UNIVERSIDAD DE CONCEPCIÓN
Dirección De Postgrado
Facultad De Ciencias Forestales
Programa de Magister en Ciencias Forestales

EFECTO DEL REEMPLAZO DE BOSQUES SECUNDARIOS POR PLANTACIONES FORESTALES SOBRE VARIABLES BIOGEOQUÍMICAS Y RESERVORIOS DE CARBONO, NITROGENO Y FÓSFORO EN SUELOS DE MINERALOGÍA CONTRASTANTE DEL CENTRO-SUR DE CHILE.

Tesis para optar al grado de Magister en Ciencias Forestales

Oscar Esteban Crovo Rondanelli
CONCEPCIÓN- CHILE

2020

Profesor Guía: PhD. Felipe Aburto Guerrero
Depto. de Silvicultura,
Facultad de Ciencias Forestales
Universidad de Concepción

EFECTO DEL REEMPLAZO DE BOSQUES SECUNDARIOS POR PLANTACIONES FORESTALES SOBRE VARIABLES BIOGEOQUÍMICAS Y RESERVORIOS DE CARBONO, NITROGENO Y FÓSFORO EN SUELOS DE MINERALOGÍA CONTRASTANTE DEL CENTRO-SUR DE CHILE.

Comisión Evaluadora:

Felipe Aburto Guerrero (Profesor guía)
Ingeniero Agrónomo, PhD. ______________________________

Rafael Rubilar Pons (Profesor co-guía)
Ingeniero Forestal, PhD. ______________________________

Randal Southard (Comisión evaluación)
Biólogo, PhD. ______________________________

Directora de Postgrado:
Darcy Ríos L.
Profesora de Biología y Química, Dra. ______________________________

Decano Facultad de Ciencias Forestales:
Manuel Sánchez O.
Ingeniero Forestal. Dr. ______________________________

DEDICATORIA

A mis padres Aurelio Crovo Asiain y Patricia Rondanelli Izquierdo por su extraordinario e incondicional apoyo que me han brindado siempre.

A mis hermanos Loreto Crovo Rondanelli y Jose Crovo Rondanelli por siempre estar cuando los he necesitados.

A mis abuelos Jose Crovo, Marta Asiain, Rafael Rondanelli y Guillermina Izquierdo por hacer todo esto posible.

A mi pareja Paula Alvarez por su incondicional apoyo en cada minuto de este proceso y su extraordinario amor.

A mi familia que siempre confío en mí.

A mi mejor amigo Igor, de quien tengo los mejores y más bonitos recuerdos. Quien se quedó acompañándome muchas veces hasta altas horas de la madrugada, quien siempre supo sacarme una sonrisa en momentos difíciles y me brindo su amor incondicional.

AGRADECIMIENTOS

A mi profesor guía Felipe Aburto Guerrero, por su apoyo, confianza, comprensión y dedicación durante el desarrollo de mi tesis. También a mi profesor co-guía Rafael Rubilar Pons por su apoyo y consejos que me brindo durante este proceso.

A la CONICYT/FONDECYT por aportar con los fondos para hacer posible esta investigación (proyecto FONDECYT INICIACIÓN 11160372) que permitió desarrollar mi postgrado.

A la corporación nacional forestal (CONAF) por su buena disposición y permiso de realizar parcelas de muestreo en la R.N Nonguén y R.N Malalcahuello. A CMPC por permitir el acceso a su predio de patrimonio para realizar parcelas de muestreo. A Forestal Arauco S.A por permitir el acceso a sus predios de patrimonio para realizar parcelas de muestreo.

Al equipo del Laboratorio de Investigación de Suelos Aguas y Bosques (LISAB) por el apoyo.

A mi familia, por sobre todo a mi padre y madre por su continuo apoyo. A mis amigos y compañeros de laboratorio y estudiaste de pre-grado quienes cooperaron con la recopilación de datos y el establecimiento de las parcelas de muestreo. A mi pareja por su apoyo en cada uno de los distintos procesos de esta tesis y por último a mis perros y a mi gato.

TABLA DE CONTENIDO

ÍNDICE DE TABLAS

ÍNDICE DE FIGURAS

RESUMEN GENERAL

El cambio de uso de suelo es un problema global con enormes implicaciones sociales, económicas y ambientales. Actualmente, muchos países muestran altas tasas de deforestación teniendo un impacto directo en las reservas de C y N del suelo y en los ciclos biogeoquímicos. A pesar de que hay un número considerable de estudios que resaltaron los efectos de la sustitución de bosques en el secuestro de C, el impacto en la estequiometria ecológica y en ciclos biogeoquímicos no se ha evaluado sistemáticamente en diferentes suelos. Para evaluar la magnitud de estas alteraciones, cuantificamos los principales grupos y flujos de C, N y P en parcelas de bosques nativos secundarios y parcelas de plantaciones de pinos exóticos en 5 suelos contrastantes en el centro-sur de Chile. Se establecieron dos parcelas permanentes adyacentes, donde se monitorearon los flujos principales del bosque durante dos años. Cuantificamos la caída y descomposición de la hojarasca, el flujo de CO2 del suelo, el I.A.F, producción anual de raíces, DOC, nitrato, amonio y DTP en lixiviados y biomasa en pie. Los reservorios totales C y N fueron significativamente diferentes entre los tipos de suelo (p = 0.02), pero no entre los tipos de bosque. El mayor promedio de C en todos los suelos se encontró en bosque Nativo (NF) (202.22 ± 82.77 Mg ha-1) en comparación con plantación (PL) (172.55 ± 87.73 Mg ha-1), mientras que el N total mostró el mismo stock tanto en NF (11.9 ± 6.5 Mg ha-1) como para PL (11,9 ± 7,18 Mg ha-1). El stock total de fósforo mostró una interacción significativa entre los factores principales (p = 0.001). Los flujos de entrada no mostraron diferencias estadísticas entre el tipo de bosque (p = 0.13), mientras que los flujos de salida tampoco mostraron diferencias estadísticas entre el tipo de bosque (p = 0.43). El almacenamiento de C en el ecosistema no mostró diferencias entre el tipo de bosque (p = 0.07). Las variables medidas mostraron una correlación positiva y significativa entre ellas, pero solo la producción de raíces mostró una correlación positiva (R2 = 0.49) con el total de C del suelo (p = 0.001

GENERAL ABSTRACT

Land use change is a global issue with tremendous social, economic and environmental implications. Currently, many countries display high rates of deforestation and forest conversion from native forest to industrial tree plantations which have a direct impact on soil C and N stocks and biogeochemical inputs. Even though, there are a considerable number of studies that highlighted the effects of forest substitution on C sequestration, the impact on ecological stoichiometry and biogeochemical cycling has not been assessed systematically across different soils. To evaluate the magnitude of these alterations we quantify the main C, N and P pools and fluxes in pair secondary deciduous native forests and exotic pine plantations plots in 5 contrasting soils in south-central Chile. Two adjacent permanent plots were established, where forest main fluxes was monitored for two years. We quantified litterfall, litter decomposition, soil CO_2 efflux, L.A.I, root annual production, DOC, Nitrate, Ammonium and DTP on leachates and standing biomass. The C and N total pools were significantly different between soil types (p=0.02), but not between forest types. The highest average C stock across all soils was found in native forest (NF) (202.22 ± 82.77 Mg ha^{-1}) compared to plantation (PL) (172.55 ± 87.73 Mg ha^{-1}), while N showed the same stock for NF (11.9 ± 6.5 Mg ha^{-1}) and for PL (11.9 ± 7.18 Mg ha^{-1}). Phosphorus total stock showed a significant interaction between the main factors (p=0.001). Input fluxes did not shown statistically differences between forest type (p= 0.13), while output C fluxes also did not shown statistically differences between forest type (p=0.43). Overall ecosystem C storage (soil and aboveground biomass) showed no differences between forest type (p=0.07). Measured variables showed positive and significant correlation between them but only root production showed a positive correlation (R^2 = 0.49) with soil total C (p=0.001). Root production differences between the forest types were observed where NF showed a significantly higher annual root production (1.76 ± 0.99 Mg ha^{-1}) than PL (0.81 ±0.88 Mg ha^{-1}) (p=0.0001).

I.INTRODUCCIÓN GENERAL

El reservorio de carbono (C) en ecosistemas terrestres depende del equilibrio entre la producción primaria y la descomposición del material orgánico (Schlesinger & Bernhardt, 2013). En este ámbito los ecosistemas forestales juegan un rol preponderante, ya que almacenan alrededor de 1240 Pg de carbono (Dixon et al., 1994) siendo hasta dos veces más que los reservorios atmosféricos (Batjes, 1996; Post et al., 1982). Más de dos tercios de este carbono está contenido en los suelos, lo que los hace el mayor reservorio de carbono terrestre (IPCC, 2013) los cuales a su vez también juegan un importante rol en los ciclos biogeoquímicos globales (Bockheim & Gennadiyev, 2010) como en el ciclo del nitrógeno (N) y fósforo (P) (Schlesinger & Bernhardt, 2013).

El estudio de la biogeoquímica de bosques es una tarea muy laboriosa debido a la gran cantidad de procesos concatenados que determinan la productividad primaria neta de los ecosistemas (Spohn & Sierra, 2018). Frente a esto, la estequiometria ecológica se ha consolidado como un pilar fundamental para el estudio del ciclaje de nutrientes y las limitantes nutricionales en ecosistemas boscosos (Sterner & Elser, 2002;Cleveland & Liptzin, 2007; Müller et al., 2017), ya que busca estudiar de manera conjunta el ciclaje multi-elemental en los ecosistemas forestales (McGroddy et al., 2004; Cleveland & Liptzing 2007; Heuck & Spohn 2016; Feng et al., 2017; Müller et al., 2017).

Dicha productividad está supeditada muchas veces por la disponibilidad de elementos como el N y el P (Elser et al., 2007; LeBaur & Treseder, 2008) y el correcto ciclaje en el material orgánico del suelo (Quan et al., 2014). En base a esto, investigadores han demostrado que una mayor incorporación de nitrógeno a un sistema conlleva a una reducción en las limitaciones de fósforo (Vitousek et al., 2010) debido a un aumento en el ciclaje de P (Olander & Vitousek 2001; Treseder and Vitousek 2001), impulsado por una mayor producción de la enzima fosfatasa

(Liu et al., 2014). A su vez, este aumento en la disponibilidad de P influye también de manera directa en el reservorio de C en el suelo a consecuencia de un aumento en la producción de biomasa (Zhou et al., 2006).

El control de la estabilización y almacenaje en el largo plazo del C se ha visto afectado por procesos antropogénicos como lo son la deforestación y el cambio de uso de suelo, los cuales han generado un desbalance y un declive en los reservorios totales de carbono en el suelo (Lal et al., 2005). Por ejemplo, procesos de habilitación de bosques hacia terrenos agrícolas han generado un agotamiento de hasta un 50% en los reservorios de C en el suelo (Guo & Gifford, 2002). Este declive se debe en gran medida a la baja en la producción de biomasa aérea y subterránea y su posterior descomposición (Lal, 2005). Sin embargo, también se ha estudiado el efecto contrario, donde el abandono de terrenos agrícolas y su posterior forestación con plantaciones, como también el desarrollo de bosques secundarios han generado un incremento en este “stock” de C en el suelo de hasta un 20% en el primer caso y un 50% en el segundo (Guo & Gifford, 2002). Demostrando así la importancia del control en el secuestro de carbono atmosférico por parte de los bosques y la posterior estabilización de este en los suelos forestales (Lal, 2005).

Por otra parte, el efecto del reemplazo de bosque nativo por plantaciones forestales sobre el carbono orgánico del suelo sigue siendo una materia de discusión. Mientras algunos autores indican aumentos en el stock de carbono en el suelo posterior al reemplazo del bosque nativo por plantaciones productivas (Jobaggy and Jackson, 2000; Chen et al., 2005; Fialho and Zinn, 2012), investigaciones más recientes demuestran lo contrario (Guo & Gifford, 2002; Liao et al., 2012; Chen et al., 2016; Cao and Chen, 2017). Esta falta de consenso sobre la influencia del reemplazo forestal en el reservorio de carbono en el suelo ha sido una oportunidad para que nuevas investigaciones aporten con el aumento de bases de datos, como también promover una estandarización en la medición y cuantificación del reservorio total de C en el suelo a nivel global (Eclesia et al., 2012; Liu et al., 2016; Chen et al., 2016).

En Chile, desde 1974, una serie de políticas públicas y subsidios estatales impulsaron una rápida expansión y producción forestal (Nahuelhual et al., 2012) resultando en 1.7 millones de hectáreas plantadas con Pinus sp (i.e *Pinus radiata* D.Don) (INFOR 2008) representando cerca del 17% de la superficie forestal de Chile, la cual se concentra casi exclusivamente en las regiones del centro –sur de Chile. (CONAF, 2011). Esta expansión significó una notable pérdida de bosque nativo en los últimos 15 años del siglo XX y en la primera década del siglo XXI (Echeverria et al., 2006; Aguayo et al., 2009; Miranda et al., 2015; Heilmayr et al., 2016). En la última década, investigadores han centrado sus esfuerzos en dilucidar el efecto del reemplazo forestal en la pérdida de biodiversidad (Altamirano and Lara, 2010), la regulación hídrica (Huber et al., 2010), pérdida de nutrientes y contaminación de cursos de agua (Oyarzun et al., 2007; Perez et al., 2015) y la fragmentación del paisaje (Echeverria et al., 2006). No obstante, el efecto del reemplazo forestal sobre los reservorios de carbono, nitrógeno y fósforo han sido poco estudiados, no solo en Chile sino que también en otras regiones del planeta (Cao and Chen, 2017). A pesar de que existen investigaciones que han marcado un precedente, estas se han centrado en procesos de degradación de bosques y no en el reemplazo del bosque nativo por plantaciones forestales (Staelents et al., 2011; Panichini et al., 2012, Rivas et al., 2012; Neculman et al., 2013). Más aún, casi ninguna investigación ha considerado el tipo de suelo como factor determinante en el almacenaje de carbono y ciclaje de nutrientes, a pesar que la mayoría de la evidencia indica que este es un factor determinante a escala regional (González-Domínguez et al., 2019).

Existe evidencia de que el reemplazo forestal no solo afecta el reservorio total de C en el suelo, sino que también el de nutrientes como el N y el P (Ross et al., 1999; Guan et al., 2015). Por ejemplo, Yan et al., (2008) concluyó que bosques siempreverdes al ser reemplazados por plantaciones monoespecíficas de *Cunninghamia sp (*Abeto chino*)* producen una disminución en el reservorio total de

C y N en el suelo. Esta disminución se atribuyó a la reducción en la entrada de materia orgánica como hojarasca y cambios micro-climáticos producidos por cambios en la composición arbórea (Yan et al., 2008). Por otro lado, Guan et al., (2015) obtuvo resultados que muestran la misma tendencia en bosques subtropicales de China al ser reemplazados por plantaciones forestales, concluyendo que las bajas en el reservorio de C y N se deben a las prácticas de manejo asociadas a la producción forestal (quema de rastrojos y tala-rasa). Ross et al., (1999) algunas décadas atrás evidenció esta baja de C y N en el suelo en bosques templados de Nueva Zelanda al ser reemplazados por plantaciones de *Pinus radiata,* atribuyéndolo a la mayor producción de CO_2 presentada por la plantación en comparación al bosque nativo. El efecto del reemplazo de bosques nativos por plantaciones forestales también afecta al reservorio total de P en el suelo, donde distintos estudios comentan sobre un aumento de este debido a la fertilización fosfatada (Ross et al., 1999; Wang et al., 2017; Cai et al., 2018). Si bien se ha logrado demostrar un descenso en los reservorios de C y N luego del reemplazo forestal (i.e. Quan et al, 2014 and Cao & Chen, 2017), el impacto sobre la estequiometria ecológica y los ciclos biogeoquímicos aún deben ser mejor entendidos.

Si bien queda claro que el reemplazo forestal conlleva prácticas de manejo asociadas como son la fertilización y el laboreo mecánico de suelo, las cuales pueden acelerar la mineralización de C y la pérdida de materia orgánica en el suelo y otros nutrientes (Mancinelli et al., 2010; Peñuelas et al., 2012), aún no es certero que procesos ligados al cambio en la composición arbórea tengan un impacto directo sobre el reservorio de C, N y P en los suelos forestales. Es por ello, que se hace necesario medir variables concomitantes como la producción de hojarasca y raíces, las que pueden ayudar a entender estas pérdidas de C y otros nutrientes en el cambio de bosque nativo a plantaciones forestales (Jobbagy and Jackson, 2000; Kasel & Bennett, 2007; Manzoni et al., 2010; Eclesia et al., 2012; Guan et al., 2015; Heuck & Spohn 2016).

La importancia de los suelos sobre el ciclo biogeoquímico de C y el ciclaje de nutrientes radica en entender la diferencia entre la acumulación y estabilización del C. Mientras la primera está regida por procesos de respiración, la segunda es controlada por la fase mineral del suelo donde el carbono es adsorbido por la superficie reactiva de los minerales (Jandl et al., 2007). Es decir, las propiedades intrínsecas del suelo como lo son la textura y la mineralogía de arcillas supeditan la estabilización fisicoquímica del carbono y la retención de nutrientes (Torn et al., 1997; Six et al., 2002; Denef & Six, 2005; Rasmussen et al., 2006).

Se ha logrado demostrar que suelos abundantes en arcillas de baja actividad (Caolinita y Vermiculita) presentan fuertes cambios en los reservorios de C frente a cambios de uso de suelo (Torn et al., 1997; Six et al., 2002; Lützow et al., 2006). Rasmussen et al. (2006) encontró, mediante el estudio de incubaciones, que suelos formados de un material granítico con minerales secundarios dominados por Vermiculita y Vermiculita con hidróxidos interlaminares, presentan una baja protección de C en comparación a suelos dominados por minerales secundarios de corto rango de ordenamiento derivados de materiales andesíticos. Esto corrobora la importancia de las propiedades intrínsecas del suelo sobre la magnitud de los cambios del reservorio de carbono (Denef et al., 2004).

En consecuencia, de lo anteriormente señalado, es que el presente documento busca cuantificar el efecto del reemplazo forestal en los reservorios de carbono, nitrógeno y fósforo, y sus relaciones estequiométricas en los principales tipos de suelos forestales del centro sur de Chile. Los suelos seleccionados representan una variada gama de propiedades intrínsecas (textura, mineralogía y pH) debido a su contrastante origen. Esta investigación también busca dilucidar cambios en las dinámicas forestales, como la descomposición de materia orgánica, producción de litera y raíces entre otros, inducidos por el reemplazo forestal y en qué medida estos cambios, si es que existen, podrían explicar la modificación de los reservorios de C, N y P. De esta forma, los resultados obtenidos podrían indicar el nivel de sensibilidad de los distintos tipos de suelos a la alteración provocada por el reemplazo del bosque nativo por plantaciones forestales en los reservorios de carbono y nutrientes y las dinámicas biogeoquímicas propias de los bosques templados del centro sur de Chile.

HIPÓTESIS GENERALES

- Los cambios en los reservorios totales de carbono, nitrógeno y fósforo, posterior al reemplazo forestal, son fuertemente controlados por el tipo de suelo y las propiedades intrínsecas de este, donde suelos con baja capacidad de estabilización de carbono debido a texturas gruesas y arcillas de baja reactividad se verán más afectados por el reemplazo forestal.
- El bosque nativo presentará mayores reservorios totales de carbono tanto a nivel aéreo como a nivel de suelo, sin embargo, debido al manejo forestal (i.e. fertilización) las plantaciones presentarán mayores pools de nitrógeno y fósforo totales como también en sus formas disponibles.
- El bosque nativo presentará una mayor productividad, mayores niveles de producción de hojarasca y raíces, repercutiendo así en un mayor reservorio total de carbono.

OBJETIVO GENERAL

- Cuantificar el efecto del tipo de suelo sobre el nivel de alteración de los reservorios de carbono, nitrógeno y fósforo posterior al reemplazo forestal de bosques templados de *Nothofagus sp* por plantaciones de *Pinus sp*.

OBJETIVOS ESPECÍFICOS

- Determinar los reservorios de carbono, nitrógeno y fósforo y sus relaciones estequeométricas tanto en bosque como en plantaciones en los principales suelos forestales del Centro-Sur de Chile.
- Cuantificar los flujos de ganancia y pérdidas en un periodo anual y comparar la distribución de estos bajo distintos manejos forestales.
- Determinar que flujos biogeoquímicos se ven más afectadas por el reemplazo forestal y determinar su impacto en los reservorios totales de carbono, nitrógeno y fósforo.

II. Differential response of soil carbon, nitrogen and phosphorous stocks and available pools to conversion from native forest to exotic plantation in soils of contrasting origin.[1]

Abstract

Land use change (LUC) is a global issue with tremendous social, economic and environmental implications. Currently, many countries display high rates of deforestation and forest conversion from native forest to industrial tree plantations which have a direct impact on soil C and N stocks. Even though, there is a considerable number of studies that highlighted the effects of forest substitution on C sequestration, the impact on ecological stoichiometry and biogeochemical cycling has not been assessed systematically across different soils. Moreover, most studies have concentrated in surface soils disregarding deep SOC pools. The soils considered in this study encompass the main forest soil types found in south central Chile representing a range of soil properties and mineralogy (crystalline to amorphous ash derived soils). To reduce confounding factors due to site history, we exclusively selected pair sampling sites (native versus plantation) that shared a similar land-use history and had close to identical soil and geomorphic conditions in which two independent 625m^2 plots were established at adjacent Native Forests (NF) and Pine Plantations (PL). To determine soil C:N:P inventories alongside N and P available pools, the plot was divided into four sub-quadrants where surface litter layer and bulk soil samples were collected at 6 depth intervals in the central soil pit and in four augers at each quadrant up to a depth of 240 cm. The C and N total pools were significantly different between soil types (p=0.02), but not between forest types. The highest average C stock across all soils was found in NF (202.22 ± 82.77 Mg ha^{-1}) compared to PL (172.55 ± 87.73 Mg ha^{-1}), while N showed the same stock for NF (11.9 ± 6.5 Mg ha^{-1}) and for PL (11.9 ± 7.18 Mg ha^{-1}). Phosphorus total stock showed a significant interaction between the main factors (p=0.001). When comparing each soil type individually, disregarding forest type, the Young Ash soil (Andisol) displayed significantly higher C and N than all the other studied soils. On the contrary, the Recent Ash soil (Entisol) displays changes in the C:N:P stoichiometry. Available Phosphorus was significantly different among sites, but not for forest types across sites. Overall, native forest exhibits higher stocks of available NO_3^- and we did not find a significant effect of forest type in NH_4^+ stocks. Our result indicates the strong influence of soil type in the magnitude of the effect of forest conversion in the total C, N, P stocks.

Keywords: Soil nutrient stocks; C:N:P stoichiometry; forest conversion; use intensification; land-use change.

---

[1] Crovo O, Aburto F, Albornoz M.F, Southard R. In Review. Differential response of soil carbon, nitrogen and phosphorous stocks and available pools to conversion from native forest to exotic plantation in soils of contrasting origin. CATENA.

1. Introduction.

Land use change (LUC) is a global issue with tremendous social, economic and environmental implications. Currently, many countries display high rates of deforestation and forest conversion from native forest to industrial tree plantations (Carlson et al., 2012; Heilmayr et al., 2016; Gaveau et al., 2019). Among the many and diverse described effects of the conversion from native forest to industrial plantation are lower biodiversity (e.g. Brockerhoff et al., 2008), increased C (Carbon) and erosion losses (e.g. Guillaume et al., 2015) and lower climate resilience (e.g. Domec et al., 2015). Even though forest productivity and soil carbon storage are highly interdependent on major nutrient pools, the alteration in nutrient and C : N : P (Carbon : Nitrogen : Phosphorus ratio) stoichiometry as a result of forest conversion to productive plantations have been generally overlooked.

The study of ecological multi-elemental cycling is a very complex task due to the many different biogeochemical processes that are involved (Spohn & Sierra, 2018). Gärdenäs et al. (2011) showed that in most ecosystems nitrogen (N) availability in soils determine the rate of SOM (Soil organic matter) cycling, in turn regulating both plant growth and forest productivity (Quan et al., 2014). Other studies have found that N additions (i.e fertilization and deposition) reduce P limitations in forest ecosystems (Vitousek et al., 2010) because of an acceleration of P cycling (Olander & Vitousek 2001; Treseder and Vitousek 2001) driven by the production of extracellular phosphatase enzyme resulting in a higher phosphorus availability but declining the total P stock (Liu et al., 2014). P availability and total stock also plays an important role on the soil total C stock (Zhou et al., 2006). Were P limitation modify C allocation, reducing standing biomass and litter production (Huang et al., 2011) but induces a higher belowground C allocation product of root exudates (Liu et al., 2014).

The relevance of the C, N and P cycling interrelations and their impact on net primary productivity (NPP) (Cleveland et al., 2013) has motivated researchers to try to disentangle multi-elemental cycling in forest ecosystems (McGroddy et al., 2004; Cleveland & Liptzing 2007; Heuck & Spohn 2016; Feng et al., 2017 ;Müller et al., 2017). These researchers have highlighted the relevance of ecological stoichiometry as a major determinant for material cycling and nutrient limitations in ecosystems (Sterner & Elser, 2002;Cleveland & Liptzin, 2007; Müller et al., 2017). And leads to interesting interrogatives, such as, how forest management intensification could alter biogeochemical C-N-P cycling in soils? And whether potentially enhance tree growth in plantation propel C storage, but also creating imbalances that can result in augmented nutrient leaching and greenhouse gas emissions (Peñuelas et al., 2012).

Recent research showed that planted pine forest have a direct impact on soil C and N stocks, litter stoichiometry and SOM mineralization (e.g. Quan et al, 2014 and Cao & Chen, 2017), yet the impact on ecological stoichiometry and biogeochemical cycling has not been well assessed. Elemental stocks alterations as a result of forest type change are the result of the response to the change in microenvironmental conditions and forest dynamics such as root production, litter quality and deposition, decomposition rates and plant nutrient acquisition (Manzoni et al., 2010; Heuck & Spohn 2016). But also on soil properties and processes that determine productivity , SOM physicochemical stabilization and nutrient retention (Torn et al., 1997; Six et al., 2002; Denef & Six, 2005; Rasmussen et al., 2006), implying that not every soil should exhibit the same biogeochemical resilience to land-use change.

Therefore, here we study the shifts in C:N:P stocks and stoichiometry using adjacent native and pine plantation pair stands across five soils types that encompass the main temperate forest soil types found in south central Chile (Carrasco & Millan, 1990; Toro & Gessel, 1999). These soils display a wide range of intrinsic properties (texture, mineralogy, pH, etc.) as a result of varying parent materials (pyroclastic and residuals) as well as depositional and developmental history (from young deposits to long residence time residuals soils) (CIREN, 1999; Toro, 1985) that have resulted in contrasting dominant secondary mineralogy among these soils (amorphous to crystalline).

This study aims to quantify the modulation effect of soil intrinsic properties on the response of C:N:P stoichiometry induced by native broadleaf (Nothofagus sp.) forest replacement by exotic coniferous plantations (Pinus sp.) in the south-central temperate region of Chile. Consequently, we expect that the differences in soil properties should yield large differences in C and nutrient stocks as well as in nutrient availability and C:N:P relationships in these soils and respective forest floors. We were particularly interested on determining the changes in depth distribution including deep soil pools (>100 cm depth). Therefore, we quantified the C, N and P contents and calculated C:N:P stoichiometry for the litter layers and organic horizons, as well as in mineral soil down to a depth of 2.4 m.

We believe the results of this investigation will provide one of the first indications of the magnitude of the alteration in C and nutrient stoichiometry produced by natural temperate forest replacement by industrial pine plantations considering a range of soils with contrasting soil properties.

We hypothesized that total soil C, N and P reservoirs are mostly driven by soil type and that the magnitude of the alteration of the C:N:P stoichiometry caused by native forest conversion to plantation will also depend on soil type. We predicted that soils with low carbon stabilization potential (i.e. low clay content and dominant crystalline low activity clays) respond more strongly to forest plantation conversion displaying a net decrease in total C and N total stocks, and a decoupled increment in available N and P (i.e. fertilization and deeper resource exploitation). We also predicted that plantation forests modify the elemental vertical distribution concentrating C, N and P in the organic layers and surface mineral horizons.

2. Materials and methods

2.1 Study sites.

To determine how forest type change has modified C, N and P stocks we selected five sites with soils of contrasting origins and mineralogy in south-central region of Chile (Figure 1). We chose areas where forest substitution has occurred at least 35 years ago (Crovo et al., In prep), and remnants of secondary native forest still persist in the surroundings (Aburto et al. In prep).

The selected soils encompassed the main forest soil types of this macro-region (Carrasco & Millan 1990) (Table 1) and are representative of the main soil types used for industrial forestry plantations (Toro & Gessel, 1999). SCH and GR soils correspond to residual soils located in the coastal mountain ranges, derived from mica-schist and granitic materials, respectively. Recent ash (RA), young ash (YA), and old ash (OA) formed from pyroclastic materials (basaltic-andesitic) with increasing deposition ages (Besoaín, 1985; CIREN, 1999). RA is located near Lonquimay volcano and the materials were deposited in continuous eruptions from the volcanic complex during the last 200 years until 1990 (Gonzalez-Ferran, 1995; SERNAGEOMIN 2018). YA and OA further west in the cordillera’s piedmont and correspond to pyroclastic materials deposited after the last glacial maximum probably (>10 Kyrs) during the last major eruptions of the Antuco volcano (Gonzalez-Ferrán, 1995) and pre LGM probably during the interglacial periods between 450.000-200.000 or 140.000-110.000 yrs. (Besoaín, 1985), respectively (Figure 1).

Because of the differences in parent materials, evolutionary history and environmental conditions, the soils included in this study greatly differ in their intrinsic properties displaying contrasting clay mineralogy (from amorphous to crystalline) and different carbon stabilization capacity (Aburto et al., In Prep).

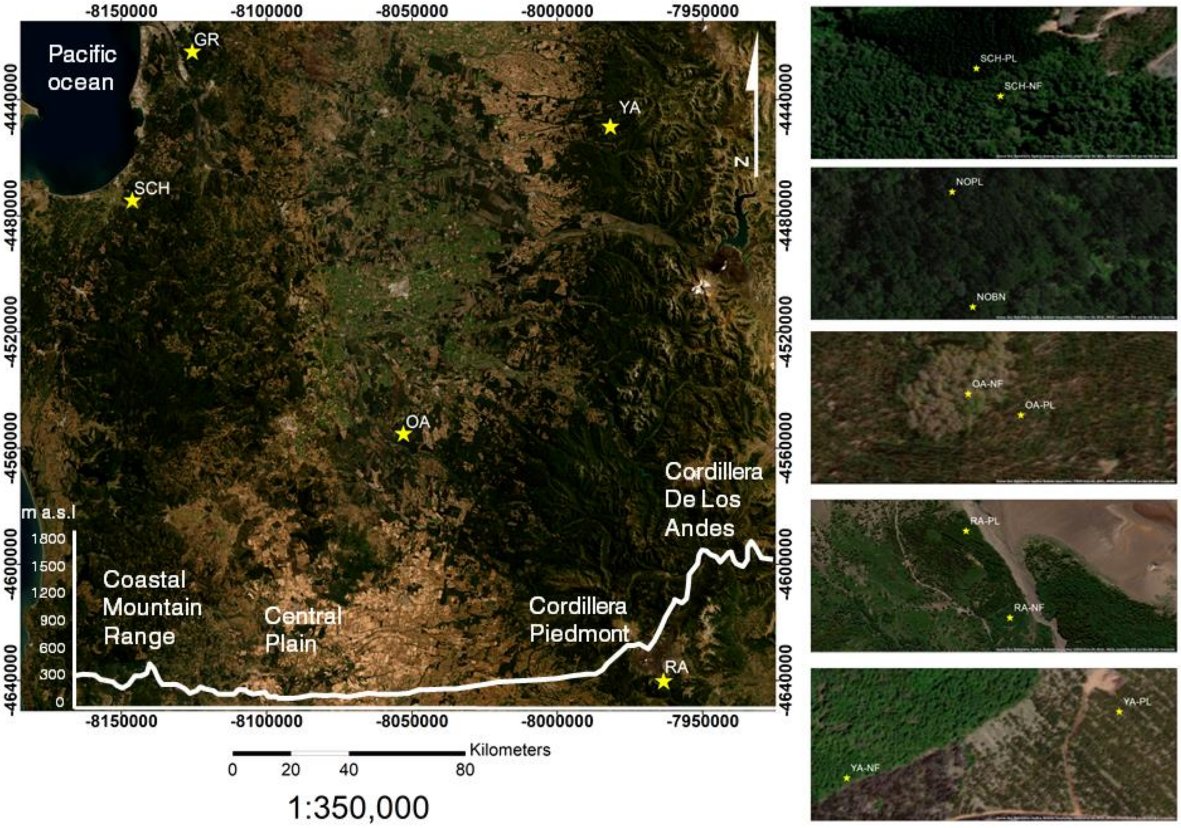

Figure 1. (Left) Locations of the five study sites, across the south-central region of Chile. (Right) Zoom in layers of the NF and PL conditions for each site. Image Source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA FSA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community**.**

Table 1. General description and locations of each site used in this study.

