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DINÁMICA DEL CARBONO Y OTRAS INTERACCIONES EN  
AGROECOSISTEMAS CON *Pinus ponderosa* (Dougl. Ex Laws) Y  
PRADERAS NATURALES ESTABLECIDOS EN SUELOS VOLCÁNICOS  
DEGRADADOS EN LA REGIÓN DE AYSÉN, PATAGONIA CHILENA

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## INTRODUCCION

## 1.1 El problema: importancia y justificación

El cambio climático mundial constituye una de las principales amenazas para el bienestar de la humanidad y se ha transformado en un fenómeno globalmente reconocido por casi todas las naciones del mundo (IPCC, 1997, 2001 y 2007). Este fenómeno, por su naturaleza global, incentivó a las Naciones Unidas a crear en 1988 el Panel Intergubernamental de Expertos sobre el Cambio Climático (IPCC), el cual es responsable de evaluar periódicamente los conocimientos sobre el cambio climático. A través de sus distintos grupos de trabajo, el IPCC ha hecho varias recomendaciones para reducir las emisiones de gases de efecto invernadero (GEI) y atenuar los efectos del cambio climático (IPCC, 2001).

Los GEI más preocupantes son el dióxido de carbono ( $\text{CO}_2$ ), el metano ( $\text{CH}_4$ ) y el óxido nitroso ( $\text{N}_2\text{O}$ ). El  $\text{CO}_2$  es responsable del 50% del efecto de los GEI (Bowman, 1990). Se estima que entre 1850 y 2010 la concentración atmosférica de  $\text{CO}_2$  aumentó de 280 a 388 ppm, siendo el incremento de 5% en los últimos 10 años (Tans, 2010). Cambios en los usos del suelo no afectan solamente las fuentes y los sumideros de C, sino que también impactan en las emisiones netas de  $\text{CH}_4$  y  $\text{N}_2\text{O}$  a la atmósfera. Pese a la baja concentración atmosférica de  $\text{CH}_4$  ( $1,7 \mu\text{mol mol}^{-1}$ ) y  $\text{N}_2\text{O}$  ( $0,32 \mu\text{mol mol}^{-1}$ ), comparada con la de  $\text{CO}_2$  ( $360 \mu\text{mol mol}^{-1}$ ) (Smith y Conen, 2004), su contribución al efecto invernadero es desproporcionada en relación a sus concentraciones, siendo de 20% y 6% para  $\text{CH}_4$  y  $\text{N}_2\text{O}$ , respectivamente (Paustian et al., 2006). Ello se debe a que el  $\text{CH}_4$  tiene un potencial de calentamiento global (GWP) 21 veces mayor al del  $\text{CO}_2$  en un plazo de 100 años, mientras que el GWP del  $\text{N}_2\text{O}$  es 310 veces superior. Además, debe tenerse en cuenta que el secuestro de  $\text{CO}_2$  es reversible, dependiendo del manejo del suelo, mientras que la emisión de  $\text{N}_2\text{O}$  no lo es.

Existen varios modelos que permiten estimar los niveles de carbono secuestrado en agroecosistemas. Entre ellos, se destaca el simulador CO2FIX versión 3.1, que permite estimar las cantidades y flujos de C en bosques nativos, plantaciones forestales, praderas y suelos, así como determinar el número de créditos de C transferibles de acuerdo con el Protocolo de



Kyoto (Schelhaas et al., 2004). El modelo usa pasos anuales para estimar el contenido de C en troncos, ramas, hojas, raíces, etc., mediante parametrización (características de especies, tipo de manejo, clima) ingresada por el usuario. El modelo incluye el sub modelo YASSO que utiliza datos climáticos básicos y estima la producción y calidad de litera para determinar el contenido de C en el suelo (0-100 cm de profundidad) y las fracciones de la materia orgánica.

El tercer informe publicado por el IPCC (2001) indica que el uso de sistemas agroforestales en tierras agrícolas degradadas tiene un enorme potencial de secuestro de C. Los árboles, por su naturaleza perenne, actúan como sumidero de CO<sub>2</sub> atmosférico de largo plazo. Por lo tanto, la introducción de árboles de rápido crecimiento en sistemas de cultivo agrícola, sobretodo cultivos perennes, puede aumentar la captura de C e incrementar a corto plazo los sumideros terrestres (Pandey, 2002). Los sistemas agroforestales permiten aumentar la producción total, combinando cultivos agrícolas, arbóreos y plantas forrajeras y/o animales, simultáneamente o secuencialmente, aplicando prácticas de manejo compatibles con la cultura de la población local (Dube et al., 2002; Garrett et al., 2000). Dada su aplicabilidad en terrenos agrícolas y en distintos programas de forestación y reforestación, la agroforestería se torna una estrategia muy atractiva y eficiente de secuestro de C (Montagnini y Nair, 2004).

La incorporación de árboles en sistemas agrícolas suele resultar en mayor cantidad de C almacenado por hectárea, sin interferir generalmente con la producción de cultivos para alimentación animal y humana; por el contrario, puede ser ventajoso, dependiendo del tipo de interacción que se produce. Políticas que favorecen su adopción ayudan a optimizar el secuestro de C y traen múltiples beneficios (Watson et al., 2000; Pandey, 2002).

El secuestro de C en suelos bajo sistemas agroforestales depende mucho de la proporción de árboles en el sistema productivo, del tipo de manejo de suelos y del uso de prácticas adecuadas de conservación de suelos. La captura de C en suelos bajo sistemas silvopastoriles con pasturas perennes no es la misma que bajo sistemas silvoagrícolas con labranza y disturbios frecuentes. Los suelos no tienen capacidad infinita para secuestrar C (Paustian et al., 2000). Las mayores acumulaciones de carbono orgánico de suelo (SOC) ocurren generalmente en suelos de textura fina. Suelos con alto contenido de arcilla y limo permiten la formación de

agregados estables que ofrecen una protección adicional al SOC (Brady, 1990; Ingram y Fernandes, 2001). Los agregados estables formados en suelos no labrados y protegidos de la erosión eólica e hídrica constituyen excelentes sumideros de C de largo plazo (Dube et al., 2009). Por otro lado, la comprensión de la dinámica de la materia orgánica del suelo (SOM) y su vínculo con la vegetación anterior o actual ayuda a determinar si el agroecosistema es sustentable, ya que el tipo de manejo de suelos afecta el contenido y la calidad de la SOM así como también la biomasa microbiana y la respiración del suelo (Carter, 2002).

El uso de sistemas agroforestales en tierras degradadas podría secuestrar globalmente  $820\text{--}2200 \times 10^6 \text{ Mg C año}^{-1}$ , en un período de 50 años. Se estima que en regiones templadas, el almacenamiento de C varía entre  $15 \text{ y } 198 \text{ Mg C ha}^{-1}$  (Dixon et al., 1994). En consecuencia, existe un interés cada vez mayor a nivel mundial por el estudio del potencial de secuestro de C y la reducción de las emisiones de otros GEI en agroecosistemas sustentables.

Una serie de estudios recientes realizados en regiones templadas del mundo han demostrado que los sistemas agroforestales pueden secuestrar más C que monocultivos agrícolas, praderas y plantaciones forestales y por lo tanto, son considerados sumideros reales de C (e.g. Nair y Nair, 2003; Abohassan, 2004; Montagnini y Nair, 2004; Sharrow e Ismail, 2004; Thevathasan y Gordon, 2004; Gordon y Thevathasan, 2005; Oelbermann et al., 2006; Peichl et al., 2006). Varios de estos estudios se realizaron en condiciones similares, aunque pocos de ellos midieron *in situ* las pérdidas de C provenientes de la descomposición de litera, respiración y lixiviación del suelo, flujos importantes a considerar para modelar los balances de C a nivel de sistemas. El trabajo realizado por Peichl et al. (2006) es probablemente el único estudio que compara los flujos y reservorios de C en sistemas agrícolas y silvoagrícolas.

Sin embargo, la mayoría de estos trabajos fueron realizados en estaciones experimentales, establecidas sobre tierras agrícolas favorables, y frecuentemente rodeadas por poblaciones rurales con necesidades socioeconómicas medianas, una serie de condiciones que no son tan comunes en otras regiones del mundo. Por ejemplo, en regiones remotas como la Patagonia chilena, que se caracteriza por sus extensas praderas degradadas y sus suelos volcánicos erosionados, la situación es otra. Allí, varios pequeños y medianos productores tienen cada vez más dificultad para

mantener la productividad de sus sitios de cria de ganado, una actividad tradicional que es también a veces su única fuente de ingresos. Ya que pocos de ellos pueden cambiar de rubro para dedicarse exclusivamente a la actividad forestal, el Instituto Forestal de Chile (INFOR) incentiva la adopción de sistemas silvopastorales en sus propiedades, como una práctica más sustentable del uso de sus tierras a fin de cubrir sus necesidades socioeconómicas y, a su vez, contribuir con la mitigación del aumento de la concentración de CO<sub>2</sub> atmosférico (Teuber y Ganderats, 2009).

En la Patagonia chilena, las experiencias con sistemas silvopastorales sustentables son muy limitadas, tanto bajo bosques nativos como en plantaciones. En la región de Aysén, gran parte de las cuencas han sido quemadas entre 1920 y 1940 para "abrir campos" a la ganadería, dejando las laderas de los valles expuestas a una inexorable erosión y degradación de los suelos formados por depósitos de cenizas volcánicas. Según Zagal y Córdova (2003), el manejo no sustentable de suelos volcánicos puede disminuir notablemente el contenido de materia orgánica. La ganadería incontrolada también contribuye a la destrucción actual de los bosques. Las áreas deforestadas, especialmente aquellas situadas en zonas de pendientes, sufren procesos erosivos tan graves que en algunos casos dificultan la reintroducción del bosque (Ortega y Rodríguez, 1994).

En la Región de Aysén la gestión de la mayoría de las propiedades ganadero-forestales medianas y grandes está caracterizada por un pensamiento pionero a corto plazo y su manejo no es sustentable, es decir, no existe una orientación hacia beneficios equitativos para las futuras generaciones (Silva et al., 1999). Según el Gobierno Regional de Aysén (2002) es necesario evaluar experiencias extranjeras y adaptarlas a la realidad de la zona, implementando por ejemplo prácticas silvopastorales. Con la reciente ratificación del Protocolo de Kyoto por Chile, la adopción progresiva de sistemas silvopastorales puede representar hoy más que nunca una forma de uso del suelo adaptada a la realidad climática y económica de la región.

Los escasos estudios sobre sistemas agroforestales en Chile, y la ausencia de investigaciones científicas que permitan modelar los balances de C en sistemas silvopastorales ubicados en la zona templada del hemisferio sur, más aun aquellos establecidos sobre suelos volcánicos degradados, hacen necesario estudiar y comparar a la brevedad los reservorios, flujos y ciclos de C de estos sistemas con aquellos encontrados en praderas naturales y bosques plantados. En la

Región de Aysén los pastizales cubren más de 1 millón ha y las plantaciones exóticas cubren un área superior a 30.000 ha (A. Sotomayor, com. pers., 2008). Por lo tanto, el objetivo general de este proyecto es investigar y modelar el potencial de captura de C en agroecosistemas típicos (bosque de *Nothofagus pumilio*, plantaciones de pino ponderosa, pradera natural, sistemas silvopastorales con *Pinus ponderosa*) de la Patagonia chilena.

## 1.2. Hipótesis y Objetivos

En este proyecto, las hipótesis específicas son:

1. Dada la utilización más eficiente de los recursos del sitio y la presencia de un microclima interno más estable y favorable, el sistema silvopastoral: (a) dará como resultado una producción anual de biomasa total y secuestro de C mayor que la pradera; además, (b) la presencia de ciclos de C aéreos y subterráneos altamente activos en el sistema silvopastoral permitirá secuestrar sustancialmente más C que la pradera y la plantación si ésta tuviera la misma densidad arbórea, pero no permitirá secuestrar más C que la plantación con el doble de densidad.
2. La cantidad de SOC a 0-20 cm de profundidad será significativamente mayor en la pradera que en la plantación, siendo intermedia en el sistema silvopastoral.
3. La mayor cantidad de N disponible en el suelo debida a la fijación de N atmosférico por las forrajeras leguminosas (*Trifolium* spp.) favorecerá (a) la acumulación de C en suelos bajo la pradera y el sistema silvopastoral, siendo mayor la cantidad de N en la pradera natural y menor en la plantación de pino ponderosa, y (b) se registrará un mayor crecimiento del componente arbóreo en el sistema silvopastoral y mayor cantidad de C secuestrado por árbol que en la plantación forestal.
4. Dada la presencia de suelo volcánico con propiedades físicas y químicas distintas de suelos no volcánicos, CO2FIX subestimaré la cantidad de C efectivamente secuestrado, y

5. Las emisiones de CH<sub>4</sub> y N<sub>2</sub>O serán máximas en la pradera y mínimas en la plantación, siendo el sistema silvopastoral una alternativa interesante para mitigar el efecto neto de los tres GEI bajo estudio, y a su vez la forma de uso del suelo más sustentable a largo plazo.

Los cinco objetivos específicos definidos para comprobar las hipótesis planteadas son:

1. Evaluar (a) la influencia de agroecosistemas típicos de la Patagonia sobre el contenido total de C de sus componentes, la biomasa microbiana y su respectiva respiración en un suelo volcánico, y sobre la distribución y (b) la calidad de la materia orgánica del suelo; además de calibrar el modelo CO2FIX para simular los contenidos de C en el suelo y la vegetación.
2. Medir (a) el tamaño de los reservorios de C secuestrado en la biomasa vegetal y suelos volcánicos en una pradera natural, una plantación de pino manejada (*Pinus ponderosa*), y en un sistema silvopastoral con *Pinus ponderosa*; y (b) medir los flujos anuales de C mediante la evaluación de las entradas (litera foliar, necromasa) y salidas (descomposición, lixiviación, respiración), considerando las variaciones de temperatura y humedad del suelo.
3. En base a lo anterior, desarrollar modelos de secuestro de C para cada tipo de uso del suelo.
4. Determinar el efecto de las leguminosas *Trifolium* spp. sobre el contenido en N y C total del suelo en los agroecosistemas, y sobre el crecimiento del componente arbóreo, y
5. Estimar las emisiones de CH<sub>4</sub> y N<sub>2</sub>O de cada tratamiento y su efecto neto en equivalente C, y evaluar cuál de ellos presenta el mejor potencial de reducción del calentamiento global.

Se investigó el Objetivo 1a en el Capítulo 1 a través de mediciones en terreno y en laboratorio, y el Objetivo 1b en el Capítulo 2 con las mismas mediciones *in situ* y mediante simulaciones usando el modelo CO2FIX. Los Objetivos 2a y 4 en el Capítulo 3 se investigaron mediante nuevas mediciones en terreno. Finalmente, se indagaron los objetivos 2b, 3 y 5 en el Capítulo 4 de esta tesis haciendo mediciones repetitivas en terreno y laboratorio y mediante una serie de cálculos que permitieron la construcción de los modelos de reservorios y flujos de carbono.

## CAPITULO 1

Dube F., Zagal E., Stolpe N.B., Espinosa M. 2009. **The influence of land-use change on the organic carbon distribution and microbial respiration in a volcanic soil of the Chilean Patagonia.** *Forest Ecology and Management* 257: 1695-1704.

**Abstract:** Depending on the rate of soil organic matter (SOM) formation and decomposition, soil-vegetation systems can be a source or sink of CO<sub>2</sub>. The objective of this study was to determine the influence of land-use change on SOM distribution, and microbial biomass and respiration in an Andisol of the Chilean Patagonia. Treatments consisted of degraded natural prairie (DNP), *Pinus ponderosa* plantations (PPP), a second-growth *Nothofagus pumilio* forest (NPF). The soil was classified as medial, amorphic, mesic Typic Hapludands. Soil microbial respiration and microbial biomass were determined in the laboratory from soil samples taken at 0-5, 5-10, 10-20 and 20-40 cm depths obtained from three pits excavated in each treatment. Physical fractionation of SOM was performed in soil of the upper 40 cm of each treatment to obtain the following aggregate-size classes: macroaggregates (>212 µm), mesoaggregates (212-53 µm) and microaggregates (<53 µm). Plant C content was 68% higher in PPP than in DNP and 635% higher in NPF than in PPP. Total soil and vegetation C content in both DNP and PPP were less than half of that in NPF. Total SOC at 0-10 cm depth decreased in the order DNP (7.82%) > NPF (6.16%) > PPP (4.41%), showing that land-use practices affected significantly ( $P < 0.01$ ) SOC stocks. In all treatments, microbial biomass C and respiration were significantly higher ( $P < 0.05$ ) in the upper 5 cm. Soil microbial respiration was also correlated positively with microbial biomass C and SOC. The different land uses affect the formation of organic matter, SOC and microbial biomass C, which in turn will affect soil microbial respiration. Conversion of DNP to PPP resulted in a 44% decrease of SOC stocks in 0-10 cm mineral soil. The largest amount of SOC was stabilized within the mesoaggregate fraction of the less disturbed system, NPF, followed by PPP. In the long term, formation of stable mesoaggregates in soils protected from erosion can behave as C sinks.

**Keywords:** Andisol, carbon sequestration, soil aggregates, soil microbial respiration.

## 1.1. Introduction

During the 1990s, terrestrial ecosystems captured approximately 36% of the total carbon liberated to the atmosphere by combustion of fossil fuels (IPCC, 2001). Consequently, there is a growing interest to study the potential for carbon sequestration in agroecosystems worldwide, including remote regions such as the Patagonia, Chile where large areas of degraded pastures and eroded soils are found, where the ranchers face increasing pressures to maintain the productivity of their land. The Aysén Region, located in the heart of Patagonia between Latitude S. 44° and 49°, covers an area of 110,000 km<sup>2</sup>, and the main forest types are evergreen (2.2 million ha), *Nothofagus pumilio* (1.4 million ha), *Nothofagus dombeyi* (0.6 million ha) and *Pilgerodendron uviferum* (0.5 million ha) (SAG, 1999).

The native forest that originally covered most of the Aysén Region has been reduced to 45% of the total because of fires that were used to "open fields" to cattle, leaving the valley slopes exposed to an inexorable erosion and degradation of the soils that formed in volcanic ash deposits. Overgrazing has also contributed to degradation of forests that are adjacent to the pastures (Gobierno Regional de Aysén, 2002). Deforested areas with steep slopes suffer extreme erosive processes, which has complicated reforestation efforts. The forest plantations that have been established on degraded soils cover approximately 30,000 ha and consist of exotic species, mostly *Pinus ponderosa*, *Pinus contorta* and *Pseudotsuga menziesii*.

Non-sustainable use of volcanic soils can notably diminish the content of organic matter, which can be reverted by sustainable management that can improve the soil physical and chemical properties, and increase organic matter content, which is known as "carbon capture or sequestration" in soils (Zagal and Córdova, 2003). The influence of land management on the carbon (C) contents in biomass and soils is well documented worldwide (Peichl et al., 2006; Huygens et al., 2005; Six et al., 2002; Ross et al., 2002, 1999; Post and Kwon, 2000) and soils are known to be the largest terrestrial C reservoir. According to Tate (1987), land-use changes can modify soil C contents because of the interactions between residue inputs to soil and the subsequent transformations mediated by soil microorganisms. Depending on the rate of soil organic matter (SOM) formation and decomposition, soil vegetation systems can

be a source or sink of CO<sub>2</sub> (Six et al., 2002). The understanding of SOM dynamics and its link with previous or current type of vegetation is also helpful to determine whether the ecosystem is sustainable, because land management affects the content and quality of SOM as well as microbial biomass and respiration in soil (Carter, 2002). However, there have been few comparative studies of the influence of land-use changes on organic matter, and microbial biomass and respiration in volcanic soils, especially in the Patagonia. In general, volcanic soils have unique physical and chemical properties compared with other soils. Volcanic soils (Andisols) commonly form in humid temperate climates and are dark in color because of high organic matter content and Al, Fe-humus complexes. Other properties include: low bulk density, fine crumb or granular structure, the presence of amorphous clay (allophane or imogolite) having a pH-dependent charge and high specific surface area, high water content at 1500 kPa tension, low exchangeable bases, and elevated retention of phosphate (Wada, 1985).

The objective of this study was to determine the influence of land-use change on SOM distribution, and microbial biomass and respiration in an Andisol of the Chilean Patagonia.

## **1.2. Materials and methods**

### *1.2.1. Site description*

The site was located in the San Gabriel Agroforestry Unit that was established in 2002 by the Forestry Research Institute (INFOR) and the Agricultural and Cattle Development Institute (INDAP) of the Chilean government. This module is located in the Mano Negra Sector, 30 km north of the city of Coyhaique, on a western exposed slope with 730 m altitude above sea level at Latitude S 45°25', Longitude W 72°00'. Within the module, there are several adjacent experimental land-use management practices (200 m apart), hereafter referred to as treatments: (1) more than 50-year-old degraded prairie with traditional management of cattle grazing - 5 ha (DNP), (2) 16-year-old thinned and pruned *P. ponderosa* exotic plantations - 5 ha (PPP), and (3) 150-year-old unmanaged second-growth *N. pumilio* forest - approximately 100 ha (NPF). It is worth mentioning that more than 50 years ago, the entire zone was originally



covered with indigenous forest, mostly *N. pumilio*, which has been burnt in diverse sectors to "open fields" to cattle-raising activities.

Treatments were imposed in March 2006 in a randomized complete design with three replicates (20-m spaced pits that consisted in one centralized pit at 100 cm depth that also served for the soil description and classification, and two adjacent ones at 40 cm depths). All replicates (or sampling plots) had a slope ranging from 10 to 15%, a uniform aspect and were located at similar elevations. In each sampling site, soil samples were taken at 0-5, 5-10, 10-20 and 20-40 cm depths, and also at 40-60, 60-80 and 80-100 cm depths in the central pit that served for classifying the soil.

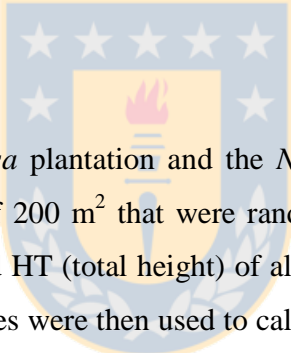
In 1991, the pine plantation was established, with an initial density of 2000 stems ha<sup>-1</sup> and a spacing of 2 m x 2.5 m; in 2003 it was thinned to 800 trees ha<sup>-1</sup>. In 2006, average height and diameter at breast height (DBH) were 6.3 m and 17.3 cm for the *P. ponderosa* trees, and 23.7 m and 32.5 cm for the *N. pumilio* trees, respectively. The natural prairie that gradually established itself after the native forest was cleared has traditional cattle-raising activities, with an animal density of 2 cows ha<sup>-1</sup>. The pastures presently consist of a mixture of perennial gramineas (*Dactylis glomerata*, *Holcus lanatus*, *Poa pratensis*), perennial leguminous plants (*Trifolium pratense*, *T. repens*), and other accompanying species. During the growth period of years 2004-2005, the prairie was fertilized with 16 kg N ha<sup>-1</sup>, 30 kg Mg ha<sup>-1</sup> and 79 kg S ha<sup>-1</sup>. In 2005-2006, 24 kg N ha<sup>-1</sup>, 17 kg Mg ha<sup>-1</sup> and 77 kg S ha<sup>-1</sup> were applied.

The climate is temperate with dry summers and cool, humid winters. Mean temperatures fluctuate between 12°C and 14°C in summer and 2°C and 3°C in winter (Silva et al., 1999). The annual precipitation varies from 1000 mm to 1500 mm. However, only 15% of the precipitations take place between December and February, coinciding with the windiest and warmest period. The soil is characterized as a well drained, deep and medium acidic, formed from relatively unaltered volcanic ash deposits (Silva et al., 1999). The soil has been classified as medial, amorphic, mesic Typic Hapludands (Soil Survey Staff, 2006).

### 1.2.2. Soil analysis

To check soil variability, the soil morphology was described in the central pit of each land-use area and samples were taken at 0-5, 5-10, 10-20, 20-40, 40-60, 60-80 and 80-100 cm depths (Soil Survey Staff, 1992). Soil samples were air-dried and sieved (2 mm mesh) for later analyses. Soil texture was determined using the hydrometer method (Day, 1965). The total C and N contents were determined by dry combustion (Wright and Bailey, 2001). Soil pH was measured using a 1:1 mixture of soil and water (McLean, 1982). Effective cation exchange capacity (ECEC) was measured using 1 M  $\text{NH}_4\text{O}_2\text{C}_2\text{H}_3$  for extraction of exchangeable cations Ca, Mg, K, and Na, and 1N KCl for determination of exchangeable Al (Thomas, 1982). Available N ( $\text{NO}_3\text{-N}$ ), P (Olsen) and K (Sadzawka et al., 2006), and % P fixation were also measured in the soils (Soil Survey Staff, 1992).

### 1.2.3. Tree and pasture biomass



An inventory of the *P. ponderosa* plantation and the *N. pumilio* forest was performed and consisted of five circular plots of 200 m<sup>2</sup> that were randomly established in each treatment, measuring the variables DBH and HT (total height) of all the trees using a diametric tape and clinometer. The measured variables were then used to calculate aboveground and subterranean biomass, using existing biomass functions for the climatic zone, that were specifically developed for pine plantations (Gayoso et al., 2002) and second-growth *N. pumilio* forest (Loguercio and Defosse, 2001; Weber, 1999). Partitioning of the aboveground biomass compartments was as follows: *P. ponderosa*, 47% of consisted of stems, 25% of branches and 28% of foliage; *N. pumilio*, 87% consisted of stems, 12% of branches and 1% of foliage.

To obtain the aboveground grass biomass in the prairie, the grazing material was manually harvested from nine randomly selected 0.25 m<sup>2</sup> quadrants. Then, the fresh biomass was dried at 65°C for 5 days and weighed once again to obtain dry biomass. To measure the subterranean biomass, the greatest quantity of roots growing under each square was carefully extracted, cleaned and washed in slow running water to remove soil from roots, and then dried at 65°C for 5 days and weighed. The values obtained were used to calculate the total grass

biomass weight per hectare. We assumed that approximately 50% of the biomass consists of carbon (IPCC, 2001) to calculate the carbon content of the vegetation.

#### *1.2.4. Soil microbial respiration*

Soil microbial respiration was determined in the laboratory from soil samples (three replicates, 0-5, 5-10, 10-20, 20-40 cm depths) obtained from the pits of each treatment. The incubation method was used to measure the CO<sub>2</sub> evolved by the microbial biomass (Rowell, 1994). Sieved 25 g moist soil samples were placed in 1-liter incubation jars (water-filled pore space (WFPS) = 60%), together with glass essay tubes containing 7.5 ml of NaOH. The same procedure was used for the controls to measure the ambient CO<sub>2</sub> concentrations in the jars, but without the presence of soil. The incubation jars were sealed with Parafilm and kept in an incubation chamber at 22°C for 3 days, after which the essay tubes were retrieved from the jars, from which 1 ml of NaOH in duplicate was extracted and added to 2 ml of BaCl<sub>2</sub> in separate essay tubes. The solutions were titrated with HCl (0.1 M), adding first a few drops of phenolphthalein. Evolved CO<sub>2</sub> was calculated as the difference between controls and samples and expressed per gram of dry soil.

#### *1.2.5. Microbial biomass C*

Soil microbial biomass C was determined through the chloroform fumigation-extraction method (Vance et al., 1987) and subsequent quantification of ninhydrin-N according to Joergensen and Brookes (1990). Sieved moist soil samples (25 g) (60% WFPS) were placed in desiccators together with a beaker containing 25 ml ethanol-free chloroform for incubation. Sieved moist soil samples were also incubated without fumigation (controls). According to the Rothamsted method (Ocio and Brookes, 1990), microbial biomass C was calculated as follows: biomass C = 31 x (ninhydrin-N). The efficiency factor is used to transform to 100% the value obtained by fumigation, since there is only partial cell destruction using the fumigation-extraction method (Jenkinson, 1988).

### 1.2.6. Physical fractionation

Physical fractionation of soil organic matter was performed in soil of the upper 40 cm of each treatment following the method described by Feller et al. (1991) to obtain three aggregate-size classes. Samples of 50 g of dry soil were weighed and placed in 250 cc polyethylene bottles, to which 180 ml of distilled water and 10 glass beads (6 mm diameter) were added. The bottles were then placed in a rotary shaker (Reax 2, Heidolph) with a frequency of 50 rpm for 16 h, after which the samples were wet sieved using stainless steel sieves with 212- and 53- $\mu\text{m}$  apertures. The following aggregate-size classes of organic matter were obtained: macroaggregates,  $>212\ \mu\text{m}$  (separated by flotation of roots and residues); mesoaggregates, 212-53  $\mu\text{m}$ ; and microaggregates,  $<53\ \mu\text{m}$  (organic matter associated to silt and clay). All the fractions were dried at  $50^\circ\text{C}$  for 48 h and weighed afterwards, and the C and N contents were then measured by dry combustion (Wright and Bailey, 2001).

### 1.2.7. Statistical analyses

The General Lineal Model procedure of SAS v.8.0 (SAS Institute Inc., 2002) for completely randomized designs was used to test the effect of treatment on soil microbial respiration and microbial biomass C. Statistical significance ( $P < 0.05$ ) between treatments was tested using Student's t test and significant differences within treatments at different depths were tested using Tukey's HSD multiple comparison test. Correlation analyses (Pearson's coefficient) were used to evaluate possible relationships among microbial biomass C, soil microbial respiration and soil organic carbon. In the case of physical fractionation of soil organic matter, Student's t test was used to verify statistical significance between treatments at different depths and in different soil fractions. Finally, Tukey's HSD multiple comparison test was used to check for significant interactions between depths and fractions in every treatment.

## 1.3. Results

The soil morphology was relatively uniform among the land-use sites, as expected, considering the soil parent material is aerielly deposited volcanic ash, the proximity and

uniformity of the topography among the sites, and the original shared vegetation of native forest. The soil texture (0-100 cm) was loamy fine sand in all sites that varied slightly to fine sandy loam in the subsoil (>20 cm depth) of PPP. The soil structure was moderate fine granular in the surface (0-30 cm), which graded to weak fine granular in the subsoil, with no structure (massive) in the lowest horizon, and the bulk density was 0.9 g cm<sup>-3</sup> or lower in all horizons. However, the chemical properties showed differences among treatments and are summarized in Table 1. In general, in the upper 20 cm of soil the contents of organic C and N were highest in DNP, but available N (NO<sub>3</sub>-N), P and K were highest in NPF. Fixation of P was also lowest in the upper 10 cm of soil of NPF.

Table 1.1. Chemical properties (organic C and N, effective cation exchange capacity, and contents of aluminum, nitrate, phosphorus and potassium) of the volcanic soil (medial, amorphic, mesic Typic Hapludands) (0-40 cm depth) under three temperate land-uses. Measurements were taken in the Chilean Patagonia in March 2006.

| Treatment | So Depth (cm) | C (%) | N (%) | C/N    | pH   | Urea-C (mg kg <sup>-1</sup> ) | Al (mg kg <sup>-1</sup> ) | NO <sub>3</sub> -N (mg kg <sup>-1</sup> ) | Available P (mg kg <sup>-1</sup> ) | Available K (mg kg <sup>-1</sup> ) | Fixation (%) |
|-----------|---------------|-------|-------|--------|------|-------------------------------|---------------------------|---|------------------------------------|------------------------------------|--------------|
| DNP       | 0-10          | 5.85a | 0.81a | 12.10a | 5.5a | 8.5a                          | 0.2a                      | 29a                                       | 8.2a                               | 200a                               | 1.9a         |
|           | 0-20          | 5.70b | 0.82b | 11.70b | 5.5a | 1.70b                         | 0.2a                      | 27a                                       | 7.9b                               | 171b                               | 6.45b        |
|           | 10-20         | 5.75c | 0.85c | 12.05a | 5.2a | 1.75c                         | 0.2a                      | 17b                                       | 11.2a                              | 97a                                | 1.9a         |
|           | 20-40         | 1.84d | 0.11d | 15.73d | 6.5b | 113d                          | 0.11b                     | 95c                                       | 97c                                | 850c                               | 840c         |
| PPP       | 0-10          | 4.85a | 0.51a | 12.10a | 5.5a | 3a                            | 0.1a                      | 15a                                       | 11.8a                              | 110a                               | 26.1a        |
|           | 0-20          | 4.00b | 0.42b | 12.70a | 5.2a | 85a                           | 0.13b                     | 17a                                       | 11.8a                              | 77ab                               | 12.0b        |
|           | 10-20         | 2.84c | 0.35c | 12.05b | 6.1b | 113b                          | 0.14b                     | 103b                                      | 11.3b                              | 59ab                               | 11.4b        |
|           | 20-40         | 2.8c  | 0.25c | 14.00c | 6.5c | 111b                          | 0.14b                     | 100b                                      | 10.1c                              | 59ab                               | 12.2b        |
| NPF       | 0-10          | 8.25a | 1.05a | 12.00a | 5.5a | 4a                            | 0.2a                      | 33a                                       | 62.2a                              | 400a                               | 11.9a        |
|           | 0-20          | 4.07b | 0.29b | 17.00b | 6.0b | 170b                          | 0.2a                      | 123b                                      | 6.1b                               | 205b                               | 21.9a        |
|           | 10-20         | 3.1c  | 0.25c | 12.20c | 6.2c | 82a                           | 0.2a                      | 17c                                       | 12.8a                              | 187b                               | 15.02b       |
|           | 20-40         | 1.61d | 0.11d | 14.20c | 6.0b | 85a                           | 0.11b                     | 20c                                       | 11c                                | 178b                               | 87.04c       |

DNP: more than 50-year-old degraded natural pasture, PPP: 16-year-old *Pinus ponderosa* exotic plantation, NPF: 150-year-old *Nothofagus pumilio* second-growth forest. Values with the same lower case letter within the different depths in the same column and one land-use are not significantly different (Tukey's HSD test, \**P* < 0.05).

The Al saturation was slightly higher in PPP (< 0.4%). The chemical characteristics at 40-100 cm depth were similar to 20-40 cm and are therefore not presented here. In general, with

increasing soil depth there were decreasing contents of C, N, available P and K, and increasing soil pH and C/N ratios. In all treatments, both organic C and N contents were significantly higher at 0-5 cm than 5-10 cm depths, at 5-10 cm than 10-20 cm depths, and at 10-20 cm than 20-40 cm depths, except in PPP where the C and N contents did not decrease between 10 and 40 cm depths. The C/N ratios also generally increased with soil depths. In DNP, the C/N ratio and ECEC were lower, and available P and K and P fixation were significantly higher in the upper 5 cm of soil, while Al was significantly lower at 20-40 cm and NO<sub>3</sub> at 10-40 cm depths. In PPP, the C/N ratio, pH, ECEC, NO<sub>3</sub> and available P had similar values among the 0-5 and 5-10 cm depths but were significantly different below the 10 cm depth, and Al was lower and available K higher at 0-5 cm depth only. In NPF, the C/N ratio, pH, NO<sub>3</sub> and available P were significantly different between the 0-5 and 5-10 cm depths but generally similar between the other depths. Al and available K were higher at 20-40 cm and 0-5 cm depths, respectively, and fixation of P significantly higher starting 10 cm depth. Also, it is interesting to note that ECEC at 5-10 cm depth largely exceeds ECEC at other depths in DNP and NPF, that available P was higher in the upper 5 cm in these two treatments, and that available K was much higher at 0-5 cm depth in every land use.

### *1.3.1. Carbon content in soil and vegetation*

Determination of soil C at 0-40 cm depth in DNP, PPP and NPF shows that DNP had the highest C contents, but was significantly higher than NPF only (Tables 2 and 3).

Aboveground and root biomass in NPF was significantly higher than PPP, but the latter contained much more leaf biomass than NPF (Table 2). Total root biomass per hectare in DNP was more than 1.5 times higher than in PPP. Total C content in DNP aboveground biomass was also significantly lower than in the other systems. However, C content in DNP root biomass was 63.5% higher than in PPP but less than half in NPF. Most C in NPF biomass was found in stems, followed by roots, branches and leaves. NPF contained almost seventeen times more C in stems than PPP, five times more C in branches, and four times more C in roots. Ponderosa pine needles contained five times more C than NPF leaves. There was more C in

NPF biomass than the soil underneath, but the opposite was observed in the other ecosystems. Total C content was similar in DNP and PPP but both were less than half of that in NPF.

Table 1.2. Biomass and carbon content ( $\text{Mg ha}^{-1}$ ) (mean  $\pm$  standard deviation) in soil at 0-40 cm depth and different plant components (trees and pasture) from three temperate land-uses. Measurements were taken in the Chilean Patagonia in March 2006.

| Treatment | Total Plant Biomass<br>( $\text{Mg DM ha}^{-1}$ ) |                  |                   |                   |                  |             |
|-----------|---|------------------|-------------------|-------------------|------------------|-------------|
|           | Stems   | Branches         | Leaves            | Total Aboveground | Total Roots      | Total Plant |
| DNP       | NA  | NA               | NA                | 3.8 $\pm$ 0.5 a   | 41.2 $\pm$ 8.0 a | 44.9 a      |
| PPP       | 23.7 $\pm$ 0.7 a                                  | 12.4 $\pm$ 0.1 a | 4.4 $\pm$ 0.1 a   | 50.5 $\pm$ 0.5 b  | 25.2 $\pm$ 0.4 b | 75.8 b      |
| NPF       | 396.2 $\pm$ 5.4 b                                 | 56.5 $\pm$ 0.5 b | 1.74 $\pm$ 0.02 b | 453.5 $\pm$ 5.9 c | 97.4 $\pm$ 0.9 c | 550.9 c     |

| Treatment | Carbon Content<br>( $\text{Mg ha}^{-1}$ ) |                 |                 |                  |                     |         |
|-----------|---|-----------------|-----------------|------------------|---------------------|---------|
|           | Stems                                     | Branches        | Leaves          | Roots            | Soil                | Total   |
| DNP       | NA  | NA              | 9 $\pm$ 0.3 a   | 29.6 $\pm$ 4.0 a | 135.8 $\pm$ 10.8 a  | 165.3 a |
| PPP       | 11.9 $\pm$ 0.2 a                          | 6.2 $\pm$ 0.1 a | 1.2 $\pm$ 0.1 b | 12.6 $\pm$ 0.2 b | 116.9 $\pm$ 10.1 ab | 136.8 a |
| NPF       | 1516 $\pm$ 2.5 b                          | 383 $\pm$ 0.3 b | 37 $\pm$ 0.00 c | 48.7 $\pm$ 0.5 c | 1235 $\pm$ 12.3 b   | 2800 b  |

DNP: more than 50-year-old degraded natural pasture, PPP: 16-year-old *Pinus ponderosa* exotic plantation, NPF: 150-year-old *Nothofagus pumilio* second-growth forest. Values with the same lower case letter within a column (Total Plant Biomass or Carbon Content) are not significantly different (Student's t test,  $*P < 0.05$ ).

### 1.3.2. Distribution of soil organic carbon

Total SOC at 0-40 cm depth of each ecosystem decreased in the order DNP > PPP > NPF (Table 3). There was a 33% increase in SOC that resulted from the conversion of NPF to DNP and a 14% decrease from DNP to PPP. SOC was significantly lower in NPF than DNP at 0-40 cm depth while no significant difference was found between other land-use practices. On the other hand, total SOC at 0-10 cm depth decreased in the order DNP > NPF > PPP. Averaged over the two soil layers, land-use practices affected significantly SOC stocks. There was a 27% increase from NPF to DNP, but a 44% decrease resulting from the conversion of DNP to PPP. It is interesting to note that SOC content in DNP was almost double the SOC content in PPP. SOC was also significantly higher in DNP and lower in PPP.

