ENVIRONMENTAL AMELIORATION POTENTIAL OF SILVOPASTORAL AGROFORESTRY SYSTEMS OF SPAIN: SOIL CARBON SEQUESTRATION AND PHOSPHORUS RETENTION

By

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A DISSERTATION PRESENTED TO THE GRADUATE SCHOOL OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

UNIVERSITY OF FLORIDA

2009
To my parents, Scott and Carol Howlett, whose love and support made this work possible
ACKNOWLEDGMENTS

I offer my deepest gratitude to Dr. P.K. Ramachandran Nair, for his kind support, advise, and understanding throughout my doctoral program. I am indebted to him for guiding my academic life over the past four years. I would also like to acknowledge my other Supervisory Committee members Dr. Vimala Nair, Dr. Nick Comerford, Dr. Wendell Cropper, Dr. Lynn Sollenberger, and Dr. M. Rosa Mosquera-Losada for supporting my academic progress.

This project was executed in conjunction with the University of Santiago de Compostela, Crop Production Department, in Galicia, Spain. Dr. M. Rosa Mosquera-Losada served as my academic advisor in Spain, and provided significant support while in Spain. Dr. Gerardo Moreno of the University of Plasencia, Forestry School also provided support for the Dehesa sampling in west-central Spain. I am the beneficiary of work done by many dedicated Spanish researchers.

My lab mates, in Florida and Spain, made life easier during my time in both areas. In Florida, I would like to acknowledge Dr. Solomon Haile, Dr. Subhrajit Saha, Dr. Asako Takimoto, Dr. Alain Michel, and Wendy Francesconi. In Spain, I would like to recognize Daniel Moran Z., Christos Paraskevopoulos, and Antonio Cunha Guimarães. Javier Santiago Freijanes, Divina Vasquez, Teresa Pineiro-Lopez, and Pablo Fernández Paradela provided invaluable assistance in analyzing the soil samples and fieldwork. I am indebted to all for their help and encouragement.

Financial support for this dissertation was provided by: the Alumni Association at the University of Florida, the School of Forest Resources and Conservation, Center for Subtropical Agroforestry, the Institute of International Education, Fulbright Scholars Program, and The Universidad de Santiago de Compostela, Crop Production Department.
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December 2009

Chair: P.K. Ramachandran Nair
Cochair: Vimala D.Nair
Major: Forest Resources and Conservation

This study investigates the environmental amelioration potential for silvopasture agroforestry systems of Spain to store and retain soil carbon (C) and phosphorus (P). Interest in C has grown due to its role in affecting global climate. Fertilizer P from can become an environmental pollutant when applied in excess of a soil’s storage capacity. To assess soil C and P retention in Spain, two study sites with were selected: a “Dehesa” silvopasture planted with cork oak (Quercus suber) and a silvopasture experiment planted with radiata pine (Pinus radiata) and birch (Betula alba). Soils underneath trees and in adjacent open pastures were sampled to 100 cm, wet sieved into four size classes (<53 µm, 53-250 µm, 250-2000 µm, and >2000 µm), and combusted for C determination. Potential for P contamination was assessed using the Phosphorus Saturation Ratio (PSR) and Soil Phosphorus Storage Capacity (SPSC), indicators of soil P saturation and storage capacities. Results from the cork oak site indicated that soils closer to the tree, as compared to away, had C greater in the 250 – 2000 µm soil fraction and that overall C was higher in the 50 – 100 cm depths. Soil C to 100 cm increased from 20.01 to 41.22 Mg C ha⁻¹, from away from the tree canopy to underneath the canopy. In the simulated silvopasture, birch had more C in the 250 – 2000 µm size class from 50 – 100 cm, as compared
to the pasture, and radiata pine exceeded pasture C storage in the 250 – 2000 µm size class in the
75 – 100 cm depth. More C was found in the <53 µm size class in the pasture, as compared to
silvopastures from 0 – 25 cm depth. Maintenance and expansion of silvopastures in Spain will
increase C storage in macroaggregate-sized soil fractions in the deeper soils, and improve long
term storage potential. In the PSR and SPSC studies, extraction of P from soils by deeply-rooted
trees in silvopasture was believed to reduce soil P in the deeper soils examined, reducing
potential P loss as compared to a pasture under similar conditions.
CHAPTER 1
AGROFORESTRY AND ENVIRONMENTAL SERVICES FOR THE 21ST CENTURY

Introduction

The human race benefitted greatly from the ‘Green Revolution’ that took place in the post World War II era. The development of petroleum based fertilizers, pesticides, and herbicides, as well as improved crop varieties, helped spawn a worldwide agricultural revolution, feeding millions of people like never before. This revolution in agriculture prevented famine and contributed to an unprecedented human population growth in the second half of the 20th century. Most Europeans and North Americans, in particular, are now dependent on industrialized agriculture for cheap and plentiful food. The technology-driven lives of millions are now subsidized by solar energy that had been captured millions of years ago by plants and is now drilled from the ground as a black sludge. As a fossil fuel, petroleum is used in combustion engines, and for the production of electricity and agricultural fertilizers. Each of these uses for fossil fuels represents a massive annual transfer of carbon from belowground stable forms to atmospheric carbon dioxide gas (CO₂). Excess CO₂ in the atmosphere is believed to be one of the principal causes of the greenhouse effect, whereby long wave energy from the sun is prevented from leaving the earth’s atmosphere by CO₂ molecules (and other gases, as well). This trapped heat contributes to what is known as ‘global warming.’ This warming may have a potentially devastating effect on global climate, and thus, the human condition. Interest has grown in recent years in determining ways to mitigate this effect. People are buying more energy-efficient cars, appliances, and homes, and the concept of one’s ‘carbon footprint’ has entered the lexicon. Managing the increase in atmospheric concentration of CO₂ by capturing, conserving, and sequestering carbon (C) are all promising means by which to combat global climate change. There may not be one single solution to contain the climate crisis, so researchers are working on
various fronts to reduce atmospheric CO₂. The conservation and enhancement of vital terrestrial C sinks will play a role in preventing excessive concentrations of CO₂ in the atmosphere.

Land use and land use change plays a significant role in how C is cycled in terrestrial ecosystems. Carbon contained in the world’s soils, 2,300 Pg, outweighs both atmospheric and plant biomass C, 770 and 610 Pg, respectively (Dixon et al., 1994, and Batjes and Sombroek, 1997, Nair et al., 2009b). Most of this soil C is found in organic forms. Thus, landscape scale improvements in management of soil organic carbon (SOC) can help to reduce atmospheric CO₂ levels. The increased storage, or sequestration of C in reservoirs such as in soil and plants, removes CO₂ from the atmosphere. Sequestration of C by woody plants is an inexpensive means by which to capture and maintain CO₂ out of the atmosphere for times scales generally equivalent to the lifespan of the plant. Additionally, maintaining and improving current soil C reservoirs is very important due to the current large scale of storage and possibilities for long term sequestration beyond that of the lifespan of the plants contributing to it. Land managers are increasingly interested in managing forests for C sequestration, whereby growth of woody biomass is considered to offset emissions of C elsewhere. Reforestation or afforestation of treeless areas leads to greater C sequestration above ground, and increases in soil C (Post and Kwon, 2000; Paul et al., 2003; Montagnini and Nair, 2004; Haile et al., 2008). Wherever climatic conditions exist for plant growth and the storage of associated residues, there is potential for sequestration of C in biomass and underlying soils. Afforestation, reforestation, and shifting to no-till agriculture (as well as other land use activities) has the potential to offset emissions of CO₂, reducing atmospheric CO₂, and the effects of global warming.

One of the great benefits of the Green Revolution was the invention and advent of chemical fertilizers. This, in addition to improvement of high yield varieties of grain, allowed
farmers to produce large, unprecedented crops, leading to cheap food for a growing world population. For the individual farmer though, managing fertilizer applications to maximize production can lead to excess fertilizer use. Insurance fertilization of crops may ensure full plant nutritional status, but water quality can be affected when fertilizer applications exceed the soil’s storage capacity and/or the plant’s capacity to take up applied nutrients (Schlegel et al., 2000; Aarts et al., 2000). In the past, farmers applied enough fertilizer to maximize food production, and were not necessarily held responsible for excess applications that led to contamination. Each soil has a limited capacity to adsorb and retain nutrients, and whatever nutrients are applied in excess, may be lost to leaching and/or erosion. From the point of view of the producer, little attention has been paid to the soil’s capacity to store and release fertilizer nutrients over time. As such, fertilizer nutrients not held by the soil have made their way from the farmer’s fields to the world’s water bodies (Graetz and Nair, 1995; Nair et al., 1998). Excess nutrients in drinking water, particularly nitrogen (N) and phosphorus (P) cause a host of human health problems. Water quality is also significantly impaired by eutrophication, as excess nutrients cause algal blooms, further reducing oxygen levels beyond that required to sustain some aquatic life (Perkins and Underwood, 2002). While the loss of fertilizer nutrients represents an economic loss to the farmer via lower fertilizer use efficiency, the migration of these nutrients also decreases water quality (Van der Molen et al., 1998; Schlegel et al., 2000). Phosphorus is a key limiting plant nutrient in natural systems whose concentration in an agricultural soil is artificially increased through fertilization and/or through management of animal wastes. Phosphorus is an element without a gaseous phase, and as such, it is not readily lost from the terrestrial system, such as in the case of nitrogen. Loss of P through excessive application can occur by leaching to groundwater, because some soils have limited ability to capture and store nutrients passing
through the rhizosphere. Phosphorus loss is generally not a problem in soils with high clay content. Sandy soils are prone to P loss because of their texture. A sandy soil generally has less ability to bank and transmit plant nutrients than a finer textured soil, and thus applications of fertilizers on sandy textured soils must be carefully monitored to prevent contamination to water sources. Researchers from the University of Florida Center for Subtropical Agroforestry have shown that the addition of trees to a traditional treeless pasture can help to reduce soil P losses in sandy soils in Florida (Michel et al., 2007), and diversify production on the land by the additional provision of timber. Fertilizer nutrients are taken up, used, and stored by deeply rooted trees that ‘recapture’ nutrients that contaminate water bodies. Trees are large biotic elements that can serve to regulate the cycling of nutrient contaminants on the agricultural landscape. Planted and maintained adequately, trees can be used to reduce high levels of soil P and N on agricultural lands, while at the same time providing valuable forest products (Allen et al., 2006 and Michel et al., 2007).

Spain is a modern western European country that benefitted from the green revolution of the 20th century. Due to a mild temperate climate from southerly ocean currents, Spain’s arable land has been cultivated for centuries. Spain can be roughly divided into two climatic zones, hot and dry Mediterranean in the south and cooler and wetter Atlantic Spain to the north. In Mediterranean south and central Spain, evergreen oak (Quercus spp.) savannah agroforestry systems called ‘Dehesas’ produce a variety of products such as firewood, cork, forage for cattle and sheep, as well as oak acorns for pig consumption. Over the past century, a reduction in tree cover has occurred, due to disease and a local belief that trees had a negative effect on forage production (Joffre et al., 1999). Land use changes in northern Atlantic Spain have seen the conversion of traditional pastures and row cropland to forest plantations of mainly exotic, fast
growing pines, *Pinus* spp., and eucalyptus, *Eucalyptus* spp. (Zas and Alonso, 2002; Marey-Perez and Rodriguez-Vicente, 2008). Traditional pasture lands are being abandoned for afforestation schemes, animal production has been reduced, and forest fires have become a greater concern (Rigueiro-Rodriguez et al, 2005). While tree cover has been reduced in the Mediterranean Dehesas, environmental services associated with this landscape, such as erosion control and microclimatic improvements, may have also been reduced. The massive conversion of traditional pastures and cropped land in Atlantic Spain to exotic forest plantations may not be a preferred land use for several reasons; as silvopasture agroforestry is an intermediary land use measure (between pasture and plantation) to reduce forest fires, preserve animal production and maintain a more attractive open landscape. Some researchers have begun to consider the secondary environmental benefits associated with these land use changes, such as C sequestration and retention of nutrient contaminants such as P (Rodriguez-Murillo, 2001; Fernandez-Nunez, 2007; Fouz et al., 2009). Spain is a signatory of the Kyoto protocol, which seeks to reduce global CO₂ emissions, and public education on climate change in Spain is strong. A more complete understanding of soil C and how land use affects it will help with the C accounting that is required for the Kyoto agreement. Also, in Spain, the contamination of water bodies from agricultural sources is a well known phenomenon, and research efforts are underway to address these issues (Hilbebrandt et al., 2008; Fouz et al., 2009). Identifying and expanding land use practices that reduce loading of fertilizers in agricultural runoff (N, P) to water bodies will improve water quality and reduce contaminant related health problems where the impairment of water bodies is a concern.

**Objectives and Hypotheses**

The overall objective of the study is to evaluate how much soil C and P are prevented from entering the atmosphere and water, respectively, by the presence of trees in pastures in
comparison with a treeless pasture. Specifically, the objective is to compare the silvopasture agroforestry practices with traditional treeless pastures in terms of their soil C sequestration and P retention capacities in two different climatic regions Spain. This will be accomplished through detailed soil sampling at various soil depths and distances from trees and analyses of the samples for C and P. Carbon storage in different soil size fractions will be determined to gain a better understanding of the mechanism of C storage under the treeless and tree-incorporated systems. As for P retention, a measure of soil P saturation and retention capacity will be calculated based on analytical data on the content of extractable soil P in relation to that of iron and aluminum.

Based on available information, it is hypothesized that C sequestration will be more under silvopasture, as compared with traditional treeless pastures. This is based on the premise that deep rooting trees, coupled with cessation in tillage operations, and consequent formation of stable soil aggregates, is likely to lead to longer-term C sequestration in soils under tree-based systems. It is also hypothesized that fertilizer applications will likely enhance tree growth, and thus increase C inputs to the soil (from above and below ground), particularly at depths below the maximum rooting depths of pasture grasses and lead to more C storage in deeper layers of silvopasture.

Phosphorus removals by deep-rooted trees will likely be greater under silvopasture at lower depths, and as such, P saturation will be reduced where tree roots proliferate. Associated soil P storage capacities will be greater under silvopasture as compared to pasture, where tree roots have removed P deeper in the soil profile. A buildup of soil P is also less likely under silvopasture than under treeless pasture at the landscape level.

**Outline of Remaining Chapters**

Chapter Two is a review of pertinent literature on silvopasture as an agroforestry practice, its use in Spain, and the potential C sequestration and P retention benefits associated with these
landscapes. Chapter Three describes site characteristics for two Spanish silvopastures used in this study, including detailed geographical, meteorological, and soil data, as well as a history of land use and experiments on the sites. Chapter Four will present the results for C sequestration potential in the whole soil, four different size fractions, and whole field estimates. Chapter Five will consider the P saturation and storage capacity underlying two Spanish silvopastures as compared to adjacent treeless pastures, as well as describe the potential for environmental pollution to occur on these sites. Finally, Chapter Six will provide a brief synthesis of results and an evaluation of the use of silvopasture as an environmental management tool for improving storage of C and P on agricultural landscape.
CHAPTER 2
LITERATURE REVIEW

Introduction to Silvopasture

Silvopasture is defined as the planting and/or maintenance of trees in a pasture production system (Clason and Sharrow, 2000; Mosquera-Losada et al., 2005; Ibrahim et al., 2005). ‘Silvo’ and ‘pasture’ describe the two vegetative components of a multiuse agroforestry system that supports the growth of forage grasses for grazing animals and a variety of forest products. The tree and pasture components can be arranged in a variety of forms. Trees can be planted in rows or randomly dispersed, and a variety of grasses or other crops can be grown underneath and between the trees. Combinations of crops and trees are made to maximize resource allocation and use. Trees may be allowed to dominate the system after pasture production wanes and the tree canopy closes, leaving a more traditional tree plantation. Numerous types of grazing animals are supported by silvopasture, and their excrement helps to fertilize the soil for tree and pasture growth. The growth of pasture and tree components differ over time, as trees tend to grow and occupy more space in the system. Two ways to manage this competition are: plant trees at a high density and later thin (remove individual trees) to a desired density, or simply plant at a lower initial density, while risking loss of potential tree growing space. With either management style, the three components of silvopasture are managed to provide maximum desired benefit from the land. Competition between the system components must be kept to a minimum to maintain each component. For example, if animals are left too long to graze, they may degrade the pasture component to a point that erosion related fertility loss in the soil leads to loss of the pasture, or even the tree component. In an alternate scenario, allowing trees to dominate, as described above, will decrease pasture production and lead to a loss of the animal component. A careful balance of management goals must be made to preserve each component.
Silvopasture is one of the most common agroforestry practices in the world, with several variations on the practice to suit local conditions and market demand for forest and animal products (Joffre et al., 1999; Clason and Sharrow, 2000; Dulormne et al., 2003; Sharrow and Ismail, 2004; Mosquera-Losada et al., 2005). Conversion from a traditional treeless pasture to silvopasture can provide several benefits to the landowner. Diversified revenue streams from a greater diversity of forest, agricultural and animal products, and payments for environmental services can make silvopasture an attractive option (Shresha and Alavalapati, 2004 and Garret et al., 2005). Several unique examples of silvopastoralism have been examined by those seeking to understand the multiple benefits they provide (Mosquera-Losada et al., 2005). The Dehesa oak system of southern Spain and pine/birch silvopastures of northwestern Spain, given distinct climatic conditions, each provide insight into silvopastoralism as a general agricultural practice.

**Dehesas of Mediterranean Spain**

**The Dehesa System**

The ‘Dehesa’ is a silvopastoral system of agroforestry production in southwestern Spain and southeastern Portugal (‘Montado’ in Portuguese), occupying about five million hectares of Europe’s Iberian Peninsula (Dupraz and Newman, 1997). The word ‘Dehesa’ is rooted in the Spanish words deffesa or defensa (defense) whereby medieval peasants would defend and tend the local elite’s pasture lands from roaming herdsmen (Montero et al., 1998). With a classically open savannah-like structure with evergreen oaks and pasture and grazing animals, this system contains the three required elements of silvopasture. Between trees, farmers plant cereals and fodder crops such as barley or wheat during times of water availability on cycles ranging from two to five years (Montero, 1998). Grain production generally occurs only on more productive Dehesa sites. Crop residues are often left (or cut and stored) for animal consumption when fresh forage is not available. Landowners increase the time between cropping cycles (or eliminate
them completely) on less productive lands as fertility is reduced. Less productive Dehesas are left to be invaded by native pasture. Trees are a planned and managed component of the system.

The typical Dehesa is planted with Holm oak (*Quercus ilex*) and/or cork oak (*Quercus suber*) that both produce acorns eaten by animals during the winter (November to February), and the cork oak can be harvested on cycles that vary from 9 – 12 years (Joffre et al., 1999). In a country where consumption of jamón serrano (cured ham) is a cultural identifier, the jamón de bellota (‘Ham from acorns’) produced on these lands, is most highly prized for its unique buttery acorn flavor. These oak trees, with low densities of around 30 to 150 stems per hectare, create a relatively open savanna or park-like woodland. Joffre et al. (1999) estimate that most Dehesa lands have between five and twenty percent tree cover. Trees lower wind speeds, reduce evaporation of limited soil moisture, improve the microclimate for animals and other plants (Joffre and Rambal, 1993, Moreno et al., 2006, Moreno et al., 2008). Trees also accrue soil nutrients in biomass and soil and serve as a habitat for floristic diversity by providing a favorable microclimate for animals to forage and for herbaceous plants to thrive (Moreno et al., 2007).

Cattle, goats, and sheep are pastured at variable stocking rates that depend on site specific conditions related to forage production, and transhumance is common in summer months when forage production wanes. Animal husbandry is the primary production goal of the Dehesa system. Each of the silvopasture elements interacts to provide positive benefits to the other components. Trees serve to reduce heat stress of animals and are pruned to provide fodder in times where forage is not available. Oaks are also pruned for fuel wood. Crops are grown to feed animals and they help to maintain floristic composition of the system by disrupting the natural plant successional process through herbage removal and tillage. Typical of many agroforestry systems, the Dehesa system provides a variety of products from its various components, such as
beef, lamb, ham, wild game, cork, and acorns, and, on soils of low fertility in a region with highly seasonal precipitation patterns.

**Biophysical and Geographic Aspects of the Dehesas**

Rainfall typical of the Dehesa is around 600 mm per year, and this rain falls almost exclusively in the winter months (October to April). The long dry season of summer is characteristic of Mediterranean climates (Joffre et al., 1999), and evaporation often exceeds rainfall (Joffre and Ramball, 1993, Montero et al., 1998). During the late summer and fall months, production of forage for animals is significantly reduced as subsurface soil moisture draws down and the landscape dries out. Cereal crops are sometimes grown during the wetter months to take advantage of higher soil moisture and after fallow periods when soil fertility is higher (Gerardo Moreno, personal communication, January 2008). When cereals are planted, the land is prepared by first disking the soil. This practice also eliminates competing woody species that invade through natural succession processes. Disking is also used to create firebreaks around individual fields.

Trees in the system have several effects on biophysical aspects of the Dehesa. First, they have a system of roots that penetrates deeper into soils (compared to forage or cereal crops), bringing water and nutrients up to subsurface horizons and improving the microclimate of the Dehesa (Jobbagy and Jackson, 2004). These effects on moisture and nutrients help extend the growing season for forage species around the trees. Prunings from oaks can also help feed animals in times when grass production is reduced during the dry season. The deep and extensive rooting ability of trees allows animals to feed on prunings while more shallow-rooted forage plants suffer moisture deprivation and reduction in net primary production (NPP). Shade from the trees also helps reduce animal weight loss due to water stress during the hot months (Moreno et al., 2007). A commonly published image of the Dehesa shows animals relaxing under the
dense shade of the evergreen oak trees. The presence of animals under trees has several effects that serve to reinforce the Dehesa system. Animal excreta, in the form of urine and feces, help mobilize and concentrate nutrients below tree canopies. Soil nutrients can thus be higher underneath tree canopies (Dupraz and Newman, 1997; Moreno et al, 2007). This may also be due to the improved microclimate for N fixing herbs. On the other hand, bulk densities can be greater with high animal traffic and accumulation of salts (from urine) can kill isolated trees in a pasture (Lynn Sollenberger, personal communication, April 2007). The maintenance of many trees helps to spread these negative effects throughout the field. Animals help maintain the biophysical integrity of the Dehesa by grazing and eliminating the natural successional process for plants. This key element must remain in place for the Dehesa system to persist. Dehesa lands left ungrazed will revert to Mediterranean woody shrubs and trees, called ‘matorral,’ which are sometimes managed for hunting deer, birds, and wild pigs. The production of cereal crops between trees also serves to maintain the floristic integrity of the Dehesa, but the lack of water and soil nutrients limits this practice to more fertile sites. Dehesas are generally not irrigated. Montero et al. (1998) maintains that cereal production is the least important aspect of the Dehesa, particularly on less fertile sites. Animals help to maintain the herbaceous pasture component by eliminating woody plant competitors. Regeneration of oaks in the Dehesas is thus limited if saplings are not protected from browsing.

The Dehesa system varies tremendously according to climatic variations found in central-southern Spain and Portugal. In climates with warmer winters, acorn production is of greater importance, and pigs are fattened during the winter months when acorns are present (Montero et al., 1998). In colder climates, tree biomass is harvested for firewood, cork, and charcoal, as acorn
production is lower and less frequent (Montero et al., 1998). Where acorn production is important, old large trees are maintained for decades, if not centuries.

Soils under the Dehesa systems are universally described as poor or lacking in fertility (Joffre et al., 1988; Gomez-Guitierrez and Perez-Fernandez, 1996; Joffre et al., 1999; Plieninger and Wilbrand, 2001; Hernanz et al, 2002; Gallardo-Lancho and Gonzalez-Hernandez, 2004; Plieninger et al., 2004; Moreno et al., 2007; Costa et al., 2008). The positive influence of individual trees on soil properties of the Dehesas is well known. Joffre et al. (1988) and Moreno et al. (2007) both report higher nitrogen, phosphorus, potassium (and cation exchange capacity, CEC) below tree canopies. Trees serve to concentrate nutrients from a large soil volume, depositing them by root turnover, leaf fall, and by attracting animals seeking shade and forage (Gomez-Guitierrez and Perez-Fernandez, 1996; Joffre et al., 1999). Animal excreta contributes to significant improvements in soil fertility. The improved soil fertility under trees in springtime helps to reinforce this cycle, as early growth of grass and herbs leads to preferential grazing underneath trees early in the growing season (Joffre et al., 1988). Summer heat also drives animals to seek shade under these evergreen oaks (personal observation, June 2006). These effects, though, are limited by the low density of some dehesas. Moreno et al. (2007) found the positive effects of individual *Quercus ilex* trees (20 stems ha⁻¹) to extend only slightly beyond the tree canopy, representing only 15% of the land area. Dehesas may occur only on poor soils, as sites of better quality would likely be used for more rotational cropping and grazing, as it represents a more profitable land use than just grazing alone (Plieninger et al., 2004). Soil type and texture are important to Dehesa productivity, as sandier soils were found to support higher cork production in *Quercus suber* in Portugal across several soil types (Costa et al., 2008). SOC in the Dehesas is driven by mean annual rainfall, as Gallardo-Lancho and Gonzalez-Hernandez
(2004) demonstrated on three sites of varying precipitation in southern Spain. Another positive effect on SOC is soil aggregation, which increases on no-till Dehesas. Hernanz et al. (2002) found almost double the water stable aggregates (% weight) in no till versus tilled Dehesa sites in central Spain. Overall, Dehesa soils are considered poor in fertility, but at the same time, they support and extensive (5 million hectares) and robust system of production (Dupraz and Newman, 1997).

**Socioeconomic Considerations of the Dehesas**

The Dehesa is an ancient system of agricultural production that dates back to before the reconquest of Spain by Christians some 800–1000 years ago (Montero et al., 1998, Joffre and Ramball, 1999). Almost all of the Dehesa lands are owned privately, and as such, their management is dependent on the decisions made by individual landowners. Regeneration of tree species has become a serious concern for the preservation of the Dehesa, as ageing stands of oak are rarely replaced by new trees. Disease in the oaks has also become a problem, reducing tree cover in the Iberian Peninsula (Brasier, 1993). The Dehesa system was developed from natural stands of existing trees several hundred years ago. Montero et al. (1998) estimate that it would take 120 years to fully regenerate a Dehesa system. Current Dehesa landowners were likely not alive during the establishment of the uniquely old production system they own and manage. Consequently, they may not understand the importance of tree regeneration, and do not wish to have any portion of their grazing lands utilized for ‘other’ purposes, i.e. taken out of production for tree regeneration (Gerardo Moreno, personal communication, January 2008). Most landowners are not interested in such long term investments, and may not fully understand the role tree regeneration plays in the whole system over time.

