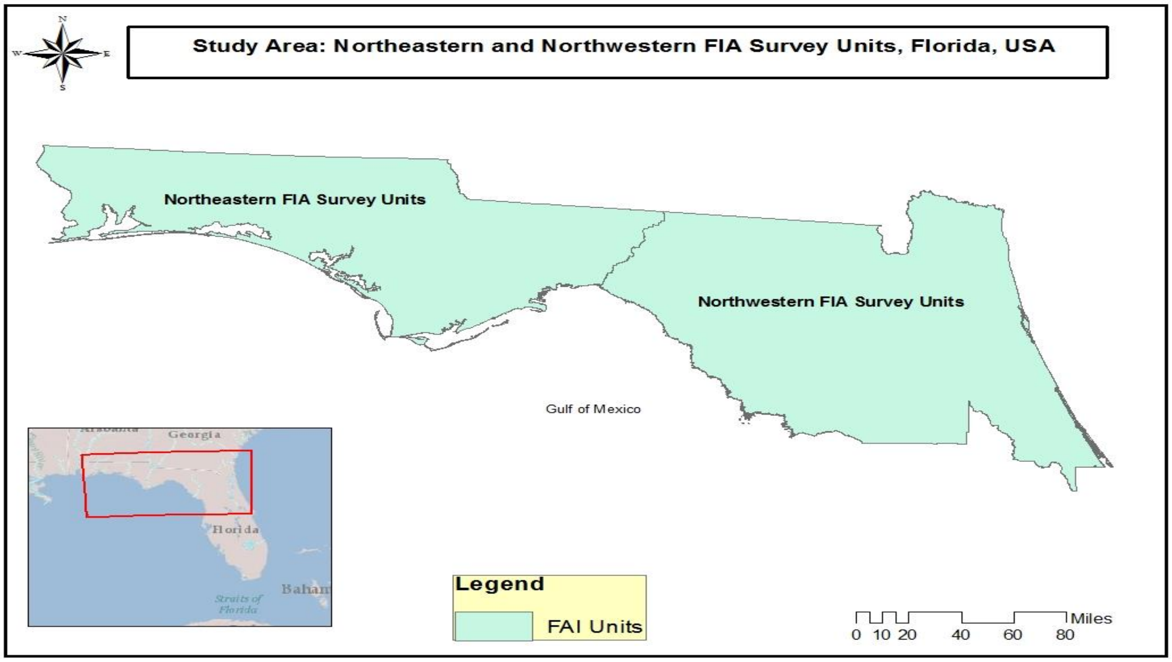

Figure 2-2. Study Area: northeastern and northwestern forest inventory analysis (FIA) survey units, Florida, USA

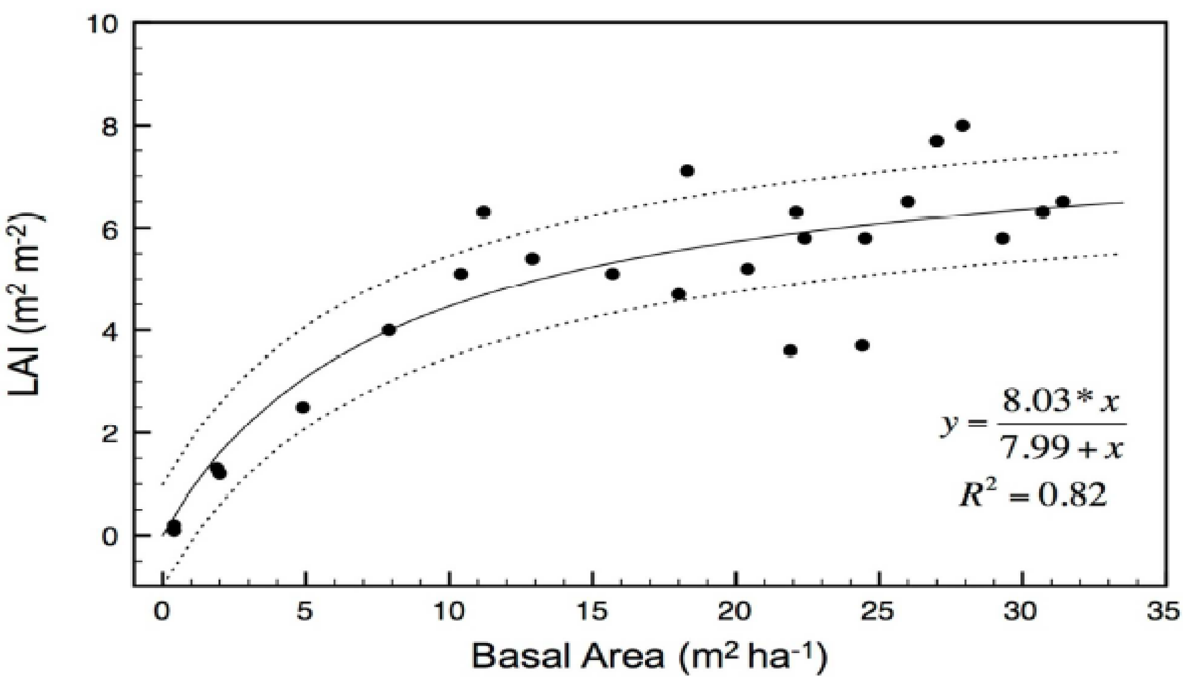

Figure 2-3. Prediction of mean annual leaf area index (LAI) from stand basal area for Florida slash pine stands. [Borrowed from McLaughlin, D., Kaplan, D., and Cohen, M. 2012. Managing Forests for Increased Regional Water Yield.. (Page 34, Figure 3). Journal of the American Water Resources Association

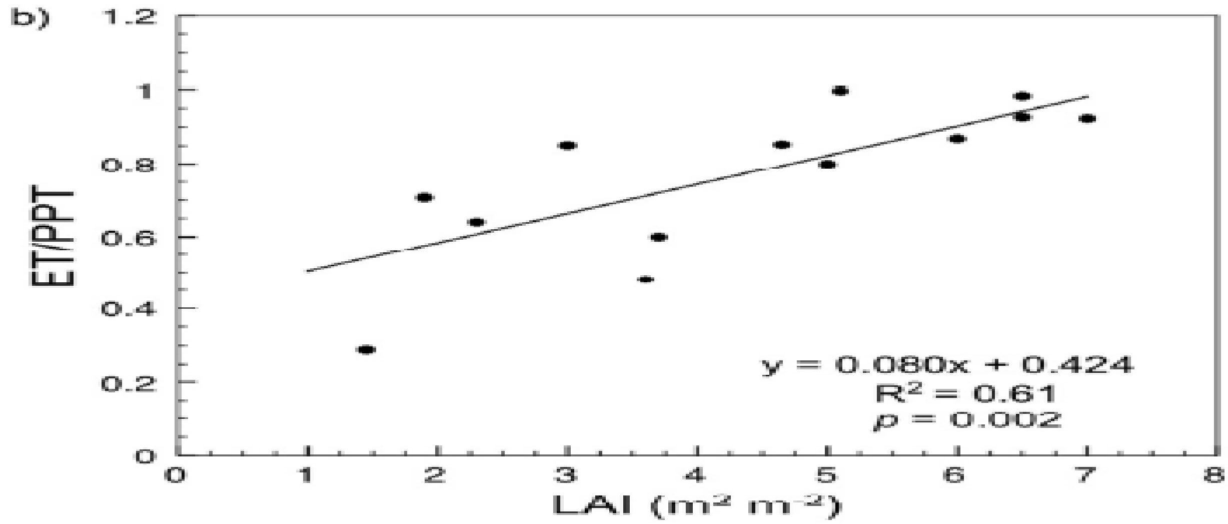

Figure 2-4. Relationship between evapotranspiration-transpiration ratio and leaf area index (LAI) for slash pine stands in the southeastern coastal region. [Borrowed from McLaughlin, D., D. Kaplan and M. Cohen (2012) Managing Forests for Increased Regional Water Yield (Page 24, Figure 2b). Journal of the american water resources association.