Table 1.3. Average soil organic carbon (SOC) and nitrogen (SON) concentrations (%) and C contents (g C m<sup>-2</sup>) at 0-10 cm and 0-40 cm depths of different temperate land-use practices. Measurements were taken in Chilean Patagonia in March 2006 (mean ± standard deviation).

| Treatment | SOC (0-10 cm)            |                        | SOC (0-40 cm)             |                           | SON (0-10 cm)            |
|-----------|--------------------------|------------------------|---------------------------|---------------------------|--------------------------|
|           | total                    | (g m <sup>-2</sup> )   | (%)                       | (g m <sup>-2</sup> )      | (%)                      |
| DNP       | 7.82 ± 0.16 <sub>a</sub> | 793 ± 117 <sub>a</sub> | 3.77 ± 0.50 <sub>a</sub>  | 1357 ± 1077 <sub>a</sub>  | 0.28 ± 0.03 <sub>a</sub> |
| PPP       | 4.4 ± 0.57 <sub>b</sub>  | 388 ± 57 <sub>b</sub>  | 3.25 ± 0.28 <sub>ab</sub> | 1188 ± 1011 <sub>ab</sub> | 0.2 ± 0.06 <sub>ab</sub> |
| NPF       | 6.6 ± 0.61 <sub>c</sub>  | 554 ± 57 <sub>c</sub>  | 2.81 ± 0.55 <sub>b</sub>  | 10217 ± 1217 <sub>b</sub> | 0.20 ± 0.01 <sub>b</sub> |

DNP: degraded pasture, PPP: *Pinus ponderosa* plantation, NPF: *Nothofagus pumilio* second-growth forest. Values with the same lower case letter within a column are not significantly different (Student's t test, \*\**P* < 0.01).

Total SOC at 0-10 cm depth was slightly more than half the total SOC at 0-40 cm depth in both DNP and NPF. However, SOC at 0-10 cm depth in PPP was nearly three times lower than the total SOC at 0-40 cm depth. Both SOC and soil organic nitrogen (SON) concentrations were significantly higher in DNP than NPF at 0-40 cm depth. In all ecosystems, SOC concentrations (%) were highest in the upper 10 cm. Finally, beyond 40 cm depth, C contents among distinct ecosystems were low and similar (data not shown).

### 1.3.3. Soil microbial respiration

Soil microbial respiration did not vary significantly among treatments at 0-40 cm depth, nor at 0-5 and 5-10 cm depths (Table 4). However, PPP had the highest soil microbial respiration at 10-20 and 20-40 cm depths. At 10-20 cm depth, it was more than double than in DNP and NPF. At 20-40 cm depth, it was 45% higher than in DNP but only 17% higher than in NPF. In all treatments, soil microbial respiration decreased with an increase of pH until a value of 6.2 (Tables 1 and 4). Above this value, soil microbial respiration remained fairly stable in DNP, but increased slightly and then decreased in PPP, until reaching a similar value to DNP at 20-40 cm depth. Soil microbial respiration in NPF followed the same trend, but decreased with a pH reduction from 6.2 to 6.0 beyond 20 cm depth. The increase of pH at further depths in DNP, PPP and NPF (Table 1) was associated with lower and stable values of soil microbial respiration (data not shown).



Table 1.4. Mean soil microbial respiration ( $\mu\text{g C-CO}_2 \text{ g}^{-1}$ ) and microbial biomass C ( $\mu\text{g C g}^{-1}$ ) at different depths (0-40, 0-5, 5-10, 10-20, 20-40 cm) under three temperate land-use practices. Measurements were taken in Chilean Patagonia in March 2006 (mean  $\pm$  standard deviation).

| Treatment  | Depth           |                   |                   |                    |                  |
|--|-----------------|-------------------|-------------------|--------------------|------------------|
|  | 0-40 cm         | 0-5 cm            | 5-10 cm           | 10-20 cm           | 20-40 cm         |
| Soil microbial respiration ( $\mu\text{g C-CO}_2 \text{ g}^{-1} \text{ day}^{-1} \text{ soil}$ ) |                 |                   |                   |                    |                  |
| DNP  | 123.6 ± 12.8 a  | 556.5 ± 15.1 aA   | 123.0 ± 18.7 ab   | 59.1 ± 8.2 aBc     | 57.7 ± 5.7 ac'   |
| PPP  | 135.2 ± 23.1 a  | 508.7 ± 10.9 aA   | 115.1 ± 16.7 ab   | 137.4 ± 10.6 bB    | 69.4 ± 5.1 bB    |
| NPF  | 137.6 ± 15.9 a  | 582.7 ± 10.6 aA   | 103.1 ± 27.8 ab   | 59.1 ± 10.9 cC     | 59.1 ± 9.3 abC'  |
| Microbial biomass C ( $\mu\text{g C g}^{-1} \text{ dry soil}$ )                                  |                 |                   |                   |                    |                  |
| DNP  | 478.1 ± 122.5 a | 3289.5 ± 396.1 aA | 1138.8 ± 227.9 aB | 419.7 ± 90.0 ac'   | 117.0 ± 15.9 ac' |
| PPP  | 222.6 ± 23.7 b  | 2001.3 ± 85.7 bA  | 856.7 ± 56.1 bB   | 284.0 ± 117.3 abB' | 82.8 ± 23.8 acC' |
| NPF  | 656.2 ± 23.7 c  | 3705.7 ± 213.1 cA | 1011.8 ± 329.6 aB | 129.0 ± 23.5 cC'   | 61.1 ± 23.9 acC' |

DNP: degraded natural pasture, PPP: *Pinus ponderosa* plantation, NPF: *Nothofagus pumilio* second-growth forest. Values with the same lower case letter within a column (soil microbial respiration or microbial biomass C) are not significantly different (Student's t test, \* $P < 0.05$ ). Values with the same higher case letter within the 0-5, 5-10, 10-20 and 20-40 cm depths and one land-use are not significantly different (Tukey's HSD test, \* $P < 0.05$ ).

In all treatments, soil microbial respiration was significantly higher in the upper 5 cm than at any other depths (Table 4). In DNP, soil microbial respiration at 5-10 cm was also higher than at 20-40 cm depth, but no significant difference was observed at 10-20 cm depth. The same trends were encountered in NPF, the only exception being at 5-10 cm where microbial respiration was higher than at 10-40 cm depth. In PPP, soil microbial respiration tended to be of a more similar magnitude between 5 and 20 cm depth.

#### 1.3.4. Microbial biomass C

Soil microbial biomass C was affected significantly by land-use treatments and decreased in the order DNP > NPF > PPP at 0-40 and 0-5 cm depths (Table 4). It was highest in DNP at all depths. At 0-40 cm depth, there was a 33% difference between DNP and NPF and a 36% difference between NPF and PPP. Additionally, microbial biomass in DNP was more than double than in PPP at depths of 0-40, 0-5 and 5-10 cm. At 10-20 cm, microbial biomass was significantly higher in DNP than NPF, but no difference was observed with PPP. No further differences were encountered beyond 20 cm depth.

In general, the greater microbial biomass C in DNP at 0-5 and 5-10 cm depths corresponded also to the higher SOC values found at these depths (Tables 1 and 3). The 0-5 and 5-10 cm depths are also those where the largest amount of live fine root biomass was encountered, especially in DNP from our field measurements (Table 2).

Soil microbial respiration values were positively correlated with microbial biomass C for all treatments together ( $R^2 = 0.86$ ,  $n = 36$ ) (Fig. 1A). Additionally, microbial biomass C was strongly correlated with SOC ( $R^2 = 0.87$ ,  $n = 12$ ) (Fig. 1B). In all treatments, microbial biomass C was significantly higher in the upper 5 cm then at any other depths (Table 2). In DNP, microbial C at 5-10 cm was also higher than at 10-40 cm depth, but no significant difference was observed between the 10-20 and 20-40 cm depths. The same trends were encountered in PPP and NPF. In PPP however, the microbial biomass C at 10-20 cm depth was not significantly different of the values at 5-10 and 20-40 cm depths.

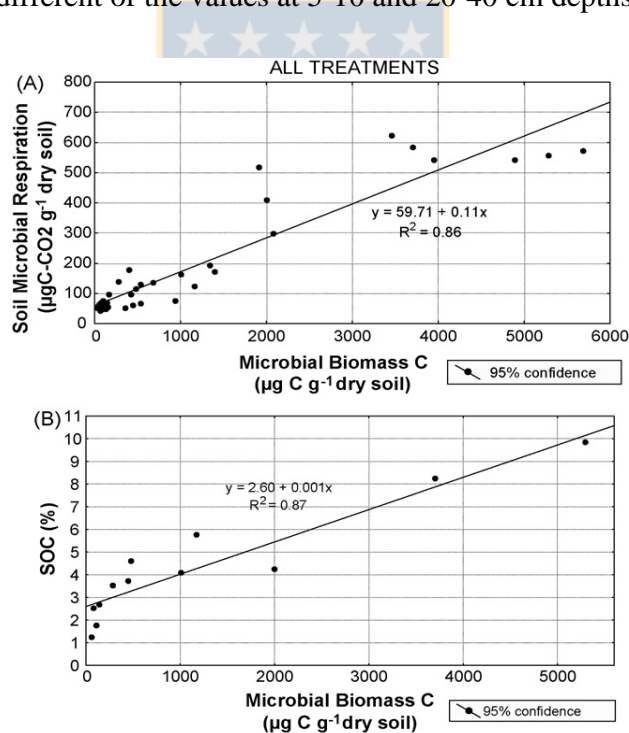


Figure 1.1. Correlation matrix showing the relationship between (A) microbial biomass C and soil microbial respiration ( $n=36$ ) and (B) microbial biomass C and soil organic carbon (SOC) ( $n=12$ ) over all treatments (degraded natural pasture, *Pinus ponderosa* plantation, *Nothofagus pumilio* second-growth forest. Measurements were taken in Chilean Patagonia in March 2006.

The soil microbial respiration to microbial biomass C ratio was negatively correlated with SOC for all treatments together ( $R^2 = 0.49$ ,  $n = 36$ ) (Fig. 2). As SOC decreased with depth, both microbial respiration and microbial biomass C also decreased because less substrate is available for the microorganisms to feed on.

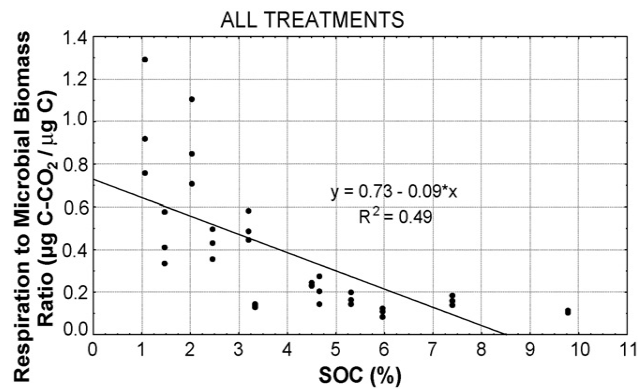


Figure 1.2. Correlation matrix showing the relationship between the ratio of microbial respiration to microbial biomass and soil organic carbon (SOC) content ( $n=36$ ) over all treatments (degraded natural pasture, *Pinus ponderosa* exotic plantation, *Nothofagus pumilio* second-growth forest. Measurements were taken in Chilean Patagonia in March 2006.

### 1.3.5. Aggregate-size distribution

Even with the large difference in SOC content (0-10 cm depth) between DNP and PPP (Table 3), there was no significant difference of aggregate-size distribution between these two ecosystems (Fig. 3). However there was a significant difference of macroaggregates ( $>212 \mu\text{m}$ ) between NPF and the two other systems. The percentage of macroaggregates (containing the light fraction of organic matter) increased from 8% in DNP to 10 and 15% in PPP and NPF, respectively. On the other hand, a small but significant difference in 212-53  $\mu\text{m}$ -sized mesoaggregates (containing the intermediate fraction of organic matter) was also observed between NPF and the two other systems. The percentage of mesoaggregates decreased from 38% in DNP and PPP to 35% in NPF. No significant differences in percentage of  $<53 \mu\text{m}$ -sized microaggregates (containing the heavy fraction of organic matter) were observed between ecosystems, although the values tended to decrease from 50 to 47 to 45% in DNP to

PPP and to NPF, respectively. In all ecosystems, the soil fractions were dominated by microaggregates (<53  $\mu\text{m}$ ), followed by mesoaggregates (212-53  $\mu\text{m}$ ) and ultimately by macroaggregates (>212  $\mu\text{m}$ ).

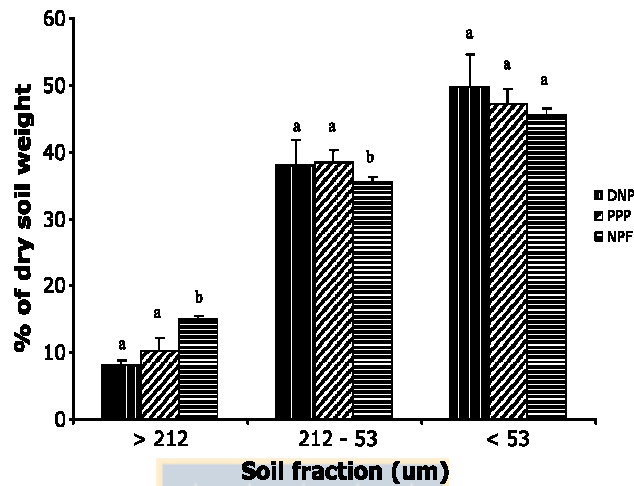


Figure 1.3. Proportion of dry soil weight at 0-10 cm depth in different soil fractions (macroaggregates (>212  $\mu\text{m}$ ), mesoaggregates (212-53  $\mu\text{m}$ ) and microaggregates (<53  $\mu\text{m}$ )) in different temperate land-uses in the Chilean Patagonia, March 2006. DNP: degraded natural pasture, PPP: *Pinus ponderosa* exotic plantation, NPF: *Nothofagus pumilio* second-growth forest. Vertical bars denote standard error of the mean. Values with the same lower case letter within a soil fraction are not significantly different (Student's t test, \* $P < 0.05$ ).

The organic C concentrations (%) within the macroaggregates at 0-10 cm depth were not statistically different among the land-use practices but were significantly higher than within the other fractions (Table 5) probably because of the presence of small root fragments. In PPP only, SOC was slightly higher in the macroaggregate fraction in the upper 5 cm, where there is a constant input of needles and cones throughout the year, which contributes to the light fraction of SOM in this treatment. In all ecosystems, SOC in macroaggregates was also significantly higher at 0-5 cm depth than at 10-20 and 20-40 cm depths. Within every ecosystem however, the organic C concentrations did not differ statistically between the mesoaggregate and microaggregate fractions at 0-5, 5-10, 10-20 and 20-40 cm depths.

Table 1.5. Mean organic carbon concentrations (%) at 0-5, 5-10, 10-20 and 20-40 cm depths in different soil fractions (macroaggregates (>212  $\mu\text{m}$ ), mesoaggregates (212-53  $\mu\text{m}$ ) and microaggregates (<53  $\mu\text{m}$ )) under three temperate land-use practices. Measurements were taken in the Chilean Patagonia in March 2006 (average  $\pm$  standard deviation).

| Depth<br>(cm) | Treatment | Organic C (%)        |                        |                      |
|---------------|-----------|----------------------|------------------------|----------------------|
|               |           | Soil Fraction        |                        |                      |
|               |           | > 212 $\mu\text{m}$  | 212 - 53 $\mu\text{m}$ | < 53 $\mu\text{m}$   |
| 0-5           | DNP       | 32.64 $\pm$ 0.32 aA  | 4.71 $\pm$ 0.34 a1B    | 5.55 $\pm$ 2.55 a1B  |
|               | PPP       | 30.48 $\pm$ 0.20 aA  | 6.42 $\pm$ 0.83 b1A    | 8.21 $\pm$ 1.59 a1A  |
|               | NPF       | 34.68 $\pm$ 1.97 aA  | 9.97 $\pm$ 2.58 c1B    | 9.45 $\pm$ 3.78 a1B  |
| 5-10          | DNP       | 27.25 $\pm$ 3.90 aAB | 4.96 $\pm$ 1.82 a1B    | 6.89 $\pm$ 2.97 a1B  |
|               | PPP       | 29.10 $\pm$ 3.86 aB  | 5.03 $\pm$ 0.49 a1B    | 6.79 $\pm$ 1.89 a1C  |
|               | NPF       | 29.53 $\pm$ 1.69 aAB | 8.85 $\pm$ 1.85 b1B    | 8.75 $\pm$ 1.83 a1B  |
| 10-20         | DNP       | 25.97 $\pm$ 4.90 aB  | 4.53 $\pm$ 2.47 a1B    | 7.98 $\pm$ 1.63 aB1B |
|               | PPP       | 22.90 $\pm$ 3.57 aB  | 4.13 $\pm$ 1.18 a1B    | 5.11 $\pm$ 1.75 a1C  |
|               | NPF       | 29.82 $\pm$ 6.37 aB  | 5.74 $\pm$ 0.41 a1B    | 8.56 $\pm$ 0.85 b1B  |
| 20-40         | DNP       | 17.22 $\pm$ 1.90 aC  | 2.06 $\pm$ 0.92 a1C    | 4.90 $\pm$ 1.3 a1B   |
|               | PPP       | 18.79 $\pm$ 4.42 aC  | 2.46 $\pm$ 0.85 a1B    | 4.19 $\pm$ 0.75 a1C  |
|               | NPF       | 18.34 $\pm$ 5.97 aC  | 7.04 $\pm$ 0.42 a1B    | 4.73 $\pm$ 0.35 a1B  |

DNP: degraded natural pasture, PPP: *Pinus ponderosa* exotic plantation, NPF: *Nothofagus pumilio* second-growth forest. Standard deviations are indicated. Values with the same lower case letter within a column and within a same depth are not significantly different (Student's t test,  $*P < 0.05$ ). Values with the same higher case letter within the four depths and one land-use are not significantly different (Tukey's HSD test,  $*P < 0.05$ ).

In general, the C contents in the mesoaggregate and microaggregate fractions were similar in both PPP and NPF, but statistically different in DNP at 0-40 cm depth. Within the mesoaggregates, the C concentration (0-10 cm depth) was significantly higher in NPF compared with DNP and PPP at 0-10 cm depth (Table 5), showing that the greatest difference between ecosystems was found in that fraction. The mesoaggregate C concentration in NPF was 64% higher than in PPP and 94% higher than in DNP. Organic C content at 0-5 cm depth in mesoaggregates decreased in the order NPF > PPP > DNP and were significantly different. There was a 36% decrease going from NPF to PPP and a 27% decrease from PPP to DNP. SOC in NPF was more than double than in DNP. At 5-10 cm depth, the organic C content follows the same trend as before, with the exception that it is significantly higher in NPF only.

No differences are observed between C contents at further depths. In the microaggregate fractions, SOC contents were similar within ecosystems at different depths, except at 10-20 cm where it was statistically higher in NPF than in PPP.

## **1.4. Discussion**

### *1.4.1. Soil microbial respiration*

In the long term, trees in second-growth deciduous forests, just as trees in old agroforestry systems (Lee and Jose, 2003a) cause continual addition of litter and root turnover, thus maintaining or increasing soil organic matter (Kimmins, 2004), thereby influencing soil microbial respiration. In an incubation study by Mallik and Hu (1997), the authors reported that soil organic matter is strongly related to soil microbial respiration and is one of the important factors controlling it. The significantly higher soil microbial respiration in PPP beyond 10 cm depth may be due to the presence of mycorrhizae in the root biomass that passed through the 2-mm mesh sieve, a fungi that also respire, and which symbiotic associations under pine trees is well known worldwide (Kimmins, 2004; Marx, 1991; Read, 1991). Most mycorrhizae may be encountered at 10-20 cm depth, thus explaining why the largest difference in soil microbial respiration is found at that depth.

On the other hand, our results show that C-CO<sub>2</sub> production had a tendency to be lower under PPP than DNP at 0-10 cm depth, but invariably not at 10-40 cm, in agreement with data obtained by Ross et al. (2002) and Saggart et al. (2001) working with *Pinus radiata* in New Zealand. In the upper 10 cm under DNP, there could be enrichment of the soil with labile C from fine roots and aboveground litter, which may explain the difference observed (Ross et al., 2002).

With respect to pH, continuous addition of above and belowground litter for the past 150 years under NPF has altered soil characteristics, making it more acidic than under the 16-year-old ponderosa pine plantation. Surprisingly, despite the addition of aboveground litter from PPP throughout the rotation, the pH values at 0-40 cm depth make the soil under the plantation

slightly less acidic than under DNP, an indication of the low buffering capacity of pasture litter. Our results are the opposite of those obtained by Lee and Jose (2003b) for *Pinus taeda* who reported a positive correlation, but in agreement for *Populus deltooides*, where the soil respiration was negatively correlated with soil pH. However, one must consider that these authors measured total soil respiration, which includes microbial and root respiration. Also in all soils, pH increased with the increase of depth, showing the influence of precipitation and vegetation in superficial soil. Soil pH in itself may not be affecting the microbial respiration, but rather lack of substrate for microbial biomass with increasing depth.

The highest microbial respiration observed at 0-5 cm depth could result from the larger amount of soil organic matter and associated SOC (Table 1), larger microbial biomass C encountered in superficial soil layers (Table 4). Kelting et al. (1998) reported that respiration by root systems and their microbial biomass components represent a significant portion of soil respiration in most ecosystems.

On the other hand, similar values of microbial biomass C in PPP at 5-10, 10-20 and 20-40 cm depths as well as similar SOC contents could explain why soil microbial respiration did not vary significantly at these depths. In NPF higher microbial respiration at 5-10 cm depth could be attributed to the large associated amount of microbial biomass at the same depth, which is significantly higher than at lower depths (Table 4). Finally, the different amounts of microbial biomass C found at 5-10 and 20-40 cm depths in DNP may also explain the trends of soil microbial respiration that were observed at the same depths.

#### *1.4.2. Microbial biomass C*

Generally, the C/N ratio (Table 1) increased with an increase of soil depth in all ecosystems, which denotes that superficial organic matter was less resistant to microbial decomposition than organic matter in the subsoil. On the other hand, our data show that all ecosystems at 0-40 cm depth were poor to very poor in available N as  $\text{NO}_3$ , with an intermediate level only at 0-5 cm depth in NPF. In any case however, this is a condition for temporal competition between plants and microorganisms for the nutrient, with prevalence that some net

immobilization in favor of the microbes may occur. There is an efficient use of nitrogen if N being mineralized is absorbed by plants as  $\text{NH}_4$  (nutrient uptake). Other nutrients as P were medium to high in all land-uses and should not be a limiting factor for the metabolism and development of microbial biomass. Neither should the low levels of available K found at all depths in PPP and at 10-40 cm depth in DNP be a limiting factor given the low amounts needed by the microbes. Considering these three nutrients, PPP is the land-use that shows the lowest available amounts available followed by DNP, NPF having the highest levels. This condition observed in PPP could perhaps constitute another reason for the higher soil microbial activity encountered at 10-20 and 20-40 cm depths, since stressed microbes generally respond by increasing  $\text{CO}_2$  evolution per unit of biomass (Chander and Brookes, 1991a, b).

Our results for microbial biomass C at 0-10 cm depth in all ecosystems (3229.1, 1242.5 and 2358.4  $\mu\text{g C g}^{-1}$  dry soil for DNP, PPP and NPF, respectively) are higher than those by Ross et al. (2002) who reported concentrations of microbial biomass C at 0-10 cm depth of 750  $\text{mg C kg}^{-1}$  in non-volcanic soil (Oxyaquic Drystrudept) under *Pinus radiata* and 1500  $\text{mg C kg}^{-1}$  in non-volcanic soil (Acric Fragiaqualf) under pasture in New Zealand. The differences observed could be explained by the unique properties of volcanic soils (Buol et al., 1997) in our study, which are generally characterized by low bulk density, the presence of allophanes, high phosphate retention and higher C contents in surface soils. Also in New Zealand in a volcanic soil (Typic Udivitrands), Ross et al. (1999) reported lower values of 1600, 858 and 1600  $\text{mg C kg}^{-1}$  in a pasture, pine plantation and indigenous forest, respectively, at 0-10 cm depth, but higher values of 482, 473 and 819  $\text{mg C kg}^{-1}$  at 10-20 cm depth.

The significantly lower microbial biomass C at 0-10 cm depth in PPP, as compared with DNP and NPF, may be attributed to the distinct quality and distribution of available substrates in the three ecosystems (Zagal and Cordova, 2005; Ross et al., 1999; Scott and Binkley, 1997). According to Ross et al. (2002) who obtained similar results for *Pinus radiata* and pasture, readily metabolizable C may be lower in the pine than pasture and deciduous tree mineral soils, resulting in the lower values of microbial biomass C. Other studies have shown reductions of microbial C and N and changes in soil organic matter quality following



afforestation of pastures with first-rotation *Pinus radiata* plantations (Beets et al., 2002; Ross et al., 1999; Giddens et al., 1997).

Lee and Jose (2003a) reported a positive correlation between soil respiration rates and microbial biomass C for old and young pecan orchards and pecan/cotton alley-cropping systems ( $R^2 = 0.52$ ,  $n = 21$ ,  $P < 0.0001$ ), although the adjustment was not as high as those obtained in our results. This could be attributed to the fact that the authors' measurement of soil respiration was total as it included root respiration, thus showing a higher variability. On the other hand, the strong correlation observed between microbial biomass C and SOC is in close agreement with another correlation of microbial biomass C and soil organic matter obtained by Lee and Jose (2003b), in which  $R^2 = 0.84$  and  $n = 21$ , and by Witter et al. (1993), in which  $R^2 = 0.82$ .

Based on our correlation results, it is clear that soil microbial respiration is highly related to microbial biomass C and soil organic carbon, but it is not possible at this point to determine the extent of each contributing factor on total soil respiration.

For every treatment, microbial biomass C decreased with an increase of pH and depth (Tables 1 and 4), a similar behavior to soil microbial respiration. Again, this is probably more related to soil depth, where there is less SOM and the associated microbial biomass. Our results are the opposite of those obtained by Lee and Jose (2003b) for *Pinus taeda* who reported a positive correlation, but in agreement with their results for *Populus deltoides*, where the microbial biomass C was strongly and negatively correlated with soil pH.

With respect to the negative correlation observed between the soil microbial respiration to microbial biomass C ratio and SOC, microbial respiration decreased at a slower rate than microbial biomass, which explains why the ratio increases. On an equivalent mass basis, the ratio could be a good indicator of microbial activity. From our data, it appears that the microbes are more active in soil from PPP, followed by NPF and lastly by DNP.

#### 1.4.3. Distribution of soil organic carbon

The SOC contents obtained are somewhat of similar magnitude as those reported by Huygens et al. (2005) for an Andisol at 0-30 cm depth in southern Chile, who observed a 31% increase and 42% decrease of SOC for the conversion from second growth *Nothofagus obliqua* forest to grass and grass to *Pinus radiata*, respectively. However, the 14% decrease of SOC at 0-40 cm depth from our investigation is much lower than the 42% decrease reported in that study, but the 44% decrease at 0-10 cm depth and 31% decrease at 0-20 cm depth in this investigation are more similar to their results.

The highest SOC contents in DNP at 0-10 cm depth could be explained by the continuous ground cover and the large root turnover and production at the surface, where aeration in the thick root mat is relatively low and reduces the decomposition rate of organic matter. According to Yakimeto (1998), more intense humus formation would occur in such environments. Jobbagy and Jackson (2000) also reported that the annual turnover of SOC from dying tree roots is less than from grass roots.

On the other hand, differences of SOC observed between tree species could be due to the distinct quality of plant material from different forest types (Lal et al., 1995), and to the high C input to the soil from NPF, where older deciduous trees provide yearly litterfall and have been in place for a much longer period of time, and where the presence of understorey provides an extra input of organic residues to soil. At 0-10 cm depth, the lower C content in PPP could also be explained by soil disturbances during the establishment causing soil erosion or loss in physical protection of SOC (Guo and Gifford, 2002). Moreover at the moment of the study, the plantation was only 16 years old, out of which the first years after its establishment were characterized by very little biomass and low litterfall rate, thus resulting in low input of SOC (Wilde, 1964). In forest environments, SOC inputs come in part from aboveground litterfall that accumulates on the surface soil. Therefore, fallen leaves decompose partially on the surface before being incorporated into the soil, resulting in a smaller accumulation of SOC and a thinner A horizon than grasslands (Paul et al. 2002).

The total soil organic carbon contents at 0-10 cm depth (7038, 3969 and 5544 g C m<sup>-2</sup> for DNP, PPP and NPF, respectively) were similar to those obtained by Huygens et al. (2005), who reported 6700, 4300 and 5800 g C m<sup>-2</sup> at the same depth in an Andisol under pasture, *Pinus radiata* plantation and *Nothofagus obliqua* second-growth forest. In their study, the sites were originally covered with the *Nothofagus* forest, which was partly converted to grassland 50 to 100 years ago and where pine plantation was established 16 years ago. Our data suggest a large potential for additional SOC sequestration at 0-10 cm depth under PPP and a smaller potential under DNP.

On the other hand, Stolpe et al. (2008) reported substantially lower C contents at 0-40 cm depth in non-volcanic soils (Ultic Palexeralfs) under *Acacia cavens* forests, known as Espinals in central Chile. They encountered 2697, 2516 and 1746 g C m<sup>-2</sup> in well-preserved (80 to 51% cover), typical (50 to 26% cover) and degraded (25 to 10% cover) Espinals.

In NPF, macroaggregation (>212- $\mu$ m) was significantly higher and mesoaggregation (212-53  $\mu$ m) lower than in the other two systems (Figure 3). This is in agreement with the results reported by Six et al. (2002) for forest and afforested systems in non-volcanic soils. The larger amount of root fragments developed over the last 150 years under the second-growth system could explain why more macroaggregates are present than in the other systems. However, our results for microaggregates (<53  $\mu$ m) are the opposite of those by Six et al. (2002), who reported the lowest amount from that fraction (<53  $\mu$ m) in forest systems.

The higher C content in the mesoaggregates at 0-10 cm depth indicates that NPF has the best soil quality, which could be attributed to the very low anthropogenic intervention over the last 100 years. The better quality of organic matter found in NPF is also reflected by the presence of more NO<sub>3</sub>-N and available P and K. SOM quality is essential for productivity and sustainability of the ecosystem as it helps retain more available nutrients and water, which enhances plant growth and results in larger amounts of C being captured in the vegetation.

Our results did not indicate any significant differences in C concentrations within the microaggregates among the ecosystems, although SOC tended to be higher in NPF. Our

results show the importance of stable mesoaggregate fractions as the primary site of long-term carbon sequestration in soils. The largest amount of SOC was stabilized within the mesoaggregate fractions associated with mineral particles of lower size of NPF, which was the less disturbed ecosystem. In such environments, there is a macroaggregate turnover that permits the formation of stable mesoaggregates within macroaggregates, which protect SOC during extended periods of time (Six et al. 1998 and 2000). Other studies (Six et al., 2002; Balesdent et al., 2000; Christensen, 2001) have illustrated that enrichment of stable mesoaggregates can act as a carbon sink in the long term.

#### *1.4.4. Carbon content in soil and vegetation*

The deep thick root mat encountered in DNP easily explains why there is still more C contained therein than PPP, but this may not be always the case as the plantation gets older and develops bigger taproots and coarse roots (Kimmins, 2004), as already observed in the 150-year-old NPF. On the other hand, the large difference in C contents in leaves between PPP and NPF would tend to decrease as the pines age and reduce the proportion of their crowns and therefore, the amount of C allocated to that part of the tree (Litton et al. 2007).

Generally, native forest and forest plantation soils have less C in their superficial horizons than prairies (Buol et al., 1997). Grass and leguminous forages have a shorter life cycle as compared with trees, contributing to larger C cycling coming from leaves and roots in superficial soil, which are therefore easily decomposed. On the other hand, trees provide organic matter from leaves that fall on the ground but do not get incorporated to the soil in the short term. The DNP soil has more total C than the other systems, although it is more degraded, as evidenced by the lower C concentrations in the macroaggregate and mesoaggregate fractions (Table 5). Non-degraded pasture soil would have even larger total C, especially in these soil fractions. On the other hand, the decreasing C and N concentrations from 0 to 40 cm depth in every treatment were expected as SOM contents decreased with increasing depths, with the highest values in the upper 5 cm (Table 1). Also, the greater C/N ratios with depth indicated that the stability of SOM increased with soil depth in every treatment.

## 1.5. Conclusions

A basic assumption in this comparative study is that the soils were all similar in their biological, chemical and physical properties initially in each ecosystem. Although we cannot test this assumption, it is likely that the soils were similar to at least 40 cm depth of mineral soil, since they all originated from the same volcanic source.

In summary, soil microbial respiration was slightly greater in NPF in almost all depths. In all ecosystems, it was always greater at 0-5 cm depth than at any other depths. Soil microbial respiration beyond 10 cm depth in PPP was significantly higher than in the other systems, which could be attributed to the presence of mycorrhizae that have their own additional respiration. Microbial biomass C was significantly greater in DNP and lower in PPP. The different quality of available substrates and distribution of C inputs in the three ecosystems may explain why. Soil microbial respiration was also correlated positively with microbial biomass C and SOC, but we cannot at this point determine the contribution of each factor on soil respiration. The different land uses affect the formation of organic matter, SOC and microbial biomass C, which in turn will affect soil microbial respiration.

Conversion of DNP to PPP resulted in a 44% decrease of SOC stocks in 0-10 cm mineral soil, emphasizing the decline in soil organic matter in the short term, and also associated with disturbance of the site when PPP was established. But changes were less pronounced and non significant at 0-40 cm depth. However, the data suggest a larger potential for additional SOC sequestration under PPP at 0-10 cm depth, due to the unique properties of volcanic soils, which are characterized by the presence of allophanic clays and higher C contents in surface soils. Long-term SOC sequestration can also be enhanced by physical protection of SOC through aggregation. The largest amount of SOC was stabilized within the mesoaggregate fraction of the less disturbed system, NPF, followed by PPP. In the long term, formation of stable mesoaggregates in soils protected from erosion can behave as carbon sinks. Soil disturbance through wind and water erosion, just like tillage, accelerates the destruction of soil organic matter and results in increased loss of SOC.

The organic matter content was higher in DNP but of better quality in NPF, which is reflected by the presence of more  $\text{NO}_3\text{-N}$  and available P and K, a higher proportion of  $>212 \mu\text{m}$  organic matter, and intermediate contents of microbial biomass C. Soil organic matter quality is essential for productivity and long-term sustainability of the ecosystem as it helps retain more available water and nutrients that can be used by plants, therefore enhancing their growth, which ultimately results in more C being sequestered in the vegetation.

There was more C in NPF biomass than the soil underneath, but the opposite was observed in the other ecosystems. Total C content was slightly higher in DNP than PPP. However, total C content in NPF was more than twice the content in PPP and DNP.

The introduction of silvopastoral practices in pine plantations by thinning trees could optimize soil organic matter and SOC formation, microbial biomass C and soil microbial respiration because of the interaction of trees and pasture in the same unit of land, and sequester larger amounts of C in both the aboveground and belowground biomass. The establishment of leguminous pasture, such as clover, can increase soil fertility through N fixation, thus benefiting tree growth. Also, trees growing in pasture may protect the soil against erosion, constitute efficient windbreaks and provide extra incomes to the landowners throughout the forest rotation, including C credits, as outlined in the Kyoto Protocol.

## **1.6. Acknowledgements**

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## 1.7. References

- Balesdent J., Chenu C., Balabane M. 2000. Relationship of soil organic matter dynamics to physical protection and tillage. *Soil Tillage Research* 53: 215-230.
- Beets P.N., Oliver G.R., Clinton P.W. 2002. Soil carbon protection in podocarp/ hardwood forest, and effects of conversion to pasture and exotic pine forest. *Environmental Pollution* 116: S63-S73.
- Buol S.W., Hole F.D., McCracken R.J., Southard R.J. 1997. *Soil genesis and classification*. 4<sup>th</sup> ed., Iowa State Univ. Press, Ames, Iowa, USA, 527 pp.
- Carter M. 2002. Soil quality for sustainable management: organic matter and aggregation indicators that maintain soil functions. *Agronomy Journal* 94: 38-47.
- Chander K. and Brookes P.C. 1991a. Microbial biomass dynamics during the decomposition of glucose and maize in metal-contaminated and non-contaminated soils. *Soil Biology and Biochemistry* 23: 917-925.
- Chander K. and Brookes P.C. 1991b. Effects of heavy metals from past application of sewage sludge on microbial biomass and organic matter accumulation in a sandy loam and silty loam U.K. soil. *Soil Biology and Biochemistry* 23: 927-932.
- Christensen B.T. 2001. Physical fractionation of soil and structural and functional complexity in organic matter turnover. *European Journal of Soil Sciences* 52: 345-353.
- Day P.R. 1965. Particle fractionation and particle size analysis. In: *Methods of soil analysis, physical and mineralogical properties*. Black C.A. (Ed.), American Society of Agronomy, Madison, WI. pp. 545-567.
- Feller C., Burtin G., Gérard B., Balesdent J. 1991. Utilisation des résines sodiques et des ultrasons dans le fractionnement granulométrique de la matière organique des sols: intérêts et limites. *Sciences du Sol*, 29 (2): 77-94.
- Gayoso J., Guerra J., Alarcón D. 2002. Contenido de carbono y funciones de biomasa en especies nativas y exóticas. In: *Medición de la capacidad de captura de carbono en bosques de Chile y promoción en el Mercado mundial*. Proyecto FONDEF D98I1076, Informe Técnico, Univ. Austral de Chile, Valdivia, 157 pp.
- Giddens K.M., Parfitt R.L., Percival H.J. 1997. Comparison of some soil properties under *Pinus radiata* and improved pasture. *New Zealand J. Agricultural Research* 40, 409-416.

- Gobierno Regional de Aysén. 2002. Plan Regional, Ordenamiento Territorial, Región de Aysén. Gobierno Regional de Aysén, Chile.
- Guo L.B. and Gifford R.M. 2002. Soil carbon stocks and land use change: a meta analysis. *Global Change Biology* 8: 345-360.
- Huygens D., Boeckx P., Van Cleemput O., Oyarzun C., Godoy R. 2005. Aggregate and soil organic carbon dynamics in South Chilean Andisols. *Biogeosciences* 2: 159-174.
- IPCC. 2001. Climate change 2001: the scientific basis. In: Houghton J.T., Ding Y., Griggs D.J., Noguier M., van der Linden P.J., Dai X., Mashell K., Johnson C.A. (Eds.), Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univ. Press, Cambridge, UK, 881 pp.
- Jenkinson D.S. 1988. Determination of microbial biomass carbon and nitrogen in soil. In: Advances in nitrogen cycling in agricultural ecosystems. Wilson J.R. (Ed.), CAB International, Wallingford, UK, pp. 368-386.
- Jobbagy E.G. and Jackson R.B. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications* 10: 423-436.
- Joergensen R.G. and Brookes P.C. 1990. Ninhydrin-reactive nitrogen measurements of microbial biomass in 0.5 M K<sub>2</sub>SO<sub>4</sub> soil extracts. *Soil Biology Biochemistry* 22: 1023-1027.
- Kelting D.L., Burger J.A., Edwards G.S. 1998. Estimating root respiration, microbial respiration in the rizosphere, and root-free respiration in forest soils. *Soil Biology and Biochemistry* 30: 961-968.
- Kimmins J.P. 2004. Forest ecology: a foundation for sustainable forest management and environmental ethics in forestry. 3<sup>rd</sup> Ed. Prentice Hall, Upper Saddle River, NJ, 611 pp.
- Lal R., Kimble J.M., Levine E., Stewart B.A. 1995. World soils and greenhouse effect: an overview. In: Soils and global change. Lal R., Kimble J.M., Levine E., Stewart B.A. (Eds.), CRC Lewis Publishers, Boca Raton, pp. 1-7.
- Lee K.H. and Jose S. 2003a. Soil respiration and microbial biomass in a pecan - cotton alley cropping system in Southern USA. *Agroforestry Systems* 58: 45-54.
- Lee K.H. and Jose S. 2003b. Soil respiration, fine root production, and microbial biomass in cottonwood and loblolly pine plantations along a fertilizer gradient. *Forest Ecology and Management* 185: 263-273.