Another problem that reduces management quality of the Dehesa system is Spain’s rising property values. Investors can earn money just owning a property, as capital gains are made from
increasing land values. This sometimes leads to poor management, and landowners that are less interested in economic production of the Dehesa agricultural system. Moreover, the Dehesa system is an attractive land use system, and it is recognized as a national source of pride for those of the Iberian Peninsula. Media produced by the travel industry for this part of Spain often uses images of the Dehesa landscape. The current king of Spain, Juan Carlos I, enjoys hunting on Dehesa lands. These cultural attributes, in some cases, may help sway landowners towards maintaining these traditional landscapes and the system of agriculture they support.

Silvopasture as a Land Use Intervention in Northern Spain

There is a broad swath of land in northern Spain that is bathed in moist currents from the Atlantic Ocean, keeping this portion of the Iberian Peninsula green with trees and agricultural production. Upon viewing a satellite image of Spain, one will notice this difference in landscape mostly in summer, when the rest of Spain is dry and brown. Northern Spain contains the most productive forest land in the country, due to its higher average precipitation, less intense dry seasons, and generally cooler temperate maritime climate (Rodriguez-Murillo, 1997). In more southern regions of Spain, a Mediterranean climate dominates, with a more pronounced dry season, which limits primary production to the wetter winter months when temperatures are lower. Of the four northern provinces of Spain, Galicia province (northwestern Spain) has been identified as the region in the country with the highest C storage in standing biomass, after forest land surveys in the early 1970s, and again in the 1980s, revealed standing forest C stocks (Rodriguez-Murillo, 1997). Land use in northwestern Spain over the past forty years has shifted from traditional row crops and pastures to the production of short rotation woody crops on nutrient poor acid soils (Merino et al., 2004, Zas and Alonso, 2002; Schmitz et al., 1998). Agricultural land has been abandoned. Tree plantations of Pinus spp. and Eucalyptus spp. (as well as other exotics) have been planted on abandoned agricultural lands, and regional public
opinion on the plantations is mixed, especially where clear-cutting harvest systems are utilized (Schmitz et al. 1998). Soils after afforestation of agricultural lands in Galicia are described by Zas and Alonso (2002). After examining 186 sites that had been afforested with Pinus spp., Eucalptus spp., Castanea spp., and Quercus rubra, they found mostly acidic (4.9 pH), sandy (68.4% sand), high organic matter (12.7%) soils that had developed on granites, schists, slates, and sedimentary parent materials. Mean precipitation was the dominant factor in soil level of development, with regional range of 600 – 2,500 mm for the province.

Recent research has elucidated some useful information about the effects of afforestation on soil nutrient characteristics in northwestern Spain (Merino et al., 1999; Rigueiro-Rodriguez et al., 2000; Mosquera-Losada et al., 2001; Zancada et al., 2003; Merino et al., 2004). Merino et al. (1999) and Merino et al. (2004) demonstrated the negative effects on soil properties for site preparation and harvesting of Pinus radiata on hilly sites in Galicia. Removal of whole trees and the humus layer increased soil erosion and loss of nutrients up to nine years after harvest. Rigueiro-Rodriguez et al. (2000) and Mosquera-Losada et al. (2001) reported that biosolid application on a simulated silvopasture (Pinus radiata and Dactylis spp./Trifolium spp. combined) had positive effects on tree and pasture growth. Zancada et al. (2003) found relatively higher soil C accumulation belowground under Eucalyptus plantations, as compared to native forests dominated by Quercus spp.

The silvopastoral system of production is one possibility for a land use intervention that may help to reduce some of the negative effects of land use change in northern Spain over the past 40 years. A more intensively managed silvopastoral system, one able to produce greater amounts of tree and forage biomass as compared to treeless pastures, is possible for parts of northern Spain (Mosquera-Losada et al., 2005). While the productive benefits of silvopasture
systems are well known, particularly in areas of Dehesa-dominated Spain and Portugal, the additional benefits of soil C sequestration is a less known environmental service that deserves greater attention.

**Climate Change, Carbon Sequestration, and Silvopasture**

Over the past fifteen years, there has been a proliferation of interest in sequestration of C in terrestrial soils and vegetation (Dixon et al. 1994; Batjes, 1997; Lal, 2004; IPCC, 2000; Post and Kwon, 2000; Ingram and Fernandes, 2001; Montagnini and Nair, 2004). The world’s interest in C sequestration grew from a political process that has progressed significantly over the past 20 years. The worldwide scientific community has recognized that the buildup of CO₂ in the atmosphere is having a significant effect on the earth’s climate, increasing temperatures and shifting precipitation patterns on a large scale, among other effects. Carbon dioxide is the greatest contributor to global warming, among the so called green house gases, and efforts are underway to reduce its atmospheric levels by enhancing and protecting C sinks. Soil C stored in terrestrial ecosystems (2,300 Pg) outweighs both plant C (560 Pg), and atmospheric C (760 Pg) combined (Dixon et al., 1994; and Batjes and Sombroek, 1997; Lal, 2004; Lal, 2008). While forests and other vegetation maintain significant C stocks above ground, soils play a dominant role in storage of the world’s terrestrial C below ground.

Beginning in the late 1980s, international organizations, including the United Nations (UN), became interested in addressing the growing problem of global warming. First, a global political consensus was gathered by the UN Intergovernmental Panel on Climate Change (IPCC), which culminated in the creation of the United Nations Framework Convention on Climate Change (UNFCCC) at the Rio Earth Summit in 1992. Since Rio, UNFCCC parties (representing all UN member states) continued to gather international support to move nations towards an agreement on reducing greenhouse gas emissions. The 1997 Kyoto Protocol was the first attempt
to set individual country emissions targets in order to reduce emissions of greenhouse gasses globally. The protocol recognized the responsibility of developed nations for historic emissions of greenhouse gases since the mid-nineteenth century, as industrialization spread over Europe and North America. Agricultural expansion (with its associated deforestation and soil tillage) was responsible for emissions of carbon dioxide into the atmosphere historically, but during the 20th century, the burning of fossil fuels surpassed land use change in CO2 emissions (Houghton, 1994). While the Kyoto protocol was originally envisioned as a measure to reduce fossil fuel use, follow up agreements to Kyoto allowed for forests, agriculture, and soils to be included in a country’s C accounting (UNFCCC, 2006, Lal, 2008). Soil organic carbon can be included in a member state’s C accounting, but the difficulty in measuring SOC, with varying methods used (Guo and Gifford, 2002, Nair et al., 2009) and issues related to monitoring of sequestration projects make this accounting problematic. As SOC represents a significant C pool in terrestrial ecosystems, future international agreements to combat climate change must include improved measurement and monitoring of SOC. In February 2005, with the accession of Russia, the Kyoto protocol went into effect with provisions for SOC inclusion in C accounting. Nonetheless, significant gap in measuring and monitoring SOC still exists.

As one of the original signatories of Kyoto, Spain had committed to reducing C emissions by various target dates over the next twenty years. Although Spain is a developed nation and member of the European Union with the financial and technical capacity to conduct an assessment of its terrestrial C stocks, data on forest C stocks, in Spain, particularly in soils, are minimal (Rodriguez-Murillo, 1997). In order to enact land use changes that will reduce CO2 emissions, the Spanish government must understand how land use affects long term storage of C in its vegetation and soils (Montero et al., 2005).
Carbon Sequestration Opportunities in Agroforestry

The enhancement of C ‘sinks’ and reduction in C ‘sources' are the two ways that Kyoto signatories can offset C emissions. Lal (2004, 2008) identifies several measures nations can take to improve C sequestration on their lands, including: woodland regeneration, no-till farming, cover crops, manuring, sludge application, and agroforestry, among others. Agroforestry, in particular, has the potential to provide C sequestration services, in addition to a host of other benefits (Nair and Nair, 2003; Montagnini and Nair, 2004 Haile et al., 2008; Takimoto et al., 2009; Saha et al., 2009, Nair et al., 2009). Agroforestry is defined as the purposeful planting of trees and crops (including forage) in combinations for multiple benefits on the same land, and silvopasture is a prime example of this type of system. Trees may also be planted in temporal succession with crops such as in fallow systems, and agroforestry systems may include an animal component as in silvopastoral grazing systems. As an example, planting trees as a windbreak can help to reduce wind speeds and crop water stress in traditional row cropping systems. The conversion of traditional agriculture (treeless pastures and row crops) to agroforestry would not only enhance C sequestration with increased standing and below ground biomass, but also reduce pressure on native forests for conversion to other uses (Nair et al., 2009). While Kyoto protocol signatories search for innovative ways to reduce C emissions, agroforestry in the tropical and temperate zones of the world has the potential to provide food, timber, fiber, medicine, and an aesthetically pleasing land use, in addition to C sequestration, improved water quality, biodiversity protection, and other environmental services.

Sharrow and Ismail (2004) demonstrated the additional C sequestration benefits from silvopastures as compared to pure pastures and forest plantations in Oregon, USA:

Agroforests could have efficient carbon and nitrogen sequestering over time because they have both forest and grassland storage patterns active. This would manifest itself as below ground storage by a vigorous grassland component in very young agroforests being
gradually augmented by storage in long residence time woody material and semi-decomposed surface organic matter (duff) in older agroforests.

The combination of trees with grasses in silvopasture helps to recycle soil nutrients as these annual pasture species translocate a relatively large proportion of C to below ground, as compared to cereals that were bred to fill above ground grains with C rich photosynthate (Kuzyakov and Domanski, 2000). Sharrow and Ismail (2004) demonstrated that silvopastures sequester more C (above and below ground) than tree plantations (740 kg ha\(^{-1}\) yr\(^{-1}\)) or pastures alone (520 kg ha\(^{-1}\) yr\(^{-1}\)). Haile et al. (2008) showed that SOC was 33% higher in a Pinus elliotii + Paspalum natatum silvopasture as compared to adjacent pastures across several sites in Florida, USA.

The pathway for organic C to become sequestered in soils is through the input of organic matter (OM) from plants, either from root exudates (in a process called rhizodeposition) and above ground from plant litter decomposition and incorporation into the mineral soil. Soils are also amended with manure, mulch and other sources of plant matter. About 50% of OM is C, and as such, the transfer of OM to the soil is an important process when considering terrestrial C cycles. OM enhances soil fertility in many ways by improving soil moisture and cation exchange capacity, and by the formation of stable aggregates for improved soil structure, among others. As plants input OM into the soil, soil microorganisms break down C bonds and release C to the atmosphere by oxidation and release as CO\(_2\) gas. While most of the C input to the soil is lost by oxidation, some is held in the soil for the long term. Enhancing this long term storage component of soil C is of great significance, as it represents a C sink that, in some cases, can last from decades to centuries (Parton et al., 1987, Ussiri and Lal, 2005)


Long Term Storage of Organic Carbon in Soils

Reviewing the processes that lead to long term storage of SOC in the soil profile, Christensen (2001) and Ussiri and Lal (2005) identified and defined three mechanisms for long term storage of SOC. First, organomineral complexes can promote physical protection of SOC within soil aggregates, or in micropores. Second, SOC is protected by its biochemical recalcitrance, as the chemical composition of the organic substance is difficult to degrade, such as in high lignin content materials. Third, the chemical stabilization of SOC results from the interaction of organic C with minerals in soil, such as adsorption to clay surfaces (exchange complex).

For each of these mechanisms, the input of OM into the soil is the starting point for long term sequestration of SOC. Management practices that improve the input of OM into the soil will generally lead to improvements in long term SOC storage. Maintaining crop and tree residues, plant material leftover from harvest, is one way producers can improve soil C inputs (Lal, 2004; Merino et al., 2004). Initiation of no-till practices on previously tilled soils is another way to reduce SOC losses. Six et al. (2000a) and Gonzales and Laird (2003), among others, have shown that no-till farming helps to preserve soil aggregate structure and reduce oxidation of surface soil horizons. The maintenance of microaggregates held within macroaggregates, in particular, has been identified as one way to improve C storage (Six et al., 2002a). Larger macroaggregates shield microaggregate-held C from microbial attack (and decomposition) as they are formed and maintained over time. Tillage has been shown to reduce C-rich macroaggregates and increase C-depleted microaggregates (Six et al., 2000a). The maintenance of stable microaggregates within macroaggregates in no-till agriculture and forest systems leads to greater storage of SOC as compared to unprotected SOC (Six et al., 2000a). Six et al. (2002b) have shown that soil size fractionation separates the soil into particle sizes that each contain SOC that is varyingly

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sensitive to land use, management, and climate. Also, each size class demonstrates differences in C residence time. The binding agents that serve to form and maintain aggregates help us to understand the residence times associated with each size class (Christensen, 2001). Macroaggregates are formed with newly incorporated OM, and smaller sized aggregates, and bound with fine roots, fungal hyphae, and plant and microbial residues. Carbon tracer studies have shown the redistribution of C from macroaggregates to microaggregates over time, as microaggregates are formed within macroaggregates (Six et al., 2002a). The binding agents that form microaggregates are microbial polymers, root exudates, and polyvalent cations. Silt +clay sized aggregates contain little occluded OM, and mineral sources become important binding agents (organomineral complexes). Mean residence times for macroaggregates, microaggregates, and silt+ clay sized aggregates vary from 1 – 10, 25, and 100 – 1000 years (Parton et al., 1987; Schimel et al., 1994). As one considers the three aggregates sizes in descending order, C:N ratios and occluded OM decrease, as binding agents shift from biotic to more mineral sources (Christensen, 2001). Where soil OM is the main binding agent, a hierarchy of protection has been demonstrated as smaller aggregates are stored in increasingly larger size, as silt+clay sized aggregates are protected within macroaggregates that are protected within macroaggregates (Tisdale and Oades, 1982; Six et al., 2000b). Since mineral complexes are generally very stable in the soil, the protection of this hierarchical stabilization mechanism will promote long term C storage in the soil. The protection of macroaggregates, in particular, helps to protect C stored in smaller aggregate sizes.

To obtain soil fractions (with similarly sized aggregates) by wet sieving, soil is first placed in water, in a process called slaking, which preserves only water stable aggregates (Elliot, 1986). The soil is then sieved to different fraction sizes (≤53 μm, 53 – 250 μm, 250 – 2000 μm). Each of
these size classes coincides with silt + clay size aggregates, microaggregates and macroaggregates, respectively. Using this technique, Six et al. (2000b) found significantly greater SOC in the microaggregate and microaggregate size fraction in native grasslands as compared to adjacent tilled sites on four Midwestern American sites. Six et al. (2002b) have also shown that significant, long term SOC storage occurs in the smaller aggregate size class in soils. Haile et al. (2008) found significantly higher SOC in the macroaggregate fraction on a slash pine (Pinus elliotti) + Bahiagrass (Paspalum notatum) silvopasture on as compared to immediately adjacent open treeless pastures on two sites in Florida. Each study demonstrated a useful procedure for determining the proportion of relatively protected SOC. Estimating the total C content of these various soil fractions improves our understanding of SOC pools of varying residence time and the potential for long term C storage.

Other management techniques that can promote storage of SOC include soil amendments that either directly add C sources to soils, and/or promote the growth of plants on less fertile soils. Application of biosolid sludges (either from wastewater or other industrial processes) has been tested as a measure to increase OM on degraded sites such as on reclaimed mine soils. Ussiri and Lal (2005) report that amending degraded soils with biosolid sludges improves soil characteristics such as OM, CEC, soil nutrient status, moisture retention, and reduces bulk density. Application of biosolids can help reduce the negative effects of intensive farming and forestry practices such as deep soil ripping by replacing OM where it was lost due to exposure of soil profiles to oxidation (Merino et al., 2004). Rigueiro-Rodriguez et al. (2000) and Mosquera-Losada et al. (2001) reported on two sites in northwestern Spain where sewage and milk processing sludge were applied to Pinus radiata plantations and silvopasture, respectively, where biosolid application resulted in improved growth of tree and forage components. Mosquera-
Losada et al. (2001) observed that the use of biosolids as a fertilizer to silvopasture leads to the accumulation of heavy metals (such as zinc, lead, and cadmium) in soils and vegetation, with adverse effect on forage quality for animals. Crohn (1995) found similar results for regular application of 4 Mg ha\(^{-1}\) (dry mass) of municipal sewage sludge on a managed forest at the Hubbard Brook Experimental Forest in New Hampshire, USA. Although harvestable timber increased 26\% over controls, concerns for groundwater contamination by heavy metals remained a serious concern.

While the direct benefits of sludge application include improved soil characteristics and plant growth, research addressing the effects of biosolid application on C sequestration on agricultural lands is rare. Hartenstein (1981) demonstrated the characteristics of stable (non-labile) sludge, which was correlated with the higher concentrations of humic and fulvic acids. The humification of sludge in soil is the goal for increasing C sequestration, with reduced leaching of heavy metals as well as a reduction in bad odors. Biosolids, while promising as a soil amendment, require careful consideration of the negative effects of high concentrations of heavy metals in the soil and vegetation. It may be though, another mechanism for increasing long term storage of SOC. Additionally, municipalities that must treat and dispose of sewage have a strong incentive to find alternative disposal mechanisms, and will even pay to transport sludge to field application sites (Ussiri and Lal, 2005). The disposal of biosolids on silvopasture is yet another way that nutrient cycling loops can be closed, as the tree component takes up excess nutrients that are normally lost to (and thus a pollutant of) the ground water system. Biosolid application to silvopasture not only directly adds C to the soil, but has been shown to improve tree and forage growth by increasing soil nutrients, and given all other factors equal, increases inputs to the soil through increased litter deposition and rhizodeposition of tree exudates.
As NPP is a significant indicator of terrestrial C sequestration potential, management practices that improve growth of the overlying vegetation will eventually lead to generally greater storage of SOC. Application of fertilizers, as in biosolid application or traditional mineral fertilizer improves SOC storage. Other practices that include mulching, either with organic or inorganic materials, will lead to improved soil fertility, which, again, improves SOC storage (Ussiri and Lal, 2005). Mulch simulates a forest leaf litter layer and helps reduce moisture losses from soil, as well as directly provides nutrients from decomposition, incorporation, and mineralization of mulched materials. Liming is another example of a management intervention that will improve plant growth, as well as stimulate storage of SOC in soils. Low pH soils have limited capacity to grow plants and liming helps to raise pH and reduce toxicity of Al, Fe, and Mn (Ussiri and Lal, 2005).

Growing trees versus other less deeply rooted vegetation is a management technique that will help to improve storage of SOC, especially in deeper horizons. The deep rooting ability of trees helps them to maintain structural stability, as well as to exploit soil water and nutrients in deeper soil profiles. Sternberg et al. (1998) found significant, while heterogeneous, masses of tree roots at 4 m below the soil surface in the Brazilian Amazon. As deeper soil profiles are not as prone to common oxidation processes (fire, erosion, and other aboveground disturbances), the opportunity exists for SOC to be stored for the long term due to deep tree rooting. The obvious difficulty in examining SOC additions by deeply rooted trees (greater than 100 cm) makes studies such as Stone and Comerford (1994) and Sternberg et al. (1998) unique. Although this aspect is not well studied, the deep rooting ability of trees, in agroforestry systems in particular, is one of the key factors that help some production systems persist in dry conditions. The addition to SOC
pools in agroforestry systems to deep soil profiles is a major knowledge gap in agroforestry research (Nair et al., 2009).

While land use change in northern Spain during the past four decades have lead to increases in short rotation forest plantations of fast growing exotic trees, alternative land use trajectories have received less attention. Agroforestry, and silvopastures in particular, can help meet the need for timber (or pulpwood as is the case for northern Spain), in addition to providing food commodities. When considering the additional benefit of soil C sequestration, promotion of silvopastoral systems may be yet another way that Spain can meet its Kyoto CO₂ emission reduction targets. The development of these silvopasture systems will require researchers to consider the best ways to design and promote silvopasture, from choosing the right forage and tree combinations, to considerations of spacing, fertilizer applications, and social factors. Spain has a long history of use of silvopasture (as in the Dehesa), and the promotion of a unique, productive silvopasture for wetter conditions in northern Spain will foment the diverse production of agricultural and forest products (compared to treeless pastures and monoculture croplands), while at the same time help to meet its international political commitments. More research is needed to assess the potential long term storage of SOC in silvopasture in Spain. This, in turn, will provide useful information to land use planners who may be considering C sequestration through agroforestry practices as a mitigation technique to reduce atmospheric CO₂.

**Phosphorus Retention and Silvopasture in a Fertilized Landscape**

Phosphorus (P) is one of the most important elements for growth of plants. Phosphorus is a key element in the DNA of all life forms and it provides energy transfer capacity in photosynthesis to plants. In natural ecosystems, P is internally cycled within natural terrestrial systems, providing a limit on growth and reproduction of plants. On agricultural lands with
annual removals of biomass through harvest, P is constantly removed and thus must be supplied through fertilizers and other amendments such as manures and composts. Typical mineral fertilizers supply plants with nitrogen (N), P, and potassium (K) in proportions that match the growth requirements of a particular crop and/or stage of development for a crop. Application rates for these elements are typically dictated by N requirements of the target crop (Attiwill and Adams, 1993). When farmers choose to meet N requirements with manure inputs, there is a tendency to over fertilize with P because of high N to P ratios of such manures, leading to losses of P through leaching and surface flow (Sharpley et al., 2000; Chrysostome et al., 2007). Insurance fertilization also occurs, whereby farmers will over apply fertilizer to ensure high nutrient status of the soil. Excess N and P are blamed for the eutrophication of lakes and other water bodies, causing algal blooms that reduce drinking water quality, and cause massive fish die offs and outbreaks of cyanobacteria (Kotak et al., 1993; Sharpley et al., 2000). In order for this contamination to occur in a particular water body, two things must occur: there must be a P source from the soil, and transport of P to the water body (Sharpley et al., 2000). If high inputs of P occur on a field and there is no transport mechanisms (no water outlet), then no problem will occur. It’s the combination of these two scenarios (P source and a transport mechanism) that is responsible for loss of P from agricultural soils and the eutrophication of water bodies. Fertilizer, manure, and other amendments that input nutrients into the soil can become sources of P. Facility in P transport is related to several factors, including: soil texture, topography, climate, etc. P moves through the soil more slowly than N (Barber, 1995), and as such, is generally not an environmental concern. Exceptions to this occur on sandy, low clay soils where water can be transmitted quickly through macropores and other open spaces in the soil (Sharpley et al., 2000). In the Netherlands, sandy soils and intense dairy production have lead to P leaching and
eutrophication of fresh and salt water. Drinking water, recreation, biodiversity, and commercial fishing have all been negatively affected (Ver der Molen et al., 1998). Aarts et al (2000) estimated that 67% of P purchased through fertilizers for dairy farms in the Netherlands does not leave the farm in useful products (meat or milk), and ends up in soils, where buildup leads to eventually loss (32 kg ha\(^{-1}\) yr\(^{-1}\)). Soils of Florida, USA, are very sandy in a state where a mild subtropical climate promotes a major agriculture and forestry industry. Annual fertilizer applications on Florida’s sandy soils have lead to reductions in water quality (Graetz and Nair, 1995). The perfect combination of favorable crop growing conditions and sandy soils in Florida has, unfortunately, has lead to the contamination of pristine aquatic environments. Spain also enjoys a mild climate that is excellent for the production of many agricultural products in its diverse climatic zones. Water quality is also effected by excess fertilization in Spain, and P from agricultural lands, in particular, has been identified as a contaminant of water bodies (Ramos and Martinez, 2004; Fouz et al, 2009).

**Use of Indices to Assess Potential P Loss from Agricultural Lands**

The ability to access potential loss of P from agricultural lands must be specific to a region where soils have at least some common characteristics for P transport. The United States Department of Agriculture, Agricultural Research Service maintains a database of soil P indices to be used to access potential P loss from soils (USDA-ARS, 2008). These indices are used to help farmers assess whether or not there is potential for P loss on their field, and guide them in proper fertilization applications. The combination of P inputs and high potential for loss in Florida sandy soils has driven the development a P- index. On November 13, 2000, the USDA, in consultation with a group of scientists from the Phosphorus Index Core Team (PICT), officially adopted the Florida P-index as a risk assessment tool for land use planners to address loss of P from agricultural lands (UF-IFAS, 2008). The P-index ‘tool’ is a worksheet that is
easily used by land managers, without need for a laboratory, to determine potential P loss risk. The worksheet is divided into an assessment of P transport due to site characteristics and management, each contributing to a general assessment of risk. Site characteristics such as topography, erosion, and proximity to water bodies are used to determine the P loss potential due to site characteristics, and fertilizer application rates and timing, and water management are factors that determine P loss due to management. Given the results from the Florida P Index, a land use planner can better manage fertilizer applications to reduce potential losses, improving water quality in the state.

For a better technical assessment of P loss potential, more precise indices have been developed to address P contamination. In the Netherlands, the degree of soil P saturation (DPS) concept evolved, which is a comparison of P saturation from the surface soils to the dissolved P in the solution that is leached from that profile (Del Campillo et al., 1999; Chrysostome et al., 2007). This is mostly used to assess soil subsurface losses of P using oxalate as an extracting agent. The ratio of oxalate extracted P to Al and Fe is used to calculate a percentage that, if relatively low, demonstrates lower risk for P loss. Since P forms highly insoluble complexes with Fe and Al oxides, the presence of Fe and Al in the soil is a strong indicator of the P sorptive capacity of the soil (Giesler et al., 2005). Nair et al. (2004) adapted the DPS concept to use Mehlich 1-P, Al, and Fe to calculate the Phosphorus Saturation Ratio (PSR) for Florida:

\[
\text{PSR} = \frac{\text{Mehlich } 1-P}{\text{Mehlich } 1(\text{Fe} + \text{Al})}
\] (2-1)

A ratio of greater than 0.15 indicates that soils are saturated with P, and that leaching may occur (in conjunction with of the factors such as facility of P movement to water bodies). This ratio is determined by increases in water soluble P that occur as the PSR increases. It represents a change point (0.15) where increased P saturation leads to P leaching in the soil solution. Taking
the PSR concept a step further, Nair and Harris (2004) estimated the Soil Phosphorus Storage Capacity (SPSC), the amount of P that can be safely added to a given soil without environmental risk:

$$\text{SPSC} = (0.15-\text{PSR}) \times (\text{Mehlich 1-Fe/55.8} + \text{Mehlich 1-Al/27}) \times 31 (\text{mg P/kg})$$

This tool provides a useful approximation of the “safe” P storage capacity of the soil that reflects past P applications and potential for P contamination, making it relatively easily to calculate additional amounts of P that can be applied to a soil without risking loss from the system. Negative SPSC values (i.e. when soil PSR is >0.15) indicate oversaturation of P in the soil, and an increased likelihood that the soil will be a source of contamination.