Site	**Location**	**Parent Material** [a]	**Landform**	**Soil Taxonomy**[b]	**Dominant mineralogy type**	**MAP**[d] **(mm)**[c]	**MAT**[e] **(°C)**[c]	**Native dominant specie**	**Coordinates (UTM)**
SCH	Caranpangue	Schist	Hillslope	Typic Haploxerults	Crystalline High activity	1160[1]	12.5[1]	*Nothofagus obliqua.*	18S 661491,5875430
GR	R.N Nonguen	Granite	Hillslope	Mollic Palexeralfs	Crystalline Low activity	1021[2]	12.5[1]	*Nothofagus obliqua.*	18S 678858,5915778
RA	R.N Malalcahuello	Recent Volcanic Tephra	Fluvial Terrace	Vitrandic Udortents	Amorphous Low activity	2090[3]	8.5[4]	*Nothofagus pumilio*	19S 278320,5744197
YA	Yungay	Young Ash Deposit	Piedmont	Pachic Melanudands	Amorphous pseudo-crystalline High activity	1769[5]	12.2[6]	*Nothofagus obliqua.*	19S 259865,5893739
OA	Mulchen	Old Ash Deposit	Piedmont	Typic Rhodoxeralfs	Crystalline Low activity	1316[7]	13.3[7]	*Nothofagus obliqua.*	18S 734093,5810121

[a] CIREN 1999, Besoain, E. 1995, and González-Ferrán, O. 1995.

[b] Soil Survey Staff 2014.

[c] CR^2 Explorador climático 2019 (Data from 1980-2019) at the following climatic stations: [1] Carriel sur meteorological station, Concepción, [2] Carampangue meteorological station, Carampangue, , [3] Malalcahuello meteorological station, Malalcahuello, [4] Las Cruces meteorological station, Yungay, , [5] Diguillin meteorological station, Recinto, [6] Collipulli meteorological station, Collipulli, Chile, [7] La Mona meteorological station, Collipulli, Chile

[d] Mean anual precipitation.

[e] Mean anual temperature.

2.2 Soil sampling strategy and laboratory analyses

To reduce confounding factors due to site history, we exclusively selected pair sampling sites (native versus plantation) that shared a similar land-use history and had close to identical soil and geomorphic conditions. This was achieved by preliminarily selecting and exploring remote sensed images and when possible interviewing local forest managers to reconstruct site history. After this, pre-selected pair-sites were validated on the ground (geomorphic and soil reconnaissance) after which the five final sampling sites were selected. In order to reduce soil and forest variability we preferred sites where the plantation and native forests were at close proximity (from a minimum of ten meters to a maximum of a few hundred meters). In these areas two independent 25m by 25m plots were established at each forest condition (plantation and native) at each of the five pair-sites (10 plots total). The two plots at each site had similar topographic conditions (landform, elevation, slope gradients and exposure).

At the center of each plot in each of the five sites and in each condition (plantation and native forest), one soil pit was hand-dug up to a depth of 240 cm. No further digging could be achieved because of physical limitations such as lithic contact or the presences of water table beyond this depth at some sites. Each soil pit was fully described and sampled by soil genetic horizons following the standard protocol described by Schoeneberger et al. (2012). Soil description and other soil properties are included in a companion paper (Aburto et al., In prep.). In addition, to determine C:N:P inventories the plot was divided into four sub-quadrants (12.5 x 12.5 m). Bulk soil samples were collected at 6 depth intervals in the central soil pit and in four supplementary augers (0-20cm, 20-40cm, 40-80cm, 80-120cm, 120-180cm and 180-240cm). In the central pit, 3 soil volumetric rings (100cm^3) were collected at the mentioned depth intervals and taken to the laboratory to determine soil bulk density. At the five sampling points (1 soil pit and 4 augers in sampling plot) we obtained a volumetric sample of litter layer and organic horizons using a

314 cm^2 cylinder (n=5 per plot). These samples were collected in sterile and hermetic bags, stored in cold (<4ºC) and taken to the lab to avoid further decomposition before performing analytical procedures. The sampling campaign extended from the end of July to the begging of December of 2017. 50 soil profiles were obtained totaling 290 soil mineral depth intervals and 100 organic soil and litter samples that were collected and analyzed.

Mineral soil samples were air-dried at 25°C, sieved (<2mm) removing all visible root and plant remains and then dried to 105°C as a pre-process for carbon and nutrient analysis. For available N (NO_3^- and NH_4^+) determination, we separated an aliquot (10g) in the field of the bulk samples from both the soil pits and the auger samples. These aliquots were kept cold (<4°C) until they were taken to the laboratory for KCl extraction that was performed within the next 24h. Both mineral soil and organic samples (litter layer and O's horizons) were ground using a steel ball mill (Spex Certiprep 8000M©) after which total C and N were determined by Dumas combustion using a SERCON© EA CHNS analyzer. Available NO_3^- and NH_4^+were determined following the protocol detailed by Miranda et al. (2001). Total, organic and inorganic phosphorus (P, $P_{organic}$ and $P_{inorganic}$) were determined following the protocol described by Bowman & Moir (1993) and Gregorich & Carter (2007) utilizing sequential NaOH and Na_2EDTA for $P_{inorganic}$ determination and $K_2S_2O_8$ and H_2SO_4 for P determination and $P_{organic}$ by difference. Available phosphorus was measured by sodium bicarbonate extraction using the standard Olsen extraction method adapted by Sadzawka et al. (2006). All P fractions were determined colorimetrically with the blue ascorbic acid method using a Shimadzu UV-mini 1240 spectrophotometer (Watanabe & Olsen, 1965). All analysis were performed at the Soils, Water and Forest Research Laboratory (LISAB) of the University of Concepción (Concepción, Chile.)

2.3 Data treatment and statistical analysis

We calculated the C, N and P stocks for each soil interval (n = 290) at each soil profile (n = 50) using Equation 1.

$$\boldsymbol{Soil\ Stock_i} = \sum_{i=0}^{n} \frac{\boldsymbol{Xi}}{\boldsymbol{100}} * \boldsymbol{BD_i} * \frac{\boldsymbol{SLT_i}}{\boldsymbol{100}} * \left[\boldsymbol{1} - \left(\frac{\boldsymbol{CF_i}}{\boldsymbol{100}}\right)\right] \qquad \text{Equation 1}$$

Where X_i represents the element (C,N and P) concentration at a given soil interval *i* (%), BD_i the soil bulk density at the same depth interval (Mg m^{-3}), SLT_i is the soil interval *i* thickness (cm) and CF (%) represents the percentage of coarse fragments (>2mm) at that interval. The results for all layers were either summed to estimate the total stocks of soil C, N and P to a depth of 240 cm (deep soil compartment) or 120 cm (shallow soil compartment) and expanded to an hectare level. Consequently, we obtained average values for each site (SCH, GR, RA, YA, OA) and condition (native vegetation and pine plantations). The same procedure was performed to calculate C, N and P stocks for the litter layer and organic horizons using the bulk density estimated based on the extraction cylinder used to collect the litter and organic horizon samples. We also estimated C:N, C:P, N:P and C:N:P on a mass basis for litter layer and organic horizon as well as for the whole soil profile using both a depth weighted average for elemental content and elemental stocks.

All statistical analysis were performed using R software (R Core Team, 2018). Depth distributions were plotted and analyzed using the AQP package (Beaudette et al., 2013). For comparisons of the content of C, N, P, and available N, and P among the different sites and conditions we used ANOVA following the general linear model of the “CAR” R package (Fox et al., 2012), followed by a posthoc multiple comparisons (LSD test) with $p \leq 0.05$ considered as the threshold for statistical significance. Before performing ANOVA, all the data was checked to meet assumptions of normality and homogeneity of variance. For non-normal data we first performed a Box-Cox transformation using the MASS package (Ripley et al., 2013) and if after the transformation the assumptions were still not met, we performed an aligned rank transformed for nonparametric factorial ANOVA (Kay & Wobbrock, 2019). In the presence of left-censored data (i.e. P-Olsen) we used a Tobit regression model approach (Fox et al., 2015) performing the *vglm* function on the VGAM package (Yee, 2015).

3. Results

3.1 Soil C, N, P pools and depth distributions

The C and N total pools were significantly different between soil types (p=0.02 and p<0.001 respectively), but not between forest types. We did not find interactions between the main effects (i.e. soil and forest types). The highest average soil C stock across all sites was found in native forest stands (NF) with 202.22 ± 82.77 Mg ha^{-1} while pine plantations (PL) averaged 172.55 ± 87.73 Mg ha^{-1} .The average soil N stock for NF and PL across all sites was practically identical (NF = 11.9 ± 6.15 Mg ha^{-1} and PL = 11.9 ± 7.18 Mg ha^{-1}). When comparing each soil type individually, disregarding forest type, the Andisol (YA) displayed significantly higher C and N than all the other studied soil-sites with 280.8 Mg ha^{-1} and 16.67 Mg ha^{-1}, respectively (Figure 2). On the contrary, the Entisol (RA) soil displayed the lowest C and N stocks. C content depth distributions were always higher at NF sites in the upper 30 cm (Figure 3). We found that for all sites and conditions more than 70% of the C and N stocks were allocated in the upper 120 cm of the soil profile (Figure 4). For NF and PL, 74 and 79% of the C stocks, respectively, were in the shallow soil compartment (<120). However, we found significant differences at specific depth intervals between forest conditions (Table 2). For example, the planted forest at GR displayed a significantly lower (p=0.03) C stock at the 120-180 cm interval, contributing to the whole soil C stock differences found between NF and PL at this site (p=0.006). We didn’t found significant differences neither in the whole soil C nor in specific depth intervals at SCH. N stock differences between forest types in SCH were only found in the first depth-interval (0-20 cm) (p = 0.047), where NF was higher (Table 3).

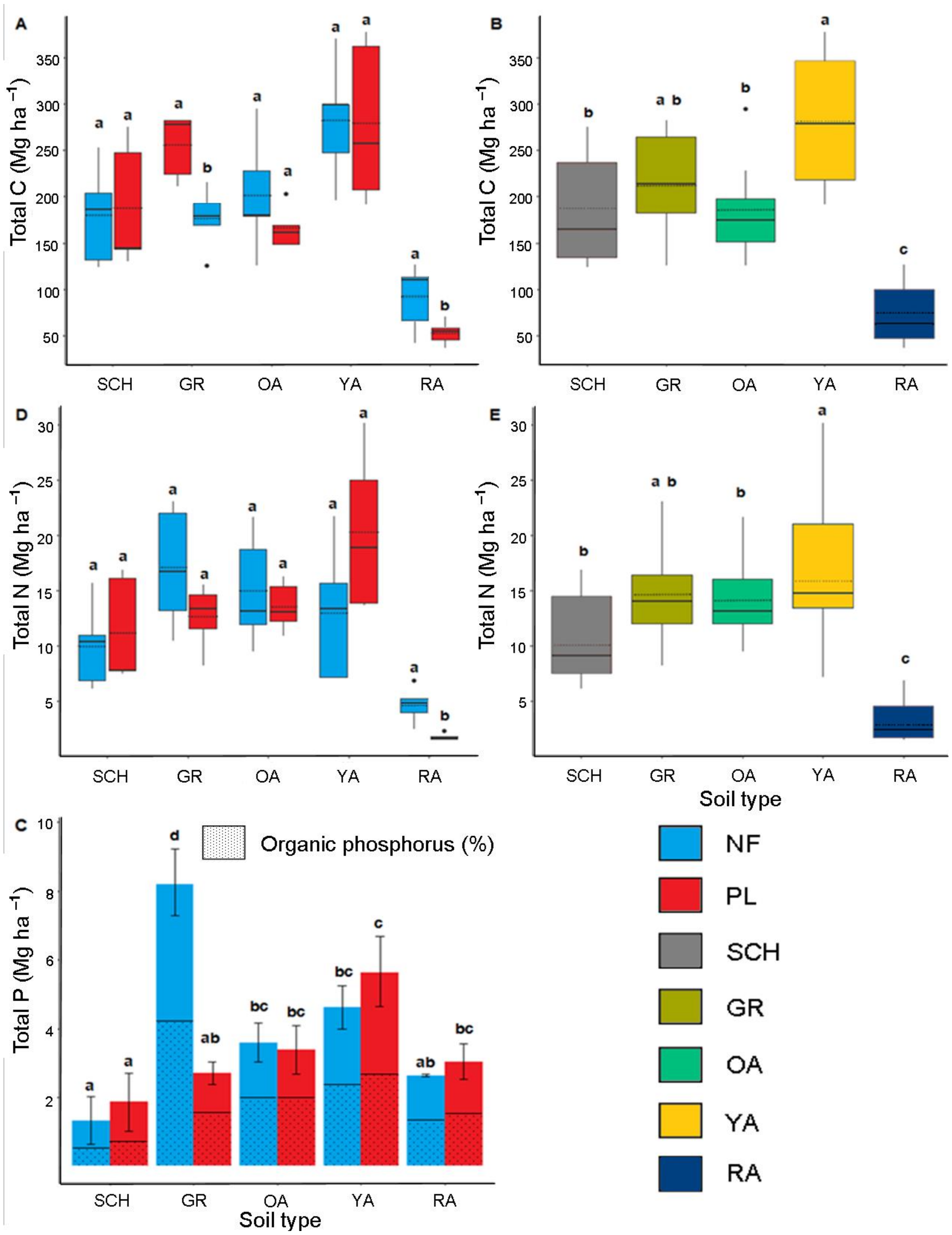

Figure 2 . Total C, N and P stocks for each soil type (B, C, E), each treatment between each soil type (A, D,) and the proportion of organic Phosphorus in the total Phosphorus stock (C). Different letters indicate significant differences and dots outside the box indicate outliers. For A, B, D and E the solid line indicates the median and the dashed line the mean.

It is important to notice that at this site, distribution by depth and overall C stocks were only calculated for the upper 120 cm because of the paralytic contact starting at approximately at 120 cm (Figure 4). In the OA soil, differences in C between NF and PL were displayed in the 20-40 cm ($p=0.047$) and 80-120 cm ($p=0.01$) depth intervals (Figure 3). However, no differences were found for the whole soil C stock between NF (201.440±63.48 Mg ha^{-1}) and PL (166.2±22.65 Mg ha^{-1}) across soil types. N content and stocks followed a similar depth trend pattern than C (Figure 3), and differences in N stocks between NF and PL were only found at the 20-40 cm depth-interval ($p =0.009$).

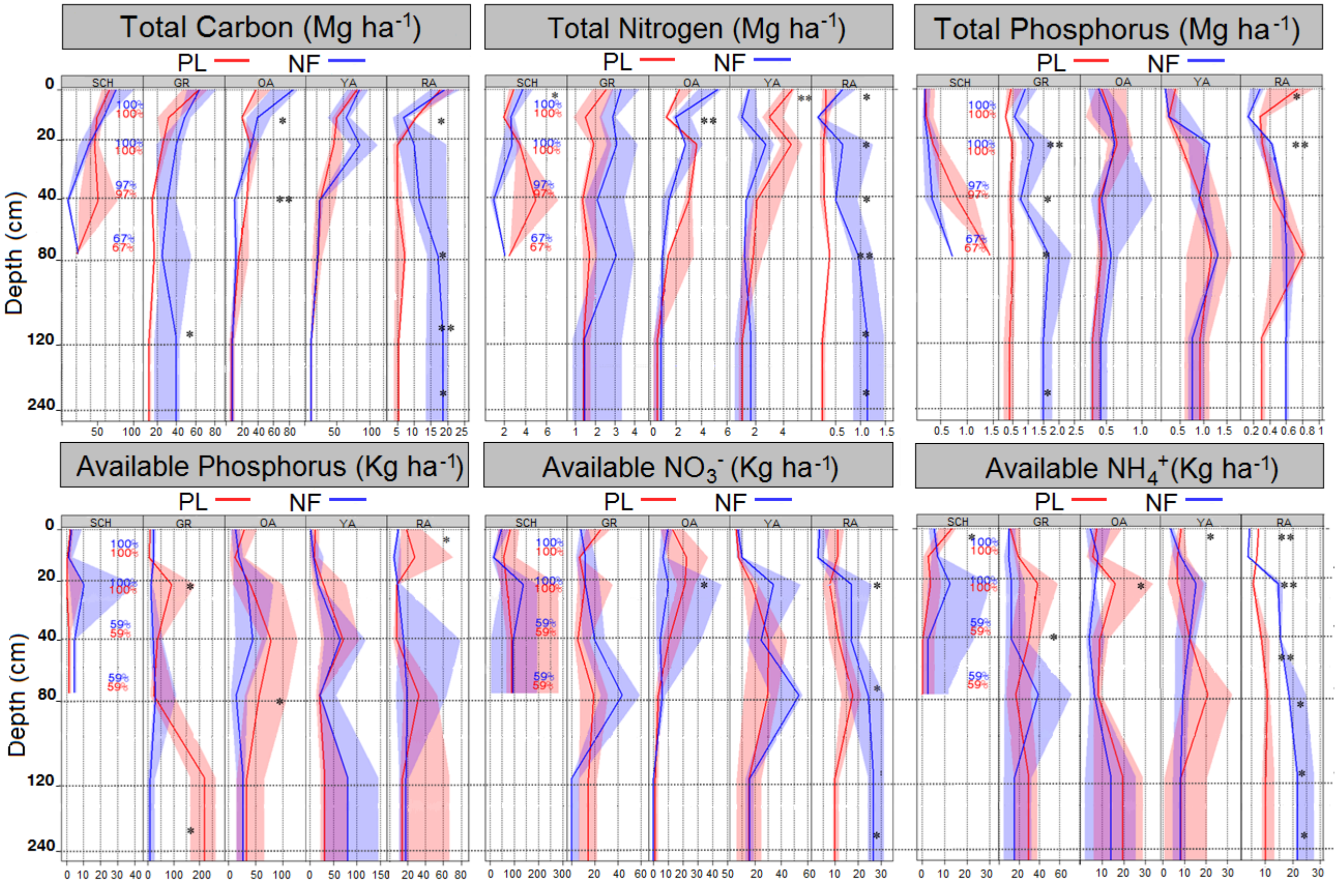

Figure 3. Carbon, Nitrogen and Phosphorous content (%) and available fractions (mg/Kg$_{suelo}$) depth distribution for each site and treatment. The continued line is the median while the casted shadow shows the variation between the data (25th and 75th percentiles). Percentages in the right of some plots show the amount of data that contributed for the median. Only for SCH the 100% of the data was not used since there was a paralithic contact at approximately 120cm.

The recent-ash derived soil (RA) showed a significant difference in C stock ($p=0.05$) between NF (91.85 ±35.8 Mg ha^{-1}) and PL (52.72 ±12.96 Mg ha^{-1}). Similarly, the NF display significantly higher C stock in most depth intervals; 20-40 cm ($p=0.034$), 80-120 cm ($p=0.028$), 120-180 cm ($p=0.01$) and 180-240 cm ($p=0.02$). N also displayed significant differences between NF and PL at the same depth intervals (Table 3, Figure 1). At the YA site (Andisols) we found no significant differences between NF and PL neither in the total whole soil C stock and N stock nor at depth intervals.

Total phosphorous stock was the only variable that showed an interaction between the main effects forest type and soil ($p = 0.001$). The interaction yielded four distinct groups, where GR-NF whole soil P stock was significantly higher than any other combinations (Figure 2). SCH-NF and SCH-PL presented significantly lower whole soil P stock (0.61 ±0.29 and 1.49 ±1.32 Mg ha^{-1}, respectively). In average, GR-NF had 53% of P in the top 120 cm, while GR-PL contained 68%. GR-PL the P content did not vary much with depth, whereas in GR-NF P content was lower in the upper half of the soil profile and then consistently increased. GR-NF had significantly higher total P stock than GR-PL in almost all depth intervals (20-40, 40-80, 120-180 and 180-240 cm) (Table 3, S. Figure 1). This was also evident for average P content depth-distribution that was significantly higher in NF for all soil depth intervals (Figure 3).

Table 2. Carbon stocks for each soil type in both treatments and its proportion by depth intervals.

Depth Intervals (cm)	**Total Carbon stock (Mg ha^{-1})**									
	SCH (Ultisol)		**GR (Alfisol)**		**OA (Alfisol)**		**YA (Andisol)**		**RA (Entisol)**	
	NF	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**
0 - 20	81.83 ±46.9	66.06 ±36.6	72.36 ±33.5	60.22 ±15.7	76.31 ±31.5	53.52 ±32.7	88.02 ±22.0	80.7 ±9.3	21.36 ±11.0**a**	19.81 ±5.1**b**
20 - 40	54.72 ±31.6	52.6 ±29.2	49.01 ±13.9	36.7 ±25.7	47.47 ±31.5**a**	21.36 ±3.5**b**	52.45 ±30.4	48.93 ±8.9	5.97 ±2.9	10.44 ±2.6
40 - 80	33.41 ±19.3	41.66 ±23.1	37.28 ±19.2	29.07 ±7.3	31.7 ±9.1	32.25 ±7.1	83.24 ±37.4	64.74 ±44.3	13.95 ±11.4	4.41 ±1.7

80 - 120	8.98 ±2.1	50.68 ±11.2	31.31 ±16.5	15.13 ±2.7	12.41**a** ±4.8	27.94**b** ±5.5	25.89 ±5.1	49.4 ±6	13.18 ±6.9**a**	5.45 ±1.5**b**
120 - 180	NA	NA	35.40 ±18.8**a**	17.06 ±2.6**b**	21.95 ±24.9	20.48 ±8.9	23.95 ±10.5	23.19 ±10.3	16.15 ±5.7**a**	5.98 ±6.9**b**
180 - 240	NA	NA	30.28 ±17.5	18.33 ±15.2	11.6 ±10.3	7.66 ±4.2	11.88 ±2.9	12.23 ±2.9	21.24 ±16.0**a**	5.64 ±2.8**b**
Sum 0-120	**173.1** ± 43.3	**180.5** ± 56.9	**190.0** ± 10.6	**141.1** ± 24.2	**168.8** ± 71.9	**138.1** ± 30.1	**246.6** ± 54.5	**243.8** ± 74.9	**54.4** ± 25.3	**40.1** ± 7.4
Sum 0-240	**NA**	**NA**	**255.6** ± 34.9	**176.5** ± 33.3	**201.4** ± 63.5	**166.2** ± 22.6	**282.4** ± 65.1	**279.2** ± 86.3	**91.8** ± 35.8	**52.7** ± 12.9

Mean ± sd, different letters indicate statistical differences for each soil depth ($p<0.05$) between the NF (Native forest) and PL (Plantation) paired plots.

NA, no mean was calculated for these depths because of the lack of repetitions (augers) at this site due to paralithic contact.

In almost all soils and forest types, the organic phosphorus fraction P accounts for more than the 40% of the P pool, being organic P higher in the PL condition in most cases, except from SCH (Figure 2). The higher contributions of organic P to the total P stock could be found in OA-PL (67%) and YA-PL (65%) and the lowest at SCH-PL (40%) and RA-NF (41%).

Table 3. Nitrogen stocks for each soil type in both treatments and its proportion in depth intervals.

Depth Intervals (cm)	**Total Nitrogen stock (Mg ha^{-1})**									
	SCH (Ultisol)		**GR (Alfisol)**		**OA (Alfisol)**		**YA (Andisol)**		**RA (Entisol)**	
	NF	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**
0 - 20	3.81 ±0.9	2.73 ±0.3	3.66 ±1.1	2.60 ±1.1	4.65 ±1.7	3.21 ±2.1	1.85 ±1.4**a**	4.4 ±0.7**b**	0.70 ±0.2**a**	0.35 ±0.2**b**
20 - 40	2.60 ±0.4	2.09 ±0.3	2.51 ±0.8	2.27 ±2.5	2.97 ±2.3**a**	1.23 ±0.1**b**	2.1 ±1.5	3.25 ±0.9	0.14 ±0.1**a**	0.30 ±0.1**b**
40 - 80	2.64 ±1.2	3.51 ±0.5	3.39 ±1.8	2.17 ±0.7	3.33 ±0.9	3.38 ±0.5	3.5 ±3.3	5.12 ±1.2	0.99 ±0.6**a**	0.24 ±0.1**b**
80 - 120	0.97 ±1.7	4.97 ±2.1	2.25 ±1.5	1.67 ±0.6	1.91 ±1.7	2.99 ±0.5	2.01 ±1.5	4.79 ±5.9	0.82 ±0.4**a**	0.25 ±0.1**b**
120 - 180	NA	NA	3.18 ±1.3	1.78 ±0.5	0.94 ±0.4	2.18 ±1.3	2.35 ±0.5	1.8 ±1.0	1.21 ±0.5**a**	0.37 ±0.1**b**
180 - 240	NA	NA	2.11 ±1.7	2.22 ±1.7	1.18 ±1.4	0.58 ±0.5	1.23 ±2.8	0.97 ±0.3	1.30 ±0.5**a**	0.27 ±0.1**b**
Sum 0 - 120	**9.40** ±2.8	**10.32** ±3.9	**11.80** ±2.8	**8.68** ±2.1	**12.87** ±4.5	**10.81** ±2.4	**9.45** ±6.5	**17.51** ±6.1	**2.43** ±1.1**a**	**1.14** ±0.2**b**

Sum **0 - 240**	**NA**	**NA**	**17.09** ±5.5	**12.68** ±2.9	**14.99** ±5.01	**13.56** ±2.2	**13.02** ±6.1	**20.31** ±7.2	**4.63** ±1.6**a**	**1.78** ±0.3**b**

Mean ± sd, different letters indicates statistical differences at each soil depth ($p<0.05$) between the NF (Native forest) and PL (Plantation) paired plots.

Na, no mean was calculated for these depths because of the lack of repetitions (augers) at this site due to paralithic contact.

3.2 Available N and P pools and depth distributions.

Available phosphorus (P_{Olsen}) was significantly different among sites ($p<0.001$), but not for forest types across sites ($P=0.12$) and no interaction between the main effects occurred (Table 4). Among sites, SCH showed the lowest available P 8.59±8.5, while YA exhibited the highest with 69.28 ± 31.9. No differences in available P were found between NF and PL, except for GR ($p=0.05$) and OA ($p=0.029$) where PL was significantly higher than NF (Figure 5). In this site we only found significant differences in the available P distribution in the 80-120 cm interval ($p=0.04$). The NF in the GR showed little vertical variation in available P stock. However, PL displayed a bimodal depth-trend, with significantly higher available P stock than NF in the 20-40 cm ($p=0.04$) and in the deepest 180-240($p=0.02$) cm intervals (Table 2, Figure 3).

Table 4. Phosphorus stocks for each soil type in both treatments and its proportion in depth intervals.

Depth Intervals (cm)	**Total Phosphorus (Mg ha^{-1})**									
	SCH (Ultisol)		**GR (Alfisol)**		**OA (Alfisol)**		**YA (Andisol)**		**RA (Entisol)**	
	NF	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**
0 - 20	0.31 ±0.2	0.45 ±0.2	0.9 ±0.4	0.58 ±0.3	0.5 ±0.4	0.73 ±0.7	0.24 ±1.1	0.4 ±0.2	0.27 ±0.09**a**	0.83 ±0.62**b**
20 - 40	0.27 ±0.2	0.47 ±0.3	0.75 ±0.5**a**	0.24 ±0.04**b**	0.67 ±0.5	0.7 ±0.7	0.46 ±0.3	0.32 ±0.2	0.13 ±0.03**a**	0.38 ±0.2**b**
40 - 80	0.35 ±0.2	0.55 ±0.3	1.48 ±1.03**a**	0.54 ±0.2**b**	0.67 ±0.2	0.66 ±0.2	0.9 ±0.4	0.78 ±0.6	0.38 ±0.2	0.3 ±0.1
80 - 120	0.41 ±0.1	0.43 ±0.4	0.85 ±0.6	0.48 ±0.2	0.73 ±0.6	0.46 ±0.1	0.96 ±0.3	1.27 ±0.9	0.61 ±0.2	0.48 ±0.2

120 - 180	NA	NA	2.05 ±1.3**a**	0.49 ±0.3**b**	0.62 ±0.3	0.47 ±0.1	1.2 ±0.5	1.63 ±1.5	0.65 ±0.2	0.68 ±0.3
180 - 240	NA	NA	1.5 ±0.7**a**	0.38 ±0.2**n**	0.4 ±0.2	0.35 ±0.1	0.88 ±0.5	1.21 ±1.01	0.58 ±0.1	0.37 ±0.2
Sum 0-120	**0.98** ±0.8	**1.58** ±1.3	**3.98** ±1.6	**1.84** ±0.5	**2.58** ±1.02	**2.56** ±1.5	**2.54** ±0.7	**2.78** ±1.9	**1.41** ±0.3	**1.96** ±0.1
Sum 0-240	**NA**	**NA**	**8.2** ±3.3	**2.71** ±0.7	**3.6** ±1.2	**3.39** ±1.6	**4.61** ±1.4	**5.63** ±4.3	**2.64** ±0.1	**3.05** ±1.1

Mean ± sd, different letters indicates statistical differences at each soil depth ($p<0.05$) between the NF (Native forest) and PL (Plantation) paired plots.

NA, no mean was calculated for these depths because of the lack of repetitions (augers) at this site due to paralithic contact.

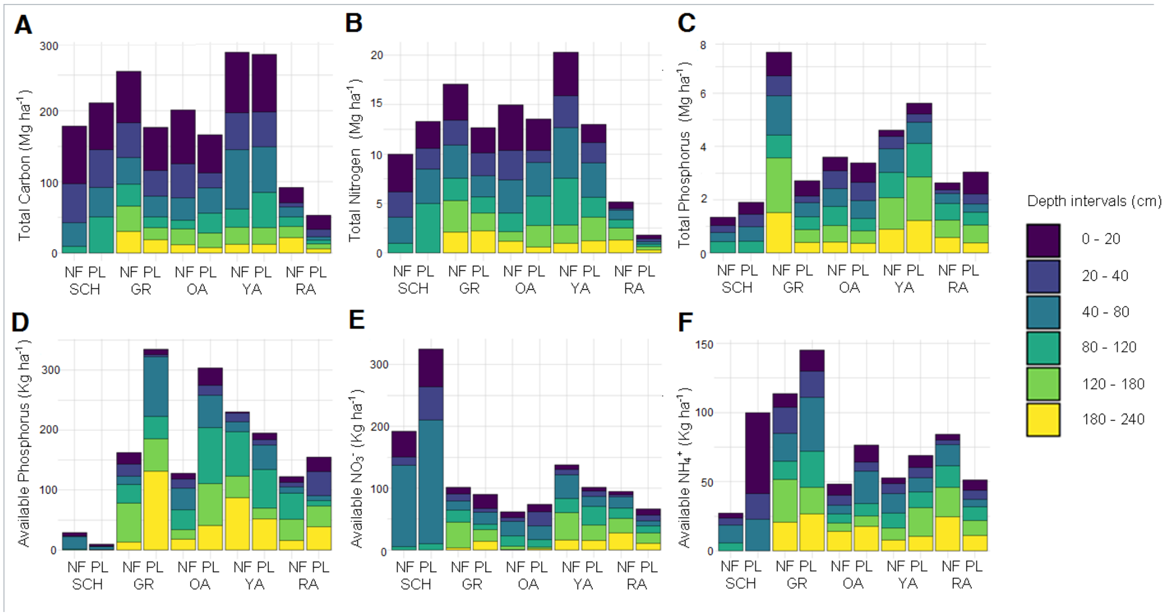

Figure 4. Relative depth-interval contribution to the total (A) Carbon Stock, (B) Nitrogen Stock, (C) Phosphorus Stock, (D) Available P Stock, (E) Available NO_3^- and (F) Available NH_4^+ for each forest and soil type.

At the RA soil, PL exhibited significantly higher available P in the upper depth interval (0-20cm) ($p=0.02$) (Figure 3). In RA for both forest types, the top 120cm allocated more than the 50% of available P (Figure 4). At YA, both NF (69.88 ± 45.4 Kg ha^{-1}) and PL (68.68 ± 17.9 Kg ha^{-1}) exhibited very similar trends in the available P depth distribution and with no differences in the available P stock. However, available P compartmentalization in the YA soil was notoriously different between forest types, where NF only allocated 47% of available P, while PL allocated approximately 64% (Figure 4).