- Litton C.M., Raich J.W. Ryan M.G. 2007. Carbon allocation in forest ecosystems. *Global Change Biology* 13: 2089-2109.
- Loguercio G.A. and Defosse G. 2001. Ecuaciones de biomasa aérea, factores de expansión y de reducción de la lenga *Nothofagus pumilio* (Poepp. et Endl.) Krasser en SO del Chubut, Argentina. In: Proceedings of Simposio internacional de medición y monitoreo de captura de carbono en ecosistemas forestales, 18-20 October 2001, Valdivia, Chile, pp. 1-11.
- Mallik A.U. and Hu D. 1997. Soil respiration following site preparation treatments in boreal mixedwood forest. *Forest Ecology and Management* 97: 265-275.
- Marx D.H. 1991. The practical significance of ectomycorrhizae in forest establishment. In: Ecophysiology of ectomycorrhizae of forest trees. The Marcus Wallenberg Foundation, Symposia Proc.7, Falun, Sweden, pp. 54-90.
- McLean E.O. 1982. Soil pH and lime requirement. In: Methods of soil analysis, 2<sup>nd</sup> Ed., Part 2. Chemical and microbiological properties. Page A.L. (Ed.), Madison, WI, pp. 199-224.
- Ocio J.A. and Brookes P.C. 1990. An evaluation of methods for measuring the microbial biomass in soils following recent additions of wheat straw and the characterization of the biomass that develops. *Soil Biology and Biochemistry* 22: 685-694.
- Paul K.I., Polglase P.J., Nyakuengama J.G., Khanna P.K. 2002. Change in soil carbon following afforestation. *Forest Ecology and Management* 168: 241-257.
- Peichl M., Thevathasan N.V., Gordon A.M., Huss J., Abohassan R. 2006. Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. *Agroforestry Systems* 66: 243-257.
- Post W.M. and Kwon K.C. 2000. Soil carbon sequestration and land-use change: Processes and potentials. *Global Change Biology* 6: 317-328.
- Read D.J. 1991. Mycorrhizal fungi in natural and semi-natural plant communities. In: Ecophysiology of ectomycorrhizae of forest trees. The Marcus Wallenberg Foundation, Symposia Proc.7, Falun, Sweden, pp. 27-53.
- Ross D.J., Tate K.R., Scott N.A., Wilde R.H., Rodda N.J., Townsend J.A. 2002. Afforestation of pastures with *Pinus radiata* influences soil carbon and nitrogen pools and mineralisation and microbial properties. *Australian Journal of Soil Research* 40: 1303-1318.

- Ross D.J., Tate K.R., Scott N.A., Feltham C.W. 1999. Land-use change: effects on soil carbon, nitrogen and phosphorus pools and fluxes in three adjacent ecosystems. *Soil Biology and Biochemistry* 31: 803-813.
- Rowell D.L. 1994. *Soil science, methods and applications*. Longman Scientific and Technical, Essex, UK, 350 pp.
- Sadzawka A., Carrasco M., Grez R., Mora M., Flores H., Neaman A. 2006. Recommended analytical methods for Chilean soils. Ministry of Agriculture-INIA, Santiago, Chile.
- Saggar S., Hedley C.B., Salt G.J. 2001. Soil microbial biomass, metabolic quotient, and carbon and nitrogen mineralisation in a 25-year-old *Pinus radiata* Agroforestry regime. *Australian Journal of Soil Research* 39: 491-504.
- SAG-Gobierno Regional de Aysén. 1999. Informe Final Proyecto FNDR “Levantamiento para el Ordenamiento de los Ecosistemas de Aysén”. SAG, Ministerio de Agricultura, Gobierno Regional de Aysén, Chile.
- SAS Institute Inc. 2002. *SAS user's guide: Statistics*. SAS Institute Inc., Cary, NC, USA.
- Scott N.A. and Binkley D. 1997. Litter quality and annual net N mineralization: comparison across North American forest sites. *Oecologia* 111: 151-159.
- Silva F., Ahumada M., Cerda J. 1999. Guide for pasture condition in the Intermediate Temperate ecological region of Aysén. Regional Government of Aysén, Chile. Ministry of Agriculture-SAG, Santiago, Chile.
- Six J., Elliott E.T., Paustian K., Doran J.W. 1998. Aggregation and soil organic matter accumulation in cultivated and native grassland soils. *Soil Science Society of America Journal* 62: 1367-1377.
- Six J., Elliott E.T., Paustian K. 2000. Soil macroaggregate turnover and microaggregate formation: A mechanism for C sequestration under no-tillage agriculture. *Soil Biology and Biochemistry* 32: 2099-2103.
- Six J., Callewaert P., Lenders S., De Gryze S., Morris S.J., Gregorich E.G., Paul E.A., Paustian K. 2002. Measuring and understanding carbon storage in afforested soils by physical fractionation. Division S-7 Forest and Range Soils. *Soil Science Society of America Journal* 66: 1981-1987.
- Soil Survey Staff. 1992. *Soil survey laboratory methods manual*. Soil Conservation Service, USDA, Washington, DC, USA.

- Soil Survey Staff. 2006. Keys to Soil Taxonomy. Natural Resources Conservation Service, USDA, Washington, DC, USA.
- Stolpe N., Muñoz C., Zagal E., Ovalle C. 2008. Modeling soil carbon storage in the "Espinal" agroecosystem of Central Chile. *Arid Land Resource Management* 22: 148-158.
- Tate R.L. 1987. Organic matter transformation: ecosystem examples. In: *Soil organic matter, biological and ecological effects*. Wiley, New York, USA, pp. 26-53.
- Thomas G.W. 1982. Exchangeable cations. In: *Methods of soil analysis, 2<sup>nd</sup> Ed., Part 2, Chemical and microbiological properties*. Page A.L. (Ed.), Madison, WI, pp.159-165.
- Vance E.D., Brookes P.C., Jenkinson D.S. 1987. An extraction method for measuring microbial biomass C. *Soil Biology and Biochemistry* 19: 703-707.
- Wada K. 1985. The distinctive properties of Andosols. In: *Advances in soil sciences, Vol. 2*, Stewart B.A. (Ed.), Springer Verlag, New York, USA, pp. 174-229.
- Weber M. 1999. Kohlenstoffvorräte eines *Nothofagus*-Primärwaldes auf Feuerland. *Forstwissenschaftliches Centralblatt* 118: 156-166.
- Wilde S.A. 1964. Changes in soil productivity induced by pine plantations. *Soil Sciences* 97: 276-278.
- Witter E., Martensson A.M., Garcia F.V. 1993. Size of the soil microbial biomass in a long-term experiment as affected by different N-fertilizers and organic manure. *Soil Biology and Biochemistry* 25: 659-669.
- Wright A.F. and Bailey J.S. 2001. Organic carbon, total carbon, and total nitrogen determinations in soils of variable calcium carbonate contents using a Leco CN-2000 dry combustion analyzer. *Communications in Soil Science and Plant Analysis* 32: 3243-3258.
- Yakimento E.Y. 1998. Soil comparative evolution under grasslands and woodlands in the forest zone of Russia. In: *Management of carbon sequestration in soil*. Lal R., Kimble J.M., Follet R.F., Stewart B.A. (Eds.), CRC Press, Boca Raton, FL, USA pp. 391-404.
- Zagal E. and Córdova C. 2003. Fraccionamiento físico de la materia orgánica en un Andisol cultivado. In: *Manejo Sustentable de Suelos Chilenos*, Vidal I. (Ed.), University of Concepción, Chillán, Chile. pp. 282-290.
- Zagal E. and Córdova C. 2005. Soil organic matter quality indicators in a cultivated Andisol. *Agricultura Técnica* 65(1): 186-197.

## CAPITULO 2

Stolpe N.B., Dube F., Zagal E. 2010. **Calibration of CO2FIX to native forest, pine plantation, and pasture on a volcanic soil of the Chilean Patagonia.** Acta Agriculturae Scandinavica Section B – Soil and Plant Science, 60(3): 235-244.

**Abstract:** Terrestrial stocks of soil carbon (C) are an important part of the global C budget, and are considered in the Kyoto Protocol. However, there have been few studies in forested regions over the influence of land-use changes on volcanic soils (Andisols) and its simulation through models. The objective of this study was to determine the influence of native forest, pine plantation, and pasture on the C content and organic matter quality in an Andisol of the Chilean Patagonia, and to simulate the effects using the CO2FIX model. The model was calibrated to each land-use area, and C content was determined in lenga (*Nothofagus pumilio*) and pine (*Pinus ponderosa*) trees using previously derived allometric equations, and in pasture by direct measurement. The C content was measured in soils (0-100-cm depth) (Typic Hapludands; Umbric Andosols), and the light (>212  $\mu\text{m}$ ), intermediate (212-53  $\mu\text{m}$ ), and heavy (<53  $\mu\text{m}$ ) fractions of organic matter (0-40-cm depth), which have progressively lower labilities. Total C (vegetation + soil) in the lenga forest was greater than in the pine plantation or pasture, but the C content in soil was greater in the pasture than in the other areas. Over all sites, there was more C in the light fraction than in the intermediate and heavy fractions at 0-5-cm depth, but C content was greater in the heavy fraction with increasing soil depths. However, the quality of soil organic matter was better in the lenga forest compared with the other sites, as indicated by higher ratios of light fraction C between 0-20- and 20-40-cm depths. CO2FIX adequately simulated the C contents in the vegetation, and recalibration for slower degradation rates of soluble organic compounds in Andisols provided better approximation of C content in soil, but overestimated the content of labile C.

**Keywords:** Andisols, carbon, organic, sequestration, simulation.

## 2.1. Introduction

There is interest in quantifying and modeling carbon (C) stocks in vegetation and soils because of increasing awareness of global warming and its potential mitigation through C sequestration in terrestrial ecosystems, which is considered in the Kyoto Protocol. Global stocks have been estimated at 1500 and 500 Pg C for soils and plants, respectively, compared with 730 Pg C in the atmosphere. Land-use changes may significantly affect the C cycle, and climate change, through biological processes and feedbacks which determine whether the soil and vegetation act as a "source or sink" for atmospheric C. For example, the CO<sub>2</sub> content in the atmosphere has increased by 30% since pre-industrial times, and this rise has been attributed to the combustion of fossil fuels, and to deforestation (IPCC, 2001). Consequently, deforestation in remote regions of the world, such as southern Chile, may have affected the local terrestrial C stocks, as well as influenced the concentration of atmospheric CO<sub>2</sub>.

### 2.1.1. The Aysén Region of Chile

The Aysén Region of the Chilean Patagonia is located between latitudes 44° and 49° S, and includes 110,000 km<sup>2</sup> of forested mountain ranges, volcanoes, and valleys with numerous lakes and rivers. Approximately 25% of the region is protected as wildlife reserves or national forests. The forest is classified as Humid Boreal, and the principal trees include evergreen (2.2 million ha), coigüe (*Nothofagus dombeyi*) (0.6 million ha), lenga (*N. pumilio*) (1.4 million ha), and Guaitecas cypress (*Pilgerodendron uviferum*) (0.5 million ha), but the forest composition in a given area is determined by the climatic gradient between the mountains and the coast, as well as by local microclimatic conditions (Silva, 2004; SERPLAC, 2005). Although glaciers covered much of the region during the Pleistocene, more recent deposits of volcanic breccias and ash are the parent materials of many soils that are found in the region (Ferrando and Muñoz, 1979).

The region is transversally classified into several eco-zones based on physiographic-climatic characteristics that grade from the Temperate Humid Zone along the coast, to the Cold Steppe Zone near the Chile-Argentina border (Silva, 2004). The climate of the Intermediate

Temperate Zone of interest has a mean annual precipitation that ranges from 900-1300 mm, with the lower values generally found in areas more distant inland from the coast; and mean monthly temperatures of 12-14°C in the summer (December to February) and 2-3°C in winter (June to August). November to February is normally frost free, and may include strong westerly and southwesterly winds. Representative areas of the Intermediate Temperate Zone include the Simpson Valley, Coyhaique, and Emperador Guillermo (Silva et al., 1999).

### 2.1.2. Change of soil use

The native forest that once covered large portions of the Aysén Region has been reduced to an area of 4.7 million hectares (43% of the region) because between 1920 and 1940 many areas were burned by settlers in order to create pastures and to claim ownership of lands (Veblen et al., 1995). Overgrazing of the pastures and transitional areas has also caused soil erosion and further degradation of the adjoining native forest (Silva, 2004). In some areas, sloping soils are so severely eroded that it has become difficult to reestablish the native forest, either by natural means or by human intervention (Ortega and Rodriguez, 1994). However, forest plantations of introduced species, principally *Pinus ponderosa*, *Pseudotsuga menziesii*, and *Pinus contorta*, have been made on approximately 27,000 ha of degraded soils, but other cleared areas continue to be used as pastures (Silva, 2004).

In general, volcanic soils (Andisols), such as those found in southern Chile, form in humid temperate climates and have unique properties compared with non-volcanic soils, including a dark color of the A horizon due to the high content of organic matter caused by the accumulation of stable Al, Fe-humus complexes (Shoji et al., 1993). Other properties include: low bulk density, fine crumb or granular structure, the presence of poorly crystalline clay (allophane or imogolite) having a pH-dependent charge and high specific surface area, high content of water at 1500 kPa tension relative to the measured clay content, low exchangeable bases, and elevated retention of P (Wada, 1985). Although Andisols cover just 0.84% of the world's land surface, they are an important land resource because many occur in areas of the Pacific Rim that have a high density of population (Shoji et al., 1993). Intensive use of volcanic soils in central Chile has been shown to significantly decrease the organic matter (and

C) content, and diminish the overall soil quality, principally due to lower return of organic debris to soil, lower recycling of plant nutrients, excessive oxidation of soil organic matter (SOM), and erosion (Sandoval et al., 2003; Zagal and Córdova, 2005).

A previous investigation in New Zealand found lower contents of C and N in volcanic soil (Typic Udivitrands; 0-20cm depth) under a 19-year plantation of *Pinus radiata* than under broadleaf native forest or pasture (Ross et al., 1999). In southern Chile, near Paillaco, the C content in volcanic soil (0-10 cm depth) was higher under pasture (6.7 kg C m<sup>-2</sup>) than under native forest of *Nothofagus oblique* (5.8 kg C m<sup>-2</sup>) and plantation of *Pinus radiata* (4.3 kg C m<sup>-2</sup>), but the rate of C mineralization was greater in soil under native forest, with decreasing rates in soils of pine plantation and pasture, respectively (Huygens et al., 2005). Additionally, the conversion of native forest into pastures and crops on volcanic soils of southern Chile has been found to significantly lower the content of N in high-molecular-weight fractions of SOM (humin and humic acid) (Borie et al., 2002).

### 2.1.3. Fractions of soil organic matter

Organic matter can be fractionated in the laboratory according to its lability, which refers to the relative ease by which it can be degraded by soil microorganisms thereby releasing N and other nutrients that can be subsequently used by plants. Lability decreases with increasing aromaticity of the fraction, greater occlusion within soil aggregates, or adsorption to clay minerals (Christensen, 2001; Krull et al., 2003). The most labile fraction has been used as an indicator of sustainable management because the amount in soil changes on a monthly to yearly basis depending upon vegetation, management, and the balance between the additions of plant residues and their rate of decomposition in soil. The most labile fraction has been designated as the mineralizable C, microbial biomass, particulate organic matter, and also the light fraction with a diameter >212 µm (Powlson, 1994; Christensen, 2001; Zagal et al., 2002). The intermediate fraction of SOM is less labile and is associated with macroaggregates in soil (212-53 µm in diameter), and the heavy fraction is the most stable and is associated with microaggregates in soil (<53 µm in diameter) (Balesdent et al., 1991; Christensen, 2001). In a previous investigation, the mean residence time of organic C was 100 years in the light and

intermediate fractions, and 575 years in the heavy fraction (Monreal et al., 1997). In volcanic soil (Typic Haploxerands) of central Chile, the contents of light fraction C and total C (0-20 cm depth) decreased with increasing intensity of agricultural management (Zagal et al., 2002), and the ratio of C content in the light fraction between 0-20 and 20-40 cm depths has been used as an indicator of sustainable management (Stolpe et al., 2008).

#### *2.1.4. The CO2FIX Model*

The CO2FIX model was developed to estimate the quantity and fluxes of C in natural forests and plantations, silvopastoral systems, and soils to determine the number of transferable C credits according to the Kyoto Protocol (Schelhaas et al., 2004). In general, the model uses annual steps to estimate the C content in trunks, branches, leaves, roots, etc. based on user-input parameters of climate, tree species, and management. The model includes the sub model YASSO that uses basic climatic data and estimates of forest-litter production and litter quality to determine the C content in soil (0-100 cm depth) and fractions of organic matter. The sub model fractions have progressively slower rates of C turnover in soil and include soluble compounds ( $< 0.82 \text{ year}^{-1}$ ), cellulose ( $0.30 \text{ year}^{-1}$ ), lignin-like compounds ( $0.22 \text{ year}^{-1}$ ), humus 1 ( $0.012 \text{ year}^{-1}$ ), and humus 2 ( $0.0012 \text{ year}^{-1}$ ) (Liski et al., 2005).

The objective of this study was to determine the influence of lenga forest, pine plantation, and pasture in the Chilean Patagonia on the total C content and organic matter quality in volcanic soil, and to calibrate the CO2FIX model to simulate C contents in the soil and vegetation.

## **2.2. Materials and methods**

### *2.2.1. Site description*

The study area was located in the Mano Negra Sector, 30 km north of the city of Coyhaique, Chile (Lat S  $45^{\circ}25'$ , Long W  $72^{\circ}00'$ ). The area has a western exposed slope (6-8%), an elevation of 730 m above sea level, and adjacent land uses (200 m apart) that include: 1) 150+- year stand of unmanaged lenga forest (*Nothofagus pumilio*) (100 ha); 2) 16-year stand of pine



plantation (*Pinus ponderosa*) (5 ha); and 3) 50+-year pasture (5 ha) (Figure 1). Originally, the entire study area was covered with indigenous forest (mostly *Nothofagus pumilio*) before the pasture established itself in the sectors that were burned, as previously described, and the pine plantation was later established over pasture.



Figure 2.1. Sites of lenga forest (*Nothofagus pumilio*) (A), ponderosa pine plantation (*Pinus ponderosa*) (B), and natural pasture (C) on volcanic soil (Typic Hapludands; Umbric Andosols) in the Chilean Patagonia.

The pasture is currently used for cattle grazing (2 cows ha<sup>-1</sup>), and consists of a mixture of perennial grasses (*Dactylis glomerata*, *Holcus lanatus*, *Poa pratensis*), leguminous plants (*Trifolium pratense*, *T. repens*) and other accompanying species. The pasture was fertilized during the 2004-2005 growing season with 16 kg N ha<sup>-1</sup>, 30 kg Mg ha<sup>-1</sup>, and 79 kg S ha<sup>-1</sup>; and in 2005-2006 with 24 kg N ha<sup>-1</sup>, 17 kg Mg ha<sup>-1</sup>, and 77 kg S ha<sup>-1</sup>. The pine plantation was established in 1991 with a density of 2000 trees ha<sup>-1</sup> and in 2003 it was thinned down to 800 trees ha<sup>-1</sup>. In March 2006 (the beginning of this investigation), the average tree height and diameter at breast height (DBH) were 6.3 m and 17.3 cm, respectively. The lenga forest had a density of 780 trees ha<sup>-1</sup>, average tree height of 23.7 m, and DBH of 32.5 cm.

The study area has climatic conditions that are classified as "Trans Andean with Steppe regeneration" (DMC, 2009), with dry summers and cool, humid winters (Figure 2), and the mean annual temperature and precipitation are 8.1°C and 1206 mm, respectively. Normally, only 15% of the annual precipitation occurs between December and February, coinciding with

the windiest and warmest period (Silva et al., 1999; DGF, 2008). The soil formed from relatively unaltered volcanic ash deposits and is characterized as deep, well drained, moderately acidic (Silva et al., 1999), and has been classified as Typic Hapludands (Soil Survey Staff, 2006), or Umbric Andosols (IUSS Working Group, 2006).

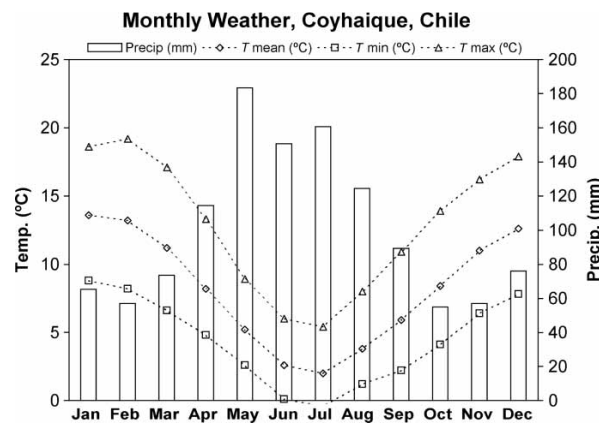


Figure 2.2. Mean monthly precipitation and temperatures (maximum, minimum, and median) near Coyhaique, Chile (Lat S 45°38', Long W 72°07' 310 m elevation) (DGF, 2008) as input into the CO2FIX model.

### 2.2.2. Soil analysis

The experimental design was Completely Randomized, and soil samples were collected in March, 2006 at three sub sites within the lenga forest, pine plantation, and pasture (0.5-ha areas). Representative, bulk samples of soil were taken at 0-5, 5-10, 10-20, and 20-40 cm depths in each sub site, stored in plastic bags and transported to the laboratory where they were air dried. The SOM was fractionated using a modified procedure from Balesdent et al. (1991) in which the soil was mechanically dispersed, followed by a physical separation through wet sieving. Briefly, 50 g of air-dried soil and 10 small glass balls (6-mm diameter) were placed in 180 ml of distilled water and shaken in a rotator agitator (REAX 2, Heidolph Instrument, Schwabach, Germany) at 50 cycles per minute for 16 hours, causing the mechanical rupture of soil macroaggregates. Then, the soil was wet sieved through 53- and 212- $\mu$ m stainless steel sieves thereby separating three fractions of organic matter: a) light fraction (above 212  $\mu$ m) that was separated from the sand by flotation and sedimentation in

water, b) intermediate fraction (53-212  $\mu\text{m}$ ), and c) heavy fraction (below 53  $\mu\text{m}$ ) without ultrasonic dispersion. The C content of the light, intermediate and heavy fractions was then measured by dry combustion (Wright and Bailey, 2001), and the ratio of C content in the light fraction between 0-20 and 20-40 cm depths was calculated as an indicator of sustainable management, whereby soil quality decreases with lowering ratios (Stolpe et al., 2008). Additional, deeper, samples of soil were collected in the sub sites, and total C content in whole soil (0-100 cm depth) was determined by dry combustion (Wright and Bailey, 2001). Analysis of variance (ANOVA) and least significant difference (LSD) of means were used to test for differences among soils of the lenga forest, plantation, and pasture (Steele and Torrie, 1980).

Soil morphology was described at a sub site in each land-use area in order to check the soil uniformity and classification among the lenga forest, pine plantation, and pasture. Representative, bulk samples were taken at 0-20, 20-40, 40-60, 60-80, and 80-100-cm soil depths (Soil Survey Staff, 2004). In the laboratory, the soil samples were air dried and sieved (2-mm mesh) for the subsequent characterization analyses. Soil texture was determined using the hydrometer method (Day, 1965), soil water contents at 33 and 1500 kPa were measured using a pressure plate apparatus, soil pH was determined using a 1:1 mixture of soil and water, and the % P sorption was measured in the soil (Soil Survey Staff, 2004). Effective cation-exchange capacity (ECEC) was measured using 1 N  $\text{NH}_4\text{OCOCH}_3$  for extraction of the exchangeable cations Ca, Mg, K, and Na, and 1 N KCl for exchangeable Al, and available N ( $\text{NO}_3\text{-N}$ ), P (Olsen), and K were also measured in the soils (Sadzawka et al., 2006). Additionally, the content of  $^{13}\text{C}$  and total C in the soils (0-100 cm depth) was determined using an elemental analyzer (ANCA-SL, PDZ-Europa, UK) coupled to an Isotope Ratio Mass Spectrometer (20-20, PDZ Europa, UK), and the  $^{13}\text{C}/^{12}\text{C}$  ratios were used to obtain the  $\delta^{13}\text{C}$  values (Huygens et al., 2005). For the latter analysis, the surface soil was sampled at finer increments of 0-5, 5-10, and 10-20 cm in addition to the deeper samples to 100 cm.

### 2.2.3. Tree and pasture biomass

The lenga forest and the pine plantation were inventoried in March, 2006 using five circular plots of 200  $\text{m}^2$  that were randomly established in each site, measuring the total height of all

the trees in the plots and the DBH using a clinometer and diametric tape, respectively. The measured variables were then used to calculate aerial and subterranean biomass, using existing biomass functions for the climatic zone that were developed for *Nothofagus pumilio* (Weber, 1999; Loguercio and Defosse, 2001), and *Pinus ponderosa* (Gayoso et al., 2002). Partitioning of the aerial biomass compartments was as follows: 1) *Nothofagus pumilio*, 87% consisted of stems, 12% of branches, and 1% of foliage; and 2) *Pinus ponderosa*, 47% of aerial biomass consisted of stems, 25% of branches, and 28% of foliage. The live ground cover in the lenga forest and pine plantation was estimated at <1 Mg dry matter ha<sup>-1</sup> due to extensive litter coverage of leaves and needles, respectively.

To obtain the aboveground biomass in the pasture, the vegetation was clipped at ground level from 9 randomly selected 0.25-m<sup>2</sup> quadrants. In the laboratory, the vegetative material was dried at 65°C for 5 days and weighed. To measure the subterranean biomass, the live roots of each quadrant were removed with a spade, rinsed in the laboratory in slow-running water in order to remove soil from the roots, and then dried and weighed as previously described. The above- and below-ground values were summed to calculate the total pasture biomass per hectare.

#### 2.2.4. Calibration of CO2FIX

The C content of the lenga forest, pine plantation, and pasture biomass was simulated using the CO2FIX (v. 3.1) model (Schelhaas et al., 2004); and soil using the sub model YASSO (Liski et al., 2005). Model calibration to regional climatic conditions included the mean monthly temperature and precipitation of the growing season (August through May) (Figure 2); with season totals of 2961 Growing Degree Days above 0°C, 500 mm of potential evapotranspiration, and 895 mm of precipitation. The weather data were taken from the nearest station located in Coyhaique, which has similar climatic conditions as the study area (DMC, 2009). Based on interviews with the landowner, the tree-rotation lengths were input at 150 and 35 years for *Nothofagus pumilio* and *Pinus ponderosa*, respectively, with 60% of the pine biomass removed (thinned) after 13 years. The C content in stems, foliage, branches, and roots was set to 50%, and the current annual increments (CAI) of both species were calculated

according to published values (Loguercio and Defosse, 2001; Quiroz and Rojas, 2003). The pasture was simulated, in effect, by inputting an annual rotation of grass as a "tree" having very small stem volume, no branches, but with a lot of foliage and roots (Schelhaas et al., 2004). Other initial soil parameters were taken from the sample files for broadleaf trees, pine, and pasture included in the CO2FIX model (Table 1). In each scenario, the model was run for a period representing 300 years, and recalibration was based on comparison of the simulated and measured contents of total C in soil, as is described in the following section.

Table 2.1. Initial vegetative and soil parameters input to the CO2FIX model for the lenga forest, ponderosa pine plantation, and natural pasture sites of southern Chile.

| Parameter   | Lenga forest | Pine plantation | Pasture |
|---|--------------|-----------------|---------|
| Maximum biomass Mg ha <sup>-1</sup>                     | 7.00         | 1.50            | 0       |
| Stem density Mg dry matter m <sup>-3</sup>              | 0.25         | 0.47            | 1.00    |
| Stem annual C Mg C ha <sup>-1</sup>                     | 298.98       | 11.85           | 0.01    |
| Foliage annual C Mg C ha <sup>-1</sup>                  | 1.17         | 1.14            | 1.38    |
| Foliage turnover time year                              | 1.00         | 0.25            | 0.80    |
| Branch annual C Mg C ha <sup>-1</sup>                   | 28.30        | 0.20            | 0       |
| Branch turnover time year                               | 0.03         | 0.05            | 0       |
| Roots annual C Mg C ha <sup>-1</sup>                    | 48.71        | 11.56           | 1.04    |
| Root turnover time year                                 | 0.10         | 0.05            | 0.80    |
| Soil initial C broadleaf litter Mg C ha <sup>-1</sup>   | 0.07         | 0.95            | 0.20    |
| Soil initial C pine wood litter Mg C ha <sup>-1</sup>   | 0.09         | 0.80            | 0       |
| Soil initial C coarse wood litter Mg C ha <sup>-1</sup> | 0.24         | 0.71            | 0       |
| Soil initial available compounds Mg C ha <sup>-1</sup>  | 0.07         | 0.50            | 1.01    |
| Soil initial total dissolved N Mg N ha <sup>-1</sup>    | 1.13         | 1.54            | 0.05    |
| Soil initial available nitrogen Mg N ha <sup>-1</sup>   | 1.04         | 1.50            | 0.25    |
| Initial available P Mg P ha <sup>-1</sup>               | 0.04         | 0.06            | 0.05    |
| Initial available K Mg K ha <sup>-1</sup>               | 11.26        | 16.16           | 40.45   |
| N <sub>2</sub> O flux per soil mineralization Mg N      | 0.00         | 0.00            | 0.00    |
| Decomposition rate of soil compounds kg C               | 0.00         | 0.50            | 0.80    |

## 2.3. Results and discussion

### 2.3.1. Soils and total carbon

The soil morphology was relatively uniform among the lenga forest, pine plantation, and pasture, as evidenced by the similar sequence of horizons (A1, A2, Bw, BC, C1, C2) (Table 2), which was expected considering that the parent material was arially deposited volcanic ash, the topographic uniformity, and the native forest that once covered the entire sector. However, the lenga forest soil had a thin (2-cm thick) organic horizon on the surface, which was not present in the other sites, and NO<sub>3</sub>-N, available P, and K were higher in the lenga site

(0-20 cm depth) than in the pasture and pine sites (Table 3). Overall, the ratios of 1500 kPa water to clay were well above 0.6, which is a characteristic of volcanic soils. High ratios indicate poor dispersion of clay during particle-size analysis, and under measurement of the actual clay content in the soil (Stolpe and Lewis, 1990).

Table 2.2. Descriptions of morphology of the volcanic soil (Typic Hapludands; Umbric Andosols) under lenga forest, pine plantation, and natural pasture in the Chilean Patagonia.

| Site and depth (cm) | Horiz.          | Roots    | Color/moist. soil | Texture          | Structure                     | Microconsistency |
|---------------------|-----------------|----------|-------------------|------------------|-------------------------------|------------------|
| <b>Lenga</b>        |                 |          |                   |                  |                               |                  |
| 0-5                 | A1              | Vertical | 10YR 2/1          | 1:1 <sup>1</sup> | Structure: irregular columnar | Finable          |
| 5-9                 | A1              | Vertical | 10YR 2/1          | 1:1.5            | Block size 1.50 v. columnar   | Finable          |
| 9-15                | A2              | Many     | 10YR 3/1          | 1:1.5            | Block size 1.50 v. columnar   | Finable          |
| 15-25               | Bw              | Vertical | 10YR 4/2          | 1:1.5            | Block size 1.50 v. columnar   | Very loath       |
| 25-35               | Bt <sup>2</sup> | 20       | 10YR 4/2          | 1:1.5            | Weak fine granular            | Very loath       |
| 35-40               | C1              | 20       | 10YR 4/2          | 1:1.5            | Clayey                        | Fin <sup>3</sup> |
| 40-100              | C2              | Some     | 10YR 4/2          | 1:1.5            | Clayey                        | Fin <sup>3</sup> |
| <b>Pine</b>         |                 |          |                   |                  |                               |                  |
| 0-15                | A1              | Many     | 10YR 2/1          | 1:1.5            | Block size 1.50 v. columnar   | Finable          |
| 15-17               | A2              | Vertical | 10YR 3/1          | 1:1.5            | Block size 1.50 v. columnar   | Very loath       |
| 17-30               | Bw              | 20       | 10YR 3/1          | 1:1.5            | Weak fine granular            | Very loath       |
| 30-40               | Bt <sup>2</sup> | 20       | 10YR 4/2          | 1:1.5            | Clayey                        | Very loath       |
| 40-100              | C1              | 20       | 10YR 4/2          | 1:1.5            | Clayey                        | Fin <sup>3</sup> |
| 100-150             | C2              | Some     | 10YR 4/2          | 1:1.5            | Clayey                        | Fin <sup>3</sup> |
| <b>Pasture</b>      |                 |          |                   |                  |                               |                  |
| 0-15                | A1              | Many     | 10YR 2/1          | 1:1.5            | Block size 1.50 v. columnar   | Finable          |
| 15-24               | A2              | Many     | 10YR 3/1          | 1:1.5            | Block size 1.50 v. columnar   | Finable          |
| 24-28               | Bw              | Many     | 10YR 3/1          | 1:1.5            | Weak fine granular            | Very loath       |
| 28-30               | Bt <sup>2</sup> | Many     | 10YR 4/2          | 1:1.5            | Clayey                        | Very loath       |
| 30-38               | C1              | Vertical | 10YR 4/2          | 1:1.5            | Clayey                        | Fin <sup>3</sup> |
| 38-100              | C2              | 20       | 10YR 4/2          | 1:1.5            | Clayey                        | Fin <sup>3</sup> |

1:1.5 = fairly fine sandy soil texture; 1:1.5L = fine silty clay soil texture

Table 2.3. Chemical properties of the volcanic soil (Typic Hapludands; Umbric Andosols) under lenga forest, ponderosa pine plantation, and natural pasture in the Chilean Patagonia.

| Site and depth (cm) | C   |     | pH  | C/EC                               |      | SO <sub>4</sub> -S  |      |     | Cation exchange capacity | Water-soluble clay |
|---------------------|-----|-----|-----|------------------------------------|------|---------------------|------|-----|--------------------------|--------------------|
|                     | %   | %   |     | cmol <sub>c</sub> kg <sup>-1</sup> | Al   | mg kg <sup>-1</sup> | P    | K   |                          |                    |
| <b>Lenga</b>        |     |     |     |                                    |      |                     |      |     |                          |                    |
| 0-20                | 4.2 | 0.3 | 6.0 | 10.9                               | 0.02 | 13.7                | 12.8 | 223 | 50                       | 3.31               |
| 20-40               | 1.6 | 0.1 | 6.0 | 6.9                                | 0.03 | 2.0                 | 1.1  | 178 | 83                       | 3.27               |
| 40-60               | 1.4 | 0.1 | 6.7 | 7.4                                | 0.01 | 0.8                 | 1.0  | 318 | 82                       | 3.43               |
| 60-80               | 1.4 | 0   | 6.4 | 7                                  | 0.01 | 0.5                 | 3.9  | 178 | 84                       | 3.67               |
| 80-100              | 1.6 | 0.1 | 6.5 | 7.6                                | 0.01 | 0.3                 | 4.4  | 125 | 84                       | 3.37               |
| <b>Pine</b>         |     |     |     |                                    |      |                     |      |     |                          |                    |
| 0-20                | 3.6 | 0.3 | 6.2 | 8.9                                | 0.03 | 1.2                 | 1.3  | 54  | 71                       | 2.71               |
| 20-40               | 2.8 | 0.2 | 6.5 | 11.1                               | 0.04 | 1.0                 | 10.3 | 36  | 72                       | 2.68               |
| 40-60               | 2.1 | 0.1 | 6.8 | 12.4                               | 0.04 | 1.3                 | 5.0  | 37  | 72                       | 2.65               |
| 60-80               | 1.4 | 0.1 | 6.8 | 11.8                               | 0.01 | 0.5                 | 3.3  | 33  | 81                       | 2.38               |
| 80-100              | 1.7 | 0.1 | 6.8 | 5.1                                | 0.01 | 0.3                 | 2.0  | 32  | 82                       | 2.11               |
| <b>Pasture</b>      |     |     |     |                                    |      |                     |      |     |                          |                    |
| 0-20                | 6.1 | 0.4 | 6.1 | 14.1                               | 0.02 | 1.7                 | 21.7 | 152 | 71                       | 3.95               |
| 20-40               | 1.8 | 0.1 | 6.5 | 11.8                               | 0.01 | 0.8                 | 6.7  | 86  | 84                       | 2.70               |
| 40-60               | 1.7 | 0.1 | 6.4 | 6.1                                | 0.01 | 0.5                 | 5.5  | 85  | 88                       | 2.99               |
| 60-80               | 1.4 | 0.1 | 6.5 | 8.8                                | 0.01 | 0.3                 | 4.4  | 106 | 85                       | 2.63               |
| 80-100              | 1.0 | 0.1 | 6.4 | 8.0                                | 0.01 | 0.3                 | 4.7  | 128 | 87                       | 2.04               |

Effective cation-exchange capacity.

The total C content in soil (0-100-cm depth) was higher ( $P < 0.05$ ) under pasture ( $286.7 \pm 10.8 \text{ Mg C ha}^{-1}$ ) than those in soil under pine ( $257.8 \pm 10.1 \text{ Mg C ha}^{-1}$ ) and lenga ( $226.7 \pm 12.5 \text{ Mg C ha}^{-1}$ ), although the total C in vegetation and soil of the lenga forest ( $504.7 \pm 16.0 \text{ Mg C ha}^{-1}$ ) was greater than that of both the pine plantation ( $295.7 \pm 10.7 \text{ Mg C ha}^{-1}$ ) and the pasture ( $309.2 \pm 15.1 \text{ Mg C ha}^{-1}$ ) (Figure 3). The lenga forest had similar C content in soil and in the wood (trunks and branches), but the pine plantation and pasture had more C content in the soil than in the biomass component.

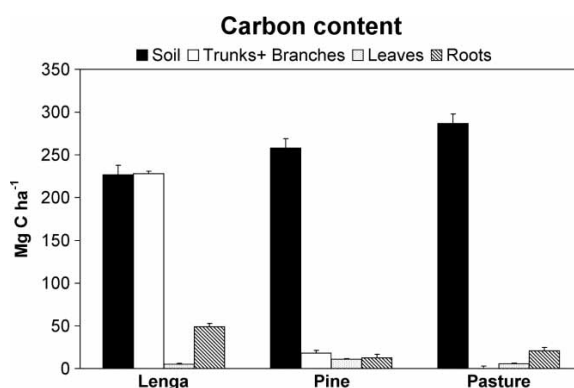


Figure 2.3. Mean carbon content and standard deviation ( $\text{Mg C ha}^{-1}$ ) in volcanic soil (Typic Hapludands) (0-100 cm depth), and vegetation of lenga forest, ponderosa pine plantation, and natural pasture in the Chilean Patagonia.

### 2.3.2. Characterization of soil organic matter

The natural abundance of  $^{13}\text{C}$  in soils of the lenga forest, pine plantation, and pasture showed an enrichment beginning at 5 cm depth, possibly associated with an isotopic discrimination during the decomposition of organic residues in soil which resulted in enrichment in the residual C of the heavier isotope ( $^{13}\text{C}$ ) (Figure 4) compared with the vegetative biomass. In contrast with a previous investigation (Huygens et al., 2005), the soil of the lenga forest had lower values of  $\delta^{13}\text{C}$  throughout the profile compared with the pine plantation and pasture and this may have resulted from the decomposition of organic residues with lower initial content of  $^{13}\text{C}$  such as hemicelluloses, cellulose, and lignin, or higher decomposition rates of organic residues in the lenga soil.