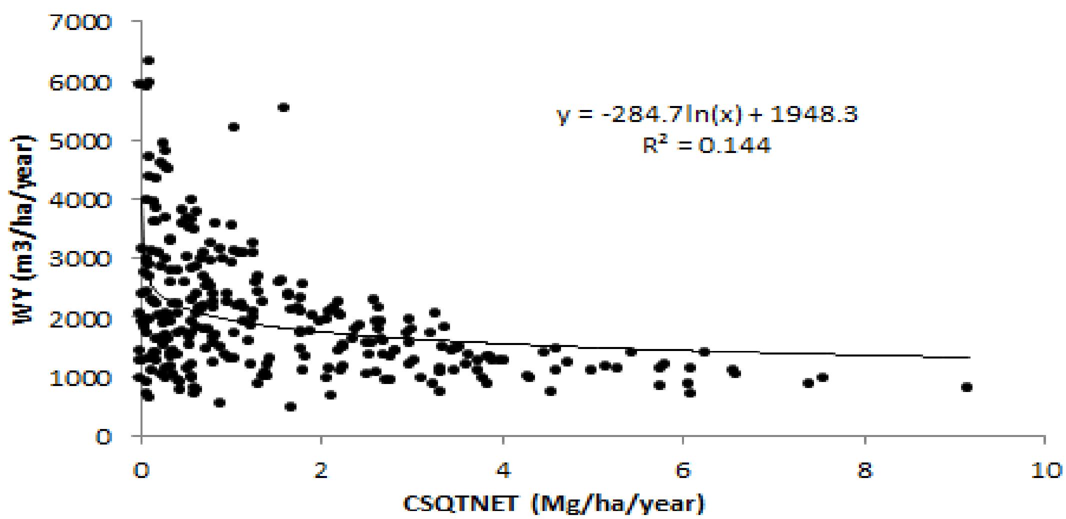

Figure 2-5. Water yield as function of net carbon sequestration for forest inventory analysis (FIA) slash pine plots, in Florida northeastern and northwestern survey units, 2002-2011

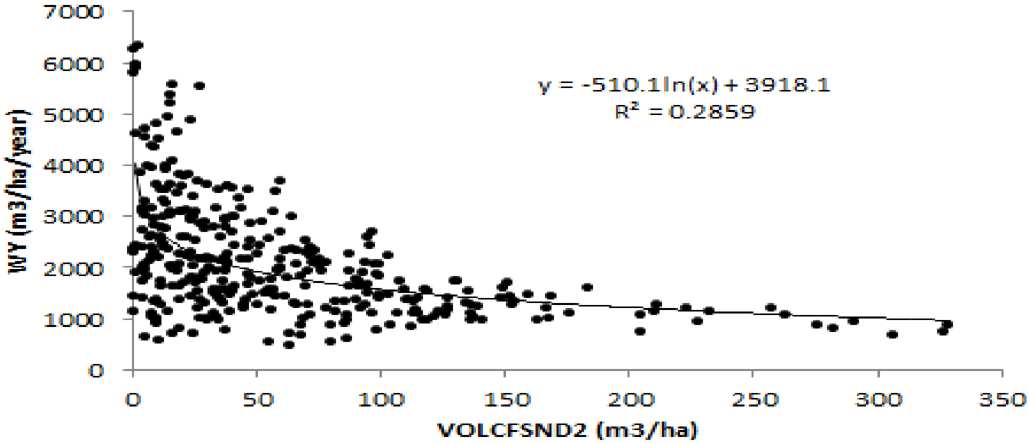

Figure 2-6. Water yield as function of timber volume for forest inventory analysis (FIA) slash pine plots, in Florida northeastern and northwestern survey units, 2002-2011

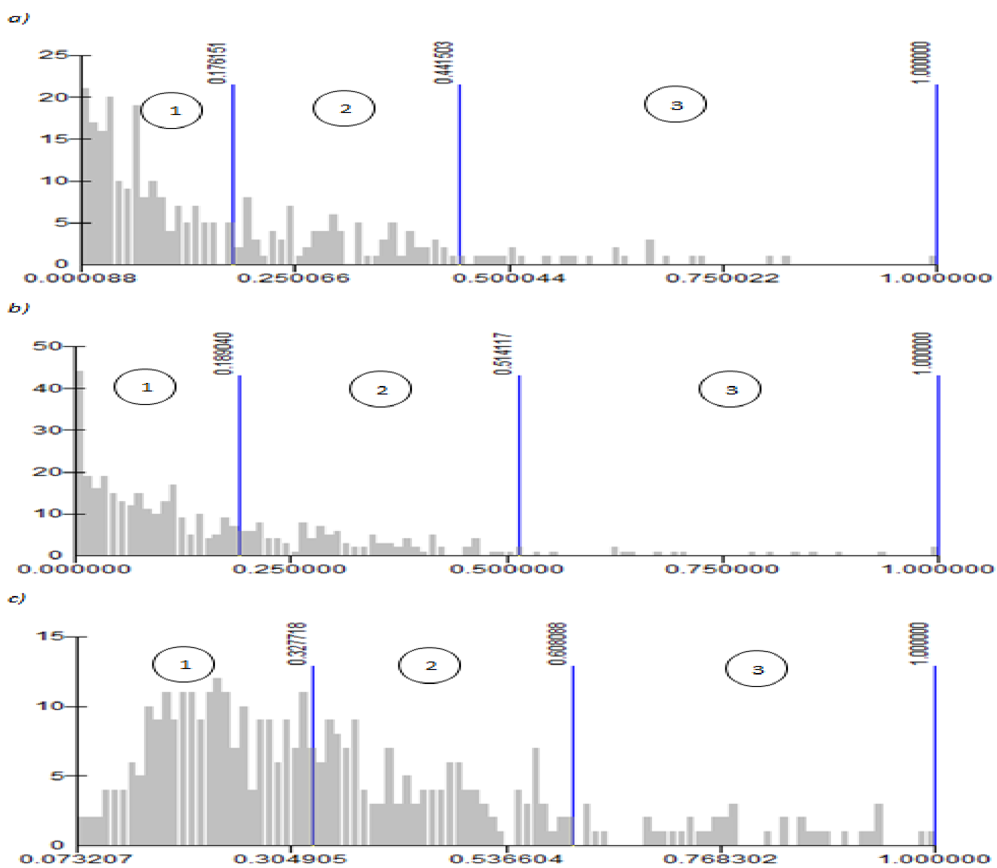

Figure 2-7. Distribution of the normalized values on a 0-1 scale for the variables (a-net carbon sequestration, b- timber volume, c- water yield) using a 3-class classification scheme generated by the natural breaks algorithm from ArcGIS 10. Note, 1 = low, 2= medium and 3 = high provision levels.