**Silvopasture to Reduce P Losses from Sandy Agricultural Soils**

Florida and Spain both have the mixed blessing of a mild climate excellent for growing many crops, and in several places, sandy soils that have a reduced capacity to hold moisture and retain nutrients. As such, producers must carefully manage fertilizer applications in order to avoid low fertilizer use efficiency and/or environmental contamination. One measure to remove buildup of P in the soil profile is the addition of trees to the agricultural landscape (Allen et al., 2006; Nair et al., 2009). In particular, silvopasture agroforestry practices have been recognized for their role in regulating the loss of P from agricultural lands (Nair et al., 2007a; Nair et al., 2007b; Michel et al., 2007). Agroforestry systems in general are known for their nutrient recapturing ability, whereby fertilizers that move down the soil profile out of reach of traditional agronomic crops, maize, wheat, etc. are recaptured by deeply rooted trees in the system (Nair et al., 1999). Soil nutrients that would have been lost to the system, perhaps contaminating local water bodies, are thus retained. Researchers from the University of Florida have recently shown, utilizing the SPSC as in indicator, that traditional treeless pastures demonstrate significantly less capacity to retain P in the soil as compared to silvopasture on immediately adjacent sites.
Removal of excess P from the soil by trees leads to greater storage capacity in the soil, lessening the risk associated with loss of P on the agricultural landscape. Nair et al. (2007a) found significantly higher water soluble P (WSP) concentrations underneath treeless pastures of bahiagrass (*Paspalum notatum*) as compared to native and slash pine (*Pinus elliottii*) and bahiagrass silvopastures on sandy Spodosols. The spodic horizon in Spodosols soils can serve as a source or sink for P, depending on management history (Graetz and Nair, 1995). Michel et al. (2007) also demonstrated lower WSP in the silvopastures compared to treeless pastures on Spodosols and Ultisols on three sites in Florida. Evidence from several researchers indicates that, given the difference between treeless pastures and silvopasture, silvopasture is a preferred land use, if reduction of P losses from the soil is a management goal. In order to draw more general conclusions about how silvopasture implementation affects P status in sandy soils, investigation in other sites is necessary. Using the indicators of P saturation and storage potential as applied to Florida, the Netherlands, and elsewhere, this study will address how silvopasture agroforestry practices in two different climatic regions of Spain affects P losses from agricultural lands.

**Conclusion**

The management of C and P in agricultural landscapes has recently become a priority for land managers as they attempt to improve terrestrial C sinks and reduce C sources to reduce global warming and improve water quality that is affected by nutrient losses from agricultural lands. Implementation of tree-based land uses, and silvopasture agroforestry practices in particular, is believed to improve both carbon sequestration in deep soil horizons and P retention in fertilizer amended soils. Several researchers are now working to elucidate the soil C and P dynamics under silvopasture agroforestry as a land use alternative that maintains two systems of production, a fast growing forage component combined with long lived trees, for the amelioration of soils that have lost C through cultivation and seen the buildup of P through poor
nutrient management. The fractionation of soils into different sizes based on discrepancies in mean C residence time allows us the opportunity to access how different land uses affect pools of SOC that are relatively resistant to decomposition, and thus contribute to long term soil C sequestration. The promotion of land uses that preferentially protect and promote long term storage of C will reduce atmospheric levels of CO₂ and thus, the greenhouse effect. The development of P saturation indicators for soil gives land managers a practical tool that can be used to assess whether or not a soil has the propensity for contributing to poor water quality (the eutrophication of rivers, lakes, etc.). Based on these soil P saturation levels, one can calculate the storage capacity that a specific soil has to hold P in forms that are not easily lost to the system, thus contributing to proper management of fertilizers and animal wastes. The implementation of silvopasture on soils has been shown to reduce P losses from highly amended soils, on sites where the potential for soil P loss is high (sandy soils). Use of soil physical fractionation techniques and P saturation indicators has given us a greater understanding of nutrient management on agricultural lands.

Land use in Spain is changing, as poor management and/or abandonment of agricultural lands has created the possibility for alternative land uses in northwestern Spain, and reinvigorated interest in maintaining an ancient system of production as in the Dehesa lands of south and central Spain. Spain has a significant agricultural sector, and improving environmental aspects of production is of great interest. Reducing soil C emissions from land use and land use change will help Spain meet its political commitments as a Kyoto protocol signatory, reduce the country’s effect on global climate change, and improve soil fertility through improved organic matter management. Loss of P from agricultural lands in Spain has been identified as a cause of poor water quality, and its retention within productive agricultural systems helps increase
fertilizer use efficiency and increase harvest yields. The use and adaptation of soil fractionation techniques and P saturation indicators on silvopastures as compared with treeless pastures in Spain will help producers, regulatory agencies, and researchers more fully consider the options they have for environmental amelioration on agricultural lands.
CHAPTER 3
CHARACTERISTICS OF STUDY SITES

Introduction

Two contrasting silvopastoral study sites were selected to evaluate C sequestration and P retention in two climatic zones that are representative of Spain’s varied ecosystems (Figure 3-1). The St. Esteban evergreen oak Dehesa is a working farm that is located in south-central Spain (Extremadura province), with a Mediterranean climate that is typical of the southern Iberian Peninsula. Annual summer dry seasons in this part of Iberia promote the development of sclerophyllous vegetation that is well adapted to survive extended periods with low precipitation (Kurz-Besson et al., 2006). This site is likely on the lower end of land use that can be used for any type of agriculture in Spain due to dry conditions. The second site, the Castro de Rey farm, is located in cooler and wetter northwestern Atlantic Spain (Galicia province), where summer dry periods exist, but are much shorter than in the Mediterranean, promoting broadleaf forests that are more typical of parts of Europe further to the north. The site is located on a farm that is in part managed as a simulated silvopastoral experiment, with several treatments and two locally utilized tree species. Mean precipitation in this part of Spain (1,350 mm) is more than double that which is found in the Mediterranean site. NPP is highest in this Atlantic region, and the vast majority of forestry production occurs in this northern Atlantic region of Spain (Mosquera-Losada et al., 2006). As such, the two sites selected represent some of the lowest and highest agricultural production potential areas in Spain. A comparison of the environmental services they provide will help demonstrate how low and high production silvopastoral systems contribute to environmental amelioration.
Dehesa Silvopasture at the St. Esteban Farm

The St. Esteban farm is a privately owned oak Dehesa silvopasture located approximately five kilometers south of the town of Plasencia in the Municipality of Malpartida de Plasencia in Extremadura province, central-western Spain (Figure 3-1). The farm occupies a relatively flat upland area, with steeper slopes towards the northwest, and a gentler ~2-3% slope towards the east (Figure 3-2). Provincial road EX-208 forms the eastern border of the property and provides an access point at its northern-most point.

The St. Esteban farm is managed for cattle, timber, cork, and acorn production. Cattle are present in the system, returning organic matter to the soil through excreta and maintaining a continuously open landscape by grazing grasses and browsing lower hanging branches on the oaks. The mix of *Quercus ilex* and *Quercus suber* provides an annual acorn “crop” from November to January, and cork can be harvested from the cork oaks every 9 – 12 years (Carvahlo-Mendes and Graca, 2009). When acorns are present, pigs are allowed to enter the farm (otherwise, their rooting activity damages pasture). The ham from these animals is the most highly prized and expensive in Spain, known as Jamón de Bellota [ham from acorns]. Cork is harvested from the cork trees as men with specialized knives climb ladders to remove the spongy bark by hand. Mostly, the cork is used for tapping of wine bottles (Carvahlo-Mendes, and Graca, 2009). Both oaks in this system are pruned to four branches to maximize acorn production and to homogenize tree canopy size, and the larger sized prunings are used as fuel wood (personal observation, January 2008). In this region of Spain, there is a demand for wood-fired ovens for bread and pizza and the dense oak wood makes an ideal fuel. During the study period, 2007 – 2008, tree canopies were rarely greater than 10 meters in diameter, regardless of tree diameter at breast height. It is estimated that the mature trees at St. Esteban farm are over 100 years old, and there is evidence of recruitment after an initial establishment, as a few smaller trees are present.

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This farm has no record of fertilizer application. Pasture improvement activities are also unknown for this site.

**Climate (Mediterranean)**

The climatic patterns at the St. Esteban farm are influenced by the nearby Atlantic Ocean and Mediterranean Sea, as onshore flows move unimpeded from the south and west through lowland Portugal. Summers are long and dry and winters mild and wet (Figure 3-3). Most of the annual precipitation (78%) falls between October and April, with an annual range of 7 to 87 mm of rain falling in July and December, respectively, and the annual mean total precipitation is 523 mm (AEMET, 2009). Mean monthly temperatures range between 7.9°C in January and 25.8°C in July, with an annual mean temperature of 16.1°C (AEMET, 2009). Evapotranspiration often exceeds precipitation in a given year.

**Soils at St. Esteban**

Soils on the St. Esteban farm are classified by the United Nations Food and Agriculture Organization (FAO) system as Cambisols or Luvisols, (United States Department of Agriculture (USDA): Alfisols). These upland soils are shallow, with depth to a slate bedrock sometimes less than 100 cm. Organic A horizons (to ~20 cm) were less developed, and well developed argillic horizons began below 50 cm and continued to weathered rock. Inorganic accumulations of carbon are not found in these acidic soils. Relatively low in fertility, with limited moisture and nutrient storage capacity, these soils support seasonal pasture growth and a low density of trees. Soils of better quality (greater depth and nutrient status) would likely be used for annual row crops without trees, and inferior soils would be used for occasional grazing (Gerardo Moreno, personal communication, January 2008). For centuries, these poor quality soils supported a robust agricultural production system that is an example of sustainable production with a limited resource base.
Simulated Silvopasture at the Castro de Rey Farm in Galicia, Spain

Located one km west of highway LU-113, the Castro de Rey farm is owned and managed by the local provincial government and is used for cattle, sheep, and hog production (Figures 3-1 and 3-4). The farm is utilized by the University of Santiago de Compostela for field laboratories, demonstrations, and field research. In 1995, the Crop Production Department at USC installed a simulated silvopasture experiment on a 1.6 hectare plot, which was formerly under potato (Solanum tuberosum) cultivation at the Castro de Rey farm, with the purpose of examining several aspects of regional silvopasture management (Rigueiro-Rodriguez et al., 2000). As a simulated silvopasture experiment, no animals are pastured on the plots, but annual forage removals were simulated by mechanical harvesting in order to determine understory production (Fernandez-Nunez, 2007). To assess adaptability for silvopastoral use, two species of trees common in Atlantic Spain were planted: Monterrey or Radiata Pine (Pinus radiata) and European Birch (Betula alba). In each plot, 25 individuals of each tree were planted in five rows of five in two different spacings, 2x2 m and 3x4 m, forming two sizes of plots at the farm (64 m² or 192 m² in area). All experimental measures were completed within the interior 3x3 grid of trees (N=9) within the 5x5 parcels, in order to reduce edge effects. Plots were fertilized with one of three different treatments (Mosquera-Losada et al., 2006):

- Mineral fertilizer alone, according to local recommendations of 500 kg of 8:24:16 (N: P₂O₅:K₂O) in March with 40 kg calcium ammonium nitrate (CAN) applied in May of each year
- Biosolids fertilizer: dairy residue sludge application made in the establishment year (1995) and then mineral fertilizer applied annually since 1998
- No fertilizer application

Half of the total plots at the Castro farm were planted with perennial ryegrass (Lolium perenne) and the other with cocksfoot (Dactilo glomerata). Ryegrass was competed out of existence relatively fast (<1 year), and as such, plots planted with this species were not included.
in this study. Combinations of each tree, spacing, and fertilizer treatment were replicated three times.

**Climate (Atlantic)**

Northwestern Spain is bathed in the moisture of the Atlantic Ocean currents that bring relative warmth to high latitude regions of Europe (Figure 3-5). Winters are long and wet, with mild dry summers. Oak (*Quercus* spp.), pine (*Pinus* spp.), and eucalyptus (*Eucalyptus* spp.) grow well in this relatively verdant region of Spain. The vast majority of forestry production in Spain occurs on this northern Atlantic strip (Mosquera-Losada et al., 2006). Soil can become moisture-limited in fall and then excessively water saturated in winter/spring; record drought conditions and then overland water flow were both observed in the silvopasture experimental plots in the study period of 2007 – 8. Mean precipitation at the Castro de Rey farm is 1350 mm (range of 40 – 140 mm month$^{-1}$, July to December) and the annual temperature is 12° C (monthly range of 6° C to 18° C from January to July).

**Soils at Castro de Rey**

Soils at the Castro de Rey farm developed along the relatively flat riparian floodplain that feeds the nearby Miño River. A gentle ~1% slope feeds a small stream to the south that joins the Miño River less than 2 km from the farm towards the west. Soils formed on sedimentary materials that have been dispersed over time by the nearby Miño river. Soils are defined as Umbrisols according to FAO (USDA: Inceptisols), with an A horizon of 32 cm in depth, with some exceeding 40 cm. Argillic horizons began at a mean depth of 58 cm, According to the USDA soil classification system, these soils are Inceptisols, with some horizon development, the eluviation of clay-sized particles to deeper horizons. These acidic and seasonally wet soils do not have accumulations of inorganic carbonates. At the beginning of the silvopasture experiment, the soils were near neutral pH (6.8) in the upper horizon (top 25 cm) and were managed for potato...
crop (Mosquera-Losada, 2006). Over a ten year course of the experiment (1995 – 2005), pH was reduced to a range of 5.5 to 6.0 in the various treatment plots in the 0-25 cm depth, and those planted with *Pinus radiata* demonstrated generally lower pH than those planted with *Betula alba* (Fernandez-Nunez, 2007).

**Simulated Silvopasture Experiments**

Results from the Castro de Rey silvopasture experiment have shed light on several aspects of silvopasture management, from understory pasture development and tree growth under various fertilizer regimes (including biosolids), to changes in soil chemical properties. Rigueiro-Rodriguez et al. (2000) demonstrated the efficacy of the fertilizer treatments on pasture growth and composition, as well as on diameter and height of one of the tree species, *Pinus radiata*. In pasture composition, grass production was favored under mineral fertilized plots, and *Trifolium* spp. was found to be more prevalent under biosolids fertilizer, improving overall forage quality. Plots fertilized with either mineral or biosolids wastes showed significantly greater pasture production in the first two years of the experiment (under a *Dactylis glomerata* mix and a *Lolium perenne* mix understory), as compared with non-fertilized plots. Pine diameter and height was reduced in the mineral fertilized plots, as compared to biosolids and no fertilizer application. Rigueiro-Rodriguez et al. (2000) attributed this result to competition with fast growing pasture grasses that utilize mineral fertilizer nutrients more efficiently than two year old *Pinus radiata*. Plant-available nutrients from biosolids fertilizers were believed to be released more slowly than mineral fertilizer in the plots and cold spring temperatures, in particular, helped to reduce availability of nutrients to trees at this time. As the annual summer droughts has a smaller effect on soil moisture of deeply rooted trees, the more slowly released nutrients from the biosolids were able to continue fertilizing the trees while the pasture grass production was reduced. As such, use of biosolids for silvopasture fertilization demonstrated similar pasture production and
significantly greater growth of *Pinus radiata* when compared with mineral fertilization. While providing a useful function in agroforestry production, use of biosolids fertilizers also helps to dispose of large-scale amounts of waste from industrial processes. In this case, use of dairy residue sludge was shown to be just as effective as mineral fertilizers in pasture growth, and more effective in promoting tree growth (Rigueiro-Rodriguez et al., 2000; Mosquera-Losada et al., 2006; Fernandez-Nunez, 2007).

Mosquera-Losada et al. (2006) reported on changes in soil chemical properties in plots from the Castro de Rey silvopasture experiment after seven years of tree and pasture growth. Soil acidity increased over the course of the experiment, as the agricultural field became a silvopasture. Fertilizer treatments had a significant effect on soil pH to 25 cm, as mineral fertilizers maintained a high pH compared to biosolid and no fertilizer application. This trend was present in plots planted with *Lolium perenne* and in *Dactilo glomerata* grasses, but was only significantly different underneath *Lolium perenne*. Cation exchange capacity (CEC) was higher under fertilized plots, except underneath biosolid fertilizer plots planted with *Lolium perenne*. Calcium and sodium occupy ~75% of exchange sites in these soils to 25 cm depth, with significantly more aluminum and magnesium in mineral fertilized plots (*p < 0.05*). Trends in *Pinus radiata* diameter and height from Mosquera-Losada et al. (2006) were similar to those reported by Rigueiro-Rodriguez et al. (2000) on the same site. In plots planted with *Lolium perenne* and *Dactilo glomerata*, *Pinus radiata* growth in height and diameter was significantly less under mineral fertilization than with biosolid and no fertilizer. Annual pasture production showed a steady decline as trees became larger, only rebounding slightly after a pruning of *Pinus radiata* (to 2 m height) lead to more sunlight reaching the understory. This is an acceptable loss, as landowners utilizing this system may switch to pure silviculture management, removing the
animal element from the landscape. In order to continue with pasture production, a reduction in
tree density would be necessary. More than any fertilization treatment, the most important factor
driving pasture and tree production is rainfall, as a severe drought in 1997 lead to a significant
reduction in forage production (Mosquera-Losada et al., 2006; Fernandez-Nunez, 2007).
Fertilization with mineral fertilizer did help to maintain grass dominance in the plots, as shrubs
tended to invade plots fertilized with biosolids or those not fertilized. Overall, Mosquera-Losada
et al. (2006) demonstrated the continued utility of biosolids fertilizers in the establishment of
silvopastures in NW Spain. Fertilizers help to maintain botanical composition, reducing invasion
of dicotyledons that leads to competition for K. Thus, fertilizer management in silvopasture can
also reduce herbicide applications.

Fernandez-Nunez (2007) provides details on many aspects of the silvopasture experiments
at the Castro de Rey farm, which support what was reported in Mosquera-Losada et al. (2006)
and Rigueiro-Rodriguez et al. (2000). Detailed information about the development and growth of
the pasture and tree component (and their interaction) in Castro de Rey are given. Fernandez-
Nunez (2007) also provides estimates of above and below ground biomass and soil carbon using
this type of silvopasture. Carbon from aboveground biomass was estimated to a range of 22.61 –
99.58 Mg ha\(^{-1}\) for *Pinus radiata* and 6.17 – 15.86 Mg ha\(^{-1}\) for *Betula alba*. Root biomass was
estimated at ~30% of total biomass for both trees. Biomass was converted to carbon using a
conversion factor of 0.50. Soil bulk density was estimated at 1.1 g/cm\(^3\), and SOC was estimated
for the 0 – 25 cm soil depth (Fernandez-Nunez, 2007). Table 3-1 shows results of soil carbon
estimates from selected plots in 2005 at Castro de Rey farm (ten years after establishment).
Carbon storage in the top 25 cm is most in more in the 2x2 than 3x4 spacing for *Pinus radiata*,
but no difference between planting density is found in non-fertilized plots of *Betula alba*. 

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Reported ranges for soil carbon are 135.9 Mg C ha\(^{-1}\) for *Betula alba* with 3x4 spacing and mineral fertilizer to 202.26 Mg C ha\(^{-1}\) for *Pinus radiata* planted in 2x2 spacing with biosolids fertilizer (Fernandez-Nunez, 2007).

The botanical composition experiments at Castro de Rey are no longer as useful, as the pine trees now have a closed canopy and pasture production is very limited. Some of the original experimental factors no longer exist; the presence of *Lolium perenne* was eliminated after only one year of the experiment. While the original purpose of the silvopasture experiments at Castro de Rey has been fulfilled, the trials continue to provide useful information about growth of the two tree species under two different spacings with three fertilizer treatments. Data for surface soils to 25 cm (pH, Mehlich-3 extracts, etc.) has been recorded continuously since the inception of the experiment, providing a wealth of information on the development of soil properties under the plots. Continued study of the soils under these trials will provide important information about the underground processes occurring under several land-use options.

**Conclusion**

Two study sites were selected based on differences in production capacities in very different climatic regions of Spain. The evergreen oak Dehesa at St. Esteban farm represents land uses typical of southern Mediterranean Spain, while the managed silvopasture experiment at the Castro de Rey farm represents future land use options in cooler and wetter ecosystems in northern Spain. Several publications have resulted from the work at the Castro de Rey farm (Rigueiro-Rodriguez et al. 2000; Mosquera-Losada et al., 2005; Mosquera-Losada et al., 2006; Fernandez-Nunez, 2007), while the current study is the first to examine aspects of Dehesa silvopasture at St. Esteban. Both are silvopasture agroforestry systems with adjacent treeless pastures that provide a comparison of silvopasture influenced landscape versus traditional pastures. Spain has a long history of use of silvopasture in the Dehesa-dominated landscapes, and
A locally modified version of the practice is being promoted in northern Spain as an alternative to pure tree plantation. The following chapters will consider the soil C sequestration and P retention capacities of each system, given drastically different climates, productive capacity, species usage, and management history.
Table 3-1. Estimates of SOC to a depth of 25 cm from selected plots in 2005 at Castro de Rey Farm, Galicia, Spain.

<table>
<thead>
<tr>
<th>Tree Species</th>
<th>Spacing, m</th>
<th>Fertilizer</th>
<th>C, Mg C ha(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Pinus radiata</em></td>
<td>2x2</td>
<td>Mineral</td>
<td>182.48</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Biosolid</td>
<td>202.26</td>
</tr>
<tr>
<td></td>
<td>3x4</td>
<td>Mineral</td>
<td>130.48</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Biosolid</td>
<td>130.48</td>
</tr>
<tr>
<td><em>Betula alba</em></td>
<td>2x2</td>
<td>Biosolid</td>
<td>191.26</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No fertilizer</td>
<td>182.8</td>
</tr>
<tr>
<td></td>
<td>3x4</td>
<td>Mineral</td>
<td>135.9</td>
</tr>
<tr>
<td></td>
<td></td>
<td>No fertilizer</td>
<td>181.69</td>
</tr>
</tbody>
</table>

Source: Fernandez-Nunez, 2007
Figure 3-1. Study site location in Spain.
Figure 3-2. Images of St. Esteban Farm, Extremadura, Spain. A) Aerial photograph of sampling area. B) Photograph near sampling points.
Figure 3-3. Mean monthly rainfall and temperature for Cáceres, Spain.

Source: State Meteorological Agency of Spain (AEMET)
Figure 3-4. Images of Castro de Rey Farm, Galicia, Spain.  A) Aerial photograph.  B) Photograph of silvopasture plots.
Figure 3-5. Mean monthly precipitation and temperature for Lugo, Spain.

Source: State Meteorological Agency of Spain (AEMET)
CHAPTER 4
SOIL CARBON SEQUESTRATION UNDER SILVOPASTORAL SYSTEMS OF SPAIN

Introduction

Strategies for reducing atmospheric carbon dioxide include the creation, protection, and expansion of C reservoirs. Soil organic carbon (SOC) is a significant pool in the global cycle, representing roughly two-thirds of terrestrial C (Dixon et al., 1994; Batjes and Sombroek, 1997; Lal et al., 2007; and Lal, 2008). Afforestation, reforestation, and other tree-based conversion of row crop land or pasture helps to increase SOC, enhancing the soil’s potential to sequester C (Lal, 2004). The deeper rooting ability of trees, compared to treeless pastures, and the relatively greater recalcitrance of tree roots (Matamala et al., 2003; Blanco-Canqui and Lal, 2004) suggests that potential exists for C sequestration in deeper soil horizons with tree-based land use changes.

Agroforestry has been identified as a promising land use for the sequestration of C (Montagnini and Nair, 2004; Haile et al., 2008; Nair, et al., 2009; Nair et al. (in press); Saha et al., 2009; Takimoto et al., 2009). Montagnini and Nair (2004) suggested that planting just 5% of the cropland in the United States with windbreaks would sequester 58 Tg of C over the next twenty years. Agroforestry practices combine the benefits of trees and traditional crops on the same land, and may also be a preferred method for promoting soil C storage in soil horizons that are less affected by the oxidizing effects of tillage, fire, and erosion. Haile et al. (2008) demonstrated greater C storage in deeper soils in silvopasture compared to treeless pastures, growing on Spodosols in Florida, USA. Takimoto et al. (2009) found greater C storage in soils underlying improved tree-based practices such as live fencing and fodder bank agroforestry than in unimproved systems in the West African Sahel region of Mali. While tree-based agroforestry systems are a preferred method for aboveground biomass C sequestration as compared to treeless
pastures, there is also evidence that C storage in deep soil horizons is greater (Haile et al., 2008; Takimoto et al., 2008)

Spain is a signatory to the Kyoto Protocol, which seeks to limit emissions of greenhouse gases for industrialized nations (Bolin, 1998). As such, Spain has vested interest in reducing CO₂ emissions, improving energy use efficiency, and limiting national sources of C such as that which occurs with forest fires and land use conversion. Accurate accounting of soil C sequestration will enable Spain to determine if soils are a source or sink for atmospheric C. Current land use and vegetation types in Spain, including forests and agricultural lands, are major indictors of stored C (Rodriguez-Murillo, 2001). The Dehesa silvopasture production system has sustainably produced meat, fiber, cork and firewood for centuries. While the productive benefits of silvopasture agroforestry have long been known in central southern Spain, their secondary environmental benefits are more recently being considered (Joffre et al., 1999; Gallardo-Lancho et al., 2004; Moreno et al., 2007). Maintenance of Quercus-dominated Dehesas will help us realize the full productive potential of this ancient system. Because of disease, abandonment (increasing fire risk), and a local belief that trees reduce forage production, density of oaks in the Dehesa system has been lowered over the past century (Joffre et al., 1999). These causes have reduced the production from oaks, and lessoned the positive interaction between the grazing animal, tree, and forage components. Taking full measure of the benefits provided by these system components may, in part, help preserve a unique and robust system of production in Mediterranean Spain.

Land use in the northwestern province of Galicia, Spain is changing rapidly, as traditional row crops and pastures are being converted to mostly short rotation exotic tree plantations, or abandoned (Zas and Alonso, 2002; Marey-Perez and Rodriguez-Vicente, 2008). Silvopasture
may be a preferred alternative to wholesale conversion to forest plantation in Galicia for a host of benefits, including reduced fire risk. Forest fires represent a significant loss of C from above-ground biomass on the Spanish landscape, as 192,000 ha burned in 1989 (Mosquera-Losada et al., 2006). Silvopasture may also be preferred by landowners for the consistent annual (or otherwise periodic) revenue stream that it can provide (Kurtz, 2000). The provision of high quality saw timber from European birch, in particular, may sway landowners towards implementation of silvopasture. The annual production of forage mixed with long rotation trees provides a stable system of mixed growth characteristics (Sharrow and Ismail, 2004). As pasture lands quickly become forest plantations in Galicia, silvopasture is an alternative land use option that preserves the growth characteristic of fast growing forage species and the maintenance of animals in the system. Compared to traditional treeless pastures and row crops, silvopasture may be a preferred land use for other environmental benefits, including additional C sequestration in biomass and soils.