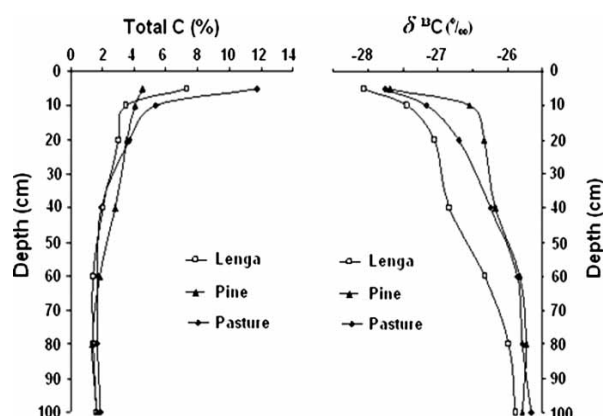


Figure 2.4. Total content of C (%) and  $\delta^{13}\text{C}$  (‰) in volcanic soil (Typic Hapludands; Umbric Andosols) under lenga forest, pine plantation, and pasture in the Chilean Patagonia.

Physical fractionation of SOM of the lenga forest, pine plantation, and pasture showed that, among all sites, the light fraction ( $>212 \mu\text{m}$ ) of the surface (0-5 cm) contained more C ( $P < 0.05$ ) than did the intermediate (212-53  $\mu\text{m}$ ) and heavy ( $<53 \mu\text{m}$ ) fractions (Table 4), but there was more C in the heavy fraction at 5-10, 10-20, and 20-40 cm depths, which indicated that the SOM was more stabilized (less labile) with increasing soil depth. In general, with increasing soil depth there was no significant difference of C content in each fraction among the lenga, pine, and pasture sites, except that the lenga soil had more C in the intermediate fraction at 0-5 and 5-10 cm depths than did the other soils ( $P < 0.05$ ), and that the pine soil had less C in the heavy fraction at 10-20 cm depth did than the other soils. However, the C content in the light fractions of the pine plantation and pasture may have possibly been elevated by charcoal C (which has a longer residence time in soil) from burning of the native forest (Murage et al., 2007). Nevertheless, the ratio of C content of the light fraction between the soil depths of 0-20 and 20-40 cm ( $C_{20/40}$ ) was higher in soil of the lenga forest (9.35), with decreasing values in the pine plantation (7.16) and pasture (6.33) ( $P < 0.05$ ;  $\text{LSD} = 2.97$ ). Lower ratios have been used to indicate the negative effect of more intensive management on the quality of SOM (Stolpe et al., 2008).



Table 2.4. Mean carbon content and standard deviation (Mg C ha<sup>-1</sup>) in the light, intermediate, and heavy fractions of organic matter of volcanic soil (Typic Hapludands; Umbric Andosols) under lenga forest, pine plantation, and pasture in the Chilean Patagonia.

| Site    | Depth (cm) | Soil organic matter        |                       |                | LSD <sup>a</sup> |
|---------|------------|----------------------------|-----------------------|----------------|------------------|
|         |            | C (Mg C ha <sup>-1</sup> ) |                       |                |                  |
|         |            | Light fraction             | Intermediate fraction | Heavy fraction |                  |
| Lenga   | 0-5        | 4395 ± 10.46               | 14.75 ± 0.54          | 17.38 ± 4.67   | 1.00             |
| Pine    | 0-5        | 3385 ± 20.94               | 10.75 ± 1.37          | 14.68 ± 2.72   | 0.25             |
| Pasture | 0-5        | 2498 ± 1.41                | 7.71 ± 0.30           | 11.75 ± 0.95   | 0.08             |
| LSD     |            | 21.44                      | 3.62                  | 4.47           |                  |
| Lenga   | 5-17       | 468 ± 1.22                 | 15.38 ± 0.24          | 21.85 ± 0.55   | 4.50             |
| Pine    | 5-17       | 575 ± 3.94                 | 8.57 ± 1.13           | 11.38 ± 2.20   | 2.28             |
| Pasture | 5-17       | 308 ± 1.27                 | 8.58 ± 2.01           | 11.75 ± 4.82   | 0.35             |
| LSD     |            | 1.82                       | 4.14                  | 1.89           |                  |
| Lenga   | 17-27      | 420 ± 1.07                 | 12.70 ± 0.05          | 20.55 ± 0.20   | 5.57             |
| Pine    | 17-27      | 570 ± 3.24                 | 14.38 ± 0.15          | 20.45 ± 8.00   | 8.35             |
| Pasture | 17-27      | 570 ± 3.84                 | 14.98 ± 0.25          | 21.08 ± 8.54   | 1.05             |
| LSD     |            | 1.75                       | 5.14                  | 11.22          |                  |
| Lenga   | 27-47      | 575 ± 1.55                 | 14.25 ± 0.94          | 20.20 ± 0.25   | 4.29             |
| Pine    | 27-47      | 528 ± 1.75                 | 16.08 ± 0.91          | 20.20 ± 0.47   | 8.10             |
| Pasture | 27-47      | 425 ± 3.57                 | 14.45 ± 1.18          | 21.25 ± 1.49   | 5.12             |
| LSD     |            | 1.87                       | 8.35                  | 10.14          |                  |

<sup>a</sup> LSD values are different at  $p < 0.05$ .

### 2.3.3. Modeling of soil carbon

The first simulation using the CO2FIX model indicated that the C content in vegetation would be higher in the lenga forest than in the pine plantation and pasture, that the soil C contents (0-100 cm depth) would be higher in the lenga forest than in the other sites, and that the least labile humus 2 fraction (Liski et al., 2005) would be higher in the pasture soil and lower in the lenga soil (Table 5). The measured C contents in the vegetation were similar to the simulated values, but the soil C contents (Figure 3) were four to eight times higher than those simulated by the model, and the least labile heavy fraction (<53 μm) was generally similar among the soils (Table 4). The soil discrepancies were attributed to the initial calibration of the sub model YASSO that was based on climate and typical decomposition rates of vegetative litter in non-volcanic soils (Liski et al., 2005). However, Andisols have unique properties that are strongly influenced by characteristics of the parent material, such as a high content of organic matter as a result of the stabilization of dissolved organic C by active Al and Fe from volcanic ash (Shoji et al., 1993; Huygens et al., 2005). Therefore, the first simulation for the lenga forest, pine plantation, and pasture resulted in underestimation of C content of the volcanic soils.

Table 2.5. Carbon content (Mg C ha<sup>-1</sup>) in vegetation and soil (0-100 cm depth) of lenga forest, ponderosa pine plantation, and natural pasture in the Chilean Patagonia, as simulated using the CO2FIX model.

| CO2FIX Output<br>Variable | Lenga forest simulations |        | Pine plantation simulations |        | Pasture simulations |        |
|---------------------------|--------------------------|--------|-----------------------------|--------|---------------------|--------|
|                           | 1                        | 2      | 1                           | 2      | 1                   | 2      |
|                           | Mg C ha                  |        |                             |        |                     |        |
| Biomass trunks            | 188.46                   | 188.46 | 1.84                        | 1.84   | 0.0                 | 0.0    |
| Foliage                   | 0.78                     | 0.78   | 3.28                        | 3.28   | 1.47                | 1.47   |
| Branches                  | 31.66                    | 31.66  | 5.87                        | 5.87   | 0.00                | 0.00   |
| Roots                     | 8.85                     | 8.85   | 6.83                        | 6.83   | 1.1                 | 1.1    |
| Total biomass             | 211.75                   | 211.75 | 17.82                       | 17.82  | 2.59                | 2.59   |
| Soil non-waterly litter   | 1.23                     | 1.23   | 0.76                        | 0.76   | -1.50               | -1.50  |
| Soil fine litter          | 1.49                     | 1.49   | 0.4                         | 0.4    | 0.00                | 0.00   |
| Soil coarse litter        | 0.00                     | 0.00   | 1.0                         | 0.0    | 0.06                | 0.06   |
| Soil soluble              | 0.5                      | 171.6  | 0.4                         | 256.7  | 1.7                 | 252.06 |
| Soil cellulose            | 1.69                     | 1.69   | 0.60                        | 0.5    | 1.93                | 1.93   |
| Soil lignin               | 1.92                     | 1.77   | 1.44                        | 1.27   | 2.99                | 2.48   |
| Soil humus 1              | 10.30                    | 8.42   | 1.83                        | 8.5    | 11.74               | 11.27  |
| Soil humus 2              | 18.7                     | 15.79  | 17.60                       | 17.09  | -12.94              | 18.79  |
| Soil total                | 33.85                    | 202.55 | -13.6                       | 286.00 | 68.87               | 281.06 |
| Total biomass + soil      | 275.60                   | 414.30 | 71.43                       | 313.82 | 71.66               | 292.05 |

<sup>1</sup>Initial conditions as defined in Table 1. <sup>2</sup>KO<sub>sol</sub> = 0.0001, and the lenga forest is followed by pasture (50 years), and pine plantation (16 years). The other initial conditions for lenga forest are as defined in Table 1, and those for natural pasture and pine plantation are the Simulation 2 outputs for the lenga forest and pasture, respectively (as listed in this Table).

The model was re-calibrated to better simulate C content in volcanic soil of the lenga forest by lowering the KO<sub>sol</sub> variable (the decomposition rate of soluble organic compounds in soil) of the YASSO sub model (Schelhaas et al., 2004; Liski et al., 2005) to the minimum value of 0.0001, and the simulation was run again for 300 years (2 rotations of 150 years). The subsequent C content in soil (0-100 cm depth) was 202.6 Mg C ha<sup>-1</sup> (Table 5), which was nearer to, but 11% lower than the measured value of 226.7 Mg C ha<sup>-1</sup> (Figure 3). Although the CO2FIX model is not structured to permit sequences of different vegetative species in the same simulation, they can be achieved by using the final output of one simulation as the initial conditions for the next. Therefore, in the second simulation for the pasture, the KO<sub>sol</sub> was similarly set to 0.0001 and the initial soil values were set according to the output soil values for the second simulation of the lenga forest (Table 5) (justified since pasture followed

burning of the native forest). The projected total C content in soil after 50 years of pasture was 289.5 Mg C ha<sup>-1</sup>, which was 1% above the measured value of 286.7 Mg C ha<sup>-1</sup>. Then, in the second simulation for the pine plantation the  $KO_{sol}$  was set to the minimum and the initial soil values were set according to the output values for the second simulation of the pasture (justified since the pine plantation followed pasture). The projected total C content in soil after 16 years of pine plantation was 286.0 Mg C ha<sup>-1</sup>, which was 11% above the measured value of 257.8 Mg C ha<sup>-1</sup>. However, the second simulations for the lenga forest, pasture, and pine plantation indicated that soluble (labile) compounds would be approximately 85% of the total SOM (Table 5), while the laboratory fractionation indicated that the labile light fraction (> 212 µm) was approximately 55% of total C in the 0-5 cm soil depth, 11% in the 5-10 cm depth, and 7-8% from 10-40 cm depth (Table 4). Nevertheless, the simulations may be correct in the sense that soluble C is a major source of organic matter in Andisols, as previously described. Finally, the C content of the biomass of the lenga forest, pasture, and pine plantation was unchanged between the first and second simulations.

The land-use changes in the study area of the Chilean Patagonia have modified the C content in vegetative biomass and volcanic soil, and the results illustrate the relative importance of volcanic soil to store C in the lenga forest, pine plantation, and pasture of the Chilean Patagonia. Although it is doubtful that all of the pastures will one day be reconverted into native forests or pine plantations in order to affect the total C content stored in vegetation and soil, there is growing interest in the use of silvopastoral systems as a balance between the economic and environmental concerns of land use in the area. A collateral investigation is currently under way to characterize the effects of this type of management.

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## 2.5. References

- Balesdent J., Pétraud J.P., Feller C. 1991. Effets des ultrasons sur la distribution granulométrique des matières organiques des sols. *Science Sol*, 29: 95-106 (in French).
- Borie G., Peirano P., Zunino H., Aguilera S. 2002. N-pool in volcanic ash derived soils in Chile and its changes in deforested sites. *Soil Biology and Biochemistry*, 35: 197-198.
- Christensen B. 2001. Physical fractionation of soil, and structural and functional complexity in organic matter turnover. *European Journal of Soil Science*, 52: 345-353.
- Day P.R. 1965. Particle fractionation and particle size analysis. In: Black C.A. (Ed.), *Methods of Soil Analysis, Physical and Mineralogical Properties*. American Society of Agronomy, Madison, WI, USA, pp. 545-567.
- DGF. 2008. *Interactive Meteorology of Chile*. Faculty of Science and Mathematics, University of Chile. <http://www.atmosfera.cl/>. Accessed in December, 2008.
- DMC. 2009. *Climates of Chile*. Directorate of Meteorology, Government of Chile, Santiago, Chile. <http://meteo Chile.cl/ climas/climas.html>. Accessed in February, 2009.
- Ferrando F. and Muñoz M. 1979. *Perspectives for Development of the Resources of the Region of Aysén*. Publication No. 26, IREN - Agricultural Ministry, Government of Chile, Santiago, Chile, 69 pp. (in Spanish).
- Gayoso J., Guerra J., Alarcon D. 2002. Content of carbon and functions of biomass in native and exotic species. In: Urrutia J. (Ed.), *Measurement of the Capacity to Capture Carbon in Chilean Forests, and its Promotion in the World Market*. Project FONDEF D9811076. University Austral of Chile, Valdivia, Chile, 157 pp. (in Spanish).
- Huygens D., Boeckx P., Van Cleemput O., Oyarzun C., Godoy R. 2005. Aggregate and soil carbon dynamics in south Chilean Andisols. *Biogeosciences*, 2: 159-174.
- IPCC. 2001. *Climate change 2001: the scientific basis*. In: Houghton J.T., Ding Y., Griggs D.J., Noguier M., van der Linden P.J., Dai X., Mashell K., Johnson C.A. (Eds.), *Contribution of Working Group I to Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge UK, 881 pp.
- IUSS. Working Group WRB. 2006. *World reference base for soil resources 2006*, 2<sup>nd</sup> ed. *World Soil Resource Report No.103*. FAO, Rome, 132 pp.

- Krull E.S., Baldock J.A., Skjemstad J.O. 2003. Importance of mechanism and processes of the stabilization of soil organic matter for modeling carbon turnover. *Functional Plant Biology*, 30: 207-222.
- Liski J., Palosuo T., Peltoniemi M., Sievänen R. 2005. Carbon and decomposition model YASSO for forest soils. *Ecological Modeling*, 189: 168-182.
- Loguercio G. and Defosse G. 2001. Equations for aerial biomass, expansion and reduction factors of lenga *Nothofagus pumilio* (Poepp. Et Endl.) Krasser, In: El Sol del Chabut, Argentina. The Book of Summaries of the International Symposium of Measurement and Monitoring of the Capture of Carbon in Forestry Ecosystems, Valdivia, Chile, pp. 1-11. October 18-20, 2001 (in Spanish).
- Monreal C.M., Schulten H.R., Kodama H. 1997. Influence of aggregate architecture and minerals on living habitats and soil organic matter. *Can. J. Soil Sc.*, 77: 379-388.
- Murage E.W., Voroney P., Beyaert R.P. 2007. Turnover of carbon in the free light fraction with and without charcoal as determined using the  $^{13}\text{C}$  natural abundance method. *Geoderma*, 138: 133-143.
- Ortega R. and Rodriguez I. 1994. *Manual of Environmental Management*. Mapfre Foundation. Madrid, Spain, 364 pp. (in Spanish).
- Powlson D.S. 1994. The soil microbial biomass: before, beyond and back. In: Ritz K., Dighton J., Giller K.E. (Eds.), *Beyond the biomass: Compositional and functional analysis of soil microbial communities*. John Wiley & Sons, Chichester, UK, pp. 3-20.
- Quiroz I. and Rojas Y. 2003. Ponderosa pine and Douglas fir: conifers for southern Chile. INFOR, Government of Chile, Valdivia, Chile, 316 pp. (in Spanish).
- Ross D., Tate K., Scott N., Feltham C. 1999. Land use change: effects on soil carbon, nitrogen and phosphorus pools and fluxes in three adjacent ecosystems. *Soil Biology and Biochemistry*, 31: 803-813.
- Sadzawka A., Carrasco M., Grez R., Mora M., Flores H., Neaman A. 2006. *Recommended Analytical Methods for Chilean Soils*. INIA-Agricultural Ministry, Government of Chile, Santiago, Chile, 164 pp. (in Spanish).
- Sandoval M., Stolpe N., Zagal E., Mardones M., Junod J. 2003. Carbon sequestration in agriculture and its importance in global warming. *Theoria*, 12: 65-71 (in Spanish).

- Schelhaas M.J., van Esch P.W., Groen T.A., de Jong B.H.J., Kanninen M., Liski J., Masera O., Mohren G.M.J., Naaburs G.J., Palosuo T., Pedroni L., Vallejo A., Vilen T. 2004. CO2FIX V.3.1. - Description of a Model for Quantifying Carbon Sequestration in Forest Ecosystems and Wood Products, ALTErrA Report 1068. Wageningen, The Netherlands, 122 pp.
- SERPLAC. 2005. Atlas of the Aysén Region, 1:50,000. Planning Ministry, Government of Chile, Santiago, Chile, 43 pp. (in Spanish).
- Shoji S., Nanzyo M., Dahlgren R. 1993. Volcanic Ash Soils, Genesis, Properties and Utilization. Elsevier, New York, NY, USA, 288 pp.
- Silva F. 2004. Final Report on the Ecological Status of the Aysén Region. SAG-Agricultural Ministry, Government of Chile, Santiago, Chile, 143 pp. (in Spanish).
- Silva F., Ahumada M., Cerda J. 1999. Guide for Pasture Condition in the Intermediate Temperate Ecological Region of Aysén. SAG-Agricultural Ministry, Government of Chile, Santiago, Chile, 127 pp. (in Spanish).
- Soil Survey Staff. 2004. Soil Survey Laboratory Methods Manual, Soil Survey Investigations Report No. 42. Natural Resources Conservation Service, USDA, Washington, DC, USA, 700 pp.
- Soil Survey Staff. 2006. Keys to Soil Taxonomy. Natural Resources Conservation Service, USDA, Washington, DC, USA, 332 pp.
- Steele R.G.D. and Torrie J.H. 1980. Principles and Procedures of Statistics, a Biometrical Approach, 2<sup>nd</sup> Ed., McGraw-Hill, New York, NY, USA, 633 pp.
- Stolpe N.B. and Lewis D.T. 1990. Mineralogical implications of high ratios of 1500 kPa water content to percent measured clay for two soils of western Nebraska. *Geoderma*, 47: 1-16.
- Stolpe N.B., Muñoz C., Zagal E., Ovalle C. 2008. Modeling soil C storage in the Espinal agroecosystem of central Chile. *Arid Land Research and Management*, 22: 148-158.
- Veblen T.T., Kitzberger T., Burns B., Rebertus A. 1995. Perturbations and dynamics of regeneration of Andean forests in southern Chile and Argentina. In: Armesto J., Villagran C., Arroyo M. (Eds.), *Ecology of the Native Forests of Chile*, pp. 169-198. University Editorial, University of Chile, Santiago, Chile (in Spanish).
- Wada K. 1985. The distinctive properties of Andosols. In: Stewart B.A. (Ed.), *Advances in Soil Sciences*, Vol. 2. Springer Verlag, New York, NY, USA, pp. 174-229.

- Weber M. 1999. Carbohydrate reserves in primary forest of Nothofagus in Tierra del Fuego. Forstwissenschaftliches Centralblatt, 118: 156-166 (in German).
- Wright A.F. and Bailey J.S. 2001. Organic carbon, total carbon, and total nitrogen determinations in soils of variable calcium carbonate contents using a Leco CN-2000 dry combustion analyzer. Communications Soil Science Plant Analysis, 32: 3243-3258.
- Zagal E. and Córdova C. 2005. Indicators of quality of organic matter in a cultivated Andisol soil. Agricultura Tecnica, 65: 186-197 (in Spanish).
- Zagal E., Rodriguez N., Vidal I., Flores B. 2002. The light fraction of organic matter in a volcanic soil under different agricultural management as an indicator of changes in the labile organic matter. Agricultura Tecnica, 62: 284-296 (in Spanish).



### CAPITULO 3

Dube F., Espinosa M., Stolpe N.B., Zagal E. **Carbon storage in silvopastoral systems with *Pinus ponderosa* and *Trifolium spp.*, in plantations and prairies on an Andisol in Patagonia, Chile.** Enviado el 8 de diciembre de 2010 a Forest Ecology and Management.

**Abstract:** This study was undertaken to quantify and compare C pools in three predominant ecosystems in the Chilean Patagonia, which included: *Pinus ponderosa* - based silvopastoral systems (SPS), 18-year-old pine plantations (PPP) and natural pasture (PST). The C contents of trees and pasture were determined by destructive sampling and dry combustion. Soil moisture (0-20 cm) and superficial air temperature (+5 cm) were measured every two hours over a 24-month period. Soil samples were taken at 0-5, 5-20, 20-40 cm depths in order to determine soil C and N. For PPP and SPS, respectively, 64 and 69% of total tree C were stored aboveground, whereas 36 and 31% were stored below ground. In addition, 32 and 39% of total C in PPP and SPS, respectively, were stored in branches, twigs, cones and needles, which represent a potential C input to soil organic carbon (SOC) via litterfall and pruning. Tree diameter at breast height (DBH) increased 1 and 2 cm per year in PPP and SPS, respectively, and was significantly higher in SPS than PPP. In SPS and PST, SOC content at 0-20 cm depth was 70 and 55% higher, respectively, than that of PPP. At 0-40 cm depth, the total C and N contents decreased significantly in the order SPS>PST>PPP. Soil analyses indicated that the conversion of PPP to SPS resulted in a 30% increase in SOC at 0-40 cm depth, while the establishment of PPP on PST caused a 16% decrease in SOC. Mean annual soil moisture in the pasture alleys of SPS was double that of PST, and superficial air temperature was substantially higher in SPS than in the other treatments during the growing seasons. Although the area available for pasture was nearly 25% lower in SPS as compared with PST, the aboveground yield of forage was similar.

**Keywords:** C pools, legumes, N storage, productivity, silvopasture, SOC, volcanic soils



### 3.1. Introduction

Between 1850 and 2000, the atmospheric concentrations of carbon dioxide (CO<sub>2</sub>) rose from 280 to 369 ppm, and increased further to 388 ppm by August 2010, which is a 5.1% rise over the last 10 years (Tans, 2010). Changes in land-use (e.g. deforestation) and combustion of fossil fuels are two important anthropogenic factors that have contributed to this increase. The influence of land management and practices on the C contents in soils and biomass has been extensively documented worldwide (Ross et al., 2002; Huygens et al., 2005; Dube et al., 2009; Stolpe et al., 2010) and it is well known that soils constitute the largest C reservoir in terrestrial ecosystems. Changes in land-uses can modify soil C contents because of the interaction between the amount and quality of organic residue inputs to soil and their subsequent use by soil macro and microorganisms, which affects the net rate of accumulation of organic matter in soil.

The Climate Change 2007 Synthesis Report (IPCC, 2007) proposed several management strategies in the agricultural sector in order to mitigate CO<sub>2</sub> concentrations in the atmosphere including sustainable management of cropped and grazing lands to increase soil C storage, and the restoration of soils that were previously degraded by intensive agriculture. Agroforestry systems rate high in this regard, because with appropriate management, and the use of perennial grasses and fast growing tree species, they increase soil C sequestration in the short term and therefore are effective CO<sub>2</sub> sinks (Nair et al., 2010). Furthermore, recent studies in temperate regions have shown that agroforestry systems have greater C sequestration potential than monocropping systems, forest plantations, or prairies (Montagnini and Nair, 2004; Sharrow and Ismail, 2004; Gordon and Thevathasan, 2005; Oelbermann et al., 2006; Peichl et al., 2006; Bambrick et al., 2010).

In the remote region of the Chilean Patagonia, ranchers are increasingly challenged to maintain pasture and livestock productivity because of the windy inhospitable climate, steep topography, and eroded volcanic soils. The latter is a consequence of the large forest fires that occurred between 1920 and 1940, when extensive areas were burned to create pastures for cattle (Veblen et al., 1995), leaving the valley slopes exposed to inexorable erosion. Consequently, the *Instituto*

*Forestal* - INFOR (Forestry Research Institute) has recently implemented various incentives for landowners to adopt sustainable agroforestry on their properties, mostly silvopastoral systems and windbreaks, in order to improve the overall productivity of the land, control erosion processes, and increase C sequestration capacity.

The non-sustainable use of volcanic soils can notably diminish organic matter (O.M.) contents (Zagal et al., 2009). Intensive use of volcanic soils in central Chile with decreased additions of crop residues to soil, has lowered the cycling of plant nutrients and increased erosion (Pizarro et al., 2003), resulting over time in substantial decreases in soil O.M. and C content, and soil fertility. Although volcanic soils (Andisols) occupy less than 1% of the world's surface, they are common to central and southern Chile. Volcanic soils, in general, have unique properties, such as low bulk density, the presence of non-crystalline clays with high specific area, high O.M. content and phosphate retention, and elevated water content at 1500 kPa tension (Shoji et al., 1993).

In a study of the C sequestration potential in *Pinus radiata* plantations in southern Chile, Espinosa et al. (2005) reported that Chilean plantations could absorb 1.3% of the global total C (11.8 Gt) captured by plantations in temperate regions between 2005 and 2060. The rotation age was the variable that was best correlated with the quantity of captured C, followed by the mean annual increment (MAI) and the wood C content.

The natural fertility of soil can be a limiting factor in agriculture or forestry plantations. Additionally, mixed stands can sequester C more efficiently than mono specific plantations (Kimmins, 2004; Nair et al., 2010). In this sense, agroforestry systems represent a good trade-off since they are frequently designed and implemented to improve soil fertility, and often combine several interdependent species. The tree component is often managed with pruning, thinning, and harvesting, leaving a significant quantity of organic debris on the site that will eventually be returned to the soil through natural processes of biological cycling (Montagnini and Nair, 2004). Nevertheless, there is a limited number of studies on Chilean agroforestry systems and a general lack of scientific research on C pools in silvopastoral systems located in the temperate areas of the Southern Hemisphere, and especially those established on degraded volcanic soils. Since introduced species, such as *Pinus ponderosa* and *Pseudotsuga menziesii*

already occupy more than 30,000 ha in the Chilean Patagonia, and natural prairies cover more than one million ha (A. Sotomayor, pers. comm., 2008), their respective C pools must be quantified as soon as possible.

The current study is probably the first to evaluate C sequestration potentials (aboveground and in soil) in ecosystems on degraded Andisols of the Chilean Patagonia. The objectives of this work were to (1) measure the size of the C pools that are present in the vegetation biomass and volcanic soils in a natural pasture, a managed *Pinus ponderosa* plantation, and a pine-based silvopastoral system, considering variations of soil moisture and superficial air temperature, and (2) assess how leguminous pasture (*Trifolium* spp.) affect the content of soil C and tree growth.

## 3.2. Material and methods

### 3.2.1. Site description

The study took place in the Intermediate Agro-ecological Zone of the Aysén Region of Patagonia, one of the most climatically extreme and southern zones of the Chilean territory. The forest is classified as Humid Boreal and covers 4.7 million ha, or 44% of the region. The main forest types are evergreen (2.2 million ha), *Nothofagus pumilio* (1.4 million ha), *Nothofagus dombeyi* (0.6 million ha), and *Pilgerodendron uviferum* (0.5 million ha) (Dube et al., 2009). Specifically, the site was located in the San Gabriel Agroforestry Unit within the Mano Negra Sector. It was established in 2002 by the Agricultural and Cattle Development Institute (INDAP) and Forestry Research Institute (INFOR) 30 km north of the city of Coyhaique, on a westerly exposed slope at 730 m altitude, Lat S 45°25' and Long W 72°00'. The study area included several adjacent land-management practices (200 m apart), hereafter referred to as treatments: 1) natural pasture with traditional cattle grazing (PST), 2) 18-year-old plantations of thinned and pruned *Pinus ponderosa* (PPP), 3) silvopastoral systems of *P. ponderosa*-arranged in strips (SPS), with pasture alleys of 21 m width between the tree strips. Originally, the entire area was covered with native forest, mostly *Nothofagus pumilio* (lenga), which was burned to open pasture for cattle grazing, as previously described.

In 1991, *P. ponderosa* plantations were established over the pasture, with a density of 2000 trees ha<sup>-1</sup> and a spacing of 2 m x 2.5 m. In 2003, the density had decreased to 1514 trees/ha, the mean tree height (HT) was 6.7 m and diameter at breast height (DBH) was 11.4 cm. In 2003, part of the plantation (5 ha) was thinned to 800 trees ha<sup>-1</sup> while another section was thinned to 400 trees ha<sup>-1</sup> and converted into a silvopastoral system arranged in strips (5 ha).

The pasture in the open alleys has a stocking density of 0.5 cows ha<sup>-1</sup>, and consists of a mixture of perennial grasses (*Dactylis glomerata*, *Holcus lanatus*, *Poa pratensis*), legumes (*Trifolium pratense*, *T. repens*), and other accompanying species (*Acaena magellanica*, *A. pinnatifida*, *Hypochoeris radicata*, *Taraxacum officinale*). *Trifolium pratense* and *T. repens* can add up to 150 and 250 kg N ha<sup>-1</sup> y<sup>-1</sup>, respectively, through symbiotic fixation of atmospheric N (Brady, 1990; Stevenson and Cole, 1999). However, during the 2004-2005 growing period, PST and SPS were also fertilized with 16 kg N ha<sup>-1</sup>, 30 kg Mg ha<sup>-1</sup> and 79 kg S ha<sup>-1</sup>; and in 2005-2006, with 24 kg N ha<sup>-1</sup>, 17 kg Mg ha<sup>-1</sup> and 77 kg S<sup>-1</sup> (Dube et al., 2009). Since then, no further fertilization took place.

The annual precipitation varies from 1000 mm to 1500 mm. However, only 15% of the precipitation occurs between December and February, coinciding with the warmest and windiest period. Mean temperatures fluctuate between 12°C and 14°C in summer and 2°C and 3°C in winter (Silva, 2004). During the summers, strong westerly winds occur, provoking seasonal water deficits and wind erosion, which may reduce the organic matter content of the surface soil. In general, the soils are deep, well drained and moderately acidic, formed from relatively unaltered volcanic ash deposits, and they present naturally high fertility levels with low Al saturation (Silva et al., 1999). Specifically, the soil pH varies from 5.9-6.5 among treatments; the soil texture in PPP and SPS is a fine sandy loam, while in PST it is a loamy fine sandy loam. The mineral horizons have Andic soil properties that include low apparent density values (< 0.9 g cm<sup>-3</sup>), high P fixation values (65-89%), and high water content at 1500 kPa tension. The soil was classified as medial, amorphic, mesic Typic Hapludands (Stolpe et al., 2010).

### *3.2.2. Experimental design*

Treatments were established in October 2007 in a completely randomized design with three replicates (or sampling plots). The plots had a uniform westerly exposure and a slope of 10-15%. Each plot measured 15 m x 27 m and was located at least 5 m from the border of the treatment (i.e. land-use management practices). In SPS, each plot included three strip rows of pines (6 m wide) and half strip (10.5 m) of pasture on either side. In both PPP and PST, the plots had only pine and pasture, respectively. All plots were fenced with barbed wire and chicken wire to exclude animals.

### *3.2.3. Ponderosa pine biomass and carbon content*

An inventory of PPP and SPS was performed to determine the tree characteristics and growth under the distinct types of management. All of the trees within the replicates were evaluated, measuring the DBH and HT variables using a diameter tape and clinometer. The measured variables were then used to calculate aboveground and subterranean biomass in PPP only, using the existing biomass functions for the climatic zone, that were specifically developed for ponderosa pine plantations (Gayoso et al., 2002). To validate the results obtained in PPP and obtain the aboveground and subterranean biomass of the trees in SPS, destructive sampling was performed to determine the weight of different tree components. Three trees in PPP and SPS, respectively, were selected for the mean basal area using the average tree method. This technique assumes that the tree with the average size of the stand also contains its average biomass (Teller, 1988).

Once the trees were felled, the fresh biomass of trunks, branches, twigs, needles and cones were measured using a 45-kg dynamometer (Salter Brecknell Electro Samson Scale, Raco Industries, Cincinnati, OH, USA). Thick root (> 5 mm diameter) biomass was estimated using subterranean biomass functions for the climatic zone, which relate the root biomass to the DBH (Gayoso et al., 2002). The annual biomass production of fine root biomass (<5 mm diameter) was estimated as a percentage of the annual litterfall (Abohassan, 2004).

Three sub-samples from each tree component were then taken, including cross sectional disks of tree stems obtained at the initial crown height, breast and stump heights, in order to determine the moisture and C content. The sub-samples were weighed, dried at 105°C for 48 hours and weighed again, and the mean dry weight of the distinct sub-samples was then extrapolated to the entire stand (Peichl et al. 2006). The C content of all the sub-samples (trunks, branches, roots, leaves) was determined by grinding the samples using a Cyclotec 1093 Sample Mill (Tecator, Sweden), oven drying at 65°C, and analysis for total C using a Fisons EA1108 CHNS-O Elemental Analyzer (Fisons Instrument, CA, USA) following the dynamic flash combustion technique (Fisons Instrument, 1990).

#### *3.2.4. Pasture biomass and carbon content*

In order to determine the annual aboveground net primary productivity (ANPP) of pasture, and simulate animal grazing, the forage material was harvested three times during each growing season over a two-year period. Vegetation was cut within nine randomly selected 0.5 m<sup>2</sup> quadrats that were permanently established in the plots of every treatment, placed in plastic bags and taken to the laboratory where it was dried at 65°C for 5 days, and weighed again to obtain the dry biomass. Since it was not possible to measure on site the below ground net primary productivity (BNPP), it was estimated with a known algorithm (Gibson, 2009). The total C content of the forage was determined using the dry combustion technique described above.

#### *3.2.5. Soil carbon and nitrogen*

In March 2009, soil samples were taken at 0-5, 5-20, 20-40 cm depths in order to determine C and N contents in each treatment (Dube et al., 2009). In each SPS plot, the samples were taken following two transects perpendicular to the pine strip, one in each direction at 2.5 m intervals from the border of the strip. In PPP and PST, triplicate samples were randomly collected at the same depths in each plot. All the samples were taken using a split-core soil sampler, stored in plastic bags, and air-dried in the laboratory before being sieved (250-µm mesh). The total organic and inorganic C and total N were determined using the flash combustion method,

whereby soil samples were placed in a muffle furnace at 500°C for 12 hours for complete combustion of organic compounds. Given the genesis of volcanic soils, the inorganic carbon content was almost non-existent and it was concluded that total C was equivalent to organic C.

### 3.2.6. Soil temperature and moisture

The soil moisture (0-20 cm depth) and superficial air temperature (+5 cm) were measured every two hours over a 24-month period using EC-20 soil moisture sensors and ECT temperature sensors, respectively, that were connected to EM-5B Data Loggers (Decagon Devices Inc., Pullman, WA, USA). Within PPP and PST, loggers were randomly installed in different locations, whereas in the SPS, a logger was placed in the tree strip and another one at 2 m from the strip.

### 3.2.7. Statistical analyses

All treatments were analyzed with the General Lineal Model procedure of SAS v.9.0 (SAS Institute Inc., 2003) for completely randomized designs to test the treatments effects on the parameters that were described in the previous sections. All the data were examined for homogeneity of variance and normality. The analysis of variance was performed with ANOVA, and Student's *t*-test for independent populations was used to check for significant differences between the treatment means (comparison by pairs) ( $P < 0.05$ ).

Within SPS, the sampling distances from the tree strips were treated as sub-treatments and the values obtained for each parameter at a specific distance were compared to each other. Statistical difference ( $P < 0.05$ ) between the means of sub-treatments was tested using Tukey's HSD multiple comparison test. Finally, measurements of soil moisture and superficial air temperature were adjusted to the monthly averages using the least minimum square procedure, and the monthly averages of soil moisture and superficial air temperatures were compared among treatments using a two-way ANOVA with Tukey's HSD test.

### 3.3. Results

#### 3.3.1. Distribution of carbon contents in ponderosa pines

Carbon contents of the distinct compartments of above and belowground biomass from the 18 year-old ponderosa pines in PPP and SPS are shown in Table 1. The C concentrations (%) of individual and total aboveground components were similar in both treatments and slightly higher in the belowground biomass of the plantation, but there was a significant difference only in the pine needle component, with that being higher in SPS.

Table 3.1. Carbon content (% , kg tree<sup>-1</sup> and kg ha<sup>-1</sup>) of different tree components from two adjacent temperate agroecosystems in the Chilean Patagonia (mean ± standard deviation).

| Tree components   | C %                      |                          | kg tree <sup>-1</sup>    |                          | kg ha <sup>-1</sup>        |                            |
|-------------------|--------------------------|--------------------------|--------------------------|--------------------------|----------------------------|----------------------------|
|                   | PPP                      | SPS                      | PPP                      | SPS                      | PPP                        | SPS                        |
| Trunk             | 41.4 ± 0.10 <sup>a</sup> | 41.7 ± 0.10 <sup>a</sup> | 22.0 ± 4.8 <sup>a</sup>  | 23.0 ± 4.8 <sup>a</sup>  | 1.650 ± 0.220 <sup>a</sup> | 1.616 ± 0.220 <sup>a</sup> |
| Branches          | 42.8 ± 0.10 <sup>a</sup> | 43.4 ± 0.10 <sup>a</sup> | 21.0 ± 5.0 <sup>a</sup>  | 22.0 ± 5.0 <sup>a</sup>  | 1.500 ± 0.220 <sup>a</sup> | 1.554 ± 0.220 <sup>a</sup> |
| Twigs             | 43.7 ± 0.10 <sup>a</sup> | 43.2 ± 0.10 <sup>a</sup> | 1.5 ± 1.2 <sup>a</sup>   | 1.5 ± 1.2 <sup>a</sup>   | 0.02 ± 0.02 <sup>a</sup>   | 0.03 ± 0.02 <sup>a</sup>   |
| Cones             | 47.2 ± 0.10 <sup>a</sup> | 47.2 ± 0.10 <sup>a</sup> | 0.5 ± 0.2 <sup>a</sup>   | 0.5 ± 0.2 <sup>a</sup>   | 0.006 ± 0.003 <sup>a</sup> | 0.01 ± 0.003 <sup>a</sup>  |
| Needles           | 47.5 ± 0.10 <sup>a</sup> | 47.5 ± 0.10 <sup>b</sup> | 0.2 ± 0.2 <sup>a</sup>   | 0.2 ± 0.2 <sup>b</sup>   | 0.001 ± 0.001 <sup>a</sup> | 0.001 ± 0.001 <sup>b</sup> |
| Roots             | 43.2 ± 0.10 <sup>b</sup> | 42.5 ± 0.10 <sup>b</sup> | 1.1 ± 2.4 <sup>b</sup>   | 2.4 ± 4.8 <sup>b</sup>   | 0.705 ± 0.022 <sup>b</sup> | 1.612 ± 0.125 <sup>b</sup> |
| Total aboveground | 43.7 ± 0.10 <sup>a</sup> | 43.8 ± 0.10 <sup>a</sup> | 24.1 ± 11.0 <sup>a</sup> | 23.1 ± 11.0 <sup>a</sup> | 1.683 ± 0.220 <sup>a</sup> | 1.612 ± 0.220 <sup>a</sup> |
| Total belowground | 43.2 ± 0.10 <sup>b</sup> | 42.5 ± 0.10 <sup>b</sup> | 1.1 ± 2.4 <sup>b</sup>   | 2.4 ± 4.8 <sup>b</sup>   | 0.705 ± 0.022 <sup>b</sup> | 1.612 ± 0.125 <sup>b</sup> |
| Total tree        | 43.6 ± 0.10 <sup>a</sup> | 43.6 ± 0.10 <sup>b</sup> | 25.2 ± 13.4 <sup>a</sup> | 25.5 ± 15.8 <sup>b</sup> | 2.388 ± 0.242 <sup>a</sup> | 3.224 ± 0.345 <sup>b</sup> |

PPP: *Pinus ponderosa* plantation; SPS: pine-based silvopastoral system. Values with the same lower case letter within a tree component and between treatments (PPP and SPS) for every group (% , kg tree<sup>-1</sup> or kg ha<sup>-1</sup>) are not significantly different (Student's t test, \*\**P* < 0.01). Values with the same higher case letter within a column and among tree components or totals (aboveground and belowground) are not significantly different (Tukey's HSD test, \**P* < 0.05).