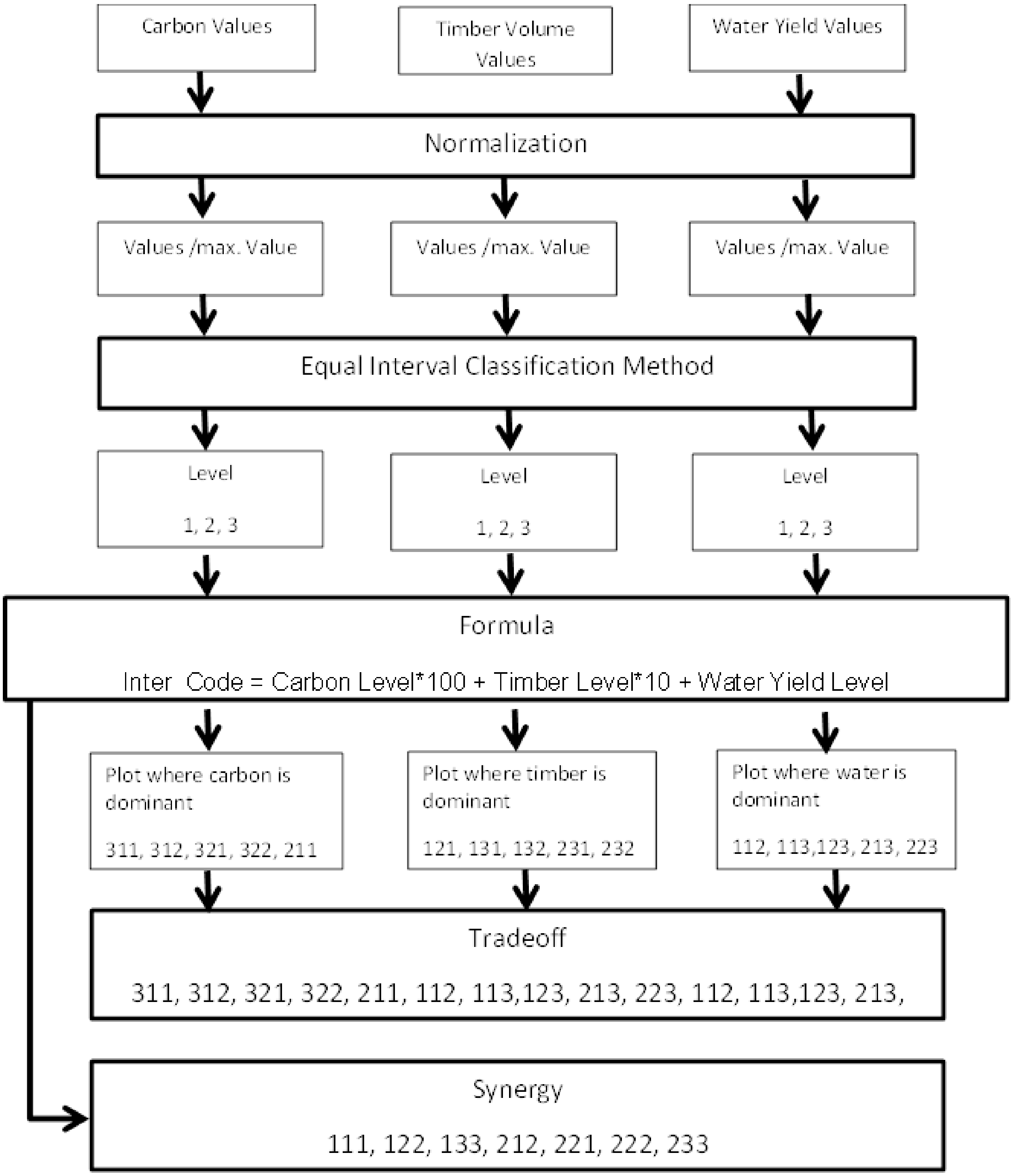

Figure 2-8. Diagram of the interaction classification framework

CHAPTER 3
RESULTS

Ecosystem Services Estimation and Interaction

According to the descriptive statistic for the variables reported in Table 3-1, during the period 2002-2010, the classification framework developed in this study estimates that around 72% of the plots provide low (mean= -0.511Mg C/ha/year), 22% medium (mean=2.685 Mg C/ha/year) and 6% high (mean=5.725 Mg C/ha/year), levels of C sequestration. As for timber volume, approximately 67% of the plots were classified as providing low (mean= 22.639 m^3/ha), 28% as medium (mean=101.538 m^3/ha) and 5% as high (mean=243.339 m^3/ha) levels. Finally, for water yield, nearly 55% of the plots were found in the low provision level, 35 as medium and 10 as high, with means of 1363.46, 2772.29 and 5001.95 m^3/ha/year, respectively. Overall, the classification framework indicated that for all the variables, the data values were skewed left, as approximately 70% of the plots were classified as providing low levels for all the services. This similarity in the distribution of the data determines the direction (i.e., trade-off or synergy) of the interaction between the services. However, as shown in Figures 3-1 to 3-3 the spatial distribution of high water yield values mostly overlap with low and medium values for net carbon sequestration and net timber volume growth.

A portion of the plots studied (around 20%) showed low synergy interactions among the three services, i.e., the services were supplied at the low provision level of 1. Most of the trade-off interactions found in this study were because of the dominance of water yield (38% of the cases), timber volume (13%) and net carbon sequestration (over 11%). However, as reported in Table 3-2, our framework did identify a few areas (only 1%) where the 3 ecosystem services in the bundle were in synergy at the intermediate provision level. Finally, in a few cases there was synergy between timber and water (around 4%); between carbon and water (2%); and between carbon and timber (over 10%).

Mapping ecosystem service provision at the plot level, shown in figures 3-1, 3-2 and 3-3, displayed the spatial distribution of the three provision levels across the study area. Moreover, the spatial distribution in Figure 3-4 displays the pattern of plots where trade-offs (i.e., crosses) occur compared to plots where there is synergy (i.e. circles). In the bundle, synergistic interactions were identified among the three services or between pair of services.

Effect of Drivers on Individual Ecosystem Service

The results from the regression analysis indicated that stand age, treatment and site quality were significant drivers of net carbon sequestration, at $p=0.0052$, $p<0.0001$ and $p<0.0023$, respectively (Table 3-3). Therefore, the older the forest stand, the lower the net increase in carbon sequestration. In addition, carbon sequestration decreased as a result of implementing silvicultural treatments. Furthermore, site quality was also a significant predictor of net carbon sequestration, as higher productive sites (i.e., lower class codes) were associated with higher carbon sequestration rates. Although ownership was not a statistically significant driver, net carbon sequestration was associated more so with private forests than those under public land tenure. Finally disturbance regime was not statistically significant, but on average, disturbed plots positively increased net carbon sequestration.

For timber volume, three of the drivers, stand age, silvicultural treatment and site quality, had a statistically significant effect on timber volume as indicated in Table 3-4, where timber volume was higher in older stands than in younger ones ($p=0.0455$).