No differences were found in NO_3^- stocks among sites or between forest type conditions. However, the distribution of NO_3^- contents were highly variable between sites, conditions and also among depth intervals showing high standards deviations (Figure 5). Overall, NF exhibits higher stocks of available NO_3^-, with the exception of SCH and OA (Figure 5). On the contrary, at RA the NF exhibits a higher NO_3^- than PL in the 20-40 ($p=0.03$), 120-180($p=0.0475$) and 180-240cm ($p<0.001$) depth intervals (Figure 3). OA was the only soil where plantation displayed a significantly higher NO_3^- content, but only in the 20-40 cm interval ($p= 0.04$).

OA had the lowest NO_3^- pool among sites and for both NF and PL allocated a significantly higher percentage of soil available NO_3^- in the upper soil compartment (90% and 93%, respectively). OA by depth-distribution displayed an NO_3^- a relative enrichment in the upper meter of soil being significant for the 20-40 depth interval (Figure 3).

In contrast, In the GR soil NF had a higher NO_3^- pool than PL (101.2 ± 47.4Kg ha^{-1} and 90 ± 61.1 Kg ha^{-1}, respectively). NO_3^- depth allocation in this site was also different, and PL allocated more of the nitrate in the top 120 cm (63.4 %) than in NF (55.4 %) (Figure 4). RA shows a similar depth distribution of NO_3^- content and stocks between forest types (Figure 3), however the average soil nitrate stock was far higher in NF than in PL (94.6 ± 30.7 Kg ha^{-1} versus 66±9.6 Kg ha^{-1}). We did not find a significant effect of forest type in NH_4^+ stocks (Figure 5). In the OA soils NH_4^+ stock was significantly higher in PL ($p =0.025$), while in SCH and RA NH_4^+ was significantly higher in NF than in PL ($p =0.021$) (Figure 5). Regarding depth distribution (Figure 3), PL in OA and GR displayed a higher NH_4^+ content in the 40-80 cm depth-interval that resulted in a significant higher stock ($p=0.02$ and $p=0.04$, respectively). Likewise, in the YA, PL exhibited a higher NH_4^+ content and stock ($p=0.01$) site in the 0-20cm depth-interval, but no differences were found in the whole soil NH_4^+ stock for this site. Similarly, PL in SCH displayed a higher NH_4^+ content and stock in the upper 0-20cm depth-interval (p=0.013), but sharply decreased with depth being higher in the NF in the 20-40cm depth-interval. The RA soil under NF was the only that show consistently higher ammonium contents for all depth-intervals.

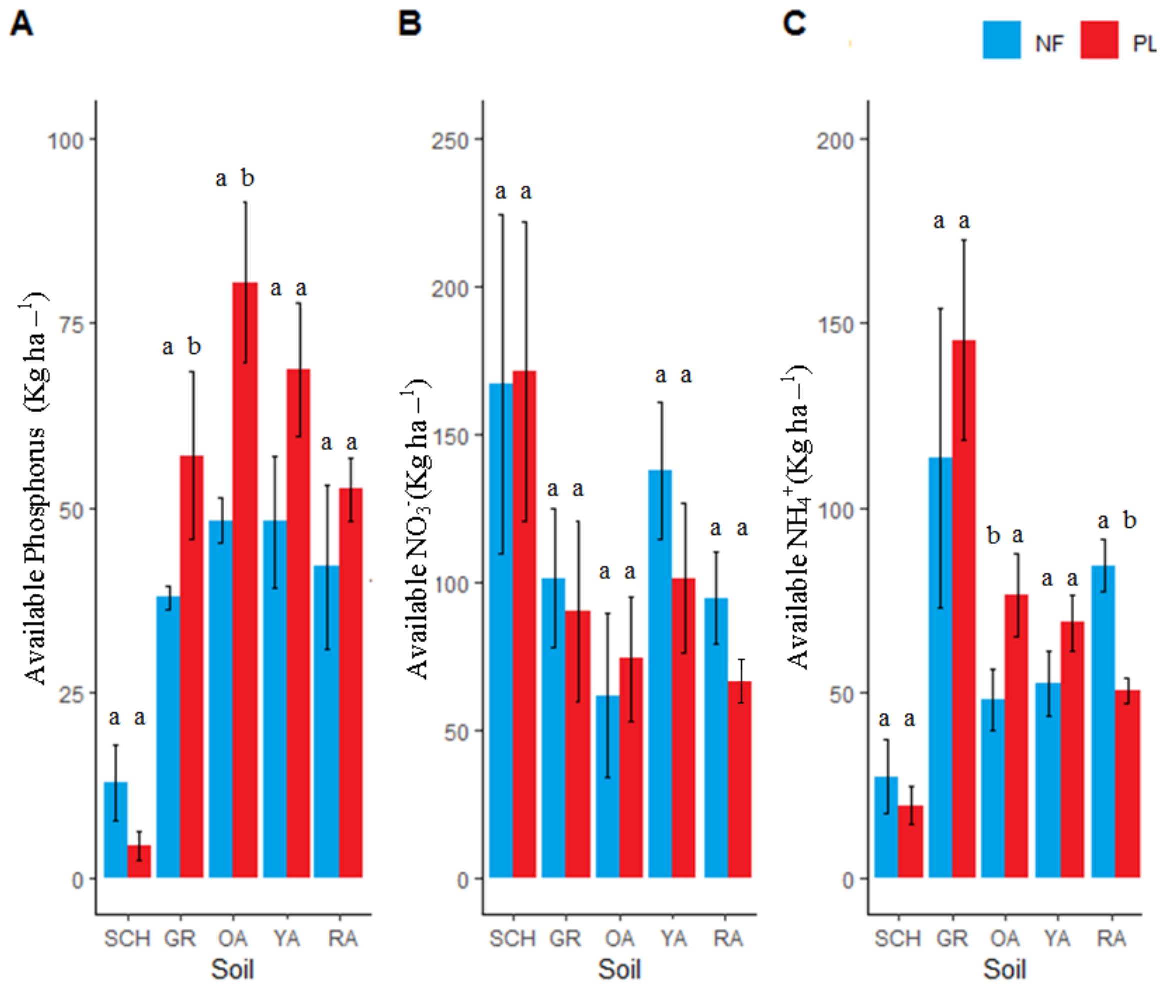

Figure.5 Available Olsen Phosphorous (A), NO^{3-}(B), NH^{4+}(C) stocks for each soil and forest type. Different letters indicate significant differences between forest type for the same soil-site type.

3.3 Organic and Litter Layers C, N and P.

We found significant differences in the C content (%) of the organic horizon among soils types (p=0.01), but not between forest type across sites (p=0.32). Similarly, we did not find significant differences between forest types at each soil type. There was a significant variability in elemental contents in the organic horizons within sites, which may have reduced our ability to test the effect of forest types (Table 5).

Similarly, we found significant differences in the C stock in organic horizons among sites (p=0.009), but not between forest types across sites (p=0. 1). Although not significant, GR under PL had a relatively higher C stock in the organic layer than NF (5.31 ± 3.9 Mg ha^{-1} and 4.42 ± 1.9 Mg ha $^{-1}$, respectively). The organic horizon in SCH and NF displayed a higher average C stock (8.74 ± 5.1 Mg ha^{-1}) than in PL (5.46 ± 4.7 Mg ha^{-1}) (Table 6). We found significant differences in organic horizons among soils (p= 0.01) and forest types (p= 0.02). Across all sites NF has a significantly higher average N concentration than PL (0.8±0.4% versus 0.6±0.3%). The most enriched organic horizon occurred in the YA soil (1.13 ±0.32%) and the significant lowest N content occurred RA (0.27 ± 0.12%).

Table 5. Carbon, Nitrogen and Phosphorus (%) concentrations in the forest floor.

	Organic Horizon									
	SCH (Ultisol)		**GR (Alfisol)**		**OA (Ultisol)**		**YA (Andisol)**		**RA (Entisol)**	
	NF	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**
C (%)	17.26 ±3.9	15.88 ±5.7	16.79 ±2.4	14.66 ±5.1	25.42 ±4.6	23.4 ±5.4	26.42 ±3.9	24.39 ±3.7	10.34±2.8	11.88 ±2.9
N (%)	0.68±0.1	0.49±0.19	0.71±0.3	0.49±0.3	1.08±0.2	0.85±0.2	1.23±0.4	1.03±0.2	0.22±0.1	0.32±0.1
P (%)	0.03 ±0.01	0.02 ±0.01	0.04 ±0.01	0.04±0.01	0.04 ±0.01	0.03 ±0.01	0.05 ±0.02	0.07±0.01	0.03 ±0.01	0.04 ±0.01
	Litter Layer									
C (%)	36.12 ±8.3	40.48 ±9.3	34.46±7.9	45.51 ±2.7	39.16 ±3.99	41.32 ±2.7	35.29±6.6**a**	45.89 ±3.4**b**	25.34 ±6.2	27.44 ±10.9
N (%)	0.88 ±0.1	0.97±0.3	1.24 ±0.3	1.13±0.2	1.12±0.3	0.88±0.3	0.85±0.3	1.12±0.1	0.54 ±0.2	0.53±0.5
P (%)	0.04 ±0.01	0.04 ±0.01	0.07 ±0.02	0.05 ±0.01	0.06±0.02	0.06 ±0.01	0.06 ±0.01	0.06 ±0.02	0.06 ±0.01**a**	0.06±0.01

Mean ± sd, different letters indicates differences ($p < 0.05$) between means with no interaction.

For N stocks, we found significant differences only between soil types ($p < 0.001$), but not between forest types (NF vs PL), and no interaction among main effects were found. Similarly, no differences between forest types within soil types were found. For SCH, GR and OA NF exhibit a higher N stock in the organic horizon whereas in RA and YA the stocks were higher, but not significant in PL (Table 5). We only find significant differences in the P content between soil types ($p=0.01$). The largest P content occurred at YA soils (0.06 ± 0.01%) and the lowest in the SCH soils (0.025 ± 0.01 %). Likewise, the P stock in the organic layer displayed significant differences only between soils types ($p = 0.001$). In GR, RA and YA the average P stock was higher in PL while in OA and SCH the higher P stock was found in NF (Table 5). In SCH we found the largest difference in P stock in the organic layer, but still, no statistical differences were found.

The litter layer in NF has significantly higher C contents than PL ($p=0.01$) across sites and between sites ($P<0.001$). N and P were significantly different between sites ($p<0.001$ and $p=0.01$, respectively), but not for forest type. We found neither difference in C stock for the litter layer between neither sites nor interactions between the main effects (soil and forest types). The native forest in SCH and GR exhibited higher C stocks in the litter than PL, but for OA, RA and YA the opposite was found (Table 5). The PL forest in YA had a significantly higher C stock ($p = 0.004$) than NF (10.17 ± 1.6 Kg ha $^{-1}$ and 5.12 ± 2.3 Kg ha^{-1}, respectively). The N stock in the litter layer showed interaction between soil and forest type ($p= 0.005$). GR-BN (314.5 ± 165.4 Kg ha $^{-1}$) exhibited a significantly higher N stock in the litter layer than GR-PL (178.39 ± 53.9 Kg ha^{-1}) ($p=0.01$). In contrast, YA-PL had a higher ($p = 0.0024$) N stock (265.15 ± 57.9 Kg ha^{-1}) than YA-NF (128.01 ± 89.4 Kg ha^{-1}). On the other hand, P was not different between soil types and we only found significant differences between forest types for YA ($p = 0.04$), where PL had a higher P stock of (13.2 ± 3.6 Kg ha^{-1}) than NF (8.28 ± 2.9 Kg ha^{-1}). In the same way as N, in SCH and GR, NF had a higher P stock, while in OA and RA, PL exhibited a higher P stock (Table 6)

Table 6. C, N and P stocks in forest floor compartments.

Organic Horizon

Element	**SCH (Ultisol)**		**GR (Alfisol)**		**OA (Alfisol)**		**YA (Andisol)**		**RA (Entisol)**	
	NF	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**
C (Mg ha^{-1})	8.8 ±5.1**a**	5.47 ±4.7**a**	4.43 ±1.9**a**	5.3 ±3.9**a**	4.3 ±2.5**a**	3.4 ±2.1**a**	9.13 ±4.8**a**	7.32 ±3.2**a**	3.43 ±1.7**a**	3.6 ±1.8**a**
N (Kg ha^{-1})	344.2±198.02**a**	170.7 ±146.7**a**	168.8 ±60.6**a**	188.3 ±176.4**a**	167.8 ±71.1**a**	121.0 ±74.9**a**	412.32 ±208.6**a**	310.31 ±144.8**a**	61.51 ±29.1**a**	97.7 ±64.9**a**
P (Kg ha^{-1})	14.66 ±9.6**a**	6.64 ±4.3**a**	10.7 ±4.5**a**	13.5 ±10.9**a**	5.9 ±3.5**a**	4.8 ±3.3**a**	17.52 ±11.2**a**	19.36 ±7.04**a**	9.58 ±4.3**a**	9.92 ±4.1**a**

Litter Layer

Element	**SCH (Ultisol)**		**GR (Alfisol)**		**OA (Alfisol)**		**YA (Andisol)**		**RA (Entisol)**	
	NF	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**	**NF**	**PL**
C (Mg ha^{-1})	10.19 ±9.9**a**	6.5 ±1.7**a**	9.8 ±6.45**a**	7.22 ±2.1**a**	8.0 ±1.5**a**	10.07 ±5.6**a**	5.12 ±2.3**a**	10.18 ±1.6**b**	5.78 ±1.9**a**	6.8 ±5.9**a**
N^{a} (Kg ha^{-1})	179.9 ±110.4**AB**	156.3 ±56.9**A**	314.5 ±165.4**B**	178.4 ±53.9**A**	226.1 ±60.5**ABC**	190.60 ±45.2**AB**	128.01 ±89.4**A**	265.16 ±57.9**B**	122.4 ±60.5**AB**	88.7 ±60.7**AB**
P (Kg ha^{-1})	8.7 ±6.1**a**	6.96 ±2.6**a**	21.4 ±14.6**a**	7.4 ±2.2**a**	12.6 ±4.26**a**	15.73 ±9.7**a**	8.28 ±2.9**a**	13.20 ±3.6**b**	13.8 ±7.3**a**	16.3 ±10.9**a**

Mean ± sd, different capitals letters indicates significant differences between means and interaction among main effects (forests and soil types), and different lower-case letters indicates differences between means with no interaction.

[a] Means are reported, but capital letters indicated differences in the estimated marginal means after aligned ranks factorial nonparametric ANOVA (emmeans).

.

3.4 C: N: P stoichiometry in mineral soils and forest floors

The C: N, C: P and N: P in the mineral soils were calculated as depth-weighted average of both elemental contents (%).We found significant interaction between main effects for C: N, C: P and N: P content ratios ($p = 0.03$, $p = 0.04$, $p < 0.001$). For C:N there were two significantly different groups, where YA-NF, RA-NF, and RA-PL displayed the largest ratio (Table 7). For C: P we identified four significantly different groups, where the highest C: P value occurred in SCH-NF, while the lowest values were found in RA-PL, RA-NF and GR-NF. Similarly, N: P formed five distinct groups, where the significantly largest value was found in SCH-NF, while the lowest ratios in RA-PL and RA-NF (Table 7)

Table 7. Depth-weighted average ratios of C:N, C:P and N:P in the five soil types and forest types.

Soil	**Treatment**	**C:N**[a]	**C:P**[a]	**N:P**[a]	**C:N:P**
SCH	NF	19 ±1.4 **a**	250 ±130 **d**	13 ±7 **e**	250:13:1
SCH	PL	19 ±2.7 **a**	175 ±63 **cd**	9±4 **de**	175:9:1
GR	NF	16 ±4 **a**	39.2 ±10 **ab**	3±1**bc**	39:3:1
GR	PL	15 ±1 **a**	72 ±27 **bc**	5 ±2 **bcde**	72:5:1
OA	NF	13 ±2**a**	50 ±10 **b**	4 ±1.1**bcd**	50:4:1
OA	PL	12 ±2**a**	49 ±18 **b**	4 ±2 **bcd**	49:4:1
YA	NF	34 ±20 **b**	103 ±59 **bcd**	3 ±1**bc**	103:3:1
YA	PL	15 ±1 **a**	95 ±61 **bc**	6 ±4 **cde**	95:6:1
RA	NF	21 ±4 **ab**	39 ±16 **ab**	2 ±1 **ab**	39:2:1
RA	PL	25 ±4 **ab**	17 ±7 **a**	1±0.2 **a**	17:1:1

Mean ± sd ratios where different letters indicate significant differences in that variable with interaction between main effects.

[a] Separated by groups by the differences in the estimated marginal means (emmeans).

The C:N ratio of the litter layer was significantly different among soil (p <0.001) and forest types (p <0.001). RA had a significantly higher C:N ratio than all the other sites (Table 8). The NF litter layer in average had lower C:N ration than PL (41:1 and 49:1 respectively). However, only two sites showed significant differences on the litter C:N ratio between forest types, where NF exhibit significantly lower ratios (GR and RA, p =0.04) (Table 8). Litter C: P ratios also display significant differences between soil and forest types (p < 0.001 and p = 0.03), respectively being higher in PL. Among different soils SCH display significantly higher C:P content ratios than OA and RA, likewise the C:P ratio in GR was higher than in RA. Litter N:P ratio was significantly different between sites (p < 0.001), but not between forest types (Table 8).

Table 8. Ratios of C:N, C:P and N:P in the litter layer in the five soil types and each treatment.

Soil	Condition	C:N	C:P	N:P
SCH	NF	41.03±7.1 **a**	916.7±291.3 **a**	22.2±4.7 **a**
SCH	PL	42.61±5.3 **a**	990.1±311.1 **a**	23.5±7.9 **a**
GR	NF[a]	31.1±6.9**a**	566.1±247.5 **a**	18.4±6.9 **a**
GR	PL[a]	40.6±4.4**b**	975.5±119.9 **b**	24.1±2.6 **a**
OA	NF	36.2±6.7 **a**	674.5159.2 **a**	19.7±7.6 **a**
OA	PL	50.6±16.7 **a**	664.2±156.9 **a**	14±4.1 **a**
YA	NF	44.2±12.5 **a**	604.4±156.9 **a**	15.2±8 **a**
YA	PL	39.1±7.1 **a**	802.8±161.2 **a**	20.9±5.1 **a**
RA	NF[a]	50.4±14.4**a**	472.4±129.9 **a**	10.1±4 **a**
RA	PL[a]	72.95±15.9**b**	426.4±191.7 **a**	6±2.9 **a**

Mean ± sd ratios where different letters indicate significant differences in that variable with interaction between main effects.

[a] We only display letters in sites were significant differences existed between forest types.

For the organic layer, C:N content ratio was significantly different between soils (p<0.001), but not for forest types. The highest C:N ratio occurred in RA (49.77) which was significantly higher than in any other soil. We found a significant difference between PL and NF in SCH were PL was significantly higher (33:1vs 25:1, p = 0.002) (Table 9). C:P content ratio in the litter layer was significantly different between soils (p = 0.001), but not between forest types (NF: 578:1 vs. PL 566:1). The highest C:P content ratio was obtained in OA (801:1), which was significantly higher than GR (433:1) and RA (381:1) (Table 9).

Table 9. Ratios of C:N, C:P and N:P in the organic layer in the five soil types and each treatment.

Soil	**Treatment**	**C:N**	**C:P**	**N:P**
SCH	NF[a]	25.1 ± 1.5 **a**	645 ± 187.9	25.6 ± 6.9
SCH	PL[a]	32.8 ± 4.2 **b**	825.9 ± 548.3	26.5 ± 20.8
GR	NF	26.9 ± 11.3	418.9 ± 88.2	17.8 ± 7.9
GR	PL	35.3 ± 13.3	447.8 ± 256.8	15.1 ± 10.9
OA	NF	23.9 ± 4.5	793.0 ± 436.9	32.2 ± 14.4
OA	PL	29.3 ± 10.4	807.9 ± 201.4	29.6 ± 10.3
YA	NF	22.6 ± 4.7	642.7 ± 449.7	30.3 ± 22.9
YA	PL	24.6 ± 6.21	373.1 ± 50.7	15.8 ± 3.4
RA	NF	61.1 ± 27.2	388.1 ± 205.1	8.2 ± 7.3
RA	PL	40.2 ± 15.4	375.7 ± 164.3	9.5 ± 3.3

Different letters indicate differences among treatments.

[a] We only display letters for sites were significant differences existed between forest type

4. Discussion.

The contrasting clay contents and mineralogy of these soils offers a range of mechanisms for soil organic matter physicochemical stabilization and protection, which could explains some of the differences in C and nutrient stocks found in the studied soils (Torn et al., 1997; Six et al., 2002; Denef & Six, 2005). The soils of native forests display significant differences in C contents and stocks, with decreasing stocks from YA, GR, SCH, OA, and RA in the shallower soil compartment (< 120 cm). Despite the differences among sites, C stocks in the deep soil contribute significantly to the total soil C stocks (up to 40 %), and it was especially relevant in the least reactive soils (namely GR, OA and RA). The ranges in C stocks reported here are slightly higher than others in the literature, which could be expected since we directly measured C (not modeled) and we included deeper soil compartments. Liao et al. (2010) reported ranges that average 103.9 and 128.8 Mg ha^{-1} for planted and native forests in the first 100cm, respectively. However, as mentioned by these authors, most research to date had only considered shallower depths averaging 33cm and most of their values for deeper compartment were modeled. In our study we purposely sampled deeper depths since these contrasting soils should display different capacity to store carbon at deeper compartments. The soils in this study presented a range of clay contents and mineralogy that included soils dominated by: HIV, vermiculite and kaolinite (GR), kaolinite and chlorite (SCH), short-range order minerals (YA), halloysite and goethite (OA), and amorphous iron and aluminum oxy-hydroxides and cristoballite (RA) (Aburto, et al In prep). Interestingly, although clay contents were the highest in SCH and OA, these soils did not display the highest C stock, highlighting the relevance of soil mineralogy on the stabilization potential for long term C storage over total clay content (Feng et al., 2013). The C pools of the soils included in this study did not equally respond to the perturbations produced by native forest conversion to exotic planted forests. For example, we found no significant differences in C pools in the SCH, OA, YA, and we only found a net significant decrease in C stock in the RA and GR soils under plantation.

The presence of non-crystalline minerals in the YA soils (Besoain 1985) with high surface area and variable charge provided a higher C stabilization (Torn et al., 1997; Garrido & Matus, 2012), and resulted in a far higher C stock than any other soil in our study, disregarding of management intensification effect (ranging from 244 to 247 Mg ha^{-1} in NF and PL, respectively, in the upper 120 cm). Similarly, Cromack et al. (1999) found that soils with andic properties could store approximately 176 Mg ha^{-1} of C in the upper 100cm of a Douglas fir plantation which was similar to what was stored by other natural forest in the Coast of Oregon (USA). Lal et al. (2013) emphasized the relevance of clay activity in C storage, indicating that volcanic origin soils and spodosols displayed the largest C stock among all soils in temperate forests (138 and 143 Mg ha^{-1}, respectively). According to Matus et al. (2006) and Panichini et al. (2017), the strong organo-mineral interaction that occurs in Andisols stabilizes the C and N pools even under intense perturbations. The fact that YA does not display a significant change in C stock between forest types can be interpreted as a naturally higher resilience of these soils in more intensively managed plantations. The specific mechanisms that provide this higher potential biogeochemical resilience must be evaluated.

As C contents depends strongly on the surface reactivity of the clay fractions, soils with relatively lower clay content (RA) or with dominant low activity clays (like kaolinites and vermiculite) (GR) displayed the largest alteration after forest conversion (Torn et al., 1997;Six et al., 2002; Lützow et al., 2006). Rasmussen et al. (2006), in an incubation experiment, found that granitic soils with secondary mineralogy dominated by HIV and Vermiculite displayed a low carbon protection potential compared to soils dominated by short-range order minerals in a soil derived from andesite.

The RA soil had a very low clay fraction content (< 5%) dominated by amorphous low activity minerals with low plant available water (approx. 3 % vol.) and cation exchange capacity (approx. 2 cmol^{+}Kg^{-1}) (Aburto et al., In prep.), which consequently limits productivity reducing C and N inputs and stocks. The low productivity at this site can also be partially attributed to a naturally reduced productivity at this relatively colder high elevation site (Table 1). In spite of this, the overall significant lower soil C and N stocks in RA corroborate the importance of intrinsic soil properties on controlling the magnitude of these pools in forest soils (Denef et al., 2004).

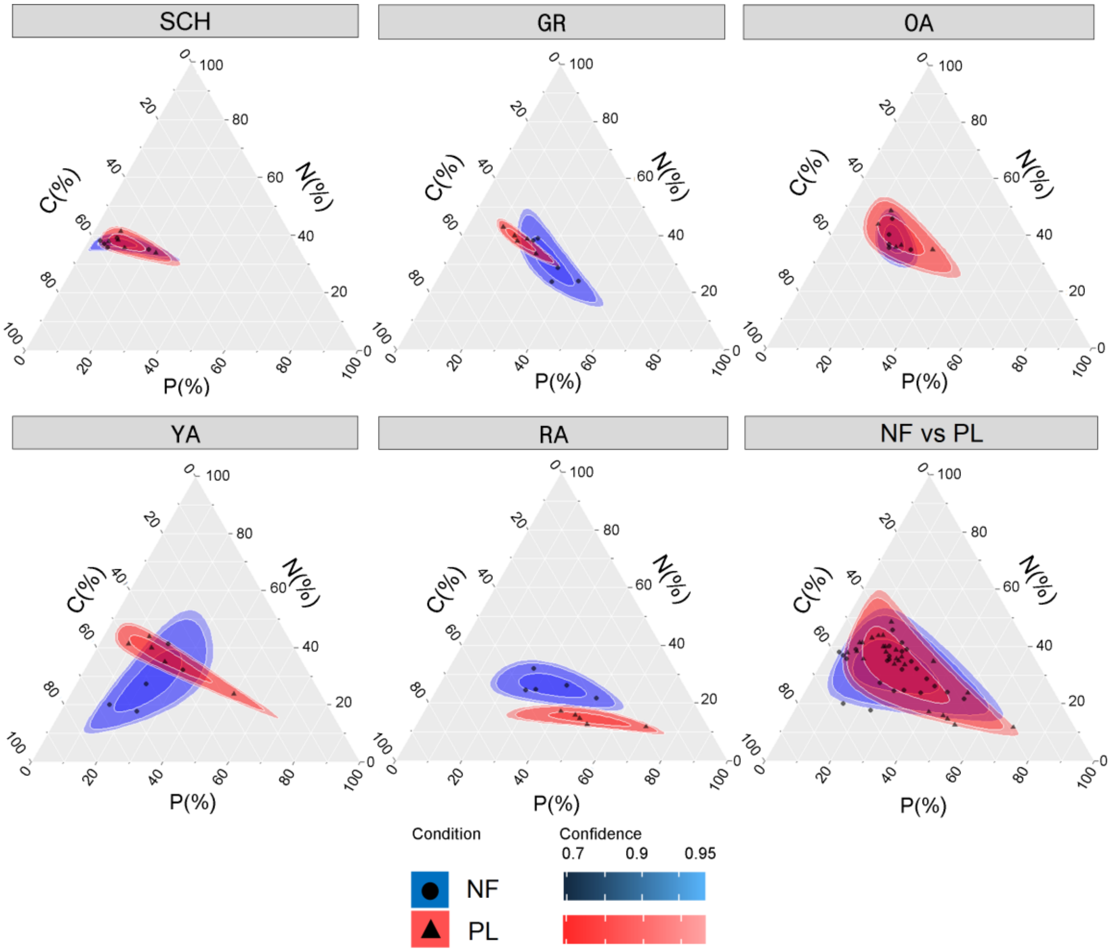

Figure 6. Stoichiometry ratios between conditions for each soil type. Each elemental content was scaled by dividing by the maximum observed value for each element and then the relative proportion between C,N,P stocks was standardized from 0-1 allowing the construction of the ternary plot (R's ggtern package Hamilton and Ferry, 2018.

Both forest conditions selected in this study were affected by recent human disturbances and all planted sites have maintain this condition for at least 40 years (except for RA). Therefore, plantation in most sites are between their second rotation. Yet, it is possible that in some other land-uses may have occurred more than 50 years ago (except in RA were plantations were directly established after a wildfire that burned the native forest in 1996). Commercial pine plantations in the Coastal Range of Chile were generally established since 1940´s in eroded lands (Toro and Gessel, 1999), but in many other places they also directly replaced degraded native forests (Heilmayr et al., 2016). Natural temperate forests in the Andes piedmont were heavily logged for firewood and clear for grazing by local farmers, consequently succeeding to shrubs and secondary forests as they were abandoned (Toro and Gessel, 1999). These former land-uses could have a legacy effects on C, N and P stocks and stoichiometry as reported by Spohn et al. (2016). In our study we tried to avoid soil alterations triggered by a previous land-use by comparing pair adjacent sites that were likely affected or pressured by similar land-uses. In fact, we only compared plantation sites to secondary forest that may have also been subjected in some extent to similar human perturbations (logging or grazing). So, we purposely avoided old-growth stands as other studies have used. We believe this approach allows a fairer and more direct comparison between forest types.

Liao et al. (2010) in a metanalysis estimated that native forest substitution by plantation reduced the C stock in soils by 32%. Chen et al. (2016) reported a significant and persistent reduction in C stocks after replacement of native evergreen broadleaf forest by Chinese fir plantations after 88 years of plantation in Southern China. They attributed SOC losses to increase soil respiration induced by augmented soil temperature and change in moisture dynamics produced by logging. Similarly, Zhang, et al. (2019) showed that intensively managed eucalyptus forests, even after 56 years, still maintain a lower C stock than natural forests. In addition, they showed that SOM in plantations have lower C recalcitrance than restored native forest and primary forest. Moreover, Liao et al. (2010) showed that most conifer plantation forests have 11, 34 and 66% lower above-ground NPP, litterfall and fine root production than broadleaf temperate forests. The lower C input, lower SOM recalcitrance and greater induced respiration could explain the lower whole soil C stock on pine plantation sites. The observed reduction in C stock by Chen et al. (2016) could also be a result of hindered plantation productivity induced by low soil nutrient availability than reported (N and P) because of the lack of fertilizer addition in these systems. We did not observe a clear reduction in soil N and P availability in most of our plantation sites, probably because fertilization is a widespread practice in most pine plantations in Chile (Toro and Gessel, 1999). However, it is important to note that we detected a reduction in the average total stock of N in three of our sites (RA, OA and GR), which were also the sites that displayed a lower C stock under plantations. Correspondingly, the two sites that displayed an increment in N stock, YA and SCH, also presented and increment in the average C stock under plantations. This was similar for the P stock, which was also higher in planted forests at these sites (YA and SCH).

There was no clear trend regarding the depth pool of C stocks in most sites. For example, GR displayed significant differences at deeper depth-intervals, while in OA the main differences occurred in the surface two depth-intervals (S. Figure 1). RA have significantly lower C and N content at almost all depths under PL, which reveal the profound effect of forest conversion in this soil type. The fact that C and N displayed significant differences in the same intervals (Figure 4) highlight the strong interrelationship between these pools (Tuo et al., 2018). As deciduous broadleaf trees allow a greater throughfall in winter season than evergreen coniferous trees, this additional water may enhance downward migration of dissolved organic carbon, which fueled deeper microbial activity and interact with minerals in the deeper compartments, increasing C sequestration (Deng et al., 2016).

Different to C and N, P stocks do not depend as strongly on SOM cycling (Feng et al., 2016) and as a result total P content does not always display the same depth distribution patterns. P distribution changes markedly between soils types, for example, in SCH, P displayed the lowest whole soil stock and the lowest content in the surface in both forest types. P at this site increased almost linearly with depth especially at the planted forest soils, which is mostly the result of an increase in the inorganic P component (not shown). In RA, P stock displayed a bi-modal vertical distribution with higher concentration on the surface depth-interval, a decrease at intermediate intervals and an increase at deeper layers (S. Figure 1) being the result of higher organic P in the surface and inorganic P in the deeper soil compartments (not shown). Even though higher litterfall and surface root inputs could increase organic phosphorus in the topsoil, the active exploitation of P by vegetation tended to produce a net decrease of P in the topsoil (Chen et al., 2016b). Therefore, significant differences found between PL and NF in the P depth-distribution in GR could be either the result of a more active deep exploration and absorption of P resources under plantation or augmented P losses from the soil system due to leaching or erosion in this highly erodible soil. On the other hand, the fresh volcanic materials in RA contained mostly inorganic P and the changes in compartmentalization mostly occurred in the surface depth-interval under PL as a result of an increment of organic P in the 0-20 cm interval and a depletion in the 20-40 cm interval mostly likely as a result of augmented root P mobilization.