The residence time for soil organic C varies as a function of aggregate size class (Parton et al., 1987; Six et al., 2002a). Carbon inputs to the soil from root decay and leaf fall are broken down by macro and micro fauna. Most C is lost in this process of decay, but some residual C does become incorporated into the soil, humified, and eventually ends up in longer term soil C pools. Over time, these resistant C pools can become quite large (Rodriguez-Murillo, 2001). Carbon associated with macroaggregate (250 – 2000 μm), microaggregate (53 – 250 μm), and silt+clay (<53 μm) size soil fractions can have mean residence times of 1 – 10, 25, and 100–1000 years, respectively (Parton et al., 1987; Schimel et al., 1994). Several studies have proposed that long term storage of C can occur in soil aggregates (Balesdent et al., 1998; Jastrow and Miller, 1998; Six et al., 2002b; Blanco-Canqui and Lal, 2004), and specifically within microaggregates
and silt+clay aggregates formed within macroaggregates. The shift from till to no-till land use practices leads to the preferential formation of macroaggregates, which are made up by microaggregates and smaller silt-clay sized aggregates (Christensen, 2001; Six et al, 2002b). This hierarchical organization of smaller sized aggregates within larger aggregates protects C stored in each occluded size fraction from microbial attack (Jastrow and Miller, 1998; Christensen, 2001) The loss of C generally occurs where tillage is implemented on previously untilled soils (Guo and Gifford, 2002), as tillage disrupts the process of macroaggregate formation, increasing bioavailable sources of C from smaller aggregates (Six et al., 2000a). The physical fractionation of soils into size class fractions allows us to consider the effects different land use practices have on the process of soil aggregation, how much C is contained in each fraction, and an estimation of residence times of SOC in a given soil (Christensen, 2001). In particular, the formation of macroaggregates and thus the formation of stable microaggregates within, is affected by land management (tillage, afforestation, etc.), which is an indicator of potential long term storage of SOC (Six et al., 2000a).

The objective of this study is to compare C storage in relation to different soil fractions at similar depths in silvopasture agroforestry and adjacent treeless pasture systems, and provide insight to the factors that lead to long term storage of SOC. At the St. Esteban Farm, individual Quercus suber (cork oak) trees provide information about changes in soil C as influenced by proximity to the canopy edge. The silvopasture treatment combinations at the Castro de Rey Farm provide a unique opportunity to study how C storage under Pinus radiata (radiata pine) and Betula alba (European birch) in two planting spacings varies with mineral, biosolids, and control fertilizer applications. It is likely that tree-based land uses will store more C in the soil, particularly at greater depth (Montagnini and Nair, 2004; Haile et al., 2008; Haile et al., 2009).
Specifically, the objectives of this study are to: (1) quantify the amount of C stored in three differently sized soil fractions and the whole soil at three distances from cork oak in four soil depth ranges to 100 cm at the St. Esteban Dehesa silvopasture in Extremadura province, (2) quantify the amount of C stored in three differently sized soil fractions and whole soil in four soil depth ranges to 100 cm under a simulated silvopasture experiment planted with radiata pine and birch in two spacings with three different fertilizer treatments each at the Castro de Rey Farm, Galicia, (3) consider treatment factors that lead to long term C storage in the three fractions and whole soil in silvopasture versus adjacent treeless pasture soils in each site, and (4) compare and contrast differences in C storage in the soil fractions and whole soil between the two sites.

Methods and Materials

Introduction to Study Sites in Spain

The study was carried out in two different regions of Spain, reflecting conditions in two major climatic zones of the country. Each site is considered a silvopastoral agroforestry system, with tree and forage production combined on the same land. The first study site is in central western Spain, the St. Esteban Farm, located about 5 km southeast of the town of Placencia, in Extremadura province (39°59' N, 6° 6'15" W). St. Esteban is a Dehesa oak silvopastoral system established at least 80 years ago, with pig, cattle, and cork production as management goals. The second site, the Castro de Rey Farm, is located approximately 12 km north of the town of Lugo in the province of Galicia (43° 9'39" N, 7° 29'54" W). The farm is host to a 1.6 hectare simulated silvopasture experiment begun in 1995, in addition to more traditional Galician agriculture such as cattle, sheep, hog and hay production.
St. Esteban Farm

Site description

Edaphic conditions are influenced by the Mediterranean climate patterns, kept warmer than other locations at the same latitude, with long hot summers and mild winters. Altitude is 400 meters above sea level. Mean rainfall is 520 mm per year (State Meteorological Agency, AEMET, 2009, Figure 3-3). Rainfall is mostly restricted to winter months, with ~90% falling between October and April. Mean temperature is 16.1° C, with a monthly range of 7.9° C and 25.8° C from January to July (Figure 3-3). Soils are Cambrisols according to the FAO classification system and Alfisols according to the USDA system. Soil texture is silt loam, with organic horizon to ~20 cm and an argillic horizon below 50 cm. No accumulations of inorganic C occur on this site. The total farm area is 120 hectares.

Soil sampling locations

Soil samples were taken from three different distances from six randomly selected individual cork oak trees. Soil samples were taken at 2, 5, and 15 meters from the tree, corresponding to about 0.5, 1.0 and 3.0 times the distance from the tree to the drip line. As tree crowns on this site are pruned to a common appearance (four widely spaced central branches), a common sampling pattern was adopted for all individual trees. Sampling was done on points on a transect corresponding to 2, 5, and 15 meters moving due west (270°) from the selected tree trunk. This direction was chosen to standardize possible edaphic differences in soils due to different sampling directions from the tree, particularly the differences in soils between north and south due to variations in incident sunlight (Moreno et al., 2007).
Castro de Rey Farm

Site description

Climate is influenced by Atlantic climatic patterns, with long cool moist winters and warm dry summers, with soil moisture limiting conditions in late summer (Mosquera-Losada, 2006, Fernandez-Nunez, 2007). Altitude is 440 meters above sea level. Mean rainfall is 1080 mm per year, and mean temperature is 11.5° C, with a range of 5.8° C to 18° C (State Meteorological Agency, AEMET, 2009). The vast majority of precipitation falls from October to April (Figure 3-5). Soils are Umbrisols (FAO classification) and Inceptisols (USDA system). Organic soil horizons can reach down to 40 cm. Texture is sandy loam, with an increase in clay below 50 cm. Gleyed clay is found below 50 cm, as well as other indicators of seasonal high water table around 60 cm depth. Bulk density was estimated by Fernandez-Nunez (2007) at 1.1 g cm⁻³ in surface horizons. Soils are acidic with no accumulations of inorganic C on this site (Fernandez-Nunez, 2007).

Soil sampling locations

The silvopastoral research plots at Castro de Rey Farm were initiated to study the effects of mineral, biosolids, and control fertilizer treatments on the growth of a pasture grass, *Dactilo glomerata*, planted under radiata pine and birch, common species utilized in Europe and northern Spain. The trees were planted in 1995, in a randomized complete block design with three replications, consisting of two spacings (2x2 and 3x4 m) each with 25 trees per plot, thus creating plots of 64 m² and 192 m². The silvopasture understory was established at the same time as trees, with a seed mix of *Dactylis glomerata* L. var. Saborto (25 kg ha⁻¹), *Trifolium repens* L. group Ladino (4 kg ha⁻¹) and *Trifolium pratense* L. var. Marino (1 kg ha⁻¹). An immediately adjacent pasture (not managed for experiments) was established at the same time with a commonly used local mix of *Trifolium repens* Huia (2 kg ha⁻¹), *Dactylis glomerata* L. var.
Saborno (10 kg ha$^{-1}$), *Trifolium repens* L. group Ladino (1 kg ha$^{-1}$), and *Lolium perenne* Tove (10 kg ha$^{-1}$). Pasture grasses and silvopasture understory are harvested and removed mechanically twice a year.

Prior to initiation of the experiment in 1995, the site (silvopasture experiments and adjacent pasture) was under potato (*Solanum tuberosum*) cultivation for more than 30 years (Rigueiro-Rodriguez et al., 2000; Mosquera-Losada et al., 2006). The pasture has not been fertilized since 2001, but from 1995 – 2001 was annually fertilized with 450 kg of NPK 8:24:16. There were a total of 36 plots in the silvopasture and six in the pasture area. Soil sampling in the silvopasture experiments was restricted to a net plot area of three by three (total nine) trees. In the adjacent pasture, six evenly spaced sampling points were chosen in a parallel transect line ~20 m to the south of the silvopasture experiments.

**Soil Sampling, Preparation, and Analysis**

Soil samples were removed from each site using a stainless steel cylinder with a cutting edge that was inserted with a powered hammer and removed with a platform-stabilized pulley (Moreno et al., 2005). Cores were measured for depth and divided in the field into four subsamples corresponding to four sampling depths: 0 – 25, 25 – 50, 50 – 75, and 75 – 100 cm. Care was taken to ensure samples were not excessively compacted. Bulk density estimates were made for each sampling depth using a cylinder of a known volume. Trees were measured within four weeks of soil sampling for diameter at breast height (1.3 m above ground) and overall height. Mortality of trees was noted for the experimental silvopasture plots at Castro de Rey.

Soil samples were immediately air dried at room temperature (~20-25° C) to a constant weight, and passed through a 2 mm sieve (#10 U.S. Standard Testing Sieve). The portion of soil that did not pass the 2 mm sieve was separated, dried overnight at 70° C (with the weight noted), and then discarded. The weight of the discarded fraction would be used to convert the eventual
data derived from 2 mm sieved fraction (whole soil, hereafter) back to field conditions (Rodriguez-Murillo, 2001; Six et al., 2002a). Whole soil (<2 mm particle size) samples were ground for homogenization and analyzed by a LECO C.N.H.S. Elemental Analyzer for percentage carbon within two weeks of being air dried. Particle size analysis was completed on a sub sample of whole soil (<2 mm) using the pipette method (Soil Survey Staff, 2004). Soil pH was measured in 0.1 M KCl using a Crison pH meter at 20° C (Crison Corporation, Barcelona, Spain).

**Soil Fractionation and Carbon Determination**

The whole soil was physically fractionated according to Elliot (1986) and Six et al. (2002b). A 25 g sample of 2 mm sieved air dry soil (of known moisture content) was placed in a 250 ml beaker. Distilled water (~150 ml), enough to completely cover the soil, was poured into the beaker to promote slaking. The slaking process breaks up water unstable aggregates in the soil, leaving water stable aggregates for further analysis. After 5 minutes, using a 250µm, and 53µm (#60 and #270 U.S. Standard Testing Sieves, respectively), slaked soil was poured on top of the 250µm sieve. Soil solution was wet sieved manually by moving the sieve up and down about 5 cm each, 50 times in two minutes. What did not pass the 250 µm sieve was backwashed, with a distilled water-filled wash bottle, into a pre-weighed and numbered aluminum plate. The remaining soil solution was next poured over the 53µm sieve, and given the same two minute manual sieving. What did not pass the 53µm sieve was backwashed into a pre-weighed and numbered aluminum plate. The remaining soil solution that passed the 53µm sieve was poured into a pre-weighed and numbered aluminum plate. The three soil fractions, 250 – 2000 µm, 53 – 250 µm, and <53µm, were dried at 60° C overnight, weighed, ground for homogenization, and stored in individually sealed and labeled plastic bags for further C analysis. Samples were then analyzed by a LECO C.N.H.S. Elemental Analyzer for percentage C within two weeks after
whole soil was air dry. All laboratory analysis (except texture determination) was conducted at
the University of Santiago de Compostela, Crop Production Department laboratories in Lugo,
Galicia, Spain. Texture was determined at the University of Florida, Soil and Water Science
Department Laboratories in Gainesville, Florida USA.

Experimental Design and Statistical Analysis

Data for statistical analysis was stored and organized using Microsoft Excel and analyses
of variance were completed using SAS statistical software version 9.2 for windows (SAS
Corporation, Carey, NC). Mean tree height and diameter were calculated from tree measures
taken from December 2007 – January 2008. Percent mortality in simulated silvopastures was
also calculated. Soil bulk density was calculated as the ratio of weight of dry bulk soil divided by
the volume of the sampling cylinder. Bulk density was used to convert C percentages in the
whole soil and fractions to Mg C per hectare (to a specified depth) basis using the following
formula:

\[
\frac{g \times C}{g < 2mm soil} \times \frac{g < 2mm soil}{g bulk soil} \times \frac{g bulk soil}{cm^3} \times \frac{g fraction soil}{g < 2mm soil} \times \frac{25 cm}{1} \times \frac{100}{1} = Mg \ C \ ha^{-1} \quad (4-1)
\]

The weight of soil fractions was calculated as a percentage of total whole soil weight.
Mean C storage in whole soil, soil fractions per hectare basis, and percentage fraction recovery
were compared using SAS ANOVA proc glm for linear models. Data were log transformed to
meet normality requirements for ANOVA. Means reported in this study were converted from
log-transformed data.

St. Esteban Farm: Differences in mean C storage at each sampling depth and soil fraction
(including whole) were compared for the three distances from the tree for each sampling depth.
A regression of tree diameter versus whole field C (0–100 cm) for each sampling distance to
cork oak was also completed to consider the effect of tree size on whole soil C storage at the
sampling distances and soil depths. Percentage recovery of soil fraction for all distances (combined and separate) from the tree was compared. Whole field (per hectare) estimates of SOC were made by multiplying the mean SOC storage (Mg C ha\(^{-1}\)) around the trees (sum of all depths) by the proportion of the whole field represented by that area.

**Castro de Rey Farm:** Mean C storage per soil depth and fraction (including whole soil) was compared for each treatment combination at each soil sampling depth. Within the silvopasture experiment, an ANOVA was carried out for all treatment factors and factor combinations. Percentage recovery of soil fraction for all distances from the tree was compared for all treatments combined, and for pine, birch, and pasture cover types. In addition, C storage in the whole soil and soil fractions at each sampling depth was compared for pooled treatment combinations: all silvopasture treatments versus treeless pasture, radiata pine versus birch versus pasture, and fertilizer and spacing treatments for each species of tree versus pasture. Mean separation for both sites was completed using the Duncan-Waller multiple range test, with \(\alpha\) <0.05.

**Results**

**St. Esteban Farm**

**Silvopasture stand characteristics**

All cork oak trees (>80 years old) at the St. Esteban Farm had been pruned to four branches between 4 – 5 m in height. The diameter at breast height of cork oak ranged from 48.6 cm to 70.1 cm, with a mean of 58.2 cm. Stand density was estimated at ~35 stems per hectare.

**Soil bulk density, texture, and acidity**

Soil bulk density at St. Esteban ranged from 1.2 g cm\(^{-3}\) for 0 – 25 cm depth to 1.34 g cm\(^{-3}\) from 25 – 100 cm. Soil texture was a silt loam to 100 cm, with an increase in clay after 25 cm from 14% in 25–50 cm to 23% at 75–100 cm (Figure 4-1). Soil pH was acidic and ranged from
3.9 to 4.1 in most soil depths, but was more acidic (pH of 3.6) in the 25–50 cm depth under the tree canopy, 2 and 5 meter sampling distances (Figure 4-2). Soil pH did not differ statistically between sampling distances at each sampling depth.

**Carbon in whole soil**

Mean whole soil C storage under the cork oak stand decreased with sampling depth and distance from the tree (Figure 4-3). C storage at 2 m was significantly greater than at 15 m (41.22 and 20.01 Mg C ha⁻¹, respectively). The majority of soil C to 100 cm depth, 74.67%, was found in the 0 – 25 cm soil depth, with 15.97% in the 25–50 cm, 5.1% in the 50–75 cm, and 4.64% in the 75–100 cm depths.

**Carbon per hectare**

The mean overall C storage at the field level (sum of all sampling depths) was estimated at 29.99 Mg C ha⁻¹. Regression of tree diameter explained 67, 62, and 43% of the variation in whole field C storage at the 2, 5, and 15 meter sampling distances, respectively.

**Carbon in soil fractions**

**Fraction recovery and distribution:** Recovery range of fractionated soil particles was between 98% and 100% by dry mass. Most of the soil mass was found in the 250–2000 µm soil fraction, with some exceptions (Figure 4-4 and Table 4-1). Dry weight recovery of the 53–250 µm fraction was similar to the 250–2000 µm fraction in the 25–50 cm depth only. In the 0 – 25 cm depth only, the 250–2000 µm fraction was higher in mass under the tree canopy (2 and 5 m) than in the 15 m distance, and the silt+clay size fraction was greater in the 15 m distance (Table 4-1). Sorting of soil size fractions less than 2 mm yielded an even distribution, with at least 20% by weight represented by all size classes at all depths examined.

**250–2000 µm fraction:** The macroaggregate size fraction had significantly more C at the 2 m distance than at the 15 m distance at three of the sampling depths (Figure 4-6). At the 5 m
sampling distance, compared with the 15 m, significantly more C was found in the shallowest (0–25 cm) and deepest (75–100 cm) sampling depths.

**53–250µm fraction:** In all sampling depths, no significant differences between distances from tree were detected (Figure 4-6).

**Less than 53µm fraction:** In all sampling depths, no significant differences between distances from tree were detected (Figure 4-7).

**Castro de Rey Farm**

**Silvopasture stand characteristics**

The height range for 13 year old radiata pine was 11.52 m for 2x2 m spacing and mineral fertilizer to 13.37 m for 2x2 m spacing and no fertilizer (Table 4-2). DBH ranged from 12.22 to 22.85 cm, for 2x2 m spacing mineral fertilizer and 3x4 m spacing no fertilizer, respectively. There were no statistical differences between height in radiata pine, but DBH was significantly higher \((p < 0.05)\) in the 3x4 m spacing treatments as opposed to the 2x2 m spacing. Heights for birch of the same age ranged from 6.61 m for 2x2 m spacing and mineral fertilizer to 10.66 m with 3x4 m spacing and no fertilizer (4-2). Diameters of birch ranged from 3.63 to 10.56 cm for 2x2 spacing mineral fertilizer and 3x4 no mineral fertilizer, respectively. Height and DBH were both significantly higher in birch for the 3x4 spacing. Mortality for each species is also shown in Table 4-2. Mortality was highest for birch in the 2x2 m spacing and similar across radiata pine treatments.

**Soil bulk density, texture, and acidity**

Soil bulk density was estimated at 1.15 g cm\(^{-3}\) for 0–25 cm, 1.25 g cm\(^{-3}\) for 25 – 50 cm, and 1.51 g cm\(^{-3}\) for 50 – 100 cm. No difference was found between pasture and silvopasture soils. Soil texture was sandy loam to 100 cm, with laboratory tests confirming a slight increase in clay from 19% to 23% in from the 0 – 100 cm depth (Figure 4-8). Soil pH was acidic, ranging
from 4.7 in the 0–25 cm depth to 3.7 in the 75–100 depth, with a consistent decreasing trend for all radiata pine, birch, and pasture treatments, with no statistical difference between treatments at each sampling depth (Figure 4-9).

**Carbon in whole soil**

For treatment combinations, the only significant difference was found in the 0 – 25 cm depth between 3x4 m spaced birch (130.55 Mg C ha\(^{-1}\)) and 3x4 m radiata pine (61.34 Mg C ha\(^{-1}\)), each with no fertilizer application (Table 4-4). Contrasts of mean C storage for pooled silvopasture treatments versus an adjacent pasture are shown in Figure 4-10. Mean pasture C storage is greater, yet not significantly, than pooled silvopasture C storage at 0–25 and 25–50 cm depths. The trend reversed in lower soil depths, with more C in silvopasture than pasture (but again, not a significant difference). Soil C underlying birch was significantly higher that the radiata pine in the 0 – 25 cm depth, and greater, but statistically similar to the pasture (Figure 4-11 and Table 4-4). All other depths had statistically similar C amounts.

No significant differences were identified in mean soil C for birch and radiata pine with three fertilizer treatments and the adjacent pasture (Figures 4-12). Contrasts of mean soil C for birch and radiata pine with two spacings treatments and an adjacent pasture were also not significant at any depth (Figures 4-13). The majority of soil C to 100 cm depth, 78.82%, was found in the 0–25 cm soil depth, with 12.9% in the 25–50 cm, 4.92% in the 50–75 cm, and 3.36% in the 75–100 cm depths.

**Carbon per hectare**

Birch planted at 3x4 m spacing showed significantly greater C storage than the 3x4 m radiata pine planted at 3x4 m spacing, each with no fertilizer (176.91 versus 80.94 Mg C ha\(^{-1}\), respectively, at \(p < 0.05\)). All other treatment combinations were statistically similar (Table 4-5).
**Simulated silvopasture experiment**

Analysis of variance results for all simulated silvopasture treatment factors (excluding the adjacent pasture) demonstrated that depth and species were significant main factors affecting soil carbon stocks in all soil fractions (Table 4-3). Additionally, the interaction between spacing and fertilizer and spacing and tree species treatments was significant.

**Carbon in soil fractions**

**Fraction recovery and distribution:** Recovery range of fractionated soil particles was between 91% and 100%. Most of the soil mass was found in the 250–2000 µm soil fraction (Figure 4-14). The 250–2000 µm soil fraction size represented a higher percentage of soil mass at the shallow soil depths 0–25 and 25–50 cm, while the 75–100 cm depth had more mass in the 53–250 µm fraction. The <53 µm fraction increased in percentage weight with soil depth, but was significantly less than the other fraction sizes. In the 0 – 25 cm depth, the 250–2000 µm soil fraction recovery was higher under both silvopasture pooled treatments as compared to the open pasture, while in the <53 µm fraction, open pasture had greater recovery (Table 4-1). Several significant differences were found between major vegetative cover types in the whole soil and three fractions, and most were found in the 0 – 25 cm depth (Table 4-6).

**250–2000 µm fraction:** In the 0-25 cm depth, a significant difference in treatment combinations was found between the 3x4 m spacing of birch (89.09 Mg C ha⁻¹) and 3x4 m radiata pine (39.57 Mg C ha⁻¹), each within the no fertilizer treatment (Table 4-4). Also, in the 50-75 cm depth, 3x4 m spacing birch has significantly higher C storage (5.68 Mg C ha⁻¹) than birch of 2x2 m spacing (0.8 Mg C ha⁻¹), each with biosolids fertilizer. At the deepest sampling depth (75 – 100 cm), 2x2 m spaced radiata pine with no fertilizer (3.16 Mg C ha⁻¹) had significantly greater C in this fraction than the pasture (0.65 Mg C ha⁻¹) in this fraction (Table 4-4).
When comparing pasture to pooled silvopasture treatments, silvopasture had more C per sampling depth than pasture alone (Figure 4-15). At lower sampling depths 50 – 75 cm, significantly more C was found in silvopasture than the pasture ($p<0.05$). Upon comparison of pasture with radiata pine and birch, birch had more C than pasture and radiata pine at 0 – 50 cm (Figure 4-16 and Table 4-6). In the 0–25 cm depth, birch showed significantly more C than radiata pine, 71.76 versus 49.71 Mg C ha$^{-1}$, respectively. Birch had significantly more C (at $p<0.05$) than pasture at lower depths, 50–75 and 75–100 cm. Carbon in this fraction, underlying radiata pine stands, was not significantly different from the adjacent pasture, except at the 75–100 cm depth (Figure 4-16).

Significantly greater C storage in the 75-100 cm depth was found for mineral and no fertilizer treatment for birch as compared to pasture (Figure 4-17). Un-fertilized radiata pine treatments also had significantly greater C storage in this fraction than the adjacent pasture. As for spacing contrasts, birch, 3x4 m spacing, showed more C through all sampling depths, with significantly more C as compared to the pasture in the 0–25, 50–75, and 75–100 cm sampling depths (Figure 4-18). The 2x2 m birch silvopasture treatments had similar C storage as the pasture across most sample depths, except between 50-75 cm. As for radiata pine spacing treatments, the 2x2 m spacing had more C at the two lowest sampling depths and as compared to the pasture (50 – 75 cm) and 3x4 m radiata pine (75–100 cm).

**53–250µm fraction:** No significant differences were identified for individual treatment combinations in this fraction (Table 4-2). In the pasture and pooled silvopasture treatments comparison, significantly more C was found in the 0–25 and 25–50 cm sampling depths in the pasture (Figure 4-19). At lower depths (50–75 and 75–100), silvopasture had slightly more C, yet not a statistically significant difference. When pooled silvopasture treatments are contrasted by
species, pasture, again, has more C storage in the 53–250µm fraction in the two shallowest sampling depths than radiata pine, and statistically similar storage as the birch (Figure 4-20 and Table 4-6). Below 50 cm, no difference in C storage was noted in the 53–250µm soil fraction.

Pooled fertilizer treatment means at Castro de Rey demonstrated no significant differences at each depth (Figure 4-21). Carbon storage in the pasture in the 53–250µm fraction exceeded each tree species and fertilizer combination in the 0–50 cm depth (but again, not at \( p<0.05 \)).

Contrasts of birch spacing treatments revealed no significant differences between mean C storage at all depths sampled (Figure 4-22). Pasture C storage was greater in the shallowest sampling depths (0–50 cm) than the 3x4 m radiata pine spacing treatment, yet statistically similar to the dense 2x2 m spacing treatment in all sampling depths.

**Less than 53µm fraction:** Individual treatment comparisons revealed a significant difference between pasture (15.84 Mg C ha\(^{-1}\)) and unfertilized birch 2x2 m spacing (7.2 Mg C ha\(^{-1}\)) for the 0 – 25 cm depth only (Table 4-4). Pooled silvopasture treatments demonstrated significantly less C in the 0–25 cm soil depth than the pasture (Figure 4-23). This trend continued to the lower sampling depths, but was not significant below 25 cm. Contrast of tree species with pasture revealed significant differences, as pasture had more C than both tree species in the 0–25 cm depth, and more C in the 25–50 cm depth than radiata pine (Figure 4-24 and Table 4-6). In the two remaining lower depths, no differences were found among pooled species treatments.

In the fertilizer comparisons for both species, C storage in pasture was significantly greater in the 0–25 depth than the silvopastures that received no fertilizer (Figure 4-25). In the birch treatments, mineral and biosolids treatments lead to a greater, yet non-significant C storage than the no fertilizer treatment in the 0–25 and 25–50 cm depths. Below 50 cm, no differences were
found among fertilizer and pasture treatments for this fraction. For radiata pine, significant
differences were found in the 0–25 cm depth, with pasture containing more C storage than
biosolids, no fertilizer treatment.