Similarly to carbon sequestration, silvicultural treatment and site quality had significant effect (both at $p<0.0001$) on this variable. Indeed, slash pine forest stands that received treatments show an increase in the merchantable timber volume. For managed (treated) slash pine forests, merchantable timber volume is more likely to increase, compared to unmanaged slash pine stands. Ownership and disturbance regime were not statistically significant drivers for timber volume. Delphin, (2012) found that timber production in most of forested areas in North Florida was under low risk of damage due to hurricanes.

Finally, for water yield, all of the drivers analyzed had a significant effect, except disturbance regime (Table 3-5). Water yield significantly decreases as the slash pine stands become older ($p<0.0001$). Also, greater water yield values were associated with treated plots ($p<0.0001$). Unlike for the other variables, ownership was a significant predictor. Slash pine stands managed under private ownership yielded more water, compared to public owned slash pine forests. Finally, in contrast, to net carbon sequestration and timber volume, higher water yield values were more associated to lower site quality.

Effect of Drivers on Ecosystem Service Interactions

Results suggest that all of the drivers analyzed were statistically significant and indicate the likelihood of having a synergy or tradeoff among the services. As indicated in Table 3-6, the coefficient for age was negative, so as a forest stand gets older the probability of having a trade-off between net carbon sequestration, timber volume and water yield increases. In contrast, plots that receive silvicultural treatment are more likely to indicate a synergy between the services. Also, stands managed under public ownership were more likely to indicate a synergy between the services than the stands managed under private ownership. Moreover, synergistic plots were more likely to be associated with more productive sites. Finally, disturbance regime, which was not a significant driver when analyzing the services individually, indicates that stands that have experienced disturbance are more likely to indicate synergy between the services.

Table 3-1. Descriptive statistics of the level of provision of each ecosystem service used in the interaction analysis. Note: Level 1= low, 2= Medium and 3= High.

Ecosystem Service	Level	N	Percent	Minimum	Maximum	Mean	Std. Deviation
Net Carbon Sequestration (Mg C/ha/year)	1	271	71.9	-11.716	1.571	-0.511	2.175
	2	83	22.0	1.613	4.042	2.685	0.698
	3	23	6.1	4.294	9.155	5.725	1.201
Timber Volume (m^3/ha/)	1	252	66.8	0.000	62.16	22.639	17.961
	2	108	28.4	62.21	168.49	101.538	28.072
	3	18	4.8	169.18	329.08	243.454	50.339
Water Yield (m^3/ha/year)	1	207	54.9	461.13	2050.292	1363.46	394.816
	2	132	35.0	2064.207	3830.178	2772.29	508.843
	3	38	10.1	3915.432	6298.726	5001.95	670.493

Table 3-2. Percentage of plot in each category of interactions between carbon sequestration, timber volume and water yield. Note: Level 1= low, 2= Medium and 3= High.

Synergy			Trade-off		
Code		Percentage of plots	Code		Percentage of plots
111	All 3 services are in low synergy	19.63%	112	Water is moderately dominant	28.12%
122	Moderate synergy between timber and water	3.71%	113, 213	Water is highly dominant	10.08%
221	Moderate synergy between carbon and timber	9.28%	121	Timber is moderately dominant	9.55%
212	Moderate synergy between carbon and water	2.12%	211	Carbon is moderately dominant	6.90%
222	All 3 services are in moderate synergy	1.06%	311, 312, 321	Carbon is highly dominant	4.77%
331	High synergy between carbon and timber	1.33%	231	Timber is highly dominant	3.45%

Table 3-3. Parameter estimates of the predictors of net carbon sequestration. (note that: treatment: 0=untreated and 1=treated; ownership: 0=public and 1=private; disturbance: 0=undisturbed, 1=disturbed; site quality=1, 2, 3, 4, 5 and 6).

Predictor	Estimate	Std Error	t Ratio	Prob>|t|
Intercept	4.0542135	0.130006	31.18	<.0001*
Age^	-0.002361	0.00084	-2.81	0.0052*
Treatment^[0/1]	-0.624561	0.047084	-13.26	<.0001*
Ownership[0/1]	0.0295234	0.044058	0.67	0.5032
Site Quality	-0.071779	0.023397	-3.07	0.0023*
Disturbance^[0/1]	0.0194269	0.059434	0.33	0.7440

Effect of predictor is statistically significant at value of α=0.05
^Drive measured in cycle 9 (2009-2011)

Table 3-4. Parameter estimates of the predictors of timber volume. (note that: treatment: 0=untreated and 1=treated; ownership: 0=public and 1=private; disturbance: 0=undisturbed, 1=disturbed; site quality=1, 2, 3, 4, 5 and 6).

Predictor	Estimate	Std Error	t Ratio	Prob>|t|
Intercept	6.3382277	0.543221	11.67	<.0001*
Age^	0.007042	0.003509	2.01	0.0455*
Treatment^[0/1]	-1.029454	0.196736	-5.23	<.0001*
Ownership[0/1]	-0.067585	0.184095	-0.37	0.7137
Site Quality	-0.641228	0.097764	-6.56	<.0001*
Disturbance^[0/1]	0.1083232	0.248339	0.44	0.6630

Effect of predictor is statistically significant at value of α=0.05
^Drive measured in cycle 9 (2009-2011)

Table 3-5. Parameter estimates of the predictors of water yield. (note that: treatment: 0=untreated and 1=treated; ownership: 0=public and 1=private; disturbance: 0=undisturbed, 1=disturbed; site quality=1, 2, 3, 4, 5 and 6).

Predictor	Estimate	Std Error	t Ratio	Prob>|t|
Intercept	684.15413	376.4522	1.82	0.0700
Age^	-17.34145	2.431407	-7.13	<.0001*
Treatment^[0/1]	1113.7285	136.3381	8.17	<.0001*
Ownership[0/1]	-324.7228	127.5775	-2.55	0.0113*
Site Quality	442.70012	67.75018	6.53	<.0001*
Disturbance^[0/1]	50.617268	172.0992	0.29	0.7688

Effect of predictor is statistically significant at value of α=0.05
^Drive measured in cycle 9 (2009-2011)

Table 3-6. Effect likelihood ratio tests and parameter estimates for synergy (1) and trade-off (0) interactions between net carbon sequestration timber volume and water yield in Florida slash pine plots. (Note that: treatment: 0=untreated and 1=treated; ownership: 0=public and 1=private; disturbance: 0=undisturbed, 1=disturbed; site quality=1, 2, 3, 4, 5 and 6).