Among sites, SCH shows the lowest available P stock (Figure 5), which was expected considering this site is deeply weathered (Ultisol) and it has also the lowest P stock (Table 4). Moreover, these soils are shallower, having a fractured bedrock contact approximately 120 cm deep. We did not see a significant trend in the P availability of volcanic soils, the highest available P stock was found in OA (Alfisols), followed by YA (Andisols) and RA (Entisols) under NF (Figure 5). YA soil is dominated by SRO minerals and low crystallinity Fe-oxides, so lower values were expected because of a higher P-fixing potential. However, the lowest values were found in RA most likely as result of low retention capacity of this soil because of a limited amount of reactive surfaces (low clay and SOM). Most soils under NF presented lower available P than PL, which may respond to a higher compartmentalization (absorption and accumulation) in vegetation (not shown). The higher biodiversity found in NF, where different assimilation strategies and root structures should be more efficient at acquiring phosphorus (Oelmann et al, 2011). Moreover, we could expected a higher P availability in PL as product of fertilization, but this effect should be relatively smaller since all plantations were mature and most P added by fertilization should have been already immobilized by vegetation, permanently fixed to mineral surface or leached. It is important to note that this was not the case for the SCH soils, where average available P pools were larger under NF ($p= 0.054$). A potential reason for this is a comparatively faster SOM mineralization at this site (not shown).

Available NO_3^- was a more relevant pool than NH_4^+ in all studied sites. However, we did not find significant differences between forest types for available NO_3^-. On the contrary, NH_4^+ stock responded to forest conversion displaying a higher whole soil NH_4^+ stock under PL at GR, OA and YA, being statistically significant in RA and OA (Figure 5). These differences are not likely the result of fertilization since these are adult plantations and fertilizer additions mostly occurred during plantation and only rarely during mid-rotation. Therefore, the higher availability in plantation is most likely the result of a higher nitrogen reabsorption and improved resource acquisition efficiency in the more diverse native deciduous broadleaf forests as well as higher N mineralization than in pine plantations, as it has been observed in other studies (Cromack et al., 1999 and Parfitt et al., 2003).

We found significant differences in the litter layers N stock between forest types in GR and YA soils. The higher N stocks in the litter can be related to the higher whole soil N stock in the NF condition for both sites. The greater N stock in NF did not result in a change in the C:N stock stoichiometry for these two sites (Table 7,8 and 9). In fact, the only site that displayed a significant shift in litter C:N stock was

RA, where PL displayed a significantly higher C:N ratio compared to the litter in NF.

The lack of significant differences in the C and nutrients stocks in the organic horizon between forest types was unexpected, considering that other authors have shown significant differences in C contents and stocks between pine and temperate broadleaf deciduous forest floors (e.g. Parfitt, 2003 and Rahman and Tsukamoto, 2013). Figure 6 shows a visual representation of the changes in C:N:P weighted average contents at each soils site, while a similar figure but calculated based on the elemental stocks was included in Supplemental Figure 2. It is evident from these representations that soil stoichiometry at each site did not respond equally to the native forest conversion to pine plantations. As with total C and N stocks, the GR and RA display a significant change in C:N:P stoichiometry visualized as separation of the confidence ellipsoid in Figure 6. Similarly, YA PL also show some departure from the NF C: N: P relationship, mostly as a result of a greater variability in the C content in NF and P content in PL. When grouping all NF and PL condition across sites, the differences in C:N:P stoichiometry became less evident, which could be expected considering the wide range of soil types we included in this study.

Quan et al. (2014) reported significant differences in C:N litter stoichiometry between broadleaf and evergreen conifers that resulted in relevant differences in C mineralization and C stocks. GR was the only site that showed significant differences in the C:P stock ratio between forest types, which is consistent with the significant differences in mineral soil P and C stocks between NF and PL found at this site. The relatively weak effect of forest condition on the organic horizon C:N:P ratio compared to soil type, suggest that soil type is the main controlling defining organic horizon stoichiometry, and forest type is only secondary across forest systems. On the other hand, litter stoichiometry was significantly different between soil and forest types, both controlling its composition.

Finally, C:N ratios consistently changed between conditions being significantly lower in PL (Table 9) as a result of the consistently higher C pool exhibited by NF across sites, which agrees with previous reports (Feng et al., 2016). For C:P and N:P no clear trend is found between conditions. Ouyang et al. (2017) described significant shift in the mineral soil C:N:P content along a forest succession from conifer to a broadleaf dominant forest. They also showed that broadleaf forests displayed a significantly higher soil C and N stocks, but lower soil P and as a result a higher C:N:P ratio. We also find a reduction in C and N stocks in most sites, but we did not find a consistent reduction in P after forest conversion along the five contrasting soils we studied. This suggests that the underlying effect of forest site (i.e. soil type and climate) may be a more relevant factor in defining P stocks and C:N:P stoichiometry than forest type.

When analyzing C: N: P ratios between conditions (Figure 6), we can reveal that some soils respond very differently to forest replacement (could also be seen when comparing C: N, C: P and N: P individually). For example, biogeochemical pools at the RA soils seems to be more sensitive to forest cover changes, where all stocks show a consistent reduction in planted forest (Figure 6). GR soil also exhibited greater sensitive to land-use changes, where we also see a drop in C and N. On the other hand, the YA soil in particular, show differences distributions rather than in their total stock, and the whole system seems to have reach a new biogeochemical C:N:P quasi-equilibrium. However, it is important to consider that in this study we have only included total and available stocks in soils, and have neither assess changes in other relevant pools like microbial biomass nor SOM quality or recalcitrance, which should be assess to determine if these stocks are equally stable in different forest types. For OA and SCH, changes in C: N: P stoichiometry are very minimal (Figure 6), suggesting that land-cover changes have only marginally affected this sites, which could also be interpreted as a higher resilience to C: N: P biogeochemical alterations.

5. Conclusions

Our paired comparisons of the effect of native broadleaf forest conversion to pine plantation allow us to explore the differential capacity that contrasting soils have to resist this major soil biogeochemical pools alteration. We provide evidence that total soil C, N and P reservoirs vary significantly between soil type and that the magnitude of the C:N:P stoichiometry modification caused by native forest conversion to plantation vary greatly between soil types. Soils with dominant crystalline low activity clays responded more strongly to forest plantation conversion displaying a net decrease in total C. Because the response to forest conversion across sites was not homogenous, C:N ratio did not display significant differences between forest types. Similarly, the C:P ratio was not equally altered across sites showing from not change to both a reduction (RA) and an increment (GR) in plantations. We also showed that plantation forests modified the elemental C, N and P vertical distribution in most sites, but C and N responded more strongly than P. In most soils under forest plantation C and N tend to be lower compared to native forest, especially in the upper 120cm.

Available nutrient pools are also significantly different between forest type, displaying a complex shift, generally showing a rise on available nitrate and phosphorous in most soils under plantation. Interestingly, organic horizons stoichiometry does not show differences between forest types or soil types and litter layer C:N:P ratios responded differently to forest substitution across sites.
The greatest changes in C:N:P contents and pools stoichiometry in the mineral soils occurred at sites with low clay content (RA, Entisols) and dominant low activity clay (GR, Alfisol), while ash-derived soils (YA, Andisol) and soils with dominant high activity clays (SCH, Ultisol and OA, Alfisol) displayed less alteration.

Our results highlight the importance of soil properties on determining C and nutrient stocks and provide valuable information for land managers to determine what soil types could be more susceptible to C and nutrient losses after conversion to planted forests. Future research should further explore on the biogeochemical mechanisms of mineral control on C stabilization and nutrient retention in forest soils and on the effect of forest management intensification and specific practices on C and nutrient stoichiometry.

Acknowedgements

The authors thank Alejandro Atenas and Maria Fernanda Albornoz for their help in the laboratory analyses and all the students, colleagues and friends for their help during field work. Two anonymous reviewers for the valuable comments and thoughts which greatly improved the manuscript. We also would like to thank CONAF Chile for allowing us to establish two paired plots in national protected areas (R.N Nonguen and R.N Malalcahuello) and also Forestal Arauco S.A and CMPC holding for allowing us to establish the other three paired plots in their patrimony. This study was supported by CONICYT Chile by FONDECYT INICIACIÓN grant n° 11160372.

6. References

Aburto F. Crovo, O. Albornoz, F. and Southard, R. (In Prep). Soil mineralogical and physicochemical alterations caused by forest substitution in contrasting soils of south central Chile.

Beaudette, D. E., Roudier, P., & O'Geen, A. T. (2013). Algorithms for quantitative pedology: a toolkit for soil scientists. Computers & Geosciences, 52, 258-268.

Besoain, E. Los Suelos. In: Tosso, J. (1985). Suelos Volcanicos de Chile. 25-95. 723 p.

Bowman, R., & Moir, J. (1993). Basic EDTA as an extractant for soil organic phosphorus. Soil Science Society of America Journal, 57(6), 1516-1518.

Brockerhoff, E. G., Jactel, H., Parrotta, J. A., Quine, C. P., & Sayer, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity?. Biodiversity and Conservation, 17(5), 925-951.

Cao, Y., & Chen, Y. (2017). Ecosystem C: N: P stoichiometry and carbon storage in plantations and a secondary forest on the Loess Plateau, China. Ecological Engineering, 105, 125-132.

Carlson, K. M., Curran, L. M., Asner, G. P., Pittman, A. M., Trigg, S. N., & Adeney, J. M. (2013). Carbon emissions from forest conversion by Kalimantan oil palm plantations. Nature Climate Change, 3(3), 283.

Carrasco, P., & Millán, J. (1990). Proyecto de suelos forestales de la VIII Región. Informe final. Universidad de Concepción, Dpto. de Cs. Forestales/Min. de Agricultura. Chillán, Chile.

Carter, M.R. and Gregorich, E.G. (2008). Soil sampling and methods of analysis 2nd ed. Canadian Society of Soil Science. CRC Press Taylor & Francis Group. ISBN-13:978-0-8493-3586-0

Centro de Información de Recursos Naturales Naturales, (CIREN). (1999). Estudio agrológico VIII región: CIREN Santiago, Chile.

Chen, X., Chen, H. Y., Chen, X., Wang, J., Chen, B., Wang, D., & Guan, Q. (2016). Soil labile organic carbon and carbon-cycle enzyme activities under different thinning intensities in Chinese fir plantations. Applied soil ecology, 107, 162-169.

Cleveland, C. C., & Liptzin, D. (2007). C: N: P stoichiometry in soil: is there a “Redfield ratio” for the microbial biomass? Biogeochemistry, 85(3), 235-252.

Cleveland, C. C., Houlton, B. Z., Smith, W. K., Marklein, A. R., Reed, S. C., Parton, W., Running, S. W. (2013). Patterns of new versus recycled primary production in the terrestrial biosphere. Proceedings of the National Academy of Sciences, 110(31), 12733-12737.

Crovo, O. Aburto, F. Rubilar, R. (In Prep.) Biogeochemical C, N, P dynamics shifts as a result of secondary native Nothofagus sp. forest substitution by Pine sp. exotic plantations.

Denef, K., & Six, J. (2005). Clay mineralogy determines the importance of biological versus abiotic processes for macroaggregate formation and stabilization. European Journal of soil science, 56(4), 469-479.

Denef, K., Six, J., Merckx, R., & Paustian, K. (2004). Carbon sequestration in microaggregates of no-tillage soils with different clay mineralogy. Soil Science Society of America Journal, 68(6), 1935-1944.

Deng, L., Zhu, G. Y., Tang, Z. S., & Shangguan, Z. P. (2016). Global patterns of the effects of land-use changes on soil carbon stocks. Global ecology and conservation, 5, 127-138.

Dixon, R. K., Solomon, A. M., Brown, S., Houghton, R. A., Trexier, M. C., Wisniewski, J. (1994). Carbon pools and flux of global forest ecosystems,*Science*, *263*(5144), 185-190.

Domec, J. C., King, J. S., Ward, E., Oishi, A. C., Palmroth, S., Radecki, A., McNulty, S. G. (2015). Conversion of natural forests to managed forest plantations decreases tree resistance to prolonged droughts. Forest Ecology and Management, 355, 58-71.

Feng, D., Bao, W., & Pang, X. (2017). Consistent profile pattern and spatial variation of soil C/N/P stoichiometric ratios in the subalpine forests. Journal of Soils and Sediments, 17(8), 2054-2065.

Feng, W., Plante, A. F., & Six, J. (2013). Improving estimates of maximal organic carbon stabilization by fine soil particles. Biogeochemistry, 112(1-3), 81-93.

Fox, G. A., Negrete-Yankelevich, S., & Sosa, V. J. (Eds.). (2015). Ecological statistics: contemporary theory and application. Oxford University Press, USA.

Fox, J., Weisberg, S., Adler, D., Bates, D., Baud-Bovy, G., Ellison, S., Graves, S. (2012). Package ‘car’. Vienna: R Foundation for Statistical Computing.

Gärdenäs, A. I., Ågren, G. I., Bird, J. A., Clarholm, M., Hallin, S., Ineson, P.,Kätterer, T., Knicker, H., Nilsson, S.I., Näsholm, T., Ogle,S., Paustian, K., Persson, T., Stendahl, J. (2011). Knowledge gaps in soil carbon and nitrogen interactions–from molecular to global scale. Soil Biology and Biochemistry, 43(4), 702-717.

Garrido, E., & Matus, F. (2012). Are organo-mineral complexes and allophane content determinant factors for the carbon level in Chilean volcanic soils? Catena, 92, 106-112. doi:https://doi.org/10.1016/j.catena.2011.12.003

Gaveau, D. L., Locatelli, B., Salim, M. A., Yaen, H., Pacheco, P., & Sheil, D. (2019). Rise and fall of forest loss and industrial plantations in Borneo (2000–2017). Conservation Letters, 12(3), e12622.

González-Ferrán, O. 1995 Volcanes de Chile. Instituto Geográfico MIlitar. ISBN 9562020541. 640p.

Gregorich, E. G., & Carter, M. R. (2007). Soil sampling and methods of analysis. CRC press.

Guillaume, T., Damris, M., & Kuzyakov, Y. (2015). Losses of soil carbon by converting tropical forest to plantations: erosion and decomposition estimated by δ13C. Global change biology, 21(9), 3548-3560.

Guo, L. B., & Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta-analysis. Global Change Biology, 8(4), 345-360. doi:10.1046/j.1354-1013.2002.00486.x

Heilmayr, R., Echeverria, C., Fuentes, R., & Lambin, E. F. (2016). A plantation-dominated forest transition in Chile. Applied Geography, 75, 71-82.

Heuck, C., & Spohn, M. (2016). Carbon, nitrogen and phosphorus net mineralization in organic horizons of temperate forests: stoichiometry and relations to organic matter quality. Biogeochemistry, 131(1-2), 229-242.

Huang WJ, Zhou GY, Liu JX (2011) Nitrogen and phosphorus status and their influence on aboveground production under increasing nitrogen deposition in three successional forests. Acta Oecologica-International Journal of Ecology 44: 20–27.

Kay, M., & Wobbrock, J. O. (2019). Package ‘ARTool’.

Lal, R. (2013). Soil carbon management and climate change. Carbon Management, 4(4), 439-462. doi:10.4155/cmt.13.31

Liu, X., Meng, W., Liang, G., Li, K., Xu, W., Huang, L., Yan, J. (2014). Available phosphorus in forest soil increases with soil nitrogen but not total phosphorus: evidence from subtropical forests and a pot experiment.*PLoS One*,*9*(2).

Lützow, M. V., Kögel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner, B., & Flessa, H. (2006). Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions–a review. European journal of soil science, 57(4), 426-445.

Manzoni, S., Trofymow, J. A., Jackson, R. B., & Porporato, A. (2010). Stoichiometric controls on carbon, nitrogen, and phosphorus dynamics in decomposing litter. Ecological Monographs, 80(1), 89-106.

Matus, F., Amigo, X., & Kristiansen, S. M. (2006). Aluminium stabilization controls organic carbon levels in Chilean volcanic soils. Geoderma, 132(1), 158-168. doi:https://doi.org/10.1016/j.geoderma.2005.05.005

McGroddy, M. E., Daufresne, T., & Hedin, L. O. (2004). Scaling of C: N: P stoichiometry in forests worldwide: Implications of terrestrial redfield-type ratios. Ecology, 85(9), 2390-2401.

Müller, M., Oelmann, Y., Schickhoff, U., Böhner, J., & Scholten, T. (2017). Himalayan treeline soil and foliar C: N: P stoichiometry indicate nutrient shortage with elevation. Geoderma, 291, 21-32.

Neculman, R., Rumpel, C., Matus, F., Godoy, R., Steffens, M., & de la Luz Mora, M. (2013). Organic matter stabilization in two Andisols of contrasting age under temperate rain forest. Biology and fertility of soils, 49(6), 681-689.

Ouyang, S., Xiang, W., Gou, M., Lei, P., Chen, L., & Deng, X. (2018). Variations in soil carbon, nitrogen, phosphorus and stoichiometry along forest succession in southern China.

Olander LP, Vitousek PM (2000) Regulation of soil phosphatase and chitinase activity by N and P availability. Biogeochemistry 49: 175–190.

Panichini, M., Neculman, R., Godoy, R., Arancibia-Miranda, N., & Matus, F. (2017). Understanding carbon storage in volcanic soils under selectively logged temperate rainforests.*Geoderma*,*302*, 76-88.

Peñuelas, J., Sardans, J., Rivas-ubach, A., & Janssens, I. A. (2012). The human-induced imbalance between C, N and P in Earth's life system. Global Change Biology, 18(1), 3-6.

Post, W. M., & Kwon, K. C. (2000). Soil carbon sequestration and land-use change: processes and potential. Global Change Biology, 6(3), 317-327. doi:10.1046/j.1365-2486.2000.00308.x

Quan, Q., Wang, C., He, N., Zhang, Z., Wen, X., Su, H., & Xue, J. (2014). Forest type affects the coupled relationships of soil C and N mineralization in the temperate forests of northern China. Scientific reports, 4, 6584.

Rasmussen, C., Southard, R. J., & Horwath, W. R. (2006). Mineral control of organic carbon mineralization in a range of temperate conifer forest soils. Global Change Biology, 12(5), 834-847.

Rasse, D. P., Rumpel, C., & Dignac, M. F. (2005). Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. Plant and soil, 269(1-2), 341-356.

Ribeiro Jr, P. J., Diggle, P. J., Ribeiro Jr, M. P. J., & Suggests, M. A. S. S. (2007). The geoR package. R news, 1(2), 14-18.

Sadzawka, A., Carrasco, M., Grez, R., Mora, M., Flores, H., & Neaman, A. (2006). Métodos de análisis recomendados para los suelos de Chile. Serie actas INIA, 163.

Schoeneberger, P. J., Wysocki, D. A., & Benham, E. C. (Eds.). (2012). Field book for describing and sampling soils. Government Printing Office

Miranda, K. M., Espey, M. G., & Wink, D. A. (2001). A rapid, simple spectrophotometric method for simultaneous detection of nitrate and nitrite. Nitric oxide, 5(1), 62-71.

Six, J., Conant, R. T., Paul, E. A., & Paustian, K. (2002). Stabilization mechanisms of soil organic matter: implications for C-saturation of soils. Plant and soil, 241(2), 155-176.

Soil Survey Staff. (2014). Keys to soil taxonomy. Department of Agriculture: Natural Resources Conservation Service. USA.

Spohn, M., & Sierra, C. A. (2018). How long do elements cycle in terrestrial ecosystems? Biogeochemistry, 139(1), 69-83.

Spohn, M., Novák, T. J., Incze, J., & Giani, L. (2016). Dynamics of soil carbon, nitrogen, and phosphorus in calcareous soils after land-use abandonment–A chronosequence study. Plant and soil, 401(1-2), 185-196.

Sterner, R. W., & Elser, J. J. (2002). Ecological stoichiometry: the biology of elements from molecules to the biosphere. Princeton university press.

Team, R. C. (2013). R: A language and environment for statistical computing.

Treseder KK, Vitousek PM (2001) Effects of soil nutrient availability on investment in acquisition of N and P in Hawaiian rain forests. Ecology 82: 946–954.

Torn, M. S., Trumbore, S. E., Chadwick, O. A., Vitousek, P. M., & Hendricks, D. M. (1997). Mineral control of soil organic carbon storage and turnover. Nature, 389(6647), 170.

Tosso, J. (1985). Suelos volcánicos de Chile. Ministerio de Agricultura. 1a edición. Instituto de Investigaciones Agropecuarias (INIA). Santiago.

Tuo, D., Gao, G., Chang, R., Li, Z., Ma, Y., Wang, S., Wang, C., Fu, B. (2018). Effects of revegetation and precipitation gradient on soil carbon and nitrogen variations in deep profiles on the Loess Plateau of China. Science of The

Total Environment, 626, 399-411. doi:https://doi.org/10.1016/j.scitotenv.2018.01.116

Vitousek PM, Porder S, Houlton BZ, Chadwick OA (2010) Terrestrial phosphorus limitation: mechanisms, implications, and nitrogen-phosphorus interactions. Ecological applications 20: 5–15.

Watanabe, F. S. and S. R. Olsen 1965. Test of an ascorbic acid method for determining P in water and NaHCO3 extracts from soil. Soil Science Society of America Proceedings 29: 677-678.

Yee, T. W. (2010). The VGAM package for categorical data analysis. Journal of Statistical Software, 32(10), 1-34.

Zhang, J., Zhao, N., Liu, C., Yang, H., Li, M., Yu, G., Wilcox, K., Yu, Q., He, N. (2018). C: N: P stoichiometry in China's forests: From organs to ecosystems. Functional ecology, 32(1), 50-60.

Zhou, GY., Liu, SG., Li, ZA., Zhang, DQ., Tang, XL., Zhou, C., Yan, J., Mo, J. (2006). Old growth forest can accumulate carbon in soils. Science, 314, 1417.

III. Effects of native forest replacement to exotic plantations on forest dynamics and C, N and P total pools in south-central Chile[2].

Abstract

Native forest substitution by managed tree plantations can significantly alter carbon and nutrient biogeochemical pools as a result of changes in forest dynamics and biogeochemical inputs. To evaluate the magnitude of these alterations we quantify the main C, N and P pools and fluxes in paired plots of secondary deciduous native forests and exotic pine plantations plots for five contrasting soils in south-central Chile. Two adjacent paired plots were established, where forest main fluxes were monitored for two years. We quantified litterfall, litter decomposition, soil CO_2 efflux, LAI, root annual production, DOC, Nitrate, Ammonium and DTP on leachates and standing biomass. Input fluxes did not shown statistically differences between forest type (p= 0.13), while output C fluxes also did not shown statistically differences between forest type (p=0.43). Overall ecosystem C storage (soil and aboveground biomass) showed no differences between forest type (p=0.07). Of the Measured variables only root production showed a positive correlation ($R^2 = 0.49$) with soil total C (p=0.001). Root at NF showed a significantly higher annual root production (1.76 ± 0.99 Mg ha^{-1}) than PL (0.81 ±0.88 Mg ha^{-1}) (p=0.0001). Our results suggest that exotic pine plantations enhance some biogeochemical fluxes such as litterfall and litter decomposition, but also diminish others such as root production. Overall, total ecosystem carbon followed by a forest replacement did not show a significant decrease. Continues monitoring of these plots should help to address temporal variability and sensitivity to future changes in environmental conditions.

Keywords: forest nutrient stocks; forest biogeochemistry; forest conversion; use intensification; land-use change; temperate forest.

---

[2] Crovo O, Aburto F, Albornoz MF, Rubilar R, Southard R. 2020. Effects of native forest replacement to exotic plantations on forest dynamics and C, N and P total pools in south-central Chile. In prep. Forest Ecology.

1. Introduction

The overall storage of carbon on land depends on the equilibrium between primary production and decomposition (Schlesinger and Bernhardt, 2013). Forest ecosystems around the globe contains about 1146 Pt of carbon (Dixon et al., 1994) were over 2/3 of this carbon in forest ecosystems is contained in soils, and being the largest terrestrial C pool (IPCC, 2013) play a significant role in global biogeochemical cycles (Bockheim and Gennadiyev, 2010). Natural factors such as temperature and precipitation are the main factors that modulate net biotic C sequestration and SOC accretion (Jobbagy and Jackson, 2000; Eclesia et al., 2012), while many anthropogenic drivers such as deforestation and land-use change can promote a general SOC decline (Lal et al., 2005).

The net effect of native forest conversion to industrial tree plantations on SOC is still matter of discussion. Some findings indicate a higher SOC stock in plantations than in natural forests (Jobaggy and Jackson, 2000; Chen et al., 2005; Fialho and Zinn, 2012), while more recent studies indicate that natural forests shows significant higher stock of SOC than productive plantations (Liao et al., 2012; Chen et al., 2016; Cao and Chen, 2017). Other authors postulate that differences in SOC vary with plantation age and management (Guo and Gifford, 2002; Yanai et al., 2003; Guan et al., 2019).

Forest replacement not only affects the total C stock but also the N and P stock (Ross et al., 1999; Guan et al., 2015). For example, Yan et al. (2008) explained that evergreen forest replacement with monospecific forest such as Chinese fir and productive bamboo plantations produce a shift in the organic matter input resulting in declines of total C and N. These changes were attributed by a reduction in the litter quality and micro climatic changes induced by shifts in plant species composition (Yan et al., 2008). Guan et al. (2015) found the same trend of decline in total C and N in the mid-subtropical region of China after secondary forest replacement with productive plantations, but changes were associated to management practices (i.e burning and clear-cutting) rather than forest dynamics. Similar findings are described in Ross et al. (1999), where a decline in soil total C and N were found after replacing a native temperate forest for a *Pinus radiata* plantation in New Zeland. The pine plantation exhibited higher CO_2 losses in comparison with the native forest. Authors have also found that total soil P stock increases after forest replacement by plantations due to phosphate fertilization (Ross et al., 1999; Wang et al., 2017; Cai et al., 2018)

Is clear that forest replacement process entails practices such as fertilization and tillage which may accelerate the mineralization and leaching of SOC and other nutrients (Mancinelli et al., 2010; Peñuelas et al., 2012), but isn't still clear what forest processes regarding shift in species composition have direct impact on C, N and P soil total stocks. While is obvious that processes such as clear cutting and the removal of understory vegetation in plantations may decrease C inputs (Cai et al., 2018), authors of the mentioned above studies agree that concomitant measures of litter and root production could benefit their investigations regarding to C decline after forest replacement (Kasel and Bennett, 2007 ; Guan et al., 2015).

In Chile, since 1974, a package of governmental policies and subsidies expanded rapidly productive plantations (Nahuelhual et al., 2012) resulting in 1.7 million planted hectares and representing approximately 17 % of all forested lands in Chile. Near 64% of them are concentrated in the south-central region of Chile (CONAF, 2011), and most corresponding to Pinus sp (i.e *Pinus radiata* D.Don) (INFOR 2008). This expansion drove significant native forest loss by conversion to plantations in the last quarter of the 20th century and the first decade of the 21th century (Echeverria et al., 2006; Aguayo et al., 2009; Miranda et al., 2015; Heilmayr et al., 2016). Over the last decade significant research has focused on elucidating the effects on biodiversity loss by Chilean native forest replacement (Altamirano and Lara, 2010), stream water quality and regulation (Huber et al., 2010), carbon and nutrient losses on river discharges (Oyarzun et al., 2007 ; Perez et al., 2015) and forest fragmentation (Echeverria et al., 2006). Nevertheless, the effect of this conversion on biogeochemical C (carbon), N (nitrogen) and P (phosphorus) dynamics and stocks has been understudied not only for Chilean temperate forest but also in temperate regions around the world (Cao and Chen, 2017). This, in spite of that forest productivity and nutrient sustainability are highly dependent on carbon and major nutrient pools (i.e N and P). Understanding the main differences in biogeochemical cycle dynamics of plantations and natural systems may provide a baseline for sustainability assessment of forest management practices. While there are a few recent studies that aim to elucidate the effect of forest substitution in major biogeochemical stocks and dynamics, only a few have included soil type as a factor that could control the magnitude of the response. In our study we determined the modulation effect of soil type on the response of whole forest biogeochemical pools and dynamics induced by native broadleaf (*Nothofagus sp.*) forest conversion to exotic coniferous plantations (*Pinus sp.)* in the south-central temperate region of Chile.

We hypothesized that forest conversion to exotic plantations, followed by management intensification, could alter carbon and major nutrients pools by creating imbalances in productivity and decomposition along with nutrient leaching and CO_2 fluxes. We also predicted that higher plant diversity and diverse forest structure at native forests will exhibit higher root production and therefore higher SOC and total ecosystem carbon.

2. Materials and methods

2.1 Study sites.

To determine how forest conversion has modified C, N and P stocks we selected five sites in the south-central region of Chile (Figure 1). Two soils are residual soils located in the Coastal Mountain Range from mica-schist (SCH) and granitic parent materials (GR). In addition, three sites corresponding to recent (RA), young (YA), and old volcanic ash (OA), are soils formed from pyroclastic materials (basaltic-andesitic) (Besoaín, 1985; CIREN, 1999). Within these soil types, areas were selected where forest substitution occurred at least 35 years ago based on interviews to local forest managers . This was further assessed by remote sensing analyses using time sequence Landsat thematic mapper (Landsat 5) images from year 1986 to 2011 with a 30 meters special resolution.

2.2 Plot preparation and stand dynamics measurements.

Once the areas where selected, field recognition targeted homogenous areas where native forest (NF) and exotic plantations (PL) occurred at a minimum of ten meters to a maximum of a few hundred meters. To reduce variability of selected plots, stand characteristics such as dominant tree species, stand age, height and breast height diameter, soil type and geomorphologic variables such as slope class and landscape position were measured (Table 1). At each location, 25m by 25m plots were established at each forest condition (plantation and native). A total of 10 plots (5 pairs) were established for all five soil-site conditions (Figure 1). At the center of each plot one soil pit was hand-dug up to a depth of 240 cm. Each soil pit was fully described and sampled by soil genetic horizons following the standard protocol described by Schoeneberger et al. (2012). Soil description and other soil properties are included in a companion paper (Aburto et al., In prep.). Plots were subdivided in four 12.5 x 12.5 m quadrants and soil auger samples were obtained at six depth intervals (0-20, 20-40, 40-80, 80-120,120-180 and 180-240 cm) at the center of each quadrate. Samples from these plots were used to estimate C, N and P reservoirs for the litter layer, organic and mineral horizons and the discussion of results are presented in a companion paper (Crovo et al., In review). Here we focus on the effect of forest replacement on litterfall, decomposition rates, roots annual production, CO_2 efflux, standing biomass and soil leachates in order to elucidate direction of change in C, N and P forest pools. At each quadrant, all assessments were measured bi-monthly over two years with the exception of soil leachates for which only one measurement was obtained bi-monthly at the center of each plot.

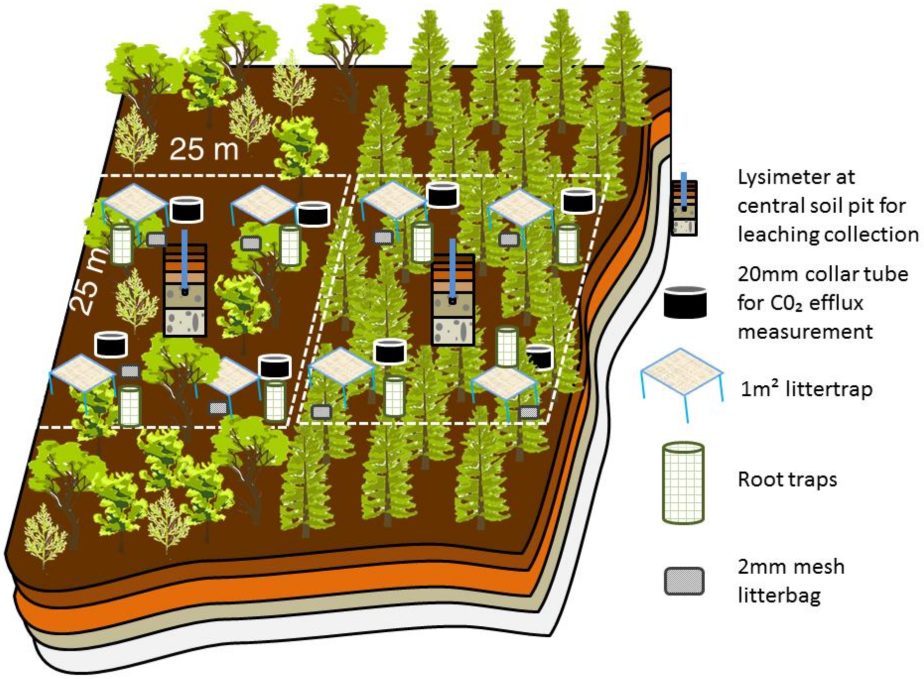

Figure 7. Paired (25 x 25m) plots and main assessments at each location. Main plots were split in four quadrants for CO2 flux, literfall, roots, decomposition and biomass assesments.