Spacing treatments for birch showed significantly more C under pasture than the birch in
2x2 m spacing in the 0 – 25 cm depth (Figure 4-26). In the 75 – 100 cm depth of birch
silvopastures, more C was found under the 3x4 m spacing than under 2x2 m. Radiata pine
spacing treatments showed significantly less C in the 0 – 25 cm depth than the pasture, and in the
25 – 50 cm depth, pasture storage exceeded the 3x4 m spacing.

Discussion

St. Esteban Farm

Tree stand characteristics

Stand density of cork oak at St. Esteban was typical of Dehesa silvopasture, given the
region’s approximate annual precipitation of 500 mm (Joffre et al., 1999). Measures of DBH and
tree height are difficult to compare, since the local practice of pruning to four primary branches
and harvesting of cork homogenizes tree physical characteristics; trees appear similar to each
other, except in diameter. On cycles of 9 – 10 years, ~5 cm of cork is removed from the diameter
of the tree (Carvalho-Mendes and Graca, 2009).

Soil bulk density and pH

Soil bulk density increased with depth at St. Esteban (Figure 4-1), possibly due to an
increase in clay with depth (Figure 4-1), or effects of tillage practices, plant rooting, and soil
macrofauna activity in surface soil horizons. Soil pH at St. Esteban decreased dramatically in the
subsurface soil layer of 25 – 50 cm (Figure 4-2). Cork oak is a known soil acidifier, and the
relatively high levels of acidity in the study site are characteristic of soils under cork oak stands
(Noble et al., 1999; Vacca, 2000; Serrasolses 2009). Serrasolses et al. (2009) attribute soil
acidification in cork oak as an adaptation to low iron availability. The trend in pH from acidifying cork roots suggests that there is less tree influence outside of the drip line, if in fact cork oak roots are found at this depth (Figure 4-2). Little is known about root length density and distribution of cork oak roots at a given distance from the tree, and it is difficult say definitively that the 15 m distance was outside the influence of the tree. Moreno et al. (2005), found similar rooting densities for *Quercus ilex* up to 20 meters from individual trees. *Quercus ilex* is found at the St. Esteban Farm (but not where sampling for the current study took place) in similar densities, height, diameter at breast height, and crown diameter, and soil type as reported in Moreno et al. (2005). Thus, there may not be an area of the St. Esteban farm that is not in some way influenced by tree roots, even with tree densities of ~35 stems ha⁻¹. Rooting in the Dehesas is limited in the upper soil horizons because of excessive drying that occurs during summer (Silva and Rego, 2003), and on this site, rooting depth is limited below 50 cm by shallow soils that were, at times, less than 100 cm deep. Through measurement of hydraulic lift in cork oak Dehesas in southern Portugal, Kurz-Besson et al. (2006) estimated that larger cork oak roots (>2 mm) increased in the 0.4 to 1.0 m soil depth, and that during summer, roots extract water from over 1.0 m in depth (they also report that fine roots in cork oak, <0.5 mm, are concentrated in surface soils). This depth corresponds with the reduction in soil pH that was observed at the St. Esteban Farm (and the rooting limitations mentioned above), and it is likely that larger cork roots proliferate in the 25-50 cm soil sampling depth.

**Whole soil carbon**

While cork oak roots may proliferate in the 0 – 100 cm range, roots from herbaceous species, inputs of organic matter from tree leaf fall and animal excreta are also likely additional sources of C input to the upper soil horizons (Moreno et al., 2005). In the shallowest sampling depth (0 – 25 cm) a doubling of soil C was noticed moving from 15 m to 2 m to the tree (Figure
The numbers of roots in herbaceous species in the Mediterranean as well as elsewhere, on the other hand, are highest closer to the soil surface and decrease significantly with depth. Silva and Rego (2003) described the root counts for several herbaceous and shrubby species in southern Portugal, demonstrating a niche separation of resource use between species such as deep-rooted shrubs that can draw on soil moisture deeper in the soil in summer and more shallow-rooted seasonal growth species that take advantage of seasonal rainfall in winter. Herbaceous species are also likely to produce more fine roots as compared to shrubs. In the St. Esteban Dehesa, cork oak can root more deeply than herbaceous species, and micro-site improvements closer to the tree (Moreno et al., 2007) improve growing conditions for herbaceous species in the upper soil profiles when water is available. Cork oaks probably also take advantage of these seasonal improvements. During field work at the St. Esteban Farm in January 2008, the growth of the native pasture grasses was noticeably enhanced under the trees (Figure 4-27). As cork trees are a source of C to the soil through root turnover and leaf fall (and by attracting animal excreta), they also facilitate a soil environment around the trees that is favorable for plant growth. Seasonal species (native pasture grasses in this case) as well as cork oak can take advantage of higher P, N, and other soil fertility improvements under the tree during the winter and spring months (Lopez et al., 2001; Moreno et al., 2007). Lopez et al. (2001) found higher annual fine root biomass for Quercus ilex in winter months in northeastern Spain when soil water was available. The combination of these actively growing tree and native pasture elements improves C inputs and storage in the upper soil horizons.

**Whole field carbon**

While SOC storage at 2 m from the tree was more than double the storage at the 15 m distance, only 27.5% of the land at St Esteban is underneath the tree canopy (assuming a 5 m drip line with 35 stems ha⁻¹). An estimate for the whole field SOC storage, 29.99 Mg C ha⁻¹, was
calculated from percent of land containing these various total carbon stocks. Moreno et al. (2007) showed that the positive influence of individual *Quercus ilex* trees on soil chemical properties only extends barely past the drip line, representing about 15% of the land on a less dense, 20 stems ha\(^{-1}\) Dehesa located in the same region of Spain. Higher density Dehesas, all other factors equal, will lead to greater tree influence on the landscape, in this case, a 13% increase in tree cover per hectare. As such, increasing densities of trees in the Dehesa, especially where trees have been lost to disease or removal, can be expected to lead to greater soil C storage on the landscape.

The estimated C storage on a hectare basis is on the low end of Spanish soils but well within the standard deviations of several reported land uses that coincide with the Dehesa agrosilvopastoral system (Rodriguez-Murillo, 2001). Whole field storage of C is similar to typical regional land uses such as olive groves (*Olea europea*), pastures, a pasture-broadleaf mix, and dry land farming (Rodriguez-Murillo, 2001), each of which may serve as a proxy for the Dehesa system under study. Regression of tree diameter and whole field C storage at varying sampling depths explained much of the variation in C storage between individual trees, particularly at the 2 and 5 meter sampling distances. As such, it is possible to predict that continued growth of trees in the smaller size classes will yield more soil C over time. Cork oak is a long lived tree (>100 years), and an increase in SOC surrounding the tree is likely as trees age.

Comparing this whole field C result with other sites reveals interesting similarities in storage in agroforestry systems in some very disparate sites. Nair et al. (in press) reports results for total soil C (given particular sampling depths) for several agroforestry systems worldwide, including 27.4 Mg C ha\(^{-1}\) for *Gmelina arborea* and crop mix in central India, 33.3 Mg C ha\(^{-1}\) for a *Faidherbia albida* parkland, in Nigeria, and between 6.9 to 24.2 Mg C ha\(^{-1}\) for a *Pinus elliotti*
and *Paspalum notatum* mix in Florida, USA. The C storage value (20–40 Mg C ha⁻¹) estimated for the Mediterranean St. Esteban Farm in this study is within range of potential C storage values for parklands and grazing lands in the arid and semiarid lands as reported by Nair et al. (in press).

**Carbon in soil fractions**

Total C analysis of the soil fractions showed that most C was also stored in the 250 – 2000 µm fraction at all depths examined (Tables 4-4 and 4-6). Differences in C storage in this fraction are generally attributed to greater C inputs, and the preferential stabilization of OM in macroaggregates as opposed to smaller size class aggregates. Fraction recovery percentages (by weight) also reflect the greater macroaggregation that took place under the tree canopy in the 0 – 25 cm depth (Table 4-1). While greater mass recovery of macroaggregate soil fractions lead to significantly greater C storage in this fraction, the same relationship is not found in the silt+clay aggregate size. Macroaggregates are known to have greater C per mass and increases in recovery of this fraction will lead to greater C storage. On the other hand, greater mass recovery of the silt+clay fraction does not lead to greater C storage, as this fraction has less C per mass (Six et al., 2000a).

Several studies have noted the protection of C within the macroaggregate size class as affected by afforestation and cessation of tillage, represented in this study as distances closer to cork oak trees (Del Gado et al., 2003; Denlef et al., 2007, Tan et al., 2007). Under the drip line and at the drip line (2 and 5 m distances), cork oak inputs from above and below ground, as well as growth of seasonal herbaceous species, lead to greater C storage in this fraction in the 0 – 25 cm depth. In lower sampling depths, greater macroaggregate C was stored in the 50 – 100 cm depths, reflecting the inputs from tree roots. Kurz-Besson et al. (2006) and Pereira et al. (2009)
estimated that cork oak roots occupy this zone, and as such, this represents soil depths where root turnover, over the decades of growth at St. Esteban, has contributed greater C that is protected in macroaggregates. While the macroaggregates themselves have relatively short turnover times compared to smaller aggregates, the addition of organic matter to the soil at this depth would provide for the formation of stable micro- and silt+clay sized aggregates. As deeper soil depths are less influenced by management activities, the inputs of C to this depth by roots are a source of C for macroaggregate formation and preservation of C for long term in smaller microaggregate and silt+clay sized aggregates (Six et al., 2000a; Fonte et al., 2009). While no significant differences were found in the whole soil at the lower sampling depths (25 – 100 cm), separation of aggregates by size demonstrates preferential C storage in the macroaggregate fractions closer the tree at all depths examined. Differences between sampling distances in SOC in the macroaggregate size fraction below 25 cm may be attributed to rhizodeposition by cork oak roots.

The smaller micro- and silt+clay fraction sizes demonstrated no differences in C storage at each sampling depth, but the C in silt+clay size aggregates follows a reverse non-significant tendency (by distance to tree) in mean C protection as compared to the macroaggregate size class, noted in the 0 – 25 cm depth (Figures 4-5 and 4-7). The SOC storage in free (un-occluded) silt+clay size aggregates away from the tree is likely greater, since under the tree canopy, improved soil edaphic conditions closer to the tree would promote inclusion of silt+clay aggregates in micro- and macroaggregates. Further fractionation of macroaggregates under the tree canopy would have to be completed to determine if micro- and silt+clay aggregates are preferentially held within. Six et al. (2000b) demonstrated preferential silt+clay protection within microaggregates and macroaggregates on no-till grassland site as compared to a tilled site. In this
study, C storage in the microaggregate fraction followed the same pattern as in macroaggregates, but significant differences between distances from the tree were not demonstrated at any depth (Figure 4-6). Protection of stable silt+clay aggregates within larger size aggregates is likely. It may be that silt+clay sized aggregates are held within micro- and macroaggregates closer to the tree, but this conjecture could only be validated by destruction and analysis of macroaggregates and microaggregates that are theorized to contain of silt+clay aggregates. While the theorized mechanism for long term SOC storage is indicated by greater macroaggregate SOC under the tree canopy, the non-significant differences between SOC storage in smaller aggregates under the tree versus open pasture do not signify that longer term storage is occurring closer to the tree. The deep rooted nature of cork oak, an adaptation for survival of lengthy summer droughts, provides a source of C to the soil at deeper depths than herbaceous species alone, and promotes the preferential storage of C in macroaggregates. Thus, storage of C in the macroaggregate size class increased under the drip line of individual cork oaks on this site as compared to the open native pasture.

**Castro de Rey Farm**

**Tree stand characteristics**

Growth of radiata pine in the 2x2 m spacing followed a similar, yet non-significant, trend in height and diameter as previously reported by Mosquera-Losada et al. (2006) and Rigueiro-Rodriguez et al. (2000). Tree height and diameter for radiata pine and birch in each fertilizer treatment in each spacing regime followed the same trend on the same site as reported by Fernandez-Nunez (2007). Reduced growth of both tree species (in height and diameter) in the mineral fertilizer treatments was attributed to competition with fast growing pasture grasses that utilized mineral sources of nutrients more efficiently (Rigueiro-Rodriguez, 2000; Fernandez-Nunez, 2007). Gautam et al. (2002) found that radiata pine had a reduced fine root length density.
when planted with a Ryegrass/Clover \((Lolium\ spp./Trifolium\ spp.)\) mix as well as with Lucerne \((Medicago\ sativa)\), as compared to bare ground. Bakker et al. (2009) found that \textit{Pinus pinaster}\ with mineral fertilizer, compared with controls, had lower fine root mass in 0 – 20 cm soil depth. Spacing treatments appear to have had an effect on height and diameter of birch, as each tree measure was significantly greater in the less dense 3x4 m spacing regime. Rozados-Lorenzo et al. (2007) also found that high density stands of \textit{Betula alba}\ (2,500 stems ha\(^{-1}\)) had reduced diameter and height as compared to lower density stands (952 stems ha\(^{-1}\)), and reduced diameter for \textit{Pinus radiata} on similarly spaced stands on the same site with higher average precipitation than the current study. Density seems to be a significant factor in birch height, as denser tree stands lead to reduced height (Table 4-2). Diameter at breast height for birch and pine was negatively affected by denser spacing.

Mortality has had a highly varied effect on stand density of both radiata pine and birch, with a range of 0 to 67% mortality in birch 3x4 no fertilizer and 2x2 mineral fertilizer treatments, respectively. In this study, mortality was not tracked over time, and as such, it is not known when these trees died, and consequently, what effect the loss of stand density over time would have had on soil carbon. For birch, mortality was higher in the denser, 2x2 m spacing treatments (mean of 41.95%) than the less dense, 3x4 m spacing (mean of 7.13%). For radiata pine, the mortality trend was reversed, with 40.73% mortality in all 3x4 m spacing treatments, and 20.97% for the 2x2 m spacing. Radiata pine appears to better tolerate high density spacing as well as other site conditions, as compared to birch.

**Soil bulk density and pH**

Estimates of soil bulk density in the 0–25 cm range were similar to those previously reported by Fernandez-Nunez, (2007), 1.1 g cm\(^{-3}\) versus 1.15 g cm\(^{-3}\). Below 25 cm, the presence
of <2 mm-sized rocks may have contributed to a higher bulk density, as well as the increase in clay sized material (Figure 4-8).

Soil acidity at Castro de Rey was previously reported to have been affected by fertilizer treatments (0 – 25 cm depth), with mineral fertilizer demonstrating a higher pH over all other treatments (Mosquera-Losada et al., 2006; Nunez-Fernandez, 2007). In the present study, pH was not affected by treatment combinations or pooled fertilizer treatments (data not shown), or for each tree species versus pasture (Figure 4-4). In the shallower 0–50 cm depths, pasture had a lower pH as compared to birch and radiata pine treatments. Below 50 cm, radiata pine had lower pH than pasture and birch (Figure 4-4). Parfit et al (1997) and Noble et al. (1999) report that *Pinus radiata* had a significant soil acidifying effect on afforested sites in New Zealand and Australia, respectively. As it relates to SOC storage, the acidifying effect has been shown to increase levels of Al and Fe in the soil, elements that are important for the formation of organomineral complexes (Noble et al., 1999; D’Angelo et al., 2009). Microbial populations, a significant controller of C decomposition, are also affected by soil pH differences (Anatya et al., 2002). Extremely low pH would likely lower the population of some decomposer communities, shifting assemblages towards those tolerate of acidity, such as fungi.

**Whole soil carbon**

Carbon in the whole soil decreased with soil sampling depth, as is typical of many SOC studies (Corre et al., 1999; Paul et al., 2002; Guo et al., 2007; Haile et al., 2008). From the point of view of enhanced soil C sequestration in the whole soil, birch appears to be the preferred tree species as compared to radiata pine (Figure 4-11 and Table 4-5). The difference between pine and broadleaf species, in particular, has been identified as a major factor affecting C storage (Paul et al., 2002). Pine stands create an understory environment less conducive to plant growth, by the formation of a slowly decomposing duff layer, which inhibits germination and growth of
other species, and over time, reduces C inputs to the soil (Paul et al., 2002). Fernandez-Nunez (2007) found a ~20 cm layer of pine needles on the site of the current study. The lack of herbaceous species in the pine understory reduces fine root growth and C inputs. Guo et al. (2007) found that soils that had been afforested with *Pinus radiata* compared with a native pasture in temperate Australia had significantly less overall root length production, shorter turnover rates, and greater longevity down to 60 cm depth. Birch, as well as other broad-leaved species, are believed to be better at improving C sequestration by the relatively high N levels (thus lower C:N ratios) in leaves that form inputs to the SOC (Paul et al., 2002). In the current study, birch had an actively growing understory, and annual measures of forage production were ongoing in January 2008, when sampling for this study was completed. Pine, on the other hand, showed almost no understory growth, especially in the 2x2 m spacing. In 2000, understory growth was reduced under both tree species in the 2x2 m spacing as compared to the 3x4 m spacing, and by 2006 understory plant growth was nil under pine in the 2x2 m spacing, and under birch, forage was still being produced in both spacings (Fernandez-Nunez, 2007). The combination of the two silvopasture components, tree and understory herbaceous species, helps to improve C inputs to the soil, and as such, can lead to greater C sequestration in the soil than in pasture and plantation alone (Sharrow and Ismail, 2004). The fertilizer treatments also had no detectable effect on SOC in the whole soil. Theodorou and Bowen (1990) found that fertilizer did increase *Pinus radiata* growth and needle duff layer, but that mineralization of the C in the duff was still more controlled by environmental conditions, as such, inputs to the soil were not different from controls. This may help to explain why there were no differences in SOC in the whole soil for pine, and for the same reasons, may explain why there were no differences in birch. The incorporation of lower lignin birch leaf litter in the soil, as compared to pine (Hobbie
et al., 2006), has likely lead to greater SOC storage. At the time of sampling for the current study in January 2008, all pine stands were essentially pure plantations, with no understory. While established as silvopastures in 1995, the forage component would likely only be revived with thinning and duff removals. The birch, on the other hand, still maintained an herbaceous component, and as such, is a preferred species for silvopasture establishment, and for C sequestration in the soil.

**Whole field carbon**

Birch appears to be a better choice for greater C storage in the 3x4 spacing for all fertilizer treatments, when compared to the pasture. In the 2x2 spacing for birch, only mineral fertilizer exceeded pasture storage in this spacing, although not in a statistically significant manner ($p > 0.05$). Pasture storage in the whole field was greater than most radiata pine treatments (except mineral fertilizer in the 2x2 spacing), but did not differ statistically. While it has been proposed that higher density tree stands will sequester more C in biomass (Nair et al., 2009), results for whole field soil C in this study do not support this trend (Table 4-5). The effects of mortality may, in particular, have reduced C inputs to the soil in the denser 2x2 m birch stands. At the time of sampling in January 2008, the 2x2 m spaced stand of each species appeared to be reaching the negative feedbacks associated with density (limited resource utilization), as tree crowns were overlapping, with lower branches not receiving as much light as in more widely spaced stands. Radiata pine, in particular, at 2500 stems ha$^{-1}$ is in need of thinning to reduce to about half the stems to about 1000–1600 stems ha$^{-1}$ (which is typical of local practices, Silvia Rodríguez Barreira, personal communication, June 2006).

Differences in mineralization of above ground inputs may help to explain why birch appears to be so much better at sequestering C in the soil than pine, as pine needles slowly decomposing on the soil surface represent a slower rate of C input to the soil as compared to
quickly decomposing birch leaves. Nunez-Fernandez (2007) demonstrated major differences at the Castro de Rey Farm in the leaf litter present under both species. Pine leaf litter weight ranged from 2.28 – 6.43 Mg dry mater ha\(^{-1}\), compared with ~0 for birch in 2006. Birch leaves may also have been blown off site or eaten by wild deer. Hobbie et al. (2006) found faster decay rates for leaf litter of *Betula pendula* as compared to *Pinus sylvestris* in a common garden plot in Poland. Radiata pine needles have piled up on the understory floor at the Castro de Ray Farm, and as such, represent a pool of C that is relatively unprotected; a fire could oxidize the pine needle C in a matter of minutes, representing a loss to the system C pool. While the differences between above ground and belowground inputs was not considered here, soil inputs from the more slowly mineralizing aboveground leaf litter of radiata pine may not contribute as much to soil C as faster decomposing birch, a possible explanation for the differences in soil C seen between the two species.

Mineral and biosolids fertilizer treatments generally yielded higher, yet non-significant differences in whole field calculations, with one exception in the 3x4 birch treatment with no fertilizer (which yielded the highest C storage). Negative interactions with planted understory grasses may help to explain why tree diameter and height growth differences (Figure 4-1) were generally lower with mineral fertilizer treatments. Fernandez-Nunez (2007) attributes this to the preferentially higher soil nutrient utilization efficiency of monocotyledon grasses. Other researchers have also found reduced fine root biomass underneath fertilized pines (Bakker et al., 2009). The non- fertilized plots contained more dicotyledonous plants in the understory (Fernandez-Nunez, 2007). Significantly more pasture production took place in plots fertilized with mineral fertilizer (Mosquera-Losada et al., 2006), and as such, nutrient extraction by understory grasses may explain why tree growth was reduced compared to other treatments.
Overall whole field C storage at the Castro de Rey Farm was much higher than that found at the St. Esteban Farm, and high for Spanish soils in general, but well within the standard deviation for the following general Spanish land use types reported by Rodriguez-Murillo (2001): bush land, broadleaved forest, coniferous forest, pastures, and meadow. Comparison of whole field C storage with previously published data on soil C on this site is made difficult because of differences in methodology (Tables 3-1 and 4-5). Fernandez-Nunez (2007) utilized the Saverlandt procedure for determination of total carbon, to a depth of 25 cm. Additionally, the amount of stoniness (fraction of soil which does not pass a 2 mm sieve) was not taken into account. A more shallow sampling depth (0–25 cm) would lead to an underestimation of whole field C storage. In the current study, approximately 74.7% of the whole soil C (in the < 2 mm soil fraction) was found in the 0–25 cm depth. Also, ignoring stoniness will lead to an overestimation of soil C, a concern common to SOC estimation studies (Murillo-Rodriguez, 1999). In the present study, all bulk density estimates were corrected to account for this large, relatively inert, fraction of soil. Estimates of whole field C storage by Fernandez-Nunez (2007) greatly exceed those of the current study (Tables 3-1 and 4-5). C storage is similar to what Nair et al. (in press) estimated for some agroforestry systems in the tropics. Improved fallows, home gardens and silvopastures in the humid lowland tropics are reported to have similar storage, while many of the other mentioned practices demonstrated lower SOC sequestration potential. In temperate Pennsylvania, the mean estimates for SOC on a northern hardwood forest were much higher than the current study, with a mean 289 Mg ha$^{-1}$ (Johnson et al., 2009). Lower estimates of whole field SOC are found in Stevens and van Wesemael (2008) for the Belgian Ardennes, from 48.6 to 75.9 Mg C ha$^{-1}$, across several varied land uses. Generally though, the whole field profile sum of SOC to 100 cm fits well within the range for similar land uses in Spain, and worldwide.
Simulated silvopasture experiment

Within the simulated silvopasture experiment at the Castro de Rey Farm (excluding the adjacent pasture), soil depth and tree species were significant factors in soil carbon storage in each of the soil fractions (Table 4-3). Jobbagy and Jackson (2000) report on the vertical distribution of soil organic C from ~2,700 soil profiles globally. Surface soil layers contain a majority of SOC, as they are most affected by the plant communities above, with forested biomes containing a higher percentage of SOC than found in grasslands and shrublands in the top 20 cm of soil, when compared to the first meter of soil (Jobbagy and Jackson, 2000). In this study, soil depth was a significant factor affecting SOC storage within the simulated silvopasture experiments.

Another significant main treatment factor that determined SOC storage is tree species (Table 4-3). Kasel and Bennett (2007) found a 30% decrease in SOC as native broadleaf forests (*Eucalyptus* spp.) were converted to exotic pine plantations in temperate Australia. Conversion of cropland (as is the case at the Castro de Rey Farm) to broadleaf species plantation had little effect on SOC, but conversion of cropland to pine plantations significantly decreased SOC (Guo and Gifford, 2002; Kasel and Bennett, 2007). While species selection for silvopasture establishment clearly had an effect on SOC storage to one meter at the Castro de Rey Farm (with birch as the preferred species for improving SOC storage), the interactive effect of species versus spacing (Table 4-3) supersedes the general explanation for differences in soil carbon found under each tree species. The 3x4 m spacing was shown to lead to greater SOC storage in birch than pine in all soil fractions, (Figure 4-13, 4-18, 4-22, 4-26). At the 2x2 m, spacing SOC storage was similar among tree species.

Neither species nor fertilizer effects were significant main treatment effects on SOC storage in each of the soil fractions, but their significant interaction demonstrated some
differences among fertilizer and spacing combinations (Table 4-3). Trees spaced at 2x2 m generally had more SOC underneath with mineral fertilizer than in 3x4 m spaced trees in all soil fractions. The reverse trend was true for biosolid fertilizer applications, as soils underneath 3x4 m spaced trees demonstrated more SOC than 2x2 m spaced trees in all soil fractions (data not shown). For control fertilizer treatments (no fertilizer application), no differences in SOC were found among tree spacing and fertilizer combinations in each of the soil fractions.

**Carbon in soil fractions**

Separation of soil particles by physical fractionation revealed that the 250 – 2000 µm size class demonstrated greater soil dry weight in the upper 50 cm of soil, as the 53–250 µm size class increased with soil depth and in the 75-100 cm depth, was greater than the 250 – 2000 µm size class in weight (Figure 4-14). Recovery in the smallest <53 µm was never more than 10% at any depth, but did increase with depth. Carbon per fraction decreased from 250 – 2000 µm > 53–250 µm > less than 53 µm, and with decreasing soil depth. Percentage recovery in the 0 – 25 cm depth demonstrated the effects of tillage (Table 4-1), as recovery of 250 – 2000 µm under both tree species was higher than the open pastures. Six et al. (2002a) also demonstrated greater 250 – 2000 µm fraction dry weight recovery in forested and afforested sites, as compared to tilled agricultural fields in Ohio. They also found the opposite trend in the < 53 µm fraction (as is the case here also), as more silt+clay sized soil fraction was found under agriculture as compared to tree-based land uses.