Driver	Estimates	DF	L-R ChiSquare	Prob>ChiSq
Age^	-0.0289166	1	29.6760995	<.0001*
Treatment^[0/1]	0.93621588	1	8.82217979	0.0044*
Ownership[0/1]	-0.596385	1	4.55557823	0.0363*
Site Quality	0.48486631	1	10.5227632	0.0015*
Disturbance^[0/1]	1.0417733	1	6.75663149	0.0139*

Effect of predictor is statistically significant at value of α=0.05
^Drive measured in cycle 9 (2009-2011)

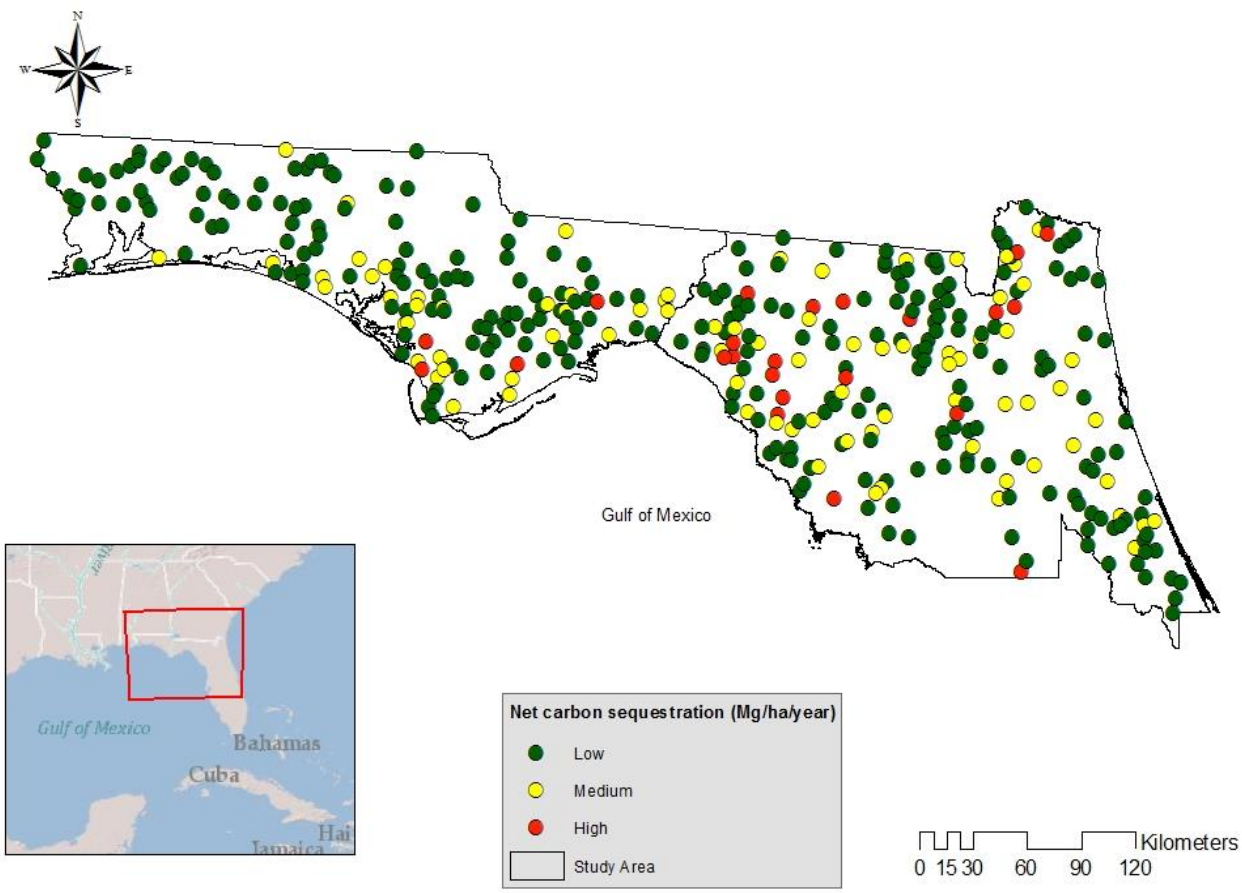

Figure 3-1. Net carbon sequestration rate provision levels for forest inventory analysis (FIA) slash pine plots, in Florida northeastern and northwestern survey units, 2002-2011.

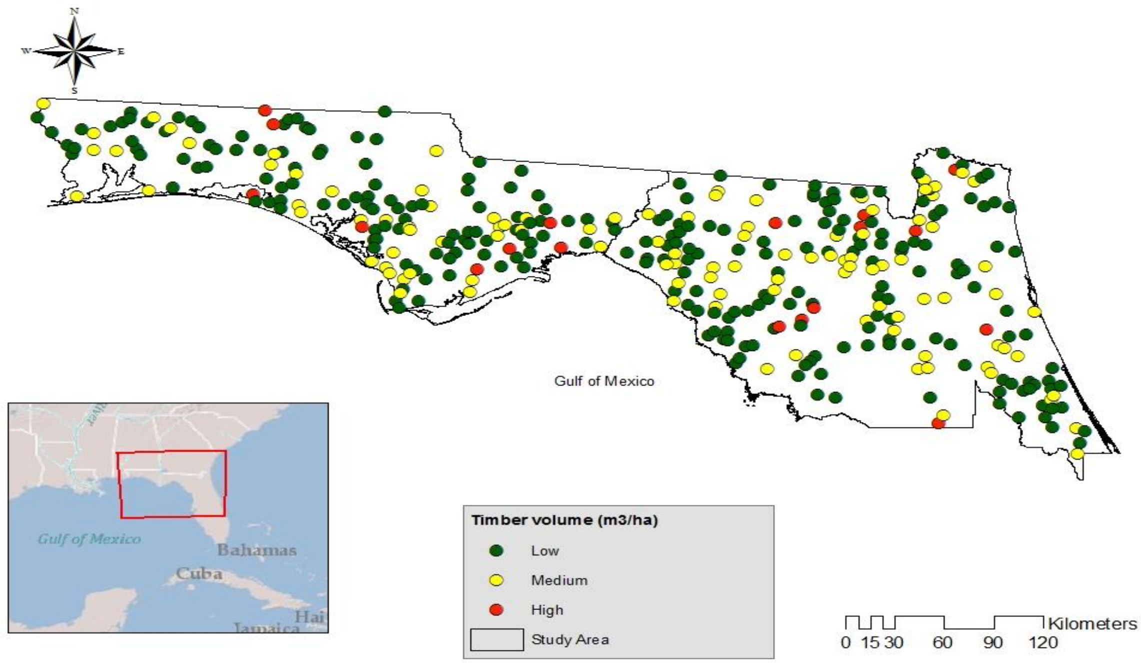

Figure 3-2. Timber volume provision levels for forest inventory analysis (FIA) slash pine plots, in Florida northeastern and northwestern survey units, 2002-2011.