The C concentrations within PPP decreased in the order Cones ≥ Needles ≥ Twigs > Roots > Branches ≥ Trunks, and Total aboveground ≥ Total belowground, but Cones, Needles and Twigs were not significantly different, nor were Branches and Trunks. There was a significant difference only between Cones and Branches, and Trunk and Roots, and Needles were also statistically different from Branches and Trunks. Within SPS, the relative C concentrations in the individual tree components was Needles > Cones > Branches ≥ Twigs ≥ Roots ≥ Trunks,



and Total aboveground  $\geq$  Total belowground, but Branches, Twigs, Roots and Trunks were not significantly different. Needles were similar to cones only, but statistically different from all other components. In both PPP and SPS treatments, Trunks was the component with the lowest C concentration, and Total aboveground and belowground C concentrations were not significantly different.

On a tree basis, the C content was higher in every component of SPS as compared with PPP (Table 1 and Figure 1a), but significant differences were found only for Twigs (61% higher), Needles (76%) and Cones (100%), Total aboveground (38%) and Total tree (28%). In contrast to C concentrations, C contents decreased in the order Roots (>5mm diameter) > Trunks > Branches > Twigs > Needles > Cones, which shows how the size of the component affects its C content. For both treatments however, high standard deviations in the average tree C contents (23-26%) indicate a certain degree of variability among individual trees that must be accounted for during the 35-year rotation.

For PPP and SPS, respectively, 64 and 69% of total tree C were stored as aboveground biomass, whereas 36 and 31% were stored within the root system. Pine roots stored 11% more C than the tree stems in PPP and only 2% more C in SPS. Together, branches, twigs, needles and cones stored the same amount of C as the trunks in PPP, while in SPS these components stored 30% more C than in the trunks. Additionally, 32 and 39% of total C in PPP and SPS, respectively, were stored in branches, twigs, cones and needles, which together represent a potential C input to the soil C pool via litterfall and branch pruning (Table 1; Figure 1b).

When looking at the data on a hectare basis, there was no significant difference between the PPP and SPS treatments with respect to the amount of C stored in branches, twigs, needles and cones, in spite of the two-fold tree density in PPP compared to SPS. However, there was significantly more C stored in roots (82% larger) and trunks (67%) in the pine plantation. It is interesting to note that the difference between PPP and SPS was reduced to only 44% when comparing Total aboveground C contents, and 56% when comparing Total tree C contents, implying more efficient C storage per tree in SPS.

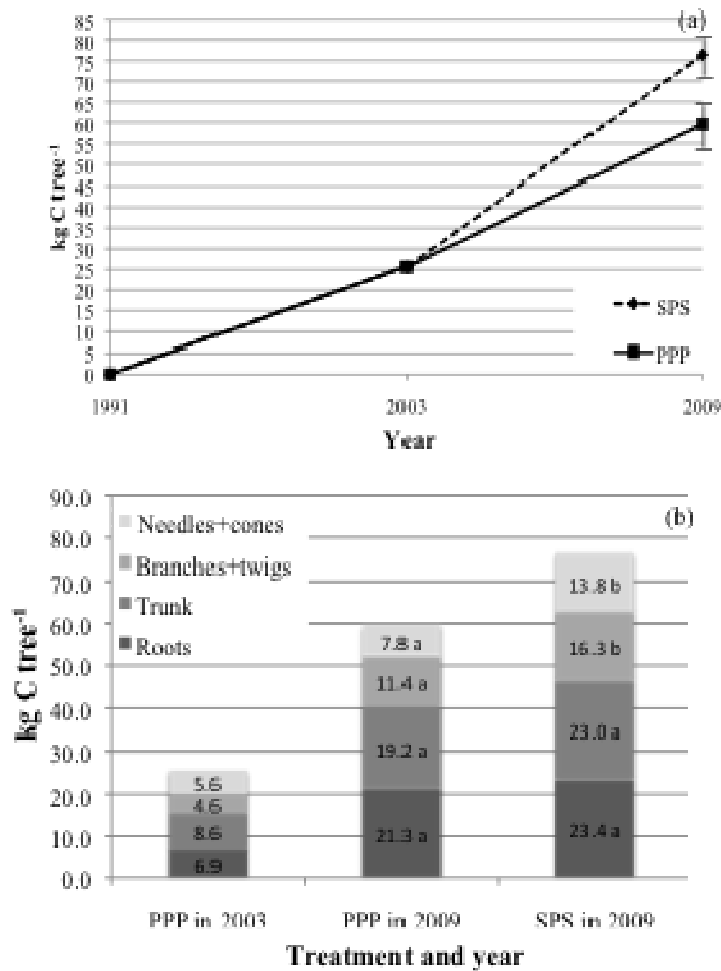


Figure 3.1. Change in C sequestration rate (a) as influenced by change in tree density, since the establishment of the pine plantation in 1991, followed by thinning in 2003 in the pine plantation (PPP) and the silvopastoral system (SPS). Vertical bars denote standard error of the mean. Distribution (b) of C content (kg) per tree compartment before thinning and in PPP and SPS six years after thinning. Values with the same lower case letter within a tree component and among treatments in 2009 are not significantly different (Student's t test, \*\* $P < 0.01$ ).

### 3.3.2. Distribution of tree biomass for energy generation

The trunks represent 32 and 30% of total tree biomass (and C content) present in PPP and SPS, respectively, or approximately one third of the tree (Table 1 and Figure 1b). In PPP, the crowns (branches, twigs, needles, cones) represent 50% of aboveground biomass and 32% of

total tree biomass, while in SPS they represent 57 and 39%, respectively. Finally, the contribution of needles to aboveground biomass is 20% in PPP and 25% in SPS. A total of 74% of aboveground and 47% of total biomass are available as a source of renewable energy (trunks and branches) in PPP, whereas 67 and 47%, respectively, are available in SPS. If twigs are included in the available biomass, the previous values climb to 80 and 51% of aboveground and total biomass, respectively, in PPP, and to 74 and 51% in SPS.

### 3.3.3. Growth and productivity

Results from annual forest inventories during the experimental period are presented in Table 2 in order to compare tree growth under the different management scenarios. Overall, the DBH increased approximately 1 and 2 cm per year in PPP and SPS, respectively, and DBH was significantly higher ( $P < 0.05$ ) each year in SPS than PPP (5, 6 and 8%, respectively, in 2007, 2008 and 2009). When comparing HT, no significant difference was observed on a yearly basis, although trees in SPS were on average 20 cm taller. There was a greater rate of gain of BA in SPS because of the larger increase in DBH, as compared with PPP. When comparing BA between treatments however, a significant difference ( $P < 0.05$ ) was observed every year, where BA was 80, 77 and 69% higher in PPP in 2007, 2008 and 2009, respectively.

Table 3.2. Increment of diameter at breast height (DBH), height (HT) and basal area (BA) over a two-year measurement period in two temperate agroecosystems in the Chilean Patagonia (mean  $\pm$  standard deviation).

| Year | Mean DBH (cm)               |                             | Mean HT (cm)               |                            | Mean BA (m <sup>2</sup> /ha) |                             |
|------|-----------------------------|-----------------------------|----------------------------|----------------------------|------------------------------|-----------------------------|
|      | PPP                         | SPS                         | PPP                        | SPS                        | PPP                          | SPS                         |
| 2007 | 20.7 $\pm$ 1.2 <sub>a</sub> | 21.7 $\pm$ 1.9 <sub>b</sub> | 6.8 $\pm$ 0.1 <sub>a</sub> | 6.9 $\pm$ 0.3 <sub>a</sub> | 27.4 $\pm$ 3.0 <sub>a</sub>  | 15.2 $\pm$ 2.7 <sub>b</sub> |
| 2008 | 21.9 $\pm$ 1.1 <sub>a</sub> | 23.2 $\pm$ 1.9 <sub>b</sub> | 7.4 $\pm$ 0.1 <sub>a</sub> | 7.6 $\pm$ 0.1 <sub>a</sub> | 30.9 $\pm$ 3.7 <sub>a</sub>  | 17.5 $\pm$ 2.8 <sub>b</sub> |
| 2009 | 23.2 $\pm$ 1.1 <sub>a</sub> | 25.2 $\pm$ 1.9 <sub>b</sub> | 8.2 $\pm$ 0.1 <sub>a</sub> | 8.3 $\pm$ 0.1 <sub>a</sub> | 34.5 $\pm$ 3.9 <sub>a</sub>  | 20.4 $\pm$ 3.1 <sub>b</sub> |

PPP: 18-year old *Pinus ponderosa* plantation; SPS: pine-based silvopastoral system arranged in strip. Values with the same lower case letter within a year and among treatments for every group (DBH, HT or BA) are not significantly different (Student's t test, \* $P < 0.05$ ).

When looking at Figure 1a, it is clear that C sequestration rates increased with a decrease of tree density in the original pine plantation, and that the sequestration rate was significantly higher after converting the plantation in 2003 to the silvopastoral system arranged in strips. Figure 1a depicts the effect of thinning the original ponderosa pine plantation in 2003 on biomass partitioning and C allocation to the distinct tree components in PPP and SPS, six years later. In the year 2009 in PPP (Figure 1b), the C stored in trunks, branches+twigs, needles+cones, and roots increased 2.2, 2.5, 1.4 and 3 times, respectively, than in 2003, while in SPS it increased 2.7, 3.5, 2.5 and 3.4 times, respectively. In other words, thinning done at different densities in 2003 resulted in more C sequestered in SPS than PPP in trunks (20% more), branches+twigs (43% more), needles+cones (77% more), and in roots of individual trees (10% more) in 2009.

#### *3.3.4. Prairie productivity and distribution of C contents*

Above and below ground net primary productivity (ANPP and BNPP) of grass growing in the ecosystems and the respective C contents are presented in Table 3. Both ANPP and BNPP and corresponding C contents were higher in 2009 than 2008. However, on a quadrant basis ( $\text{g m}^{-2} \text{y}^{-1}$ ), ANPP in SPS was 11 and 28% higher than PST in 2008 and 2009, respectively (F.Dube, unpublished data).

As depicted in Figures 3a and 3b, the soil moisture and air temperature between November 2007 and April 2008 were always highest in the grass component of SPS, the difference among treatments being significant for every month (F.Dube, unpublished data). Although the area available for pasture was 22% lower in SPS, the overall ANPP was only 14% lower in 2008 and practically the same in 2009.

Table 3.3. Above and belowground net primary productivity (ANPP and BNPP) ( $\text{kg ha}^{-1} \text{y}^{-1}$ ) and carbon content of pasture in three temperate agroecosystems over a two-year period. Measurements were taken in December 2007 and 2008, and February and May 2008 and 2009 (mean  $\pm$  standard deviation).

| Year | Above-ground dry biomass yield (ANPP)<br>( $\text{kg ha}^{-1} \text{y}^{-1}$ ) |                  |                  | Above-ground C content<br>( $\text{kg ha}^{-1} \text{y}^{-1}$ ) |                   |                |
|------|--|------------------|------------------|---|-------------------|----------------|
|      | PST  | SPS              | PPP              | PST   | SPS               | PPP            |
| 2008 | 2454 $\pm$ 1027 a  | 2548 $\pm$ 828 a | n/a              | 1312 $\pm$ 456 a  | 1159 $\pm$ 377 a  | n/a            |
| 2009 | 3391 $\pm$ 932 a   | 3368 $\pm$ 594 a | 732 $\pm$ 18 b   | 1506 $\pm$ 414 a  | 1533 $\pm$ 270 a  | 323 $\pm$ 8 b  |
| Mean | 3173 $\pm$ 309 a   | 2958 $\pm$ 580 a | 732 $\pm$ 18 b   | 1409 $\pm$ 410 a  | 1346 $\pm$ 300 a  | 323 $\pm$ 8 b  |
| Year | Below-ground dry biomass yield (BNPP)<br>( $\text{kg ha}^{-1} \text{y}^{-1}$ ) |                  |                  | Below-ground C content<br>( $\text{kg ha}^{-1} \text{y}^{-1}$ ) |                   |                |
|      | PST  | SPS              | PPP              | PST   | SPS               | PPP            |
| 2008 | 6213 $\pm$ 392 a   | 4697 $\pm$ 291 b | n/a              | 2651 $\pm$ 173 a  | 2099 $\pm$ 150 b  | n/a            |
| 2009 | 6329 $\pm$ 422 a   | 5074 $\pm$ 334 b | 1875 $\pm$ 112 c | 2791 $\pm$ 186 a†   | 2268 $\pm$ 149 b† | 851 $\pm$ 51 c |
| Mean | 6170 $\pm$ 428 a   | 4885 $\pm$ 360 b | 1875 $\pm$ 112 c | 2721 $\pm$ 189 a  | 2184 $\pm$ 161 b  | 851 $\pm$ 51 c |

PST: natural pasture; SPS: pine-based silvopastoral system; PPP: 18-year old *Pinus ponderosa* plantation. Values with the same lower case letter within a same year and among treatments are not significantly different (n/a: not applicable; Student's *t* test, \* $P < 0.01$ ; †\*\* $P < 0.05$ ).

### 3.3.5. Soil carbon and nitrogen

The mean soil C and N concentrations and soil organic carbon (SOC) contents at different depths of the ecosystems are presented in Table 4. Carbon concentrations were significantly different at 0-5, 5-20 and 20-40 cm depths within the three treatments, except at 5-20 and 20-40 cm depths in PPP. In PST and SPS, C% at 0-20 cm depth was more than double the concentration at 20-40 cm depth, while in PPP it was only 20% higher.

Among treatments, the C concentration was similar at 0-5 cm depth in PST and SPS, but almost twice as high as PPP, that difference being significant. At 5-20 and 0-20 cm depths, the concentration decreased in the order SPS > PST > PPP and the differences were also significant. In SPS and PST, respectively, the C concentration at 0-20 cm depth was 70 and 55% larger than PPP. At 0-40 cm depth, the C content also decreased in the order SPS > PST > PPP, and the differences were all significant.

Table 3.4. Soil carbon and nitrogen concentrations (%) at distinct sampling locations in the pasture and pine plantation or averaged across all distances in the silvopastoral system, C/N ratio, and mean soil organic carbon (SOC) contents (Mg ha<sup>-1</sup>) at 0-5, 5-20, 20-40, 0-20 and 0-40 cm depths under three temperate agroecosystems. Measurements were taken in the Chilean Patagonia in March 2009 (mean ± standard deviation).

| Treatment | Soil depth (cm) | C (%)            | N (%)           | C/N              | SOC (Mg ha <sup>-1</sup> ) |
|-----------|-----------------|------------------|-----------------|------------------|----------------------------|
| PST       | 0-5             | 12.77 ± 0.54 A a | 0.86 ± 0.15 A a | 12.53 ± 2.88 A a | 48.49 ± 5.34 A a           |
|           | 5-20            | 5.62 ± 0.08 A b  | 0.38 ± 0.00 A b | 14.68 ± 0.19 A b | 75.85 ± 1.05 A b           |
|           | 20-40           | 2.92 ± 0.50 A c  | 0.19 ± 0.01 A c | 15.01 ± 5.69 A c | 52.57 ± 9.08 A a           |
|           | 0-20            | 6.92 ± 0.15 A    | 0.50 ± 0.04 A   | 13.79 ± 1.70 A   | 124.53 ± 2.74 A            |
|           | 0-40            | 4.92 ± 0.28 A    | 0.33 ± 0.01 A   | 14.13 ± 0.96 A   | 177.10 ± 10.09 A           |
| PPP       | 0-5             | 6.25 ± 0.08 B a  | 0.49 ± 0.01 B a | 12.67 ± 0.19 A a | 28.12 ± 0.37 B a           |
|           | 5-20            | 3.89 ± 0.05 B b  | 0.28 ± 0.02 B b | 13.67 ± 1.15 B b | 52.49 ± 0.79 B b           |
|           | 20-40           | 3.81 ± 0.43 A b  | 0.26 ± 0.05 A b | 14.51 ± 4.29 A b | 68.63 ± 7.79 A c           |
|           | 0-20            | 4.48 ± 0.03 B    | 0.34 ± 0.02 B   | 13.20 ± 0.78 A   | 82.61 ± 0.55 B             |
|           | 0-40            | 4.13 ± 0.20 B    | 0.30 ± 0.02 B   | 13.83 ± 1.80 A   | 148.23 ± 7.34 B            |
| SPS       | 0-5             | 12.68 ± 0.35 A a | 0.86 ± 0.09 A a | 12.43 ± 0.49 A a | 48.04 ± 1.57 A a           |
|           | 5-20            | 6.51 ± 0.15 C b  | 0.49 ± 0.01 C b | 13.29 ± 0.29 B a | 87.94 ± 1.98 C b           |
|           | 20-40           | 3.20 ± 0.21 A c  | 0.21 ± 0.00 A c | 14.89 ± 1.07 A b | 57.54 ± 5.73 A c           |
|           | 0-20            | 7.57 ± 0.10 C    | 0.58 ± 0.01 C   | 13.03 ± 0.36 A   | 136.23 ± 1.86 C            |
|           | 0-40            | 5.38 ± 0.13 C    | 0.40 ± 0.01 C   | 13.33 ± 0.46 A   | 193.76 ± 4.51 A            |

PST: managed natural pasture; PPP: *Pinus ponderosa* exotic plantation; SPS: pine-based silvopastoral system arranged in strip. Values with the same higher case letter within a column and within a same depth among treatments are not significantly different (Student's *t* test, \*\**P* < 0.01). Values with the same lower case letter within a column and within a same treatment are not significantly different (Tukey's HSD test, \**P* < 0.05).

The C contents (%) at the different depths in PST and in the pasture portion of SPS are indicated in Figure 2a, whereas the C and N contents in PPP and within the tree component of SPS are shown in Figure 2b. As can be seen in Figure 2a, C contents at 5-20 and 0-20 cm depths were similar on either sides of the tree strip in SPS, but significantly higher than PST. However, at 0-40 cm depth in SPS, C% was significantly larger than PST only on the east side of the tree strip. On the other hand, the C content in SPS tree strips was 6.33% at 0-20 cm depth (Figure 2b), whereas in PPP it was 4.48%, a significantly lower value. At all depths except 20-40 cm, both C and N concentrations were significantly higher in the tree strip of SPS as compared with PPP. In SPS, N% at 0-20 cm depth was 71 and 16% larger than PPP

and PST, respectively. At 0-40 cm depth, N% decreased in the order SPS > PST > PPP and the differences were significant among all ecosystems. The C/N ratios increased with soil depths but only in PST they were significantly different at all depths. Among treatments at every soil depth, the C/N ratios were not significantly different, with the exception of PPP and SPS where the ratios were statistically lower than PST at 5-20 cm depth.

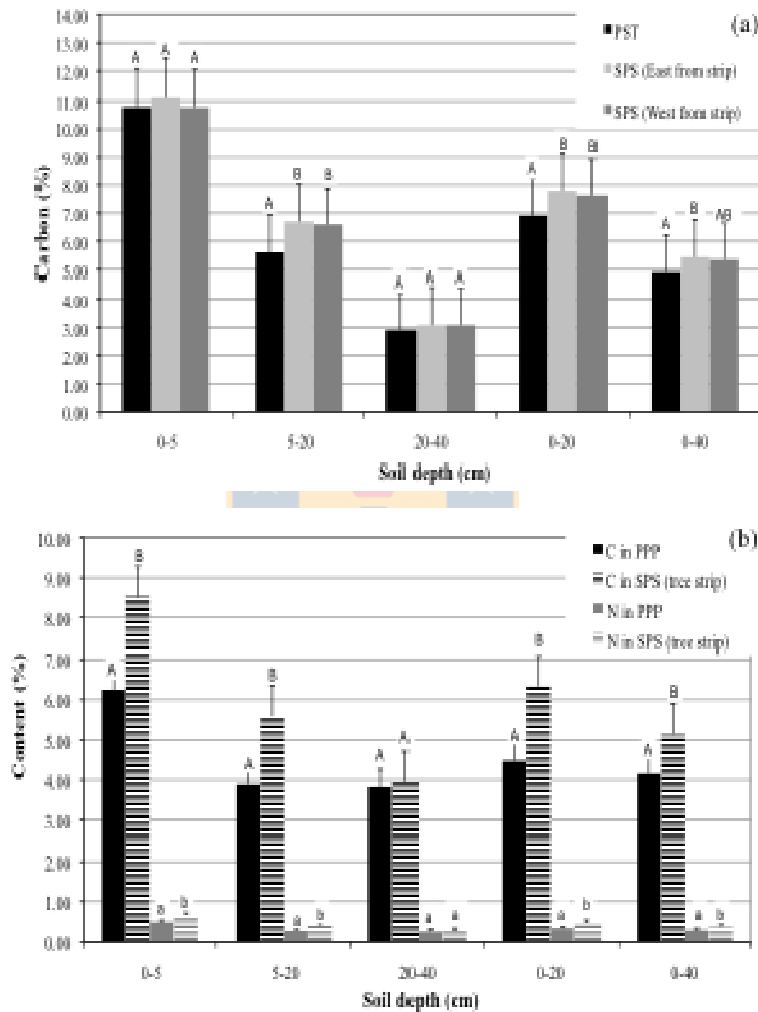


Figure 3.2. Soil C contents (%) at 0-5, 5-20, 20-40, 0-20 and 0-40 cm depths (a) in PST and at two locations in SPS (east and west of tree strip); and (b) soil C and N contents (%) at the same depths in PPP and within the tree strip of SPS. PST: managed natural pasture; PPP: *Pinus ponderosa* exotic plantation; SPS: pine-based silvopastoral system. Vertical bars denote standard error of the mean. Values with the same higher or lower case letter within a same depth among treatments are not significantly different (Student's t test,  $**P < 0.01$ ).

There was a significant difference of SOC contents ( $\text{Mg ha}^{-1}$ ) at 0-5, 5-20 and 20-40 cm depths in both PPP and SPS (Table 4). Among treatments, SOC was similar in the upper 5 cm in PST and SPS, but almost double and significantly different than in PPP. At the depths of 5-20 and 0-20 cm, SOC decreased significantly in the order  $\text{SPS} > \text{PST} > \text{PPP}$ , showing that land-use practices substantially affected the stocks of SOC. At 0-20 cm depth, the conversion of PPP to SPS resulted in a 69% increase in SOC, but the conversion of PST to PPP lowered SOC by 35%. At 0-40 cm depth, SOC generally decreased in the order  $\text{SPS} \geq \text{PST} > \text{PPP}$  but a statistical difference was only observed in PPP from the other treatments. Across this depth, there was a 30% increase in SOC that resulted from the conversion of PPP to SPS but a 16% decrease from PST to PPP. It is worth mentioning that SOC at 0-20 cm depth represents 70, 54 and 70% of total SOC in PST, PPP and SPS, respectively.

### *3.3.6. Soil moisture and superficial air temperature*

As illustrated in Figure 3a, the volumetric soil moisture at 2 m from the tree strip in SPS was significantly greater ( $P < 0.05$ ) than the other soils, at any time during the whole study period, despite the drought that occurred in the summer of 2008. For instance, mean soil moisture was 12.1% in 2008 and 15.1% in 2009; whereas in 2008 in PST, PPP and within the tree strip of SPS, it was 7.5, 7.3 and 6.1%, respectively, and in 2009 it was 9.9, 8.8 and 7.4%, respectively (F.Dube, unpublished data). Soil moisture was also larger in PPP than in the SPS tree strip, except during the drought period. The air temperature (+5 cm) (Figure 3b) was substantially higher at 2 m from the strip than other treatments during each growing season (Nov. 2007 to Apr. 2008, and Nov. 2008 to Apr. 2009) and at spring (Sep. to Nov. 2008 and 2009). The differences among treatments were minimal in June and July 2008. From May to Sep. 2009, it was significantly lower in PST (F.Dube, unpublished data).



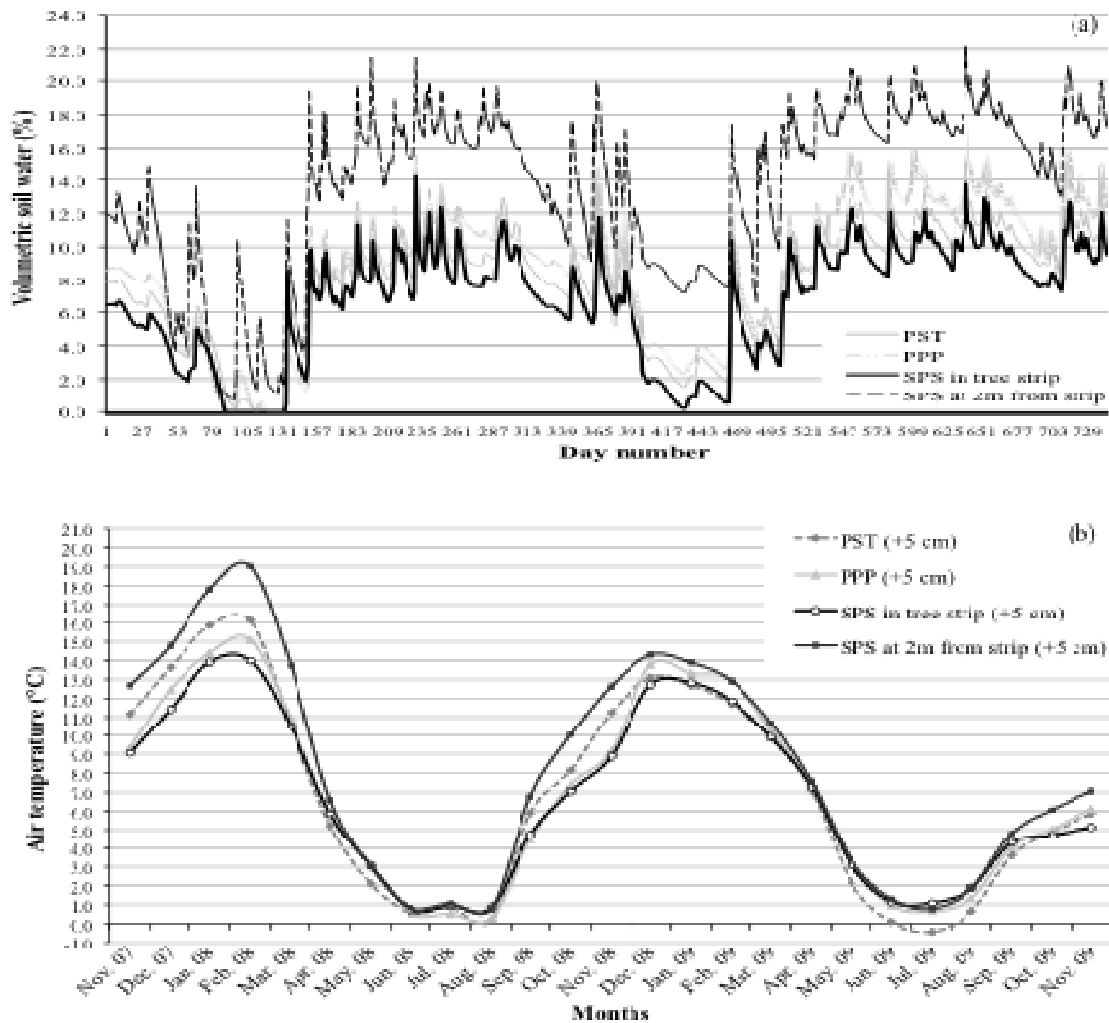


Figure 3.3. Daily soil moisture (a) content (%VWC at 0-20 cm depth) and air temperature (b) just above the soil (°C at +5 cm) measured at random in the natural pasture (PST) and ponderosa pine plantation (PPP) and at two distinct locations in the silvopastoral system arranged in strips (SPS), between November 2007 and 2009.

### 3.4. Discussion

#### 3.4.1. Distribution of carbon contents in ponderosa pines

The presence of herbaceous legumes (*Trifolium pratense* and *T. repens*), which can fix large amounts of atmospheric N (Stevenson and Cole, 1999), could perhaps explain why pine

needles in SPS contained significantly more C than any other tree components (Table 1). Clover was seeded soon after the establishment of SPS 6 years ago, and likely influenced the amounts of soil N at 0-20 and 0-40 cm depths that were measured in 2009, which were significantly higher than in PPP (Table 4). Although it may take several years before tree growth can be enhanced by soil N enrichment (Jose et al., 2004), the results from this study indicate that SPS with leguminous pasture alleys had a greater increase of tree diameter compared to PPP (Table 2). Similar results were reported by Sharrow et al. (1996) who analyzed Douglas fir-sub clover silvopastoral systems and forest monocultures of the same species. Larger trees in SPS mean that more C is allocated to various tree components, as shown in Table 1 and Figure 1b, but that less C is used for lignin in needles (F.Dube, unpublished data). Additionally, litterbag experiment indicated that in both treatments the total C contents in needles was significantly lower after 24 months compared with the initial C contents, in agreement with findings of Berg and Ekbohm (1991), and that it was lower in SPS needles than PPP needles (F.Dube, unpublished data). At the beginning of litter decomposition, there may have been a larger proportion of C in the form of soluble compounds in SPS needles, which were gradually leached and metabolized by microorganisms during the decay process, resulting in significantly lower amounts of C compared to PPP needles.

Additionally, it is known that water-soluble compounds such as sugars, lower fatty acids, amino acids, and peptides are easily taken up by microorganisms, resulting in rapid decomposition and lowering their concentrations in soil (Berg and Laskowski, 2006). Holocellulose concentrations will also decrease with time, but at a slower rate. However, litter with lower levels of lignin will show a faster change in holocellulose concentration, since holocellulose is not as encrusted in lignin and therefore, more degradable because it is accessible to microorganisms. In the current study, lignin contents increased significantly over time, but remained lower in the needles of SPS, which means that additional C contained in cellulose and hemicelluloses may also have been degraded during the 24-month decomposition period and have contributed to the decrease of total C content of SPS needles (F.Dube, unpublished data).

The lower stand density in SPS was responsible for the greater proportion occupied by the live crown (Figure 1b), where branches and twigs have larger diameters. Based on these results, it becomes clear that trees in SPS are using more efficiently the site resources (i.e. water, nutrients, light, etc.) and have their growth enhanced by the additional soil N provided by the leguminous pasture, resulting in larger amounts of C being sequestered over time. In SPS, there was a favorable microclimate between the tree strips (Figure 3), and a possible synergy resulting from the positive interaction of active aboveground and subterranean C cycles of the tree and grass components. Sharrow and Ismail (2004), compared 11-year-old Douglas fir/subclover silvopastoral systems with planted forests and grasslands, and reported a more efficient C accumulation in the silvopasture due to more active cycling patterns of nutrients resulting from the association of forest and grasses in the same parcel of land.

In this study, it was not possible to measure the tree root biomass by destructive sampling, but it was, however, estimated using biomass functions (Gayoso et al., 2002) that were specifically developed for ponderosa pine plantations growing in that climatic zone of Patagonia (Dube et al., 2009). Since these equations were developed for plantations, they would likely underestimate the amount of tree biomass in agroforestry systems, considering that trees in SPS have greater DBH and contain more C per individual components. This assumption was confirmed by applying the aboveground biomass functions to inventory data from SPS, where the calculated values obtained were up to 30% lower than those that were measured on site. Consequently, root biomass functions that depend on DBH also will probably underestimate the actual values, as well as the total below ground C content. The amount of C stored in SPS, both on a tree and hectare basis, would be greater still.

The significantly larger amounts of C stored in roots and trunks of PPP on an area basis were expected considering the large difference of the tree densities (Table 1). However, bigger trees in SPS compensate to a certain extent for the lower density, not including the additional C sequestered in grass roots and soil pools (Tables 3 and 4), thereby making of SPS the best alternative of all. Considering that individual trees in SPS sequestered almost 30% more C in the total above and belowground biomass compared to PPP suggests that a moderate increase in tree density with slight modification to the system design could further enhance C

sequestration in the tree component. There would also be increased biomass production for energy. Additionally, the use of fast growing species with high C sequestration potentials, such as poplars (Peichl et al., 2006), could permit higher C storage in shorter periods of time, but higher densities may be required to maintain the same wind-break efficiency throughout the year as that of pines.

Tree crowns at younger age can be important contributors to the generation of renewable energy, especially in SPS where it is significantly larger than PPP (Table 1; Figure 1b). However, as the trees age, the relative proportion occupied by the crowns decreases as well as the amount of carbon allocated to that part of the tree (Litton et al., 2007). Given the better tree growth and the higher C accumulation potential in SPS (Figure 1 and Table 2), the difference between available biomass for energy from PPP should diminish with time, resulting in a larger amount of trunk and branch biomass at the age of rotation.

In the northwest Patagonia of Argentina, Laclau (2003a) compared C sequestration of ponderosa pine plantations and native cypress forests that were established on volcanic soils. Mean annual rainfall was 1053 mm and temperature 10°C, and the age of trees ranged from 15 to 20 years. Carbon storage was 32100, 7200, 4800 and 8300 kg C ha<sup>-1</sup> in tree stems, branches, foliage and roots, respectively, totaling 44100 kg C ha<sup>-1</sup> in ponderosa pine aboveground biomass and 52400 kg C ha<sup>-1</sup> in total tree biomass. Carbon stored in trunks, branches, and foliage represented 73, 16 and 11%, respectively, of the aboveground C while the C stored in roots was 16% of total tree C. In the Chilean Patagonia, the C stored in trunks, branches and foliage of PPP was 50, 30 and 20%, respectively, of the total aboveground C while the C stored in roots represented 36% of total tree C. Although two times more C per hectare was stored in the Argentine tree trunks, and 1.3 times less C stored in branches and foliage compared to this investigation, caution must be exercised when making comparisons on a hectare basis if the tree density or basal area is not known. Nevertheless, it is interesting to see the substantially higher contributions of branches and foliage to the aboveground C in PPP compared to the Argentine study, whereas the contribution of branches and stems together to aboveground C was 89% in the Argentine study and 80% in this investigation in Chile.

In this study, the contribution of root C to the total tree C is more than double in PPP, compared to the Argentine investigation. These differences of C allocations could be attributed to the different mean annual summer and winter temperatures, timing and amounts of rainfalls, duration of frost-free periods, available site nutrient and water supplies, and movement of the trees caused by wind which may increase the root:shoot ratios (Laclau, 2003b). According to Cannell (1989), nutrient-rich soils can substantially increase the amount of C captured in species such as pines, modifying their capacity for light interception and C partitioning between roots and trunks. However, despite the differences of C partitioning among the different pine components, the total tree biomass C is only 10% larger in the Argentine trees, which could be non-significant if the density or basal area of the Argentine stand were higher than that of PPP. In a pure perspective of C sequestration, the large proportion of C stored in PPP roots over successive rotations could be beneficial since the residence time of C is longer and the turnover rate slower than in the adjacent natural prairie (Laclau, 2003b).

In a previous investigation in Central Oregon, where mean annual temperature and precipitation were 7.5°C and 552 mm, respectively, a 15-year-old stand of ponderosa pine had a mean tree height of 4 m and DBH of 10 cm. The C storage values were 5190 and 600 kg ha<sup>-1</sup> in wood (bark, branch and trunk) and foliage, respectively, totaling 5790 kg C ha<sup>-1</sup> in aboveground biomass, whereas C storage in root biomass reached 5630 kg C ha<sup>-1</sup> (Law et al., 2001). Although the Oregon trees were only 3 years younger, the amount of C stored in wood, foliage, total aboveground and root biomass was approximately 5, 10, 5 and 3 times lower, respectively, than in the pine plantation in the Chilean Patagonia. In addition, an inventory done in 2003 indicated that 12-year-old trees in PPP had a mean height of 6.7 m and DBH of 11.4 cm, larger already at that age than the trees in Oregon. Besides the slightly higher annual temperatures and substantially larger precipitation in the Chilean Patagonia, other factors such as the tree density and the type of soils and their influence on water supply should also be considered to explain the differences between the investigations. The Oregon forest was grown from natural regeneration after clear cutting an old-growth ponderosa pine forest, and the soils were Ultic Haploxeralfs. In Patagonia, the Andisol soils in PPP are characterized by high water contents at 1500 kPa (Stolpe et al., 2010) and higher saturated and unsaturated

hydraulic conductivity than most other types of soil (Shoji et al., 1993). Water movement and soil moisture content will therefore be greater under most conditions, affecting tree growth. The greater foliage and rooting systems in PPP, where there is more photosynthetic tissue and better access to soil water, may also explain the higher growth rate in this investigation as compared with the Oregon study (Cannell, 1989).

### 3.4.2. Growth and productivity

The climatic conditions that prevail in the study region of Patagonia, where only 15% of annual precipitations occur during the 3-months of summer, which coincides with the windiest period (Dube et al., 2009), may cause seasonal water deficits and limit the annual tree growth in both PPP and SPS. However, the significant difference of annual DBH between the Patagonian ecosystems was likely due to the smaller tree density in SPS resulting in lower individual competition and better utilization of site resources such as water, nutrients and solar radiation. In addition, the presence of clover in the pasture alleys that provides extra N to the soil is another asset of silvopastoral systems that will eventually enhance tree growth, especially trunk diameter increment. Nevertheless, it may take several years after the establishment of the legume component of pasture alleys, while N becomes sufficiently enriched by symbiotic fixation, to benefit tree growth (Jose et al., 2004). A previous investigation on the effect of clover and urea as source of N for the establishment of *Pinus radiata* plantations and silvopastoral systems, found reduced tree growth during the first three years of study. Subsequently, there was an increase of growth, where soil N and DBH were 36 and 14% higher, respectively, at the end of the seventh year (Waring and Snowdon, 1985).

The similarity of annual increase of HT between treatments (no significant difference) was probably because HT is primarily an index of site quality and productivity (Espinosa et al., 1990), and considering that PPP and SPS were established on the same soil type, there was no significant variation between the treatments. The lower tree density in SPS that reduced the competition among trees and greater N availability in soil that improved the fertility of the site, are likely the only factors that influenced the tree heights.

There were a higher growth rate of BA in SPS as compared with PPP because of the greater increase of DBH, and the significant differences observed among treatments, on a yearly basis, could be attributed to the number of trees per hectare. The effect of the double tree density in PPP is reflected by the larger basal area compared to SPS, that was initially twice as much as SPS, but was reduced with each year of growth. The decline of tree growth in PPP was related to increased competition among individual trees, which will intensify in the coming years, while the SPS should have a sustained increase of its BA. The SPS had a lower tree volume per hectare, but higher individual tree productivity (Figure 1), which is ideal to produce high quality timber for sawn wood. It should be noted that higher tree densities in the middle of the strips in SPS might affect tree growth therein, resulting in smaller DBH values of those individuals during subsequent tree inventories. If individuals within the tree strip were less closely spaced, the width of the pasture alley would be reduced and slightly less area would be available for pasture, but tree growth would be better, especially in the middle of the strip, resulting in larger mean BA.

In a study of C accumulation resulting from the encroachment of ponderosa pine in Colorado forests, Hicke et al. (2004) reported a mean BA of  $19.1 \text{ m}^2 \text{ ha}^{-1}$  corresponding to a tree density of  $841 \text{ trees ha}^{-1}$  ( $>4 \text{ cm DBH}$ ) and  $681 \text{ trees ha}^{-1}$  ( $<4 \text{ cm DBH}$ ). Another investigation of the same species in a forest in Arizona measured BA at  $18.2 \text{ m}^2 \text{ ha}^{-1}$  and a density of  $720 \text{ trees ha}^{-1}$  (Fulé et al., 1997), and a separate study in the same state reported a BA of  $27.2 \text{ m}^2 \text{ ha}^{-1}$  where the density was  $1870 \text{ trees ha}^{-1}$  (Covington and Moore, 1994). In this investigation, the positive effect of tree thinning of PPP in 2003 is evident, where the density was reduced from 1514 to  $800 \text{ trees ha}^{-1}$ . In the Colorado study where the median tree age was 79 years, the mean BA was 1.8 times less than 18-year-old trees in PPP, although the tree density was almost twice as large. In the first Arizona experiment, the density was only 10% lower but the BA was almost half that of PPP in this study, whereas in the second Arizona study, there was 20% less BA, but nearly 2.5-time higher tree density than in PPP. Besides the effect of density, the site quality and corresponding soil fertility play important roles in controlling tree growth and productivity (Kimmins, 2004).