Determination of C storage in the macroaggregate soil size fraction revealed that birch was better as promoting storage than pasture at all soil depths, and pine exceeded pasture in only the deepest sampling depth (Figure 4-16). Increased storage of C in the macroaggregate size is indicative of change in management activities, such as increased inputs from afforestation and
cessation of tillage, especially in surface horizons (Six et al., 2000a, Allison and Jastrow, 2006; Denef et al., 2007; Tan et al., 2007; D’Angelo et al., 2009). The pasture at Castro de Rey has been tilled three times since establishment in 1995. The continued mechanical disruption of macroaggregate formation under the pasture has lead to less storage in this fraction in the plow layer, 0 – 25 cm. Dry weight recovery of the 250 – 2000 µm fraction in the 0 – 25 cm depth (Table 4-1) was associated with an increase in SOC in the same fraction in birch, but not in pine (Figure 4-16). As such, birch (and the understory it promotes) contributes more to the process of macroaggregation than the pine at this soil depth. By maintaining a relatively open canopy, the birch promotes the growth of herbaceous species, which also contribute to SOC storage in the 0 – 25 cm depth. Allison and Jastrow (2006) found significantly less C associated with the macroaggregate size class on cultivated pastures as compared to native prairies in Illinois. John et al. (2005) also found more macroaggregate associated C in a forested ecosystem than in a grassland or wheat cropped ecosystem in Germany. The organic matter inputs, by trees and herbaceous species underneath, is the source of macroaggregate formation. Birch appears to improve SOC storage in the macroaggregate size class, as compared to pine and pasture. By maintaining a relative open canopy in the 3x4 m spacing (Figure 4-18) which promotes the growth of herbaceous which contribute to significantly greater SOC storage in the important 0 – 25 cm depth, where most SOC is stored.

Fertilizer and spacing treatments showed no major differences in the shallowest sampling depths (0 – 25 cm), but some differences in the deepest (75 – 100 cm) sampling depth (Figures 4-17 and 4-18). Differences in SOC storage between birch spacings, as mentioned above, are an exception (Figure 4-16). Tree-based systems were preferred for increasing C storage at the deepest depth in the macroaggregate fraction. Radiata pine planted in dense spacing contributed
more deep soil C in the macroaggregate soil size, as compared to the pasture. The limited rooting depth of pasture species likely limits C inputs to depth below 50 cm. Each deep rooted tree-based system was a preferred land use as compared to pasture for increasing macroaggregate-associated C in deeper horizons (Figure 4-15). As the whole soil analysis of C storage did not show this relationship; this further demonstrates the utility of physical fractionation to elucidate the changes in SOC that are most affected by management. The presence of more macroaggregates C in the soil is an indicator of less soil disturbance and/or more carbon inputs, and of long term storage of C in macroaggregate components (micro- and silt+clay sized aggregates).

Most of the differences in C storage in the smaller aggregate sizes (53-250 µm and <53µm) were limited to the 0 – 50 cm depth, where land management changes were most prevalent (tillage). In the 0 – 25 cm depth, tillage in the pasture lead to more free (un-occluded) silt+clay sized fraction recovery, which also translated into more SOC storage in the pasture as compared to the pine and birch stands (Table 4-1 and Figure 4-20). The smaller aggregates sizes are more stable in the soil, associated with recalcitrant plant/microbial residue and clay minerals, and have C associated with them that is relatively old (Blanco-Canqui and Lal, 2004), which represents carbon that was probably in the soil before the 1995 conversion from potato crop to pasture and silvopasture. As the process of macroaggregate formation is significantly reduced in pasture through continued tillage, and while the building blocks of macroaggregates (53-250 µm and <53µm sized aggregates) are present in pasture, the formation of new, smaller sized aggregates is also reduced (Six et al., 2000a). Carbon in stable microaggregate and silt+clay size class in the tree-based treatments is likely held in less stable macroaggregates, but the scope of the study cannot validate this conjecture. Six et al. (2000a) further separated macroaggregates by
mechanical disruption, and found micro- and silt+clay sized aggregates held within
macroaggregates, and through stable C isotope studies, have shown that new C moves from
macroaggregates to smaller aggregates over time. This process has been disrupted in pasture.
These micro- and silt+clay sized stable aggregates represent resistant SOC pools that help to
sequester atmospheric C over the long term, and while greater in pasture treatments, may not
truly reflect long term sequestration potential. Further examination of macroaggregate makeup is
needed.

A Comparison of Soil Carbon Storage at the St. Esteban and Castro de Rey Farms

Soil carbon storage to 100 cm depth was higher in the Castro de Rey Farm, as compared to
the St. Esteban Farm. A major difference between the sites is the significantly greater rainfall
that provides for a much higher NPP on the Castro de Rey Farm. Decomposing plant materials
are the building blocks for the formation of stable aggregates, and higher plant growth will lead
to higher organic matter inputs to the soil, through greater leaf fall, and root production and
turnover. Rodriguez-Murillo (2001) reported an r-squared of 0.937 for regression of mean annual
precipitation versus SOC concentration in Spain. Undoubtedly, precipitation plays a significant
role in setting the initial conditions for SOC storage to take place. Another major factor is
temperature. The Castro de Rey Farm in Galicia enjoys cooler average temperatures than found
at the St. Esteban Farm in Extremadura. Decomposition of organic matter is slower in cooler
climates, further contributing to higher SOC sequestration at Castro de Rey Farm compared with
the St. Esteban Farm. Other factors include land use and management. There is simply more tree
cover in Galicia, due to higher precipitation also, which promotes higher SOC over the
landscape.

Both sites have deep rooted trees that input carbon deeper into the soil than pasture
grasses, but a major limitation for tree growth is the limited rooting depth due to the shallow
depth to bedrock at the St. Esteban farm. Attaining samples to 100 cm was impossible in some sampling sites, as the sampling probe struck weathered rock sometimes at 80-90 cm (depths were noted to correct for C estimates). Soils at the Castro de Rey Farm are deep, and test holes more than 200 cm did not reach bedrock on this alluvial site. The additional ability of trees to explore these depths for moisture and nutrients is a major advantage over the upland St. Esteban Farm with limited soil depth. The greater exploration of soil volume by trees at Castro de Rey will not only lead to better growth of trees, but also increased volume for SOC storage.

Texture is another major difference between the two sites. The relatively coarse, sandy loam soil at Castro de Rey Farm reduces soil moisture retention capacity, negatively affecting plant growth, and increases SOC oxidation through a higher percentage of soil pore space, as compared to finer textures soils. The greater amount of clay in soils at St. Esteban, as compared to Castro de Rey, would favor SOC storage, as there is greater potential for the formation of stable organomineral complexes, a significant process for the stabilization of C for the long term (Hassink, 1997). Increased silt+clay in soils leads to greater SOC storage.

Another consideration for SOC storage is the quality and/or general recalcitrance of SOC inputs. The cork oak cambium is highly resistant to decomposition (Carvalho Mendes and Graca, 2009), and is used as a bottle stopper on vintage wines that can be stored for decades, if not centuries. It is likely that cork organic matter inputs (above and below ground) are also resistant to decay, and thus may represent soil inputs that long term in residence time. Pine needles have also been shown to be higher in lignin and resistant to breakdown compared to birch. The scope of this study does not exploration of the varying recalcitrance of SOC inputs, but it is likely that pine and cork oak, compared to birch, would be preferred for the higher relative recalcitrance of organic matter inputs, improving SOC storage. Fonte et al. (2009), on the other hand, reported
that the quality of organic input to the soil was not a significant factor in SOC storage in experimental plots in Ghana, and that tillage effects superseded input quality as a controlling factor in SOC storage.

Others have recognized the potential for SOC sequestration in Galicia as an important contributor to national C pools (Rodriguez-Murillo, 2001). As national land planners consider the most efficient means to improve SOC storage in Spain, the promotion of stable multi-component silvopasture systems in Galicia, as well as in the rest of Atlantic Spain, will preferentially increase SOC storage (as compared to drier sites) and reduce C losses from forest fires.

**Conclusion**

Silvopasture is a preferred land use option for the sequestration of C in deeper soil horizons. Results from the Dehesa silvopasture at the St. Esteban Farm indicate that there is significantly more C storage in soils influenced by individual cork oak trees as compared to further away from trees. At St. Esteban, as smaller trees in the study will eventually grow into larger size classes, this will lead to an increase in C storage in the whole field. Soil fractionation revealed significantly greater C storage in the 250–2000 µm size class at distances closer to the tree. In the past few decades, cork oak on the Iberian Peninsula has been decimated by a root pathogen (*Phytophthora cinnamomi*). This disease, along with poor management, has reduced cork oak densities in the Dehesa systems that have existed for centuries. This can only have the unfortunate secondary effect of reducing soil C inputs, and a possible reduction in existing C pools from where trees densities were once higher. Fortunately, some Dehesa landowners are replanting species to maintain higher densities, and as such, the environmental services provided by lost trees will eventually return.
At the Castro de Rey simulated silvopasture experiment, birch in the 3x4 m spacing and no fertilizer was shown to be a better improver of SOC storage than pine or pasture in the whole soil and macroaggregate size fraction. Both pine and birch based systems increased SOC storage in the deeper soil profile (50 – 100 cm), as compared to the treeless pasture. While the competition of forage species and trees for fertilizers leads to reduced tree growth in both birch and pine, soil C storage was generally increased by mineral and biosolids fertilizers in radiata pine, with mixed results for birch. Denser spacing of radiata pine in the silvopasture treatments generally lead to greater whole field C storage, with the reverse true for birch. Birch, planted at a less dense spacing, was shown to increase C inputs to deep soils, and its less dense canopy architecture (compared to radiata pine) allows for the continued production of understory forage species. The tree and forage components input carbon to the deep and subsurface soils, respectively, and, combined in silvopasture, is a preferred method for improving SOC storage in Atlantic Spain. While this study has exclusively considered soil C resources, when one includes the above + below ground C storage in biomass, silvopasture may be a preferred land use choice when C sequestration is a management goal. Fire incidence is reduced in silvopasture (a major advantage over pure plantations), and the tree and forage components may each represent significant inputs to SOC storage.

The preservation of the Dehesa silvopasture system in Mediterranean Spain and the promotion of new silvopastures in Atlantic Spain will improve SOC storage over current land uses. The Dehesa system of production is an excellent example of full resource utilization on oligotrophic soils in a semiarid environment. Dehesa species make use of highly seasonal rainfall, and produce a variety of traditional goods, and maintain a classically Spanish landscape. Its preservation maintains cultural identity and increases C storage in the soil. Where NPP is
higher in Atlantic Spain, exotic pine and eucalyptus plantations occupy significant areas of the landscape for pulp-wood production. As a land use alternative, silvopasture is being promoted to lessen fire risk, and maintain animal production, as well as meet the needs for high quality saw timber. Each system, compared to open pasture, has been shown to preferentially sequester SOC, with significant increases at greater soil depth. Spain has a vested interest in preserving and promoting these systems for increases in SOC storage.
Table 4-1. Mean mass recovery ratio of three soil fractions (fraction mass divided by whole soil mass) from wet-sieving procedure in the 0 – 25 cm depth at the St. Esteban and Castro de Rey Farms, Spain.

<table>
<thead>
<tr>
<th>Site: Treatment</th>
<th>St. Esteban Farm: distance from tree</th>
<th>Castro de Rey: vegetation cover types</th>
<th>Fraction, µm</th>
<th>2 m</th>
<th>5 m</th>
<th>15 m</th>
<th>pine</th>
<th>Birch</th>
<th>pasture</th>
</tr>
</thead>
<tbody>
<tr>
<td>250-2000</td>
<td></td>
<td></td>
<td></td>
<td>0.46a (0.04)</td>
<td>0.39ab (0.02)</td>
<td>0.31b (0.02)</td>
<td>0.69a (0.012)</td>
<td>0.68a (0.008)</td>
<td>0.62b (0.01)</td>
</tr>
<tr>
<td>53-250</td>
<td></td>
<td></td>
<td></td>
<td>0.29a (0.03)</td>
<td>0.35a (0.02)</td>
<td>0.33a (0.03)</td>
<td>0.26a (0.007)</td>
<td>0.25a (0.008)</td>
<td>0.23a (0.01)</td>
</tr>
<tr>
<td>&lt;53</td>
<td></td>
<td></td>
<td></td>
<td>0.22b (0.02)</td>
<td>0.26b (0.02)</td>
<td>0.36a (0.03)</td>
<td>0.05b (0.002)</td>
<td>0.05b (0.002)</td>
<td>0.07a (0.003)</td>
</tr>
</tbody>
</table>

Ratios for each row and site do not always sum to 1.0, as mass recovery from wet-sieving was not always 100%. Means within row and site followed by different lower case letter are statistically different at $p < 0.05$. Standard errors of the mean are shown in parenthesis.
Table 4-2. Mean height, diameter at breast height, and mortality for *Pinus radiata* and *Betula alba*, for each combination of two spacing treatments and three fertilizer treatments at the Castro de Rey Farm, Galicia, Spain.

<table>
<thead>
<tr>
<th>Spacing, m</th>
<th>2x2 m</th>
<th>3x4 m</th>
<th>2x2 m</th>
<th>3x4 m</th>
<th>2x2 m</th>
<th>3x4 m</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Fertilizer</strong></td>
<td>Biosolids</td>
<td>mineral</td>
<td>none</td>
<td>Biosolids</td>
<td>Mineral</td>
<td>none</td>
</tr>
<tr>
<td>Height, m</td>
<td>12.34a</td>
<td>11.52a</td>
<td>13.37a</td>
<td>12.17a</td>
<td>12.39a</td>
<td>13.22a</td>
</tr>
<tr>
<td>s.e.m.</td>
<td>(0.88)</td>
<td>(0.73)</td>
<td>(0.52)</td>
<td>(0.42)</td>
<td>(1.05)</td>
<td>(0.58)</td>
</tr>
<tr>
<td>Diameter, cm</td>
<td>15.26b</td>
<td>12.21b</td>
<td>15.67b</td>
<td>19.79a</td>
<td>19.45a</td>
<td>22.85a</td>
</tr>
<tr>
<td>s.e.m.</td>
<td>(1.59)</td>
<td>(1.31)</td>
<td>(1.04)</td>
<td>(1.70)</td>
<td>(2.66)</td>
<td>(1.45)</td>
</tr>
<tr>
<td>Mortality, %</td>
<td>22.2</td>
<td>37</td>
<td>3.7</td>
<td>37</td>
<td>59.3</td>
<td>25.9</td>
</tr>
</tbody>
</table>

Height and diameter and means followed by different lower case letters are significantly different at $p<0.05$. Means comparison was done within each row and species. Standard errors of the mean are reported in parenthesis.
Table 4-3. Analysis of Variance (ANOVA) results table for significant differences in mean C storage in four soil fractions between treatment factors in simulated silvopasture experiment at the Castro de Rey Farm, Galicia, Spain.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil fraction, µm</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Depth</td>
<td>&lt;0.0001</td>
<td>&lt;0.0001</td>
<td>&lt;0.0001</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td>Fertilizer</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Spacing</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>0.04</td>
</tr>
<tr>
<td>Species</td>
<td>0.0083</td>
<td>0.06</td>
<td>0.034</td>
<td>0.0065</td>
</tr>
<tr>
<td>Fertilizer*Depth</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Spacing*Dept</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Species*Depth</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Spacing*Fertilizer</td>
<td>0.008</td>
<td>0.0016</td>
<td>0.0091</td>
<td>0.0486</td>
</tr>
<tr>
<td>Species*Fertilizer</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Species*Spacing</td>
<td>&lt;0.0001</td>
<td>&lt;0.0001</td>
<td>&lt;0.001</td>
<td>0.046</td>
</tr>
<tr>
<td>Spacing* Fertilizer* Depth</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Species* Fertilizer*Depth</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>Species<em>Spacing</em>Depth</td>
<td>NS</td>
<td>NS</td>
<td>0.01</td>
<td>NS</td>
</tr>
<tr>
<td>Species<em>Spacing</em>Fertilizer</td>
<td>0.0663</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
</tbody>
</table>

Treatments combinations with no significant difference are labeled with NS; otherwise, p-values are provided. Note: this table does not include comparisons with adjacent pasture.
Table 4-4. Soil carbon (Mg C ha\(^{-1}\)) in the whole soil and three soil fractions for each treatment combination at Castro de Rey Farm, Galicia, Spain.

<table>
<thead>
<tr>
<th>Fraction</th>
<th>Spacing, m</th>
<th>Fertilizer</th>
<th>Species</th>
<th>Pinus Radiata</th>
<th>Betula alba</th>
<th>pasture</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>2x2 m</td>
<td>3x4 m</td>
<td>2x2 m</td>
<td>3x4 m</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Whole</td>
<td>0-25</td>
<td>83.12 ab</td>
<td>99.46 ab</td>
<td>74.53 ab</td>
<td>87.34 ab</td>
<td>80.45 ab</td>
</tr>
<tr>
<td></td>
<td>(s.e.m.)</td>
<td>(15.83)</td>
<td>(8.67)</td>
<td>(9.13)</td>
<td>(3.57)</td>
<td>(11.19)</td>
</tr>
<tr>
<td></td>
<td>25-50</td>
<td>28.88 a</td>
<td>23.13 a</td>
<td>19.14 a</td>
<td>16.63 a</td>
<td>14.05 a</td>
</tr>
<tr>
<td></td>
<td>(s.e.m.)</td>
<td>(15.41)</td>
<td>(2.23)</td>
<td>(4.72)</td>
<td>(2.97)</td>
<td>(4.43)</td>
</tr>
<tr>
<td></td>
<td>50-75</td>
<td>5.13 a</td>
<td>5.9 a</td>
<td>5.97 a</td>
<td>5.92 a</td>
<td>5.43 a</td>
</tr>
<tr>
<td></td>
<td>(s.e.m.)</td>
<td>(1.06)</td>
<td>(0.41)</td>
<td>(1.04)</td>
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<tr>
<td></td>
<td>75-100</td>
<td>5.11 a</td>
<td>6.77 a</td>
<td>8.4 a</td>
<td>4.93 a</td>
<td>3.41 a</td>
</tr>
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<td></td>
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<td>(1.90)</td>
<td>(0.88)</td>
<td>(0.58)</td>
<td>(0.71)</td>
</tr>
<tr>
<td></td>
<td>250 – 2000 μm</td>
<td>42.32 ab</td>
<td>58.87 ab</td>
<td>41.54 ab</td>
<td>60.25 ab</td>
<td>55.72 ab</td>
</tr>
<tr>
<td></td>
<td>(s.e.m.)</td>
<td>(5.81)</td>
<td>(3.23)</td>
<td>(3.64)</td>
<td>(3.66)</td>
<td>(8.72)</td>
</tr>
<tr>
<td></td>
<td>25-50</td>
<td>13.15 a</td>
<td>9.40 a</td>
<td>8.34 a</td>
<td>8.39 a</td>
<td>6.01 a</td>
</tr>
<tr>
<td></td>
<td>(s.e.m.)</td>
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<td>(0.89)</td>
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<td>(2.80)</td>
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<tr>
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<td>50-75</td>
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<td>1.23 ab</td>
</tr>
<tr>
<td></td>
<td>(s.e.m.)</td>
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<td>(0.23)</td>
<td>(0.36)</td>
<td>(0.29)</td>
<td>(0.30)</td>
</tr>
<tr>
<td></td>
<td>75-100</td>
<td>1.43 ab</td>
<td>1.94 ab</td>
<td>3.16 a</td>
<td>0.97 ab</td>
<td>0.56 b</td>
</tr>
<tr>
<td></td>
<td>(s.e.m.)</td>
<td>(0.35)</td>
<td>(0.11)</td>
<td>(0.77)</td>
<td>(0.04)</td>
<td>(0.02)</td>
</tr>
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</table>

For each row, treatment combination means that differ at p < 0.05. Standard errors of the mean are reported in parenthesis.
<table>
<thead>
<tr>
<th>Fraction</th>
<th>Species</th>
<th>Pinus Radiata</th>
<th>Betula alba</th>
<th>pasture</th>
</tr>
</thead>
<tbody>
<tr>
<td>Spacing, m</td>
<td>2x2 m</td>
<td>3x4 m</td>
<td>2x2 m</td>
<td>3x4 m</td>
</tr>
<tr>
<td>Depth, cm</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fertilizer</td>
<td>Biosolid</td>
<td>Mineral</td>
<td>Control</td>
<td>Biosolid</td>
</tr>
<tr>
<td>53 – 250µm</td>
<td>0-25</td>
<td>28.95 a</td>
<td>36.32 a</td>
<td>27.56 a</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(5.3)</td>
<td>(1.95)</td>
<td>(4.05)</td>
<td>(3.22)</td>
</tr>
<tr>
<td>25-50</td>
<td>10.74 a</td>
<td>9.56 a</td>
<td>9.05 a</td>
<td>5.02 a</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(4.4)</td>
<td>(1.58)</td>
<td>(1.32)</td>
<td>(1.73)</td>
</tr>
<tr>
<td>50-75</td>
<td>2.87 a</td>
<td>3.14 a</td>
<td>2.64 a</td>
<td>3.03 a</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(0.58)</td>
<td>(0.11)</td>
<td>(0.49)</td>
<td>(0.3)</td>
</tr>
<tr>
<td>75-100</td>
<td>2.18 a</td>
<td>3.77 a</td>
<td>4.3 a</td>
<td>2.31 a</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(0.23)</td>
<td>(1.08)</td>
<td>(0.66)</td>
<td>(0.42)</td>
</tr>
<tr>
<td>&lt;53µm</td>
<td>0-25</td>
<td>9.23 ab</td>
<td>9.14 ab</td>
<td>7.47 ab</td>
</tr>
<tr>
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<td>(2.08)</td>
<td>(0.78)</td>
<td>(0.69)</td>
<td>(1.78)</td>
</tr>
<tr>
<td>25-50</td>
<td>6.3 a</td>
<td>3.42 a</td>
<td>2.97 a</td>
<td>2.68 a</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(3.74)</td>
<td>(0.96)</td>
<td>(0.44)</td>
<td>(0.5)</td>
</tr>
<tr>
<td>50-75</td>
<td>0.99 a</td>
<td>1.18 a</td>
<td>1.54 a</td>
<td>1.21 a</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(0.22)</td>
<td>(0.24)</td>
<td>(0.26)</td>
<td>(0.17)</td>
</tr>
<tr>
<td>75-100</td>
<td>0.72 a</td>
<td>1.27 a</td>
<td>1.14 a</td>
<td>0.99 a</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(0.14)</td>
<td>(0.38)</td>
<td>(0.22)</td>
<td>(0.25)</td>
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</tbody>
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For each row, treatment combination means that differ at \( p < 0.05 \). Standard errors of the mean are reported in parenthesis.
Table 4-5. Whole field comparison of mean carbon storage (0 – 100 cm depth) underlying silvopastoral treatments combinations and an adjacent pasture at Castro de Rey Farm, Galicia, Spain.

<table>
<thead>
<tr>
<th>Cover</th>
<th>Pinus Radiata</th>
<th>Pasture alba</th>
<th>Pasture</th>
</tr>
</thead>
<tbody>
<tr>
<td>Spacing, m</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>2x2 m</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>3x4 m</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>3x4 m</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>2x2 m</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>3x4 m</td>
<td>--------------</td>
<td>--------------</td>
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</tr>
<tr>
<td>None</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>Fertilizer</td>
<td>Biosolid</td>
<td>Mineral</td>
<td>None</td>
</tr>
<tr>
<td>Mg C ha⁻¹</td>
<td>122.24ab</td>
<td>135.25ab</td>
<td>108.04ab</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(31.27)</td>
<td>(9.01)</td>
<td>(12.67)</td>
</tr>
<tr>
<td>Pasture</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>Mineral</td>
<td>114.25ab</td>
<td>103.34ab</td>
<td>80.94b</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(12.67)</td>
<td>(15.06)</td>
<td>(3.42)</td>
</tr>
<tr>
<td>Pasture</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>Mineral</td>
<td>96.28ab</td>
<td>161.18ab</td>
<td>105.98ab</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(8.53)</td>
<td>(18.48)</td>
<td>(10.84)</td>
</tr>
<tr>
<td>Pasture</td>
<td>--------------</td>
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<td>---------</td>
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<tr>
<td>Mineral</td>
<td>173.42ab</td>
<td>140.44ab</td>
<td>176.91a</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(19.27)</td>
<td>(22.47)</td>
<td>(11.91)</td>
</tr>
<tr>
<td>Pasture</td>
<td>--------------</td>
<td>--------------</td>
<td>---------</td>
</tr>
<tr>
<td>Mineral</td>
<td>176.91a</td>
<td>132.93ab</td>
<td>132.93ab</td>
</tr>
<tr>
<td>(s.e.m.)</td>
<td>(19.96)</td>
<td>(19.96)</td>
<td></td>
</tr>
</tbody>
</table>

Means labeled with different lower case letters (at $p < 0.05$). Standard errors of the mean are reported in parenthesis.
Table 4-6. Summary of significant differences in soil C between vegetative cover types for four soil fractions for different soil depths up to 100 cm depth at the Castro de Rey Farm, Galicia, Spain

<table>
<thead>
<tr>
<th>Cover type</th>
<th>Pinus Radiata</th>
<th>Betula alba</th>
<th>Pasture</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth range, cm</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-25</td>
<td>**</td>
<td>**</td>
<td>**</td>
</tr>
<tr>
<td>25-50</td>
<td>NS</td>
<td>NS</td>
<td>**</td>
</tr>
<tr>
<td>50-75</td>
<td>NS</td>
<td>NS</td>
<td>NS</td>
</tr>
<tr>
<td>75-100</td>
<td>NS</td>
<td>**</td>
<td>NS</td>
</tr>
</tbody>
</table>

Significant differences (at $p < 0.05$) are noted with **, those noted with NS are not significant.
Figure 4-1. Percentages of sand, silt, and clay in different soil depths up to 100 cm at the St. Esteban Farm, Extremadura, Spain.
Soil pH did not differ statistically per depth.
Figure 4-3. Soil carbon storage in the whole soil in different soil depths up to 100 cm as it varies from distance to *Quercus suber* in the whole soil at the St Esteban Farm, Extremadura, Spain.

At each depth, means that differ statistically (at $p<0.05$) are labeled with different lower case letters.
Figure 4-4. Soil size fraction recovery from wet-sieving by percent dry weight in four different soil depths up to 100 cm at the St. Esteban Farm, Spain.

At each soil depth, significant differences between fraction mean percentages are labeled with different lower case letters (at $p < 0.05$).
Figure 4-5. Soil carbon in the 250 – 2000 µm fraction in different soil depths up to 100 cm at three distances to *Quercus suber* at the St Esteban Farm, Extremadura, Spain.

At each depth, means that differ statistically (at $p<0.05$) are labeled with different lower case letters.
Figure 4-6. Soil carbon in the 53 – 250μm fraction in different soil depths up to 100 cm at three distances to *Quercus suber* at the St Esteban Farm, Extremadura, Spain.

At each depth, means that differ statistically (at $p<0.05$) are labeled with different lower case letters.
Figure 4-7. Soil carbon in the <53µm fraction in different soil depths up to 100 cm at three distances to Quercus suber at the St Esteban Farm, Extremadura, Spain.