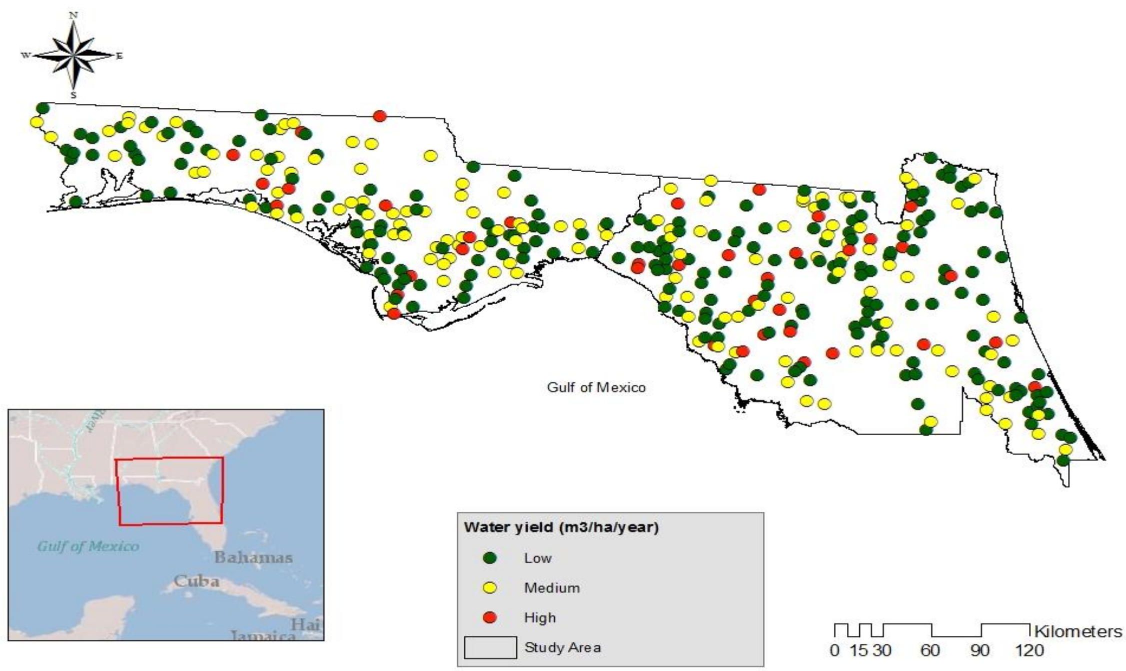

Figure 3-3. Water yield provision levels for forest inventory analysis (FIA) slash pine plots, in Florida northeastern and northwestern survey units, 2002-2011.

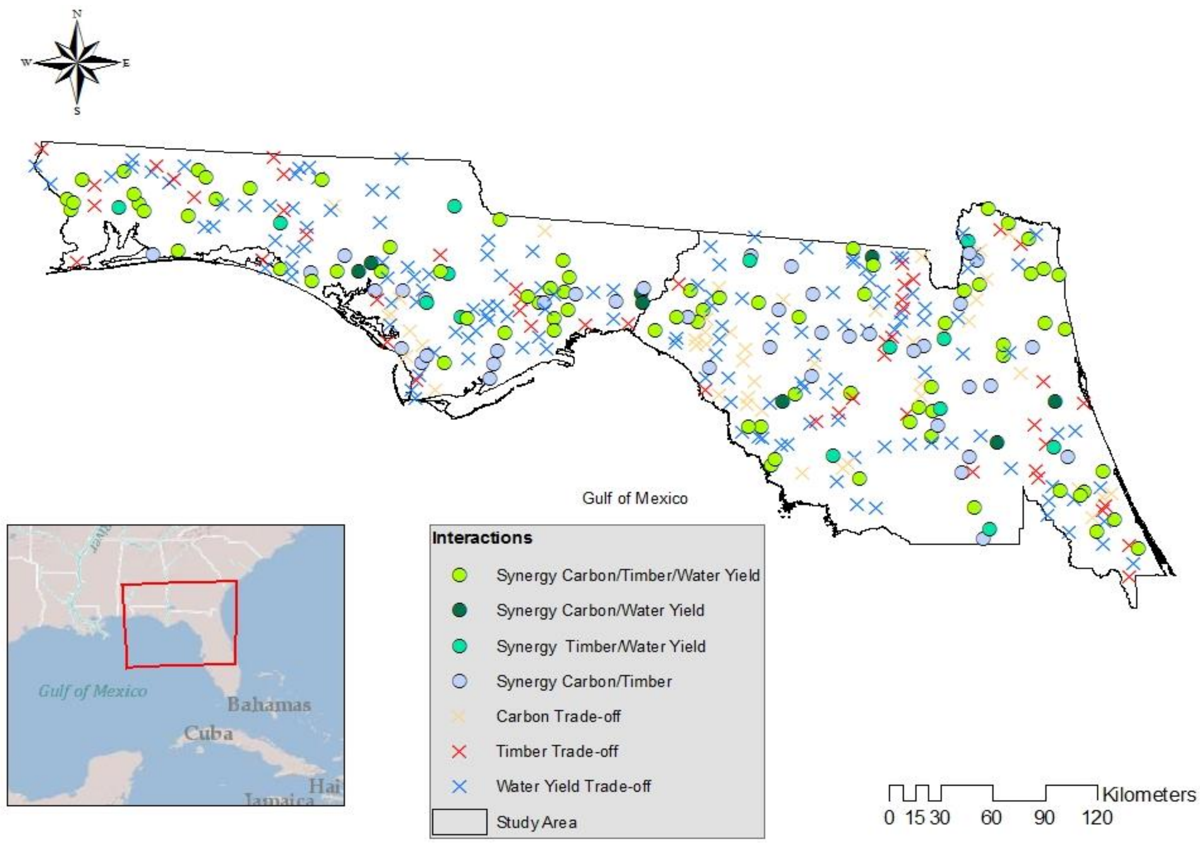

Figure 3-4. Ecosystem services interactions among net carbon sequestration, timber volume and water yield for forest inventory analysis (FIA) slash pine plots, in Florida northeastern and northwestern survey units, 2002-2011.

CHAPTER 4
DISCUSSION

Overview

This study presents a framework for quantifying and analyzing interactions among ecosystem service bundles, and their levels of provision, at the landscape-scale. Specifically, this study quantified and explored interactions among three ecosystem services: net carbon sequestration, net volume growth and water yield and also analyzed the influence of different human and ecological drivers on the provision level of the service and the resulting interactions. While the results confirmed evidence of trade-off between water yield and carbon sequestration and timber in slash pine forest in North Florida, during the period 2002 to 2011, our classification framework identified some areas with moderate synergy between the three ecosystem services or between pair of services. These areas of synergy have implications for multiple use management activities and policies that aim to maintain desired provision levels of multiple ecosystem services. Furthermore, our analysis indicated that stand age, silvicultural treatment, and site productivity were the most important drivers that influence the provision of the ecosystem services and therefore, dictate the direction of the interaction (e.g., trade-off or synergy) among them. These findings are discussed in the following sections.

Ecosystem Services Estimation and Interaction

During the period between the FIA inventory cycle 8 and 9, the slash pine stands inventoried sequestered between -11.7 and 9.0 (mean=0.6) Mg C/ha/year. These findings might be underestimating the total value as the fate of harvested products, which delay carbon emissions (Tallis et al., 2011; Timilsina et al., 2012), was not accounted for in this study. Using our spatial classification framework, 72% of the plot studied were classified as providing low ecosystem service levels, with an average of -0.511 Mg C/ha/year. Nearly 22 and 6% of the plots were categorized as medium and high provision, with averages of 2.7 and 5.7 Mg C/ha/year, respectively. While this might be a result of the classification method used, this concentration of the carbon values was consistent with the findings of Timilsina et al. (2012), which indicated that slash pine forests present a lower probability of being a hotspot for carbon storage, compared to other forest types (e.g., upland hardwood and oak hickory). In addition, the overall mean value of 0.6 Mg C/ha/year or the maximum value of 9.2 Mg C/ha/year, were slightly lower than carbon sequestration rates (10-12 Mg C/ha/year) reported by Shan et al. (2001), for a 17-year slash pine stand. This is due to the effect of stand age on aboveground carbon sequestration, suggesting that older forest sequester less carbon than younger ones (Clark et al., 2004), as 33 years was the mean age for the plot analyzed in this study.