The increased rates of C sequestration after thinning the original pine plantation of the PPP and SPS treatments demonstrates the importance to distinguish the two growth periods before and after thinning, especially in the silvopastoral system (Figure 1a). The mean C sequestration rate since the establishment of the pine plantation in 1991 would underestimate the current rate of trees in a purely silvopastoral system. Upon termination of the first rotation of the pine component in SPS, the new silvopastoral system may be established from the beginning of the next rotation rather than from a system conversion in mid rotation as was done in 2003. This promotes early tree growth, gaining several years over the first rotation. Besides that, the conversion in mid rotation is very demanding of labor and generates large amounts of slash that must be disposed of in order to establish the pasture in the alleys. The resulting stumps from fallen trees pose additional problems for pasture growth and cattle transit, and decrease the effective area that could otherwise be available for grass and herbaceous legumes. Finally, site disturbance associated with tree removal in the future pasture alleys promotes erosion and additional soil respiration, an important C output to the atmosphere (Schlichter and Laclau, 1998).

#### *3.4.3. Prairie productivity and distribution of C contents*

The drought that occurred in summer of 2008 affected grass growth in PST and SPS, but the effect appeared to be greater on a hectare basis in SPS since tree strips occupy some of the area that would otherwise be available for pasture. However, the ANPP in SPS was only slightly lower than PST in 2008 and almost the same in 2009, which illustrates the unique role played by trees in the creation of a favorable microclimate. The tree strips reduce wind speed, moisture deficit in soil and moisture loss via evapotranspiration, and increase of air and soil temperature, which help the growth of pasture in alleys (see also Figure 4) of the agroforestry system (Garrett et al., 2004). In addition, the thermal cover provided by trees may help to prevent frost damage to the pasture portion and increase the length of the growing season.

The greater tree density in PPP and increased shading can explain why the mean annual ANPP was lower compared with PST and SPS. Throughout the year, at 2 m from the tree strips in SPS, the air temperature was higher (2.3°C in 2008 and 1.1°C in 2009) and the soil moisture



was greater than PST, which helps to understand why ANPP was higher on a quadrant basis in SPS than PST. In fact, the mean soil moisture in the pasture portion of SPS was twice as high as that of PST during both growing seasons (F.Dube, unpublished data). The influence of climatic conditions on pasture growth and yield is therefore better assessed and understood when comparing data for both ecosystems on a quadrant basis since it accounts for the area unavailable for pasture in SPS. Sharrow et al. (1996), working with Douglas fir/subclover agroforests, pasture and forest monocultures in Oregon, reported an annual average of total forage production, measured over a five-year period, of 6.5 Mg ha<sup>-1</sup> in a pasture, 6.3 Mg ha<sup>-1</sup> in a cluster pattern agroforest and 2.6 Mg ha<sup>-1</sup> in a monoculture plantation of trees. Although these values are 2-4 times higher than what was obtained in Patagonia, the trend is similar, being the annual forage production similar in the pasture and the agroforest but substantially larger as compared with the plantation.

With respect to the higher annual BNPP encountered in PST as compared with SPS, this could be attributed to the harsher environment under which PST is growing, forcing the pasture to develop more extensive root systems where photosynthate reserves can be stored (Gibson, 2009). Raich and Tufekcioglu (2000) also reported that in adverse climatic conditions large proportions of the photosynthates produced by prairies were allocated to belowground roots. Indeed, soil moisture over the two-year measurement period was substantially lower in PST than SPS (Figure 4a), with a mean annual difference of 5% (F.Dube, unpublished data), whereas the annual air and soil temperatures in PST were almost 1.5 and 0.5°C less, respectively.

#### *3.4.4. Soil carbon and nitrogen*

Dube et al. (2009), working on the same site in March 2006, reported C concentrations values in PST of 9.85, 4.41 and 1.84% at 0-5, 5-20 and 20-40 cm depths, respectively, and 4.51, 3.24 and 2.80% at the same sequence of depths in PPP. No measurements were done in SPS at that time. The trends are similar to those in this study, which was to be expected. Three years later, C concentrations in PST have increased 1.1, 1.3 and 1.6 times at 0-5, 5-20 and 20-40 cm depths, respectively, and 1.4, 1.2 and 1.4 times at the same depths in PPP. However, the

different sampling intensities and patterns may not adequately address the properly the spatial variability in soil C and to understand the reasons for specific increases at different soil depths. Apart from this uncertainty, it is still clear that soil C is slowly increasing over the years, but the rate at which it does so will depend on characteristics of the ecosystems and the level of management therein. As indicated by Peichl et al. (2006), the high variability in soil C and N concentrations at 20-40 cm depth may be due to soil heterogeneity and the presence of C-rich pockets, which in this study, could consist of partly decomposed woody material coming from the original native forest, unevenly distributed in soil across the landscape.

The larger concentrations of soil N in SPS are linked to greater C concentrations and gains in SOC storage, which may result in the amelioration of soil fertility (Bambrick et al., 2010). Finally, the greater C/N ratio with increasing soil depth in every ecosystem indicates a greater stability of soil organic matter and resistance to microbial decomposition (Dube et al., 2009).

Although the initial soil C concentrations could not be measured in PST in 1991 and PPP in 2003, the data suggest that the plantation transition into silvopastoral system resulted in more C being sequestered at 0-20 and 0-40 cm depths in SPS, while there was a significant loss of soil C at the same depths after establishing PPP on the natural prairie (Table 4). Other studies have shown increases of soil C following conversion of plantations to intercropping systems (Bambrick et al., 2010) and decreases after afforestation of pastures with first-rotation pine plantations (Ross et al., 2002; Dube et al., 2009).

One possible explanation for the larger C concentration in SPS tree strips at 20 cm depth, as compared with PPP could be the different tree densities. As mentioned in the previous section, the tree strip has never been thinned since the establishment of the original plantation in 1991, resulting in a higher density with larger amount of roots per unit area therein than PPP. More root exudates may therefore be produced as well as more dead material from fine root turnover. As a consequence, these additional inputs of organic matter may contribute to increases of the soil C concentration. On the other hand, the presence of herbaceous legumes since the establishment of SPS resulted in significantly greater amounts of soil N stored at any location of SPS than within PPP, and it has been shown that soil C and N contents can be

increased by 20-100% with the presence of N-fixing plants (Johnson, 1992). Additionally, Sharrow and Ismail (2004) reported larger amounts of soil C in silvopastoral systems with subclover and ryegrass than planted forests, due to greater available soil nitrogen.

The 35% decrease of SOC associated with the establishment of PPP over pasture is in agreement with Huygens et al. (2005), who reported a 42% decrease of SOC at 0-30 cm depth after the conversion of grasslands to *Pinus radiata* plantations in south Chilean volcanic soils. The large biomass and turnover of roots in the upper soil layers of pasture in PST and SPS produce, over time, a thick root mat with limited aeration, resulting in lower decomposition rates of organic matter and accumulation of humus in the soil surface, which could therefore explain the larger SOC contained therein (Dube et al., 2009). However, as the pines age in PPP and SPS, larger structural/coarse roots of trees will develop, resulting in more biomass in the subsoil (Kimmins, 2004). Additionally, Buol et al. (1997) reported that tree litter that falls on forest ground constitutes a source of organic matter that is only partially decomposed before being incorporated into the soil, leading to lower SOC accumulations and formation of thinner A horizons as compared with pastures. In the case of prairies, the situation is the opposite since plants have a shorter life cycle and can easily decompose, resulting in larger C inputs from shoots and roots to the superficial horizons. The preponderance of SOC in the first 20 cm in relation to total SOC at 0-40 cm depth shows the importance of the upper soil in active nutrient cycling of all treatments, especially PST and SPS. This is supported by the finding of Dube et al. (2009), who reported microbial biomass C contents 2.5 times larger in PST than PPP at 0-20 cm depth on the same site of the study area.

It should be noted that although there was no significant difference in SOC ( $\text{Mg ha}^{-1}$ ) between SPS and PST at 0-40 cm depth was not significant (Table 4), the small increase of 0.46% SOC in SPS would result in  $17 \text{ Mg ha}^{-1}$  of additional SOC in the first 40 cm. This potential addition of SOC in SPS could be attributed to annual litterfall from trees (needles and cones) and fine root turnover, which will continue building up as the system ages. When comparing SOC (%) between SPS and PPP in the upper 40 cm of soil, the SOC in SPS was significantly higher by 1.24%, which resulted in an extra  $45 \text{ Mg C ha}^{-1}$  being sequestered in SPS at that depth. Regardless of the double tree density in PPP, the combination of trees and pasture in SPS

resulted in more C being sequestered in the soil, and the trees in SPS were larger (at the time of testing). This difference in tree size compensated for the lower tree density and produced similar amounts of litterfall and net additions to the soil C pools. Also, in SPS, pasture root turnover (Jobbagy and Jackson, 2000), higher microbial biomass and additions of cattle have contributed to higher SOC (Dube et al., 2009).

When comparing SOC storage among different types of agroforestry systems, it is important to consider the spatial arrangement of trees and crops/pasture. For example, in a tree-based intercropping system that resulted from the conversion of a 6-year old plantation in Quebec, Canada, Bambrick et al. (2010) reported significantly greater SOC within 0.75 m of the single poplar row than at 5 and 7.25 m from the row. The different trends between studies could be attributed to the fact in the Chilean investigation, the SPS tree strips consisted of three closely spaced pine rows where there was diminished presence of pasture, thus explaining why SOC was lower therein than at other locations within the pasture alleys.

Dube et al. (2009) working on the same site in March 2006, encountered SOC contents of 135.8 and 116.9 Mg ha<sup>-1</sup> at 0-40 cm depth in PST and PPP, respectively. The values obtained in this study are approximately 30% larger, which seems high given the relatively short period of time. However, as stated above, cautions must be followed when making direct comparisons since soil sampling intensity differed among the studies and may have underestimated the spatial variability in SOC across the field. For example, the sampling plots for the 2006 and 2009 measurements were located in different parts of the slope, which can cause variation in SOC because of the effects of soil erosion (Bambrick et al., 2010). In addition, the C-rich pockets (Peichl et al., 2006) resulting from decaying wood residues from the original *Nothofagus* forest that were unevenly distributed across the site may have been included in the second sampling period, resulting in an overestimation of SOC. Finally, the maximum sampling depth of 40 cm may not account for C inputs from tree roots that are found at greater depths (Stolpe et al., 2010), causing an underestimation of the total soil C. It is evident that questions relating to soil sampling design and intensity must be addressed in order to more accurately determine SOC storage and accumulation in adjacent ecosystems.

### 3.5. Conclusions

Several recent studies performed in temperate regions of the world have shown that agroforestry as an integrated land-use system has greater C sequestration potential than monoculture cropping systems, or even forest plantations. In the Chilean Patagonia, the adoption of silvopastoral systems appear to be a suitable and sustainable practice that optimizes land productivity, helps control erosion processes, preserves and increases C and N pools during decades or centuries, and also contributes to reduce atmospheric CO<sub>2</sub>.

Our results indicate that trees in SPS are using the site resources more efficiently and have their growth enhanced by the additional soil N provided by the leguminous pasture, resulting in larger amounts of C being sequestered. A favorable microclimate has been observed as well as a synergistic effect between trees and pasture. Individual trees in SPS have sequestered nearly 30% more C in total biomass than trees in PPP. A sustainable increase in tree density in SPS could have the added benefit of increased biomass production for renewable energy.

Any new establishment of SPS in the region may consider the recommendations from this study in terms of C sequestration. Landowners could either establish new SPS from improved seedlings, gaining several years of agroforestry benefits compared to the conversion of existing pine plantations to SPS, and avoiding all the problems related to labor requirements and disposal of slash associated with the conversion process. However, the establishment of SPS from thinned PPP may generate additional incomes from the sale of trees for bioenergy, and result in large gains in SOC related to the slow decomposition of the remaining pine stumps and coarse roots.

Finally, since all the soils in this study originated from the same volcanic source, it was assumed that they were all initially similar in their biological, chemical and physical properties to 0-40 cm depth of mineral soil when they were covered by the original *Nothofagus* forest. The presence of herbaceous legumes since the establishment of SPS resulted in significantly greater amounts of soil N stored at any location of SPS compared with PPP, and the larger soil N concentrations are linked to greater C contents and gains in SOC storage. Stumps and coarse

roots that were left over from the original pine plantation in the pasture alleys, and regular inputs to soil of lignin-rich litter from the strip trees decompose slowly and constitute a source of chemically recalcitrant C that may contribute to significant increases in SOC in SPS.

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### **3.7. References**

- Abohassan R.A. 2004. Carbon dynamics in a temperate agroforestry system in southern Ontario, Canada. M.Sc. Thesis. Dept. Envir. Biol., University Guelph, Guelph, 122 pp.
- Bambrick A.D., Whalen J.K., Bradley R.L., Cogliastro A., Gordon A.M., Olivier A., Thevathasan N.V. 2010. Spatial heterogeneity of soil organic carbon in tree-based intercropping systems in Quebec and Ontario, Canada. *Agroforest. Syst.* 79: 343-353.
- Berg B. and Ekbohm G. 1991. Litter mass-loss rates and decomposition patterns in some needle and leaf litter types. Long-term decomposition in a Scots pine forest VII. *Canad. J. Bot.* 69: 1449-1456.
- Berg B. and Laskowski R. 2006. Litter decomposition: a guide to carbon and nutrients turnover. *Adv. Ecol. Res.*, vol. 38, Elsevier Acad. Press, San Diego, USA, 428 pp.

- Brady N.C. 1990. The nature and property of soils. 10<sup>th</sup> ed. Macmillan Publ. Co., New York, USA, 621 pp.
- Buol S.W., Hole F.D., McCracken R.J., Southard R.J. 1997. Soil genesis and classification. 4th ed., Iowa State Univ. Press, Ames, Iowa, 527 pp.
- Cannell M.G.R. 1989. Physiological basis of wood production: a review. *Scand. J. For. Res.* 4: 459-490.
- Covington W.W. and More M.M. 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. *J. For.* 92: 39-47.
- Dube F., Zagal E., Stolpe N., Espinosa, M. 2009. The influence of land use change on the organic carbon distribution and microbial respiration in a volcanic soil of the Chilean Patagonia. *Forest Ecol. Manage.* 257: 1695-1704.
- Espinosa M., Acuna E., Cancino J., Munoz F., Perry D.A. 2005. Carbon sink potencial of radiata pine plantations in Chile. *Forestry* 78(1): 11-19.
- Espinosa M., Escobar R., Drake F. 1990. Silvicultura de las plantaciones forestales en Chile: pasado, presente y futuro. *Agro-Ciencia* 6(2): 131-144.
- Fisons Instrument. 1990. Instruction manual for the EA 1108 Elemental Analyzer. Rev. A2-90. San Carlos, CA, USA. 44p.
- Fulé P.Z., Covington W.W., Moore M.M. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. *Ecol. Appl.* 7: 895-908.
- Garrett H.E., Kerley M.S., Ladyman K.P., Walter W.D., Godsey L.D., Van Sambeek J.W., Brauer, D.K. 2004. Hardwood silvopasture management in North America. *Agroforest. Syst.* 61: 21-33.
- Gayoso J., Guerra J., Alarcón D. 2002. Contenido de carbono y funciones de biomasa en especies nativas y exóticas. In: *Medición de capacidad de captura de carbono en bosques de Chile*. Proyecto FONDEF D98I1076, Univ. Austral Chile, Valdivia, Chile, 157 pp.
- Gibson D.J. 2009. Grasses and grasslands ecology. Oxford Univ. Press. New York, NY, USA, 313 pp.
- Gordon A.M. and Thevathasan N.V. 2005. How much carbon can be stored in Canadian agroecosystems using a silvopastoral approach? In: *Silvopastoralism and sustainable land management*. Mosquera-Losada M.R., Rigueiro-Rodriguez, A. McAdam J. (Eds.) CABI Publishing, Wallingford, UK, pp. 210-218.

- Hicke J.A., Sherriff R.L., Veblen T.T., Asner, G.P. 2004. Carbon accumulation in Colorado ponderosa pine stands. *Can. J. For. Res.* 34: 1283-1295.
- Huygens D., Boeckx P., Van Cleemput O., Oyarzun C., Godoy R. 2005. Aggregate and soil organic carbon dynamics in South Chilean Andisols. *Biogeosciences* 2: 159-174.
- IPCC. 2007. Climate change 2007: synthesis report. [http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4\\_syr.pdf](http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr.pdf). Accessed 6 October 2010.
- Jobbagy E.G. and Jackson R.B. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecol. Applic.* 10: 423-436.
- Johnson D.W. 1992. Effects of forest management on soil carbon storage. *Water Air and Soil Pollution* 64: 83-120.
- Jose S., Gillespie A.R., Pallardy S.G. 2004. Interspecific interactions in temperate agroforestry. *Agroforest. Syst.* 61: 237-255.
- Kimmins J.P. 2004. *Forest ecology: a foundation for sustainable forest management and environmental ethics in forestry*. 3rd Ed. Prentice Hall, Upper Saddle River USA 611 pp.
- Laclau P. 2003a. Biomass and carbon sequestration of ponderosa pine plantations and native cypress forests in northwest Patagonia. *For. Ecol. Manage.* 180: 317-333.
- Laclau P. 2003b. Root biomass and carbon storage of ponderosa pine in a northwest Patagonia plantation. *For. Ecol. Manage.* 173: 353-360.
- Law B.E., Thornton P.E., Irvine J., Anthoni P.M., van Tuyl S. 2001. Carbon storage and fluxes in ponderosa pine forests at different developmental stages. *Global Change Biology* 7: 755-777.
- Litton C.M., Raich J.W., Ryan M.G. 2007. Carbon allocation in forest ecosystems. *Global Change Biology* 13: 2089-2109.
- Montagnini F. and Nair P.K.R. 2004. Carbon sequestration: an underexploited environmental benefit of agroforestry systems. *Agroforest. Syst.* 61: 281-295.
- Nair P.K.R., Nair V.D., Kumar B.M., Showalter J.M. 2010. Carbon sequestration in agroforestry systems. In: *Advances in Agronomy*. Sparks D, (Ed.), Burlington Academic Press, Vol. 108, pp. 237-307.
- Oelbermann M., Voroney R.P., Thevathasan N.V., Gordon A.M., Kass D.C.L., Schlonvoigt A.M. 2006. Soil carbon dynamics and residue stabilization in a Costa Rican and southern Canadian alley cropping systems. *Agroforest. Syst.* 68: 27-36.



- Peichl M., Thevathasan N.V., Gordon A.M., Huss J., Abohassan R. 2006. Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. *Agroforest. Syst.* 66: 243-257.
- Pizarro R., Sangüesa C., Flores J.P., Martínez E., Ponce M. 2003. Revisión y análisis de practicas tradicionales de conservación de aguas y suelos en zonas áridas y semiáridas de Chile central. Proyecto CORFO, Universidad de Talca, Talca Chile, 111 pp.
- Raich J.W. and Tufekcioglu A. 2000. Vegetation and soil respiration: Correlations and controls. *Biogeochem.* 48: 71-90.
- Ross D.J., Tate K.R., Scott N.A., Wilde R.H., Rodda N.J., Townsend J.A. 2002. Afforestation of pastures with *Pinus radiata* influences soil carbon and nitrogen pools and mineralization and microbial properties. *Aust. J. Soil Res.* 40: 1303-1318.
- SAS Institute Inc., 2003. SAS User's guide: Statistics. SAS Institute Inc., Cary, NC, USA.
- Schlichter T. and Laclau P. 1998. Ecotono estepa-bosque y plantaciones forestales en la Patagonia norte. *Ecologia Austral* 8: 285-296. (in Spanish).
- Sharrow S.H. and Ismail S. 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. *Agroforest. Syst.* 60: 123-130.
- Sharrow S.H., Carlson D.H., Emmingham W.H., Lavender D. 1996. Productivity of two Douglas fir/subclover/sheep agroforests compared to pasture and forest monocultures. *Agroforest. Syst.* 34: 305-313.
- Shoji S., Nanzyo M., Dahlgren R.A. 1993. Volcanic ash soils - genesis, properties, and utilization. *Develop. soil science* 21. Elsevier, Amsterdam, the Netherlands, 288 pp.
- Silva F. 2004. Final report on the ecological status of the Aysén Region. SAG – Agricultural Ministry, Government of Chile, Santiago, Chile, 143 pp. (in Spanish).
- Stevenson F.J. and Cole M.A. 1999. *Cycles of soil*. 2<sup>nd</sup> Ed., John Wiley & Sons, 427 pp.
- Stolpe N.B., Dube F., Zagal E. 2010. Calibration of CO2FIX to native forest, pine plantation, and pasture on a volcanic soil of the Chilean Patagonia. *Acta Agriculturae Scandinavica, Section B – Soil and Plant Science* 60(3): 235-244.
- Tans P. 2010. Trends in atmospheric carbon dioxide – Mauna Loa. NOAA/ESRL. <http://www.esrl.noaa.gov/gmd/ccgg/trends/>. Accessed 6 October 2010.
- Teller A. 1988. Biomass, productivity and wood waste evaluation in a spruce (*Picea abies*) forest (Strinchmps 983). *Commonwealth Forestry Review* 7(2): 129-148.

- Veblen T.T., Kitzberger T., Burns B., Rebertus A. 1995. Perturbations and dynamics of regeneration of Andean forests in southern Chile and Argentina. In: Armesto J., Villagran C., Arroyo M. (Eds.), Ecology of the Native Forests of Chile, pp. 169-198. University Editorial, University of Chile, Santiago, Chile (in Spanish).
- Waring H.D. and Snowdon P. 1985. Clover and urea as sources of nitrogen for the establishment of *Pinus radiata*. Aust. Forest Res. 15: 115-121.
- Zagal E., Muñoz C., Quiroz M., Córdova C. 2009. Sensitivity of early indicators for evaluating quality changes in soil organic matter. Geoderma 151: 191-198.



## CAPITULO 4

Dube F., Thevathasan N.V., Zagal E., Gordon A.M., Stolpe N.B., Espinosa M. **Carbon sequestration and net fluxes in pine-based silvopastoral systems, forest plantations and pasture monocultures on volcanic soil in Chilean Patagonia.** In: Carbon Sequestration in Agroforestry Systems. Trabajo invitado por Nair P.K.R y Kumar B.M. (Eds.) Advances in Agroforestry Series, Springer Sciences, The Netherlands. (en revisión final)

**Abstract:** This study was undertaken to model C sequestration potentials in three predominant ecosystems: 1) *Pinus ponderosa*- based silvopastoral systems arranged in strips (Silvopasture), 2) 18-year-old managed exotic plantations (Plantation) and 3) natural prairie (Prairie), in Patagonia, Chile. The C contents of trees and pasture were determined by destructive sampling and dry combustion. Litterbags were used to measure decomposition of organic material. Soil respiration was quantified with the *in situ* soda-lime technique. Soil samples were taken at 0-5, 5-20, 20-40 cm depths in order to determine soil C and N. For the Plantation and Silvopasture, total tree C was 64% and 69% of the total system, respectively. Total above and belowground C pools were 224, 199 and 177 Mg C ha<sup>-1</sup> in the Silvopasture, Plantation and Prairie, respectively. The aboveground: belowground C pool ratio was 1:10, 1:5 and 1:177 for the Silvopasture, Plantation and Prairie, respectively. Total soil respiration decreased in the order silvopasture > Prairie > Silvopasture > Plantation, and leached C decreased in the order Plantation > Prairie > Silvopasture. Estimated system net C flux was +1.8, +2.5 and -2.3 Mg C ha<sup>-1</sup> y<sup>-1</sup> for the Silvopasture, Plantation and Prairie, respectively. Based on this study and to attain C neutrality, a land area of approximately 481 km<sup>2</sup> or 0.33% of the Chilean Patagonia territory under silvopastoral systems (Silvopasture) with cattle would be sufficient to offset all C losses (CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O) from cattle-based livestock systems.

**Keywords:** Andisols, C pools and fluxes, cattle, greenhouse gases, models, *Pinus ponderosa*, silvopastoral systems.

#### 4.1. Introduction

Between 2000 and 2010, the atmospheric concentration of carbon dioxide (CO<sub>2</sub>) has increased from 369 to 388 ppm, a 5.1% increase over the last 10 years, let a lone 280 ppm in 1850 (Tans, 2010). Land-use changes and fossil fuel combustion are two important anthropogenic factors that have contributed to this increase. The influence of land management on the C content in soils and biomass is well documented worldwide (Ross et al., 2002; Huygens et al., 2005; Dube et al., 2009). Land-use changes not only affect C sources and sinks, but also impact methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) emissions.

The Climate Change 2007 Synthesis Report (IPCC, 2007) proposes key mitigation practices in the agricultural sector. Among them, the use of proven crop and grazing land management to increase soil C storage, restoration of degraded lands, improved livestock and manure management to reduce CH<sub>4</sub> and N<sub>2</sub>O emissions are a few mitigation practices related to this study. Agroforestry systems rank high for all of these strategies. Well-designed and managed agroforestry systems can be effective CO<sub>2</sub> sinks, especially with the use of perennial crops and fast growing tree species. Recent studies performed in temperate regions have shown that agroforestry systems have greater C sequestration potential than monocropping systems, prairies, or forest plantations, and should be considered as real C sinks (Montagnini and Nair, 2004; Sharrow and Ismail, 2004; Gordon and Thevathasan, 2005; Peichl et al., 2006; Bambrick et al., 2010). This chapter will focus on C sequestration potentials in three distinct ecosystems in the Chilean Patagonia region (Figure 1).

Between 1920 and 1940, large areas of the Chilean Patagonia were burned down and converted into pastures for cattle, leaving the slopes exposed to an inexorable erosion and degradation of soils formed by volcanic ash deposits. Overgrazing has also contributed to forest destruction (Silva, 2004). The deforested areas, especially those located on steeper slopes suffer extreme erosive processes, which complicate in some cases the reestablishment of native forest. The recent ratification of the Kyoto Protocol by Chile coupled with C sequestration potentials of Patagonian agroforestry systems opens the possibility for the progressive adoption of silvopastoral systems, well adapted to the climatic conditions and economic reality of Patagonia.

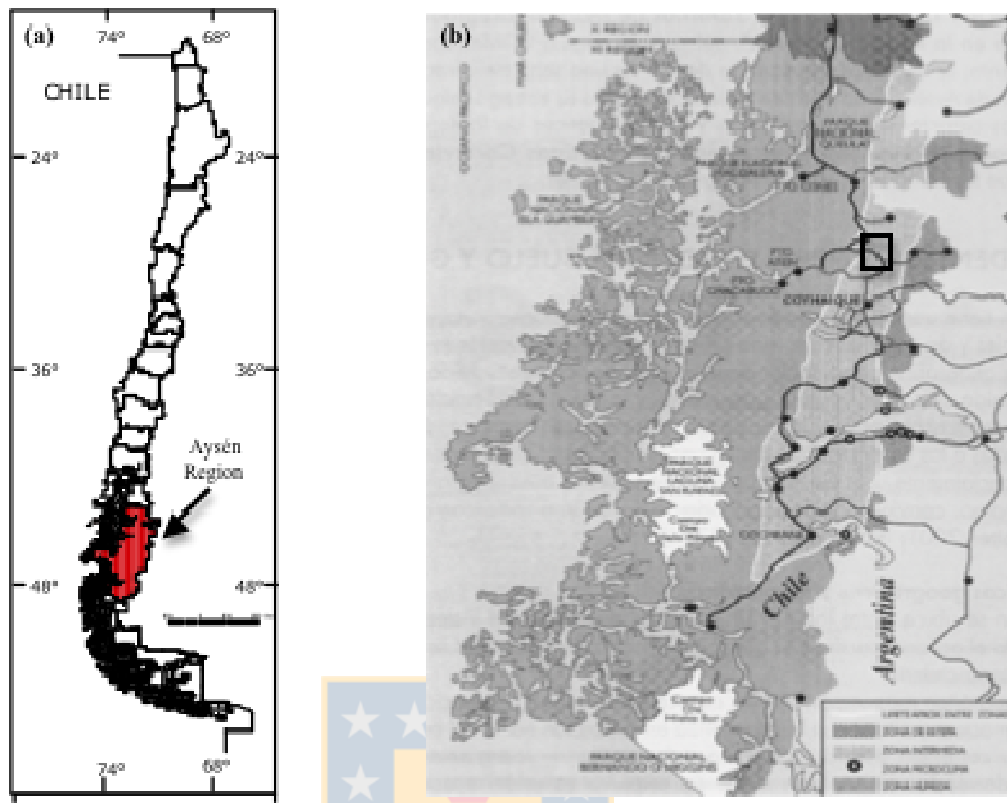


Figure 4.1. Map of Chile (a) and location of Patagonia (in black) and the Aysén Region (in red) in Patagonia (Source: INE, 2007b); (b) location of the Mano Negra Sector (insert) in the Intermediate Temperate ecological zone, Aysén Region, Chilean Patagonia where data were collected in 2007-2009 (Source: Teuber and Ganderats, 2009).

Profitability from farming, ranching and plantation forestry in Chile has decreased recently, mainly because of market globalization (Teuber and Ganderats, 2009). The harsh prevailing weather conditions, geographical isolation, high costs of production, and low technological development make it difficult for the implementation and development of new production sectors. Profitability can only be improved through innovation and the incorporation of technologies that increase the efficiency of traditional activities, one of them being the integration of forest practices and ranching on the same unit of land, resulting in a symbiosis that benefits both sectors. However, more knowledge is needed to understand the functioning of the resulting systems.

The Instituto Forestal-INFOFOR (Chilean Forestry Institute) has implemented a series of incentives to landowners to adopt agroforestry, mostly pine-based silvopastoral systems and windbreaks on their properties as a sustainable practice that satisfies their socio-economic needs while conserving the natural resource base (Teuber and Ganderats, 2009). However, there is limited number of studies on Chilean agroforestry systems and a general lack of scientific research on C sequestration aspects. In addition, since pine plantations already occupy more than 30,000 in Chilean Patagonia and natural prairies over one million ha, their C pools and fluxes must be examined as soon as possible. Therefore, the current study is the first to evaluate and model C sequestration potentials in a natural prairie (Prairie), a managed *Pinus ponderosa* plantation (Plantation), and a pine-based silvopastoral system (Silvopasture) on Andisols in the Chilean Patagonia, and to determine which ecosystem has the best potential for the long-term sequestration of carbon.

## 4.2. Material and methods

### 4.2.1. Site description and experimental design

The research site was located in the San Gabriel Agroforestry Unit within the Mano Negra Sector (Figures 1b and 2). It was established in 2002 by INFOFOR 30 km north of the city of Coyhaique in the Aysén Region, on a western exposed slope at 730 m altitude, Lat S 45°25'; Long W 72°00'. The study took place in the Intermediate agro-ecological zone of the Aysén Region of Chilean Patagonia. The annual precipitation varies from 1000 mm to 1500 mm. However, only 15% of the precipitation occurs between December and February, coinciding with the warmest and windiest period. Mean temperatures fluctuate between 12°C and 14°C in summer and 2°C and 3°C in winter (Dube et al., 2009). During summer, strong westerly winds provoke seasonal water deficits and wind erosion, which may diminish soil organic matter (SOM). The mineral soil horizons have Andic soil properties that include low bulk density ( $< 0.9 \text{ g cm}^{-3}$ ), high P fixation values (65-89%), and high water content at 1500 kPa tension relative to the measured clay content. The soil was classified as medial, amorphic, mesic Typic Hapludands (Stolpe et al., 2010).



Figure 4.2. Aerial photograph of the pine-based silvopastoral system arranged in trip (Silvopasture), surrounded by the managed natural prairie (Prairie) and the ponderosa pine plantation (Plantation), in the Mano Negra Sector, Aysén Region of the Chilean Patagonia.

Three land-uses, hereafter referred to as treatments were studied: 1) managed natural pasture (Figure 3a) with traditional cattle grazing (Prairie), 2) 18-year-old thinned and pruned *Pinus ponderosa* (Figure 3b) exotic plantations (Plantation), 3) pine-based silvopastoral systems arranged in strips (Silvopasture), where the width of pasture alley was 21 m (Figure 3c). The entire study area was initially covered with native forest, dominated by *Nothofagus pumilio*.

In 1991, *P. ponderosa* plantations were established over pasture, with a planting pattern of 2 m x 2.5 m and a density of 2000 trees ha<sup>-1</sup>. By 2003, the density had declined to 1514 trees ha<sup>-1</sup>, the mean tree height was 6.7 m, the diameter at breast height (DBH) was 11.4 cm the basal area was 15.3 m<sup>2</sup> ha<sup>-1</sup> and the crown cover was 90%. Part of the plantation (5 ha) was thinned down in 2003 to 800 trees ha<sup>-1</sup> (homogeneous spacing) while another section was thinned down to 400 trees ha<sup>-1</sup> and converted into a silvopastoral system arranged in strips (5 ha).

Although the tree density in SPS as a whole was 400 trees ha<sup>-1</sup>, the actual density within the tree strip (6 m wide) is equivalent to approximately 1444 trees ha<sup>-1</sup> with an average spacing of 2.3 m x 3 m, because that forest portion of the silvopastoral system has never been thinned since the establishment of the pine plantation in 1991. The Prairie and Silvopasture have a stocking density of 0.5 cows ha<sup>-1</sup>, and consists of a mixture of perennial grasses (*Dactylis glomerata*, *Holcus lanatus*, *Poa pratensis*), leguminous pasture (*Trifolium pratense*, *T. repens*), and other accompanying species (*Acaena magellanica*, *A. pinnatifida*, *Hypochoeris radicata*, *Taraxacum officinale*). According to a recent inventory of the botanical composition, perennial grasses, leguminous pasture, weeds and dead material represent 36, 30, 19 and 15%, respectively, of total dry matter in PST, whereas in SPS, grasses represent 29% of the total, legumes 40%, weeds 8% and dead material 23%.



Figure 4.3. Permanent plots established in the (a) managed natural prairie (Prairie) near Cerro Rosado volcano, fall 2008, (b) thinned and pruned ponderosa pine plantation (Plantation), and (c) silvopastoral system with Black Angus cattle grazing between strips of ponderosa pine (Silvopasture) on volcanic soil (mesic Typic Hapludands).





Figure 3. Continued.

Treatments were established in October 2007 in a completely randomized design with three replicates (or sampling plots). Given that the Agroforestry Unit was initially established as a demonstration site, no experimental design was considered at that time. The randomly distributed sampling plots were replicated within every treatment of the demonstration site itself but far from each other (at least 50 m). In this study, spatial interspersed replications together with the use of a systematic design were used to alleviate possible pseudo-replication problems (Stamps and Linit, 1999; K. Saez, pers. comm., 2007 and 2010). Each plot measured 15 m x 27 m and was located at 5 m from the border of the treatment. In the Silvopasture, each plot included three strip rows of pines (6 m wide) and a half strip of pasture on either side of the tree rows (10.5 m in length along the tree rows). In both the Plantation and Prairie, the plots had only pine and pasture, respectively. All the measurements took place between November 2007 and 2009.

#### 4.2.2. Tree and pasture biomass and carbon content

Destructive sampling of ponderosa pine was performed to determine the weight of different components (trunks, branches, twigs, needles and cones). Coarse root biomass was estimated using a subterranean biomass function that relates root biomass to stem DBH (Gayoso et al., 2002; Dube et al., 2009):

$$\text{Root biomass} = -13,2750 + e^{(2,4148 + 0,0743 \cdot \text{DBH})}$$

Where  $e$  is the base of the natural logarithm (2.71828) and DBH is the diameter at breast height (cm). The annual production of fine root (<5 mm diameter) biomass was estimated as a percentage of litterfall (Abohassan, 2004). The dry weight of dead pine branches left after pruning was estimated using nine 25-m<sup>2</sup> subplots per treatment. Decomposing trunks and stumps of *Nothofagus pumilio* left on the site were cut in pieces and recollected from eighteen 25-m<sup>2</sup> plots. Carbon contents of all organic materials were determined by dry combustion using a Fisons EA1108 CHNS-O Elemental Analyzer (Fisons Instrument, CA, USA).

The total standing aboveground pasture biomass was harvested from three randomly placed quadrats (0.5 m<sup>2</sup> each) per sampling plot (9 in total per treatment) to determine the aboveground net primary productivity (ANPP). The grazing material was harvested three

times a year during the growing season over a two-year period. Since it was not possible to measure belowground net primary productivity (BNPP), it was estimated using a known algorithm (Gibson, 2009):

$$\text{BNPP} = \text{BGP} * (\text{Live BGP/BGP}) * \text{turnover},$$

Where BNPP is the belowground net primary productivity ( $\text{g m}^{-2} \text{y}^{-1}$ ), BGP is the belowground productivity ( $\text{g m}^{-2}$ ), Live BGP/BGP = 0.6, and turnover =  $0.9 \text{y}^{-1}$  (Stolpe et al., 2010). In addition,

$$\text{BGP} = 0.79 * (\text{AGBIO}) - 33.3 * (\text{MAT} + 10) + 1289,$$

Where AGBIO is the peak aboveground live biomass ( $\text{g m}^{-2}$ ) during the growing season, MAT is the mean annual temperature ( $^{\circ}\text{C}$ ) of observed belowground biomass. The strength of this algorithm is given by  $R^2 = 0.54$  and  $P = 0.01$ .

#### *4.2.3. Litterfall and decomposition of organic substrates*

Circular conic  $1\text{-m}^2$  traps with 1 mm mesh screen and 60 cm tall were used in the Silvopasture and Plantation to collect the litterfall (Berg and Laskowski, 2006). The amount of litterfall was sampled monthly over 2 years, and weekly during the rainy seasons. In order to quantify the decomposition of litterfall, mixed grass root biomass and cattle faeces, polyester bags measuring 20 cm x 20 cm with 0.5 mm mesh were filled with the respective substrates (Berg and Laskowski, 2006) and placed on the Oe horizon (needles; faeces) or buried at 15 cm depth (root biomass). The pine needles were sampled every six months for two years, whereas the sampling of the grass roots and cattle feces was performed every three months for one year. The annual C contribution to the soil in the Silvopasture and Prairie by cattle was estimated using the quantity of faeces produced per animal per day (Yang et al., 2003; Byrne et al., 2007), the fecal C concentration and the animal stocking rates.

#### *4.2.4. Soil carbon and nitrogen*

In March 2009, soil samples were taken at 0-5, 5-20, 20-40 cm depths with a split-core soil sampler to determine total, organic, and inorganic C content as well as total N in each

treatment (Dube et al., 2009). Given the genesis of volcanic soils, the inorganic C content was almost non-existent and it was concluded that total C was the same as organic C.

#### *4.2.5. C content in leached soil solution and soil respiration*

The C concentrations in the leached soil solution under the pasture and pine roots were measured using tension lysimeters permanently installed at a slight angle to a soil depth of 80 cm. Sampling was done on a monthly basis and weekly during the rainy seasons. Dissolved C in rainwater and snowfall was determined six times a year during the months of greatest precipitation from three samplers located at random. Total soluble organic C was analyzed by combustion at 675°C using a TOC-V CPN Total Organic Carbon Analyzer (Shimadzu Corp., Kyoto, Japan). Carbon leaching represents the sum of different inputs, one of them being dissolved C in rain and snowfalls throughout the year. Knowing the C concentration in atmospheric depositions and using C concentration data of leached soil solution (Dube F., unpublished data, 2010), the contribution from the system itself to the total leached C could then be determined.