At each depth, means that differ statistically (at $p<0.05$) are labeled with different lower case letters.
Figure 4-8. Percentages of sand, silt, and clay in different soil depths up to 100 cm at the Castro de Rey Farm, Galicia, Spain.
Figure 4-9. Soil pH in 0.1 M KCl for pine, pasture, and birch treatments in different soil depths up to 100 cm at the Castro de Rey Farm, Galicia, Spain.

Soil pH did not differ statistically per depth.
Figure 4-10. Mean C storage in the whole soil in different soil depths up to 100 cm for contrasts between silvopasture treatments and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-11. Mean C storage in the whole soil in different soil depths up to 100 cm for contrasts between all *Pinus radiata* and *Betula alba* silvopasture treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-12. Mean C storage in the whole soil in different soil depths up to 100 cm for contrasts between silvopasture fertilizer treatments and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain. A) *Betula alba* treatments. B) *Pinus radiata*.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-13. Mean C storage in the whole soil in different soil depths up to 100 cm for contrasts between silvopasture spacing treatments and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain. A) *Betula alba* treatments. B) *Pinus radiata*.

At each soil depth, significant differences between means are labeled with different lower case letters (at \( p < 0.05 \)).
Figure 4-14. Soil size fraction recovery (by dry weight) from wet-sieving by in four different soil depths up to 100 cm at the Castro de Rey Farm, Spain.

At each soil depth, significant differences between fraction mean percentages are labeled with different lower case letters (at $p < 0.05$).
Figure 4-15. Mean C storage in the 250 – 2000 µm soil fraction in different soil depths up to 100 cm for contrasts between silvoapsture treatments and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-16. Mean C storage in the 250 – 2000 µm soil fraction in different soil depths up to 100 cm for contrasts between all Pinus radiata and Betula alba silvopasture treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-17. Mean C storage in the 250 – 2000 µm soil fraction in different soil depths up to 100 cm for contrasts between silvopasture fertilizer treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain. A) *Betula alba*. B) *Pinus radiata*.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-18. Mean C storage in the 250 - 2000 µm soil fraction in different soil depths up to 100 cm for contrasts between silvopasture spacing treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain. A) *Betula alba*. B) *Pinus radiata*.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-19. Mean C storage in the 53 – 250 µm soil fraction in different soil depths up to 100 cm for contrasts between silvoapsture treatments and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-20. Mean C storage in the 53 – 250 μm soil fraction in different soil depths up to 100 cm for contrasts between all *Pinus radiata* and *Betula alba* silvopasture treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-21. Mean C storage in the 53 – 250 µm soil fraction in different soil depths up to 100 cm for contrasts between silvopasture fertilizer treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain. A) *Betula alba*. B) *Pinus radiata*.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-22. Mean C storage in the 53 – 250 µm soil fraction in different soil depths up to 100 cm for contrasts between silvopasture spacing treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.  A) *Betula alba*.  B) *Pinus radiata*.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-23. Mean C storage in the <53 µm soil fraction in different soil depths up to 100 cm for contrasts between silvoapsture treatments and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-24. Mean C storage in the <53 µm soil fraction in different soil depths up to 100 cm for contrasts between all *Pinus radiata* and *Betula alba* silvopasture treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-25. Mean C storage in the <53 µm soil fraction in different soil depths up to 100 cm for contrasts between silvopasture fertilizer treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain. A) *Betula alba*. B) *Pinus radiata*.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p < 0.05$).
Figure 4-26. Mean C storage in the <53 μm soil fraction in different soil depths up to 100 cm for contrasts between silvopasture spacing treatments, and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain. A) *Betula alba*. B) *Pinus radiata*.

At each soil depth, significant differences between means are labeled with different lower case letters (at $p <0.05$).
Figure 4-27. Photograph of individual tree and understory growth at the St. Esteban Farm, Extremadura, Spain.
CHAPTER 5
PHOSPHORUS RETENTION UNDER SILVOPASTORAL AGROFORESTRY VERSUS TREETESS PASTURES

Introduction

Phosphorus (P) is a key nutrient for the production of agricultural and forest products, and as such, is an essential component of most fertilizers. On the fertilized agricultural landscape, P can, over time, build up in concentrations that exceed the soil’s capacity to store it (Nair and Graetz, 2002; McDowell and Stewart, 2006; Koopmans et al., 2004; Nair et al., 2007; Michel et al., 2007). If removal through harvest does not sufficiently reduce soil concentration of P, the soil can become a source of environmental contamination, as excess P is leached through ground water or overland water flow. Phosphorus, and nitrogen (N) in particular, are vital elements of fertilizers but, given improper management, can pollute water bodies, leading to eutrophication and excess growth of algae. Algal blooms can reduce oxygen in water bodies and lead to fish die offs in the worst cases (Perkins and Underwood, 2002). Improper control of animal wastes can also lead to contamination. The management of P in the agricultural landscape has increasingly become an important aspect of environmental quality. The movement and loss of P in coarse textured agricultural soils (with high sand content) is of particular interest, since P movement is limited in soils with more potential P binding sites, such as in soils with high aluminum (Al), iron (Fe), and organic matter. P is sorbed to these surfaces and maintained in the soil in stable forms, and as such, movement from the soil is limited (Geisler et al., 2005). Soils with high clay content also reduce P movement through the soil, as hydraulic conductivity is low in such soils (Hornberger et al., 1998). The relatively large size of the P molecule (PO$_4^{3-}$) in the soil matrix (as compared to nitrate) normally reduces potential losses from soils with moderate clay content, as P has a small relative diffusion coefficient (Barber, 1995).
In Florida, USA, sandy soils dominate where an extensive agricultural and forestry industry thrives, with 3.7 million ha cropland in 2007 (USDA-ERS, 2009). The loading of P through fertilizer and manure additions has been identified as a source of eutrophication in several water bodies (Reddy et al., 1995; Nair and Graetz, 2004), including Lake Okeechobee in southern Florida. Silvopasture practices, as compared to treeless pastures are now being considered as a land use intervention to reduce P loading to water bodies, as extensive and deep roots of trees (Pinus sp., in particular) have been shown to reduce P loading in the soil (Allen et al., 2006; Nair et al., 2007a; Michel et al., 2007). Tree roots take up excess nutrients, (N and P) that may not be utilized in pastures that have shallower rooting systems than trees. This ‘safety net’ role of deep rooted trees is invoked here for the recapture of nutrients that would otherwise be leached beyond the rhizosphere, which is a source of environmental pollution (Allen et al., 2004). The expansion of silvopasture in P overloaded pasture soils may reduce environmental contamination and improve fertilizer use efficiency.

In response to the loading of P in sandy soils in Florida, researchers at the University of Florida (and elsewhere) have developed a relatively simple indicator measure of P loss from soils, the Phosphorus Saturation Ratio (PSR). The PSR is the ratio of Mehlich 3-P extract to the combined Mehlich 3 Fe + Al):

\[
PSR = \frac{\text{Mehlich 3-P}}{\text{Mehlich 3 (Fe + Al)}}
\]

Since P forms relatively stable complexes with Al and Fe, this ratio is an indicator of potential P losses from the soil. Ratios exceeding 0.15 are believed to be an indicator of P loss potential (Michel et al., 2007). From this proportion, it is also possible to calculate the Soil Phosphorus Storage Capacity (SPSC), which is the amount of P that can safely be applied to a soil before the PSR ratio exceeds 0.15 and P losses become a potential environmental problem:
SPSC = (0.15-PSR) * ((Mehlich 3-Fe/55.8) + (Mehlich 3-Al/27)) * 31(mg P/kg)  \quad (5-2)

Spain is a middle income industrial economy with a significant agricultural sector. With over 27% arable land, agricultural products include: grain, vegetables, olives, wine grapes, sugar beets, citrus, beef, pork, poultry and dairy products (Solsten and Meditz, 1988). The production of many of these goods requires application of fertilizers and the control of animal wastes. As in many parts of the world, poor management of nutrients (mostly N and P) has lead to contamination of fresh water bodies in Spain (Moreno et al., 2005, Fouz et al., 2009). Sites with sandy soils, in particular, are prone to loss of P from fertilizer and manure sources. No estimates for area of sandy soils under cultivation in Spain exist. Additionally, there are few studies on potential loss of P from different land uses in Spain. As such, this study seeks to compare the PSR and SPSC for silvopasture soils versus soils influenced by traditional treeless pastures on two sites in Spain. For a comparison of very different sites with silvopasture agroforestry systems, experiments were completed at the St Esteban Dehesa *Quercus suber* (cork oak) silvopasture in Extremadura province and the *Pinus radiata* (radiata pine) and *Betula alba* (birch) simulated silvopastoral experiment in Galicia province. The study was based on the premise that, as is the case in Florida, sandy soils that are not influenced by trees are more likely a source of P for the contamination of water bodies. Fertilizer treatments at the simulated silvopasture experimental plots presented an opportunity to examine the differences in PSR and SPSC from the no fertilizer treatments, with the possibility for the densely spaced tree treatments to show less P loss potential than less dense treatments, due to higher P extraction from greater root length density.
Methods and Materials

Study Sites

This study was carried out on the St. Esteban Dehesa Oak Farm near Plasencia in Extremadura province and at the simulated silvopasture experiments at the Castro de Rey Farm near Lugo, Galicia. Both sites are described in Chapters 3 and 4. Soil sampling was carried out on each site as described in Chapter 4, with the whole soil fraction (<2 mm) used for P analysis.

Mehlich 3 Extractions and P Storage Calculations

Mehlich 3 extractions were made on whole soil for all samples (Mehlich, 1984). Mehlich 3 extractant solution consists of 0.2 M CH₃COOH, 0.25 M NH₄NO₃, 0.015 M NH₄F, 0.013 M HNO₃, and 0.001 M ethylene diamine tetraacetic acid (EDTA). Soil and extractant solution were shaken for 10 minutes, filtered, and the resultant solution was analyzed for metals (Al, Fe, and P) by atomic absorption spectrophotometer.

The Phosphorus Saturation Ratio (PSR) was calculated for each soil sampling depth as a proportion of Mehlich 3-P to the Mehlich 3-Al and Fe (Equation 5-1). The Soil Phosphorus Storage Capacity (SPSC) takes into account a predicted PSR of 0.15 as the threshold for potential environmental contamination, estimating the amount of P (in kg ha⁻¹) that can safely be applied to a field:

Statistical Analysis

Mean PSR and SPSC per soil depth were estimated from Equations 5-1 and 5-2, respectively. At each depth, individual analysis of variance (ANOVA) was carried out for mean soil P, Fe, Al, PSR and SPSC at each depth in treatment classes which differed by site. At the St. Esteban Farm, distance from individual cork oak trees was compared at each soil depth (0 – 25, 25–50, 50–75, and 75–100 cm). At Castro de Rey, a comparison of pooled treatments for the simulated silvopasture experiments was made on the same depth classes as in St. Esteban for: all
silvopasture versus pasture, all radiata pine and birch versus pasture, and within each species, fertilizer and spacing treatments versus pasture. All data were stored and arranged with Microsoft Excel and analyzed with SAS version 9.2 for Windows. Significant differences between mean soil P, Fe, Al, PSR and SPSC for each depth were separated by the Waller-Duncan test at $p < 0.05$.

Results

St. Esteban Farm

**Phosphorus Saturation Ratio:** No significant differences in P, Fe and Al concentrations were identified at any depth among the tree sampling distances from individual cork oak trees at the St. Esteban Farm (Table 5-1 and Figure 5-1). At the St. Esteban Dehesa site, PSR generally increased with soil depth, with the 15 m distance demonstrating greater P saturation in all depths except the surface 0–25 cm depth (Figure 5-1).

**Soil Phosphorus Storage Capacity:** Soil Phosphorus Storage Capacity (SPSC, in kg P ha$^{-1}$) at the St. Esteban Farm increased from the 0–25 to the 25–50 cm depths and then decreased in deeper sampling depths (Figure 5-6). Storage capacity at the distances sampled did not differ statistically at $p < 0.05$.

Castro de Rey Farm

**Phosphorus Saturation Ratio:** In the 0–25 sampling depth, the only significant difference in concentration of the tested elements was for Al, which was significantly higher in the radiata pine 3x4 m spacing than in the pasture treatment. At Castro de Rey, PSR generally decreased with soil sampling depth. When comparing pasture versus all pooled silvopasture treatments, pasture had higher PSR at all depths (Figure 5-2), with significantly more P saturation in the pasture in the 50–75 and 75–100 cm depths (at $p < 0.05$). Species comparison with pasture also revealed significant differences at depth, as P saturation in pasture exceeded radiata pine at the
50–75 depth, and under both birch and radiata pine in the 75–100 cm depth (Figure 5-3). A breakdown of fertilizer experiments showed no significant differences between fertilizer treatments each for birch or radiata pine and an adjacent pasture (Figure 5-4). A comparison of radiata pine fertilizer treatments with pasture revealed that P saturation was greater at all depths in the pasture. Phosphorus saturation in the pasture also exceeded birch fertilizer treatments in all but the 0–25 cm depth. For spacing treatments for both species, significant differences were found only in the 50–75 and 75–100 cm depths (Figure 5-5). For each species, pasture saturation exceeded both the 2x2 and 3x4 m spacing regimes in all sampling depths. In the lowest sampling depths for both species, the less dense 3x4 m spacing treatments were significantly less saturated than the pasture ($p < 0.05$), with the denser 2x2 m spacing treatments showing statistically similar saturation as under the pasture and 3x4 m spacing for both species.

**Soil Phosphorus Storage Capacity:** At Castro de Rey, SPSC generally increased with depth, and it was higher under silvopasture than under pasture in all depths examined (see Figure 5-7), the difference being significant in the 50–100 cm depth. As for differences between species and the adjacent pasture, P storage under radiata pine exceeded pasture and birch in the 0–25 cm depth (Figure 5-8). In deeper sampling depths (both 50–75 and 75–100 cm), radiata pine had greater but statistically similar storage as birch, but statistically greater storage capacity than the adjacent pasture ($p < 0.05$). For fertilizer treatments per tree species, SPSC was generally higher in the 0–25 cm sampling depth in plots that were not fertilized (Figure 5-9). Pasture showed lower, non-significantly different storage capacity for all sampling depths when compared with silvopasture plots. A comparison of spacing treatments per tree species with an adjacent pasture revealed few significant differences (Figure 5-10). Only under the radiata pine 3x4 m spacing
treatments, in the 0–25 cm depth, did the SPSC value significantly exceed ($p < 0.05$) that under pasture.

**Discussion**

**St. Esteban Farm**

**Elemental analysis**

Results from the Mehlich 3 extractions for St. Esteban revealed no significant differences in P, Fe, and Al among the three distances to cork oak trees (Figure 5-1). Phosphorus was highest outside of the tree canopy in the 15 m distance at all depths examined, with the 5 m distance generally demonstrating lowest concentrations. The pattern of P availability contradicts what Moreno et al. (2005) found for a similar site with *Quercus ilex*, as available P was significantly higher under the tree canopy, while the mean P reported was well within the range of Mehlich 3-P from the present study. Fe levels were highest under the tree canopy (2 m) in the 0–50 cm depth and generally lower in all distances from the tree in 50–100 cm soil depth, but again, no significant differences were detected. Aluminum generally decreased below 50 cm depth, and was highest in the 25–50 cm depth. The high levels of Al in this layer reflect the much lower pH measured at that depth (Table 5-1 and Figure 4-1), as well as much lower organic matter (organic horizon was to 10–15 cm depth only). Mehlich 3-Al appears to be more available with the lower pH, and/or also not taken up by association with organic matter.

**Phosphorus saturation ratio**

PSR results indicate little potential for release of P from the soils under consideration. In both sites, PSR did not exceed 0.08. At the St. Esteban Farm, an unfertilized site, PSR was much lower as compared to Castro de Rey Farm (not exceeding 0.3 in the deepest sampling depth), and the trend in saturation was opposite. PSR increased with depth at all distances from the tree, with the 15 m sampling distance generally showing more saturation (Figure 5-1). A reduction in Al
and Fe binding sites due to increasing pH at greater distance from the tree may help to explain this. P did not appreciably increase with depth, but with decreased availability of Al and Fe at lower pH, the reduction in PSR could be expected under those conditions. While no significant differences were found between saturation levels, the treeless soil demonstrated greater P saturation from 25 to 100 cm depth. This saturation may also be attributed to lower tree root densities, which can be expected to take up more P in deeper soil profiles. In the shallow sampling depths (0–50 cm), the 2 and 5 m sampling distances had P saturation similar to the 15 m depth. This depth represents the majority of root length densities of non-tree species that occupy the site at both distances from the tree. Moreno et al. (2005) found that herbaceous species in a similar grazed Dehesa Holm Oak (Quercus ilex) site in Extremadura, Spain, had most of their roots in the top 10 cm, with an exponential reduction in root length density below. The maximum rooting depth for herbaceous plants rarely exceeded 80 cm on the site studied. Root length density of the herbaceous species varied inversely with distance to the trunk of Quercus ilex, as herbaceous root length density increased after 10 m. Root length density for the Quercus ilex, on the other hand, decreased only slightly from 2.5 to 20 m from the tree (Moreno et al., 2005). While Quercus ilex rooting patterns may be a proxy for that of cork oak, the density of roots of this species at varying distances from the trunk, remains unknown. Since the rooting density of these herbaceous species is mostly limited to the shallowest depths (0–50 cm depth), the assumption is that the majority of roots below this depth are from Q. suber. As such, differences in extraction of soil P at these deeper sampling depths are more likely mediated by tree roots, especially at closer distances to the trunk. While the trends in P saturation show a reduction in saturation closer to the tree at depth, no significant differences were indentified.
Soil phosphorus storage capacity

Soil P Storage Capacity (SPSC), calculated from PSR, was not significantly affected by distance to tree at all sampling depths at the St. Esteban Farm. The 25–50 cm depth showed higher acidity (non-significant at $p < 0.05$), and as such, the storage capacity of the soil was relatively high, with Fe and Al free to form stable associations with P (Figure 5-6). In the 0–25 cm depths, SPSC was lower closer to the tree, but the differences between sampling distances were non-significant. At least in this upper soil depth (0–25 cm), extraction is mediated by tree and grass influence: but again, the trend is not significant ($p > 0.05$). Moreno et al. (2005) found similar rooting densities for Quercus ilex at up to 20 meters from the tree trunk. Even though cork oak is a different species, it occupies a similar ecological niche (semi-arid Mediterranean Europe), and its roots may explore a similar soil volume as Quercus ilex. Both species have evolved to survive lengthy summer droughts, and the development of an extensive rooting system that fully exploits available soil moisture is key to survival in the semi-arid Mediterranean (Wesemael, 1993 and Moreno et al. 2007). As such, extraction of P by cork oak, and resulting PSR and SPSC may be similar at each of the sampling distances examined. Stable isotope labeling of fertilizer treatments at St. Esteban may help future researchers determine the dynamics of P in the Dehesa oak savannahs of southern central Spain.

The semi-arid nature of the St. Esteban Farm does not indicate potential for P to become a significant source of water contamination. Given that P storage capacities range from ~15 to 60 Mg ha$^{-1}$, the soils at St. Esteban have a large capacity for P storage. There was little evidence for erosion at St. Esteban, and losses of P through erosion are not likely. The St. Esteban Farm is not located near any streams, rivers, or lakes, and as such, is not a likely source of P contamination to water bodies. Conduits to groundwater may be present, but were not considered in the current study. Overall, there is minimal risk of P loss associated with this site.
Castro de Rey Farm

Elemental analysis

At Castro de Rey, Mehlich 3 extracts of P, Fe, and Al revealed a generally decreasing trend in elemental concentration through the sampling depths, except for Al which increased slightly in the 75–100 sampling depth across some treatment combinations (Table 5-2). The organic horizon at Castro extends to 35–42 cm, into the second sampling depth of 25–50 cm. There is a greater relative reduction in P and Fe to 100 cm depth, as compared to Al, but all metals generally decreased with sampling depth. The only significant difference in Al was between the pasture and the radiata pine at 3x4 m spacing with no fertilization in the 0–25 cm depth (Table 5-2). This may be a result of fertilization of the pasture, and the acidification of soil (and the resulting solubility of Al) that occurred under radiata pine plots (Mosquera-Losada et al., 2006). While fertilization demonstrated no significant effect between treatments, non fertilized plots demonstrated much less P loading as compared to all other treatments, but this trend is less pronounced in deeper soil sampling depths (Table 5-2). Elemental concentrations generally followed a decreasing trend with depth, associated with a reduction in organic matter (organomineral complexes) across all treatments.

Phosphorus saturation ratio

The PSRs at Castro de Rey were higher than those of St. Esteban, with several significant differences in P saturation at lower sampling depths between silvopasture treatments and the adjacent pasture. The PSR for all pooled silvopasture experiments was lower in the 50–75 and 75–100 cm sampling depths, reflecting the differences in rooting depth in silvopasture versus pasture (Figure 5-2). Pasture species are limited to the surface 50 cm of soil. When silvopasture treatments are divided by tree species, some differences are seen. Radiata pine showed lower PSR at all sampling depths, and as such, may be a better extractor of P than the pasture in the
deepest 50–100 cm depths (Figure 5-3). Sudmeyer et al. (2004) found a relatively high density of *Pinus radiata* roots (root interceptions) to 100 cm, which decreased significantly below this depth on a 16 year old stand in Australia. Michel et al. (2007) also found significantly lower PSR values for radiata pine silvopasture versus an adjacent pasture in Florida, USA (*Pinus elliotti* and *Paspalum notatum* combination). Similar to the current study, Michel et al. (2007) found these significant differences from the 50–100 cm depth at one site, and at another site, found differences in PSR for the entire soil profile.

Fertilizer treatments did not have a significant effect on PSR, but the trend in saturation is what would be expected, as no fertilizer treatments had lower saturation (Figure 5-4). The effects of fertilizer are less pronounced in the deeper soil depths, but pasture fertilization lead to a greater (non-significant) PSR at all sampling depths, except for birch in the 0–25 cm sampling depth. The effect of mineral fertilizer did not vary significantly from that of biosolids fertilizer in any of the silvopasture treatments. After the third year of the experiment, mineral fertilizer was annually applied to the biosolids plots. As such, most of the experiment saw mineral fertilizer applied to biosolids fertilizer plots.

Spacing treatments for radiata pine and birch revealed significant difference in PSR (at $p < 0.05$) in the 50-100 cm soil depth (Figure 5-5). Less dense, 3x4 m spacing treatments had significantly less PSR values than the pasture treatments, and statistically similar values to the 2x2 m spacing treatments for both species. One might expect that a more densely planted silvopasture would have more dense roots (than a less dense one), and thus be a better extractor of P in the soil. In the current study, this may not the case, as less dense stands had significantly lower PSR values than the adjacent pasture. High birch mortality in the 2x2 m spacing may help to explain these differences in P extraction. But for radiata pine, mortality was greater in the 3x4
m spacing where extraction was highest (Table 4-1 and Figure 5-5). Full exploration of soil by pine roots may support the lack of difference in extraction by pine in two difference spacings. Nair et al. (2007b) also found no significant differences in PSR within rows of slash pine silvopasture (*Pinus elliotti* and *Paspalum notatum*) where root length density might be expected to be higher within rows closer to trees as opposed to further away in the alley. They attributed this finding to the fact that slash pine has an extensive root system and P removals were probably similar in both locations within and between rows of trees. Likewise, at the Castro de Rey Farm, it is also possible that spacing density treatments have not necessarily lead to major differences in root length density. Sudmeyer et al. (2004) found that the radial extension of *Pinus radiata* roots at tree heights similar to this study would have lateral roots that extend 15 to 20 m in Australia, well beyond the 3x4 and 2x2 m spacing in the current study. Mortality among the trees would temporarily reduce tree root length density where a tree has died, but closely spaced neighboring trees would quickly fill the previously occupied soil volume with roots (as would non-tree plants where sun was available). As such, perceived differences in root length density between spacing treatments for each tree may not be valid.

**Soil phosphorus storage capacity**

Soil Phosphorus Storage Capacity generally increased with depth in the silvopasture plots and was similar across depths in the adjacent pasture at the Castro de Rey Farm, as ongoing fertilizer treatments affected SPSC in silvopastures versus pasture differently (Figure 5-7). The SPSC was greater in soils under silvopasture at all sampling depths and significantly greater in the 50–100 cm depth (at *p* < 0.05), as compared to the adjacent pasture alone. Previously mentioned differences in extraction and resulting P saturation explain why SPSC is higher under pooled silvopasture treatments, particularly at lower soil depths. While PSR demonstrated differences in P saturation, SPSC followed a similar, but inverse, trend. The slightly higher soil
bulk density found under pasture may have also contributed to higher SPSC estimates. Along with the increase in clay that occurred with increasing soil depth, the associated increase in soil density would provide more binding sites for P, and increase storage. This may explain why SPSC increases with increasing soil depth on this site, in addition to being further away from surface fertilizer applications. The SPSC differences between pooled tree species treatments showed the same, yet inverse, trend as radiata pine appears to better extract P than the pasture at most depths, and is better than birch in the shallowest 0–25 cm depth (Figure 5-8). Another view may be that birch provides an understory environment for herbaceous species that quickly recycle P to the soil through relatively fast root turnover. The pine understory is no longer growing and roots present are only from pine. Guo (2007) found significantly greater fine root length, weight, and longevity below 50 cm soil depth for radiata pine versus an adjacent native pasture in temperate Australia. Brandtberg et al. (2004) found most root mass from Betula pendula and B. pubescens were found in the top 10 cm in a Swedish mixed forest, but do not provide information on root mass or density below 40 cm. The deeper rooting ability of radiata pine (Sudmeyer et al, 2004) versus pasture leads to greater P extraction at lower depth which possibly accounts for the lower PSR and higher SPSC at these depths. Lack of adequate information on birch rooting depth limits the comparison of differences between pine and birch.

Differences between SPSC for fertilizer treatments were not significant for both species, but trends follow the inverse of PSR (Figure 5-9). The non-fertilized treatments had the highest storage capacity in the 0–25 cm depth, and pasture had the lowest. The lack of fertilization likely lead to greater, even if non-significant, P storage differences in the unfertilized plots, and the P loading was less than in fertilized plots. Soil P Storage Capacity did not differ in spacing treatments for birch, but for radiata pine, significantly more storage was estimated for 3x4 m
spacing for radiata pine silvopasture treatments than the pasture at 0–25 cm depth. The trend for SPSC was also greater across all soil sampling depths, but only significantly greater (at \( p < 0.05 \)) in the 0–25 cm depth. There was no statistical difference in the radiata pine 2x2 m versus 3x4 m spacing at any depth, indicating no difference in P extraction capacity between spacing treatments. Rooting density within each spacing treatment may not differ between the radiata pine and birch plots, but radiata pine at 3x4 m spacing showed significantly more storage capacity than pasture in this study. The differences in rooting depth between tree species and pasture may explain greater SPSC in radiata pine treatments in deeper profiles, as extraction of P proceeds in deeper soil profiles where trees preferentially extract P where pasture grass root depth is limited to more surface horizons.