In the case of timber, the merchantable volume averaged approximately 56 cubic meter per hectare (m^3/ha). The classification framework grouped 67% of the slash plots as low provision, with an average of 22.6 m^3/ha. This same classification categorized 28.4 and 4.8% of the plots as medium (101.5 m^3/ha) and high (243.5 m^3/ha) provision. Since slash pine is considered the dominant softwood species in Florida (Brown et al., 2012; Shan et al., 2001), these values are within range of estimates reported by (Brown et al., 2012) for softwoods in Florida., which averaged 82.5 cubic meter per hectare (m^3/ha).

During the period of the study there was a positive water yield for all the Forest Inventory Analysis (FIA) plots considered, with values ranging from 461.0 to 6298.7

m^3/ha/year. This range of water yield values is similar to findings by McLaughlin et al. (2012) that reports annual water values ranging from 500 to over 6000 m^3/ha/year. However, water yield did decrease as a result of growth in biomass (Figure 2-5 and 2-6).

There is wide range of publications that support the paradigm of reduction in water yield (e.g., runoff) as consequence of forest plantations (Bosch & Hewlett, 1982; Farley et al., 2005; Jackson et al., 2005; van Dijk & Keenan, 2007). However, when the ecosystem functions behind the service are not taken into account, potential management practices that could enhance water use efficiency from forest plantations are often not fully considered and adopted into management (Jerome K, 2009). This study, while not attempting to reject the well-supported paradigm that forest plantations decrease the provision of water yield, does provide a framework that helps identify areas where synergies can be found among different ecosystems services provided by a forest area dominated by a single tree species.

Other studies such as Bennett et al. (2009) and Raudsepp-Hearne et al. (2010) defined trade-off among ecosystem services as the situation when the production of one service inhibits the provision of the other, i.e., the supply of one service increases while the supply of the other decreases. They also defined synergy as the situation when the provision of two ecosystem services increases or decrease simultaneously, which represent a win-win situation (Power, 2010). Indeed, Raudsepp-Hearne et al. (2010) identified trade-off patterns at the landscape scale between regulation services (e.g., carbon sequestration) and provision services (e.g., water).However, studies such as these on interactions among sets of ecosystem services often use correlation coefficients and graphics for their analysis. Therefore our approach analyzing these ecosystem service interactions, provision levels, and their drivers is a contribution to these types of studies. Although most of the provision levels of the three ecosystem services were classified as low, only 19.6% of the plots were categorized in the 111 (i.e., low synergy, as defined a priori). The reason was because in the Jenk's algorithm, the proportions are classified using natural breaks in the data ((Carr & Zwick, 2007). In addition, our analysis identified some plots with synergy between pairs of services. In the case of net carbon sequestration and timber volume interaction, the synergy or trade-off defined here in this study is based on a management goal stand point, as carbon and timber refer to the same tree biomass measured in different units. However, since the majority of above carbon biomass is encountered in merchantable part of a tree bole (Brown et al., 2012), this categorization of the biomass is important in defining and managing the forest ecosystem for a specific goal (e.g., carbon or timber) and therefore, can be used to identify which areas in north Florida and which management activities can help better reduce the negative impact of natural and managed forests on water yield.

Effect of Drivers on Individual Ecosystem Service

Usually, growth in forest stands is quantified using metrics such as increase in aboveground ground biomass (e.g., net annual carbon sequestration) or the increment in volume of the stand, e.g., net annual volume growth (Arneth, Kelliher, McSeveny, & Byers, 1998). The results reported in this study suggest that stand age, silvicultural treatments, and site quality were all significant drivers of these attributes of growth in forest carbon biomass and merchantable timber volume. Control of competition at early age benefits growth in carbon biomass of managed forest stands. However, the same growth can be obtained in older stand when the practice of understory removal is used (Shan et al., 2001). The significant effect of site quality on carbon sequestration and timber volume is plausible as this driver indicate the capacity of a land to grow biomass, on an annual basis (Woudenberg et al., 2010).

Usually, studies discuss the negative effect of forest plantations (Farley et al., 2005; Sahin & Hall, 1996; Zhang, Dawes, & Walker, 2001) on water yield without accounting for the specific forest structures and their functions associated with this effect (Jerome K, 2009). However, McLaughlin et al. (2012) suggested that management schemes that control key biophysical drivers can help increase water yield substantially in forested areas. Our study, while not proposing specific management strategies of certain biophysical drivers of water yield, found that stand age, silvicultural treatment ownership and site quality were significantly associated with water yield. The results suggested that higher stand age is associated with lower water yield values. This can be explained by the fact that leaf area index (LAI) increases as a forest becomes older (McLaughlin et al., 2012). As indicate in Table 3-4, silvicultural treatments are positively associated with higher water yield. This is because a reduction in forest biomass by thinning, for instance will reduce LAI and therefore, increases water yield (Bosch & Hewlett, 1982; Douglass, 1983; Farley et al., 2005; Hewlett & Hibbert, 1961).

Moreover, ownership was significantly associated with increase in water yield. In fact, our findings showed that slash pine stands managed under public ownership, on average, show higher increase in water yield than their counterparts managed under private land tenure. This is may be due to water yield management goals and silvicultural activities (Shan et al., 2001), as forest lands in Florida’s public areas are managed for water (e.g., water management district lands) while forests under private land tenure in the Southern region are primarily managed for timber production (Heath, Smith, Woodall, Azuma, & Waddell, 2011), which indicate more frequent silvicultural activities, which have direct effects on water yield (McLaughlin et al., 2012).

Effect of Drivers on Ecosystem Service Interactions

An important part of this study was to analyze the influence of drivers on the likelihood of a plot being associated with a tradeoff or a synergy. All of the drivers analyzed in this study, significantly predicted this outcome. However, as shown in Table 3-6, stand age, treatment and site quality were the most significant drivers. This is because these three drivers were also significant predictor of the services individually.

One method often used to analyze the interaction among multiple ecosystem services is testing how each service responds to a common driver. Opposite responses indicate trade-off while similar ones suggest synergy (Bennett et al., 2009). As developed in the method section, LAI was identified as the common driver in all three ecosystem services evaluated in this study. However, LAI was not analyzed as it was directly used in the estimation of water yield.

Although the results of this study support evidence from already published literature on interaction among forest ecosystem services (Bennett et al., 2009; Farley et al., 2005; Jackson et al., 2005), this study introduces a classification framework that has some potential applications for ecosystem management, but there are some limitations. One limitation is reflected in the classification algorithm used to group the data. Given the continuous nature of the variables, the classes generated by the natural breaks algorithm hide underlying trends in the dataset. Although 377 plots were used in the study, they were distributed across nearly 50,000 square kilometer (km^2), across north Florida, which suggests a relative small sample size.

Also, the results of the framework are limited and dependent on the classification method used, which can influence the number of plots identified in the category as trade-off or synergy. But, since the values of the ecosystem services were skewed and therefore, non-normal, the Natural Break classification method was the best suited (Carr & Zwick, 2007). However, once again this limits the use of and interpretation of the continuous data in other types of optimization and multi-criteria analyses. Finally, another limitation is related to the size of the plots (1 acre) used in this analysis, which makes the contribution of a plot, in terms of increase in water yield at a watershed scale, appear to be very minor.