Additionally, total soil respiration was quantified with the *in situ* technique of CO<sub>2</sub> absorption by soda lime in a closed chamber (Edwards, 1982). Circular chambers were installed at 5 cm depth in the soil, assuring that the area was free of live organic matter. The measurements were done monthly throughout the year and weekly in summer. The quantity of CO<sub>2</sub> produced in 24 hours was calculated for the chamber area and converted to hectare.

#### *4.2.6. Air and soil temperature and soil moisture*

The soil moisture (0-20 cm depth), soil surface temperature (0-5 cm depth) and air temperature above the soil (+5 cm) were measured every two hours over a 24-month period using Decagon Devices EM-5B Data Loggers, EC-20 soil moisture sensors and ECT soil temperature sensors, respectively (Decagon Devices Inc., Pullman, WA, USA).

#### 4.2.7. Statistical analyses

All treatments were analyzed with the General Lineal Model procedure of SAS v.9.0 (SAS Institute Inc., 2003) for completely randomized design to test the effect of treatments on the parameters described in the previous sections (Peichl et al., 2006). All data were examined for homogeneity of variance and normality. Analysis of variance was conducted using the ANOVA procedure. Student's *t*-test for independent populations was used to check for significant differences between the treatment means (comparison by pairs) ( $P < 0.05$ ).

#### 4.2.8. C flows from cattle, decomposition of woody debris, and emissions from fertilizers

Most of the data were derived from the experimental plots. However, the values for cattle respiration, methane emissions from enteric fermentation and nitrous oxide from dung and urine patches, annual mass loss for decomposition of dead branches of *Pinus ponderosa* and boles/stumps of *Nothofagus pumilio*, annual leaching rate, and annual emissions of N<sub>2</sub>O from N fertilizer application were derived from the literature. All assumptions used in the construction of C models and calculations are given in Appendix A.

### 4.3. Results

#### 4.3.1. Carbon pools

The C pools and fluxes within the *Pinus ponderosa*-based silvopastoral system arranged in strip (Silvopasture), the 18-year old pine plantation (Plantation), and the managed natural prairie (Prairie) are illustrated in Figures 4a, 4b and 4c, respectively.

Total carbon storage was 224, 199 and 177 Mg C ha<sup>-1</sup> in the Silvopasture, Plantation and Prairie, respectively. These C pools do not include *Nothofagus pumilio* (lenga) coarse woody debris and pine dead branches since they undergo a slow but constant decomposition process over the years and are therefore temporary pools. In addition, given the high annual grass root turnover in all ecosystems, C in annual fine root biomass of grasses was considered as a flux

where carbon is added to the soil C pool and not as sequestered C in grass roots (Gordon and Thevathasan, 2005). Only perennial tree roots with a diameter larger than 5 mm were considered as the belowground C pool (Abohassan, 2004). The aboveground to belowground C pool ratio was approximately 1:10, 1:5 and 1:177 for the Silvopasture, Plantation and Prairie, respectively. Eighteen years after the establishment (1991 to 2009) of the pine plantation, the total C stored in the Silvopasture and Plantation was 27 and 12% greater than that was stored in the Prairie, respectively. However, it is interesting to note that six years after the conversion (2003 to 2009) of the plantation into the silvopastoral system, the total C storage in the Silvopasture has increased 13%.

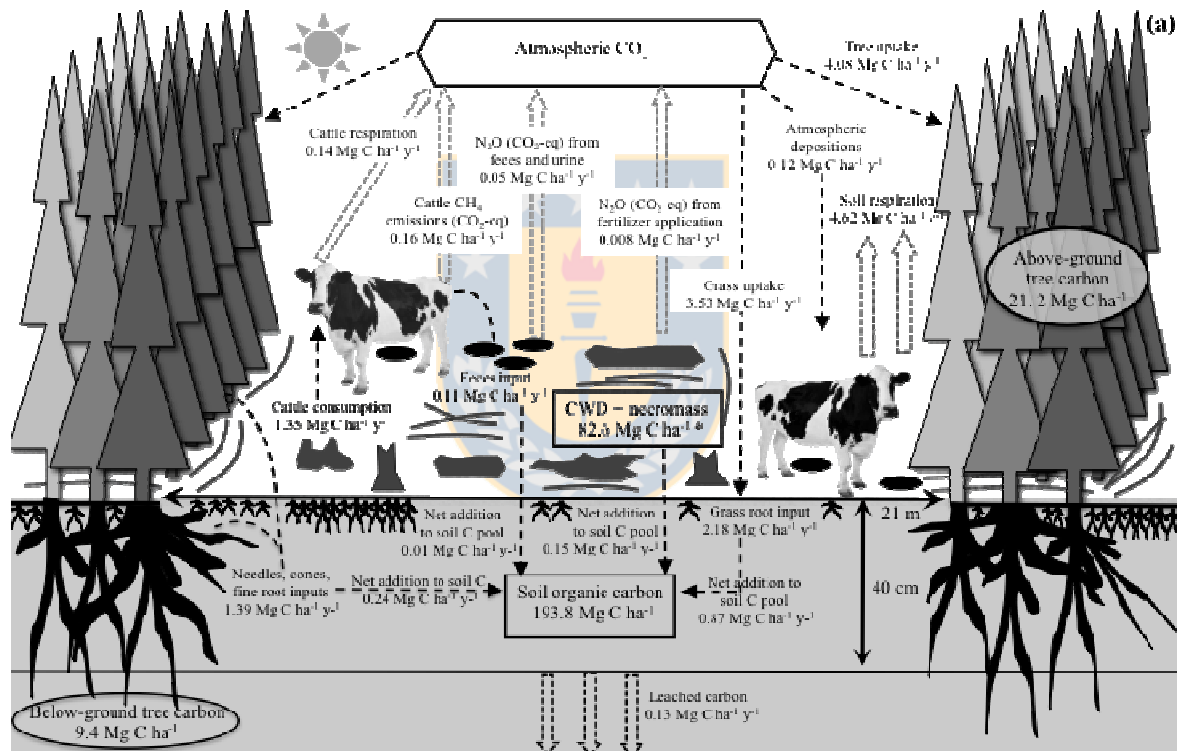


Figure 4.4. Models of C pools and fluxes for (a) a ponderosa pine-based silvopastoral system arranged in strip (Silvopasture), (b) an 18-year old *Pinus ponderosa* plantation (Plantation), and (c) a managed natural prairie (Prairie) in the Chilean Patagonia. All C pools appear in boxes and C fluxes are indicated by arrows. \*The CWD + necromass values indicated in the boxes are informative only; coarse woody debris and dead branches do not represent real C pools where storage occurs as they undergo a slow but constant decomposition process over the years. For more information on calculations, see Appendix A.

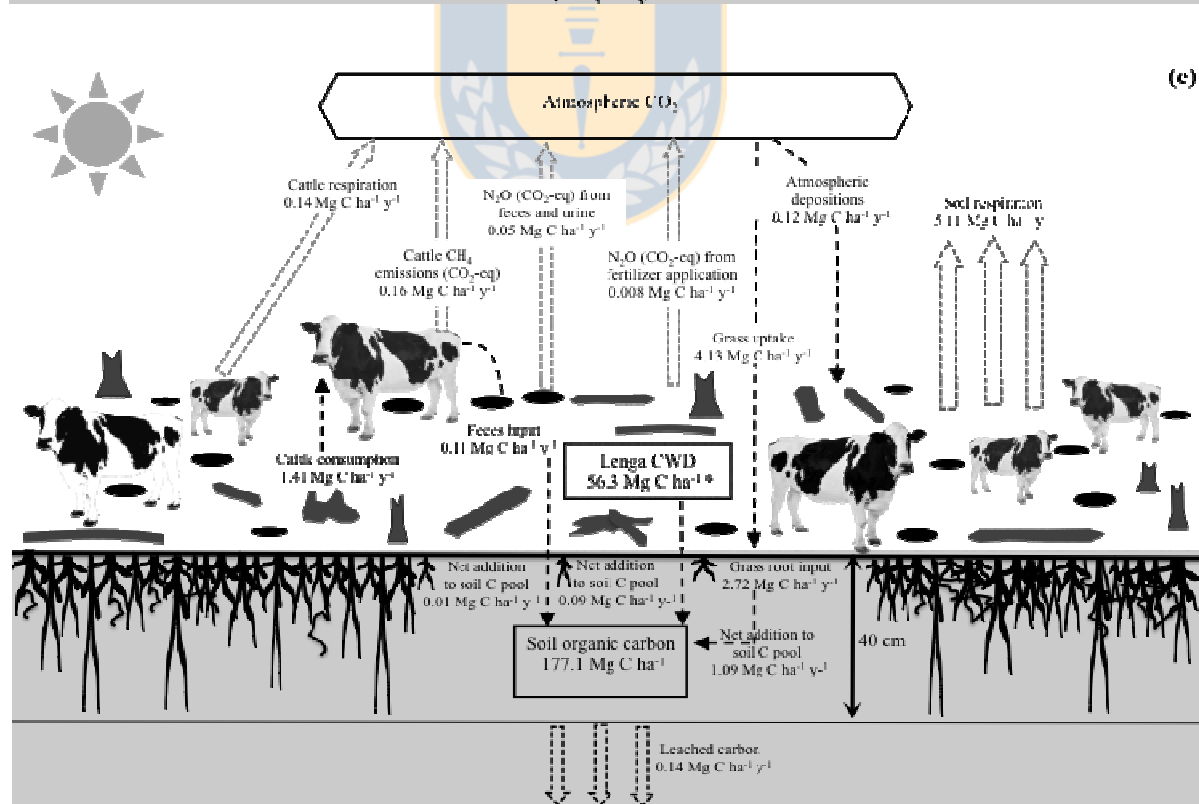
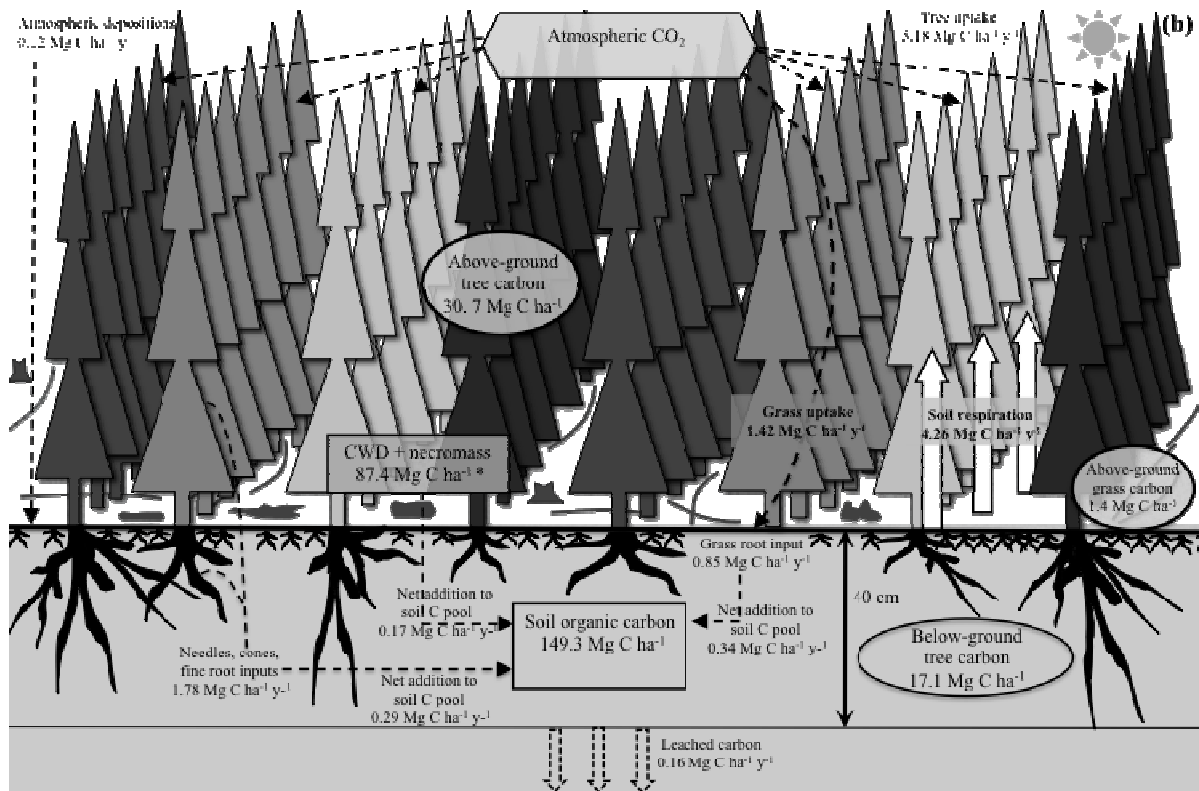


Figure 4. Continued.

As depicted in Figure 5, individual trees in the Silvopasture have sequestered almost 30% more C in the total tree biomass when compared with trees in the Plantation. On a tree basis, the C content was higher in every tree component in the Silvopasture as compared with the Plantation, but significant differences ( $P < 0.01$ ) were found only for Needles+Cones (70% higher) and Branches+Twigs (40% higher).

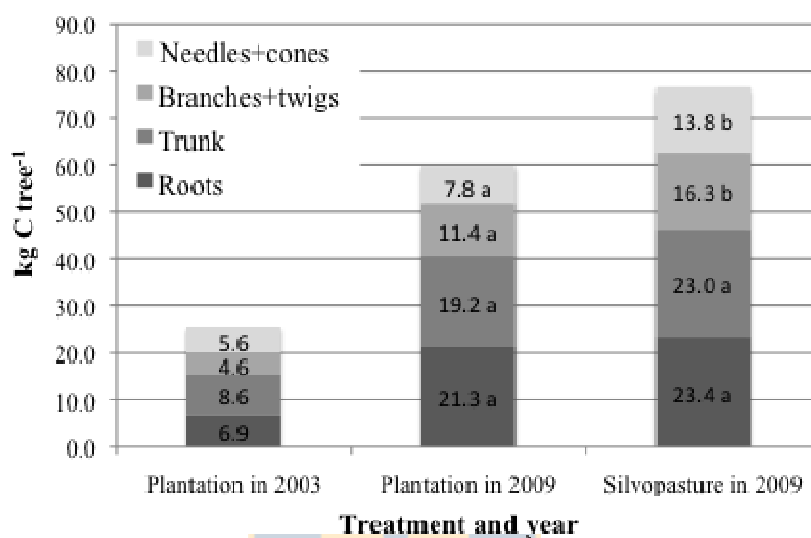


Figure 4.5. Distribution of C content (kg) per tree compartment before thinning the pine plantation in 2003 and the Plantation and Silvopasture six years after thinning. Values with the same lower case letter within a tree component and between treatments in 2009 are not significantly different (Student's t test,  $**P < 0.01$ ).

#### 4.3.2. Carbon fluxes

Net carbon flux in tested ecosystems over a two-year measurement period, were based on the following quantifications: net assimilation by trees and grass, decomposition of woody detritus, soil respiration, C leaching and atmospheric depositions, animal consumption, feces input and fertilizer applications, and was found to be as +1.8, +2.5 and  $-2.3 \text{ Mg C ha}^{-1} \text{ y}^{-1}$  for the Silvopasture, Plantation and Prairie, respectively. The highest soil respiration was observed in the Prairie ( $5.11 \text{ Mg C ha}^{-1} \text{ y}^{-1}$ ) and in the pasture component of the Silvopasture (Dube F., unpublished data, 2010). Annual C input via atmospheric deposition was  $0.12 \text{ Mg C ha}^{-1} \text{ y}^{-1}$  for all three ecosystems, and leaching C losses in the Silvopasture, Plantation and



Prairie were 0.13, 0.16 and 0.14 Mg C ha<sup>-1</sup> y<sup>-1</sup>, respectively. In relation to annual C input from litterfall and fine root production, it was 1.39 and 1.78 Mg C ha<sup>-1</sup> y<sup>-1</sup> in the Silvopasture and Plantation, respectively. Finally, the net annual C incorporation by grass roots to soil C pools was 0.87, 0.34 and 1.09 Mg C ha<sup>-1</sup> y<sup>-1</sup> in the Silvopasture, Plantation and Prairie, respectively, considering a 40% addition to the recalcitrant fraction of soil organic C (Falk, 1976).

#### **4.4. Discussion**

##### *4.4.1. Carbon pools*

The above to belowground C pool ratio depicts the preponderance of soil organic C (SOC) pools belowground. The absence of perennial woody species in the prairie is responsible for the large ratio observed in the Prairie. When comparing pools in the Silvopasture and Plantation systems, even though the tree density in Silvopasture was only 50% of that in the Plantation, the above to belowground C pool ratio was higher in the Silvopasture. The presence of highly active aerial and subterranean C cycles in the silvopastoral system (Sharrow and Ismail, 2004) could have contributed to the higher ratio in the Silvopasture in spite of lower tree density. The ratio obtained in this study for Silvopasture was the same as that reported by Peichl et al. (2006) for a spruce-barley intercropping system in Canada. In their study, above and belowground C pools were about three times smaller than those obtained in this work, which was to be expected given the lower tree density (111 trees ha<sup>-1</sup>) and non-volcanic soils in which their system was established. Volcanic soils with allophanes tend to capture larger amounts of C than non-volcanic ones (Buol et al., 1997).

After its conversion, the Silvopasture has taken only one third of the time compared to the Plantation, since its establishment, to reach similar C gains in their above and belowground pools, perhaps due to positive interactions between cattle, tree and pasture components. The large C storage potential in the Silvopasture can also be explained by higher tree growth, as shown in Figure 5. Additionally, it is worth mentioning that the proportions of total tree C pools and SOC pools in relation to the total C sequestered in the Plantation were similar to those reported by Dube et al. (2009) for the Plantation in a previous study.

If, for example, no animal grazing was allowed in all systems, the total above and belowground grass C pools would represent 6, 2 and 10% of total C sequestered in SPS, PPP and PST, respectively. The relative contribution of belowground root biomass to these values ranged from 80 to 90%. This is in agreement with the findings reported by Raich and Tufekcioglu (2000), where large proportions of the photosynthates produced by prairies were allocated to belowground roots. Furthermore, it has also been shown that grass growing in harsh environments, such as in Patagonia, tend to develop larger root systems where energy reserves can be stored (Gibson, 2009).

The total C storage potentials indicated in this research, 177 to 224 Mg C ha<sup>-1</sup>, exceed those reported in other recent studies (Dixon et al., 1994; Sharrow and Ismail, 2004; Gordon and Thevathasan, 2005; Peichl et al., 2006), demonstrating the potential of temperate agroforestry systems in C sequestration. The high C storage observed in this study could be attributed to the high C sequestration capacity of volcanic soils and their large contribution to the total system C pools. In southern Canada, Peichl et al. (2006) found total system C pools of 97, 75 and 69 Mg C ha<sup>-1</sup> for poplar and spruce intercropping and for barley sole cropping systems, respectively. Gordon and Thevathasan (2005) estimated that the C stored in all pools of a poplar-based silvopastoral system with sheep at 62 Mg C ha<sup>-1</sup> compared to 44 Mg C ha<sup>-1</sup> in a monoculture pasture system. In Oregon, Sharrow and Ismail (2004) found that in Douglas fir/ryegrass/ clover silvopastoral system, monoculture plantation and pasture systems of the same species sequestered 109, 101 and 103 Mg C ha<sup>-1</sup>, respectively. Dixon et al. (1994) estimated that agroforestry systems in temperate regions could capture between 15 and 198 Mg C ha<sup>-1</sup>.

#### *4.4.2. Carbon fluxes*

The C fluxes in this study were found to be similar to those reported by Peichl et al. (2006), who found net C fluxes of 13.2, 1.1 and -2.9 Mg C ha<sup>-1</sup> y<sup>-1</sup> for alleycropping systems with poplar and spruce and for a barley mono-cropping system, respectively. In this study and in the study mentioned above, conifer-based agroforestry systems have demonstrated net positive C fluxes. It was 1.8 Mg C ha<sup>-1</sup> y<sup>-1</sup> in this investigation and in the previous study it was 1.1 Mg C ha<sup>-1</sup> y<sup>-1</sup>, 60% higher in the Patagonian Silvopasture. Part of the difference observed could

perhaps be explained by the better growth performance of pine in relation to spruce in their respective environments, but several other factors such as the crop type must also be taken into consideration before a direct comparison can be made. For instance, Gordon and Thevathasan (2005) estimated the net annual C sequestration potential in a silvopastoral system with sheep grazing under poplars to be  $2.7 \text{ Mg C ha}^{-1} \text{ y}^{-1}$  compared to  $0.99 \text{ Mg C ha}^{-1} \text{ y}^{-1}$  in a monoculture pasture system. However, their study did not take into account C losses through leaching,  $\text{N}_2\text{O}$  emissions from fertilizers nor included long-term soil respiration data as we did in this study.

In the Silvopasture and Plantation, the net C fluxes were positive, which indicate that these systems are true C sinks. A negative net C flux in the Prairie indicates that this system is a net C source to the atmosphere. Six years after the conversion of the Plantation into the Silvopasture, the C flux of Plantation was only 39% higher than that of the Silvopasture, in spite of higher tree density and the absence of GHG emission from animals and fertilizer application. Carbon inputs from trees and grass were  $7.6$  and  $6.6 \text{ Mg C ha}^{-1} \text{ y}^{-1}$  within the Silvopasture and Plantation, respectively. These C inputs were higher than C outputs, but in the Prairie, C outputs exceeded C inputs. In the winter months, soil respiration was offset by tree photosynthesis in the Silvopasture and Plantation, although at a lower rate, but was not so in the Prairie, resulting in net  $\text{CO}_2$  emissions during the winter season.

#### 4.4.2.1. Soil respiration

The presence of trees in the Silvopasture did help to offset the high soil respiration in the pasture portion of the system, given their higher C assimilation and lower contribution to soil respiration than grass, as reflected by the values obtained within the tree strip (Dube F., unpublished data, 2010). This is also supported by a study by Raich and Tufekcioglu (2000), who reported respiration rates 20% higher in grasslands than forests growing on the same soil type and under similar environmental conditions. In addition, soil temperature and moisture are largely responsible for differences in soil respiration rates, and the moisture and temperature variations that were observed among ecosystems and at distinct locations within the Silvopasture (Table 1) may help to understand the differences recorded.

Table 4.1. Mean annual soil moisture (0-20 cm depth), superficial air temperature just above the soil (+5 cm) and soil surface temperature (0-5 cm depth) measured over a two-year period between November 2007 and 2009.

| Treatment                        | Soil moisture<br>0-20 cm<br>(% VWC) | Air temperature<br>+5 cm<br>(°C) | Soil temperature<br>0-5 cm<br>(°C) |
|----------------------------------|-------------------------------------|----------------------------------|------------------------------------|
| Prairie                          | 8.7                                 | 6.7                              | 6.5                                |
| Plantation                       | 8.0                                 | 6.8                              | 6.6                                |
| Silvopasture (within tree strip) | 6.7                                 | 6.6                              | 6.4                                |
| Silvopasture (2m from tree)      | 13.6                                | 8.0                              | 6.9                                |

%VWC: volumetric water content.

But other factors such as root activity and density, the presence of mycorrhizae (Kimmins, 2004), and the availability of C substrates for microbial biomass (Dube et al., 2009) may also have influenced soil respiration rates. In addition, it should be noted that the Aysén Region of the Chilean Patagonia was under indigenous forest in the past and had large quantities of coarse woody debris (CWD) on the ground, covering approximately one third of the total study area based on a CWD inventory data (Dube F., unpublished data, 2010). Therefore, the presence of significant amount of decaying woody material left on the soil could also have contributed to elevated soil respiration in the three studied systems, which exceeded grass C assimilation in the Prairie. Therefore, it should be noted here that a large extent of the Patagonia region (more than one million ha), which is currently under Prairie is acting as a C source based on the data from this study.

#### 4.4.2.2. Atmospheric depositions

Although the annual C input via atmospheric deposition value appears to be insignificant in the presented models, it is important to assess the atmospheric annual C inputs to the overall C budgets for the test site (Chilean Patagonia region), and verify if recent volcanic activity caused additional C depositions to the soil. Having said this, it should be mentioned that only dissolved C via rainfall and snow will remove atmospheric CO<sub>2</sub> and any C addition as a result of volcanic activities will not have any effects on atmospheric CO<sub>2</sub> removal. Monthly measurements of dissolved C in rain and snowfall after the eruption of the Chaitén volcano in May 2008 did not show evidence of additional C input from ashes (Dube F., unpublished data,

2010), although it is located at 400 km North of the site, probably because of the prevailing winds blowing eastward. It is interesting to note that 75% of the mean annual atmospheric C input is organic. Sources of organic particles may include dusts and pollens. The inorganic counterpart is very likely the result of carbonic acid ( $\text{H}_2\text{CO}_3$ ) formed in small amounts when  $\text{CO}_2$  contained in the atmosphere is dissolved in rainwater (Reay and Grace, 2007).

#### 4.4.2.3. Leached carbon

With respect to total annual C losses as leachates, its contribution to the C budgets also seems very small as compared with other studies. In Guelph, Canada, Peichl et al. (2006) encountered leachate outputs of 1.8, 1.5 and 1.8  $\text{Mg C ha}^{-1} \text{ y}^{-1}$  in alleycropping systems with poplar and spruce and a barley mono-cropping system, respectively. The values reported by these authors are 7-15 times larger than those from our models, although annual precipitation and leaching rate in Patagonia are only 45% higher, the trees are older and their density is greater. The larger leaching C losses in the Guelph studies could be attributed to the calcareous soil caused by the  $\text{CaCO}_3$  bedrock. Additionally, Undurraga et al. (2009), working with annual crops on a volcanic soil in Chile, reported dissolved organic C contents of 0.0067-0.0152%, a similar range to the values of 0.0040-0.0056% obtained in this study (Dube F., unpublished data, 2010). Volcanic soils contain a special type of clay known as allophanes, which have the capacity to retain larger amounts of organic C, resulting in lower C concentrations in leachates (Woignier et al., 2007). The Patagonian ecosystems would be more efficient in reducing C losses to ground waters than ecosystems on non-volcanic soils, and thereby reduce the amount of soluble C source in aquatic ecosystems, which is required for denitrification by microbes. As a result, the lower content of dissolved C may aid in the reduction of  $\text{N}_2\text{O}$  emissions from aquatic ecosystems.

The contribution from the system itself to the total leached C, excluding dissolved C from atmospheric depositions, was quantified to be at 77.3, 82.1 and 79.6% in the Silvopasture, Plantation, and Prairie, respectively (Dube F., unpublished data, 2010). The Silvopasture is the most efficient in terms of dissolved C retention and the Plantation tends to release higher amounts of dissolved C to the environment.

#### 4.4.2.4. Litterfall and C input through decomposition

These annual C input from litterfall and fine root production values were obtained from data collected over a 24-month sampling period, which permitted to take into consideration the effect of periodical weather patterns and forest management activities.

Measurement of litterfall, fine root production and cattle feces input helped to quantify the annual C input to soil C pool through decomposition processes. As time constraints impeded the realization of long-term decomposition experiments, the annual mass losses observed *in-situ* were complemented by data from the literature. Considering a maximum mass loss of 84.5% after a period of six years (Berg and Laskowski, 2006), the amount of C incorporated into the stable soil C pools will be 0.04 Mg C ha<sup>-1</sup> y<sup>-1</sup> in the Silvopasture and 0.05 in the Plantation. Notwithstanding, these values do not consider accumulated litterfalls from the previous years that are gradually decomposing and also being added to the soil C pool. Therefore, the sum of net annual additions to stable soil C pools were 0.24 and 0.29 Mg C ha<sup>-1</sup> y<sup>-1</sup> for the Silvopasture and Plantation, respectively.

The larger belowground biomass yield usually results in greater C addition to soil C pool, as reflected in the Prairie and Silvopasture. Gordon and Thevathasan (2005) reported net additions of 0.99 Mg C ha<sup>-1</sup> y<sup>-1</sup> in a poplar-based silvopastoral system and a ryegrass prairie in southern Canada, in agreement with the values indicated for the Silvopasture and Prairie. The differences observed could be explained by the larger grass root input in the Prairie and lower input in the Silvopasture, since approximately 25% of the spatial area is occupied by trees, whereas only 16% of the area was occupied by ryegrass in the poplar silvopastoral system.

#### 4.4.3. Greenhouse gases mitigation potential

##### 4.4.3.1. Stocking rates

Using net C sequestration values from three tested ecosystems and Global Warming Potential (GWP) values for methane and nitrous oxide, it is possible to determine how many cows per

hectare could potentially be grazed in the Prairie and the Silvopasture without resulting in net C emissions to the atmosphere. The net C sequestration values were 3.80 and 1.09 Mg C ha<sup>-1</sup> y<sup>-1</sup> for the Silvopasture and Prairie, respectively. Based on these net C sequestration values, the number of cattle that can be “C-neutrally grazed” was 5 cows per ha in the Silvopasture and only 2 in the Prairie. This is ten and four times more than the actual stocking rate of 0.5 cows per ha in the Silvopasture in Prairie, respectively (Sotomayor et al., 2009), due to the incorporation of trees into the system.

The actual stocking rate could be increased provided pasture production can support the new stocking density. It must be noted however that these maximal stocking densities are derived based on results from this study, where total C tree uptake (assimilated in woody components and returned to soil via litterfall and fine root turnover) represents the mean above and belowground C sequestration rate over 18 years during the 1991-2003 and 2003-2009 growth periods. The actual stocking rate of cattle would probably be lower during pine establishment and senescence. Also, as the trees mature, their crowns become larger and will create more shadow on the border of the strip, which may affect pasture growth at the edges of the alleys and thereby can reduce the stocking density due to lower pasture production. Higher stocking rates can also be maintained with grazing of goats, sheep or horses, given their lower individual GHG emissions (Yang et al., 2003).

The stocking rate affects the C budget because CH<sub>4</sub> emissions by ruminants increase with the number of grazing animals. Digestible C losses of 5% occur due to the CH<sub>4</sub> emissions from enteric fermentation, which contributes between 16 and 23% of global CH<sub>4</sub> emissions (Soussana et al., 2004). Well-managed prairies, using better quality grasses that increase the digestive efficiency will reduce CH<sub>4</sub> emissions because the food remains less time in the rumen, producing less CH<sub>4</sub> (DeRamus et al., 2003). A silvopastoral system with low input sustainable practices, which minimize vegetative and soil disturbances, promote the presence of perennial vegetation, recover or recycle emissions, and will contribute to the preservation of C and N pools during decades or centuries (Lal, 2005).

#### 4.4.3.2. Land area under silvopastoral management

The results from this investigation and published reports permit the estimation of the total land area required under silvopastoral management so that cattle raising could become C neutral in the Chilean Patagonia. There are approximately 260,967 cows in Patagonia, out of which 199,284 are found in the regions of Aysén (O. Teuber, pers. comm., 2010) and 61,683 in Magallanes (INE, 2007a). Using the same net C sequestration data, and considering that more than 3 million ha are either abandoned or degraded land resulting from severe forest fires in the last century, only 48,127 ha (or 481.27 km<sup>2</sup>) under silvopastoral systems (Silvopasture) with cattle would be needed in the Chilean Patagonia to offset all C losses from cattle-based livestock systems and become C neutral, out of which 36,752 and 11,375 ha are in the Aysén and Magallanes Regions, respectively. Since the Aysén and Magallanes Regions (INE, 2007b) cover an area of 108,494.4 and 38,400.8 km<sup>2</sup>, respectively, the total area needed would be only 0.33%. However, the land area required using a natural prairie (Prairie) approach climbs to a total of 167,783 ha, where 128,125 ha are in Aysen and 39,658 ha in Magallanes. That is 3.5 times more land area required as monoculture pasture systems, and represents 1.14% of the Chilean Patagonia. However, one should note that these required areas only consider silvopastoral systems with cows; smaller areas would be needed if sheep only were grazed, but larger areas would be necessary if both cattle and sheep were included. In addition, it is worth mentioning that CH<sub>4</sub>, CO<sub>2</sub> and N<sub>2</sub>O emissions contribute to 46, 39 and 15 %, respectively, to the GWP from the areas mentioned above.

On the other hand, the use of faster-growing species well adapted to Patagonian conditions, such as *Populus trichocarpa* (A. Sotomayor, pers. comm., 2009) could result in better short-term C storage or greater C storage than pines at rotation age. In addition, poplar leaves may annually release more N to the soil than pine needles (Thevathasan and Gordon, 1997) because of the different substrate quality and could enhance pasture growth. In addition, based on this study, and knowing the numerous benefits of agroforestry for soil conservation (Gordon et al., 2009), it may be a policy option to be considered to introduce tree-based pasture systems in Patagonia as the monoculture pasture systems (Prairie), regrettably, are currently acting as a C source.



## 4.5. Conclusions

Several recent studies have shown that temperate agroforestry systems have greater C sequestration potential than forest plantations, grasslands, or sole cropping systems. In the Chilean Patagonia, the adoption of Silvopasture appears to be a sustainable land-use management practice that preserves and increases soil C and N pools, contributes to reduce atmospheric CO<sub>2</sub>, and permit to offset GHG emissions from animal grazing and fertilizer applications, which could convert the entire region into effective C sinks. Our results indicate that individual trees in the Silvopasture are using more efficiently the site resources and have sequestered almost 30 % more C in total above and belowground biomass when compared with trees in the Plantation. Sustainable increase in tree density could enhance C sequestration and have an added benefit in terms of biomass production for bioenergy.

As the thinning operation resulted in higher C sequestration rates in the Silvopasture, any new establishment of silvopastoral systems in the region may follow the recommendations from this study in terms of C sequestration. We can expect significant gains in SOC in the future resulting from remaining pine stumps and coarse-root decomposition. The aboveground: belowground C pools ratio show the preponderance of SOC pools belowground and the key role played by volcanic soils in the capture of large amounts of C. Besides the higher stem density in the Plantation, the synergistic effect resulting from the combination of trees and pasture led to more C being sequestered in the Silvopasture soil.

The C fluxes suggest that the Plantation can annually sequester only 40% more C than the Silvopasture, in spite of twice as much tree density. In the Silvopasture and Plantation systems, the net C fluxes were positive, which indicate that these systems are true C "sinks". A negative net C flux in the Prairie system indicates that this system is a net C "source" to the atmosphere. Based on this study, the actual cattle stocking rate could be increased to 5 cows per ha in the Silvopasture and only 2 in the Prairie in order to be "C-neutrally grazed", provided pasture production can support the new stocking density. On the other hand, only 481 km<sup>2</sup> under Silvopasture would be needed to offset all C losses from cattle-based livestock systems in the Chilean Patagonia and become C neutral. Given that large deforested areas are

currently subject to soil erosion coupled with poor and inferior quality pasture production, the adoption of Silvopasture over large tracts of grazing lands should not be a problem in Patagonia nor a threat to other types of land uses. As the Prairie is acting as a C source, pine-based silvopastoral systems could contribute enormously towards Chilean strategies to mitigate climate change.

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#### **4.7. Appendix A**

The following assumptions and calculations were used in order to build the models of C pools and fluxes within the pine-based silvopastoral system arranged in strip (Silvopasture), the 18-year old *Pinus ponderosa* plantation (Plantation), and the managed natural prairie (Prairie):

1) The Silvopasture and Plantation have a tree density of 400 and 800 stems ha<sup>-1</sup>, respectively. Pine strips in the Silvopasture occupy 22% of the area available for pasture and have never been thinned. All trees were pruned to heights of 2.8 and 4 m in 2006 and 2009, respectively.

2) Using inventory data obtained since the establishment of the pine plantations, the mean above and belowground C sequestration rates were calculated for the 1991-2003 period at the initial tree density, and then for the 2003-2009 periods after thinning to a density of 800 trees ha<sup>-1</sup> in the Plantation and 400 trees ha<sup>-1</sup> in the Silvopasture.

3) Aboveground tree C pools include trunks, branches, twigs, needles and cones. Belowground C pools include thick roots superior to 5 mm diameter.

4) It was assumed that cattle will consume most of the aboveground pasture biomass produced during the year in the Silvopasture and Prairie (part of it returning to the system as feces, methane, nitrous oxide and respiration) and that only belowground biomass C will be added to the soil C pool. The aboveground grass biomass present in the Plantation remains in the system as no grazing occurs therein.

The C content of aboveground net primary productivity (ANPP) was determined after manually harvesting grazing material three times a year over a two-year period. Since it was not possible to measure on site belowground net primary productivity (BNPP), it was estimated with a known algorithm (Gibson, 2009). The C content of BNPP and net annual C sequestration by pasture alone could then be calculated, considering a 40% addition to the recalcitrant soil C pool (Falk, 1976).

5) Knowing the stocking rates and the amount of feces produced annually and C content, the cattle respiration (kg CO<sub>2</sub> ha<sup>-1</sup> y<sup>-1</sup>), methane emissions from enteric fermentation, and nitrous oxide emissions from dung and urine patches, as well as their CO<sub>2</sub>-equivalents (IPCC, 2001) were calculated using data published by Flessa et al. (2002), Yang et al. (2003) and Byrne et al. (2007). Carbon dioxide, CH<sub>4</sub> and N<sub>2</sub>O emissions from a single animal are estimated to be 996, 56 and 1.29 kg head<sup>-1</sup> y<sup>-1</sup>, respectively, and depend on the amount and kind of feed that is consumed. The reference weight per head-unit is 500 kg. CO<sub>2</sub>-equivalents are calculated using the Global Warming Potentials (GWP), which determine the relative contribution of a given gas to the greenhouse effect. The GWP values represent how many times more deleterious than CO<sub>2</sub> in a 100-year period are CH<sub>4</sub> (21) and N<sub>2</sub>O (310) in terms of global warming.

6) The annual mass loss values of cattle feces obtained after a 12-month litterbag decomposition experiment made possible the calculation of net C additions to soil C pools, considering that 2.2 and 2.5 years are required to get a maximum decomposition in the Silvopasture and Prairie, respectively. The net addition to soil C pool in each treatment represents therefore the sum of annual C incorporations over these periods. Hirata et al. (2009) reported similar results, where cattle dung reached a decomposition of 79.1% after 2.2 years.

7) Annual litterfall and needle decomposition in the Plantation and the Silvopasture were obtained from field measurements over a two-year period to illustrate the importance of annual C inputs and net additions to soil C pools. It was assumed that annual fine root C turnover in pines is 30 % of litterfall (Abohassan, 2004).

8) Since time constraints did not permit to undertake a long-term experiment for the decomposition of the ponderosa pine needles, a maximum mass loss of 84.5% for Scots pine needles in Scandinavia was assumed to be representative of the situation, considering the similar climatic conditions encountered and values of initial N and lignin contents found in green litter (Berg and Laskowski, 2006; Dube F., unpublished data, 2010). Theoretically, the C contribution to soil from litterfall and fine root turnover for the last 18 years was 25 and 32 Mg C ha<sup>-1</sup> in Silvopasture and Plantation, respectively. However, an average of 14.2% in the Silvopasture and 14.5% in the Plantation of the C added annually via litterfall and fine roots was released back into the atmosphere through microbial decomposition.