Above ground P storage form and amount may shed light on why radiata pine had a greater SPSC in the 0–25 cm depth. Differences in mineralization rates for each tree species’ leaf litter (Oi/e horizon) may help to explain differences in P availability, saturation and storage capacity. On all plots, several tons of forage were harvested over the course of the 14 year experiment (Fernandez-Nunez, 2007), representing loss of P from the system. Production of forage under the radiata pine was significantly reduced over the course of the experiment with the accumulation of radiata pine needles on the forest floor (Fernandez-Nunez, 2007). Mineralization of these radiata pine needles is slow compared to birch, which demonstrated no accumulation of a leaf layer. The leaf litter under radiata pine is a significant source of P to growing plants, but first mineralization must take place to make the organic P available to the plant. Wesemael (1993) found that decomposition of Pinus sp. needles began only after one year of being on the forest floor, and as such, a leaf litter (duff layer) forms. Pine needle accumulation also provides a mulching effect, improving moisture retention under radiata pine. Furthermore, Gonzales-
Hernandez et al. (2003) found 100% hydrolyzable tannins in birch. Tanins are polyphenolic aromatic compound that reduce decomposition of plant material, and they play an important role in buffering plants against nutrient losses in sites with low natural nutrient abundance (Northup et al., 1995). Pines are generally higher in tannins, and as such, mineralization of the leaf litter will be reduced in radiata pine as compared to birch (Northup et al., 1995; Gonzales-Hernandez et al., 2003; Kraus et al., 2003). Pine needles accumulate on the forest floor, essentially storing nutrients (like P) that become available to the plants over time.

The time frame for replenishing soil nutrients depends on factors that affect mineralization, including chemical composition, environmental factors, and the presence of microbial decomposers. Given all similar environmental conditions at Castro de Rey Farm, the chemical composition of pine needles helps to reduce and slowly supply P to the soil. The important role leaf litter plays in nutrient cycling is apparent from experiments in Florida, USA, as experimental removal of pine leaf litter lead to significantly lower soil P and growth in *Pinus elliotti* on a sandy soil (Lopez-Zamora et al., 2001). Phosphorus is being stored in pine leaf litter, locked up, and more slowly released to the soil over time. This may help explain why P removals were greater under pine, as the quick recycling ability of grasses in the pasture is reduced by P buildup in radiata pine needles. The relatively fast mineralization of *Betula sp.* (compared to *Pinus sp.*, Hobbs et al., 2005) may result in similar nutrient cycling (PSR/SPSC) as in the pasture in shallow depths. Reduced mineralization may be an adaptation to nutrient removing spring snow melts and floods, and existence on otherwise infertile soils, where buffering against nutrient loss helps plants survive challenging growing conditions (Northup et al., 1995; Northup et al., 1998; Gonzales-Hernandez et al., 2003). Phosphorus is, of course, also stored in all aboveground biomass, and as such, all silvopasture treatments had more overall site P in aboveground pools.
Assuming a relatively similar P composition of biomass between tree species (kg P kg$^{-1}$ dry matter), pine has much greater aboveground biomass than birch (Table 4-1 and Figure 3-3), and P extraction over time for incorporation into aboveground biomass is likely greater.

Potential for P loss from the Castro de Rey Farm is higher for the adjacent pasture as compared to silvopastoral treatments, especially at greater sampling depths. While PSR did not exceed 0.10 for any of the treatments, the continued P loading during a typical 30 year rotation may not be matched by P removals, resulting in a PSR that exceeds the threshold or critical limit for causing contamination. In particular, the P removal by pasture grasses in the adjacent pasture is not matched by removals by trees that occur in silvopasture at depths greater than 50 cm. Care must be taken to avoid potential contamination resulting from the pasture land uses with similar P loading. The Castro de Rey Farm is located less than 2 km from the Mino river, with conduit ephemeral streams bordering the site. As such, during heavy periods of precipitation, which are common in winter in Galicia, P loss by washing of soil through overland flow of water is possible. In winter, overland flow of water Castro de Rey Farm can be ‘like a river’ (Jose Javier Santiago-Freijanes, personal communication, January 2008). Losses of P from agricultural lands in Galicia in short term storms has been identified a significant source of P to water bodies (Fouz et al., 2009). While PSR and SPSC results indicate lower risk for loss of P from soils to water bodies, the continued loading from fertilizer treatments (especially on the pasture), and proximity to water bodies indicates moderate potential for P loss from the site. While underground water flows were not considered in this study, the existence of immediately adjacent overland water conduits increase risk of P contamination. Tree based land uses, such as silvopasture, are able to extract excess P from the soil, and reduce the risk of P loss during these overland water flow events. Implementation of pine silvopasture is recommended to reduce excess buildup of P.
Conclusion

For an agricultural field to become a source of environmental contamination, two criteria must be met. First, the field under question must have a relatively high P Saturation Ratio, exceeding 0.15 in this case, which indicates possible losses from the field. The second criterion is the presence of a nearby means by which contamination can reach and pollute water bodies. These can be a stream, river, or below ground conduit to a water body. Precipitation sufficient for P to make use of these conduits is also required. If soil water passes the vadose zone of other fields, P contamination from one field can be spread to another, or alternatively, P can be held where P saturation is less, avoiding potential contamination. On both sites in this study, the overall potential for P loss is not high. At St. Esteban, lack of fertilizer additions, silty soil texture, dry conditions, and the presence of trees that more fully occupy the soil volume are likely causes for more P retention in the system. As an upland rocky site, no obvious conduit to a water body is present at St. Esteban. At the Castro de Rey Farm, fertilizer applications lead to greater overall P loading in the pasture as compared to radiata pine silvopasture. At Castro de Rey, soils are sandy loams and a conduit to a significant local water body is present on site, and prevalent overland flow may allow for more P loss in the upper soil profile. As such, potential for P loss is greater at this site than at St. Esteban, but moderate nevertheless. An examination of water soluble P in the shallowest profiles may help to demonstrate practical P loss potential through overland flow, and provide an estimate of a critical PSR for local soil conditions. Radiata pine silvopasture, in particular, is better at removing P from the deeper soil profiles, and is a preferred land use when reduction of P loss is a management goal.
Table 5-1. Mean Mehlich-3 phosphorus, iron, and aluminum, and water soluble phosphorus (WSP) in different soil depths up to 100 cm at varying distances from individual *Quercus suber* trees at the St. Esteban Farm, Extremadura, Spain.

<table>
<thead>
<tr>
<th>Soil depth, cm</th>
<th>Distance to tree, m</th>
<th>Mehlich 3-P Mean (s.e.m.)</th>
<th>Mehlich 3-Fe Mean (s.e.m.)</th>
<th>Mehlich 3-Al Mean (s.e.m.)</th>
<th>WSP mean (s.e.m.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-25</td>
<td>2</td>
<td>9.21 (0.68)</td>
<td>15.9 (36.97)</td>
<td>80.3 (234.3)</td>
<td>0.897 (0.36)</td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>7.63 (0.74)</td>
<td>15.6 (27.44)</td>
<td>1077 (212.7)</td>
<td>0.335 (0.20)</td>
</tr>
<tr>
<td></td>
<td>15</td>
<td>11.12 (1.85)</td>
<td>162.4 (31.34)</td>
<td>1300 (351.3)</td>
<td>0.540 (0.20)</td>
</tr>
<tr>
<td>25-50</td>
<td>2</td>
<td>8.13 (0.83)</td>
<td>104.3 (29.22)</td>
<td>1463 (409.6)</td>
<td>0.079 (0.04)</td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>7.49 (1.34)</td>
<td>65.2 (8.4)</td>
<td>1614 (288.8)</td>
<td>0.016 (0.01)</td>
</tr>
<tr>
<td></td>
<td>15</td>
<td>10.23 (1.72)</td>
<td>63.8 (8.43)</td>
<td>1298 (268.6)</td>
<td>NA</td>
</tr>
<tr>
<td>50-75</td>
<td>2</td>
<td>7.71 (0.69)</td>
<td>57.0 (8.52)</td>
<td>895.8 (271.2)</td>
<td>NA</td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>7.91 (1.39)</td>
<td>52.0 (4.24)</td>
<td>930.8 (207.6)</td>
<td>0.46 (0.46)</td>
</tr>
<tr>
<td></td>
<td>15</td>
<td>9.31 (1.07)</td>
<td>59.1 (7.75)</td>
<td>888.1 (317.5)</td>
<td>NA</td>
</tr>
<tr>
<td>75-100</td>
<td>2</td>
<td>7.80 (0.99)</td>
<td>49.6 (6.02)</td>
<td>411.2 (109.0)</td>
<td>NA</td>
</tr>
<tr>
<td></td>
<td>5</td>
<td>7.40 (0.65)</td>
<td>49.6 (8.23)</td>
<td>585.1 (147.8)</td>
<td>NA</td>
</tr>
<tr>
<td></td>
<td>15</td>
<td>8.86 (1.28)</td>
<td>64.1 (11.52)</td>
<td>372.3 (143.9)</td>
<td>NA</td>
</tr>
</tbody>
</table>

For each column, no significant differences were found between element concentrations. Water soluble phosphorus values below detection limit are labeled with NA.
Table 5-2. Mean phosphorus, iron, and aluminum for different soil depths up to 100 cm for individual silvopasture treatments and an adjacent pasture at the Castro de Rey Farm, Galicia, Spain.

<table>
<thead>
<tr>
<th>Extraction</th>
<th>Species</th>
<th>Spacing, m</th>
<th>Fertilizer</th>
<th>Mean (s.e.m.)</th>
<th>Mean (s.e.m.)</th>
<th>Mean (s.e.m.)</th>
<th>Mean (s.e.m.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-25</td>
<td>Pine</td>
<td>2x2</td>
<td>Biosolid</td>
<td>63.71 (23.31)</td>
<td>238.6 (79.64)</td>
<td>913.4ab (152.7)</td>
<td>1.06 (0.41)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Mineral</td>
<td>71.36 (21.1)</td>
<td>386.9 (134.6)</td>
<td>820.78ab (84.03)</td>
<td>1.61 (0.42)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Control</td>
<td>21.07 (3.48)</td>
<td>226.0 (53.61)</td>
<td>815.1ab (129.0)</td>
<td>0.01 (0.01)</td>
</tr>
<tr>
<td></td>
<td>Birch</td>
<td>2x2</td>
<td>Biosolid</td>
<td>78.33 (27.8)</td>
<td>246.0 (13)</td>
<td>1183ab (150.5)</td>
<td>0.35 (0.18)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Mineral</td>
<td>74.98 (20.26)</td>
<td>205.4 (108.2)</td>
<td>1252ab (72.7)</td>
<td>1.20 (0.72)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Control</td>
<td>28.21 (2.11)</td>
<td>305.2 (30.58)</td>
<td>1538a (226.8)</td>
<td>NA NA</td>
</tr>
<tr>
<td></td>
<td>Birch</td>
<td>3x4</td>
<td>Biosolid</td>
<td>70.76 (8.98)</td>
<td>326.4 (21.11)</td>
<td>851.6ab (127.9)</td>
<td>0.63 (0.53)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Mineral</td>
<td>96.26 (19.27)</td>
<td>339.7 (43.58)</td>
<td>816.2ab (73.43)</td>
<td>0.26 (0.18)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Control</td>
<td>37.48 (2.05)</td>
<td>255.9 (66)</td>
<td>880.8ab (73.43)</td>
<td>0.87 (0.18)</td>
</tr>
<tr>
<td></td>
<td>Pasture</td>
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For each column, significant differences between means are marked with different lower case letters. Water soluble phosphorus values below detection limit are labeled with NA.
Table 5-2. Continued

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For each column, significant differences between means are marked with different lower case letters. Water soluble phosphorus values below detection limit are labeled with NA.
Figure 5-1. Mean Phosphorus Saturation Ratio in different soil depths up to 100 cm and at 2, 5, and 15 meters from individual Quercus suber trees at the St. Esteban Farm, Extremadura, Spain.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
Figure 5-2. Mean Phosphorus Saturation Ratio in different soil depths up to 100 cm between pooled silvopasture treatments versus pasture at the Castro de Rey Farm, Galicia Spain.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
Figure 5-3. Mean Phosphorus Saturation Ratio in different soil depths up to 100 cm between pooled *Betula alba* and *Pinus radiata* silvopasture treatments versus pasture at the Castro de Rey Farm, Galicia Spain.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
Figure 5-4. Mean Phosphorus Saturation Ratio in different soil depths up to 100 cm between pooled fertilizer silvopasture treatments versus pasture at the Castro de Rey Farm, Galicia Spain. A) *Betula alba*. B) *Pinus radiata*.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
Figure 5-5. Mean Phosphorus Saturation Ratio in different soil depths up to 100 cm between pooled spacing silvopasture treatments versus pasture at the Castro de Rey Farm, Galicia, Spain. A) Betula alba. B) Pinus radiata.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 

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Figure 5-6. Mean Soil Phosphorus Storage Capacity in different soil depths up to 100 cm at 2, 5, and 15 meters from individual *Quercus suber* trees at the St. Esteban Farm, Extremadura, Spain.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
Figure 5-7. Mean Soil Phosphorus Storage Capacity in different soil depths up to 100 cm between pooled silvopasture treatments versus pasture at the Castro de Rey Farm, Galicia, Spain.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
Figure 5-8. Mean Soil Phosphorus Storage Capacity in different soil depths up to 100 cm between pooled *Betula alba* and *Pinus radiata* silvopasture treatments versus pasture at the Castro de Rey Farm, Galicia, Spain.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
Figure 5-9. Mean Soil Phosphorus Storage Capacity in different soil depths up to 100 cm between pooled fertilizer silvopasture treatments versus pasture at the Castro de Rey Farm, Galicia, Spain.  A) Betula alba. B) Pinus radiata.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
Figure 5-10. Mean Soil Phosphorus Storage Capacity in different soil depths up to 100 cm between pooled spacing silvopasture treatments versus pasture at the Castro de Rey Farm, Galicia, Spain. A) Betula alba. B) Pinus radiata.

Within each sampling depth, means are labeled with standard error bars and those with different lower case letters differ statistically at $p < 0.05$. 
The expansion of carbon sinks on agricultural lands is a promising means by which to reduce atmospheric CO₂, which is believed to be a major cause of global warming. In addition to protecting current carbon sinks, afforestation, reforestation, and implementation of agroforestry techniques help to increase total carbon stored on a unit of land. With harvest in agricultural and forestry systems, most of this captured carbon is returned to the atmosphere, and long term benefits associated with removal of atmospheric CO₂ are not realized. Storage of carbon in the soil, alternatively, has the potential to sequester carbon for longer terms, in deeper soils, and in association with recalcitrant chemical complexes and aggregates. With this rationale, this study examined the potential of silvopasture agroforestry systems to preferentially store soil carbon and P as compared with traditional treeless pastures in two distinct climatic regions of Spain.

This study determined the amount of C stored in four soil size fractions (<53 µm, 53 – 250 µm, 250 – 2000 µm, and < 2 mm) to 1 m soil depth. The C present in each of these fractions is known to, have different residence times in soils, and generally, the smaller the size fraction, the older the C will be. The protection of high C microaggregates inside of macroaggregates is a mechanism identified to protect C in the long term. The smallest silt and clay aggregates (<53 µm) form microaggregates inside macroaggregates, and as such, forming a hierarchical protection mechanism. Land uses that promote the protection of C inside macroaggregates help to preserve carbon for the long term. This study sought to determine the extent of C held in each of the particle size classes, and to compare storage in silvopastoral soils with pasture soils that are not influenced by the presence of trees.

Another aspect of this study was to assess the differences in phosphorus (P) loss potential for silvopasture versus adjacent treeless pastures for two sites in Spain. The Phosphorus
Saturation Ratio (PSR) and Soil Phosphorus Storage Capacity (SPSC), based on Mehlich-3 extracts of P, aluminum (Al), and iron (Fe), were calculated for each site. The ratio of P to Al and Fe is an indicator of possible environmental pollution, if the ratio surpasses 0.15. The SPSC is the amount of P that can be safely applied before saturation becomes a potential environmental risk. This study compared PSR and SPSC in soils of silvopastoral systems and pastures that are not influenced by trees. Saturation and storage capacity indicate site specific potential for P loss. The presence of a means by which P can move towards a water body is also required for pollution of water bodies to occur (river, stream, lake, etc.).

The study took place at the St. Esteban Farm in Extremadura province in central-western Spain (39°59' " N, 6° 6'15" W), and at the Castro de Rey Farm in Galicia province, Spain (43° 9'39" N, 7°29'54" W). In Extremadura, the climate is Mediterranean with hot droughty summers and mild winters. In Galicia, wet winters and mild summers with a late summer drought are typical. Net primary production of natural ecosystems is much higher in Galicia than Extremadura. At the St. Esteban Farm, cork oak, *Quercus suber*, was planted and maintained in a density of about 35 trees per hectare. On the Castro de Rey Farm, a simulated silvopasture experiment was conducted since 1995 with *Pinus radiata* (radiata pine) and *Betula alba* (birch) planted in two densities (2500 stems ha$^{-1}$, and 833 stems ha$^{-1}$), and were either fertilized with mineral fertilizer, or biosolid followed by mineral fertilizer, there was a ‘no fertilizer’ treatment as well. Soil samples to 100 cm were taken at three distances (2, 5, and 15 m) from trees at St. Esteban, and inside silvopastoral plots and an adjacent pasture at Castro de Rey. Soil samples were divided by depth: 0 – 25, 25 – 50, 50 – 75, and 75 – 100 cm. Soils were physically fractionated into four size classes (<53 µm, 53-250 µm, 250-2000 µm, and <2000 µm) and
analyzed for total carbon. Mehlich-3 extracts were made from the whole soil (<2000 µm), and analyzed for P, Al, and Fe concentrations.

Results from the physical fractionation and carbon analysis revealed significantly greater C storage in the macroaggregate fraction (250 – 2000 µm) in silvopasture dominated soils as compared with treeless pasture soils, particularly at lower depths (50 – 100 cm). At St. Esteban, storage of C was greater closer to the trees in three of the lower sampling depths, which is an indication that increased inputs from the tree helped to increase C in the soil in this fraction. A reduction in tillage closer to trees may also help to explain why less C was found further away from the tree in this fraction. Tillage is known to disrupt the formation and maintenance of macroaggregates in the soil, and C is known to be protected in macroaggregates (250 – 2000 µm). Carbon held by the macroaggregate fraction was higher at locations closer to the tree than away from the tree at St. Esteban, which is an indication of higher C storage in tree- incorporated systems. The increased rhizodeposition of C to deeper soil depths by trees provides the starting material for the formation of macroaggregates. The protection and expansion of trees in this region will help improve carbon sequestration in the soil.

The oak Dehesas are a cultural identifier for people of the region, providing jobs, a beautiful landscape, and moderate production on oligotrophic soils. The intangible benefit provided the trees through C sequestration has hitherto been unrecognized. Unfortunately, disease (Phytophthora cinnamomi) has reduced densities of Quercus suber over the past decades on the Iberian Peninsula, and the benefits of soil carbon sequestration are likely not being fully recognized.

Storage of soil C in tree based silvopastoral systems at the Castro de Rey Farm was also significantly higher in the 250 – 2000 µm soil fraction, as compared to an adjacent pasture,
particularly below 50 cm depth. Birch stands demonstrated significantly more soil C than the
treeless pasture in the 0 – 25 cm depth as well as below 50 cm in this macroaggregate size class.
Radiata pine only exceeded soil C storage in pasture below 50 cm depth. Pine and birch have
deeper roots that are able to input carbon deeper in the soil as compared to the pasture. It is likely
that the influence of herbaceous pasture species on soil carbon is limited below 50 cm. In the 53
– 250 µm soil fraction, pasture C storage was greater than radiata pine in the 0-50 cm depths
only. In the smallest silt-clay size fraction (<53 µm), pasture had demonstrated higher C storage
in the 0 – 50 cm depth for pine, and in the 0 – 25 cm depth for birch. The differences in storage
in this fraction reflect the tillage activities that are ongoing in the adjacent pasture at Castro de
Rey (tilled three times since 1995). Carbon within this size class in the silvopasture experiments
is likely stored within larger, macroaggregates, and tillage causes breakdown of these aggregates,
which explains why macroaggregate C is not well represented in the tilled pasture soil. When the
silt+clay sized fractions (aggregates) that contain C and are usually protected inside
macroaggregates are freed up with the breakdown of macroaggregates, the C is subjected to
greater decomposition. The significantly greater C storage found in deep soil profiles in
silvopasture as compared to the adjacent pasture at the Castro de Rey Farm could be due to the
effect of tillage and other disturbances in the pasture surface soil. This would lead to greater
breakdown of macroaggregate held C.

Birch appears to be a better land usage as compared to radiata pine for storage of carbon in
the 0 – 25 cm soil depth in the important macroaggregate soil size fraction. Fertilizer effects on
soil C sequestration were found to be minimal. While physical protection of carbon from smaller
aggregates inside macroaggregates is indicated for long term sequestration of carbon in
silvopasture, other C protection mechanisms are at work. Radiata pine has been shown to
enhance the creation of a duff layer of pine needles on the forest floor, which slowly mineralizes and provides fresh C (and other elements) to the soil over time. Birch, by contrast, does not develop this layer, and leaves falling from the tree are more quickly decomposed and incorporated into the soil. The decomposition of belowground inputs was not considered in this work. The differences in chemical composition of the species may help to explain why birch is preferred for carbon sequestration in the upper soil layers. The pasture and silvopasture had similar amount of Al, which forms stable mineral complexes with organic matter, but the lower inputs of C to lower depth in pasture may lead to reduced formation of these complexes below 50 cm.

A comparison of whole field calculations revealed that the mean C storage per site was 30 Mg C ha\(^{-1}\) at the St. Esteban Farm. The mean for the Castro de Rey Farm was 127 Mg C ha\(^{-1}\). Net primary production, which sets the amount of available C inputs, was much lower in Extremadura, and soils at St. Esteban were sometimes less than 100 cm deep. Higher relative temperatures at the St. Esteban Farm would lead to reduced soil C storage due to higher decomposition. For the silvopastoral experiments at the Castro de Rey Farm, pasture generally demonstrated greater soil C storage than most radiata pine treatments, and less than the 3x4 m spaced birch treatments. No statistically significant differences were found between the pasture and any of the treatments in the whole field soil C estimations. The main differences in C storage between pasture and silvopasture are found in the macroaggregate-size soil fraction, and in the differences in soil C below 50 cm depth. Soil C in this large macroaggregate is known to be more resistant to decomposition (as compared to un-aggregated organic matter), as is soil C that is found at lower depth because of reduced microbial activity caused by reduced oxygen infiltration. Finally, when one takes into consideration the considerable aboveground biomass in
silvopasture versus pasture, overall site C storage is always higher than that found in the pasture. This work focused solely on the C that is fully humified and incorporated into the soil, in an attempt to identify C pools that are resistant to decomposition. Harvest, fire, disease or pest outbreaks could drastically and quickly reduce aboveground C storage in the silvopasture, whereas stable C deep in the soil may not be so affected by these factors.

Results from the analysis of P saturation and storage capacity revealed several differences in P retention and extraction at the Castro de Rey Farm in Galicia. Results for PSR and SPSC at St. Esteban were non-significant. At Castro de Rey, differences in PSR were found between the pasture and silvopasture plots below 50 cm depth, as pasture showed significantly greater saturation than radiata pine in the 50 – 75 and 75 – 100 cm sampling depths. The densely planted 2x2 m spacing treatments for pine and birch showed significantly less saturation of P below 50 cm as compared to pasture. The deep-rooted trees are preferentially extracting P from these depths as compared to the adjacent pasture. These differences in PSR translate to the same differences for SPSC below 50 cm. The SPSC under pine was also significantly greater than under birch and pasture in the 0-25 cm depth. Extraction by roots underlying the pine needle duff layer (Oi/e), and the relatively slower release of P from the pine needles contribute to higher P storage capacity in the 0 – 25 cm layer. The birch and pasture plots, on the other hand, still have actively growing forage components, extracting and returning P to the soil in this layer. The SPSC was higher in unfertilized plots, but not at a significant level. The major differences between silvopasture and pasture in soil P saturation and storage capacity were found bellow 50 cm depth, where rooting is limited by pasture. Pasture grasses likely cannot access P in this depth, and as such, P builds up in the soil. While no indications are given for high risk of P loss from soils from these sites, the relative ease with which P could move to a significant water body
at the Castro de Rey Farm suggests potential risk of P contamination. Buildup of P over many years at the Castro de Rey Farm could be a future source of P contamination, especially in the pasture. The washing of upper soil layers through overland flow of water (assisted by an argillic soil horizon at 60 cm, which limits infiltration) could provide occasional P to local water bodies.

Silvopasture is a preferred method for the sequestration of C below 50 cm in the sites examined. Given the improved C protection mechanisms underlying silvopasture, the deep rooted trees are able to sequester more C in deeper soil horizons. Carbon at depth is likely more protected, as is C held in stable aggregates. While results for PSR and SPSC at St. Esteban are mixed, P loss was significantly reduced below 50 cm at the Castro de Rey Farm. The expansion of silvopasture in Galicia is likely to reduce P in deeper soil layers, avoiding potential future P loading that may cause contamination of water bodies. Combined with the other benefits from silvopasture such as improved revenue streams for landowners, biodiversity, and aesthetic value, these environmental amelioration potential of silvopasture make it an attractive land use alternative to pasture in Spain.
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BIOGRAPHICAL SKETCH

David S. Howlett, a native of Las Vegas, Nevada, graduated from Vassar College with a Bachelor of Arts degree in Biology in 1995. After, he served as a Peace Corps Volunteer in the Dominican Republic, managing an integrated conservation and development project in the country’s central highlands. Following Peace Corps service, David earned two Masters Degrees in International Affairs at Columbia University’s School of International and Public Affairs in 2000 and in Forest Science at Yale University’s School of Forestry and Environmental Studies in 2002. Returning to the Caribbean, David coordinated of the Forest Stewardship Program at the Virgin Islands Department of Agriculture for three years, administering a technical and financial assistance program for island residents. In August 2005, he began doctoral study at the University of Florida, School of Forest Resources and Conservation, with Dr. P.K. Ramachandran Nair as his major adviser. In December 2009, David graduated from the University of Florida with a Ph.D. in Forest Resources and Conservation, a minor in Soil and Water Science, and a certificate in Agroforestry.