CHAPTER 5
CONCLUSION

Based on the conceptual framework which defines ecosystem services as the direct benefits from natural ecosystems to humans, this study investigated interactions between carbon sequestrations, timber volume production and water yield for slash pine forests in north Florida. This study is novel in that the provision levels of each of these 3 ecosystem services was quantified using georeferenced field data from the Forest Inventory Analysis (FIA) program of the US Forest Service. In the calculation, the net values of each services was estimated over a 7-year time period (2002-2011). Secondly, a classification framework was developed to determine which plots exhibited tradeoff or synergy interactions among the three ecosystem services or between pair of services. And finally, the study tested the effect of some human and ecological drivers on the individual services and also on the interactions in an ecosystem service bundle (e.g., trade-off or synergy).

Results indicated that biomass measured as carbon sequestration or timber volume reduced water yield during the study period. Nevertheless, this trade-off interaction varied across space, as revealed by the lack of correlation in the model of water yield as function of net carbon sequestration or timber volume growth. The classification framework developed in this study also accounted for this spatio-temporal variability, and therefore could be used to identify interest where trade-off and synergy (i.e., “win-win”) areas occur. Also, the results indicated that, although the effect of some drivers was not statistically significant on individual services (e.g., disturbance regime and ownership on timber volume and carbon sequestration), all the drivers analyzed affect the interaction among the services studied, with stand age, treatment and site quality, the most significant.

Generally, the management of natural resources tends to focus on a single resource or objective, overlooking the multiple functions of an ecosystem (Nelson et al., 2009). This approach often ignores the capability of an ecosystem to, when properly managed, generate a wide range of, and multiple, services to people, and therefore maximize profits as well as enhance sustainability (Bennett et al., 2009; Tallis & Polasky, 2009; Tallis et al., 2011). Thus, managing natural ecosystems properly and sustainably requires information on how to reach multiple benefits, objectives, or desired provision level of different ecosystem services in a bundle (Tallis et al., 2011). However, reaching desired levels of these multiple services and identifying their spatio-temporal characteristics is difficult. Geospatial and statistical modeling however, as used in this study can provide useful information on how to manage the natural ecosystem in order to reduce trade-offs (Tallis & Polasky, 2009).

These objectives can also be understood using hierarchical sets of statement that define different goals (Carr & Zwick, 2007). For example, a timber company may want to achieve a certain timber volume from its managed forests. But, an environmental institution involved in climate change regulation may want the forest to sequester the maximum amounts of carbon per year. Similarly, a private or public entity managing the supply of water resources may have an interest in knowing the quantity of water a forested area can yield every year. In these three cases, if the entities depend on the same ecosystems (e. g., managed forests) and a single institution wants to be responsible for its multiple management, one of the strategies to attain sustainability would be to prioritize actions on areas or ecosystems where these goals (or ecosystem service provision levels) are provided at the highest level.

As environmental protection becomes more important due to the anthropogenic impacts on natural systems, ecosystem services emerge as a relevant conceptual framework to study and manage natural ecosystems and monitor the effects of these unprecedented rates of land use change (Daily, 2000). Some of the most important changes are the increase in carbon dioxide concentration in the atmosphere (Vitousek, Mooney, Lubchenco, & Melillo, 1997) and alteration of species composition (Tilman et al., 1997). These environmental changes, which affect ecosystem processes and the flow of goods and services, (de Groot et al., 2010), are felt worldwide (Daily, 2000). For example, natural ecosystems in North Florida, once dominated by Long Leaf pine, have been replaced by Slash pine (Clark et al., 2004). While this change in forest cover can provide substantial benefits to one sector (forest industry), it may affect the overall balance of the regional ecosystems and this affects other process (e.g., water balance, biodiversity). In the US southeastern lower coastal plain, environmental conditions (e.g., soil and wildfire) coupled with management practices have greatly influenced the spatio-temporal variability of biophysical structures, such as leaf area index (LAI), composition and stand densities. Similarly, water and energy fluxes are sensible to these change in landscape conditions (Gholz & Clark, 2002).

Most of the studies that assess the interaction of multiple ecosystem services use biodiversity as a proxy. One of the key aspects of this study is to illustrate that even a single-species forest can provide multiple ecosystem services. Unlike many of the studies cited above, our analytical approach is not using any simulation or weighting of the ecosystem service considered. We let the data dictate the ranking (high, medium, and low) of the services individually by applying the same classification system.

Future research that take into account the issues explained above to investigate provision levels and interactions among biomass growth and water quantity can improve this study. Also, water resources entities interested in knowing the impact of forest management on water yield by their watersheds might want to use datasets which include more frequent inventories than those by the FIA program. One other approach for evaluating interactions between multiple ecosystem services is multi-criteria decision analysis (MCDA) or multiple criteria decision support. For example (Schwenk et al., 2012) used this analytical approach to evaluate forest management alternatives on the provision of carbon sequestration, timber production, and biodiversity. Their methodology included weight assignment to different management objectives that target a specific ecosystem service. Results can be used to assess how multiple forest management approaches supply different (opposite) services and how to balance trade-offs and maximize the provision of the ecosystem services.

Environmental planning, decision making, and management of natural ecosystems for multi-functional uses, can create synergies between conservation (e.g., water services or carbon sequestration) and economic growth (e.g., timber volume). To achieve that goal, it is important to assess the relationship between ecosystem management and the provision of all the linked services provided, and therefore detect optimal management option (de Groot et al., 2010). For example, this study, which assesses spatial and temporal analysis of evapotranspiration which is an important ecosystem process that links carbon sequestration, timber production and water yield, as a result of afforestation, is compelling to better understand the tradeoffs involved with the co-management of services (Balvanera et al., 2001; Kearns, Inouye, & Waser, 1998; Monica, 1998). This spatial assessment, which includes mapping and visualization, is one of the instruments often used to analyze management changes on natural systems.

In conclusion, most studies focus on the quantification or valuation of a single ecosystem service without accounting for the interactions among multiple ecosystem services and their different levels of provision. The method used in this study identified these areas where land managers can have acceptable levels for three key ecosystem services and an understanding of the forest management variables driving that interaction. Therefore, results and methods from this work could be used to improve the understanding and use of the ecosystem service framework and also help managers who rely on spatial information to know how and where management can be most effective and efficient at providing ideal bundles of ecosystem services provided by natural ecosystems.

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BIOGRAPHICAL SKETCH

Ronald Cademus was born in Maniche, Haiti. He graduated from Universidad ISA in Dominican Republic in 2008 with a bachelor degree in forestry, where he learned the importance of the importance of ecosystem services for a more sustainable world. Upon graduation Ronald went back to Haiti and joined the Fondation Seguin, the leading national environmental NGO in Haiti, working La Visite National Park. During 2 years, Ronald led a conservation project where he trained local farmers in tree nursery and soil conservation. In June 2010, he was awarded a 2 scholarship grant from the USAID/WINNER Project in Haiti for his master program here at University of Florida. He received his master from the Interdisciplinary Ecology program from the School of Natural Resources and Environment (SNRE) under the direction of Dr. Francisco Escobedo.