Based on the results of the decomposition experiment, approximately six years (84.5% / 14.2% y<sup>-1</sup> in the Silvopasture and 84.5% / 14.5% y<sup>-1</sup> in the Plantation) would be required to obtain maximum needle decomposition. This represents 0.20 Mg C ha<sup>-1</sup> y<sup>-1</sup> (25 Mg C ha<sup>-1</sup>/18 yrs \* 14.2% y<sup>-1</sup>) in the Silvopasture and 0.26 Mg C ha<sup>-1</sup> y<sup>-1</sup> (32 Mg C ha<sup>-1</sup>/ 18 yrs \* 14.5% y<sup>-1</sup>) in the Plantation that are lost due to decomposition. Considering a maximum mass loss after a period of six years, the amount of C incorporated into the stable soil C pools will be 0.04 Mg C ha<sup>-1</sup> y<sup>-1</sup> (25 Mg C ha<sup>-1</sup>/18 yrs \* 15.5% /6 yrs) in the Silvopasture and 0.05 (32 Mg C ha<sup>-1</sup>/18 yrs \* 15.5% /6 yrs) in the Plantation. However, these values represent what is lost and gained from the annual litterfall, and do not consider accumulated litterfalls from the previous years

that are gradually decomposing and also being added to the soil C pool. Taking this process into account, the sum of annual losses as decomposition during the six year period in the Silvopasture and Plantation reached 1.2 and 1.6 Mg C ha<sup>-1</sup> y<sup>-1</sup>, respectively, whereas the net annual additions to stable soil C pools were 0.24 and 0.29 Mg C ha<sup>-1</sup> y<sup>-1</sup> for the Silvopasture and Plantation, respectively.

9) With respect to decomposition of necromass and coarse woody debris (CWD), the annual mass loss was determined using published  $k$  values (y<sup>-1</sup>) for decomposition of dead branches of *Pinus ponderosa* (Hart et al., 1992; Yin, 1999; Hall et al., 2006) and boles/stumps of *Nothofagus pumilio* (Frangi et al., 1997). Knowing the dry weight of dead branches after two years of decomposition and CWD, and assuming a 95% loss of initial weights, it was possible to calculate their mass losses and net addition to soil C pools. It should be noted that decomposition of the duff needle layer is not considered here as it has already been accounted for in the calculation of annual litterfall decomposition.

10) Soil C sequestration for the upper 0-40 cm layer was determined using weighted averages of C contents at three measured depths and a bulk density of 0.9 g cm<sup>-3</sup> (Dube et al., 2009). In the silvopastoral system, an average value was calculated from the C contents obtained within the tree strips and at 2.5 m intervals on either side of strips (up to 10.5 m, corresponding to the middle of the 21-m wide pasture strip).

11) Soil respiration values refer to total respiration, including tree root, mycorrhizae and microbial respiration, and annual decomposition losses of needles, fine roots, cattle feces, necromass and coarse woody debris. Annual soil respiration for the three ecosystems was calculated from the monthly respiration rates presented in this study. For the months that soil respiration was not measured, estimates were done as follows: A regression between soil respiration and air temperature (+5 cm) was adjusted for every treatment ( $R^2 = 0.94$ ), using the values obtained in the field. Knowing the mean monthly superficial air temperatures, these equations were then used to estimate monthly soil respiration and check the values calculated initially, the differences being less than 5%. Within the Silvopasture, it was assumed that soil respiration in the tree strip accounts for 22% from the spatial area, while respiration from 1

and 7.5 m from the tree strip accounts for 78%. Since respiration chambers were installed within the pine strips, at 1 m and at 7.5 m from the strips, tree roots growing into the grass band could be taken into consideration in the calculations.

12) In order to determine the annual amount of leaching C, it was assumed that 24% of the annual rainfall leaches to the ground water (Gisi, 1997; Peichl et al., 2006). Annual rainfall at the research site is 1206 mm out of which 290 mm ha<sup>-1</sup> y<sup>-1</sup> is lost as leaching. The mean annual total C concentrations of leached soil solution from the land-uses were then used to estimate the annual leached C losses in conjunction with total annual leaching losses. As above, C leaching within the tree strip was assumed to account for 22% of the spatial area, whereas leaching from 1 and 7.5 m from the tree strip account for 78% of the area.

13) Since approximately 1.25% of N fertilizer applied to the soil is lost in the form of N<sub>2</sub>O emissions (IPCC, 1997), and knowing the amount of N-fertilizer applied to the pasture every 3 years, annual emissions of N<sub>2</sub>O and CO<sub>2</sub>-equivalent were estimated.

14) Carbon storage in ecosystems pools was calculated using the following equation:

$$C_{\text{pools}} = C_{\text{agt}} + C_{\text{bgt}} + C_{\text{agg}} + C_{\text{soil}},$$

Where  $C_{\text{pools}}$  = total carbon stored in ecosystem pools,  $C_{\text{agt}}$  = aboveground tree carbon,  $C_{\text{bgt}}$  = belowground tree carbon,  $C_{\text{agg}}$  = aboveground grass carbon in the Ponderosa pine plantation and  $C_{\text{soil}}$  = soil organic carbon pool.

15) Positive or negative carbon flux into or out of the ecosystems was calculated using the following equation:

$$C_{\text{flux}} = C_{\text{TrU}} + C_{\text{GrU}} + C_{\text{AtD}} + C_{\text{FecS}} + C_{\text{CwdS}} - C_{\text{SRes}} - C_{\text{Lch}} - C_{\text{Fert}} - C_{\text{AnC}},$$

Where  $C_{\text{flux}}$  = net carbon flux in the ecosystem,  $C_{\text{TrU}}$  = carbon input via total tree uptake,  $C_{\text{GrU}}$  = carbon input via total grass uptake,  $C_{\text{AtD}}$  = carbon input through atmospheric depositions (rain and snow),  $C_{\text{FecS}}$  = net addition to soil carbon pool via feces input,  $C_{\text{CwdS}}$  = net addition to soil carbon pool via coarse woody debris and necromass decomposition,  $C_{\text{SRes}}$  = carbon output via total soil respiration,  $C_{\text{Lch}}$  = carbon leachate output from the soil solution,  $C_{\text{Fert}}$  = volatile carbon-equivalent output from fertilizer application, and  $C_{\text{AnC}}$  = carbon output through pasture

consumption by animals (divided between cattle fattening, feces production and GHG emissions). Therefore, losses as animal respiration, CH<sub>4</sub> emissions from enteric fermentation and N<sub>2</sub>O from feces have already been accounted for in cattle consumption

#### 4.8. References

- Abohassan R.A. 2004. Carbon dynamics in a temperate agroforestry system in southern Ontario, Canada. M.Sc. Thesis. Dept. Envir. Biol., University Guelph, Guelph, 122 pp.
- Bambrick A.D., Whalen J.K., Bradley R.L., Cogliastro A., Gordon A.M., Olivier A., Thevathasan N.V. 2010. Spatial heterogeneity of soil organic carbon in tree-based intercropping systems in Quebec and Ontario, Canada. *Agroforest. Syst.* 79: 343-353.
- Berg B. and Laskowski R. 2006. Litter decomposition: a guide to carbon and nutrients turnover. *Adv. Ecol. Res.*, vol. 38, Elsevier Acad. Press, San Diego, USA, 428 pp.
- Buol S.W., Hole F.D., McCracken R.J., Southard R.J. 1997. Soil genesis and classification. 4th ed., Iowa State Univ. Press, Ames, Iowa, 527 pp.
- Byrne K.A., Kiely G., Leahy P. 2007. Carbon sequestration determined using farm scale carbon balance and eddy covariance. *Agric., Ecosyst. and Environ.* 121: 357-364.
- DeRamus H.A., Clement T., Giampola D., Dickison P. 2003. Methane emissions of beef cattle on forages: efficiency of grazing management systems. *J. Environ. Qual.* 32: 269–277.
- Dixon R.K., Wimjum J.K., Lee J.J., Schroeder, P.E. 1994. Integrated systems: assessing of promising agroforest and alternative land use practices to enhance carbon conservation and sequestration. *Clim. Change* 30: 1-23.
- Dube F. 2010. Carbon dynamics and other interactions in *Pinus ponderosa* (Dougl. Ex Laws) and natural prairie agroecosystems established on degraded volcanic soils in the Aysén Region, Chilean Patagonia. Ph.D. Thesis (in preparation), Faculty of Forest Sciences, University of Concepcion, Concepcion, VIII Region, Chile.
- Dube F., Zagal E., Stolpe N., Espinosa, M. 2009. The influence of land use change on the organic carbon distribution and microbial respiration in a volcanic soil of the Chilean Patagonia. *Forest Ecol. Manage.* 257: 1695-1704.
- Edwards N.T. 1982. The use of soda-lime for measuring respiration rates in terrestrial systems. *Pedobiologia* 23: 321-330.

- Falk J.H. 1976. Energetics of a suburban lawn ecosystem. *Ecology* 57: 141-150.
- Flessa H., Ruser R., Dörsch P., Kamp T., Jimenez M.A., Munch J.C., Beese, F. 2002. Integrated evaluation of greenhouse gas emissions (CO<sub>2</sub>, CH<sub>4</sub>, N<sub>2</sub>O) from two farming systems in southern Germany. *Agric., Ecosyst. and Environ.* 91: 175-189.
- Frangi J.L., Richter L.L., Barrera M., Aloggia M. 1997. Decomposition of *Nothofagus* fallen woody debris in forests of Tierra del Fuego, Argentina. *Can. J. For. Res.* 27: 1095–1102.
- Gayoso J., Guerra J., Alarcón D. 2002. Contenido de carbono y funciones de biomasa en especies nativas y exóticas. Proyecto FONDEF D98I1076, Informe Técnico, Univ. Austral de Chile, Valdivia, Chile, 157 pp. (in Spanish).
- Gibson D.J. 2009. *Grasses and grasslands ecology*. Oxford Univ. Press. New York, NY, USA, 313 pp.
- Gisi U. 1997. *Bodenökologie*. 2. Auflage. Georg Thieme Verlag, Stuttgart, Germany.
- Gordon A.M. and Thevathasan N.V. 2005. How much carbon can be stored in Canadian agroecosystems using a silvopastoral approach? In: *Silvopastoralism and sustainable land management*. Mosquera-Losada M.R., Rigueiro-Rodriguez, A. McAdam J. (Eds) CABI Publishing, Wallingford, UK, pp. 210-218.
- Gordon A.M., Thevathasan N.V., Nair P.K.R. 2009. An agroecological foundation for temperate agroforestry. In: *North American Agroforestry: An integrated science and practice*, 2nd edition, H.E. Garrett (ed.) Madison, USA.
- Hall S.A., Burke I.C., Hobbs N.T. 2006. Litter and dead wood dynamics in Ponderosa pine forests along a 160-year chronosequence. *Ecol. Applic.* 16(6): 2344-2355.
- Hart S.C., Firestone M.K., Paul E.A. 1992. Decomposition and nutrient dynamics of ponderosa pine needles in a Mediterranean-type climate. *Can. J. For. Res.* 22: 306-314.
- Hirata M., Hasegawa N., Nomura M., Ito H., Nogami K., Sonoda T. 2009. Deposition and decomposition of cattle dung in forest grazing in southern Kyushu, Japan. *Ecol. Res.* 24: 119-125.
- Huygens D., Boeckx P., Van Cleemput O., Oyarzun C., Godoy R. 2005. Aggregate and soil organic carbon dynamics in South Chilean Andisols. *Biogeosciences* 2: 159-174.
- INE. 2007a. VII Censo agropecuario y forestal. Existencia de ganado en las explotaciones agropecuarias y forestales por especie. Santiago, Chile. <http://www.censoagropecuario.cl/noticias/09/07042009.html> (accessed 20 August 2010).



- INE. 2007b. División Político-Administrativa y Censal, 2007. 2007b. [http://www.ine.cl/canales/chile\\_estadistico/territorio/division\\_politico\\_administrativa/pdf/DPA\\_COMPLETA.pdf](http://www.ine.cl/canales/chile_estadistico/territorio/division_politico_administrativa/pdf/DPA_COMPLETA.pdf) (accessed 18 October 2010).
- IPCC. 1997. Revised 1996 IPCC guidelines for national greenhouse gas inventories. IPCC/OECD/IEA, IPCC, Geneva, Switzerland.
- IPCC. 2001. Climate change 2001: the scientific basis. In: Houghton J.T., Ding Y., Griggs D.J., Noguer M., van der Linden P.J., Dai X., Mashell K., Johnson C.A. (Eds.) Contribution of Working Group I to Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univ. Press, Cambridge, UK, 881 pp.
- IPCC. 2007. Climate change 2007: synthesis report. [http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4\\_syr.pdf](http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr.pdf). Accessed 6 October 2010.
- Kimmins J.P. 2004. Forest ecology: a foundation for sustainable forest management and environmental ethics in forestry. 3rd Ed. Prentice Hall, Upper Saddle River USA 611 pp.
- Lal R. 2005. Forest soils and carbon sequestration. For. Ecol. Mgmt. 220: 242-258.
- Montagnini F. and Nair P.K.R. 2004. Carbon sequestration: an underexploited environmental benefit of agroforestry systems. Agroforest. Syst. 61: 281-295.
- Peichl M., Thevathasan N.V., Gordon A.M., Huss J., Abohassan R. 2006. Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. Agroforest. Syst. 66: 243-257.
- Raich J.W. and Tufekcioglu A. 2000. Vegetation and soil respiration: Correlations and controls. Biogeochem. 48: 71-90.
- Reay D.S. and Grace J. 2007. Carbon dioxide: Importance, sources and sinks. In: Greenhouse gas sinks. Reay D.S., Hewitt C.N., Smith K., Grace J. (Eds.) CAB International, Wallingford, UK, 289 pp.
- Ross D.J., Tate K.R., Scott N.A., Wilde R.H., Rodda N.J., Townsend J.A. 2002. Afforestation of pastures with *Pinus radiata* influences soil carbon and nitrogen pools and mineralization and microbial properties. Aust. J. Soil Res. 40: 1303-1318.
- SAS Institute Inc., 2003. SAS User's guide: Statistics. SAS Institute Inc., Cary, NC, USA.
- Sharrow S.H. and Ismail S. 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. Agroforest. Syst. 60: 123-130.

- Silva F. 2004. Final report on the ecological status of the Aysén Region. SAG – Agricultural Ministry, Government of Chile, Santiago, Chile, 143 pp. (in Spanish).
- Sotomayor A., Teuber O., Moya I. 2009. Resultados y experiencia sobre manejo de sistemas silvopastorales en la Region de Aysén. In: *Sistemas agroforestales para la región de Aysén*. Teuber O. (ed.) INIA-INFOR, Coyhaique, Chile, 211 pp. (in Spanish).
- Soussana J.P., Loiseau P., Vuichard N., Ceschia E., Balesdent J., Chevallier T., Arrouays D. 2004. Carbon cycling and sequestration opportunities in temperate grassland. *Soil Use Manage.* 20: 219-230.
- Stamps W.T. and Linit M.J. 1999. The problem of experimental design in temperate agroforestry. *Agroforest. Syst.* 44: 187-196.
- Stolpe N.B., Dube F., Zagal E. 2010. Calibration of CO2FIX to native forest, pine plantation, and pasture on a volcanic soil of the Chilean Patagonia. *Acta Agriculturae Scandinavica, Section B – Soil and Plant Science* 60(3): 235-244.
- Tans P. 2010. Trends in atmospheric carbon dioxide – Mauna Loa. NOAA/ESRL. <http://www.esrl.noaa.gov/gmd/ccgg/trends/>. Accessed 6 October 2010.
- Teuber O. and Ganderats S. 2009. Características geográficas y edafoclimáticas de la región de Aysén. In: *Sistemas agroforestales para la región de Aysén: Cortinas cortaviento y silvopastoreo*. Teuber O. (ed.) INIA-INFOR, Coyhaique, Chile, 211 pp.
- Thevathasan N.V. and Gordon A.M. 1997. Poplar leaf biomass distribution and nitrogen dynamics in a poplar-barley intercropped system in southern Ontario, Canada. *Agroforest. Syst.* 37: 79-90.
- Undurraga P., Zagal E., Sepúlveda G., Valderrama N. 2009. Dissolved organic carbon and nitrogen in Andisol for six crop rotations with different soil management intensity. *Chilean J. Agric. Res.* 69(3): 445-454.
- Woignier T., Pochet G., Doumenc H., Dieudonné P., Duffours, L. 2007. Allophane: a natural gel in volcanic soils with interesting environmental properties. *J. Sol-Gel Sci. Techn.* 41:25-30.
- Yang S.S., Liu C.M., Liu Y.L. 2003. Estimation of methane and nitrous oxide emission from animal production sector in Taiwan during 1990-2000. *Chemosph.* 52: 1381-1388.
- Yin X. 1999. The decay of forest woody debris: numerical modeling and implications based on some 300 data cases from North America. *Oecologia* 121: 81-98.

## DISCUSION GENERAL

En los Capítulos 1 y 2 se determinó que los diferentes usos del suelo (pradera natural - PST, plantación de *Pinus ponderosa* - PPP, bosque de *Nothofagus pumilio* - NPF) han afectado la formación de materia orgánica (calidad de substratos disponibles, distribución de los ingresos de C), el C orgánico del suelo (SOC) y la biomasa microbiana C, los que terminan afectando la respiración microbiana del suelo. Los diferentes usos del suelo también han modificado el contenido de C en la biomasa vegetal. Los resultados demuestran la capacidad que este tipo de suelo volcánico tiene para almacenar grandes cantidades de C. Por otro lado, se ha encontrado que la protección física del SOC proporcionada por los agregados del suelo ha permitido mejorar el secuestro de C a largo plazo. La mayor cantidad de SOC se estabilizó en la fracción intermedia de la materia orgánica (mesoagregados) del ecosistema menos perturbado, el bosque nativo (NPF), seguido de la plantación de pino (PPP). A través de los años, los mesoagregados estables formados en suelos no perturbados por la erosión, la labranza y la cosecha mecanizada, pueden comportarse como verdaderos sumideros de C (Six et al., 2002).

La alta calidad de la materia orgánica del suelo (SOM) en NPF promueve el crecimiento y productividad de las plantas, dando como resultado más secuestro de C en la vegetación y, en última instancia, la sustentabilidad del ecosistema a largo plazo. Aunque no ha sido posible estudiar detalladamente la calidad y distribución de la SOM en el sistema silvopastoral con pino (SPS), los resultados obtenidos sugieren que la interacción de árboles y pasto leguminoso en una misma área podría permitir la optimización de la biomasa microbiana C y la respiración microbiana y, probablemente, la formación de mesoagregados estables en el suelo. Según Sharrow e Ismail (2004), existe una sinergia resultante de la interacción de los ciclos de C aéreos y subterráneos de los árboles y el pasto.

La simulación del C del suelo utilizando el modelo CO2FIX demostró que los contenidos de C en estos suelos volcánicos fueron cuatro a ocho veces más bajo que aquellos medidos experimentalmente en NPF, PPP y PST. La subestimación se atribuye a las altas tasas

predefinidas de descomposición de la litera vegetal en suelos no volcánicos del sub modelo YASSO, junto con datos climáticos típicos (Liski et al., 2005). Sin embargo, los suelos volcánicos bajo estudio tienen altos contenidos de materia orgánica. Al disminuir al mínimo la tasa de descomposición de los compuestos orgánicos solubles en el suelo, los contenidos de C simulados fueron más cercanos, pero aún 11% más bajos que los valores medidos en laboratorio. Además, la recalibración sobrestimó la proporción de compuestos solubles de la SOM total (Stolpe et al., 2010). En consecuencia, los resultados del Capítulo 2 llevan a la aceptación de la Hipótesis N°4, la cual estipula que dada la presencia de suelos volcánicos, CO2FIX subestimaré la cantidad de C secuestrado.

La adopción de SPS es una práctica sustentable que optimiza la productividad del suelo, preserva y aumenta los reservorios de C y N, y también contribuye con la reducción de CO<sub>2</sub> atmosférico. Los árboles en SPS en particular han secuestrado casi un 30% más de C en la biomasa total que los árboles en PPP (Capítulo 3). Las mayores dimensiones de los árboles en SPS compensan, hasta cierto punto, la menor densidad arbórea, sin olvidar el C adicional secuestrado en los reservorios subterráneos (raíces del pasto y suelo). El crecimiento de los árboles (sobre todo el incremento del diámetro) en SPS aumenta debido al N adicional del suelo que provee el trébol, el que, a su vez, se beneficia del microclima favorable que se crea en las fajas de pasto, dando como resultado mayores cantidades de C secuestrado. Por lo tanto, los resultados planteados en el Capítulo 3 llevan a la aceptación de la Hipótesis N°3b, que sostiene que la mayor cantidad de N en el suelo por acción de las leguminosas permitirá un mayor crecimiento del árbol en SPS y mayor cantidad de C secuestrado por árbol que en PPP.

Aunque la faja de árboles ocupa casi el 25% del área disponible para el pasto en SPS, y a pesar de la sequía ocurrida durante el año 2008, la productividad anual neta de pasto por hectárea no fue significativamente diferente en SPS y PST. Esto demuestra el rol inigualable que juegan los árboles en la creación de un microclima favorable (mayor temperatura del aire durante las estaciones de crecimiento; doble humedad del suelo durante todo el año) para el crecimiento del pasto en el sistema agroforestal (Garrett et al., 2004). Sin embargo, usando como base de comparación el cuadrante de medición de pasto, que permite compensar por el área no disponible para pastura, la productividad anual de pasto es significativamente mayor en SPS.

La conversión de la plantación de pino a SPS dio como resultado más SOC secuestrado a 0-40 cm de profundidad en el suelo, mientras que hubo una pérdida significativa de SOC a la misma profundidad luego de haberse establecido la PPP en la pradera natural. La presencia de trébol desde el establecimiento del SPS resultó en cantidades significativamente mayores de N almacenado en el suelo de SPS comparado con PPP. Según Johnson (1992), se ha demostrado que los contenidos de C y N del suelo pueden aumentar de 20-100% con la presencia de plantas fijadoras de N atmosférico. Por otro lado, los ácidos húmicos marrones que se producen como resultado de la descomposición de la litera de pinos (Stevenson y Cole, 1999) han reducido la actividad de la biomasa microbiana y disminuido el C rápidamente metabolizable en PPP (Ross et al., 2002), lo que conlleva a una reducción de la formación de SOC luego del establecimiento de las coníferas en la pradera natural. Esto corrobora los resultados obtenidos por Dube et al. (2009) quienes encontraron contenidos de biomasa microbiana C hasta 2,5 veces más baja en PPP que en PST a 0-20 cm de profundidad en el mismo sitio de estudio. Por lo tanto, los resultados del Capítulo 3 llevan a rechazar la Hipótesis N°3a, que señala que la mayor cantidad de N en el suelo por acción de forrajeras leguminosas se encontrará en suelos bajo PST.

Así mismo, los resultados del Capítulo 3 han determinado que SOC fue significativamente mayor a 0-20 cm de profundidad en SPS que en PST, lo que lleva a rechazar la Hipótesis N°2. La gran contribución de SOC a 0-20 cm de profundidad en relación al SOC total demuestra la importancia del ciclaje activo de nutrientes en la capa superficial del suelo de todos los tratamientos, en especial PST y SPS. Las mayores concentraciones de N en el suelo están relacionadas con los mayores contenidos de C y ganancias en almacenamiento de SOC, los que resultan en una mejora de la fertilidad del suelo (Bambrick et al., 2010). Debido a sus propiedades únicas, tales como altos contenidos de materia orgánica, baja densidad aparente y presencia de arcillas alofánicas, los suelos volcánicos tienden a capturar mayores cantidades de C que la mayoría de los suelos. Además, las propiedades especiales de los alófanos que contienen los suelos volcánicos, los que retienen grandes cantidades de SOC (Woignier et al., 2007), y la estabilización del C orgánico disuelto controlada por el Al y Fe activos de las cenizas volcánicas (Shoji et al., 1993), podrían ser responsables de las menores pérdida de C en los lixiviados del suelo. En la escala de balances de C, SPS es más eficiente que PPP y PST

para reducir las pérdidas de C a las aguas subterráneas y, de este modo, permite disminuir las cantidades de C soluble en los ecosistemas acuáticos. Esto podría contribuir a reducir las emisiones de N<sub>2</sub>O a la atmósfera, ya que los microbios acuáticos requieren C soluble para realizar la desnitrificación (Stevenson y Cole, 1999).

Los resultados del Capítulo 4 muestran que el almacenamiento aéreo y subterráneo total de C fue de 224, 199 y 177 Mg C ha<sup>-1</sup> en SPS, PPP y PST, respectivamente, y las relaciones de reservorio de C aéreo / subterráneo fueron de 1:10 (SPS), 1:5 (PPP) y 1:177 (PST). Las relaciones demuestran la preponderancia de los reservorios subterráneos de C en el contenido total de C secuestrado en los ecosistemas. La ausencia de árboles explica la alta relación observada en PST. Cuando se comparan los reservorios de C en SPS y PPP, aunque la densidad de árboles en SPS sea sólo el 50% de PPP, la proporción de C aéreo / subterráneo fue más alta en SPS. Como consecuencia, los resultados del Capítulo 4 llevan a la aceptación de la Hipótesis N°1a y al rechazo de la Hipótesis N°1b: la primera (a) que afirma que SPS permitirá una mayor producción anual de biomasa total y secuestro de C que PST, y la segunda (b) que SPS no secuestrará más C que PPP con el doble de densidad.

Los flujos netos positivos de C en SPS y PPP indican que estos ecosistemas pueden funcionar como verdaderos sumideros de C, mientras que el flujo neto negativo de C en PST indica que este sistema actúa como una fuente neta de C. Como consecuencia, la adopción de SPS en grandes áreas con pradera natural en la Patagonia podría contribuir a las estrategias de mitigación del cambio climático de Chile. Por consiguiente, los resultados del Capítulo 4 llevan a la aceptación de la Hipótesis N°5, la cual señala que las emisiones de CH<sub>4</sub> y N<sub>2</sub>O serán máximas en PST y mínimas en PPP, siendo SPS una alternativa interesante para mitigar el efecto neto de los GEI bajo estudio. La presencia de árboles en SPS contribuye a contrarrestar la alta respiración del suelo, gracias a su mayor asimilación de C y menor aporte a la respiración total del suelo que el pasto. Además, la significativa cantidad de madera en descomposición sobre el suelo por la quema histórica del bosque nativo, contribuye a la elevada respiración, la que sobrepasó la capacidad de asimilación de C del pasto en la pradera. Por lo tanto, una gran extensión de la Patagonia (más de un millón de ha), que actualmente se encuentra como pradera natural, estaría actuando como una fuente de C a la atmósfera.

## CONCLUSIONES GENERALES

Prácticas de manejo sustentable que promueven la formación de agregados en el suelo, sobretodo mesoagregados estables, pueden actuar como sumideros eficientes de C por largos períodos. Para ello, es necesario estudiar la calidad y distribución de la SOM en nuevas formas de uso del suelo promisorias para la Patagonia, tal como los sistemas silvopastorales. También evaluar los niveles de biomasa y respiración microbiana a distintas profundidades del suelo, la abundancia natural de  $^{13}\text{C}$  a través del perfil del suelo, y las propiedades químicas del suelo incluyendo el pH,  $\text{NO}_3\text{-N}$ , y P y K disponibles. Los cambios en pH del suelo podrían afectar las poblaciones de hongos en el suelo, lo que, a su vez, puede estar ligado a variaciones en contenidos de C disuelto en lixiviados (W. Blum, comunicación personal, 2010), hipótesis que requiere ser evaluada. No obstante, para comprender por completo la variabilidad espacial del SOC en el campo y determinar con precisión el almacenamiento y ganancias de C en diferentes estudios en el mismo sitio, es necesario revisar la metodología relacionada con el diseño e intensidad de muestreo de suelo.

Nuestra investigación ha demostrado que la recalibración del modelo CO2FIX permite simular mejor los contenidos de C en los suelos volcánicos, pero sobrestima los contenidos de C solubles, al compararlos con los resultados obtenidos por fraccionamiento físico en laboratorio. Futuros estudios deberán considerar el ajuste de otros parámetros del modelo a fin de obtener simulaciones más cercanas a la realidad.

Debido al cambio en las tasas de secuestro de C en los árboles de SPS y PPP después de ralea la plantación inicial de pino, cualquier nuevo establecimiento de SPS en la Región de Aysén puede seguir las recomendaciones de este estudio en términos de secuestro de C y otros beneficios que derivan de él. El establecimiento de sistemas silvopastorales con plántulas mejoradas permitiría ganar varios años de beneficios agroforestales durante la rotación, y evitar los problemas relacionados con el trabajo requerido para convertir las plantaciones de pino y disponer de los desechos de cosecha que podrían formarse en el proceso. Por otro lado,

el establecimiento de SPS a partir del raleo de PPP podría generar grandes ganancias en SOC debidas a la lenta descomposición de los restos de tocones y raíces gruesas. Sin embargo, los disturbios del sitio asociados a la remoción de árboles en los sectores correspondientes a las futuras fajas de pasto pueden afectar la formación de mesoagregados estables y promover la erosión y la respiración adicional del suelo, una salida importante de C (Schlichter y Laclau, 1998). Esta pérdida de C del sistema no ha sido incluida cuando se calcularon los flujos de C y se desarrollaron los modelos de secuestro de C en este estudio, pero debería ser considerada en estudios futuros, especialmente cuando PPP se convierta a SPS a mitad de rotación.

El que cada árbol en SPS secuestre un 30% más de C en la biomasa total implica que un aumento moderado en la densidad arbórea, junto con algunas modificaciones al diseño del sistema, podría aumentar el secuestro de C en los árboles, con el beneficio agregado de la producción de biomasa para distintos usos (e.g. energía). Una mayor densidad arbórea reduciría la contribución de la respiración del pasto a la respiración total del suelo, terminando en una menor respiración total anual en SPS. Lo que tendría un efecto benéfico en los flujos netos de C, haciendo aún más positivo y más cercano al valor obtenido para la PPP (+2.5 Mg C ha<sup>-1</sup> y<sup>-1</sup>), comparado con el valor obtenido en este estudio para SPS (+1.8 Mg C ha<sup>-1</sup> y<sup>-1</sup>).

Basado en los resultados de este estudio, podría incentivarse la introducción de SPS en la Patagonia, ya que los pastizales (sobre un millón de ha) actúan como fuentes de C. El establecimiento de menos de 500 km<sup>2</sup> bajo SPS en praderas degradadas, o la conversión de PPP a SPS permitiría contrarrestar todas las pérdidas de C de los sistemas pecuarios con ganado bovino en la Patagonia chilena. Por otro lado, la introducción del manejo silvopastoral en bosques de *Nothofagus pumilio* puede ser una buena alternativa de manejo para los bosques nativos ubicados cerca de las estancias ganaderas, siempre que los árboles sean raleados para abrir su dosel, y que la pastura leguminosa sea establecida en el sotobosque para optimizar el secuestro de C en los componentes vegetales y el suelo. La utilización de árboles nativos de larga vida en sistemas silvopastorales permitiría el secuestro de grandes cantidades C en la biomasa y el suelo, preservaría la calidad del SOC y mantendría un adecuado balance entre las preocupaciones medioambientales y económicas relacionadas al uso de la tierra en la Región.



## BIBLIOGRAFIA

- Abohassan R.A. 2004. Carbon dynamics in a temperate agroforestry system in southern Ontario, Canada. M.Sc. Thesis. Dept. Environ. Biol., University of Guelph, Guelph, ON, Canada, 122 pp.
- Bambrick A.D., Whalen J.K., Bradley R.L., Cogliastro A., Gordon A.M., Olivier A., Thevathasan N.V. 2010. Spatial heterogeneity of soil organic carbon in tree-based intercropping systems in Quebec and Ontario, Canada. *Agroforest. Syst.* 79: 343-353.
- Bowman A.F. 1990. *Soils and the Greenhouse Effect*. John Wiley & Sons, New York, USA.
- Brady N.C. 1990. *The nature and property of soils*. 10<sup>th</sup> ed. Macmillan Publ. Co., New York, NY, USA, 621 pp.
- Carter M. 2002. Soil quality for sustainable management: organic matter and aggregation indicators that maintain soil functions. *Agronomy Journal* 94: 38-47.
- Dixon R.K., Wimjum J.K., Lee J.J., Schroeder P.E. 1994. Integrated systems: assessing of promising agroforest and alternative land use practices to enhance carbon conservation and sequestration. *Clim. Change* 30: 1-23.
- Dube F., Couto L., Silva M.L., Leite H.G., Garcia R., Araujo G.A. 2002. A simulation model for evaluating technical and economic aspects of an industrial eucalyptus-based agroforestry system in Minas Gerais, Brazil. *Agroforest. Syst.* 55: 73-80.
- Dube F., Zagal E., Stolpe N., Espinosa, M. 2009. The influence of land use change on the organic carbon distribution and microbial respiration in a volcanic soil of the Chilean Patagonia. *Forest Ecol. Manage.* 257: 1695-1704.
- Garrett H.E., Rietveld W.J., Fisher R.F. 2000. *North American Agroforestry: an integrated science and practice*. Am. Soc. Agron., Madison, WI, USA, 401 pp.
- Garrett H.E., Kerley M.S., Ladyman K.P., Walter W.D., Godsey L.D., Van Sambeek J.W., Brauer, D.K. 2004. Hardwood silvopasture management in North America. *Agroforest. Syst.* 61: 21-33.
- Gobierno Regional de Aysén. 2002. *Plan Regional, Ordenamiento Territorial, Región de Aysén*. Gobierno Regional de Aysén, Chile.

- Gordon A.M. y Thevathasan N.V. 2005. How much carbon can be stored in Canadian agroecosystems using a silvopastoral approach? In: *Silvopastoralism and sustainable land management*. Mosquera-Losada M.R., Rigueiro-Rodriguez A., McAdam J. (Eds.), CABI Publishing, Wallingford, UK, pp. 210-218.
- Ingram J.S.I. y Fernandes E.C.M. 2001. Management carbon sequestration in soils: concepts and terminology. *Agric. Ecosyst. Environ.* 87: 111-117.
- IPCC. 1997. Revised 1996 IPCC guidelines for national greenhouse gas inventories. IPCC/OECD/IEA, IPCC, Geneva, Switzerland.
- IPCC. 2001. Climate change 2001: the scientific basis. In: Houghton J.T., Ding Y., Griggs D.J., Noguer M., van der Linden P.J., Dai X., Mashell K., Johnson C.A. (Eds.) *Contribution of Working Group I to Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge Univ. Press, Cambridge, UK, 881 pp.
- IPCC. 2007. Climate change 2007: synthesis report. [http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4\\_syr.pdf](http://www.ipcc.ch/pdf/assessment-report/ar4/syr/ar4_syr.pdf). Accessed 6 October 2010.
- Johnson D.W. 1992. Effects of forest management on soil carbon storage. *Water Air and Soil Pollution* 64: 83-120.
- Liski J., Palosuo T., Peltoniemi M., Sievänen R. 2005. Carbon and decomposition model YASSO for forest soils. *Ecological Modeling*, 189: 168-182.
- Montagnini F. y Nair P.K.R. 2004. Carbon sequestration: an underexploited environmental benefit of agroforestry systems. *Agroforest. Syst.* 61: 281-295.
- Nair P.K.R. y Nair V.D. 2003. Carbon storage in North American Agroforestry Systems. In: Kimble J., Heath L.S., Birdsey R.A., Lal R. (Eds.), *The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect*. CRC Press, Boca Raton, FL, USA, pp. 333-346.
- Oelbermann M., Voroney R.P., Thevathasan N.V., Gordon A.M., Kass D.C.L., Schlonvoigt A.M. 2006. Soil carbon dynamics and residue stabilization in a Costa Rican and southern Canadian alley cropping systems. *Agroforest. Syst.* 68: 27-36.
- Ortega R. y Rodríguez I. 1994. *Manual de gestión del medio ambiente*. Ed. Mapfre, 343 pp.
- Paustian K., Six J., Elliott E.T., Hunt H.W. 2000. Management options for reducing CO<sub>2</sub> emissions from agricultural soils. *Biogeochemistry* 48: 147-163.

- Paustian K., Antle J.M., Sheehan J., Paul E.A. 2006. Agriculture's role in greenhouse gas mitigation. Pew Center on Global Climate Change. Arlington, VA, USA. 87 pp.
- Pandey D.N. 2002. Carbon sequestration in agroforestry systems. *Climate Policy* 2: 367-377.
- Peichl M., Thevathasan N.V., Gordon A.M., Huss J., Abohassan R. 2006. Carbon sequestration potentials in temperate tree-based intercropping systems, southern Ontario, Canada. *Agrof. Syst.* 66: 243-257.
- Ross D.J., Tate K.R., Scott N.A., Wilde R.H., Rodda N.J., Townsend J.A. 2002. Afforestation of pastures with *Pinus radiata* influences soil carbon and nitrogen pools and mineralisation and microbial properties. *Australian Journal of Soil Research* 40: 1303-1318.
- Schelhaas M.J., van Esch P.W., Groen T.A., de Jong B.H.J., Kanninen M., Liski J., Masera O., Mohren G.M.J., Naaburs G.J., Palosuo T., Pedroni L., Vallejo A., Vilen T. 2004. CO2FIX V. 3.1. - Description of a model for quantifying carbon sequestration in forest ecosystems and wood products, ALTERRA Report 1068. Wageningen, The Netherlands, 122 pp.
- Schlichter T. y Laclau P. 1998. Ecotono estepa-bosque y plantaciones forestales en la Patagonia norte. *Ecologia Austral* 8: 285-296. (in Spanish).
- Sharrow S.H. e Ismail S. 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western Oregon, USA. *Agroforest. Syst.* 60: 123-130.
- Shoji S., Nanzyo M., Dahlgren R.A. 1993. Volcanic ash soils - genesis, properties, and utilization. *Develop. soil science* 21. Elsevier, Amsterdam, the Netherlands, 288 pp.
- Silva F., Ahumada F., Cerda J. 1999. Guías de condición para los pastizales de la Ecorregión Templada Intermedia de Aysén. Departamento de protección de los recursos naturales renovables, Servicio Agrícola y Ganadero, Ministerio de Agricultura, Gobierno Regional de Aysén, Chile. 127 pp.
- Six J., Callewaert P., Lenders S., De Gryze S., Morris S.J., Gregorich E.G., Paul E.A., Paustian K. 2002. Measuring and understanding carbon storage in afforested soils by physical fractionation. Division S-7 Forest and Range Soils. *Soil Science Society of America Journal* 66: 1981-1987.
- Smith K.A. y Conen F. 2004. Impacts of land management on fluxes of trace greenhouse gases. *Soil Use Management.* 20: 255-263.
- Stevenson F.J. y Cole M.A. 1999. *Cycles of soil.* 2<sup>nd</sup> Ed., John Wiley & Sons, USA, 427 pp.

- Stolpe N.B., Dube F., Zagal E. 2010. Calibration of CO2FIX to native forest, pine plantation, and pasture on a volcanic soil of the Chilean Patagonia. *Acta Agriculturae Scandinavica, Section B – Soil and Plant Science* 60(3): 235-244.
- Tans P. 2010. Trends in atmospheric carbon dioxide – Mauna Loa. NOAA/ESRL. <http://www.esrl.noaa.gov/gmd/ccgg/trends/>. Accessed 6 October 2010.
- Teuber O. y Ganderats S. 2009. Características geográficas y edafoclimáticas de la región de Aysén. In: *Sistemas agroforestales para la región de Aysén: Cortinas cortaviento y silvopastoreo*. Teuber O. (Ed.) INIA-INFOR, Coyhaique, Chile, 211 pp.
- Thevathasan N.V. y Gordon A.M. 2004. Ecology of tree intercropping systems in the North temperate region: experiences from southern Ontario, Canada. *Agroforest. Syst.* 61: 257-268.
- Watson R.T., Noble I.R., Bolin B., Ravindranath N.H., Verardo D.J., Dokken D.J. 2000. Land use, land-use change, and forestry. IPCC Special Report. Cambridge Univ. Press, New York, USA. [http://www.grida.no/climate/ipcc/land\\_use/index.htm](http://www.grida.no/climate/ipcc/land_use/index.htm). Accessed in May 2005.
- Woignier T., Pochet G., Doumenc H., Dieudonné P., Duffours, L. 2007. Allophane: a natural gel in volcanic soils with interesting environmental properties. *J. Sol-Gel Sci. Techn.* 41:25-30.
- Zagal E. y Córdova C. 2003. Fraccionamiento físico de la materia orgánica en un Andisol cultivado. In: *Manejo Sustentable de Suelos Chilenos*. Vidal I. (Ed.). Universidad de Concepción, Chillán, Chile. pp. 282-290.