For each plot, diameter at breast height (DBH) and tree height of all trees were measured to estimate standing biomass (leaves, branches and stems) using published species-specific allometric equations (Gayoso, 2001; Gayoso. 2013; Milla et al., 2013; Valenzuela et al., 2017). Concentration of C, N and P of tree compartments were estimated by sampling leaves, branches and stem of the dominant species in each plot quadrant. In cases of a multi-species understory we sampled the three most representative species. Since most of the sites were on protected areas wood samples were obtained using an increment borer dug at breast height of 15 individuals at each subplot. A destructive harvesting method for leaves and branches at 2/3 of the tree crown was used, while for stem tissues increment borer samples were used. The C, N and P concentrations found in this ecosystem compartments where multiplied by the biomass of the same compartments in order to calculate the total C, N and P found aboveground. Samples were ground using a steel ball mill (Spex Certiprep 8000M©) after which total C and N were determined by Dumas combustion using a SERCON© EA CHNS analyzer. Total Phosphorus was determined colorimetrically in a nitro-vanadium-molibdate solution method (Sadzawka et al., 2006) using a Shimadzu UV-mini 1240 spectrophotometer (Watanabe & Olsen, 1965). All analysis were performed at the Soils, Water and Forest Research Laboratory (LISAB) of the University of Concepción (Concepción, Chile.).

Litterfall was collected bi-monthly from four 1 m^2 - 50 cm height collectors established at each plot (4 per main plot) from November 2017 to November 2019. After collection litterfall was transported to the laboratory and oven dried at 65°C to constant weight. In-situ decomposition rates were determined using 2mm mesh decomposition litterbags. Initially, 48 litterbags were filled with 10g of oven dried fresh leaf litter collected in July -2017 and installed in the forest floor of each plot quadrant in September – 2017 by grouping 12 litterbags per each sub-plot. Litterbags were collected bi-monthly from November 2017 to July 2019, at each sampling date one litterbag was collected per plot quadrant (four per plot) and oven dried to compare pre and post-decomposition sample mass. At each date leaf area index (LAI) was estimated also bi-monthly in order to assess stand productivity (Schlesinger and Bernhardt, 2013). Leaf area index was measured at each sub-plot using an automated fisheye lens equipped measuring device (Solariscope SOL300, Behling, Germany) that was placed at 1.8 meters height above the ground. Solariscope device calculates LAI estimates by taking a hemispherical photograph and subsequently analyzing it at seven standard contrast threshold values.

Root biomass was measured using in-growth cores (Marthews et al., 2013). At each plot-quadrant a 30 cm depth and 12 cm diameter wide hole was dug. Root material from the core sample was separated from the soil and carried to the laboratory for oven drying at 65°C up to constant weight in order to measure root biomass. At each root-free core hole, in-growth cores of 40 cm high and 12 cm diameter with a 1.5 cm^2 mesh were installed and covered up to the first 30 cm depth with root-free soil (10 cm of the in-growth was kept above ground for field recognition purposes). The cores were installed between July and September 2017 and removed in the same date for year 2018 in order to estimate annual root production. Once the first year of measurements was done, the in-growth core was re-installed with the same procedure and re-measured in 2019. In order to evaluate CO_2 efflux, a series of eight PVC collars (20 cm in diameter) per plot were installed into the soil (1 collar for each plot-quadrant). In order to separate autotrophic (root derived respiration) from heterotrophic respiration, a trenching technic approach was used (Subke et al., 2006), were four 40 cm high PVC collars were installed into the first 30 cm of soil, removing every root in this volume of soil and leaving a measurable 10 cm high PVC collar aboveground. For total respiration (root derived respiration + heterotrophic respiration) four 11 cm high PVC collars were installed into the soil to a depth of 1 cm with 1 to 2 meters separation from the heterotrophic PVC collar, depending on field conditions (i.e shrub and bamboos patches). Collars where installed between September and November 2017 and left for 2 month of stabilization prior field measurement. Field measurements were carried out with a 20 cm soil chamber and an IRGA (LI-8100A,Licor Inc., Lincoln, NE , USA) and measured bi-monthly between January 2018 and January 2020, considering three measurements a day for two minutes for every soil collar in each plot (morning, noon and night) for a better representativeness of the day CO_2 efflux. Field measurements were interrupted when field conditions were not appropriate (i.e rain, snow or hardware failure) but efforts were made to measure at least one observation point for each collar at each plot.

For soil solution sampling a ceramic cup lysimeter (SK-20, METER Group Inc, USA) was installed in the hand dug soil pit at the center of each plot to a depth of 1.8m. Soil solutions were obtained bi-monthly using a vacuum pump at a constant suction of 85 kPa for at least 8 hours, and collected in glass bottles during night and transported in closed cool chambers where they were protected from uv-light and sharp variations in temperature until arrival to the laboratory. Soil solution were analyzed for total organic carbon (TOC), dissolved organic and mineral nitrogen and phosphorus (TDN, DON, TDP). Samples where filtered through a 0.45 µm filter and analyzed for TOC in a TOC analyzer (Vario TOC Cube, Elementar, Germany). In addition, TON was measured in a SERCON© EA CHNS analyzer while NO_3^- and NH_4^+ were measured colorimetrically using a Shimadzu UV-mini 1240 spectrophotometer. DON concentrations were estimated by subtracting inorganic N concentrations (NO_3^- and NH_4^+) from total N concentrations. TDP was measured by colorimetry after a sulfiric acid digestion with an UV-visible spectrophotomer (UV-1700,Shimadzu Corporation, Kyoto, Japan).

2.3 Data treatment and statistical analysis

Litter collection and root collection were analyzed as replicates and averaged to create per-plot estimates for both years of study. Litterbag data was expressed as mass loss % of original mass remaining (% mass remaining) as a dependent variable and time (days) as an independent variable where the litter decomposition rates where determined using a single-pool exponential decay model (Olson, 1963):

$$M(t) = M(0)e^{-kt} \tag{1}$$

Where *M(t)*= mass at time *t*; *M(0)* = mass at time 0; *k*= exponential decay coefficient; and *t* = time in days.

Daily soil CO_2 efflux data (umol m^{-2} s^{-1}) was generated from a weighed average from three observations of daily measurements (morning, noon and night) in order to better represent the daily variations of site conditions that have influences on soil respiration such as soil moisture, soil temperature, air temperature and relative humidity (Perez-Quezada et al., 2016). Daily average of each bi-monthly sampling was extrapolated to the period and the sum of the period in one year where considered as the annual soil CO_2 efflux (Mg ha $^{-1}$ year $^{-1}$). Plot quadrant LAI estimates were analyzed as replicates and averaged to create a per-plot estimate for each year of the study. The Hydrus 1D model, based on Richard's equation for simulating one dimensional water flow (Simunek et al. 2008), was used to quantify water flow at the depth where lysimeters were installed at the center of each plot. Fluxes of dissolved elements (C, N and P) were calculated bi-monthly for each soil type and forest stand by multiplying nutrient concentrations in the soil solution bytotal flux of leached water estimated from the Hydrus 1D model.

Two-way ANOVAs were used for comparisons between soil types and to assess the effect of both treatments (native forest v/s pine plantation) on studied fluxes, following the general linear model of the "CAR" R package (Fox et al., 2012) followed by a posthoc multiple comparisons (LSD test) with $p \leq 0.05$ considered as the threshold <for statistical significance. Before performing ANOVAs, all the data was checked to meet assumptions of normality and homogeneity of variance. For non-normal data we first performed a Box-Cox transformation using the MASS package (Ripley et al., 2013). If after transformation assumptions were still not met, an aligned rank factorial analysis was carried out (Kay & Wobbrock, 2019). For soil leachates, statistical analysis of paired plots at each soil type were used as replicates for native forest and pine plantations stands respectively. Statistical differences between estimates were assessed by one way ANOVA when data was normally distributed, and when normality was not obtained a non-parametric test was used (Kruskall-wallis). Pearson correlation was used to test for the relationship between stand dynamics versus soil Carbon, Nitrogen, Phosphorus total pools and the relationship between the measured variables. Correlation coefficients with $p < 0.05$ were considered significant. All statistical analysis where performed using the R software (R Core Team, 2018).

3. Results

3.1 Stand characteristics and aboveground biomass.

Tree ring analysis proved that, in general, our study stands have at least 25 years of no logging activity or land use change (Table 10), with the exception of YA-PL that is the youngest plantation with 20 years. The five sites showed similar stand ages 20 to 26 years for forest plantations and 39-50 years for native forests. Among other characteristics, stand mean height showed to be pretty similar among the five native forest stands and the five pine plantations plots in contrast with basal area and stem density that showed more variation (Table 10). Managed PL stands at OA and YA showed the lower stockings (576 and 832 trees ha^{-1}, respectively) and basal area (34.16 and 23.7 m^{2} ha^{-1}, respectively), and the RA unmanaged PL stand showed the largest stocking with 2048 trees ha^{-1} and 111.1 m^{2} ha^{-1} of basal area. The *Nothofagus pumilio* stand showed the higher stocking and basal area values with 2736 trees ha^{-1} and 74.1 m^{2} ha^{-1}, respectively.

Table 10. Stand characteristics for each paired plot at each soil type (site).

Site	Forest Type	Dominant species	Tree height (m)	Stem density (trees ha-1)	G (m2 ha-1)	Age (year)	Management
SCH	Pine plantation	*Pinus radiata*	17 ± 3.1	1536	51.3	25	Unmanaged
	Native forest	*Nothofagus obliqua*	11.6 ± 4.6	1296	30.06	41	Unmanaged
GR	Pine plantation	*Pinus radiata*	16 ± 3.9	1600	53.84	26	Unmanaged
	Native forest	*Nothofagus obliqua*	10.9 ± 5.1	1312	56.87	39	Unmanaged
OA	Pine plantation	*Pinus radiata*	24.8 ± 1.6	576	34.16	26	Thining and prunning
	Native forest	*Nothofagus obliqua*	11.3 ± 5.3	1888	61.72	50	Unmanaged
YA	Pine plantation	*Pinus radiata*	13 ± 2.7	832	23.73	20	Thining and prunning
	Native forest	*Nothofagus obliqua*	10.9 ± 6.1	2400	32	43	Unmanaged
RA	Pine plantation	*Pinus ponderosa*	13.6 ± 3.4	2048	111.06	38	Unmanaged
	Native forest	*Nothofagus pumilio*	10.8 ± 3.8	2736	74.06	48	Unmanaged

SCH corresponds to the site in the mountain coastal ranges formed from schist and slate parent material.

GR corresponds to the site in the mountain coastal ranges formed from granite parent material.

OA corresponds to the site in the central valley formed from old ash deposits.

YA correspond to the site in the Cordillera piedmont formed from young ash deposits.

RA corresponds to the site in the Cordillera de los Andes formed from recent ash deposits (Andesitic-basaltic parent material).

Overall, in every single soil type (site) the native forest stands (NF) exhibit higher biomass than the plantation stands (PL) (Figure 8). For example in SCH, the native forest presents 487.3 Mg ha $^{-1}$ while the pine plantation shows an aboveground biomass of less than the half (201.45 Mg ha $^{-1}$). This proportion also occurs in YA, where the NF stand presents 216.47 Mg ha $^{-1}$ and the PL stand 81.78 Mg ha $^{-1}$. In both GR and OA the differences are yet higher, where the NF stands threefold their plantation pair with 636.73 versus 192.05 Mg ha $^{-1}$ in GR and 519.79 versus 173.44 Mg ha $^{-1}$ in OA respectively. In the recent ash (RA) site the stands presents a more equal quantity of aboveground biomass with 320.39 Mg ha $^{-1}$ in NF and 293.53 Mg ha $^{-1}$ in PL. When using the five sites as replicates for forest type the NF exhibit a significantly higher (p = 0.017) aboveground biomass than PL with 436.17 ± 166.98 versus 188.45 ± 75.54 Mg ha $^{-1}$ respectively. The stem is the compartment that has a greater contribution to the total aboveground biomass (Figure 9) from values that ranges from the 44% (NF-YA) to 71% (NF-GR) but with most values going around a 65% of the total.

For aboveground Carbon, in the recent ash (RA) site the PL stand showed more aboveground total Carbon (142.2 Mg ha $^{-1}$) than in NF (138.4 Mg ha $^{-1}$). For the Schist derived soil site (SCH) the NF stand yielded a twofold aboveground total Carbon in comparison with the PL stand (223.4 and 92.9 Mg ha $^{-1}$ respectively). For the granite derived soil site (GR) the amount of aboveground total Carbon in NF exceeds for four times the one founded in PL (384.7 and 89.9 Mg ha $^{-1}$ respectively). For the Old ash derived soil site (OA) a threefold in aboveground total Carbon is shown between the NF stand and PL stand, where for the NF stands exhibit a 230.3 Mg ha $^{-1}$ versus the 83.02 Mg ha $^{-1}$ found in PL. In the case of YA the NF a threefold in aboveground total Carbon is also shown where the NF stand exhibit a 93.04 Mg ha $^{-1}$ versus the 30.2 shown in PL. When using the sites as replicates for NF and PL the NF stands shows a significantly higher yield (p=0.044) in the aboveground total Carbon with 214 ± 111.6 than the PL stands 87.7 ± 39.8 Mg ha $^{-1}$.

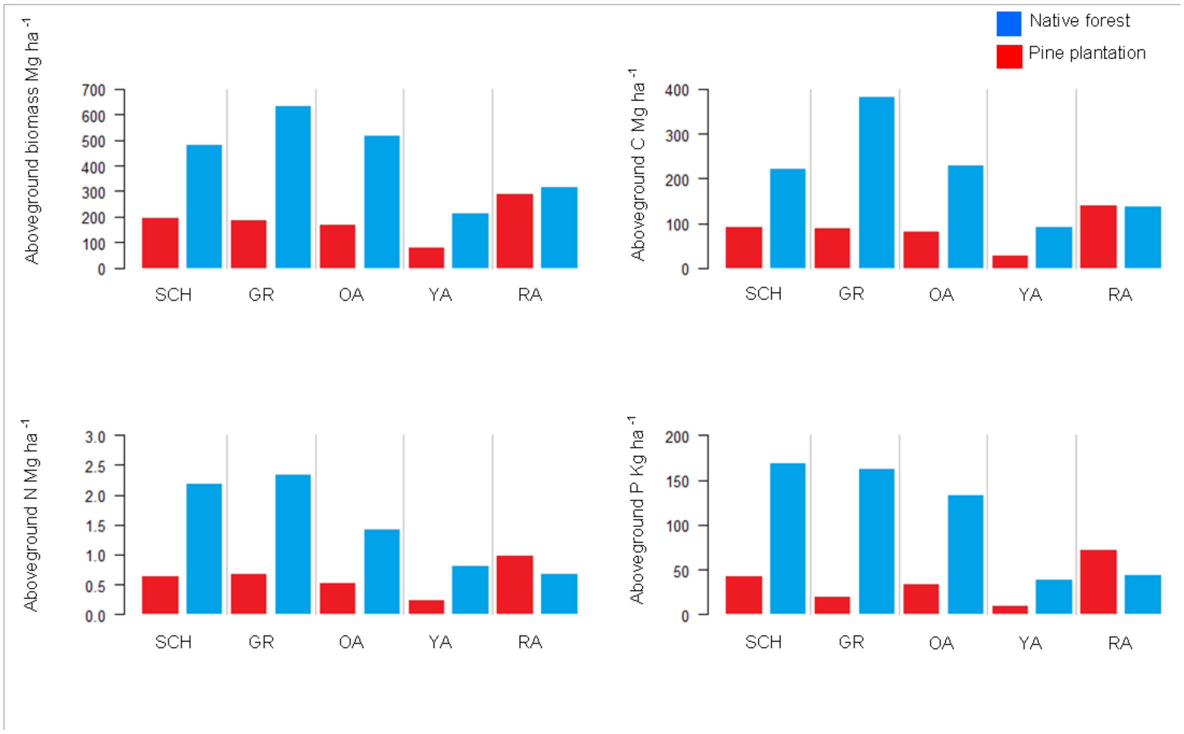

Figure 8. Aboveground biomass, Carbon (C), Nitrogen (N) and Phosphorus (P) for each stand (Native forest and Pine plantation) at each soil type.

A very similar trend is shown for aboveground total Nitrogen (Figure 8) where in almost every site the total Nitrogen pool is higher in the NF stands with again, RA being the exception where the PL stands shows a higher total N pool with 0.99 Mg ha $^{-1}$ in comparison with the 0.69 Mg ha $^{-1}$ founded in NF stand. In the rest of the site, a threefold higher total N aboveground pool is found in the NF stands in comparison with the NF. For example, SCH NF shows a 2.2 Mg ha^{-1} of aboveground total Nitrogen in comparison with the 0.65 founded in the PL stand of the same site. A very similar proportion is founded in the GR site, where the NF stand yielded a 2.36 Mg ha $^{-1}$ versus the PL stand with 0.69 Mg ha $^{-1}$ of the total aboveground N. The NF stand in the OA site exhibit a higher total Nitrogen pool in contrast with the PL stand (1.43 versus 0.55 Mg ha $^{-1}$ respectively) in the same way than in the YA site (0.82 versus 0.26 Mg ha $^{-1}$). When using the five stands as replicates, we found that NF (1.5 Mg ha $^{-1}$) exhibits a significantly higher ($p=0.042$) aboveground total Nitrogen than PL.

For total aboveground Phosphorus, the recent ash site (RA) is the only case where PL shows a higher total Phosphorus (72.9 Kg ha $^{-1}$) than NF (44.4 Kg ha $^{-1}$). The biggest difference is found in the GR site where NF shows 163.5 Kg ha $^{-1}$ and PL only 20.2 Kg ha $^{-1}$. While in SCH (170.3 versus 43.9 Kg ha $^{-1}$), OA (133.7 versus 34.4 Kg ha $^{-1}$) and YA (40.01 versus 10.8 Kg ha $^{-1}$) the total aboveground total phosphorus found is four times higher in NF than in PL respectively. Upon using the sites as replicates the NF forest type also shows a significantly higher (p=0.041) aboveground total phosphorus with 110.4 ± 63.8 Kg ha $^{-1}$ in contrast with the 36.4 ± 24 Kg ha $^{-1}$ found in PL.

When analyzing the concentrations of Carbon, Nitrogen and Phosphorus in different aboveground compartments (leaves, branches and stem) a two-way ANOVA(Site x Forest type) showed that in the case for the C concentrations in leaves no interaction among factors was found (p=0.17), and differences in concentrations where only found in NF (p= 0.024) (Table 11). An interaction effect between factors was found in the C concentrations in branches (p= 0.003) where the mean found in OA-NF (4.3 ± 2.7 %) was significantly lower than OA-PL (48.3 ± 0.5 %) and RA-NF (47.8 ± 0.7 %). No differences were found in the Carbon concentrations in the stem. For the Nitrogen concentrations found in the leaves, a significant interaction between factors was found (p= 0.002) where the principal differences are found between NF and PL in the different sites (Table 3). In the same way, the N concentration in branches has a significant interaction between the factors (p= 0.02) with differences between forest types in each site and also across sites (Table 11). No differences were found in the Nitrogen concentrations in the stem compartment. For the Phosphorus concentrations in the different forest compartments, no differences were found in the leaves, while in branches differences were found only between forest types (p= 0.04) where Phosphorus content in branches in the native forest stand is significantly higher (0.03 ± 0.01 %) than the one in the pine plantation stand (0.021 ± 0.01 %). The P concentrations found in the stem compartment shows an interaction between the main factors (p= 0.0001) with differences between forest types in each site and also across sites (Table 11).

Table 11. C, N and P concentrations (%) of each forest compartment (Leaves, branches and Stem) in each stand and each site.

Compartment	Forest Type	C Concentration (%)				
		SCH	GR	OA	YA	RA
Leaves	BN	46.8(0.5)**b**	43.7(2.2)**a**	45.7(0.8)**ab**	48(0.8)**b**	46(1.5)**ab**
	PL	46.4(1.3)	46(2.4)	46.4(1.4)	46.2(2.1)	46.3(0.9)
Branches	BN	47.1(1.1)**AB**	45.6(0.3)**AB**	43.3(2.7)**A**	46(1.1)**AB**	47.8(0.7)**B**
	PL	46.6(1.5)**AB**	46.5(2)**AB**	48.3(0.5)**B**	45.7(1.7)**AB**	45(2.2)**AB**
Stem	BN	46.7(1.4)	46.1(1.5)	46.7(1.5)	45.6(1.2)	47(1)
	PL	46.8(1.4)	46.3(1)	46.2(1.6)	47.8(1.2)	46.9(1.7)
Compartment	Forest Type	N Concentration (%)				
		SCH	GR	OA	YA	RA
Leaves	BN	1.43(0.17)**A**	0.78(0.08)**B**	1.06(0.44)**AB**	1.06(0.44)**AB**	0.83(0.17)**B**
	PL	1.28(0.28)**AB**	1.47(0.17)**A**	1.08(0.11)**AB**	1.19(0.07)**AB**	1.7(0.21)**C**
Branches	BN	0.89(0.36)**BC**	0.25(0.01)**A**	0.35(0.03)**A**	0.4(0.03)**AB**	0.47(0.29)**AB**
	PL	0.65(0.15)**AB**	0.73(0.17)**AB**	0.61(0.16)**A**	0.75(0.06)**AB**	0.88(0.01)**BC**
Stem	BN	0.07(0.01)	0.1(0.02)	0.07(0.01)	0.07(0.01)	0.07(0.01)
	PL	0.06(0.02)	0.04(0.01)	0.07(0.02)	0.08(0.04)	0.09(0.02)
Compartment	Forest Type	P Concentration (%)				
		SCH	GR	OA	YA	RA
Leaves	BN	0.08(0.01)	0.09(0.03)	0.08(0.004)	0.09(0.004)	0.08(0.01)
	PL	0.09(0.02)	0.07(0.02)	0.1(0.01)	0.09(0.01)	0.1(0.02)
Branches	BN	0.02(0.01)	0.03(0.01)	0.04(0.01)	0.03(0.01)	0.03(0.01)
	PL	0.02(0.01)	0.02(0.01)	0.02(0.01)	0.02(0.01)	0.02(0.01)
Stem	BN	0.03(0.003)**BC**	0.04(0.01)**BC**	0.05(0.01)**C**	0.007(0.006)**A**	0.04(0.01)**B**
	PL	0.007(0.006)**A**	0.01(0.02)**A**	0.01(0.002)**A**	0.02(0.01)**A**	0.01(0.003)**A**

Mean ± sd, different capitals letters indicates significant differences between means in the interaction among main effects (forests and soil types), and different lower-case letters indicates differences between means with no interaction.

3.2 Forest dynamics.

3.2.1 Litterfall

The results of the two way ANOVA indicates that there's a statistically significant interaction between the effects of Site (soil type) and forest type on total annual litterfall ($p < 0.0001$). Tukey's HSD post hoc test show us that the means where separated into 5 groups (Figure 9) where differences between site and forest type varied through the analysis. GR-PL showed the highest total annual mean with 1.17 ± 0.04 Mg ha^{-1} year^{-1} being statistically significant (p = 0.0002) with GR-NF (0.79 ± 0.07 Mg ha^{-1} year^{-1}); OA-PL 0.65 ± 0.04 Mg ha^{-1} year^{-1} and OA-NF 0.57 ± 0.11 Mg ha^{-1} year^{-1} with p < 0.0001 in both cases; with YA-PL 0.84 ± 0.09 Mg ha^{-1} year^{-1} and YA-NF 0.71 ± Mg ha^{-1} year^{-1} (p=0.0013 and p< 0.0001 respectively); finally with RA-PL 0.61 ± 0.16 Mg ha^{-1} year^{-1} and RA-NF 0.25 ± 0.02 Mg ha^{-1} year^{-1}. This last group (RA-NF) exhibits statically significant lower mean than YA-PL,YA-NF, SCH-PL,SCH-NF, GR-PL and GR-NF (p<0.0001) also with OA-PL (p=0.0001),RA-PL (p=0.0005) and OA-NF (p=0.0023).

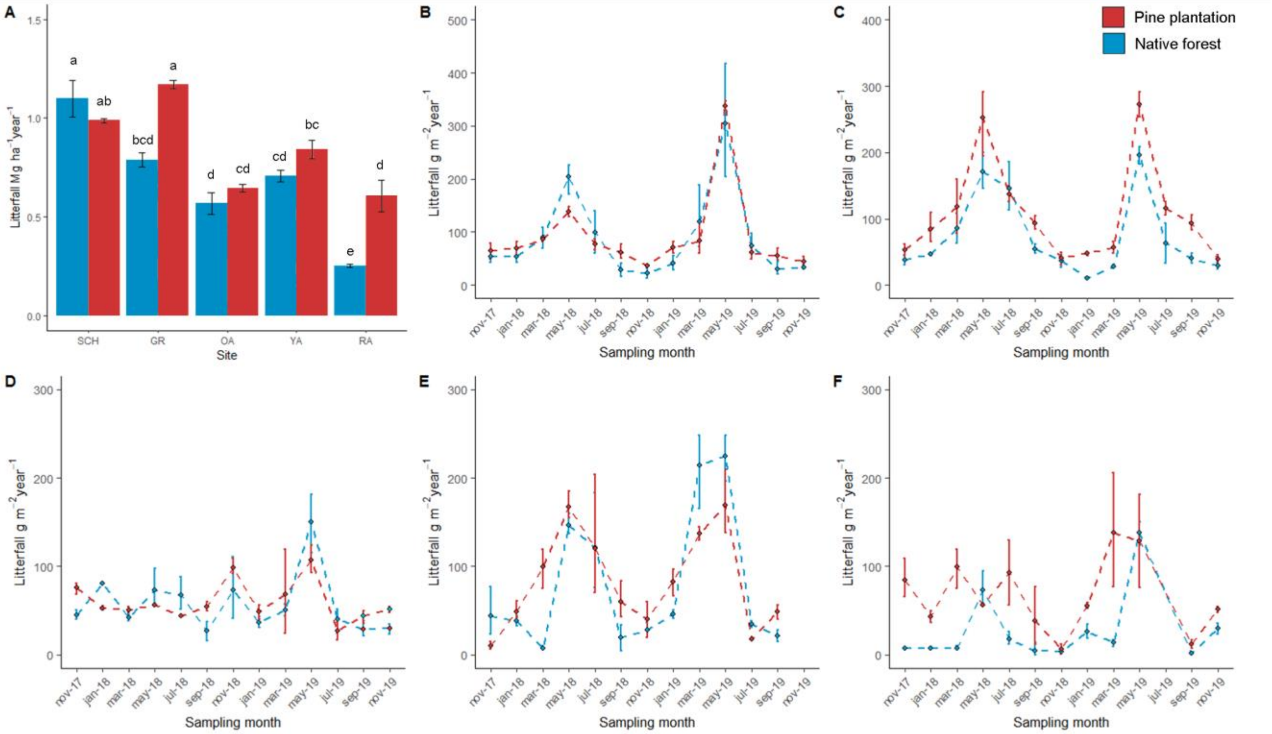

Figure 9. Annual litterfall for each paired plot at each site where different letters indicates significant differences (A) and Annual litterfall distribution for the five sites SCH(B),GR(C),OA(D),YA(E) and RA (F)

Upon analyzing the distribution of the litterfall in the two year period a very clearly season effect is shown between March to July in both years (2018 and 2019) when the peaks of litterfall collection are present (bi-modal distribution) for both PL and NF. Even though, NF has higher seasonal peaks than PL, with GR being the exception, the litter input through the year is higher for PL (Figure 9).

3.2.2 Litter decomposition

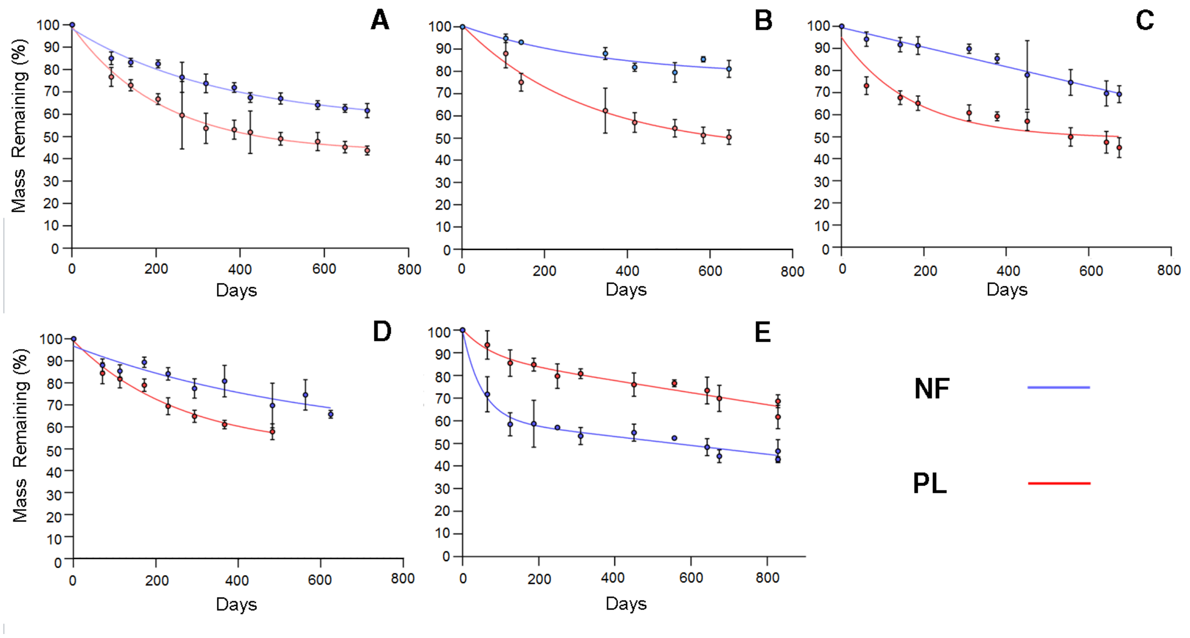

Figure 10. Litter Mass remaining (%) after 1 year of study for the paired plots at SCH (A), GR (B), OA (C), YA(D) and RA (E).

After the two-year period of decomposition, mean mass loss varied from 43 to 81.1 % (Figure 10). Decomposing litter mass was better described by the one phase decay exponential model (Figure 10, Table 12), with the exception of both paired plots in RA and OA where a double decay exponential model was more parsimonious. Given the fact that the response variable between models are different (k rate for the one phase exponential decay and a labile fraction decay (k_1) and recalcitrant fraction decay (k_2) for the two phase model) comparison between forest type were carried out at a site level and not across sites. For the schist derived soil (SCH) even though the decomposing leaf litter mass differ significantly between forest type ($p<0.001$) the decay rate (k) was not different between NF (0.0028 ± 0.001) and PL (0.0047 ± 0.001). The same response occurred for the granite derived soil (GR) where the significant differences between the leaf litter mass remaining ($p<0.001$) did not match with the decay rate (k) were the differences are not significant between NF (0.0029 ±0.001) and PL (0.0033 ± 0.001). For the old ash derived soil (OA) differences between the leaf litter mass remaining ($p= 0.02$) matches with differences in k_1 and k_2 ($p=0.0002$). While for the young ash derived soil (YA) no differences were found in the leaf litter mass remaining and in the decay rate. The leaf litter decay in the recent ash derived soil (RA) was better explained by a double exponential decay model for both NF and PL (Table 12), with no differences in in k_1 (p=0.25) and k_2 (p=0.34).

Table 12. Decomposition variables for each plot.

Site	Forest	MR(%)	One Phase exponentenial decay		Two phase exponential decay			R2
			k	AICc	k1	k2	AICc	
SCH	NF	61.6(3.2)	0.003(0.001)	**115.1**	0.12(0.11)	0.002(0.002)	117.7	0.93
SCH	PL	43.8(1.9)	0.005(0.001)	**171.8**	0.06(0.01)	0.002(0.002)	176.6	0.89
GR	NF	81.1(3.8)	0.003(0.001)	**80.1**	0.003(0.001)	0.0024(0.0016)	85.9	0.83
GR	PL	50.4(3.3)	0.003(0.001)	**112.7**	0.07(0.12)	0.0025(0.0016)	118.6	0.92
OA	NF	69.3(3.9)	0.000004(0.002)	147.2	0.00017(0.002)	0.0002(0.0003)	**145.6**	0.76
OA	PL	45.1(4.5)	0.006(0.002)	144.4	0.25(0.28)	0.00185(0.002)	**110.5**	0.88
YA	NF	65.7(1.8)	0.002(0.0006)	**145.7**	1.7(0.75)	0.00011(0.0002)	146.6	0.75
YA	PL	61.1(1.9)	0.004(0.0009)	**87.2**	0.12(0.17)	0.002(0.0019)	91.1	0.94
RA	NF	43(1.23)	0.011(0.002)	172.8	0.02(0.03)	0.004(0.003)	**160.4**	0.87
RA	PL	61.7(5.2)	0.002(0.001)	159.6	0.03(0.04)	0.0001(0.0002)	**157.9**	0.81

Mean (sd) for each decomposition variable, different letters indicates significant differences (p<0.05). Numbers in bold indicates the better fit exponential decay model.

3.2.3 Root biomass production.

Root data was squared root transformed to follow normality (p=0.05). No interaction among factor was observed (p=0.91), but each main factor was significantly different by itself (p<0.0001). Using the paired plots as replicates across the different soil types, differences between the forest types was observed where NF showed a significantly higher annual root production (1.76 ± 0.99 Mg ha^{-1}) than PL (0.81 ±0.88 Mg ha^{-1}). Among every soil type, NF showed a higher production than PL (Figure 11) but only in OA, NF (1.51 ± 0.56 Mg ha^{-1}) showed a statistically higher root annual production (p=0.03) than PL (0.37 ± 0.26 Mg ha^{-1}). The YA soil type showed the higher values for NF (3.26 ± 0.52 Mg ha^{-1}) and for PL (2.31 ± 1.56 Mg ha^{-1}) with no differences between them (p=0.33), followed by the GR paired plots with 2.2 ± 1.1 and 0.86 ± 0.41 Mg ha^{-1} for NF and PL respectively and no significantly differences between them (p=0.09). The SCH soil type showed the lowest values for NF (0.9 ± 0.45 Mg ha^{-1}) and a low root productivity in

PL (0.32 ± 0.17 Mg ha^{-1}) with no differences between them ($p=0.36$). The RA soil type is the least productive and showed a low value for NF (0.94 ± 0.53 Mg ha^{-1}) and the lowest for PL (0.19 ± 0.05 Mg ha^{-1}) with no differences between forest types ($p=0.11$).

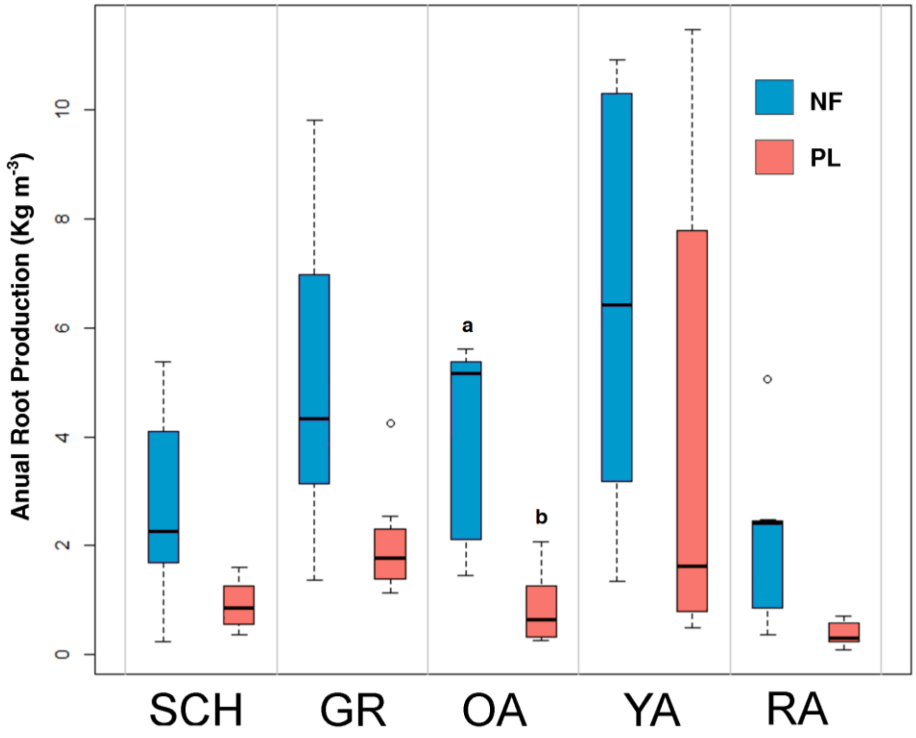

Figure 11. Annual root production for native forest (NF) and pine plantations (PL) at each site. Different letters indicates significant differences ($p<0.05$).

3.2.4 Soil CO_2 efflux.

For soil CO_2 efflux data no transformation passed the normality tests so an aligned rank transformed for nonparametric factorial ANOVA test was carried out. The analysis showed a highly significant interaction effect between factors ($p<0.0001$), Sidak's multiple comparison post hoc test showed that the means where separated into 3 groups (Figure 12) where differences between site and forest type varied through the analysis. The greatest soil CO_2 efflux was exhibited in GR-NF with 2.64 ± 0.23 Mg CO_2 ha $^{-1}$ year $^{-1}$, being significantly different ($p=0.0003$) with his paired-plot GR-PL (1.70 ± 0.14 Mg CO_2 ha $^{-1}$ year $^{-1}$). While the lowest was OA-PL with 1.49 ± 0.32 Mg CO_2 ha $^{-1}$ year $^{-1}$ with differences ($p=0.007$) with his paired plot OA-NF (2.04 ± 0.1 Mg CO_2 ha $^{-1}$ year $^{-1}$). The YA paired plots, was the only case where the PL stand showed a greater soil CO_2 efflux (2.19 ± 0.17 Mg CO_2 ha $^{-1}$ year $^{-1}$) than the NF stand (2.16 ± 0.13 Mg CO_2 ha $^{-1}$ year $^{-1}$) but with no differences between them (p>0.9). In a very similar trend, the paired plots in RA exhibit non-significant differences (p=0.3) between NF (1.54 ± 0.1 Mg CO_2 ha $^{-1}$ year $^{-1}$) and PL (1.52 ± 0.48 Mg CO_2 ha $^{-1}$ year $^{-1}$).

The SCH-NF (2.61 ± 0.6 Mg CO_2 ha $^{-1}$ year $^{-1}$) was the only plot not significantly different with GR-NF (p=0.9) neither with his paired plot SCH-PL (p= 0.45) that present values of 2.21 ± 0.25 CO_2 ha $^{-1}$ year $^{-1}$. Upon analyzing the annual distribution of the soil efflux it is shown that the CO_2 efflux increases between November and March (spring and summer) and start decreasing between the months of May and September (autumn and winter) with its lowest in July for both NF and PL (Figure 12). When separating the source of soil CO_2 efflux, the measurements from root exclusion (heterotrophic respiration) accounts for more than the 60% in all cases, where the values ranges from 62% to
85% of the total respiration

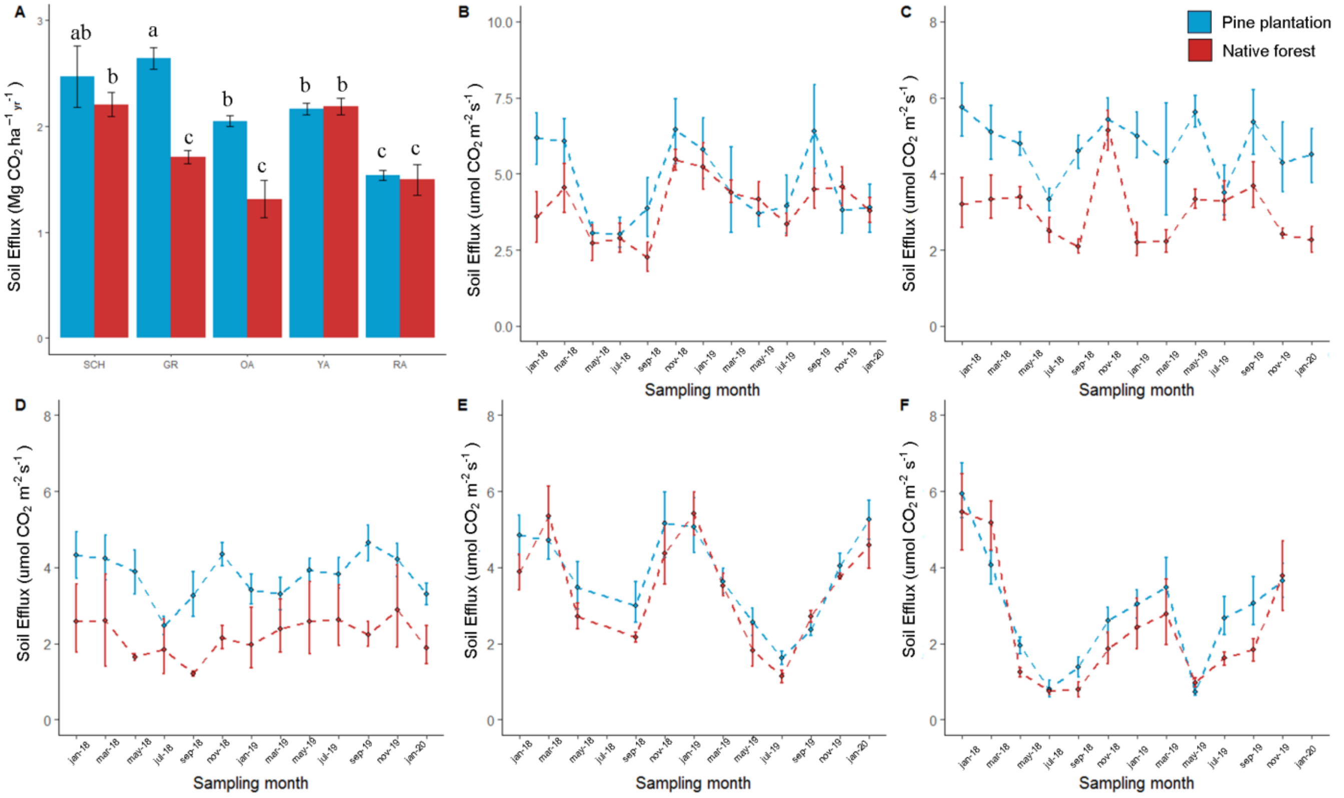

Figure 12. Annual soil CO_2 efflux for each paired plot at each site where different letters indicates significant differences (A) and Annual soil CO_2 efflux distribution for the five sites SCH(B),GR(C),OA(D),YA(E) and RA (F).

3.2.5 Leaf area index.

Because of contrasting canopy types considered in this study (deciduous versus evergreen) the comparison of the leaf area index (L.A.I) was carried out at the prime of the growing season (January) where the forest productivity is at its peak. Data showed to be normally distributed (p=0.104) and the ANOVA demonstrate a highly significant interaction effect between the main factors (p<0.0001). Sidak's multiple comparison post hoc test showed that the means where separated into 2 groups (Figure 13). The plot that showed the highest L.A.I was the GR-NF with 3.0 ± 0.6 being highly significant (p<0.0001) only with YA-PL (1.2 ± 0.1) and OA-PL (1.6 ± 0.1) while the his direct neighbor (GR-PL) showed a L.A.I of 2.7 ± 0.1 being highly significant only with YA-PL and OA-PL as well. The paired plots located in the schist derived soil showed no differences between them (p=0.9) with 2.9 ± 0.6 for NF and 2.8 ± 0.1 for PL and both showing significant differences only with OA-PL and YA-PL (p<0.0001). The paired plots located in the recent ash derived soil (RA) was the only case where the PL stand showed a higher L.A.I value (2.6 ± 0.3) than NF (2.5 ± 0.3) but no significant differences between was found (p=0.9) while both of them where only significantly different with YA-PL and OA-PL (p=0.0004 and p<0.0001 respectively). The young ash derived paired plots where highly significant between them (p<0.001) where the NF stand showed a much higher L.A.I value (2.7 ± 0.1) in comparison with PL (1.2 ± 0.1), the NF stand was also significantly different with OA-PL (p<0.0001). The OA-PL plot was not significantly different only with YA-PL (p=0.89) and showed a much lower L.A.I (1.6 ± 0.1) than his direct neighbor OA-NF (2.6 ± 0.6). When studying the year distribution of the L.A.I, big differences between all the NF and PL stands are shown despite the soil type. As the NF is dominated by deciduous trees, a considerable decrease of the L.A.I is shown between the months of March and September (Figure 8) showing his minimum in the month of July in the winter season. As expected, the value tends to increase gradually between September and January showing its peak in the growing season (November and January). As for the PL stands, the L.A.I follows a very similar trend in every soil type being constant through the year but with some small peaks in the months of November and January.

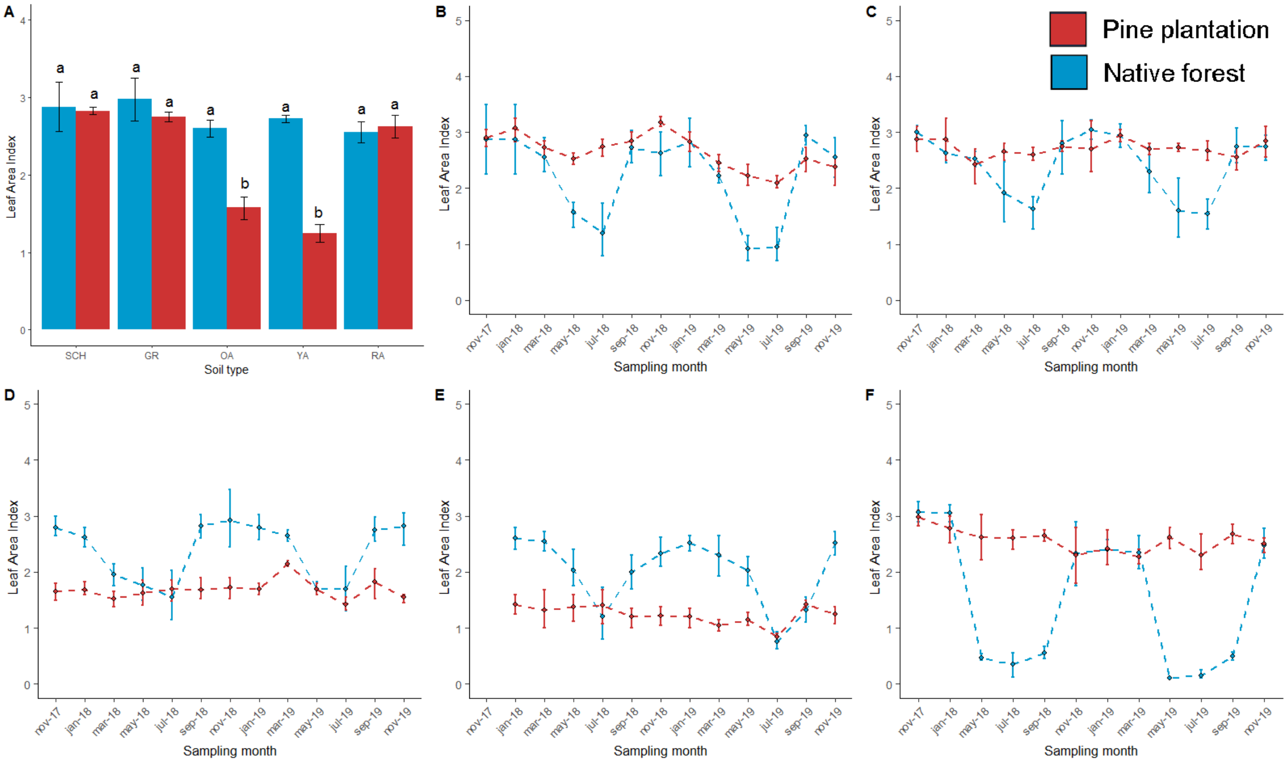

Figure 13. Leaf area index for each paired plot at each site where different letters indicates significant differences (A) and Leaf Area Index distribution for the five sites SCH(B),GR(C),OA(D),YA(E) and RA (F).

3.2.6 Soil leachates.

Dissolved organic carbon (DOC) follows a very rain dependent trend; where in the periods of higher precipitations (Complementary data) losses by DOC tend to augment (Figure 14). In most of the soil types, paired plots shows a peak in July, where The PL plots show higher DOC losses with values than ranges from 3.9 g m^{-2} in GR to 101.59 g m^{-2} in RA. On the other side, the NF plots show values that range from 0.86 g m^{-2} in GR to 13.13 g m^{-2} in YA. Upon comparison between NF with PL plots, even though the losses in PL are a fourth-fold higher (31.01 ± 40.8 g m^{-2} yr^{-1}) than the NF (7.9 ± 4.8 g m^{-2} yr^{-1}) no differences were found between them (p=0.35) due to the high standard deviation.

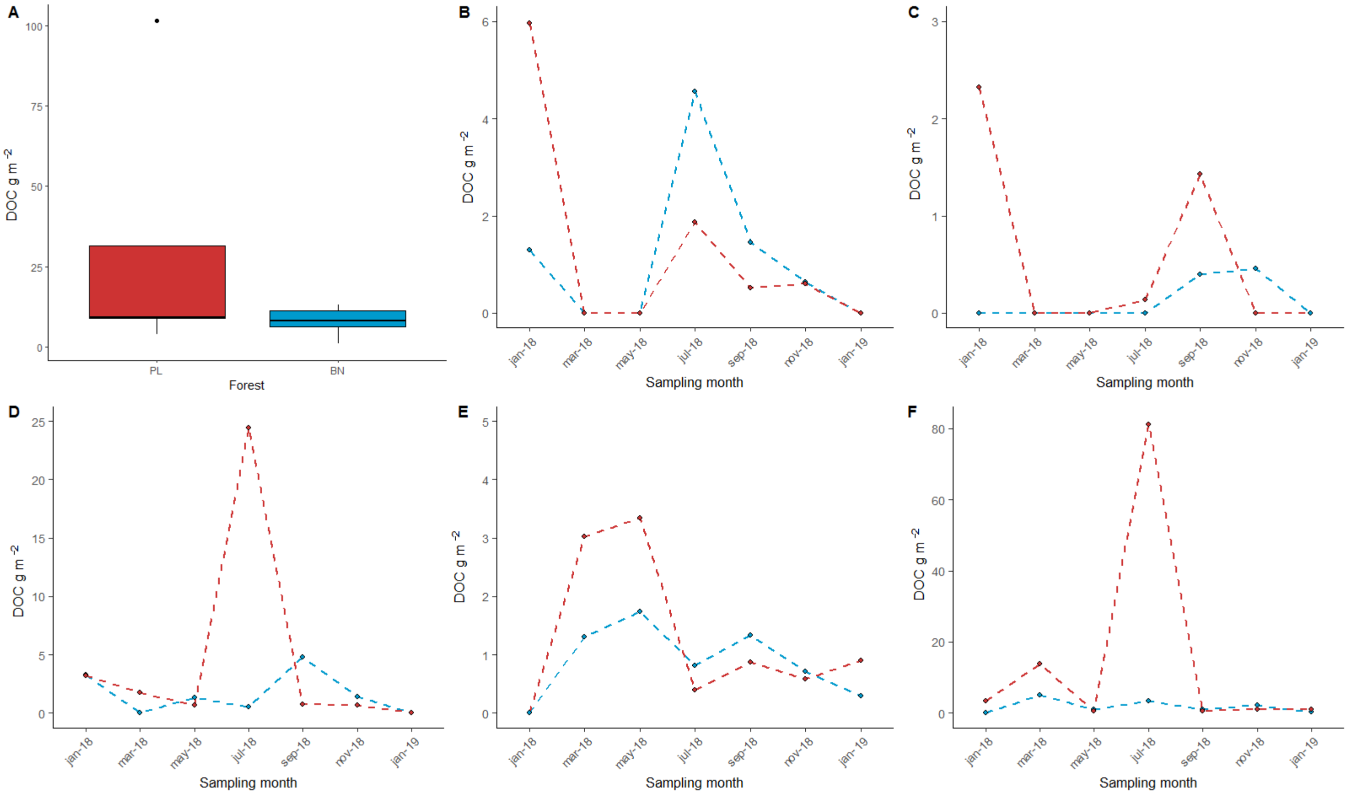

Figure 14. Annual dissolved organic carbon losses for each paired plot at each site (A) and annual dissolved organic carbon distribution for the five sites SCH(B),GR(C),OA(D),YA(E) and RA (F).

Nitrate leaching follows a similar trend for both forest type and soil type (Figure 14), where the peaks of Nitrate losses occurs at the end of the rain season, where YA-PL exhibits the highest monthly loss of 8.1 g m^{-2} followed by OA-NF with 5.18 g m^{-2} in the same period of the year. No statistical differences were found in the total annual Nitrate loss (p=0.92) between PL (4 ± 2.9 g m^{-2}) and NF (4.2 ± 2.4 g m^{-2}). For NH_4^+ the magnitude of the loss and the peaks are exhibits very similar trends between NF and PL in each of the soil types studied (Figure 16), the highest NH_4^+ loss was observed RA-NF in the month of march (0.9 g m^{-2}) followed by RA-PL in the same period of the year (0.26 g m^{-2}). No statistical differences (p=0.24) were found between NF (0.42 ± 0.14 g m^{-2}) and PL (0.21 ± 0.36 g m^{-2}) in the annual NH_4^+ loss although the observed losses were higher at the NF stands.

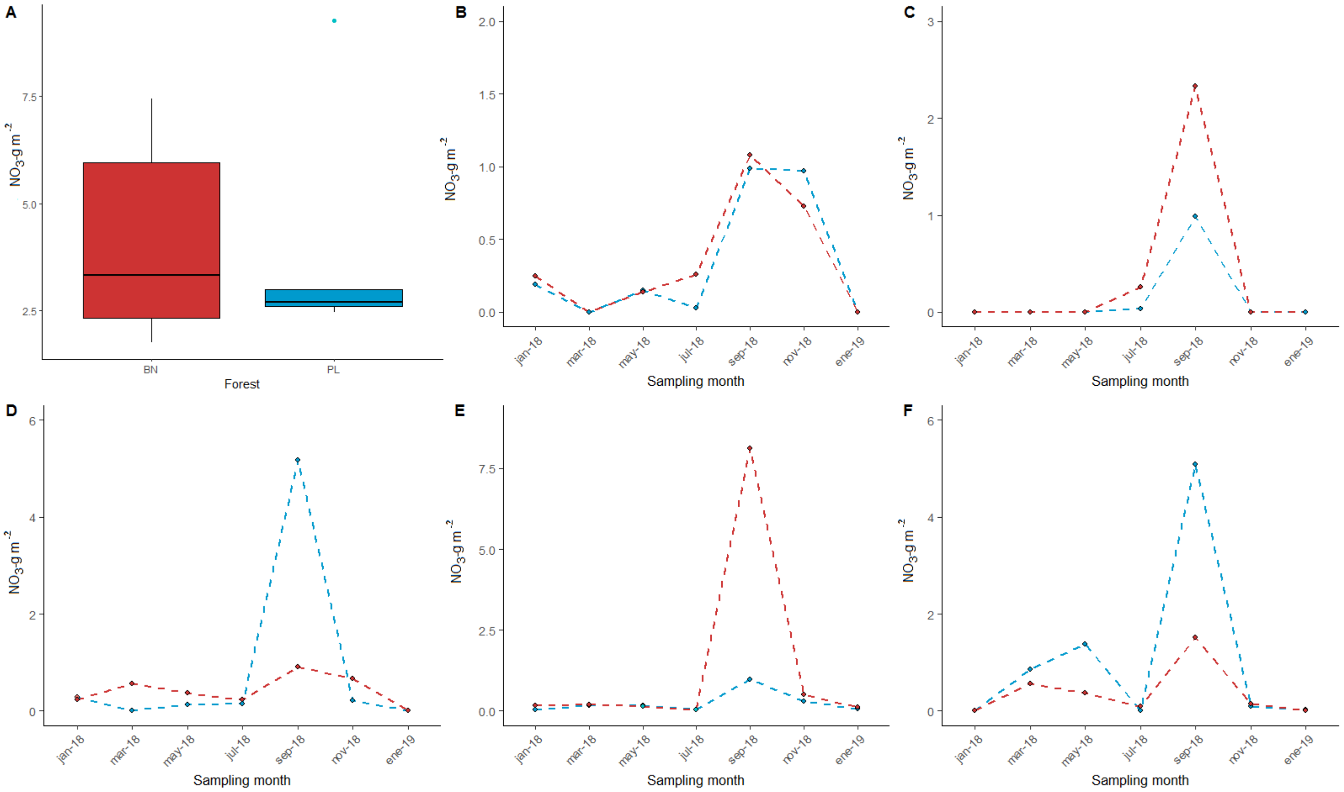

Figure 15. Annual NO_3+ losses for each paired plot at each site (A) and NO_3+ losses distribution for the five sites SCH(B),GR(C),OA(D),YA(E) and RA (F).

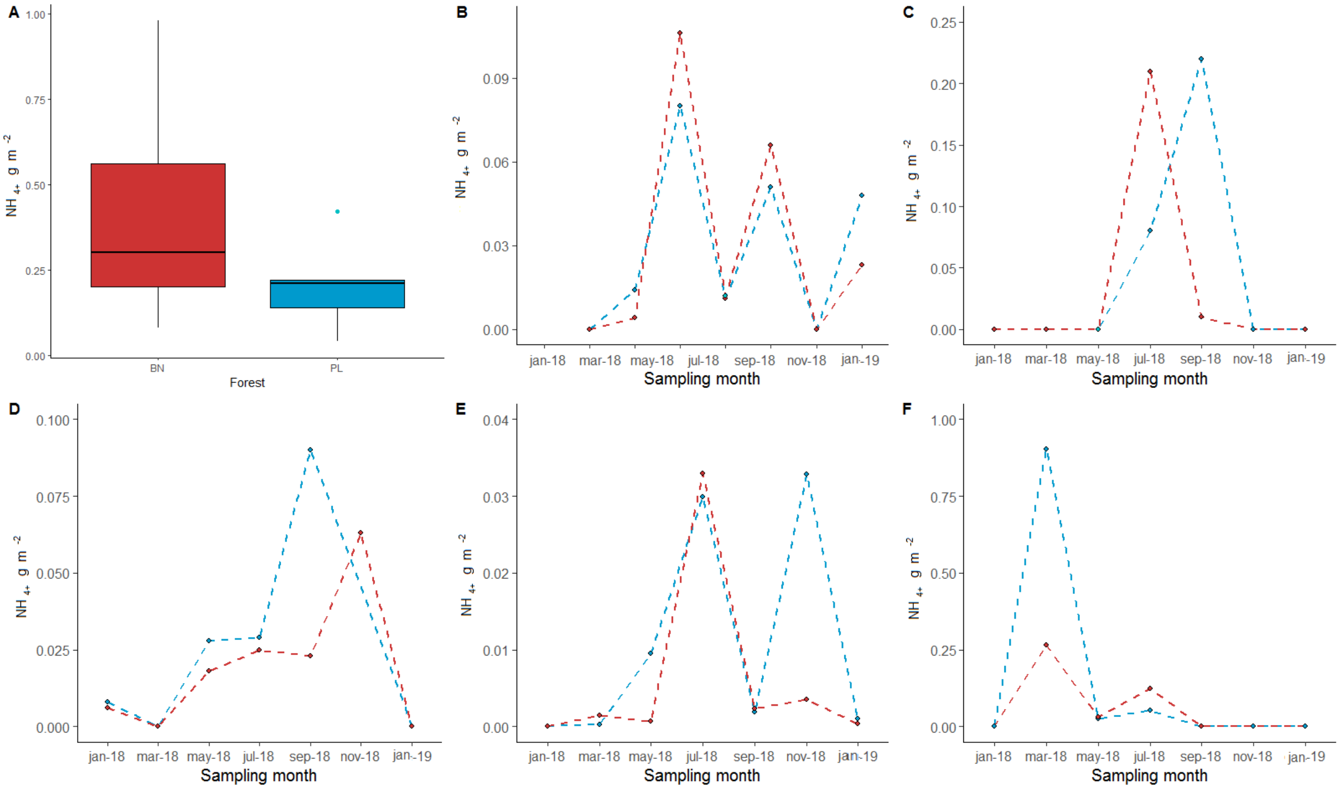

Figure 16. Annual NH_4- losses for each paired plot at each site (A) and NH_4- losses losses distribution for the five sites SCH(B),GR(C),OA(D),YA(E) and RA (F).

For dissolved total Phosphorus (DTP) the peaks of the losses were concentrated in winter time with exception of YA and RA that the peak was at early spring (Figure 17). The pine plantations stand shows higher losses than the native forest ones, where the higher loss of DTP was observed at OA-PL in September with 591.8 g ha $^{-1}$ followed by 245.6 g ha $^{-1}$ in RA-PL. Even though the losses are higher in PL stands, no statistical differences (p= 0.78) were found in the total annual loses of DTP between PL (966.44 ± 276.9 g ha $^{-1}$) and NF (616.44 ± 342.1 g ha $^{-1}$).

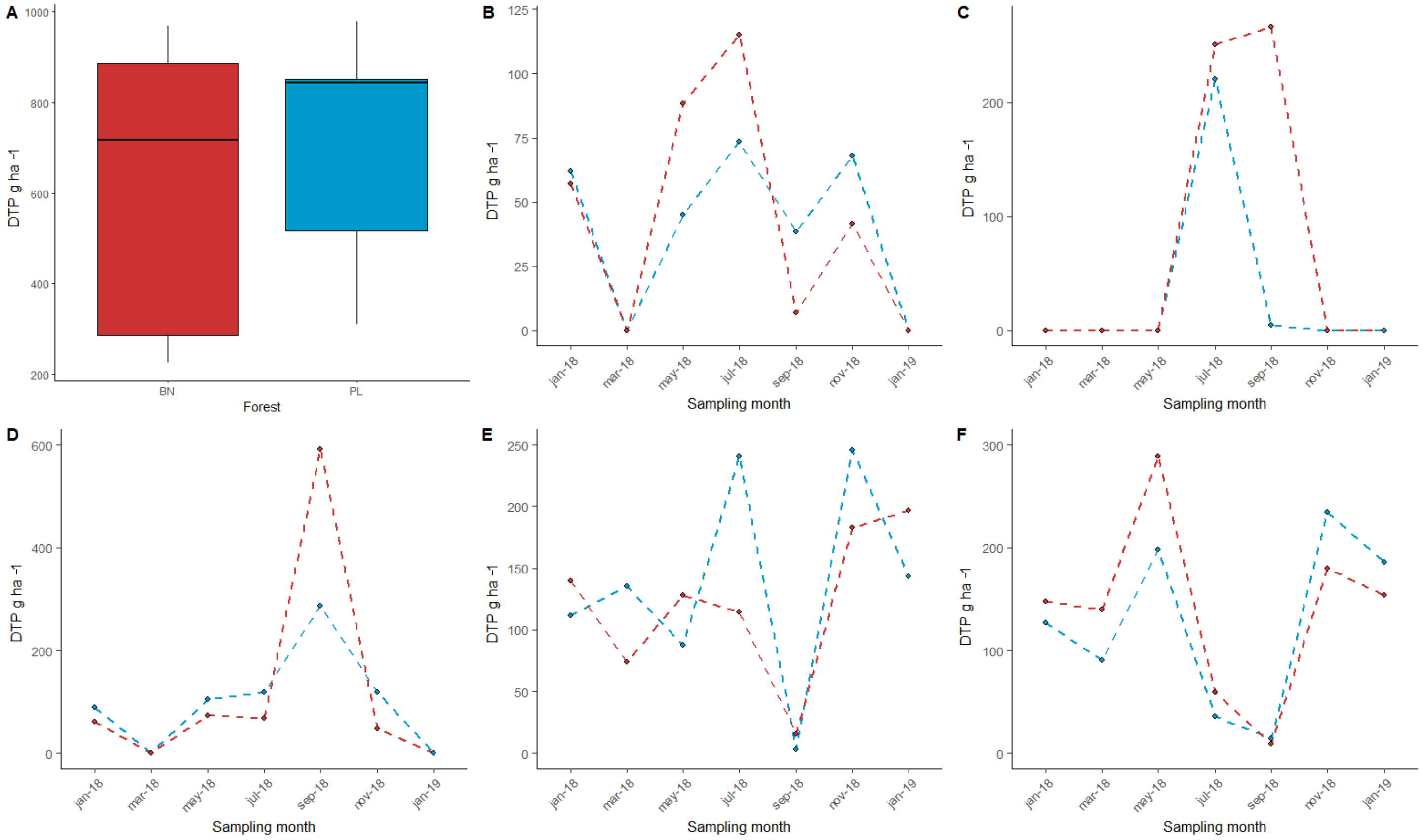

Figure 17. Annual total Phosphorus- losses for each paired plot at each site (A) and total Phosphorus losses distribution for the five sites SCH(B), GR(C), OA(D), YA(E) and RA (F).

3.3 Soil ecosystem Carbon balance.

Total annual inputs (Litterfall and Roots) values ranges from 0.5 to 3.61 Mg ha^{-1} being the lower at RA-PL and the higher value at YA-BN (Table 13). Input fluxes showed no difference between the native forest (NF) and plantation (PL) stands ($p=0.13$). For the total annual outputs values ranges from 0.5 to 1.42 Mg ha^{-1} of Forest Carbon losses, with no statistical differences ($p=0.43$) the NF and PL stands. When adding the Carbon storage of the standing biomass and the soil and litter layer reservoirs data from Crovo et al., (In Review) and subtracting the inputs with the outputs we get the actual Carbon storage of every plot analyzed in this investigation. The values of the total Carbon storage range from 206.2 to 652.3 Mg ha^{-1} with no statistical difference ($p=0.07$) between PL (272.9 ± 43.9 Mg ha^{-1}) and NF (428.2 ± 148.4 Mg ha^{-1}).

3.4 Effect of replacement in forest dynamics on Carbon storage.

Among the analyzed variables (forest dynamics) annual litterfall showed a significant ($p= 0.008$) positive correlation with the total Carbon stock in soil (Figure 18) with an R^2 equal to 0.42 but no correlation with the total aboveground biomass or any biomass compartment. It also showed a significant positive ($p=0.001$) correlation (Figure 18) with soil CO_2 efflux with an R equal to 0.5. Soil CO_2 efflux showed a positive correlation ($R^2= 0.45$) and significant ($p=0.004$) with soil total Carbon, with root biomass ($R^2= 0.49$ and $p=0.001$) and with the total aboveground biomass ($R^2= 0.4$ and $p=0.02$). L.A.I showed a significant ($p= 0.02$) negative correlation ($R^2=-0.4$) with soil total Nitrogen and a positive correlation with total aboveground biomass ($R^2= 0.3$ and $p= 0.03$).

Annual root production showed a significant ($p= 0.0001$) and positive correlation ($R^2=0.57$) with soil total Carbon. Total aboveground biomass showed a significant ($p < 0.0001$) and positive correlation with soil total Phosphorus ($R^2= 0.68$).

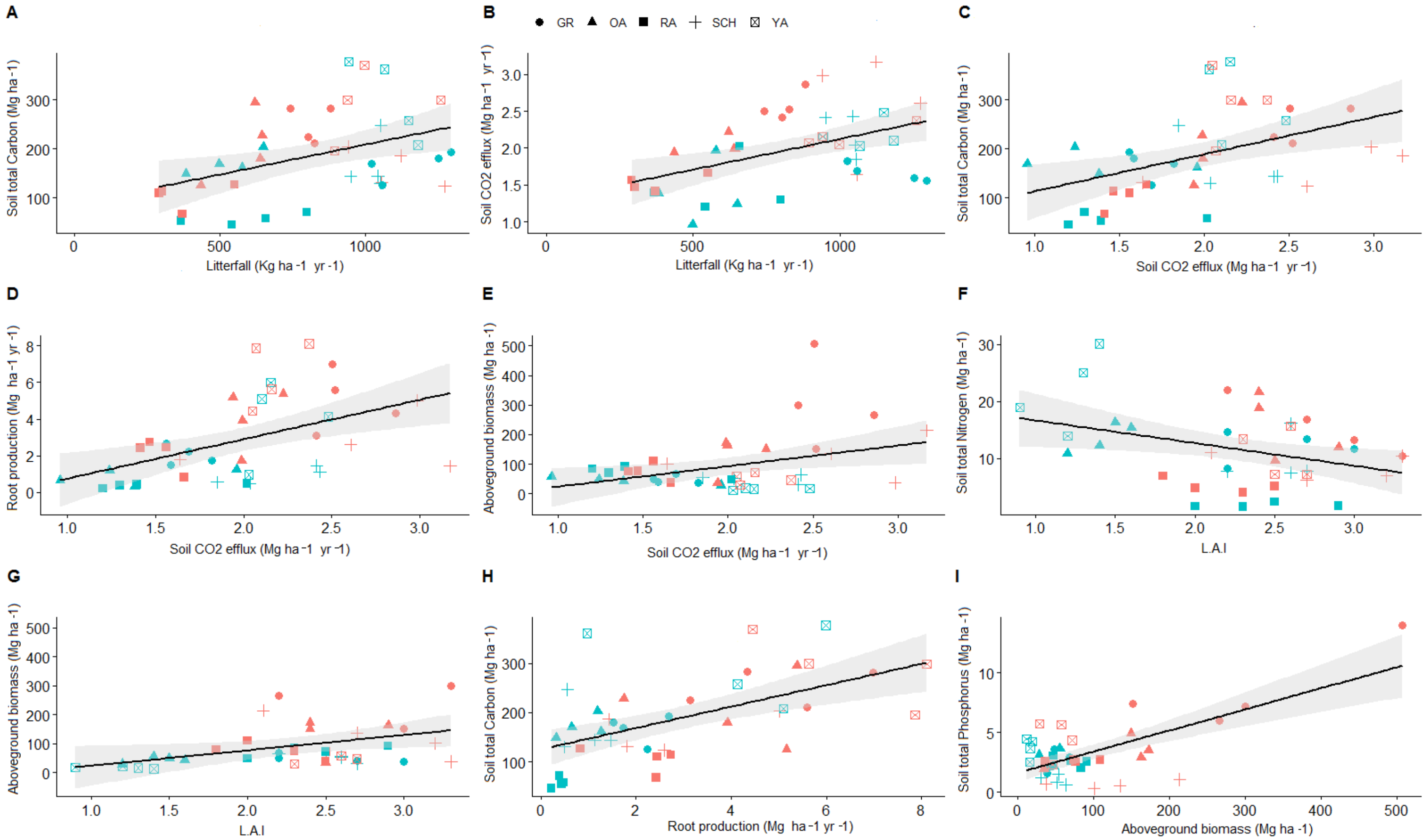

Figure 18. Pearson correlations between variables that showed to be significant ($p<0.05$). The light-grey shadow within the regression line is the confidence interval (95%) of the correlation.

Table 13. Carbon mass balance (Inputs, outputs and storages) for every plot a each site.

Site	Forest	Storage				Annual Input		Annual Output		Balance	Total Carbon
		Standing Biomass (Mg ha^{-1})	Litter layer (Mg ha $^{-1}$)	Organic Horizon (Mg ha $^{-1}$)	Soil (Mg ha $^{-1}$)°	Litterfall (Mg ha^{-1})*	Roots (Mg ha -1)*	CO2 efflux (Mg ha -1)'	DOC (Mg ha - 1)	(Inputs - Outputs) (Mg ha^{-1})	Storage (Mg ha $^{-1}$)
SCH	PL	92.98	6.46	5.46	188.1	0.49	0.46	0.60	0.09	0.26	292.7
SCH	BN	223.39	10.19	8.74	179.7	0.55	1.36	0.71	0.08	1.12	420.9
GR	PL	89.89	7.22	5.31	176.5	0.59	1.02	0.47	0.04	1.10	277.8
GR	BN	384.68	9.79	4.42	255.6	0.39	2.50	0.72	0.01	2.17	652.3
OA	PL	83.02	10.07	3.43	166.2	0.32	0.43	0.36	0.32	0.08	262.6
OA	BN	230.32	7.99	4.27	201.4	0.28	2.03	0.56	0.11	1.64	442.3
YA	PL	30.23	10.17	7.32	279.2	0.42	2.02	0.59	0.09	1.76	325.2
YA	BN	93.04	5.12	9.12	282.4	0.35	3.26	0.59	0.06	2.96	386.7
RA	PL	142.18	6.84	3.59	52.7	0.30	0.19	0.40	1.02	-0.93	206.2
RA	BN	138.44	5.77	3.43	91.8	0.13	1.05	0.42	0.13	0.63	238.8

Data from Crovo et al., (In Review) was used for Carbon storage in the litter layer, organic horizon

4. Discussion.

The stand characteristics of the native *Nothofagus sp.* forests presented in this study (Table 2) differ from other studies that inform higher stem density and basal area values. This may be explained because our study focused on relatively young secondary forest (25-50 years) while the majority of the past research have focused on older secondary forest between 60 – 120 years (Oyarzun et al., 2007; Rivas et al., 2009; Staelens et al., 2011). It is expect that this older stands will show different ecological processes such as gap dynamics and companion species composition, which could explain higher stem density and basal area. Differences in site productivities and on the level of anthropogenic perturbations (i.e. grazing or logging) could also explain these differences.

The YA-NF stand in our study shares similar edaphoclimatic characteristics with the deciduous unmanaged stand presented in Staelens et al.,(2011) and Oyarzun et al., (2007) being the only stand in our study that shares similar stand characteristics with the secondary *Nothofagus obliqua* forest existing literature. On the other hand, Rivas et al., (2009) did soil processes research under commercial *Pinus radiata* plantations showing similar stand characteristics such as density height and basal area with the ones described in our study, which are common characteristics in pre-harvest *P.radiata* plantations in south-central Chile. Likewise, the biomass and allometric characteristics in the *Nothofagus pumilio* stand in RA are very similar to the ones showed by Caldentey et al., (2001).

The differences of stands characteristics in the Nothofagus stands presented in our study, rely basically on the differences in soil intrinsic properties (Aburto et al., In Prep) and mean annual precipitation (Complementary Data). While the principal differences in the plantations stands are because of the different management, where OA and YA are thinned and pruned for wood production (300-500 trees. Ha $^{-}$

^{1}), while GR and SCH were not (>1000 tress. Ha $^{-1}$). The RA stand remained unmanaged for soil conservation purposes (CONAF, 1996). Regardless this numerical observation, BHD frequency distributions follows the same trend for NF stands and PL stands (Log-normal and Weibull respectively) indicating that the stands follows the same horizontal structure and are comparable for ecological purposes (Donoso et al., 2014).

The aboveground biomass obtained values for the PL stands are in the range for typical values obtained in the coastal range in south-central Chile (Camacho, 2016) while Carbon concentrations are similar for those that Gayoso (2001) found in a 23 year *Pinus radiata* stand in the south of Chile. The recent advances in allometric equations for Nothofagus sp. secondary forests stands (Valenzuela et al., 2017) allowed a consistent estimation of the aboveground biomass because specific equations where modeled in the same edaphoclimatic zones at the south-central region that we aimed. Even though our results are consistently higher than does used as references values for temperate forest literature (Vitousek et al., 1988 in Schlesinger & Bernhardt., 2013) and doubled what recent investigations found in Chinese temperate forest (Cao & Chen, 2017), they are in the ranges for what Gayoso (2001) proposed for various temperate forest in the south regions of Chile. The lack of information for allometric equations for companion species in secondary oak forest may have misestimated the total biomass value for some shade-tolerant companion species. In spite of this, NF stands across different soil types in the south-central region of Chile display aboveground biomass values that more than double the standing biomass and carbon storage of their paired monospecific pine plantation stand. Crovo et al., (In Review) found statistical differences in the soil carbon pool in two pair sites we studied here, which may be explained by the much higher productivity reported in the NF plots for these sites (GR and RA).

Even though no statistical differences were found at the whole ecosystem level for carbon storage, we found statistical differences in the soil carbon pool in soil types where the aboveground carbon storage in secondary oak forest was much larger than those founded at pine plantations (SCH, GR and OA). Our study confirms with the postulate that soil acts as the main carbon pool in temperate forest ecosystem, and that the tree species have an important effect on the total carbon storage (Li et al., 2012; Schlesinger & Bernhardt, 2013; Cao & Chen., 2017). We also confirm the fact that the forest floor (litter layer plus organic horizon) contributes significantly with the total forest carbon (Domke et al., 2016) with values similar to the ones reported for other forest in temperate regions (Cao & Chen 2017)

As expected, carbon concentrations where higher in more lignified compartments such as branches and stem (Table 3), consequently the lowest C:N ratios were exhibited in the leaf compartment. PL leaves displayed a higher nitrogen concentration, than leaves in their pair plots. This is opposite to what other investigations have suggested, where native broadleaf forest display higher nitrogen concentrations than pine plantations (Cao & Chen, 2017). This probably a result of a more effective nitrogen uptake and ecosystem elemental recycling under PL. As expected, phosphorus concentrations were highest at leaves compartments, while the N:P ratios is less than 14 in most cases, suggesting a N limitations (Richardson et al., 2008). Consistently, other authors have documented that temperate forest in Chile are mostly N limited ecosystems (Hedin et al., 1995; Staelens et al., 2009).

Considering that the annual accumulation of organic matter per of unit of land (i.e. litterfall) is a measure for NPP (Schlesinger & Bernhardt, 2013), the SCH and GR soils from the coastal mountain ranges could be consider the most productive sites for both PL and NF stands. On the other hand, the stands located at highest altitude showed to be the least productive for both PL and NF (RA). Our litterfall results showed to be lower than other *Nothofagus* studies (Perez et al., 1998; Leiva & Godoy 2002; Staelens et al., 2011) where the values ranges from 1.7 to 7.4 Mg ha $^{-1}$ year $^{-1}$. However, these investigations studied Nothofagus stands in a more southern distribution, which generally display higher overall productivity.

Our results confirms the conclusions of Bray & Gorham (1964) and Millar (1974) that at a regional scale coniferous stands exceeds the total annual amount of litterfall than deciduous stands. Despite, GR and RA under PL showed significantly higher annual litterfall (PL > NA), this does not translate in a higher soil total carbon annual stock after 25 years of replacement, respectively.

The litter mass remaining in the NF stands is similar to the findings of Staelens et al., (2011) which ranges between the 80 and 60%. Unlike other research our decomposition rates did not follow a two-phase decay rate (Lusk et al., 2001; Staelens et al., 2011). This may be due to the fact that we only include leaf material in our litterbags without lignified tissues such as twigs or branches. Taking this into consideration our findings show patterns that differ from the common notion that coniferous litter decomposes slower than deciduous litter (Schlesinger & Bernhardt, 2013). However, our findings are supported by the observations of Lusk et al., (2001).

In this research they compare litter decomposition of *Nothofagus obliqua*, *Pinus radiata* and an evergreen stand in the coastal mountain ranges of Chile, showing that the litter from the *P.radiata* stand decompose faster than the evergreen stand, composed by two companion species we found in our NF plots (*Cryptocaria alba* and *Peumus boldus*) although slower than pure Nothofagus obliqua litter. This may explain why the PL stands showed a higher litter decomposition than the NF stands. Heng (1980) also found that the litter from *P.radiata* stands decomposes similarly than Nothofagus stands in New Zealand. Lusk et al., (2001) concluded that litter from *P.radiata* decomposes faster than other commercial Pine species such as *Pinus pinaster* and *Pinus patula*. Our results also indicate that *P.radiata* litter decomposes faster than litter produced by *Pinus ponderosa* because of a higher nutritional value and lower sclerophyll rate of the former than the later (Lusk et al., 2001). The *Nothofagus pumilio* stand in RA showed the faster litter decomposition with mass remaining values like those presented by Caldentey et al., (2001) but with a different decomposition rate because our stand followed a double-pool exponential decay. The faster decomposition in the RA-NF stand could be associated with the washing of soluble substances due to the higher precipitation at this site. It is important to note that we only consider two year for analysis, so further analyses of litter decomposition should be carried out in both PL and NF stands with more studied factors such as site variables (i.e Indirect site factor, soil pH and water content) as well as nutritional parameters (i.e C: N, Lignin:N). Even though litter decomposition itself cannot be considered as the main driver for mineral soil C, N ,P reservoirs, it still has important implications for other functions and processes such as soil CO2 efflux.

Root annual production showed to be consistently higher in all the NF stands, but also showed to be quite variable among sites. The YA soil exhibits particularly good physical soil properties such as lower bulk density and higher water holding capacity (Aburto et al., In Prep), which could explains why the annual root production is high in both paired plots. It is generally found that, fine root annual production increases in direct proportion to aboveground biomass (Aragao et al., 2009). It has been showed that root production is a major driver in the soil total Carbon pool (Rasse et al., 2005 ; Jackson et al., 2017) . Replacement of secondary *Nothofagus sp*. for *P.radiata* plantations will promote a decline in the annual root carbon input translating in an overall decline of the total carbon storage.

Because of the high equipment and operational cost, soil CO_2 efflux data of temperate forest ecosystem in south-central Chile is scarce. As reported elsewhere, soil CO_2 efflux showed to follow a seasonal pattern driven by environmental conditions (Raich & Schlesinger, 1992; Savage et al., 2008). The highest annual soil efflux (Mg CO_2 ha^{-1} year $^{-1}$) was exhibited in the stands with highest litterfall and root productions, corroborating that carbon mineralization influenced bythe soil priming effect (Sayer et al., 2001). . NF stands showed a higher CO_2 production than the PL stands, in contrary of the findings of Ross et al., (1999) which showed that *P.radiata* plantations produce more CO_2 after forest replacement in New Zealand.

As leachates fluxes are expected to be larger in higher precipitation sites with low retention and fast drainage DOC losses were higher in the sandy RA sites. On the other hand, losses were lower in the granite derived soil (GR) with lower precipitation and the presence of a thick argilic horizon (Bt), which may contributes for DOC sorption (Kalbitz et al., 2000). Moreover, higher evapotranspiration of the pine plantations stands reduced the total infiltrated water translating in lower total DOC losses. For most sites, DOC losses have near negligible impact in the ecosystem outputs. Losses by soil leachates in our stands fit between the ranges for temperate ecosystem DOC reported fluxes (3 – 70 g C m^{-2} y^{-1}) (Neff & Asner, 2001). Contrary to the findings of Oyarzun et al., (2007) that reported that *Pinus radiata* plantations generate more N losses than native forest at catchment level, we did not find enough evidence to support that conclusion. Even though both NF and PL display an efficient N retention capacity, future changes in management practices and logging activity could produce high peaks of N leaching. Phosphorus losses were higher in soils that exhibit higher mean annual precipitations. Losses in GR-PL double-fold the ones in the NF stand and agrees for the potential P deficit found in this soil (Crovo et al., In Review). It is relevant to note that total annual fluxes were calculated disregarding the data gaps generated by insufficient water samples in summer period (November to March). Also 2018 had a particular dry winter, which may not be representative of the long term leaching losses. Therefore, it is suggested to consider this data as a low range estimate for leachate losses.

5. Conclusion

Our paired comparisons of the effect of secondary native *Nothofagus sp.* forest conversion to pine plantation allow us to explore the magnitude of changes of the principal variables that regulates intra ecosystem cycling and productivity, and how this could have direct impact in ecosystem Carbon and nutrients reservoirs such as Nitrogen and Phosphorus.

This research contributes with relevant information about the sensitivity or resilience of different soils to the effect of native forest conversion to pine plantations Our result indicates that exotic pine plantations enhance some biogeochemical fluxes such as litterfall and litter decomposition, but also diminish other such as root production in most soils and leaching losses in susceptible soils. To assess if the effect of forest conversion on biogeochemical fluxes and pools is consistent at larger scales, it is necessary to increase the number of sites, ideally considering other environmental (climate and topography) and site management factors (i.e. stand age, residue management, etc). Overall, total ecosystem carbon followed by a forest replacement did not show a significant decrease. Continues monitoring of this plots should help to address temporal variability and sensitivity to future change in environmental conditions. We partially prove our work hypothesis, while exotic pine plantations produces C, N and P imbalances, we did not find a consistent decline in the total pools across sites. The N:P ratios indicates that both native forest and plantations stands exhibits potential deficit on N.

We also corroborate that native forest have significantly higher root production than planted stands, which seems to be explaining the relatively higher C total stock in certain soils and other processes such as soil CO_2 efflux (C mineralization). We consider this differential root production a critical aspect that need to be further assessed.

This investigation provided evidence about the relevance of soil type for determining the sensitivity forest C and nutrient biogeochemical cycling to LUC and we hope it promote further investigations.

Acknowedgements

The authors thank the crew members from LISAB,UdeC for their help in the laboratory analyses and all the students, colleagues and friends for their help during field work. We also would like to thank CONAF Chile for allowing us to establish two paired plots in national protected areas (R.N Nonguen and R.N Malalcahuello) and also Forestal Arauco S.A and CMPC holding for allowing us to establish the other three paired plots in their patrimony. This study was founded by CONICYT Chile by FONDECYT INICIACIÓN grant n° 11160372.

6. References

Aburto F. Crovo, O. Albornoz, F. and Southard, R. (In Prep). Soil mineralogical and physicochemical alterations caused by forest substitution in contrasting soils of south central Chile.

Aguayo, M., Pauchard, A., Azócar, G., & Parra, O. (2009). Cambio del uso del suelo en el centro sur de Chile a fines del siglo XX: Entendiendo la dinámica espacial y temporal del paisaje. Revista chilena de historia natural, 82(3), 361-374.

Altamirano, A., & Lara, A. (2010). Deforestación en ecosistemas templados de la precordillera andina del centro-sur de Chile. Bosque (Valdivia), 31(1), 53-64.

Besoain, E. Los Suelos. In: Tosso, J. (1985). Suelos Volcanicos de Chile. 25-95. 723 p.

Bockheim, J. G., & Gennadiyev, A. N. (2010). Soil-factorial models and earth-system science: A review. Geoderma, 159(3-4), 243-251.

Cai, X., Lin, Z., Penttinen, P., Li, Y., Li, Y., Luo, Y.,Yue, T., Jiang , P. Fu, W. (2018). Effects of conversion from a natural evergreen broadleaf forest to a Moso bamboo plantation on the soil nutrient pools, microbial biomass and enzyme activities in a subtropical area. Forest ecology and management, 422, 161-171.

Caldentey, J., Ibarra, M., & Hernández, J. (2001). Litter fluxes and decomposition in Nothofagus pumilio stands in the region of Magallanes, Chile. Forest Ecology and Management, 148(1-3), 145-157.

Camacho Focacci, F. I. (2016). Productividad primaria bruta y neta en plantaciones de Pinus radiata D. Don en la Región del Bío Bío, Chile.

Cao, Y., & Chen, Y. (2017). Ecosystem C: N: P stoichiometry and carbon storage in plantations and a secondary forest on the Loess Plateau, China. Ecological Engineering, 105, 125-132.

Centro de Información de Recursos Naturales Naturales, (CIREN). (1999). Estudio agrológico VIII región: CIREN Santiago, Chile.

Chen, G. S., Yang, Y. S., Xie, J. S., Guo, J. F., Gao, R., Qian, W. (2005). Conversion of a natural broad-leafed evergreen forest into pure plantation forests in a subtropical area: effects on carbon storage. Annals of forest science, 62(7), 659-668.

Chen, X., Chen, H. Y., Chen, X., Wang, J., Chen, B., Wang, D.,Guan, Q. (2016). Soil labile organic carbon and carbon-cycle enzyme activities under different thinning intensities in Chinese fir plantations. Applied soil ecology, 107, 162-169.

CONAF, 2011. Catastro de los recursos vegetacionales nativos de Chile: Monitoreo de cambios y actualizaciones. In: C.N.Forestal (Ed.), Santiago, Chile.

CONAF, R. M. Plan de manejo de la Reserva Nacional Malalcahuello. Documento de Trabajo, 1996, vol. 247.

Dixon, R. K., Solomon, A. M., Brown, S., Houghton, R. A., Trexier, M. C., Wisniewski, J. (1994). Carbon pools and flux of global forest ecosystems,*Science*, *263*(5144), 185-190.

Domke, G. M., Perry, C. H., Walters, B. F., Woodall, C. W., Russell, M. B., Smith, J. E. (2016). Estimating litter carbon stocks on forest land in the United States. Science of the Total Environment, 557, 469-478.

Donoso, C., González, M., Lara, A. (2014). Ecología forestal: bases para el manejo sustentable y conservación de los bosques nativos de Chile. Valdivia, Chile. Ediciones Universidad Austral de Chile.

Echeverría, C., Coomes, D., Salas, J., Rey-Benayas, J. M., Lara, A., Newton, A. (2006). Rapid deforestation and fragmentation of Chilean temperate forests. Biological conservation, 130(4), 481-494.

Eclesia, R. P., Jobbagy, E. G., Jackson, R. B., Biganzoli, F., Piñeiro, G. (2012). Shifts in soil organic carbon for plantation and pasture establishment in native forests and grasslands of South America. Global Change Biology, 18(10), 3237-3251.

Fialho, R. C., Zinn, Y. L. (2014). Changes in soil organic carbon under eucalyptus plantations in Brazil: a comparative analysis. Land Degradation & Development, 25(5), 428-437.

Fox, J., Weisberg, S., Adler, D., Bates, D., Baud-Bovy, G., Ellison, S., Graves, S. (2012). Package ‘car’. Vienna: R Foundation for Statistical Computing.

Gärdenäs, A. I., Ågren, G. I., Bird, J. A., Clarholm, M., Hallin, S., Ineson, P.,Kätterer, T., Knicker, H., Nilsson, S.I., Näsholm, T., Ogle,S., Paustian, K., Persson, T., Stendahl, J. (2011). Knowledge gaps in soil carbon and nitrogen interactions–from molecular to global scale. Soil Biology and Biochemistry, 43(4), 702-717.

Garrido, E., Matus, F. (2012). Are organo-mineral complexes and allophane content determinant factors for the carbon level in Chilean volcanic soils? Catena, 92, 106-112. doi:https://doi.org/10.1016/j.catena.2011.12.003

Gayoso J. 2013. Funciones alométricas para la determinación de existencias de carbono forestal para la especie Nothofagus obliqua (Mirb.) Oerst. (roble). Santiago, Chile. CONAF. 41 p

Gayoso, J., Guerra, J., Schlegel, B. (2001). Manual de procedimientos para inventarios de carbono en ecosistemas forestales.

Guan, F., Xia, M., Tang, X., Fan, S. (2017). Spatial variability of soil nitrogen, phosphorus and potassium contents in Moso bamboo forests in Yong'an City, China. Catena, 150, 161-172.

Guo, L. B., Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta-analysis. Global Change Biology, 8(4), 345-360. doi:10.1046/j.1354-1013.2002.00486.x

Hedin, L. O., Armesto, J. J., Johnson, A. H. (1995). Patterns of nutrient loss from unpolluted, old-growth temperate forests: Evaluation of biogeochemical theory. Ecology, 76(2), 493-509.

Heilmayr, R., Echeverria, C., Fuentes, R., Lambin, E. F. (2016). A plantation-dominated forest transition in Chile. Applied Geography, 75, 71-82.

Heng, S. (1980). The addition, decomposition and accumulation of organic matter in some native Nothofagus spp. forests and Pinus radiata plantations in the South Island of New Zealand (Doctoral dissertation, Lincoln College, University of Canterbury).

Huber, A., Iroumé, A., Mohr, C., & Frêne, C. (2010). Efecto de plantaciones de Pinus radiata y Eucalyptus globulus sobre el recurso agua en la Cordillera de la Costa de la región del Biobío, Chile. Bosque (Valdivia), 31(3), 219-230.

IPCC, W. (2013). Working Group I Contribution to the IPCC Fifth Assessment Report: Climate Change 2013: The Physical Science Basis, Summary for Policymakers. IPCC, UN.

Jackson, R. B., Lajtha, K., Crow, S. E., Hugelius, G., Kramer, M. G., Piñeiro, G. (2017). The ecology of soil carbon: pools, vulnerabilities, and biotic and abiotic controls. Annual Review of Ecology, Evolution, and Systematics, 48, 419-445.

Jobbágy, E. G., Jackson, R. B. (2000). The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological applications, 10(2), 423-436.

Kalbitz, K., Solinger, S., Park, J. H., Michalzik, B., Matzner, E. (2000). Controls on the dynamics of dissolved organic matter in soils: a review. Soil science, 165(4), 277-304.

Kasel, S., Bennett, L. T. (2007). Land-use history, forest conversion, and soil organic carbon in pine plantations and native forests of south eastern Australia. Geoderma, 137(3-4), 401-413.

Kay, M., Wobbrock, J. O. (2019). Package ‘ARTool’.

Lal, R. (2005). Forest soils and carbon sequestration. Forest ecology and management, 220(1-3), 242-258.

Leiva, J., Godoy, R. (2001, September). Production and decomposition of litterfall in Nothofagus forest in Southern Chile. In Comparison of ecosystem functioning and biogeochemical cycles in temperate forests in Southern Chile and Flanders. Proceedings of the workshop held at Ghent University, Belgium (pp. 17-19).

Liao, C., Luo, Y., Fang, C., Chen, J., Li, B. (2012). The effects of plantation practice on soil properties based on the comparison between natural and planted forests: a meta-analysis. Global ecology and biogeography, 21(3), 318-327.

Lusk, C. H., Donoso, C., Jiménez, M., Moya, C., Oyarce, G., Reinoso, R., Saldaña A., Villegas, P., Matus, F. (2001). Descomposición de hojarasca de Pinus radiata y tres especies arbóreas nativas. Revista chilena de historia natural, 74(3), 705-710.

Mancinelli, R., Campiglia, E., Di Tizio, A., Marinari, S. (2010). Soil carbon dioxide emission and carbon content as affected by conventional and organic cropping systems in Mediterranean environment. Applied Soil Ecology, 46(1), 64-72.

Marthews, T. R., Metcalfe, D., Malhi, Y., Phillips, O., Huaraca Huasco, W., Riutta, T. (2013). Measuring Tropical Forest Carbon Allocation and Cycling: A RAINFOR-GEM Field Manual for Intensive Census Plots (v2. 2); 2012.

Milla, F., Emanuelli, P., Sartori, A. (2013). Compendio de funciones alométricas para la estimación de biomasa de especies forestales presentes en Chile: Elemento clave para la Estrategia Nacional de Bosques y Cambio Climático (ENBCC). Corporación Nacional Forestal.

Millar, C. S. (1974). Decomposition of coniferous leaf litter. Biology of plant litter decomposition, 1, 105-128.

Miranda, A., Altamirano, A., Cayuela, L., Pincheira, F., Lara, A. (2015). Different times, same story: Native forest loss and landscape homogenization in three physiographical areas of south-central of Chile. Applied Geography, 60, 20-28.

Nahuelhual, L., Carmona, A., Lara, A., Echeverría, C., González, M. E. (2012). Land-cover change to forest plantations: Proximate causes and implications for the landscape in south-central Chile. Landscape and urban planning, 107(1), 12-20.

Neff, J. C., Asner, G. P. (2001). Dissolved organic carbon in terrestrial ecosystems: synthesis and a model. Ecosystems, 4(1), 29-48.

Olson, J.S., 1963. Energy storage and the balance of producers and decomposers in ecological systems. Ecology 44, 322–331

Oyarzun, C., Aracena, C., Rutherford, P., Godoy, R., Deschrijver, A. (2007). Effects of land use conversion from native forests to exotic plantations on nitrogen and phosphorus retention in catchments of southern Chile. Water, air, and soil pollution, 179(1-4), 341-350.

Peñuelas, J., Sardans, J., Rivas-ubach, A., Janssens, I. A. (2012). The human-induced imbalance between C, N and P in Earth's life system. Global Change Biology, 18(1), 3-6.

Pérez, C. A., DeGrandpre, M. D., Lagos, N. A., Saldías, G. S., Cascales, E. K., Vargas, C. A. (2015). Influence of climate and land use in carbon biogeochemistry in lower reaches of rivers in central southern Chile: Implications for the carbonate system in river-influenced rocky shore environments. Journal of Geophysical Research: Biogeosciences, 120(4), 673-692.

Pérez, C. A., Hedin, L. O., Armesto, J. J. (1998). Nitrogen mineralization in two unpolluted old-growth forests of contrasting biodiversity and dynamics. Ecosystems, 1(4), 361-373.

Raich, J. W., Schlesinger, W. H. (1992). The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. Tellus B, 44(2), 81-99.

Rasse, D. P., Rumpel, C., Dignac, M. F. (2005). Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. Plant and soil, 269(1-2), 341-356.

Richardson, S. J., Allen, R. B., Doherty, J. E. (2008). Shifts in leaf N: P ratio during resorption reflect soil P in temperate rainforest. Functional Ecology, 22(4), 738-745.

Rivas, Y., Oyarzun, C., Godoy, R., Valenzuela, E. (2009). Mineralización del nitrógeno, carbono y actividad enzimática del suelo en un bosque de Nothofagus obliqua (Mirb) Oerst y una plantación de Pinus radiata D. Don. del centro-sur de Chile. Revista chilena de historia natural, 82(1), 119-134.

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(6), 803-813. Temperate conifer forest soils. Global Change Biology, 12(5), 834-847.

Sadzawka, A., Carrasco, M., Grez, R., Mora, M., Flores, H., Neaman, A. (2006). Métodos de análisis recomendados para los suelos de Chile. Serie actas INIA, 163.

Savage, K. E., Davidson, E. A. (2003). A comparison of manual and automated systems for soil CO2 flux measurements: Trade-offs between spatial and temporal resolution. Journal of Experimental Botany, 54(384), 891-899.

Sayer, E. J., Heard, M. S., Grant, H. K., Marthews, T. R., Tanner, E. V. (2011). Soil carbon release enhanced by increased tropical forest litterfall. Nature Climate Change, 1(6), 304-307.

Schlesinger, W. H., Bernhardt, E. S. (2013). Biogeochemistry: an analysis of global change. Academic press.

Schoeneberger, P. J., Wysocki, D. A., Benham, E. C. (Eds.). (2012). Field book for describing and sampling soils. Government Printing Office Miranda, K. M., Espey, M. G., & Wink, D. A. (2001). A rapid, simple spectrophotometric method for simultaneous detection of nitrate and nitrite. Nitric oxide, 5(1), 62-71.

Simunek, J., Van Genuchten, M. T., Sejna, M. (2005). The HYDRUS-1D software package for simulating the one-dimensional movement of water, heat, and multiple solutes in variably-saturated media. University of California-Riverside Research Reports, 3, 1-240.

Staelens, J., Ameloot, N., Almonacid, L., Padilla, E., Boeckx, P., Huygens, D., Verheyen, K., Oyarzun, C., Godoy, R. (2011). Litterfall, litter decomposition and nitrogen mineralization in old-growth evergreen and secondary deciduous Nothofagus forests in south-central Chile. Revista chilena de historia natural, 84(1), 125-141.

Staelens, J., Oyarzún, C., Almonacid, L., Padila, E., Verheyen, K. (2009). Aboveground nutrient cycling in temperate forest ecosystems of southern Chile. In Ecological advances on Chilean temperate rainforests (pp. 103-116). Academia Press.

Subke, J. A., Inglima, I., Francesca Cotrufo, M. (2006). Trends and methodological impacts in soil CO2 efflux partitioning: a metaanalytical review. Global Change Biology, 12(6), 921-943.

Valenzuela, C., Acuña, E., Muñoz, F., Ortega, A., Cancino, J. (2017). Generalized allometric equations of total volume generated by meta-analysis for roble, raulí and coigüe in second growth forests in Chile. Forestry: An International Journal of Forest Research, 90(5), 661-674.

Vitousek, P. M., Fahey, T., Johnson, D. W., Swift, M. J. (1988). Element interactions in forest ecosystems: succession, allometry and input-output budgets. Biogeochemistry, 5(1), 7-34.

Wang, J., Ren, C., Cheng, H., Zou, Y., Bughio, M. A., Li, Q. (2017). Conversion of rainforest into agroforestry and monoculture plantation in China: Consequences for soil phosphorus forms and microbial community. Science of the Total Environment, 595, 769-778.

Watanabe, F. S. and S. R. Olsen 1965. Test of an ascorbic acid method for determining P in water and NaHCO3 extracts from soil. Soil Science Society of America Proceedings 29: 677-678.

Yan, E. R., Wang, X. H., Huang, J. J., Li, G. Y., Zhou, W. (2008). Decline of soil nitrogen mineralization and nitrification during forest conversion of evergreen broad-leaved forest to plantations in the subtropical area of Eastern China. Biogeochemistry, 89(2), 239-251.

Yanai, R. D., Currie, W. S., Goodale, C. L. (2003). Soil carbon dynamics after forest harvest: an ecosystem paradigm reconsidered. Ecosystems, 197-212.

III. CONCLUSIONES GENERALES

Nuestras comparaciones pareadas sobre el efecto de la conversión de bosques secundarios hacia plantaciones de pinos nos permiten explorar la capacidad diferencial que tienen los suelos contrastantes para resistir esta importante alteración de los reservorios biogeoquímicos del suelo. Se evidencia de que los reservorios totales de suelo C, N y P varían significativamente entre el tipo de suelo y que la magnitud de la modificación de estequiometria C: N: P causada por la conversión del bosque nativo a plantaciones varía mucho entre los tipos de suelo. Los suelos con arcillas cristalinas dominantes de baja actividad respondieron más fuertemente a la conversión de plantaciones forestales mostrando una disminución neta en el total C.

Debido a que la respuesta a la conversión forestal en los sitios no fue homogénea, la relación C: N no mostró diferencias significativas entre los tipos de bosque. Del mismo modo, la relación C: P no se alteró por igual en todos los sitios que mostraban el no cambio a una reducción (RA) y un incremento (GR) en las plantaciones.

También se evidencia que las plantaciones modificaron la distribución vertical elemental de C, N y P en la mayoría de los sitios, pero C y N respondieron con más fuerza que P. En la mayoría de los suelos bajo plantaciones forestales, C y N tienden a ser más bajos en comparación con los bosques nativos, especialmente en los 120cm superiores

Los mayores cambios en el contenido de C: N: P y la estequiometria de los reservorios en los suelos ocurrieron en los sitios con bajo contenido de arcilla (RA, Entisoles) y suelos con arcilla dominante de baja actividad (GR, Alfisol), mientras que los suelos derivados de cenizas (YA, Andisol) y los suelos con arcillas dominantes de alta actividad (SCH, Ultisol y OA, Alfisol) mostraron menos alteración.

Esta investigación también contribuye con información relevante sobre la sensibilidad o resistencia de diferentes suelos al efecto de la conversión de bosques nativos a plantaciones de pinos. Nuestro resultado indica que las plantaciones de pinos exóticos mejoran algunos flujos biogeoquímicos como la caída y la descomposición de la hojarasca, pero también disminuyen otros como la producción anual de raices en la mayoría de los suelos.

Para evaluar si el efecto de la conversión forestal en los flujos y pooles biogeoquímicas es consistente a grandes escalas, es necesario aumentar el número de sitios, idealmente teniendo en cuenta otros factores ambientales (climáticos y de topografía) y de manejo del sitio (manejo de residuos, densidad y fertilización)

Nuestros resultados destacan la importancia de las propiedades del suelo en la determinación de las reservas de C y nutrientes y proporcionan información valiosa para los administradores de la tierra para determinar qué tipos de suelo podrían ser más susceptibles a la pérdida de C y nutrientes después de la conversión a bosques plantados. La investigación futura debería explorar más a fondo los mecanismos biogeoquímicos del control de minerales en la estabilización de C y la retención de nutrientes en los suelos forestales y en el efecto de la intensificación del manejo forestal y las prácticas específicas sobre la estequiometría de C y nutrientes.

IV. REFERENCIAS GENERALES

Aburto F. Crovo, O. Albornoz, F. and Southard, R. (In Prep). Soil mineralogical and physicochemical alterations caused by forest substitution in contrasting soils of south central Chile.

Aguayo, M., Pauchard, A., Azócar, G., & Parra, O. (2009). Cambio del uso del suelo en el centro sur de Chile a fines del siglo XX: Entendiendo la dinámica espacial y temporal del paisaje. Revista chilena de historia natural, 82(3), 361-374.

Altamirano, A., & Lara, A. (2010). Deforestación en ecosistemas templados de la precordillera andina del centro-sur de Chile. Bosque (Valdivia), 31(1), 53-64.

Beaudette, D. E., Roudier, P., & O'Geen, A. T. (2013). Algorithms for quantitative pedology: a toolkit for soil scientists. Computers & Geosciences, 52, 258-268.

Besoain, E. Los Suelos. In: Tosso, J. (1985). Suelos Volcanicos de Chile. 25-95. 723 p.

Bockheim, J. G., & Gennadiyev, A. N. (2010). Soil-factorial models and earth-system science: A review. Geoderma, 159(3-4), 243-251.

Bowman, R., & Moir, J. (1993). Basic EDTA as an extractant for soil organic phosphorus. Soil Science Society of America Journal, 57(6), 1516-1518.

Brockerhoff, E. G., Jactel, H., Parrotta, J. A., Quine, C. P., & Sayer, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity?. Biodiversity and Conservation, 17(5), 925-951.

Cai, X., Lin, Z., Penttinen, P., Li, Y., Li, Y., Luo, Y.,Yue, T., Jiang , P. Fu, W. (2018). Effects of conversion from a natural evergreen broadleaf forest to a Moso bamboo plantation on the soil nutrient pools, microbial biomass and enzyme activities in a subtropical area. Forest ecology and management, 422, 161-171.

Caldentey, J., Ibarra, M., & Hernández, J. (2001). Litter fluxes and decomposition in Nothofagus pumilio stands in the region of Magallanes, Chile. Forest Ecology and Management, 148(1-3), 145-157.

Camacho Focacci, F. I. (2016). Productividad primaria bruta y neta en plantaciones de Pinus radiata D. Don en la Región del Bío Bío, Chile.

Cao, Y., & Chen, Y. (2017). Ecosystem C: N: P stoichiometry and carbon storage in plantations and a secondary forest on the Loess Plateau, China. Ecological Engineering, 105, 125-132.

Carlson, K. M., Curran, L. M., Asner, G. P., Pittman, A. M., Trigg, S. N., & Adeney, J. M. (2013). Carbon emissions from forest conversion by Kalimantan oil palm plantations. Nature Climate Change, 3(3), 283.

Carrasco, P., & Millán, J. (1990). Proyecto de suelos forestales de la VIII Región. Informe final. Universidad de Concepción, Dpto. de Cs. Forestales/Min. de Agricultura. Chillán, Chile.

Carter, M.R. and Gregorich, E.G. (2008). Soil sampling and methods of analysis 2nd ed. Canadian Society of Soil Science. CRC Press Taylor & Francis Group. ISBN-13:978-0-8493-3586-0

Centro de Información de Recursos Naturales Naturales, (CIREN). (1999). Estudio agrológico VIII región: CIREN Santiago, Chile.

Chen, G. S., Yang, Y. S., Xie, J. S., Guo, J. F., Gao, R., Qian, W. (2005). Conversion of a natural broad-leafed evergreen forest into pure plantation forests in a subtropical area: effects on carbon storage. Annals of forest science, 62(7), 659-668.

Chen, X., Chen, H. Y., Chen, X., Wang, J., Chen, B., Wang, D.,Guan, Q. (2016). Soil labile organic carbon and carbon-cycle enzyme activities under different thinning intensities in Chinese fir plantations. Applied soil ecology, 107, 162-169.

Cleveland, C. C., & Liptzin, D. (2007). C: N: P stoichiometry in soil: is there a "Redfield ratio" for the microbial biomass? Biogeochemistry, 85(3), 235-252.

Cleveland, C. C., Houlton, B. Z., Smith, W. K., Marklein, A. R., Reed, S. C., Parton, W., Running, S. W. (2013). Patterns of new versus recycled primary production in the terrestrial biosphere. Proceedings of the National Academy of Sciences, 110(31), 12733-12737.

CONAF, 2011. Catastro de los recursos vegetacionales nativos de Chile: Monitoreo de cambios y actualizaciones. In: C.N.Forestal (Ed.), Santiago, Chile.

CONAF, R. M. Plan de manejo de la Reserva Nacional Malalcahuello. Documento de Trabajo, 1996, vol. 247.

Crovo, O. Aburto, F. Rubilar, R. (In Prep.) Biogeochemical C, N, P dynamics shifts as a result of secondary native Nothofagus sp. forest substitution by Pine sp. exotic plantations.

Denef, K., & Six, J. (2005). Clay mineralogy determines the importance of biological versus abiotic processes for macroaggregate formation and stabilization. European Journal of soil science, 56(4), 469-479.

Denef, K., Six, J., Merckx, R., & Paustian, K. (2004). Carbon sequestration in microaggregates of no-tillage soils with different clay mineralogy. Soil Science Society of America Journal, 68(6), 1935-1944.

Deng, L., Zhu, G. Y., Tang, Z. S., & Shangguan, Z. P. (2016). Global patterns of the effects of land-use changes on soil carbon stocks. Global ecology and conservation, 5, 127-138.

Dixon, R. K., Solomon, A. M., Brown, S., Houghton, R. A., Trexier, M. C., & Wisniewski, J. (1994). Carbon pools and flux of global forest ecosystems,*Science*, *263*(5144), 185-190.

Domec, J. C., King, J. S., Ward, E., Oishi, A. C., Palmroth, S., Radecki, A., & McNulty, S. G. (2015). Conversion of natural forests to managed forest plantations decreases tree resistance to prolonged droughts. Forest Ecology and Management, 355, 58-71.

Domke, G. M., Perry, C. H., Walters, B. F., Woodall, C. W., Russell, M. B., Smith, J. E. (2016). Estimating litter carbon stocks on forest land in the United States. Science of the Total Environment, 557, 469-478.

Donoso, C., González, M., Lara, A. (2014). Ecología forestal: bases para el manejo sustentable y conservación de los bosques nativos de Chile. Valdivia, Chile. Ediciones Universidad Austral de Chile.

Echeverría, C., Coomes, D., Salas, J., Rey-Benayas, J. M., Lara, A., Newton, A. (2006). Rapid deforestation and fragmentation of Chilean temperate forests. Biological conservation, 130(4), 481-494.

Eclesia, R. P., Jobbagy, E. G., Jackson, R. B., Biganzoli, F., Piñeiro, G. (2012). Shifts in soil organic carbon for plantation and pasture establishment in native forests and grasslands of South America. Global Change Biology, 18(10), 3237-3251.

Feng, D., Bao, W., & Pang, X. (2017). Consistent profile pattern and spatial variation of soil C/N/P stoichiometric ratios in the subalpine forests. Journal of Soils and Sediments, 17(8), 2054-2065.

Feng, W., Plante, A. F., Six, J. (2013). Improving estimates of maximal organic carbon stabilization by fine soil particles. Biogeochemistry, 112(1-3), 81-93.

Fialho, R. C., Zinn, Y. L. (2014). Changes in soil organic carbon under eucalyptus plantations in Brazil: a comparative analysis. Land Degradation & Development, 25(5), 428-437.

Fox, G. A., Negrete-Yankelevich, S., Sosa, V. J. (Eds.). (2015). Ecological statistics: contemporary theory and application. Oxford University Press, USA.

Fox, J., Weisberg, S., Adler, D., Bates, D., Baud-Bovy, G., Ellison, S., Graves, S. (2012). Package ‘car’. Vienna: R Foundation for Statistical Computing.

Gärdenäs, A. I., Ågren, G. I., Bird, J. A., Clarholm, M., Hallin, S., Ineson, P.,Kätterer, T., Knicker, H., Nilsson, S.I., Näsholm, T., Ogle,S., Paustian, K., Persson, T., Stendahl, J. (2011). Knowledge gaps in soil carbon and nitrogen interactions–from molecular to global scale. Soil Biology and Biochemistry, 43(4), 702-717.

Garrido, E., Matus, F. (2012). Are organo-mineral complexes and allophane content determinant factors for the carbon level in Chilean volcanic soils? Catena, 92, 106-112. doi:https://doi.org/10.1016/j.catena.2011.12.003

Gaveau, D. L., Locatelli, B., Salim, M. A., Yaen, H., Pacheco, P., & Sheil, D. (2019). Rise and fall of forest loss and industrial plantations in Borneo (2000–2017). Conservation Letters, 12(3), e12622.

Gayoso J. 2013. Funciones alométricas para la determinación de existencias de carbono forestal para la especie Nothofagus obliqua (Mirb.) Oerst. (roble). Santiago, Chile. CONAF. 41 p

Gayoso, J., Guerra, J., Schlegel, B. (2001). Manual de procedimientos para inventarios de carbono en ecosistemas forestales.

González-Ferrán, O. 1995 Volcanes de Chile. Instituto Geográfico MIlitar. ISBN 9562020541. 640p.

Gregorich, E. G., Carter, M. R. (2007). Soil sampling and methods of analysis. CRC press.

Guan, F., Xia, M., Tang, X., Fan, S. (2017). Spatial variability of soil nitrogen, phosphorus and potassium contents in Moso bamboo forests in Yong'an City, China. Catena, 150, 161-172.

Guillaume, T., Damris, M., Kuzyakov, Y. (2015). Losses of soil carbon by converting tropical forest to plantations: erosion and decomposition estimated by δ13C. Global change biology, 21(9), 3548-3560.

Guo, L. B., Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta-analysis. Global Change Biology, 8(4), 345-360. doi:10.1046/j.1354-1013.2002.00486.x

Hedin, L. O., Armesto, J. J., Johnson, A. H. (1995). Patterns of nutrient loss from unpolluted, old-growth temperate forests: Evaluation of biogeochemical theory. Ecology, 76(2), 493-509.

Heilmayr, R., Echeverria, C., Fuentes, R., Lambin, E. F. (2016). A plantation-dominated forest transition in Chile. Applied Geography, 75, 71-82.

Heng, S. (1980). The addition, decomposition and accumulation of organic matter in some native Nothofagus spp. forests and Pinus radiata plantations in the South Island of New Zealand (Doctoral dissertation, Lincoln College, University of Canterbury).

Heuck, C., & Spohn, M. (2016). Carbon, nitrogen and phosphorus net mineralization in organic horizons of temperate forests: stoichiometry and relations to organic matter quality. Biogeochemistry, 131(1-2), 229-242.

Huang WJ, Zhou GY, Liu JX (2011) Nitrogen and phosphorus status and their influence on aboveground production under increasing nitrogen deposition in three successional forests. Acta Oecologica-International Journal of Ecology 44:20–27.

Huber, A., Iroumé, A., Mohr, C., & Frêne, C. (2010). Efecto de plantaciones de Pinus radiata y Eucalyptus globulus sobre el recurso agua en la Cordillera de la Costa de la región del Biobío, Chile. Bosque (Valdivia), 31(3), 219-230.

IPCC, W. (2013). Working Group I Contribution to the IPCC Fifth Assessment Report: Climate Change 2013: The Physical Science Basis, Summary for Policymakers. IPCC, UN.

Jackson, R. B., Lajtha, K., Crow, S. E., Hugelius, G., Kramer, M. G., Piñeiro, G. (2017). The ecology of soil carbon: pools, vulnerabilities, and biotic and abiotic controls. Annual Review of Ecology, Evolution, and Systematics, 48, 419-445.

Jobbágy, E. G., Jackson, R. B. (2000). The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological applications, 10(2), 423-436.

Kalbitz, K., Solinger, S., Park, J. H., Michalzik, B., Matzner, E. (2000). Controls on the dynamics of dissolved organic matter in soils: a review. Soil science, 165(4), 277-304.

Kasel, S., Bennett, L. T. (2007). Land-use history, forest conversion, and soil organic carbon in pine plantations and native forests of south eastern Australia. Geoderma, 137(3-4), 401-413.

Kay, M., & Wobbrock, J. O. (2019). Package 'ARTool'.

Lal, R. (2005). Forest soils and carbon sequestration. Forest ecology and management, 220(1-3), 242-258.

Lal, R. (2013). Soil carbon management and climate change. Carbon Management, 4(4), 439-462. doi:10.4155/cmt.13.31

Leiva, J., Godoy, R. (2001, September). Production and decomposition of litterfall in Nothofagus forest in Southern Chile. In Comparison of ecosystem functioning and biogeochemical cycles in temperate forests in Southern Chile and Flanders. Proceedings of the workshop held at Ghent University, Belgium (pp. 17-19).

Liao, C., Luo, Y., Fang, C., Chen, J., Li, B. (2012). The effects of plantation practice on soil properties based on the comparison between natural and planted forests: a meta-analysis. Global ecology and biogeography, 21(3), 318-327.

Liu, X., Meng, W., Liang, G., Li, K., Xu, W., Huang, L., & Yan, J. (2014). Available phosphorus in forest soil increases with soil nitrogen but not total phosphorus: evidence from subtropical forests and a pot experiment.*PLoS One*,*9*(2).

Lusk, C. H., Donoso, C., Jiménez, M., Moya, C., Oyarce, G., Reinoso, R., Saldaña A., Villegas, P., Matus, F. (2001). Descomposición de hojarasca de Pinus radiata y tres especies arbóreas nativas. Revista chilena de historia natural, 74(3), 705-710.

Lützow, M. V., Kögel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner, B., & Flessa, H. (2006). Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions–a review. European journal of soil science, 57(4), 426-445.

Mancinelli, R., Campiglia, E., Di Tizio, A., Marinari, S. (2010). Soil carbon dioxide emission and carbon content as affected by conventional and organic cropping systems in Mediterranean environment. Applied Soil Ecology, 46(1), 64-72.

Manzoni, S., Trofymow, J. A., Jackson, R. B., & Porporato, A. (2010). Stoichiometric controls on carbon, nitrogen, and phosphorus dynamics in decomposing litter. Ecological Monographs, 80(1), 89-106.

Marthews, T. R., Metcalfe, D., Malhi, Y., Phillips, O., Huaraca Huasco, W., Riutta, T. (2013). Measuring Tropical Forest Carbon Allocation and Cycling: A RAINFOR-GEM Field Manual for Intensive Census Plots (v2. 2); 2012.

Matus, F., Amigo, X., Kristiansen, S. M. (2006). Aluminium stabilization controls organic carbon levels in Chilean volcanic soils. Geoderma, 132(1), 158-168. doi:https://doi.org/10.1016/j.geoderma.2005.05.005

McGroddy, M. E., Daufresne, T., & Hedin, L. O. (2004). Scaling of C: N: P stoichiometry in forests worldwide: Implications of terrestrial redfield-type ratios. Ecology, 85(9), 2390-2401.

Milla, F., Emanuelli, P., Sartori, A. (2013). Compendio de funciones alométricas para la estimación de biomasa de especies forestales presentes en Chile: Elemento clave para la Estrategia Nacional de Bosques y Cambio Climático (ENBCC). Corporación Nacional Forestal.

Millar, C. S. (1974). Decomposition of coniferous leaf litter. Biology of plant litter decomposition, 1, 105-128.

Miranda, A., Altamirano, A., Cayuela, L., Pincheira, F., Lara, A. (2015). Different times, same story: Native forest loss and landscape homogenization in three physiographical areas of south-central of Chile. Applied Geography, 60, 20-28.

Miranda, K. M., Espey, M. G., & Wink, D. A. (2001). A rapid, simple spectrophotometric method for simultaneous detection of nitrate and nitrite. Nitric oxide, 5(1), 62-71.

Müller, M., Oelmann, Y., Schickhoff, U., Böhner, J., & Scholten, T. (2017). Himalayan treeline soil and foliar C: N: P stoichiometry indicate nutrient shortage with elevation. Geoderma, 291, 21-32.

Nahuelhual, L., Carmona, A., Lara, A., Echeverría, C., González, M. E. (2012). Land-cover change to forest plantations: Proximate causes and implications for the landscape in south-central Chile. Landscape and urban planning, 107(1), 12-20.

Neculman, R., Rumpel, C., Matus, F., Godoy, R., Steffens, M., & de la Luz Mora, M. (2013). Organic matter stabilization in two Andisols of contrasting age under temperate rain forest. Biology and fertility of soils, 49(6), 681-689.

Neff, J. C., Asner, G. P. (2001). Dissolved organic carbon in terrestrial ecosystems: synthesis and a model. Ecosystems, 4(1), 29-48.

Olander LP, Vitousek PM (2000) Regulation of soil phosphatase and chitinase activity by N and P availability. Biogeochemistry 49: 175–190.

Olson, J.S., 1963. Energy storage and the balance of producers and decomposers in ecological systems. Ecology 44, 322–331

Ouyang, S., Xiang, W., Gou, M., Lei, P., Chen, L., & Deng, X. (2018). Variations in soil carbon, nitrogen, phosphorus and stoichiometry along forest succession in southern China.

Oyarzun, C., Aracena, C., Rutherford, P., Godoy, R., Deschrijver, A. (2007). Effects of land use conversion from native forests to exotic plantations on nitrogen and phosphorus retention in catchments of southern Chile. Water, air, and soil pollution, 179(1-4), 341-350.

Panichini, M., Neculman, R., Godoy, R., Arancibia-Miranda, N., & Matus, F. (2017). Understanding carbon storage in volcanic soils under selectively logged temperate rainforests.*Geoderma*,*302*, 76-88.

Peñuelas, J., Sardans, J., Rivas-ubach, A., & Janssens, I. A. (2012). The human-induced imbalance between C, N and P in Earth's life system. Global Change Biology, 18(1), 3-6.

Pérez, C. A., DeGrandpre, M. D., Lagos, N. A., Saldías, G. S., Cascales, E. K., Vargas, C. A. (2015). Influence of climate and land use in carbon biogeochemistry in lower reaches of rivers in central southern Chile: Implications for the carbonate system in river-influenced rocky shore environments. Journal of Geophysical Research: Biogeosciences, 120(4), 673-692.

Pérez, C. A., Hedin, L. O., Armesto, J. J. (1998). Nitrogen mineralization in two unpolluted old-growth forests of contrasting biodiversity and dynamics. Ecosystems, 1(4), 361-373.

Post, W. M., & Kwon, K. C. (2000). Soil carbon sequestration and land-use change: processes and potential. Global Change Biology, 6(3), 317-327. doi:10.1046/j.1365-2486.2000.00308.x

Quan, Q., Wang, C., He, N., Zhang, Z., Wen, X., Su, H., & Xue, J. (2014). Forest type affects the coupled relationships of soil C and N mineralization in the temperate forests of northern China. Scientific reports, 4, 6584.

Raich, J. W., Schlesinger, W. H. (1992). The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. Tellus B, 44(2), 81-99.

Rasmussen, C., Southard, R. J., & Horwath, W. R. (2006). Mineral control of organic carbon mineralization in a range of temperate conifer forest soils. Global Change Biology, 12(5), 834-847.

Rasse, D. P., Rumpel, C., & Dignac, M. F. (2005). Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. Plant and soil, 269(1-2), 341-356.

Ribeiro Jr, P. J., Diggle, P. J., Ribeiro Jr, M. P. J., & Suggests, M. A. S. S. (2007). The geoR package. R news, 1(2), 14-18.

Richardson, S. J., Allen, R. B., Doherty, J. E. (2008). Shifts in leaf N: P ratio during resorption reflect soil P in temperate rainforest. Functional Ecology, 22(4), 738-745.

Rivas, Y., Oyarzun, C., Godoy, R., Valenzuela, E. (2009). Mineralización del nitrógeno, carbono y actividad enzimática del suelo en un bosque de Nothofagus obliqua (Mirb) Oerst y una plantación de Pinus radiata D. Don. del centro-sur de Chile. Revista chilena de historia natural, 82(1), 119-134.

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(6), 803-813. Temperate conifer forest soils. Global Change Biology, 12(5), 834-847.

Sadzawka, A., Carrasco, M., Grez, R., Mora, M., Flores, H., Neaman, A. (2006). Métodos de análisis recomendados para los suelos de Chile. Serie actas INIA, 163.

Savage, K. E., Davidson, E. A. (2003). A comparison of manual and automated systems for soil CO2 flux measurements: Trade-offs between spatial and temporal resolution. Journal of Experimental Botany, 54(384), 891-899.

Sayer, E. J., Heard, M. S., Grant, H. K., Marthews, T. R., Tanner, E. V. (2011). Soil carbon release enhanced by increased tropical forest litterfall. Nature Climate Change, 1(6), 304-307.

Schlesinger, W. H., Bernhardt, E. S. (2013). Biogeochemistry: an analysis of global change. Academic press.

Schoeneberger, P. J., Wysocki, D. A., Benham, E. C. (Eds.). (2012). Field book for describing and sampling soils. Government Printing Office

Simunek, J., Van Genuchten, M. T., Sejna, M. (2005). The HYDRUS-1D software package for simulating the one-dimensional movement of water, heat, and multiple solutes in variably-saturated media. University of California-Riverside Research Reports, 3, 1-240.

Six, J., Conant, R. T., Paul, E. A., & Paustian, K. (2002). Stabilization mechanisms of soil organic matter: implications for C-saturation of soils. Plant and soil, 241(2), 155-176.

Soil Survey Staff. (2014). Keys to soil taxonomy. Department of Agriculture: Natural Resources Conservation Service. USA.

Spohn, M., & Sierra, C. A. (2018). How long do elements cycle in terrestrial ecosystems? Biogeochemistry, 139(1), 69-83.

Spohn, M., Novák, T. J., Incze, J., & Giani, L. (2016). Dynamics of soil carbon, nitrogen, and phosphorus in calcareous soils after land-use abandonment–A chronosequence study. Plant and soil, 401(1-2), 185-196.

Staelens, J., Ameloot, N., Almonacid, L., Padilla, E., Boeckx, P., Huygens, D., Verheyen, K., Oyarzun, C., Godoy, R. (2011). Litterfall, litter decomposition and nitrogen mineralization in old-growth evergreen and secondary deciduous Nothofagus forests in south-central Chile. Revista chilena de historia natural, 84(1), 125-141.

Staelens, J., Oyarzún, C., Almonacid, L., Padila, E., Verheyen, K. (2009). Aboveground nutrient cycling in temperate forest ecosystems of southern Chile. In Ecological advances on Chilean temperate rainforests (pp. 103-116). Academia Press.

Sterner, R. W., & Elser, J. J. (2002). Ecological stoichiometry: the biology of elements from molecules to the biosphere. Princeton university press.

Subke, J. A., Inglima, I., Francesca Cotrufo, M. (2006). Trends and methodological impacts in soil CO2 efflux partitioning: a metaanalytical review. Global Change Biology, 12(6), 921-943.

Team, R. C. (2013). R: A language and environment for statistical computing.

Torn, M. S., Trumbore, S. E., Chadwick, O. A., Vitousek, P. M., & Hendricks, D. M. (1997). Mineral control of soil organic carbon storage and turnover. Nature, 389(6647), 170.

Tosso, J. (1985). Suelos volcánicos de Chile. Ministerio de Agricultura. 1a edición. Instituto de Investigaciones Agropecuarias (INIA). Santiago.

Treseder KK, Vitousek PM (2001) Effects of soil nutrient availability on investment in acquisition of N and P in Hawaiian rain forests. Ecology 82: 946–954.

Tuo, D., Gao, G., Chang, R., Li, Z., Ma, Y., Wang, S., Wang, C., Fu, B. (2018). Effects of revegetation and precipitation gradient on soil carbon and nitrogen variations in deep profiles on the Loess Plateau of China. Science of The Total Environment, 626, 399-411. doi:https://doi.org/10.1016/j.scitotenv.2018.01.116

Valenzuela, C., Acuña, E., Muñoz, F., Ortega, A., Cancino, J. (2017). Generalized allometric equations of total volume generated by meta-analysis for roble, raulí and coigüe in second growth forests in Chile. Forestry: An International Journal of Forest Research, 90(5), 661-674.

Vitousek PM, Porder S, Houlton BZ, Chadwick OA (2010) Terrestrial phosphorus limitation: mechanisms, implications, and nitrogen-phosphorus interactions. Ecological applications 20: 5–15.

Vitousek, P. M., Fahey, T., Johnson, D. W., Swift, M. J. (1988). Element interactions in forest ecosystems: succession, allometry and input-output budgets. Biogeochemistry, 5(1), 7-34.

Wang, J., Ren, C., Cheng, H., Zou, Y., Bughio, M. A., Li, Q. (2017). Conversion of rainforest into agroforestry and monoculture plantation in China:

Consequences for soil phosphorus forms and microbial community. Science of the Total Environment, 595, 769-778.

Watanabe, F. S. and S. R. Olsen 1965. Test of an ascorbic acid method for determining P in water and NaHCO3 extracts from soil. Soil Science Society of America Proceedings 29: 677-678.

Yan, E. R., Wang, X. H., Huang, J. J., Li, G. Y., Zhou, W. (2008). Decline of soil nitrogen mineralization and nitrification during forest conversion of evergreen broad-leaved forest to plantations in the subtropical area of Eastern China. Biogeochemistry, 89(2), 239-251.

Yanai, R. D., Currie, W. S., Goodale, C. L. (2003). Soil carbon dynamics after forest harvest: an ecosystem paradigm reconsidered. Ecosystems, 197-212.

Yee, T. W. (2010). The VGAM package for categorical data analysis. Journal of Statistical Software, 32(10), 1-34.

Zhang, J., Zhao, N., Liu, C., Yang, H., Li, M., Yu, G., Wilcox, K., Yu, Q., He, N. (2018). C: N: P stoichiometry in China's forests: From organs to ecosystems. Functional ecology, 32(1), 50-60.

Zhou, GY., Liu, SG., Li, ZA., Zhang, DQ., Tang, XL., Zhou, C., Yan, J., Mo, J. (2006). Old growth forest can accumulate carbon in soils. Science, 314, 1417.